Universitat Autònoma de Barcelona

The key role of ecosystem services in forests: spatial relationships, conservation implications and risk to climate change hazards

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The key role of ecosystem services in forests: spatial relationships, conservation implications and risk to climate change hazards

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Abstract

Forest ecosystems provide a wide variety of benefits for human well-being, commonly referred to as ecosystem services (ES). Understanding how these ES are distributed across the landscape and identifying their main drivers is essential to inform policy to protect, enhance and restore these ecosystems. Besides, protected areas (PAs) are fundamental for biodiversity conservation and the provision of ES, yet their effectiveness in maintaining ES and biodiversity is still unclear. Currently, forests are increasingly under pressure from climate change, resulting in changes in disturbance regimes (e.g., wildfires, drought, insect-outbreaks and windstorms). Predicting where these natural hazards will occur in the future and to what extent forest ES will be affected are also fundamental research challenges.

The general objective of this thesis is to analyze the spatial distribution of forest ES, their relevance in conservation and their vulnerability and risk to climate change hazards, especially wildfires. To do so, 1) we have analyzed the spatial distribution, relationship and drivers of forest carbon stocks and biodiversity in two regions (Spain and Quebec) and five subclimates (steppe, dry Mediterranean, humid Mediterranean, temperate and boreal); 2) we have determined the role of PAs in preserving ES and biodiversity in forests and shrublands of Catalonia (NE Spain); 3) we have developed a general framework of forest vulnerability and risk of losing ES due to different climate change hazards; and 4) we have assessed the spatial patterns and drivers of forest vulnerability to wildfires and the corresponding risk of losing ES in Catalonia (NE Spain). We have found a general positive relationship between carbon stocks and biodiversity, with the highest values in northern Spain (humid Mediterranean subclimate) and southern Quebec (temperate subclimate). High density and structural diversity have simultaneously favored carbon stocks, tree and overall biodiversity. The variables positively affecting carbon and biodiversity have been also driving their hotspots, emphasizing the viability of 'win-win' solutions. Regarding PAs, we have found more carbon stocks, coverage of community-interest habitats, priority-habitats and geological-interest sites in PAs than in buffer zones, but none of the biodiversity variables considered (i.e., tree and bird richness) have showed differences between PAs and buffer zones. PAs with higher degree of protection (i.e., moderate vs partial protection) have not provided higher levels of ES and biodiversity, or vice versa. Furthermore, we have proposed a general framework to assess forest vulnerability and risk based on the components of exposure, hazard magnitude, susceptibility and lack of

adaptive capacity. We have suggested a standardized procedure to define and combine these components, as well as a list of indicators readily applicable to the main climate change-related hazards to forests. Finally, we have applied this general framework to the particular case of wildfires in Catalonia. The results have indicated that hazard magnitude is the most important factor defining ES at risk from wildfires. Climate is the main driving factor of ES at risk under average conditions, but forest functional type - in particular non-Mediterranean conifers that have low adaptive capacity - have gained importance under extreme conditions. The highest increases in risk have been found in relatively wet forests with currently low risk, which according to climate trends will become common in the future.

Overall, this thesis has gained evidence on the positive relationship between carbon stocks and biodiversity and their main drivers in five subclimates, and has showed that the conservation strategy in Catalonia is only effective at maintaining some of the ES and conservation variables considered. It has also contributed with an innovative conceptual framework of forest vulnerability and risk of losing ES due to climate change hazards, constituting a basis for a systematic operationalization of forest risk and vulnerability. The application of this framework to the case of wildfires has showed relevant implications on the future risk of losing ES due to wildfires, which could contribute to future-oriented policies by anticipating conditions associated with particularly high risks and guiding efficient forest management.

Resum

Els boscos proveeixen d'una àmplia varietat de beneficis pel benestar humà, anomenats serveis ecosistèmics (SE). Entendre com i per què aquests SE es distribueixen en el paisatge és essencial per dotar a les polítiques d'informació per protegir, millorar i restaurar aquests ecosistemes. A més, les Àrees Protegides (AP) són fonamentals per a la conservació de la biodiversitat i la provisió de SE, però la seva efectivitat en el manteniment dels SE i la biodiversitat encara no està clara. Actualment, els boscos estan cada vegada més sotmesos a la pressió del canvi climàtic, amb canvis en el règim de pertorbacions (com ara incendis, sequera, plagues o ventades). Predir on aquestes pertorbacions tindran lloc en el futur i fins a quin punt els SE dels boscos s'hi veuran afectats són reptes fonamentals en la recerca.

L'objectiu general d'aquesta tesi és analitzar la distribució espacial dels SE dels boscos, la seva rellevància en la conservació i la seva vulnerabilitat i risc enfront pertorbacions del canvi climàtic, especialment els incendis forestals. Per assolir aquest objectiu, 1) hem analitzat la distribució espacial dels estocs de carboni i la biodiversitat, així com la relació entre ells i les seves causes, en boscos de dues regions (Espanya i Québec) i cinc subclimes (estèpic, Mediterrani sec, Mediterrani humit, temperat i boreal); 2) hem determinat el rol de les AP en la preservació dels SE i la biodiversitat a Catalunya; 3) hem desenvolupat un marc conceptual per avaluar la vulnerabilitat dels boscos i el seu risc de pèrdua de SE degut a pertorbacions del canvi climàtic; i 4) hem avaluat els patrons espacials i les causes de la vulnerabilitat dels boscos a incendis i el risc associat de pèrdua de SE a Catalunya. La relació entre els estocs de carboni i la biodiversitat és en general positiva, amb valors més elevats al nord d'Espanya (subclima Mediterrani humit) i al sud del Québec (subclima temperat). Valors de densitat i diversitat estructural elevats han afavorit al mateix temps els estocs de carboni, la biodiversitat d'arbres i la biodiversitat global. Les variables amb un efecte positiu en el carboni i en la biodiversitat també han tingut un efecte positiu en els seus valors més elevats (els *hotspots*), destacant la viabilitat de solucions de guany mutu (*win-win*). Respecte a les AP, hem trobat més estocs de carboni, cobertura d'hàbitats d'interès comunitari, hàbitats prioritaris i llocs d'interès geològic a l'interior de les AP que a les seves àrees d'influència (o *buffer zones*), però cap dels indicadors de biodiversitat (riquesa d'arbres i d'aus) ha mostrat diferències entre les AP i les àrees d'influència. Les AP amb nivells de protecció més elevats (protecció moderada respecte parcial) no han proveït de més SE i biodiversitat, o viceversa. A més, hem proposat un

marc conceptual per avaluar la vulnerabilitat dels boscos i el risc de pèrdua de SE, basat en els components d'exposició, magnitud de la pertorbació, susceptibilitat i manca de capacitat adaptativa. Hem suggerit un procediment estàndard per definir i combinar aquests components, a més d'una llista d'indicadors fàcilment disponibles i aplicables a les pertorbacions degudes al canvi climàtic que són més rellevants en boscos. Finalment, hem aplicat aquest marc general als incendis forestals de Catalunya. Els resultats han mostrat que la magnitud de la pertorbació és el component més important que defineix el risc de pèrdua de SE degut a incendis. El clima és la causa més important dels SE en risc sota condicions mitjanes, però el tipus funcional de bosc - especialment les coníferes no Mediterrànies que tenen poca capacitat adaptativa - guanya importància sota condicions extremes. L'augment de risc més gran s'ha trobat en boscos relativament humits que actualment tenen un risc baix, situació que segons les tendències climàtiques actuals passarà a ser més comuna en un futur.

En general, aquesta tesi ha contribuït a augmentar l'evidència científica de la relació positiva entre els estocs de carboni i la biodiversitat, així com les seves causes en cinc subclimes. També ha mostrat que la conservació a Catalunya només és efectiva en el manteniment d'alguns dels SE i variables de conservació considerades. També ha contribuït amb un marc conceptual innovador sobre la vulnerabilitat dels boscos i el risc de pèrdua de SE degut a pertorbacions del canvi climàtic, assentant les bases per avaluar la vulnerabilitat i el risc d'una manera operativa i sistemàtica. L'aplicació d'aquest marc conceptual als incendis forestals ha mostrat implicacions rellevants en el risc de pèrdua de SE en un futur, fet que podria contribuir en el desenvolupament de polítiques futures mitjançant l'anticipació del risc, i ser una guia per la gestió forestal eficient.

Resumen

Los bosques proveen una amplia variedad de beneficios para el bienestar humano, comúnmente llamados servicios ecosistémicos (SE). Entender cómo y por qué estos SE se distribuyen en el paisaje es esencial para dotar a las políticas de información para proteger, mejorar y restaurar estos ecosistemas. Además, las Áreas Protegidas (AP) son fundamentales para la conservación de la biodiversidad y la provisión de SE, pero su efectividad en el mantenimiento de los SE y la biodiversidad aún no está del todo clara. Actualmente, los bosques están cada vez más sometidos a la presión del cambio climático, con cambios en el régimen de perturbaciones (por ejemplo, incendios, sequía, plagas o vientos severos). Predecir en qué lugares estas perturbaciones se darán en un futuro y hasta qué punto los SE se verán afectados son retos fundamentales de investigación.

El objetivo general de esta tesis es analizar la distribución espacial de los SE de los bosques, su relevancia en la conservación y su vulnerabilidad y riesgo frente a perturbaciones del cambio climático, especialmente los incendios forestales. Para cumplir este objetivo, 1) hemos analizado la distribución espacial de los stocks de carbono y la biodiversidad en los bosques, así como la relación entre ellos y sus causas, en dos regiones (España y Québec) y cinco subclimas (estépico, Mediterráneo seco, Mediterráneo húmedo, templado y boreal); 2) hemos determinado el rol de las AP en la preservación de los SE y la biodiversidad en Cataluña; 3) hemos desarrollado un marco conceptual para evaluar la vulnerabilidad de los bosques y su riesgo de pérdida de SE debido a perturbaciones del cambio climático; y 4) hemos evaluado los patrones espaciales y las causas de la vulnerabilidad de los bosques a incendios y el riesgo asociado de pérdida de SE en Cataluña. La relación entre los stocks de carbono y la biodiversidad es en general positiva, con valores más elevados en el norte de España (subclima Mediterráneo húmedo) y en el sur del Québec (subclima templado). Valores de densidad y diversidad estructural elevados han favorecido los stocks de carbono, la biodiversidad de árboles y la biodiversidad global. Las variables con un efecto positivo en el carbono y en la biodiversidad también han tenido un efecto positivo en sus valores más elevados (los *hotspots*), destacando así la viabilidad de soluciones de doble beneficio (*win-win*). Respecto a las AP, hemos encontrado más stocks de carbono, cobertura de hábitats de interés comunitario, hábitats prioritarios y lugares de interés geológico dentro de las AP que en sus áreas de influencia (o *buffer zones*), pero ninguno de los indicadores de biodiversidad (riqueza de árboles y aves) ha mostrado diferencias entre las AP y sus áreas de influencia. Las AP con

niveles de protección más elevados (protección moderada respecto a parcial) no han proveído de más SE y biodiversidad, o viceversa. Además, hemos propuesto un marco conceptual para evaluar la vulnerabilidad de los bosques y el riesgo de pérdida de SE, basado en los componentes de exposición, magnitud de la perturbación, susceptibilidad y ausencia de capacidad adaptativa. Hemos sugerido un procedimiento estándar para definir y combinar estos componentes, además de una lista de indicadores fácilmente disponibles y aplicables a las perturbaciones debidas al cambio climático más relevantes en bosques. Finalmente, hemos aplicado este marco general a los incendios forestales en Cataluña. Los resultados muestran que la magnitud de la perturbación es el componente más importante que define el riesgo de pérdida de SE debido a incendios. El clima es la causa más importante de los SE en riesgo bajo condiciones medias, pero el tipo funcional de bosque - especialmente las coníferas no Mediterráneas que tienen una menor capacidad adaptativa - gana importancia bajo condiciones extremas. El aumento de riesgo más grande está en bosques relativamente húmedos que actualmente estan en menor riesgo, situación que según las tendencias climáticas actuales pasará a ser más común en un futuro.

En general, esta tesis ha contribuido a aumentar la evidencia científica de la relación positiva entre los stocks de carbono y la biodiversidad, así como sus causas en cinco subclimas. También ha mostrado que la conservación en Cataluña solo es efectiva para mantener algunos de los SE y variables de conservación consideradas. También ha contribuido con un marco conceptual innovador sobre la vulnerabilidad de los bosques y el riesgo de pérdida de SE debido a perturbaciones del cambio climático, constituyendo la base para evaluar la vulnerabilidad y el riesgo de un modo operativo y sistemático. La aplicación de este marco conceptual a los incendios forestales ha demostrado implicaciones relevantes en el riesgo de pérdida de SE en un futuro, que podrían ser útiles para el desarrollo de políticas futuras mediante la anticipación del riesgo, pudiendo servir de guía para la gestión forestal eficiente.

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Chapter 1

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Chapter 2

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Chapter 3

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Chapter 4

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List of Acronyms

AEMET	Agencia Estatal de Meteorología
B	Overall biodiversity
BAL	Basal Area of Larger trees in the plot
B+	Hotspots of overall biodiversity
B-	Coldspots of overall biodiversity
Bb	Bird richness
Bb+	Hotspots of bird richness
Bb-	Coldspots of bird richness
Bt	Tree richness
Bt+	Hotspots of tree richness
Bt-	Coldspots of tree richness
C	Forest carbon stocks
C+	Hotspots of forest carbon stocks
C-	Coldspots of forest carbon stocks
CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
CWD	Climatic Water Deficit
CWM	Community Weighted Means

DBH	Diameter at Breast Height
E	Exposure, Exposed values
ES	Ecosystem Services
FVI	Forest Vulnerability Index
FWI	Fire Weather Index
HM	Hazard Magnitude
ICGC	Catalan Geographical and Cartographical Institute
IEFC	Forest Inventory of Catalonia
IFN2	Second Spanish National Forest Inventory
IFN3	Third Spanish National Forest Inventory
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
JRC	Joint Research Centre
LAC	Lack of Adaptive Capacity
LAI	Leaf Area Index
LIDAR	Laser Imaging Detection and Ranging
LM	Linear Model
MA	Millennium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
MCSC	Catalan Land Cover Map
MFE	Spanish Forest Map
NCP	Nature's Contributions to People
PAs	Protected Areas

PCA	Principal Component Analysis
PET	Potential Evapotranspiration
PET4	4ème Inventaire du Placettes-échantillons Temporaires du Québec
PEP4	4ème Inventaire du Placettes-échantillons Permanentes du Québec
PFC	Projected Forest Cover
PI	Persistence Index
REDD+	Reducing Emissions from Deforestation and Forest Degradation
RUSLE	Revisited Universal Soil Loss Equation
S	Susceptibility
SIFORT	Sistème d'Information Forestière par Tesselle
SLA	Specific Leaf Area
SMC	Servei Meteorològic de Catalunya
SWB	Soil Water Balance
TEEB	The Economics of Ecosystems and Biodiversity
V	Vulnerability
WDPA	World Database on Protected Areas

Introduction

The framework of ecosystem services

Humans have been always benefiting from nature, yet it was not until the late 1970s that the concept of ecosystem services emerged. Ecosystem services (hereafter ES) are the direct and indirect contributions of ecosystems to human well-being (Millennium Ecosystem Assessment, 2005). Previous literature emphasized the societal value of ecosystem's functions in the 1970s, when nature's services concept was initially used to show that ecosystem functions are vital to society to raise awareness on conservation (Odum, 1971; Westman, 1977; Daily, 1997)(Fig. 0.1). In a context of growing market environmentalism, other authors started to use the concept to frame ecological concerns in economic terms (Costanza, 1980; Martínez-Alier, 1987) (Fig. 0.1). But what undoubtedly contributed to popularizing the concept of ES was the paper by Costanza et al. (1997), which valued the world's ES in monetary terms (see also their updates in Costanza et al. (1998, 2014)). It is probably the most cited article in this area (>23,000 citations). Afterwards, the ES concept started to be introduced in the policy agenda through the Global Biodiversity Assessment (Heywood and Watson, 1995) and especially, with the Millennium Ecosystem Assessment (MA) (Millennium Ecosystem Assessment, 2005) (Fig. 0.1). The MA highlighted the human dependency on ecosystem direct services, as well as on ecosystem functions and processes that indirectly contribute to human well-being (Millennium Ecosystem Assessment, 2005). Other relevant initiatives such as The Economics of Ecosystems and Biodiversity (TEEB) and the Common International Classification of Ecosystem Services (CICES) illustrate the increased use of the ES concept, but also the benefits of biodiversity to human well-being (Fig. 0.1).

During the last decade, several outstanding initiatives related to ES and biodiversity have also been established. In 2011, the EU Convention of Biological Diversity Strategy 2020 was created to stop biodiversity loss and to ensure healthy ecosystems providing essential services to people, which includes among its targets the improvement of knowledge on ecosystems and ES (Convention on Biological Diversity, 2010). In 2012, The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was founded as an independent intergovernmental body 'to strengthen the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development' (IPBES, 2012). IPBES has gone beyond some of the initiatives stated before (i.e., MA, TEEB) because it included comprehensive interdisciplinary approaches on the

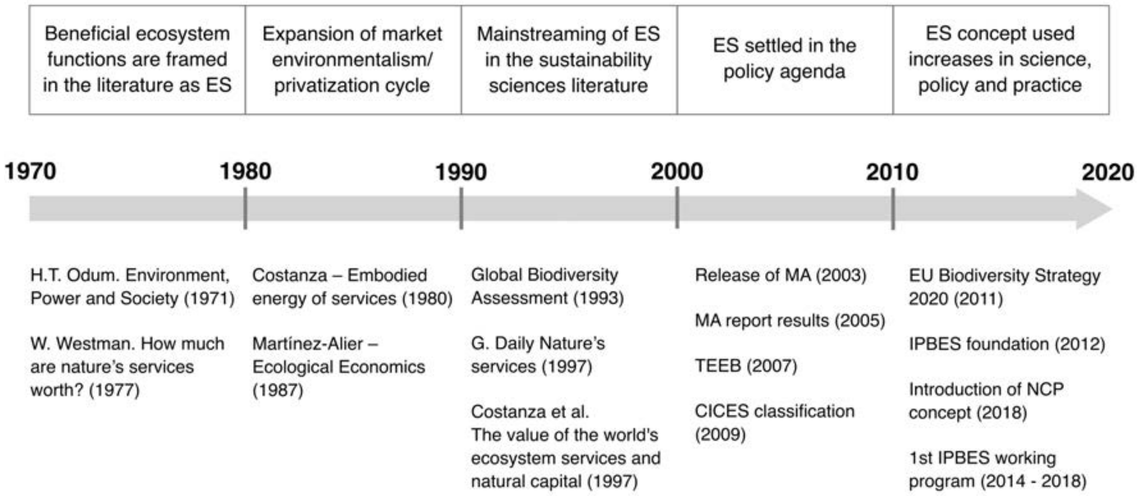


FIGURE 0.1: Timeline of the ecosystem services concept (adapted from Gómez-Baggethun et al. (2010)).

state of biodiversity and ES across key thematic areas and methodological issues from regional to global scales (Vadrot et al., 2018; Timpte et al., 2018). Later on, Díaz et al. (2018) introduced the concept of nature's contributions to people (NCP) as an alternative to the ES concept, stating that ES was only based on knowledge from natural sciences and economics, while NCP – defined as the positive and negative contributions of living nature to people's quality of life – was more inclusive because it engaged perspectives from the social sciences and humanities, thus being more likely to be incorporated into the policy arena (Díaz et al., 2018). Although IPBES promoted the use of the NCP concept, it is still a controversial concept with several criticisms (Kenter, 2018; Braat, 2018).

Given the variety of approaches and terminology used to refer to the contributions of ecosystems and nature to human well-being, in this thesis we will focus on the term 'ecosystem services'. We consider that it is today the most widely accepted term, which has been already adopted by scientists and policy-makers.

The potential of mapping ecosystem services

The spatial analysis of ES and biodiversity is essential to understand how ES are distributed across the landscape and where synergies and trade-offs arise. Maps of ES and biodiversity are effective and comprehensive products that are commonly used to support land-use planning and decision making. Within the target 2 of the EU Biodiversity Strategy to 2020, Member States were called to map and assess the state of ecosystems and their services with the aim to inform policy to protect, enhance or restore ecosystems (Convention on Biological Diversity, 2010). Their main outcome is the EU initiative on Mapping and Assessment of Ecosystems and their Services (MAES), which is supported by the national ecosystem assessments (e.g., for Spain, the Spanish National Ecosystem Assessment (Santos-Martin et al., 2014)). Nevertheless, land-use planning and decision making through the mapping and assessment of multiple ES could become challenging when synergies and trade-offs between ES arise. For instance, biodiversity conservation in forests could benefit from climate change mitigation policies if high carbon stocks co-occur with high biodiversity (Soto-Navarro et al., 2020). Contrarily, enhancing forest carbon stocks could lead to large reductions in other ES (e.g., decrease in water provisioning due to an increase of water used by forests (Brauman et al., 2007)). Sometimes synergies occur when multiple ES have the same drivers of change, whereas trade-offs could respond to either direct (e.g., forest harvest increases wood provision but also decreases carbon sink capacity) or indirect interactions (e.g., forest harvest decreases forest cultural services). Therefore, understanding the driving forces that determine the spatial distribution of ES across spatial scales is essential to ensuring effective implementation of land-use planning, conservation strategies, and climate change mitigation policies (Naidoo et al., 2008; Blumstein and Thompson, 2015).

Mapping areas with high values (also called 'hotspots') for one or more ES and biodiversity is a common way to identify priority areas for management and conservation. In the case of forest carbon stocks and biodiversity, some studies have shown a poor overlap between hotspots for global biodiversity and carbon storage (Anderson et al., 2009; Egoh et al., 2009), while others have found spatial convergence of carbon storage hotspots with plant diversity (Locatelli et al., 2013; Labrière et al., 2016). Hence, more local and regional studies are needed to better understand the spatial distribution of hotspots for different ES and the mechanisms behind these relationships. This could be particularly helpful to identify win-win future priority areas for conservation in action plans, such as the Convention on Biological Diversity's 2020 targets.

Incorporating ecosystem services into conservation priorities

Protected areas (PAs) represent 15% of the earth's surface and should have been increased to 17% by 2020 (Convention on Biological Diversity, 2010), a target that seems unlikely to be met. PAs have been established to avoid deforestation, preserve iconic landscapes and ecosystem representativeness, as well as to protect biodiversity and charismatic or endangered species (Eken et al., 2004; Hannah, 2008). The success of PAs in halting biodiversity loss has been until now insufficient, as global biodiversity is nowadays declining and it is expected to continue decreasing over the 21^{st} century (Hoffmann et al., 2010; Pereira et al., 2010; Tittensor et al., 2014). In Europe, Natura 2000 is the largest coordinated network of PAs in the world and covers 18% of the EU's land area. The aim of Natura 2000 is 'to ensure the long-term survival of Europe's most valuable and threatened species and habitats, listed under the Birds Directive and the Habitats Directive' (`https://ec.europa.eu/environment/nature/natura2000/`). Thus, it includes areas for protecting species and/or habitats listed as priorities in the Habitats (92/43/EEC) and the Birds Directives (2009/147/EC). Previous studies showed that Natura 2000 succeeded in covering threatened species included in the Directives (Donald et al., 2007; Maiorano et al., 2015; Kukkala et al., 2016), while others suggested a poor relationship between PAs and bird richness (Albuquerque et al., 2013) and that tighter control on intensive agriculture within PAs should be carried out to ensure conservation goals (Hermoso et al., 2018). In fact, to monitor Natura 2000 conservation status, EU Member States inform about their progress every 6 years. For instance, recent results for Catalonia (NE Spain) show that 48% of habitats are in Unfavourable-Inadequate conservation status (it includes 71% of forests in an Unfavourable status) and that 54% of the species in the Habitats Directive are in Unfavourable-Bad status. In the case of birds, 21% of breeding species are showing a decreasing trend at the short term (Sainz de la Maza et al., 2019). In addition, each country and region has its own protected area network that sometimes overlaps with Natura 2000 sites. In other cases, the overlapping between European and regional PAs is limited (Moilanen and Arponen, 2011). Hence, to evaluate the management and conservation effectiveness of PAs, local studies should consider the different levels of protection in each region.

While it is still not clear whether original conservation purposes are being fulfilled, ES have been suggested to be incorporated into conservation policies. Hence, part of the conservation community have claimed that conservation should look at protecting, restoring and enhancing the services that nature provides to people (Doak et al., 2014). Others have suggested that both biodiversity and ES (i.e., intrinsic and instrumental values, respectively) need to be included to achieve conservation objectives (Reyers et al., 2012). As PAs have not initially been established to preserve ES, there is a lack of agreement on the effectiveness of conservation strategies in maintaining ES. Previous studies have showed less provisioning ES inside than outside PAs (Castro et al., 2015; Mukul et al., 2017), but more regulating services inside than outside PAs (e.g., more carbon storage capacity of forests due to the role of PAs in preventing deforestation (Rodríguez et al., 2013; Vačkář et al., 2016)). PAs also supply multiple cultural services and socioeconomic benefits for local people (Palomo et al., 2013; Oldekop et al., 2016). Additionally, PAs exist within broader landscape mosaics that allow or interfere in the movement of species (Wiens, 2009). Conservation strategies should be focused not only inside PAs as an isolated system, but also in their surrounding buffer zones (Cox and Underwood, 2011). In this sense, conserving and reconnecting fragmented natural areas while enhancing the maintenance of multiple ES is one of the main objectives of the EU Green Infrastructure (`https://ec.europa.eu/environment/nature/ecosystems/`). Thus, incorporating ES into conservation priorities requires an in-depth analysis of the spatial distribution of ES in the whole study area (e.g., considering the PAs but also their buffer zones), since benefits from PAs could be enhanced by increasing their scale and connectivity.

The key role of forest ecosystem services in the face of climate change

Forests provide a wide variety of ES. Some of them are fundamental for society and human well-being, while at the same time are being increasingly under pressure from climate change. Forests supply provisioning services such as timber, fiber, bioenergy and clean water, as well as regulating services such as climate regulation, water purification and erosion control (Millennium Ecosystem Assessment, 2005). Forests are also relevant for cultural services, as they provide opportunities for recreation, education and spiritual development. One of the most studied ES in forests is climate regulation (Mengist and Soromessa, 2019). Forest carbon stocks may partially offset global greenhouse gas emissions, but do so differently depending upon the scale and biome (Canadell and Raupach, 2008; Pan et al., 2011). At the global scale, tropical and boreal forests store the most carbon, followed by temperate forests (Pan et al., 2011). Climate is one of the main drivers of carbon stocks through precipitation and temperature effects on net primary productivity (Keith et al., 2010; Stegen et al., 2011; Fischer et al., 2014). At regional scales, forest age, structure or species composition can affect carbon storage capacity (Nabuurs et al., 2008; Schwenk et al., 2012).

The ongoing climate change poses significant threats to forest ecosystems and their services. The IPCC stated that 'continued high emissions would lead to mostly negative impacts for biodiversity, ES and economic development and amplify risks for livelihoods and for food and human security' (IPCC, 2014). One of the main effects of climate change is the increase in the occurrence and severity of extreme climate events, as well as changes in disturbance regimes. Although natural hazards and disturbances are an integral part of forests that alter their structure, composition and functions (Turner, 2010), the fact that forests are characterized by immobile and long-lived vascular plants makes them more difficult to adapt to the high rate of climate change (Lindner et al., 2010; Seidl et al., 2016). Future changes in climate are expected to occur within one generation of trees (or even less), thus compromising the capacity of tree species to adapt through migration, regeneration or recovery processes.

Climate change hazards and their effect on forest ecosystem services

The most relevant climate change-related hazards affecting forest throughout the world are wildfires, drought, insect-outbreaks and windstorms, with distinct importance depending on the biome considered (Fig. 0.2). Tropical forests are the most affected by fires, yet due to deforestation for the expansion of agriculture (Mouillot and Field, 2005; Flannigan et al., 2009; Van Lierop et al., 2015). Boreal and temperate forests are increasingly affected by wildfires, especially associated with warmer and drier conditions (Abatzoglou et al., 2018; Seidl et al., 2020). Wildfires are also a main disturbance in Mediterranean forests, where recent studies show a decrease in fire activity (Turco et al., 2016; Silva et al., 2019), but also an increase of extreme wildfire events (Bowman et al., 2017). Future fire-prone areas could expand to the north and to Mediterranean mountains, while fuel

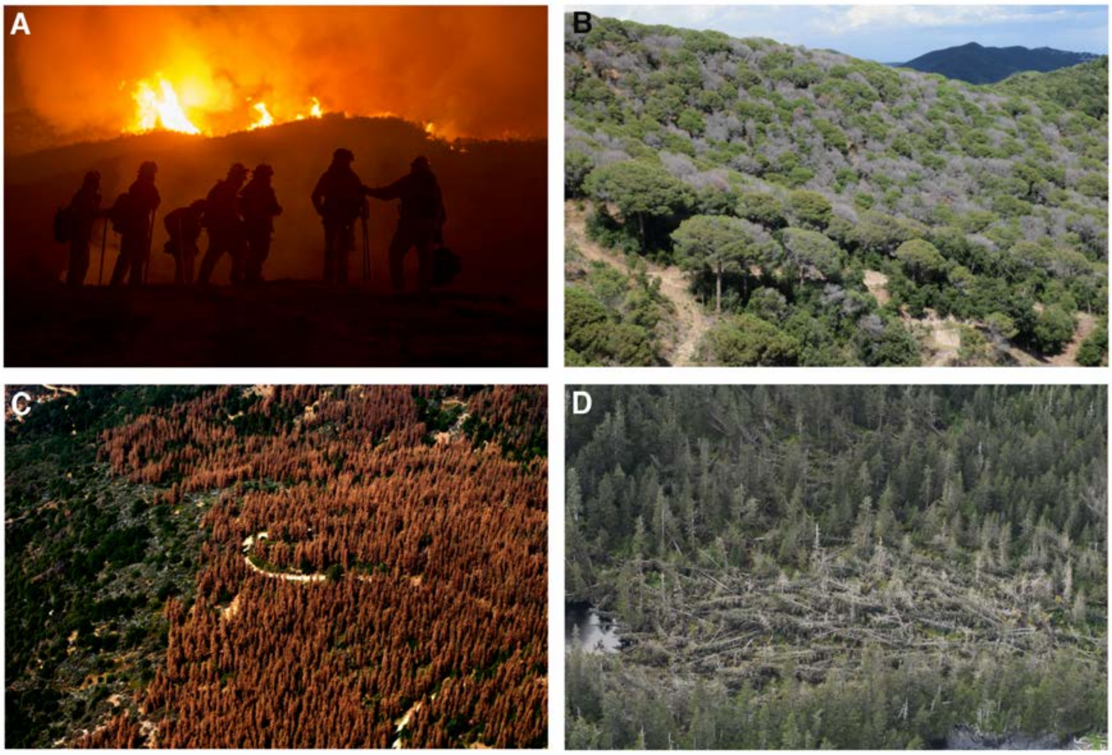


FIGURE 0.2: The main climate- change hazards in forests are (A) wildfires; (B) drought; (C) insect-outbreaks and (D) windstorms. (Credit images for (A) and (C) USDA Forest Service. Pacific Southwest Research Station; (B) JL. Ordóñez); and (D) USDA Forest Service, Forest Health Protection Program, Alaska Region).

availability could be limiting wildfire in the most arid areas (Dupuy et al., 2020). Recent examples of forest dieback due to drought and heat stress have been documented across all forested biomes (Allen et al., 2010; Allen et al., 2015). In tropical forests, increased mortality associated with drought was found in Amazonia (Feldpausch et al., 2016), but also in other areas such as Asia and Central America associated with El Niño-Southern Oscillation events (Aiba and Kitayama, 2002; Chazdon et al., 2005). Drought events also produced high tree mortality rates in temperate and boreal forests, which includes examples from the USA (Kane et al., 2014; Allen et al., 2015) and Canada (Hogg et al., 2008; Michaelian et al., 2011). Tree mortality associated with water stress and drought was also reported in Mediterranean forests (Gea-Izquierdo et al., 2014; García de la Serrana et al., 2015). In the case of insect-outbreaks, the most relevant attacks were found in boreal and temperate forests (Kurz et al., 2008; Senf et al., 2017), but examples in the Mediterranean Basin were also found (Gazol et al., 2019). Besides, windstorms are a major disturbance in temperate and boreal forests. Although stand-level (e.g., height-diameter ratio, crown density, etc) and topographic characteristics (e.g., slope) influence the impact of windstorms on forests (Scott and Mitchell, 2005; Klaus et al., 2011), future climate scenarios suggest an increase in the frequency of severe storms and its associated wind damage (Schelhaas et al. (2010); Klaus et al. (2011), but see Blennow et al. (2010); Saad et al. (2017)). Adding to these particular effects of the different climate change hazards is the fact that they also interact with mainly positive interaction effects (e.g., drought and wind facilitate insect-outbreaks and fire, Anderegg et al. (2015); Seidl et al. (2017)), resulting in the amplification of forest disturbances (Seidl et al., 2017).

The need for a comprehensive framework of forest vulnerability and risk

Vulnerability originates from the Latin word *vulnerare*, meaning 'to wound'. This term has been used in various contexts and in a wide range of disciplines, commonly understood as a predisposition to be harmed if an extreme event or hazard occurs. The origin of the word risk is more uncertain, from French *risque* or Italian *riscare*, meaning 'run into danger', as it normally involves the notion of probability of occurrence. According to the IPCC, vulnerability is 'the propensity or predisposition to be adversely affected by a hazard, including sensitivity or susceptibility to harm and lack of capacity to cope and adapt' (IPCC, 2018). Risk is defined as 'the potential for consequences where something of value is at stake and where the outcome is uncertain, which results from the interaction of vulnerability (of the affected system to a given hazard), its exposure over time to the hazard, as well as the (climate-related) hazard and the likelihood of its occurrence' (IPCC, 2018). These terms have been applied to create global and local indices, such as the World Risk Index, used to assess the risk and vulnerability of societies towards natural hazards at the country scale (Welle and Birkmann, 2015), or the Social Vulnerability Index to climate change at the local scale (Nguyen et al., 2017). However, these frameworks are not directly applicable to forests because of the particular nature of forest ecosystems and the principal hazards they are exposed to.

In forests, vulnerability and risk to climate change hazards have been defined in different ways. They have been hazard-specific (i.e., only considering one hazard), or have only used particular indicators or IPCC components. Fire vulnerability has been simply defined by changes in simulated future area burned relative to historical area burned (Buotte et al., 2019), or by considering specific indicators such as fuel moisture or frequency of fires (McWethy et al., 2013). It has also been defined using indicators of adaptive capacity such as vegetation recovery (Aretano et al., 2015) and reproductive strategies (Schelhaas et al., 2010). Besides, the study of Duguy et al. (2012) has assessed vulnerability of forests to wildfires accounting for susceptibility and adaptive capacity, but without including risk and exposure. Vulnerability to wildfires using exposure, sensitivity and coping capacity including different dimensions (e.g., physical and human) has been analyzed in the Mediterranean Europe, but without including risk (Oliveira et al., 2018). Examples of drought include the forest vulnerability index (FVI) based on forest stress defined by water and energy exchange processes caused by drought and high temperatures (Mildrexler et al., 2016). Resistance and resilience have been analyzed to assess vulnerability to insect-outbreaks (Sánchez-Pinillos et al., 2019*b*), while structural characteristics have been used to determine wind resistance (Schelhaas et al., 2010; Anyomi et al., 2017). Vulnerability of forests to climate change has been assessed by considering only particular indicators such as mortality, regeneration or productivity (Halofsky et al., 2018). Moreover, a recent study has analyzed vulnerability of southwestern forests in the USA to climate change by considering exposure, sensitivity and adaptive capacity scored subjectively in 10 regional forest types, but it has been based in a single study region (Thorne et al., 2018). Therefore, a general framework of forest vulnerability and risk to the main climate change-related hazards that uses the components defined by the IPCC and being applicable to forest ecosystems in different biomes is still lacking. This framework should facilitate the visualization of the complexity of vulnerability and risk using simple yet meaningful metrics to identify specific targets for vulnerability and risk reduction, so that it can be understood by policymakers and land-use managers.

Thesis objectives and outline

The general objective of this thesis is to analyze the spatial distribution of forest ES, their relevance in conservation and their vulnerability and risk to climate change hazards. The specific aims of this thesis are: 1) to analyze the spatial distribution, relationship and drivers of forest carbon stocks and biodiversity in two regions (Spain and Quebec) and five subclimates (steppe, dry Mediterranean, humid Mediterranean, boreal, and temperate); 2) to determine the role of protected areas in preserving ES and biodiversity in Catalonia (NE Spain); 3) to develop a general framework of forest vulnerability and risk that includes the IPCC components and can be applied to different forest types and hazards; and 4)

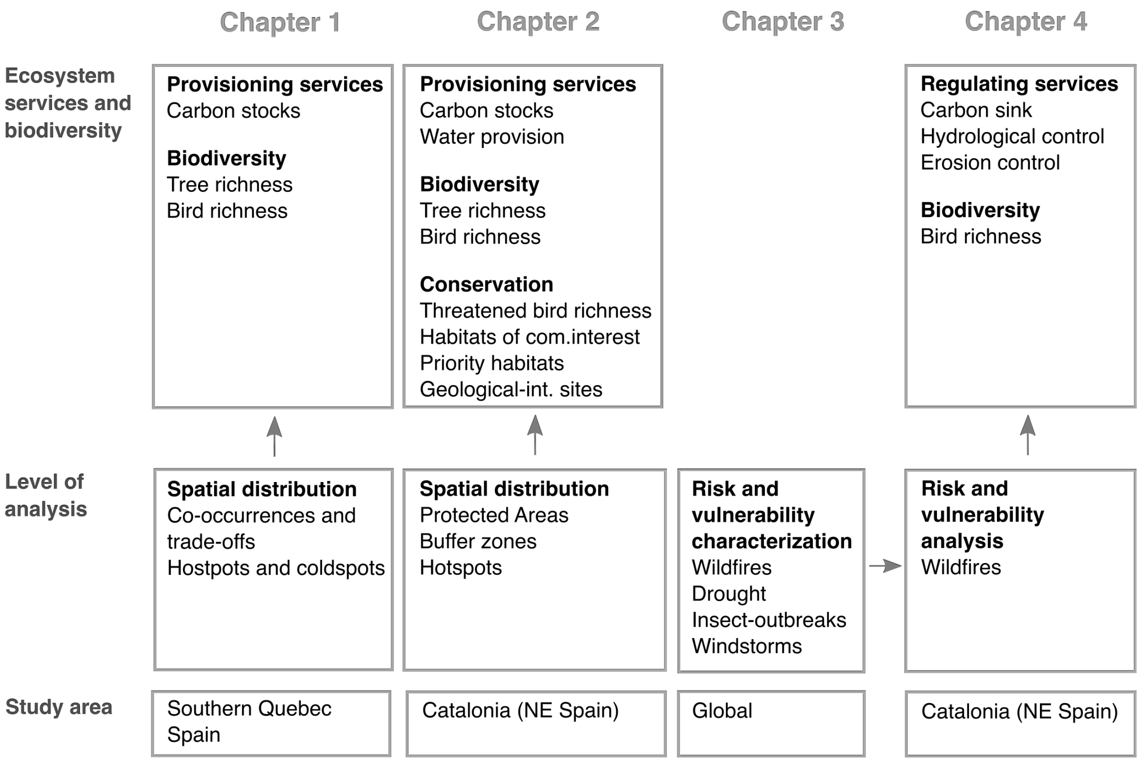


FIGURE 0.3: General overview of the thesis chapters regarding the ecosystem services, level of analysis and study areas considered.

to assess the spatial patterns and drivers of vulnerability of forests to wildfires and the corresponding risk of losing ES in Catalonia (NE Spain), using the general framework presented in the third objective. To accomplish these objectives, different indicators of ES, levels of analysis and study areas have been considered (Fig. 0.3). The indicators of the ES considered are related with provisioning services (carbon stocks and water provision) and regulating services (carbon sink as climate regulation, erosion control and hydrological control). We have also considered biodiversity variables (tree richness and bird richness), as well as conservation variables (threatened bird richness, habitats of community interest, priority habitats and geological-interest sites). The levels of analysis range from spatial relationships between ES (synergies or trade-offs) and distributions (on PAs and buffer zones, on the level of protection) to the development of composite indexes of risk and vulnerability and their application to wildfires. The study areas range from global to regional (Fig. 0.3). Specifically, this thesis is structured in four chapters with the following titles and objectives.

Chapter 1. The positive carbon stocks–biodiversity relationship in forests: co-occurrence and drivers across five subclimates

In this chapter, we determine the spatial distribution and relationship of forest carbon stocks and biodiversity by comparing two regions, Spain and Quebec, in two continents (Europe and North America), covering five subclimates (steppe, dry Mediterranean, humid Mediterranean, temperate and boreal). First, we determine the spatial patterns of forest carbon stocks and biodiversity (bird richness, tree richness, and overall biodiversity) and we examine the factors that influence them. Second, we establish the relationships between forest carbon stocks and biodiversity in the two regions and the different subclimates. Third, we define and characterize the areas of high values (hotspots) and low values (coldspots) of carbon stocks and biodiversity and quantify the degree of spatial overlap between them. To do so, we integrate information from two main data sets related to carbon and biodiversity in the two regions: National Forest Inventories and Breeding Bird Atlases.

Chapter 2. Are protected areas preserving ecosystem services and biodiversity? Insights from Mediterranean forests and shrublands

As protected areas (PAs) have not initially established to preserve ES, in this chapter we want to know to what extent they could be a useful tool to preserve not only biodiversity but also ES. Thus, we aim to determine the role of PAs in preserving ES and biodiversity, using forest and shrublands in Catalonia (NE Spain). First, we want to know whether the spatial distribution of ES (carbon stocks and water provision), biodiversity (woody and bird richness) and conservation variables (threatened bird richness, habitats and geology) vary between PAs (with different protection status) and buffer zones. Then, we quantify and compare the percentage of high values (hotspots) of ES, biodiversity and conservation variables inside PAs (with different protection status) and buffer zones.

Chapter 3. Characterizing forest vulnerability and risk to climate change hazards

The ongoing climate change will increase the occurrence and severity of extreme climate events. Previous vulnerability and risk assessments in forests were only focused on one of the IPCC components or on a single hazard. In this chapter, we present a general framework to evaluate forest vulnerability to climate change hazards and risk of ES loss that can be applied to different forest types and hazards. First, we define a general framework of forest vulnerability and risk using the latest IPCC components (i.e., exposed values, hazard magnitude, susceptibility and lack of adaptive capacity) structured in a time frame before, during and after the hazard. We also characterize each component with intrinsic and extrinsic factors. Second, we provide explicit examples of indicators for each of the components of vulnerability and risk. Third, we propose a methodology to combine the components of vulnerability and risk that considers their strong interdependencies in forests. Finally, we suggest a list of the methodological steps to apply this framework and discuss some of the applications and future research challenges.

Chapter 4. Assessing the risk of losing forest ecosystem services due to wildfires under average and extreme hazard conditions

Once the characterization of forest vulnerability and risk has been detailed in Chapter 3, the next step is applying this framework using real data. Hence, in this chapter we aim to evaluate the vulnerability of forests to wildfires and their risk of losing ES in Catalonia (NE Spain). First, we assess the influence of exposed values, hazard magnitude, susceptibility and lack of adaptive capacity on the spatial variability of the risk of losing forest ES due to wildfires. Second, we determine the effect of climate (i.e., mean annual temperature and annual precipitation) and forest functional type (i.e., broadleaf evergreen, broadleaf deciduous, Mediterranean conifer and non-Mediterranean conifer) on the risk of losing forest ES due to wildfires under average and extreme hazard conditions. Finally, we determine the influence of climate and forest functional type on the increase in risk associated to extreme vs. average hazard conditions.

1 The positive carbon stocks - biodiversity relationship in forests: co-occurrence and drivers across five subclimates

Abstract

Carbon storage in forests and its ability to offset global greenhouse gas emissions, as well as biodiversity and its capacity to support ecosystem functions and services are often considered separately in landscape planning. However, the potential synergies between them are currently poorly understood. Identifying the spatial patterns and factors driving their co-occurrence across different climatic zones is critical to more effectively conserve forest ecosystems at the regional level.

Here, we integrated information of National Forest Inventories and Breeding Bird Atlases across Europe and North America (Spain and Quebec, respectively), covering five subclimates (steppe, dry mediterranean, humid mediterranean, boreal and temperate). In particular, this chapter aims to 1) determine the spatial patterns of both forest carbon stocks and biodiversity (bird richness, tree richness and overall biodiversity) and the factors that influence them; 2) establish the relationships between forest carbon stocks and biodiversity; and 3) define and characterize the areas of high (hotspots) and low (coldspots) values of carbon and biodiversity, and ultimately quantify their spatial overlap.

Our results show that the factors affecting carbon and biodiversity vary between regions and subclimates. The highest values of carbon and biodiversity were found in northern Spain (humid Mediterranean subclimate) and southern Quebec (temperate subclimate) where there was more carbon as climate conditions were less limiting. High density and structural diversity simultaneously favored carbon stocks, tree and overall biodiversity, especially in isolated and mountainous areas, often associated with steeper slopes and low accessibility. In addition, the relationship between carbon stocks and biodiversity was positive in both regions and all subclimates, being stronger where climate is a limiting factor for forest growth. The spatial overlap between hotspots of carbon and biodiversity provides an excellent opportunity for landscape planning to maintain carbon stocks and conserve biodiversity. The variables positively affecting carbon and biodiversity were also driving the hotspots of both carbon and biodiversity, emphasizing the viability of 'win-win' solutions. Our results highlight the need to jointly determine the spatial patterns of ecosystem services and biodiversity for an effective and sustainable planning of forest landscapes that simultaneously support conservation and mitigate climate change.

1.1 Introduction

The concept of ecosystem services (ES) has been increasingly used to support land-use planning, conservation strategies and climate change mitigation policies. Understanding the driving forces that determine the spatial distribution of services and benefits that people obtain from ecosystems across spatial scales is essential to ensuring an effective implementation of these actions (Naidoo et al., 2008; Blumstein and Thompson, 2015).

Forest carbon (C) stocks may partially offset global greenhouse gas emissions, but do so differently depending upon the scale and biome (Canadell and Raupach, 2008; Pan et al., 2011). At the global scale, annual precipitation is positively correlated with forest biomass through effects on net primary productivity across a wide range of climates and biomes (Zhao and Zhou, 2006; Stegen et al., 2011; Fischer et al., 2014). Likewise, temperature can affect primary productivity by modifying stomatal conductance, carboxylation rates and nitrogen uptake (Keith et al., 2010; Urban et al., 2017), as well as through effects on soil organic matter decomposition and soil abiotic processes such as mineral weathering (Campbell et al., 2009). However, various studies in tropical, boreal and temperate climates have shown that the relationship between temperature and C stocks is not consistent (Raich et al., 2006; Stegen et al., 2011). Other studies suggested positive relationships where precipitation was not scarce (Armenteras et al., 2015; Duchesne et al., 2016) or negative relationships where an increase of temperature reduced precipitation in water-limited climates (Zhao and Zhou, 2006). At regional scales, different studies have shown that structurally and functionally diverse forests can use resources more efficiently, so they can have greater forest productivity (Paquette and Messier, 2011; Ruiz-Benito et al., 2014) and C storage (Vayreda et al., 2012; Mensah et al., 2016). Additionally, C stocks can be affected by natural disturbances (e.g., wildfires and insect outbreaks) by means of direct combustion of live trees or accelerated decomposition of dead biomass (Stinson et al., 2011; Heon et al., 2014). Moreover, forest management influences C stocks by promoting changes in forest age, structure or species composition (Nabuurs et al., 2008; Schwenk et al., 2012).

Biodiversity supports numerous ecosystem functions and services at multiple spatial and temporal scales (Gamfeldt et al., 2013; Jiang et al., 2013; Balvanera et al., 2014). It is needed for maintaining primary productivity and nutrient uptake because high-diverse ecosystems can have more energy inputs such as light interception (Naeem et al., 1994; Isbell et al., 2011). Biodiversity can also improve water quality by removing nitrates through niche partitioning (Cardinale, 2011). Specifically, bird biodiversity is essential to promote seed dispersal (Tomback and Linhart, 1990), pollination or pest control (Whelan et al., 2015). Previous broad-scale research has suggested a strong association between climate and species richness (Hawkins and Diniz-Filho, 2004; Kreft and Jetz, 2007). At higher latitudes, species richness is mostly determined by annual temperature (Kier et al. (2009); Kreft and Jetz (2007), but see Young et al. (2016)). In contrast, at lower latitudes (i.e., in tropical and arid climates), water related variables such as annual precipitation or actual evapotranspiration are the most important drivers of species richness (Kreft and Jetz, 2007). Moreover, other studies found high diversity in mature stands with multi-layered structure (Gil-Tena et al., 2007; Gao et al., 2014) and in stands with an increasing heterogeneous canopy structure (Calladine et al., 2017), suggesting that forest structure can be critical for maintaining forest biodiversity (Lindenmayer et al., 2006).

Previous studies have suggested a positive relationship between forest productivity and biodiversity at global scales (Liang et al., 2016), as well as at the regional level in tropical (Manhães et al., 2016), Mediterranean (Vilà et al., 2013), temperate and boreal forests (Paquette and Messier, 2011). However, other studies have found weak (Armenteras et al., 2015) or even negative relationships (Naidoo et al., 2008; Murray et al., 2015). These differences can be attributed to the scale of the analysis (global, national or subnational), as well as to the type of forest and measure of biodiversity used. Therefore, a better understanding of the complex and often interacting relationships between biodiversity and forest C storage across scales and biomes is urgently needed. The identification of areas with high values (hereafter referred to as 'hotspots') for both forest C stocks and biodiversity is crucial for improving sustainable management of natural resources as these areas could simultaneously support conservation and climate change mitigation. Several studies have shown poor overlap between hotspots for global biodiversity and C storage (Anderson et al., 2009; Egoh et al., 2009) or plant diversity and C storage (Jiang et al., 2013), because habitats having more biodiversity (grasslands and heathlands) were spatially dissociated from those storing more carbon (peatlands and forests). On the contrary, others have shown spatial convergence of C storage hotspots with plant diversity because of biogeographical factors, niche complementarity and dominance effects (Locatelli et al., 2013; Labrière et al., 2016). Thus, the identification of hotspots and the main drivers underlying their spatial overlap is essential to enhancing synergies and mitigating trade-offs between C and biodiversity.

The general objective of this chapter is to determine the spatial distribution and relationship of forest C stocks and biodiversity by comparing two regions, Spain and Quebec, in two continents (Europe and North America), covering five subclimates (steppe, dry mediterranean, humid mediterranean, boreal and temperate). The specific objectives are 1) to determine the spatial patterns of both forest C stocks and biodiversity (bird richness, tree richness and overall biodiversity) and examine the factors that influence them; 2) to establish the relationships between forest C stocks and biodiversity in the two regions and the different subclimates; and 3) to define and characterize the areas of high values (hotspots) and low values (coldspots) of C stocks and biodiversity, and quantify the degree of spatial overlap between them. To accomplish these objectives, we integrated information from two main datasets related to C and biodiversity in the two regions: National Forest Inventories and Breeding Bird Atlases.

1.2 Methods

Study area

The study region comprises peninsular Spain (hereafter Spain, in southern Europe) and southern Quebec (hereafter Quebec, in eastern North America) (Fig. 1.1), encompassing five subclimates (Kottek et al., 2006): steppe, dry Mediterranean and humid Mediterranean for Spain, and temperate and boreal for Quebec (see Fig. 1.1 and Table 1.1).

Table 1.1: Mean ± standard deviation of the climate variables (Mean Annual Precipitation (Prec, in mm), Mean Annual Temperature (Temp, in ºC) WorldClim, (Hijmans et al., 2005)), tree C stocks (C) (Mg C/forest ha), Forest bird richness (Bb), tree richness (Bt) and overall biodiversity (B), for the two regions (Spain and Quebec) and five subclimates (steppe, dry Mediterranean, Humid Mediterranean, temperate and boreal).

	Regions		Subclimates				
	Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Tempe-rate	Boreal
Prec	636 ± 238	993 ± 88	408 ± 53	535 ± 100	787 ± 271	993 ± 78	994 ± 106
Temp	12.9 ± 2.6	1.9 ± 1.6	14.6 ± 1.4	14.6 ± 1.9	10.7 ± 1.9	2.7 ± 1.3	0.4 ± 0.8
C	14.4 ± 15.4	45.2 ± 21.6	2.8 ± 2.8	9.2 ± 8.0	22.0 ± 18.8	49.4 ± 21.8	36.1 ± 18.3
Bb	29.3 ± 11.3	63.9 ± 10.3	18.7 ± 8.3	26.0 ± 10.1	36.4 ± 8.9	65.8 ± 9.4	55.6 ± 9.8
Bt	2.6 ± 1.3	4.1 ± 1.4	1.9 ± 0.8	2.3 ± 1.2	3.0 ± 1.4	4.6 ± 1.2	3.0 ± 1.0
B	0.8 ± 0.3	1.1 ± 0.2	0.6 ± 0.2	0.7 ± 0.2	1.0 ± 0.2	1.2 ± 0.2	0.9 ± 0.2

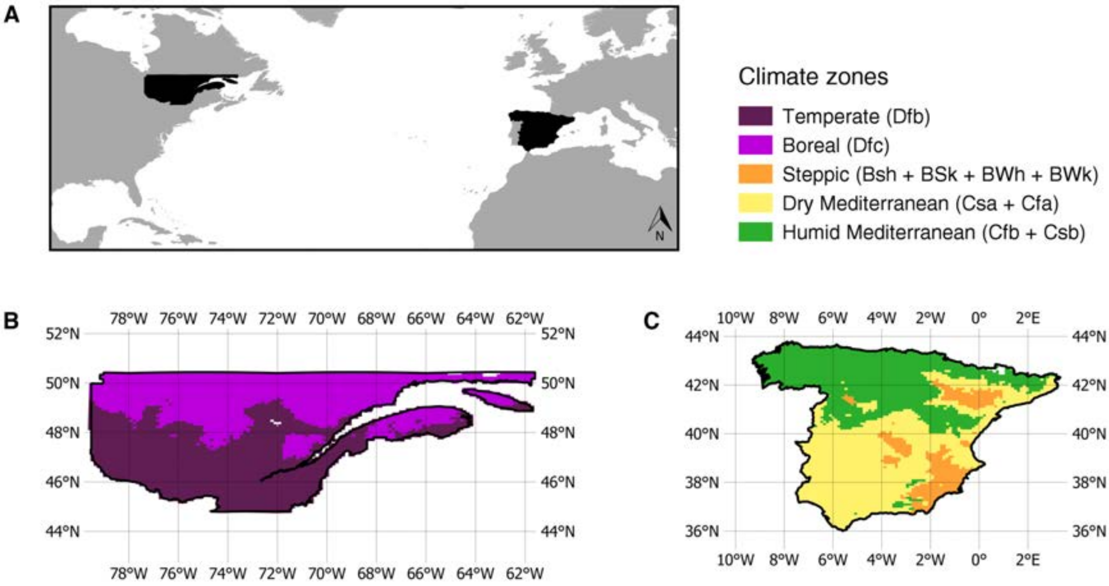


Figure 1.1: Study areas (in black, (A)) and subclimates in Quebec (B) and Spain (C) grouped from the Koppen-Geiger classification (Kottek et al., 2006).

Datasets: Forest Inventories and Breeding Bird Atlases

To compare datasets with different spatial scales, we up-scaled all variables at the coarser spatial scale, i.e., Bird Atlases, at 10 x 10 km (see Fig. 1.2).

Forest tree C stocks

For Spain, forest C stock data was obtained from the third Spanish National Forest Inventory (IFN3) conducted between 1997-2007 (Ministerio de Medio Ambiente, 2007*b*). The data consisted of a systematic sampling of permanent plots with a sampling density of one plot in every 1 km^2 of forest area, where trees above 7.5 cm were identified and measured within variable circular size plots (5 m radius for trees with DBH $\geqslant$ 7.5 cm, 10 m radius for trees with DBH $\geqslant$ 12.5 cm, 15 m radius for trees with DBH $\geqslant$ 22.5 cm and 25 m radius for trees with DBH $\geqslant$ 42.5). For Quebec, forest C stock data was obtained from the fourth Quebec Forest Inventory of temporary and permanent plots (PET4 and PEP4, respectively) conducted between 2000 and 2010, where all trees with a DBH above 9.1 cm were identified and measured. For the two regions, we selected plots not previously affected by disturbances (e.g., wildfires, pests, windfall, etc) or human intervention (e.g., cutting) to avoid their effecting our analyses. Moreover, we excluded plots without trees, resulting in a total number of plots of 51,677 and 76,016 for Spain and Quebec, respectively.

Tree biomass (above-ground + below-ground) of each live tree in each forest inventory plot were computed from DBH using species-specific allometric equations developed by Lambert et al. (2005) for Quebec and by Gracia et al. (2004) and Montero et al. (2005) for Spain. We then applied the widely established relationship of 1:0.5 between tree biomass and carbon (McGroddy et al., 2004). To up-scale to a 10 x 10 km cell grid, we computed the average value of all plots in each 10 x 10 km cell of the reference grid (Fig. 1.2). However, as that the number of plots within each 10 x 10 km cell was not proportional to the forest surface, we multiplied the average value of C by the percentage of forest surface. Thus, we produced maps in two different units: 1) mean forest C stocks of plots in each cell (Mg C/ forest ha); and 2) mean forest C stocks of plots per cell multiplied by the percentage of forest surface (Mg C/cell ha). Given that the values from both maps were highly correlated (r^2=0.89, p<0.01), we decided to use the second one (C stocks/cell ha).

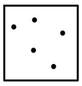
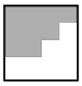
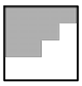
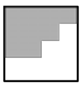
Variables	Source of information	Data type and resolution	Example in a 10 x 10 km cell	Upscaling method
C stocks (Mg/ha) C stocks (Mg/forest ha) Tree species richness Basal area (m^2/ha) Tree density (trees/ha)	Third Spanish National Forest Inventory Fourth Quebec Forest Inventory	Circular plots (diameter = 5 - 25 m) Circular plots (diameter = 22.6 m)		Average value of all plots in each 10x10 km cell
Bird species richness	Breeding Bird Atlas of Spain Second Breeding Bird Atlas of Quebec	10 x 10 km cell grid		
Percentage of forest	Spanish Forest Map (MFE)	Shape (1:50,000)		% of forest surface in each 10x10 km cell
Forest type	Système d'Information Forestière par Tesselle (SIFORT IV)	Raster (25 m)		Forest type with most of the surface in each 10x10 km cell
Mean slope	Digital Elevation Model of Spain Digital Elevation Model of Quebec	Raster (200 m) Raster (116 m)		Average value in each 10x10 km cell
Mean Annual Temperture Mean Annual Precipitation	WorldClim database (Hijmans et al., 2005)	Raster (30 seconds)		Average value in each 10x10 km cell
Subclimate	Koppen-Geiger climate classification	Raster (200 m)		Subclimate iwth most of the surface in ach 10x10 km cell

FIGURE 1.2: Source of information, data type, resolution, example and up-scale method of the variables used in the study.

Biodiversity

We computed biodiversity from two taxonomic groups of species: trees and birds. Trees can provide different habitats, whereas birds represent the group of vertebrates where we have an in- depth knowledge of their relationships with forest age, structure and composition with global coverage (Drapeau et al., 2000; Gil-Tena et al., 2007; Drever et al., 2008). Furthermore, bird assemblages and trees can be considered as complementary when used as indicators of biodiversity (Kati et al., 2004), thus increasing biodiversity surrogacy (Larsen et al., 2012). Therefore, we defined an overall biodiversity value as the standardized sum of tree and bird species richness.

Tree Species Richness (Bt) Using the previous forest inventory plots, we counted the number of tree species in each plot and computed the average tree species richness per plot in each 10 x 10 km cell. This provides an average diversity value at the stand scale.

Bird Species Richness (Bb) Bird data was obtained from the Breeding Bird Atlas of Spain and the second Breeding Bird Atlas of Quebec, which includes information in 10 x 10 km cells on the distribution of bird species during the period 1999-2002 for Spain (Martí and del Moral, 2003) and 2010-2014 for Quebec (Atlas des oiseaux nicheurs du Québec, 2016). We used the accumulative number of species detected in each 10-km cell to compute the species richness. Given that tree carbon and tree species richness were assessed in forested areas, we only considered forest bird species. Two groups of bird species were defined according to their degree of specialization in forest habitats: 1) forest specialists, species tightly linked to forested habitats both for feeding and nesting; and 2) forest generalists, species with a preference for forest habitats but can thrive in a wide range of environmental conditions and use a variety of different resources, e.g., feed in set-asides and grassland patches. For Quebec, only the fully surveyed cells were used, i.e., those with a minimum of 20 hours of sampling effort (Atlas of the Breeding Birds of Quebec, 2010). For Spain, field assistants were recommended to cover all habitats within each grid, with the aim to have visual or auditory contact with the maximum number of species in each habitat (Martí and del Moral, 2003).

Overall biodiversity (B) This variable was the result of the combination of bird and tree biodiversity within the 10 x 10 km grid cells. As tree species richness and bird species richness had different ranges of values, the same weighting of each group was ensured by standardizing the two variables to proportional values from 0 to 1. We then added the two values for each plot to account for overall biodiversity per grid cell.

Variables affecting forest C stocks and biodiversity

Forest stand variables

We obtained the following variables from each inventory plot: 1) basal area (m^2/ha); 2) tree density (trees/ha); and 3) structural diversity index (Hd), defined by Lei et al. (2009) as the tree size diversity index as follows

$$Hd = -\sum_{i=1}^{d} p_i \cdot log(p_i) \tag{1.1}$$

where p_i is the proportion of basal area for the ith diameter class and d is the number of diameter classes. We considered diameter classes of 4 cm. This index was used since tree diameter distribution is a good indicator of stand diversity (Buongiorno et al., 1994).

Landscape variables

We computed the following variables for each 10 x 10 km cell grid (Fig. 1.2): 1) percentage of forest, taken from the Spanish Forest Map (MFE) for Spain (Ministerio de Medio Ambiente, 2007*a*), and the Fourth Système d'information forestière par tesselle (SIFORT-4) for Quebec (MRNF 2007); 2) major forest type: deciduous, conifer or mixed forest, also from the MFE for Spain and SIFORT-4 for Quebec; and 3) mean slope, assessed from Digital Elevation Models (with a cell size of 200 m for Spain and 116.30 m for Quebec) (Fig. 1.2).

Climate variables

We obtained the following variables for each 10 x 10 km cell grid: 1) mean annual temperature; 2) annual precipitation, from the WorldClim database with 30 second resolution (Hijmans et al., 2005); and 3) subclimate type, defined by grouping the climate classifications from Koppen-Geiger (Kottek et al., 2006) into 5 subclimates: 1) Steppe: BSh + BSk + BWh + BWk; 2) Dry Mediterranean: Csa + Cfa; 3) Humid Mediterranean: Cfb + Csb; 4) Temperate: Dfb; and 5) Boreal: Dfc.

Hotspots/Coldspots delineation

We computed hotspots (areas with high values) and coldspots (areas with low values) for the following variables: overall biodiversity (B), forest tree richness (Bt), forest bird richness (Bb), and forest carbon stocks (Mg C/cell ha) (C).

After reviewing the previous articles for different methods to define hotspots (Alessa et al., 2008; Wu et al., 2013; Armenteras et al., 2015; Bagstad et al., 2015; Mora et al., 2016), we applied the 20th percentile, which is neither too broad nor too narrow, and defines 20% of the cells with the highest values (hotspots) and 20% with the lowest values (coldspots) for each variable across each region (Spain and Quebec) and within each subclimate. We then produced maps with the highest and lowest 20% values of B, Bt, Bb and C for each region (Spain and Quebec) and subclimate (steppe, dry Mediterranean, humid Mediterranean, temperate and boreal). For each region and subclimate, we overlapped the hotspots/coldspots map of C with those of the three biodiversity variables (B, Bt, Bb). Each of these three combined maps indicates the overlap of hotspots and coldspots of the two variables: 1) areas with hotspots of C and B (C+ B+, C+ Bb+ and C+ Bt+); 2) areas with hotspots of C but coldspots of B (C+ B-, C+ Bb- and C+ Bt-); 3) areas with coldspots of C but hotspots of B (C- B+, C- Bb+ and C- Bt+); 4) areas with both coldspots of C and B (C- B-, C- Bb- and C- Bt-); and 5) intermediate areas, without hotspots or coldspots in any of the variables considered.

Data analysis

To determine the factors that influence forest C stocks and biodiversity, we used Linear Models (LM) with C (in Mg C/cell ha), B, Bb and Bt as response variables. Tree density, structural diversity (Hd), slope, forest type, annual mean temperature and annual precipitation were selected as explanatory variables. We excluded basal area from the analysis since it was strongly correlated with both density and structural diversity (r=0.72 and 0.76, p-value<0.001, respectively) (Fig. A1.1 and Table A1.1). We also excluded the percentage of forest for its high correlation with tree C stocks (r=0.78, p-value<0.001)(Fig. A1.1 and Table A1.1).

To determine the relationships between forest C stocks and biodiversity, we established correlations of pairs of variables using the Pearson correlation test at p-value<0.001, after data transformation, to reach normality (we carried out a square root transformation of forest carbon stocks, both in Mg C/forest ha and Mg C/cell ha).

To quantify the degree of spatial overlap between areas of high (hotspots) and low (coldspots) values of C and biodiversity, we accounted for the percentage of 10 x 10 km cells in each of the following combinations: 1) areas with hotspots of C and B (C+ B+, C+ Bb+ and C+ Bt+); 2) areas with hotspots of C but coldspots of B (C+ B-, C+ Bb- and C+ Bt-); 3) areas with coldspots of C but hotspots of B (C- B+, C- Bb+ and C- Bt+) and 4) areas with both coldspots of C and B (C- B-, C- Bb- and C- Bt-). We considered that the highest percentage of overlap would be 100% (e.g., when the highest 20% of C values overlapped with the highest 20% of B values). To characterize these areas, we used generalized linear models with multinomial response (i.e., a dependent variable with 4 levels: C+ B+, C+ B-, C- B+ and C- B-, with any of the three biodiversity variables, B, Bb and Bt), with the same explanatory variables as the LM analyses. This analysis was not computed in the boreal subclimate for the overlapping of C with B and C with Bb because there was insufficient data.

All the analyses were carried out at the regional level (considering Spain and Quebec separately) and at the subclimate level (considering the five subclimates separately).

1.3 Results

Spatial patterns and factors influencing forest C stocks and biodiversity

The mean values of C stocks, bird richness, tree richness and overall biodiversity were, in general, higher in Quebec than in Spain (Fig. 1.3 and Table 1.1). For the spatial distribution in Spain, the highest values of C stocks and bird richness were located in northern Spain, while those of tree richness and overall biodiversity were detected in north-eastern Spain (in the humid Mediterranean subclimate). In Quebec, the highest values of C stocks and biodiversity (either Bb, Bt or B) were located in the southern area, corresponding to the temperate subclimate.

The factors determining C stocks were in general similar from those determining biodiversity, but varied between regions and subclimates (Table 1.3). For forest stand variables, both density and, especially, structural diversity had a strong positive effect on C stocks in both study regions and all subclimates considered (p-value < 0.001). Stand density and structural diversity were also positively associated with tree richness and overall biodiversity in the two regions and in most subclimates. Bird richness was significantly correlated with stand structural diversity in Spain across all subclimates (p-values

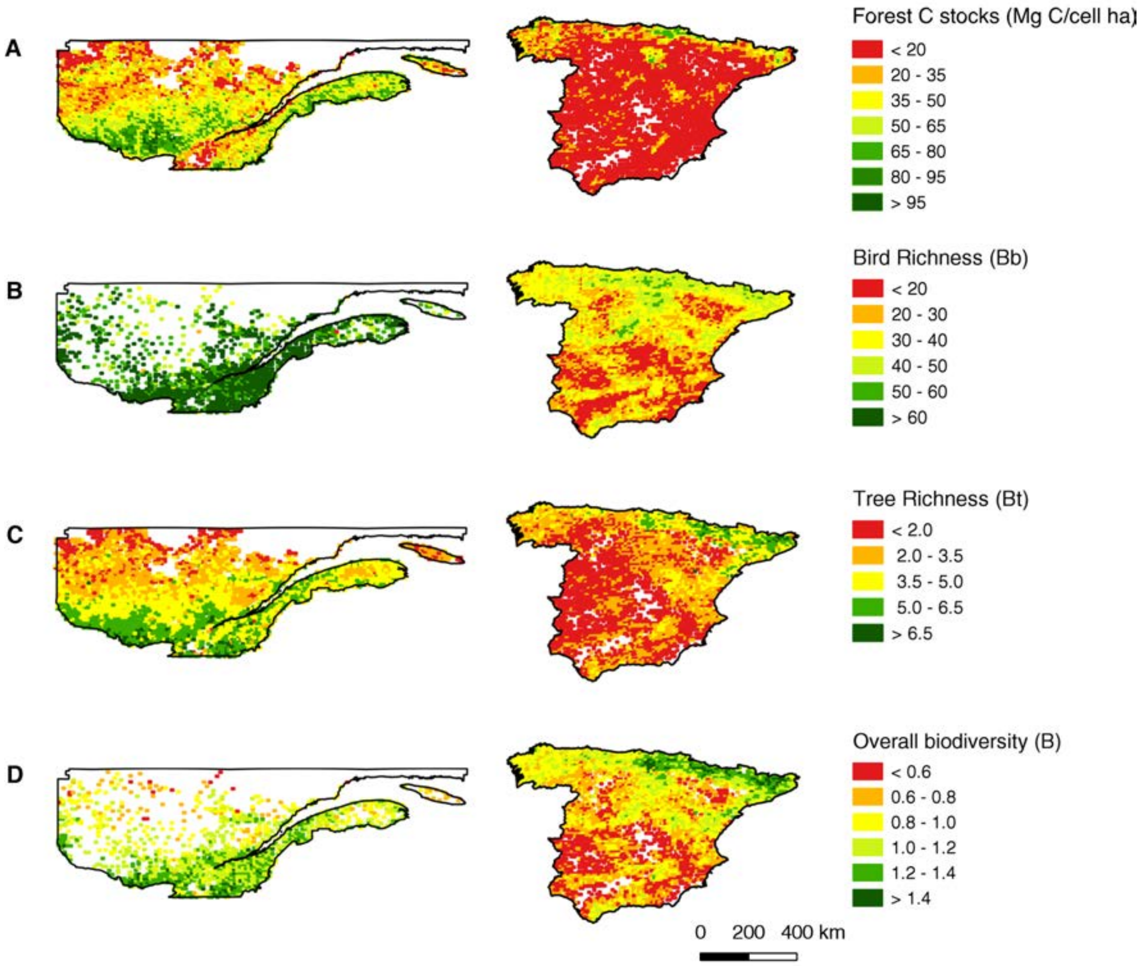


FIGURE 1.3: Spatial distribution of (A) C stocks (*C*, in Mg C/cell ha); (B) Bird Richness (*Bb*); (C) Tree richness (*Bt*); and (D) Overall biodiversity (*B*), for Quebec and Spain.

< 0.001 and < 0.01) but was not significantly associated with stand density (p-value > 0.1), except for the humid Mediterranean subclimate of Spain (p-value < 0.001). Regarding the landscape variables, slope had a significant positive effect on all variables investigated in both regions and all subclimates considered (p-value < 0.001, Table 1.3). Conifer forest cover had a strong negative effect on C stocks, both in Spain and Quebec, but it was particularly strong in the dry Mediterranean subclimate (t-value= -11.4; p-value < 0.001). The effect of broadleaves on C stocks was only positive in Quebec and for the boreal subclimate (p-value < 0.001). The effect of the climate variables varied between regions and among subclimates (Table 1.3). Mean annual temperature negatively affected C stocks in Quebec and in three of the five subclimates, steppe, dry Mediterranean and temperate. Mean annual temperature also had a negative effect on forest bird richness and overall forest biodiversity in Spain. On the other hand, mean annual precipitation

Table 1.2: Correlation tests (Pearson correlation) between carbon stocks (C) (in Mg C/cell ha) and the different components of biodiversity (Bird richness, Bb; Tree richness, Bt; overall biodiversity, B).

	Regions		Subclimates				
	Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Tempe-rate	Boreal
C - Bb	0.53 ***	0.14 ***	0.47 ***	0.41 ***	0.36 ***	0.08 *	0.24 ***
C - Bt	0.47 ***	0.53 ***	0.34 ***	0.25 ***	0.46 ***	0.42 ***	0.59 ***
C - B	0.59 ***	0.29 ***	0.52 ***	0.41 ***	0.51 ***	0.23 ***	0.50 ***

*$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

had a positive effect on C stocks and forest biodiversity (either Bb, Bt or B) in Spain, but a negative effect in both subclimates of Quebec.

Relationships between C stocks and biodiversity

There was a general significant and positive correlation between tree C stocks for the three biodiversity indices used both in Spain and Quebec and across the 5 subclimates (Table 1.2). Specifically, the correlation between tree C stocks and overall biodiversity was stronger in Spain than in Quebec (0.59 and 0.29, respectively). Among the studied subclimates, the strongest correlation was in the steppe (0.52) and humid Mediterranean subclimates (0.51) (Table 1.2). The correlation between tree C stocks and bird richness was also stronger in Spain than in Quebec (0.53 and 0.14, respectively), whereas the correlation between tree C stocks and tree richness was stronger in Quebec than in Spain (0.53 and 0.47, respectively), being the strongest in the boreal subclimate (0.59) (Table 1.2).

C and B hotspots and coldspots distribution and characterization

Overall, there was more spatial overlap of areas with high values of C and B (C+ B+, for any of B, Bb or Bt), or low values of C and B (C- B-, for any of B, Bb or Bt), than the rest of the combinations (Table A1.3). In Spain, 'win-win' areas (C+ B+ areas), were located in the north (in the humid Mediterranean subclimate), whereas overlapping of coldspots of C and B (C- B- areas) were located in the center and south-east (Fig. 1.4). In Quebec, 'win-win' areas for C and B (C+ B+) and C and Bb (C+ Bb+) were located in the south, whereas C+ Bt+ areas were located in the south-west (Fig. 1.4). But when we applied the same hotspots delimitation method within each subclimate (considering the overlap of the highest and lowest 20% values of C and B in each subclimate), we found more overlapping of coldspots of C and B in the south-western area of the humid Mediterranean subclimate (Fig. A1.2). We also found a higher percentage of overlap of both high and low values of C and B (C+ B+ and C- B- areas, respectively) in the boreal than in the temperate subclimate, and more trade-offs of C and Bb (more C+ Bb- and C- Bb+ areas) in the temperate subclimate (> 5.0%) than in the other subclimates (Table A1.3).

Table 1.3: Results of the linear models (t value and level of significance) for C stocks (C), bird richness (Bb), tree richness (Bt), and overall biodiversity (B).

	Regions		Subclimates				
Parameter	Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Temperate	Boreal
Forest carbon stocks, C							
Forest stand variables							
Density	17.2***	13.0***	3.6***	5.8***	17.1***	5.5***	20.1***
Structural diversity, Hd	30.9***	51.1***	7.9***	19.7***	26.1***	44.4***	30.9***
Landscape variables							
Slope	15.1***	19.8***	10.2***	15.1***	11.1***	23.9***	1.2
Forest type							
Conifer	-7.4***	-6.9***	1.4	-11.4***	2.5*	-2.2*	-2.8**
Broadleaf	-4.2***	3.4***	1.3	5.0***	-12.9***	-0.2	3.1**
Mixed	-1.8†	-2.7**	-0.2	-3.1**	-0.6	-3.6***	-2.2*
Climate variables							
Mean annual temperature	-1.5	-9.4***	-3.0**	-5.8***	8.4***	-16.1***	8.8***
Mean annual precipitation	20.8***	-3.8***	1.6	6.4***	6.6***	-1.8†	-0.7
df	4,261	3,797	342	2,007	1,896	2,608	1,181
R^2	0.58	0.61	0.45	0.35	0.60	0.61	0.71
Bird richness, Bb							
Forest stand variables							
Density	5.0***	-1.8†	0.3	-0.6	6.2***	-1.3	-0.4
Structural diversity, Hd	11.8***	0.8	2.9**	7.8***	7.3***	-0.3	0.9
Landscape variables							
Slope	11.1***	2.9**	3.6***	15.5***	4.9***	3.3***	1.0

Table 1.3: (continued)

	Regions		Subclimates				
Parameter	Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Temperate	Boreal
Forest type							
Conifer	-2.9**	-1.3	1.0	-1.2	0.6	-0.9	9.1***
Broadleaf	41.7***	18.7***	3.6***	19.6***	28.2***	16.6***	NA
Mixed	1.2	3.9***	0.2	1.2	0.8	3.2**	0.2
Climate variables							
Mean annual temperature	-25.0***	8.9***	-2.7**	-12.7***	-7.8***	6.5***	4.1***
Mean annual precipitation	5.0***	-4.1***	0.5	5.7***	-0.3	-3.6***	-2.1*
df	4,248	1,405	335	2,002	1,895	1,148	250
R^2	0.39	0.20	0.12	0.27	0.19	0.07	0.15
Tree richness, Bt							
Forest stand variables							
Density	18.7***	10.2***	0.0	12.9***	12.9***	10.6***	5.6***
Structural diversity, Hd	11.3***	26.6***	5.8***	7.3***	6.8***	23.2***	15.0***
Landscape variables							
Slope	14.0***	12.7***	0.5	10.5***	11.8***	12.5***	3.1**
Forest type							
Conifer	13.6***	-6.8***	2.7**	15.5***	6.1***	-4.6***	-0.5
Broadleaf	1.2	14.5***	-2.2*	2.6**	-4.6***	8.0***	4.5***
Mixed	13.5***	2.2*	0.0	12.4***	7.2***	2.0*	0.7
Climate variables							
Mean annual temperature	2.2*	34.6***	1.6	-5.6***	8.1***	27.0***	14.9***
Mean annual precipitation	9.1***	-11.4***	5.6***	11.7***	1.2	-9.9***	-4.3***

Table 1.3: (continued)

Parameter	Regions		Subclimates				
	Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Temperate	Boreal
df	4,261	3,797	342	2,007	1,896	2,608	1,181
R^2	0.38	0.71	0.20	0.43	0.31	0.60	0.58
Overall biodiversity, B							
Forest stand variables							
Density	14.8***	3.1**	0.3	7.0***	12.4***	4.1***	1.4
Structural diversity, Hd	14.9***	8.4***	5.2***	10.0***	9.0***	7.1***	3.7***
Landscape variables							
Slope	16.1***	6.0***	3.0**	17.6***	10.9***	7.2***	1.3
Forest type							
Conifer	6.3***	-4.0***	2.2*	7.9***	4.5***	-2.5*	9.6***
Broadleaf	12***	17.4***	1.6	16.2***	14.5***	14.0***	NA
Mixed	9.1***	3.4***	0.2	8.0***	5.3***	2.9**	2.9**
Climate variables							
Mean annual temperature	-15.7***	16.3***	-1.3	-12.7***	0.5	13.2***	6.5***
Mean annual precipitation	9.0***	-5.8***	3.2**	11.0***	0.7	-4.9***	-2.9**
df	4,248	1,405	335	2,002	1,895	1,148	250
R^2	0.48	0.52	0.19	0.45	0.32	0.34	0.48

Note: intercept is the forest type broadleaf.

†$P < 0.1$, *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

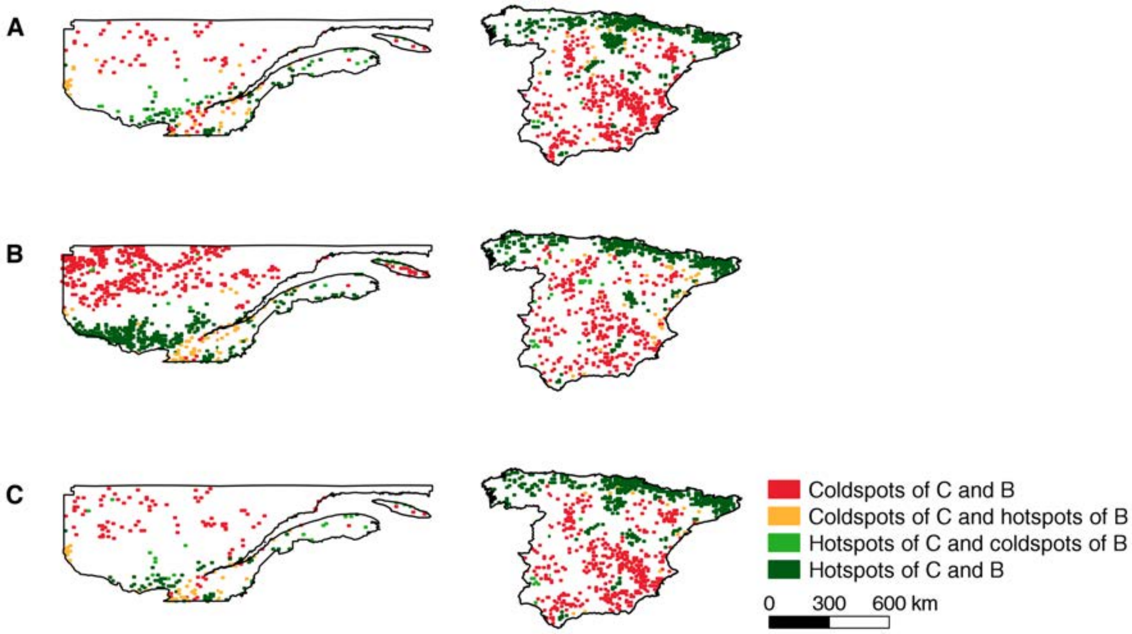


FIGURE 1.4: Synergies and trade-offs between C and B, i.e., overlap of the highest 20% (hotspots) and lowest 20% (coldspots) values of (A) Carbon (*C*) + bird richness (*Bb*); (B) Carbon (*C*) + tree richness (*Bt*) and (C) Carbon (*C*) + overall biodiversity (*B*) for the study areas.

The multinomial logistic models showed that both forest density and structural diversity positively affected C+ B+ areas (either Bb, Bt or B) in Spain and Quebec (Table A1.4, Table A1.5 and Table A1.6). The effect of density was only positive for C+ Bb+ in the humid Mediterranean subclimate, and for C+ B+ in the dry and humid Mediterranean subclimate (Table A1.4, Table A1.5 and Table A1.6). Slope positively affected C+ B+ (hotspots) areas (either Bb, Bt or B), both in Spain and Quebec, and in the majority of the subclimates. The conifer forest type showed a positive effect on C+ Bt+ in Spain in the dry Mediterranean subclimate (Table A1.5), whereas their effects were negative in C+ Bb+ in Quebec and the dry Mediterranean subclimate (Table A1.4). Moreover, broadleaves had a negative effect in C+ B+ areas (either Bb, Bt or B) in Spain and Quebec and in all their subclimates. Mean annual temperature had a negative effect on C+ Bb+ and C+ B+ areas in Spain, with the strongest effect on the dry Mediterranean subclimate, whereas it had a positive effect in the temperate subclimate (Table A1.4 and Table A1.6). Mean annual temperature had also a positive effect on C+ Bt+ areas in Quebec and the humid Mediterranean and boreal subclimates, but a negative effect in the dry Mediterranean subclimate (Table A1.5). Regarding mean annual precipitation, it positively affected C+ B+ areas (either Bb, Bt or B) in Spain and in the majority of the subclimates, and only negatively affected C+B+ in the temperate subclimate.

1.4 Discussion

Spatial patterns and factors influencing forest C stocks and biodiversity

Our results show high values of C stocks in Quebec, as well as in the temperate and humid Mediterranean subclimates (Fig. 1.3 and Table 1.1). These patterns are consistent with studies at global and regional scales that have also shown higher levels of C stocks in the temperate and humid Mediterranean subclimates than in the rest of subclimates of our study (Liu et al., 2012; Duchesne et al., 2016). Carbon storage is clearly associated with water availability in both regions, which is in line with previous studies (Bunker et al., 2005; Zhao and Zhou, 2006; Fischer et al., 2014). Biodiversity (either bird, tree or overall diversity) was also higher in Quebec than in Spain (Fig. 1.3 and Table 1.1). This agrees with previous studies showing that tree richness and productivity were higher in the temperate and boreal forests than in the Mediterranean forests (Liang et al., 2016).

In addition, density and structural diversity had a strong positive effect on C stocks and on tree and overall forest biodiversity (Table 1.3). Other studies have shown that forests with high stem density can be more productive (Vilà et al., 2013), while forests with a diverse structure contain species that occupy different vertical and horizontal layers favoring a better use of resources (e.g., light-adapted and shade-tolerant species) and reduce competition, thus enhancing productivity and C stocks (Paquette and Messier, 2011; Ruiz-Benito et al., 2014). Furthermore, it is well known that birds are affected by forest structure (Drapeau et al., 2000; Camprodon and Brotons, 2006; Nikolov, 2009). In our study, we found that bird richness in Spain was positively affected by structural diversity. Forests with a diverse structure are generally associated with a wider provision of microhabitats and understory plant species tied to each canopy species (Cavard et al., 2011) thereby supplying more diverse food resources and nest sites for different bird species than those with a more homogeneous structure (Gil-Tena et al., 2007; Ferger et al., 2014). In Quebec, neither stand characteristics of structural diversity nor stem density were associated with bird richness. Although other studies have shown negative effects of tree density on bird richness (Smith et al., 2008; Ameztegui et al., 2018), we found no effect in the majority of the subclimates considered (Table 1.3). This suggests that other habitat characteristics (e.g., amount of dead trees or single big trees) may be also important drivers (Drapeau et al., 2009; Grinde et al., 2017).

Regarding landscape variables, slope had a significant positive effect on C stocks and biodiversity in both the regions and all subclimates considered (Table 1.3). Some studies have suggested that slope was negatively correlated with bird richness because areas with steeper slopes are often characterized with more stressed conditions (Huang et al., 2009), lower resources and higher extinction rates (Kattan and Franco, 2004; Díaz, 2006). However, we found a positive effect in our study that might be related to difficult human accessibility and low-intensity management of isolated and mountainous areas often associated with steeper slopes (Vilà et al., 2013). Among the forest types considered, conifer forests had a strong negative effect on C stocks, because conifers in these two regions and at the 10 x 10 km cell grid scale are mainly found in less productive areas with more severe climatic conditions (Fig. A1.3d). This is, however, not always the case since in many parts of the world, higher tree productivity is often associated with conifer species, especially in plantations (e.g., *Pinus radiata*) (Romanyà and Vallejo, 2004; Ivković et al., 2016). In addition, the effect of forest type on bird richness in both regions agrees with numerous studies that show a higher bird richness in broadleaf than in conifer forests (Calviño-Cancela, 2013; Charbonnier et al., 2016; Cadieux and Drapeau, 2017).

Concerning climate variables, mean annual temperature negatively affected C stocks in all subclimates, except for the humid Mediterranean and the boreal zones, where the effect was positive (Table 1.3). Compared with the other subclimates of Spain, the humid Mediterranean subclimate had the most favorable growth conditions, as it has the highest amount of precipitation (Fig. A1.3). Therefore, we could expect an increase in forest productivity in the context of global warming, as long as precipitation does not become a limiting factor. Productivity in boreal forests depends strongly on temperature and solar radiation (Beer et al., 2010; Babst et al., 2013), so increasing temperature might also favor growth, productivity and C stocks (Pan et al., 2013). In contrast, mean annual temperature was found to negatively affect forest bird richness and overall forest biodiversity in Spain. Most of the forest birds in Spain are cold-dwelling species, as they are located in the southern limit of their distribution in Europe, so an increase in temperature can have a detrimental effect on these forest bird communities (Regos et al., 2017). On the other hand, mean annual precipitation was found to have a negative effect on C stocks and biodiversity in Quebec. Previous studies have shown that water scarcity limits biomass in dry ecosystems (Sankaran et al., 2005; Beer et al., 2010), but excessive water availability can have detrimental effects on other resources for plant growth, such as nutrient availability, decreasing productivity and C stocks (?Paquette et al., 2017).

Relationships between C stocks and biodiversity

The positive correlation between C stocks and tree diversity has been demonstrated in previous articles at both global and regional scales (Paquette and Messier, 2011; Gamfeldt et al., 2013; Liang et al., 2016), where forests with high tree diversity can use resources in a more efficient way through niche partitioning, thus having greater levels of productivity (Zhang et al., 2012). In our study, the positive correlation between C stocks (C) and overall biodiversity (B) and bird richness (Bb) was stronger in Spain than in Quebec (0.59 and 0.29, respectively), emphasizing that the gradients both in terms of C stocks and forest biodiversity were more contrasted in Spain than in Quebec (Table 1.2).

Looking at subclimates, the strongest correlations were in the steppe and the boreal subclimates likely due to high-contrasting climatic conditions (0.52 and 0.59, respectively). In addition, competition tends to be reduced in favor of complementarity in more stressful environments, and species interactions improve the availability and efficient use of resources (Paquette and Messier, 2011; Prior and Bowman, 2014; Forrester and Bauhus, 2016). Previous studies have suggested positive C/B relationships where climatic conditions limit productivity (Toïgo et al., 2015; Jucker et al., 2016). These results need to be considered at the first-stage of landscape planning and conservation, as a decrease of biodiversity could also have negative consequences on carbon storage.

C and B hotspots and coldspots distribution and characterization

Although significant relationships between ecosystem services are not always coupled with high spatial overlap between their hotspots (Chan et al., 2006), we found a high percentage of overlap between hotspots of C and B (from 29.5% to 89.5%, Table A1.3). In other words, we found more synergies than trade-offs (Fig. 1.4, Table A1.3), which implies that areas maintaining greater C stocks will also likely conserve higher levels of biodiversity, and vice versa. Landscape planning and conservation strategies with the aim to maintain both C stocks and biodiversity should be focused in the north of Spain and in southern (for C+ B+ and C+ Bb+) and south-western (for C+ Bt+) Quebec (Fig. 1.4). A high percentage of overlapping hotspots was also described in other studies (Egoh et al. (2009); Bai et al. (2011); Gos and Lavorel (2012), but see Anderson et al. (2009)), suggesting that carbon stocks support biodiversity, and vice versa. However, the selection of areas for strategic planning can vary greatly depending on the level of analysis, i.e., from regions to subclimates. Thus, coldspots that were not found in the regional delineation showed up in the humid Mediterranean subclimate (Fig. A1.2), while trade-offs of C and Bb emerged in the temperate subclimate (Table A1.3). We found that some of the variables driving C stocks and B separately were also drivers of their hotspots overlap (Table A1.4, Table A1.5 and Table A1.6). When density and structural diversity increase, the probability of having 'win-win' areas also increases while some trade-offs decrease. Our study also showed that slope enhanced C+ B+ (either Bb, Bt or B) areas. Given that steeper areas are less profitable for harvesting, these areas may be easily favored in conservation plans. Regarding forest type, conifers had a positive effect on C+ Bt+ areas in Spain, in the dry Mediterranean subclimate. Even though the effect of conifers on C was negative in this subclimate, this effect changed when high values of Bt were included, as tree richness enhanced forest productivity (Paquette and Messier, 2011; Ruiz-Benito et al., 2014). The negative effect of forest type on C+ Bb+ areas, and especially of broadleaves on C+ Bb-, may be because forests with high C values had other more important variables affecting Bb than forest type per se, such as having enough large and old trees, important for nesting for some species of birds (Remm et al., 2008). Finally, climate variables had similar effects on synergies than on C and B separately.

1.5 Conclusions

Implications for conservation and landscape planning

This study provides relevant insights into the spatial patterns of Carbon stocks (C) and Biodiversity (B) across five subclimates in Spain and Quebec. Our results highlight the importance of determining the spatial patterns of C and B, their relationships and drivers. Specifically, we have showed the 'win-win' areas where landscape planning strategies should be focused by developing conservation policies at the national or regional level that maintain C stocks and at the same time preserve biodiversity. Although our scale of analysis is quite broad, other studies suggested that patterns at this scale are robust for ecological applications (Anderson et al., 2009; Xu et al., 2017), being therefore critical to identify areas to develop specific management plans at more detailed scales. Our approach provides an essential framework for the fist-stage in strategic landscape planning to define forest conservation strategies and policies at global and regional levels. At global scales, such information is needed to help identify future priority areas for conservation in action plans such as the Convention on Biological Diversity's 2020 targets. This methodology could also be applied in tropical and subtropical forests for programs such as REDD+. At regional scales, this study is useful to define nation-wide green infrastructures, which contribute to biodiversity conservation and benefit human populations through the maintenance and enhancement of ecosystem services (Naumann et al., 2011). Besides, this information on C stocks and biodiversity hotspots could be incorporated, together with other ecosystem services indicators, into landscape decision support systems to help environmental decision making (e.g., exploitation of green areas or urban expansion). Furthermore, the strategic level of planning of this study could also be used to develop general objectives for all other management plans, providing long-term strategies for regional land use and landscape management (Tittler et al., 2001).

Climate change is expected to have far-reaching negative effects in the Mediterranean area, where forest productivity and its associated biodiversity are predicted to decline in the next decades due to more severe and frequent droughts as well as forest fires (Lindner et al., 2010). The temperate forests have been greatly affected by drought, insect outbreaks and wildfires, and the large amount of C stored in the cold or frozen soils of boreal forests has also been extremely sensitive to climate change (Gower et al., 2001). Forest and environmental planners should therefore integrate hotspots for C stocks and biodiversity into their strategic planning to promote 'win-win' solutions, in order to define national and regional actions that simultaneously support conservation and mitigate climate change.

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2 Are Protected Areas Preserving Ecosystem Services and Biodiversity? Insights from Mediterranean forests and shrublands

Abstract

Protected areas (PAs) are essential for biodiversity conservation and the provision of ecosystem services (ES), representing 15% of the earth's surface and targeted to increase until 17% by 2020. But previous studies showed different results on the effectiveness of PAs in preserving ES and biodiversity, which has implications for landscape conservation.

The main objectives of this chapter are 1) to know whether the spatial distribution of ES (carbon stocks and water provision), biodiversity (woody and bird richness) and conservation variables (threatened bird richness, habitats and geology) varies between PAs (with different protection status) and buffer zones; and 2) to quantify and compare the percentage of high values (hotspots) of ES, biodiversity and conservation variables inside PAs (with different protection status) and buffer zones. We analyzed 108 PAs from a Mediterranean region using linear mixed models with ES, biodiversity and conservation variables as response factors, and type of zone (PA vs buffer) and protection status (moderate vs partial) as fixed factors.

We found higher values of carbon stocks in PAs than in buffer zones. We also found more coverage of community-interest habitats, priority-habitats and geological-interest sites in PAs than in buffer zones. However, PAs with higher degree of protection did not provide higher levels of ecosystem services and biodiversity, or vice versa. We found more hotspots of woody richness, bird richness and threatened bird richness in buffer zones than in PAs. This study highlights the importance of landscape planning in conservation, which should include PAs within broader landscapes by considering also their buffer zones and non-PAs. It also emphasizes the importance of integrating ES and biodiversity to define effective conservation policies.

2.1 Introduction

Protected Areas (hereafter PAs) are the main focus of conservation strategies. Currently, PAs represent 15% of the earth's surface and should increase to 17% by 2020 (Convention on Biological Diversity, 2010). PAs have been established to avoid deforestation, preserve iconic landscapes and ecosystem representativeness, as well as to protect biodiversity and charismatic or endangered species (Eken et al., 2004; Hannah, 2008). However, a considerable proportion of the world's PAs is being ineffective at achieving conservation targets, such as maximizing biodiversity and maintaining species populations (Wiersma and Nudds, 2009; Geldmann et al., 2013).

Part of the conservation community claims that conservation should look at protecting, restoring and enhancing the services that nature provides to people (Doak et al., 2014), whereas others suggest that both biodiversity and ecosystem services (i.e., intrinsic and instrumental values, respectively) need to be included to achieve conservation objectives (Reyers et al., 2012). Although PAs provide multiple ecosystem services (hereafter ES), previous studies have generated diverse results regarding the type of ES. Some studies have showed that provisioning ES are more often found outside than inside PAs (Castro et al., 2015; Mukul et al., 2017), whereas others have found no negative impact of protection on provisioning services (Eastwood et al., 2016). Concerning regulating services, PAs are found to maintain carbon stocks and mitigate climate change by preventing deforestation, especially because forests are one of the most protected ecosystems (Rodríguez et al., 2013; Vačkář et al., 2016). Moreover, water-regulating services that are essential for ecosystem functioning (e.g., streamflow and erosion control) and for human well-being (e.g., water quality) are found to be more preserved inside than outside PAs (Quijas et al., 2012; Dos Santos et al., 2018). PAs also supply multiple cultural services such as environmental education, recreation and aesthetic values (Palomo et al., 2013; Vlami et al., 2017), as well as socioeconomic benefits for local people (Oldekop et al., 2016). Therefore, there is not a decisive agreement regarding the effectiveness of conservation strategies in maintaining ES. One of the reasons of this divergence is that PAs should be considered with a landscape broader perspective, considering their surrounding land cover and land uses, because landscapes and species are dynamic so that buffer zones surrounding PAs as well as other non-PAs can be relevant in conservation planning (Wiens, 2009).

Biodiversity conservation has been one of the main priorities for establishing PAs. Some studies have indicated that PAs are an effective tool to maintain global (Butchart et al., 2012) and tropical biodiversity (Bruner et al., 2001), whereas others have suggested that current PAs are not sufficiently effective in conserving biodiversity at the global level, as PAs are neither ecologically representative nor efficiently allocated (Pimm et al., 2014; Venter et al., 2014). However, effective conservation strategies need to take into account that biodiversity has multiple organizational levels and spatial scales (Wu, 2008) and that PAs exist within broader landscape mosaics that allow or interfere in the movement of species (Wiens, 2009). PAs have been proved to be beneficial for species of conservation interest, such as endemic birds and endangered species (Le Saout et al., 2013). Together with the protection of particular species, the conservation of habitats has been claimed to be integrated in PA management (Brilha, 2002), because habitats with a good conservation status can provide more biodiversity and ES than habitats with unfavorable conservation status (Maes et al., 2012). Specifically, habitats of biological value, in danger of disappearance or with a small natural range have been a focus of protection (92/43/EEC). Previous studies have showed that PAs have lower rates of habitat loss than non-PAs (Joppa and Pfaff, 2011; Geldmann et al., 2013), but others have suggested that PAs have not been effective in preventing habitat conversion (Clark et al., 2013). In addition, particular geologies (e.g., geological heritage sites) are also claimed to be considered inside PAs (Gordon et al., 2017) as geology is considered a framework for life on Earth (Brilha, 2002). The importance of geological heritage conservation has been recognized by international institutions such as UNESCO and IUCN (IUCN, 2008).

The degree of protection in natural areas might influence the ES they provide. Strictly PAs (e.g., natural reserves) are found to provide the highest carbon storage by preventing the conversion from forests to agriculture or tourism areas (Castro et al., 2015). However, PAs with non-strict protection (e.g., Natura 2000 sites) are also found to be important for preserving ecosystem services and biodiversity (Gaston et al., 2008*a*; Bastian, 2013). In fact, regulating and provisioning services (e.g., fodder and water) are the highest in areas with lower-levels of protection, as local stakeholders are allowed to maintain traditional management of ecosystems that ensure the delivery of ES (Castro et al., 2015). Specifically, non-strict PAs are important for regulating services such as water purification or regulation (Castro et al., 2015; Manhães et al., 2016) and for cultural services such as socioeconomic outcomes for local people (Oldekop et al., 2016).

The identification of areas with high values (hereafter 'hotspots') for different ES and biodiversity is essential to know whether priority areas for ES and biodiversity are located inside or outside PAs. Some studies have suggested that most of the ES hotspots are included inside PAs (García-Nieto et al., 2013), indicating that the conservation strategy provides ES. In contrast, other studies have showed that substantial portions of hotspots of ES are located outside PAs (Davids et al., 2016).

Given the above considerations, we aim to determine the role of PAs in preserving ES and biodiversity, with Catalonia as a case study. It is a region located in the Mediterranean Basin with around 60% of its surface covered by forests and shrublands. PAs in Catalonia are under different administrative levels (European, national, regional and provincial) and different protection levels (from strict to partial PAs). Together, all PAs in Catalonia cover 31% of the territory. Specifically, we aim 1) to know whether the spatial distribution of ES (carbon stocks and water provision), biodiversity (woody and bird richness) and conservation variables (threatened bird richness, habitats and geology) varies between PAs (with different protection status) and buffer zones; and 2) to quantify and compare the percentage of high values (hotspots) of ES, biodiversity and conservation variables inside PAs (with different protection status) and buffer zones. We considered the following hypotheses:

- *Hypothesis 1*: PAs in this region are a useful tool to preserve ES and biodiversity, thus there are higher levels of ES, biodiversity and conservation variables (in the whole range of the variables and their hotspots) in PAs than in buffer zones, especially carbon stocks (*hypothesis 1.1*) and biodiversity (*hypothesis 1.2*). There is more water provision in buffer zones than in PAs because there PAs have more forest than buffer zones and highly vegetated areas consume more water and contribute to the avoidance of water runoff, resulting in less water provision (*hypothesis 1.3*).
- *Hypothesis 2*: regarding the protection status, higher levels of protection have higher levels of ES, biodiversity and conservation variables (in the whole range of variables and hotspots) (*hypothesis 2.1*), except for bird richness and habitats of interest, that are higher in partial than moderate PAs because these areas are mainly Natura 2000 sites, designated for these purposes (*hypothesis 2.2*).

2.2 Methods

Study area

The study was carried out in Catalonia (NE Spain), a region located between 40º 5’ and 42º 9’ latitude North and 0º 2’ and 3º 32’ longitude East (Fig. 2.1). Catalonia has a heterogeneous geomorphology and a large climatic gradient. It encompasses mountainous areas such as the Pyrenees (up to 3,143 m.a.s.l), inland agricultural plains and coastal zones along the Mediterranean Sea. The climate is Mediterranean, with a mean annual temperature of 12.5 ºC and a mean annual precipitation of 739 mm (Hijmans et al., 2005). Around 60% of the area is covered by forests and shrublands (CREAF, 2005), the ecosystems considered in this chapter.

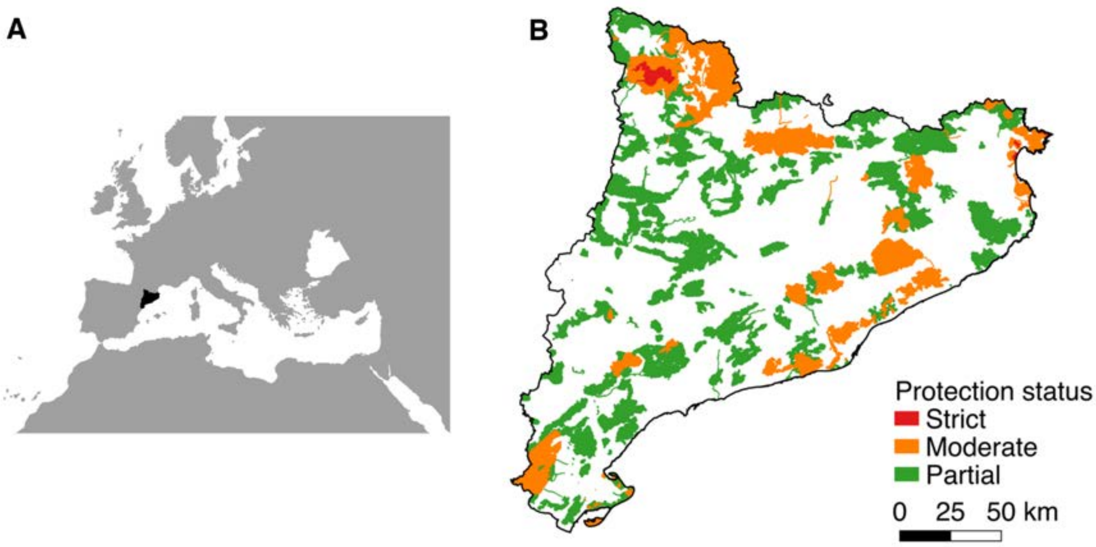

FIGURE 2.1: (A) Location of the study area (in black) and (B) protected areas with different protection status.

Type of zones

To determine the role of PAs in preserving ES and biodiversity in Catalonia, we considered three types of zones (Fig. 2.1): a) protected areas; b) buffer zones; and c) other unprotected areas. These three types are described as follows:

Protected areas

In Catalonia, 31% of the surface is under some degree of protection. As the World Database on Protected Areas (WDPA) does not reflect the regional variability on protection status in Catalonia, we grouped the existing PAs in Catalonia in 3 groups depending on their protection status (Fig. 2.1):

Strict protected areas Natural areas with a high ecological and cultural values, with little transformation from exploitation or human activities. Their protection is due to the beauty of their landscapes, the representativeness of their ecosystems or the singularity of their flora, fauna, geology or geological formations, that have important ecological, aesthetic, cultural, educative and scientific values and whose conservation deserves priority attention (`http://www.mapama.gob.es/es/red-parques-nacionales/sig/`). In the study area, this category includes a national park and two strict nature reserves.

Moderate protected areas Natural areas with a medium conservation level that allow traditional and cultural management practices. This includes 11 natural parks, 53 small partial natural reserves, 1 buffer zone of national park, 1 buffer zone of natural park, 7 national interest sites and 12 provincial parks.

Partial protected areas It includes 83 Natura 2000 areas without any of the abovementioned protected status. Natura 2000 is a network aiming to ensure long-term survival of most valuable and threatened species and habitats of Europe, listed in the Birds Directive (2009/147/EC) and the Habitats Directive (92/43/EEC). In cases of overlapping polygons with the same protection status (e.g., partial natural reserves inside natural parks), we dissolved them into one polygon. In cases of overlapping of polygons with different protection status, we classified them into the highest protection status.

Buffer zone

To provide similar environmental conditions and to avoid heterogeneity caused by location, we defined a buffer area of 5 km from each PA boundary (Fig. 2.1). These areas were used to compare the effect of protection with PAs in the provision of ES, biodiversity and conservation variables and covered the 54% of the study area.

Ecosystem services, biodiversity and conservation variables

In order to have the same scale for the two main data sets (forest inventory plots and Breeding Bird Atlas), we defined a reference grid at 1 x 1 km resolution. All variables were computed at this scale and were afterwards upscaled at the PA/buffer scale (Table A2.1).

Ecosystem services

Carbon stocks There were four different grid nodes including different sources of information: a) nodes including a forest inventory plot, according to the Land Cover Map of Catalonia (CREAF, 2005); b) nodes without forest inventory plot but including trees; c) nodes without forest inventory plot and corresponding to shrubland; and d) nodes corresponding to other land cover types (grasslands, outcrops, agricultural areas, etc). We computed carbon stocks for the first three types of nodes.

- Forest with inventory data. The third Spanish National Forest Inventory (IFN3) was conducted in Catalonia between 2000 and 2001 (Ministerio de Medio Ambiente, 2007*b*). The data consisted of a systematic sampling of permanent plots with a sampling density of one plot in every 1 km^2 of forest area, where woody species were identified and measured within variable circular size. Tree biomass (aboveground + belowground) of each live tree in each forest inventory plot was computed from DBH using species-specific allometric equations developed by Gracia et al. (2004) and Montero et al. (2005).
- Forest without inventory data. In these plots, the dominant tree species were identified using the Land Cover Map of Catalonia (CREAF, 2005) (Appendix 2 - Carbon stocks methodology). Aboveground biomass was computed with statistical models from LIDAR data, that were calibrated using forest inventory plots from the third Spanish National Forest Inventory (Ministerio de Medio Ambiente, 2007*b*). Belowground biomass was computed using the same species-specific allometric equations as outlined for plots with inventory data (previous section).
- Shrubland. After selecting the nodes without forest inventory plot but classified as shrubland, according to the Land Cover Map of Catalonia (CREAF, 2005), we selected the data from the third Spanish National Forest Inventory (Ministerio de Medio Ambiente, 2007*b*) that was analogous to shrublands (i.e., open-forests inventory plots with basal area < 5 m^2/ha). We grouped them into similar biomass

groups depending on their main species following Pasalodos-Tato et al. (2015) to obtain mean values of shrub carbon in relation with their dominant shrub species. In each node of this category, we carried out a photointerpretation of aerial images to estimate shrub-cover, and dominant species were identified using the Catalan Habitats Map (Carreras and Ferré, 2014). Then, shrub biomass in each node was computed following the expression:

$$Shrub\ biomass = \overline{b_i} \cdot c \tag{2.1}$$

where $\overline{b_i}$ is the average value of biomass of the species i and c is the percentage of shrub coverage at the node (Appendix 2 - Carbon stocks methodology). To obtain carbon stocks from biomass data in the three types of nodes above stated, we applied the relationship of 1:0.5 between biomass and carbon (McGroddy et al., 2004).

Water provisioning Blue water was defined as the sum of water exported daily via runoff and deep drainage. Annual sums of daily blue water were calculated for each 1 km cell by applying the water balance model described in De Cáceres et al. (2015). Details of each of the eco-hydrological processes considered and species parameter values of the model are given in De Cáceres et al. (2015) and in Appendix 2 - Water balance model.

We calculated the proportion of annual blue water over annual precipitation, and selected the 10-year period 1993-2002 to account for the interanual variability in the water balance and to adjust to the periods of the rest of the data (Table A2.1).

Biodiversity

We computed biodiversity from two taxonomic groups of species: woody species and birds. We used these two taxonomic groups of species because woody species include trees and shrubs so they can provide different habitats, whereas birds represent the group of vertebrates where we have one of the most extensive databases and an in-depth knowledge of their relationships with forests at global scale (Gil-Tena et al., 2007; Drever et al., 2008).

Woody species richness Woody species richness was computed in the same three grid nodes as in the Carbon stocks section.

- Forest with Inventory data. In the nodes with data from the IFN3, we counted the number of woody species (tree and shrubs) in each plot.
- Forest without Inventory data. We developed two linear models, one for tree species richness and the other one for shrub species richness as response variables, respectively (Table A2.4). The explanatory variables in the models were: location (x and y coordinates), shrub carbon stocks (Mg C/ha), tree carbon stocks (Mg C/ha), slope (º), mean annual temperature (Hijmans et al., 2005), mean annual precipitation (Hijmans et al., 2005) and main forest species. We therefore applied these models (R^2=0.24, p-value < 0.001 and R^2=0.51, p-value < 0.001 for tree and shrub richness, respectively) to forest nodes without inventory data. We computed woody species richness in a plot by adding tree and shrub species richness values obtained for this plot (Table A2.4).
- Shrubland. We developed a linear model for shrub species richness using data of open forest inventory plots (i.e., plots with basal area < 5 m^2/ha, the most similar to shrublands) (Appendix 2 - Tree and shrub richness linear models). The explanatory variables were location (x and y coordinates), shrub carbon stocks (Mg C/ha), slope (º), mean annual temperature (Hijmans et al., 2005) and mean annual precipitation (Hijmans et al., 2005). This model (R^2=0.49, p-value < 0.001) was applied to obtain shrub species richness in nodes without forest inventory plot and corresponding to shrubland, according to the Land Cover Map of Catalonia (CREAF, 2005) (Table A2.4).

Bird richness We used the accumulative number of bird species observed in each 1 x 1 km cell from the second Catalan Breeding Bird Atlas (Estrada et al., 2004). It was conducted in 1999-2002 at a 10 x 10 km cell and downscaled to 1 x 1 km cell. Given that ecosystem services were only assessed in forest and shrub areas, we only considered those bird species having fore st (both forest specialist and forest generalist) and shrub habitats.

Conservation variables

As conservation strategies rely on PAs, we included variables directly related to conservation, most of them coming from two of the powerful international legal tools for nature protection (Birds and Habitats Directives).

Threatened bird richness The Birds Directive aims 'to protect the 500 wild bird species naturally occurring in the European Union' (2009/147/EC). From these, 194 species are particularly threatened and are included in the Annex I from the Directive. We selected bird species with forest and shrub habitats that were listed in the Birds Directive annex (2009/147/EC), because they represent a conservation value at the European extent and are subject of special conservation measures. We therefore counted the number of these species within each 1 x 1 cell from the second Catalan Breeding Bird Atlas (Estrada et al., 2004).

Habitats The Habitats Directive (92/43/EEC) has listed habitats of European concern to ensure biodiversity through the conservation of rare and characteristic natural habitat types (92/43/EEC). As one of the most important aspects in conservation strategies is the maintenance of habitats, we have defined two types of habitats from the Habitats Directive (92/43/EEC), from which we quantified their percentage of surface in each PA and buffer zone: habitats of Community interest and priority habitats.

- Habitats of Community interest, defined as natural habitats types that '1) are in danger of disappearance in their natural range; or 2) have a small natural range following their regression or by reason of their intrinsically restricted area; or 3) present outstanding examples of typical characteristics of one or more of the five following biogeographical regions: Alpine, Atlantic, Continental, Macaronesian and Mediterranean' (92/43/EEC). As the map has different habitats in the same polygon, we selected the habitats classified as forested and shrublands habitats having the highest coverage within each polygon.
- Priority habitats. We selected the habitats from the previous classification defined as 'natural habitat types in danger of disappearance [...] and for the conservation of which the Community has particular responsibility in view of the proportion of their natural range which falls within the territory', also defined as 'threatened to disappear in the EU' (92/43/EEC).

Geological-interest sites Geology has important conservation values for being part of all natural systems, providing framework for life on Earth, thus it is claimed to be integrated into PA management (Brilha, 2002). The inventory of Catalan geological-interest sites is a map with a selection of rocky outcrops and geological-interest sites needed to be preserved as geological heritage. This inventory is defined as a framework for decision-making in landscape planning and management (Carreras and Druguet, 2010). We quantified the percentage of surface of geological-interest sites in each PA and buffer zone.

Hotspot delimitation

We applied the 20th percentile (Armenteras et al., 2015; Mora et al., 2016), which defines 20% of the cells with the highest values for each variable. We computed each variable at the 1 x 1 cell scale (Table A2.1) to obtain a numerical value for each variable in each cell. We then selected the 20 % of cells with the highest values for each variable and defined them as hotspots. We generated one hotspot map for each ES, biodiversity and conservation variable.

Comparison between protected areas and buffer zones

To quantify and compare PAs (with different protection status) and buffer zones in terms of ES, biodiversity and conservation variables considering the whole range of the variables (i.e., not their hotspots), we computed the average value of each variable of all nodes in the PA and in the buffer zone. As the surface of forest and shrubs was not proportional to the total surface of the protected/buffer area, we weighted the average value of each forest and shrub variable by the percentage of surface of forest and shrubs, respectively (except for geological-interest sites, that were not dependent on forest and shrub surface). To obtain the averaged value for each variable in each PA/buffer area, we summed and divided them by the total percentage of forest and shrubs, as follows:

$$X = \frac{(\overline{x}_f \cdot a_f) + (\overline{x}_s \cdot a_s)}{a_f + a_s} \tag{2.2}$$

where $\overline{x}_f$ and $\overline{x}_s$ are the average of the variable in forested (f) and shrubland (s) areas, respectively; and a are the percentage of forest (f) and shrub (s) area within each protected/buffer area.

In the case of hotspots, we quantified the percentage of hotspots of each variable (i) in each PA/buffer zone (j) as follows:

$$hotspots = \frac{s_i^j}{s_i} \cdot 100 \tag{2.3}$$

where s_i^j is the surface of hotspots of the variable i in the PA/buffer zone j and s_i is the total area of hotspots of the variable i in the whole study area.

Data analysis

To determine the differences in ES, biodiversity and conservation variables between PAs and buffer zones, as well as in PAs with different protection status, we used linear mixed models with the ES, biodiversity and conservation variables as response variables (carbon stocks (Mg/ha), water, biodiversity – woody richness, bird richness –, threatened bird richness, habitats of community interest (%), priority habitats (%) and geological-interest sites (%)). Data on habitats of community interest, priority habitats and geological-interest sites were transformed (using square root, logarithm and square root, respectively) to meet the assumptions of normality of residuals. Fixed factors were type of zone (PA or buffer zone), protection status (moderate and partial, because due to the low sampling size, i.e., 3, we excluded strict-PAs from the analyses) and their interaction. We include each PA, together with their buffer zone, as a random effect to account for masking-effect of location. In the case of hotspots analysis, the same analyses were carried with percentage of hotspots surface as response variable. We log-transformed carbon stocks and geological-interest sites and calculated the square root of the rest of the hotspots variables to reach normality. In this case, we also included the area of forests and shrubs in each PA as a fixed factor.

2.3 Results

Ecosystem services, biodiversity and conservation variables in protected areas and buffer zones

The results of linear mixed models showed that carbon stocks were significantly higher in PAs than in buffer zones, whereas water provision was not significant (Table 2.1 and Fig. 2.2). None of the biodiversity variables considered showed differences between PAs and buffer zones. However, conservation variables of community-interest habitats, priority habitats and geological-interest sites showed more coverage inside PAs than in buffer zones (Table 2.1 and Fig. 2.2).

Focusing on PAs, we found that carbon stocks and geological-interest sites were significantly different depending on the protection status (moderate or partial), as there were more carbon stocks and more surface of geological-interest sites in moderate than in partial PAs (Table 2.1 and Fig. 2.2).

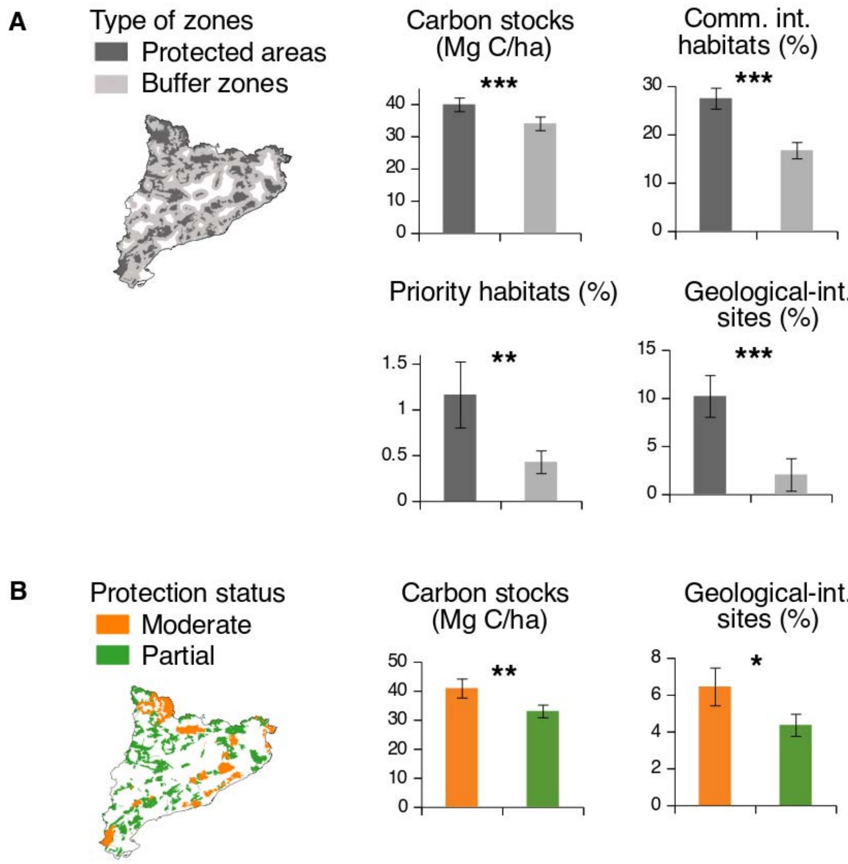

Figure 2.2: Plot bars of mean values and standard error of the studied variables showing significant differences between a) the types of zones (protected and buffer) and b) the protection status (moderate and partial). Signification codes: *** < 0.001, ** < 0.01, * < 0.05.

Hotspots of ES, biodiversity and conservation variables inside and outside PAs

The highest values (i.e., hotspots) of the variables showed a different pattern than considering their whole range (Table 2.2 and Fig. 2.3). Although hotspots of carbon stocks, bird richness, community-interest habitats and priority-habitats showed more coverage in PAs than in buffer zones (Table A2.3), the results varied when applying the mixed models. Thus, we found more hotspots of woody richness, bird richness and threatened bird richness in buffer zones than in PAs (Table 2.2 and Fig. 2.3). However, hotspots of priority habitats and geological-interest sites followed the same pattern as when considering the whole range of the variables (i.e., higher in PAs than in buffer zones). When considering protection status, none of the variables showed significant differences between protection levels (Table 2.2).

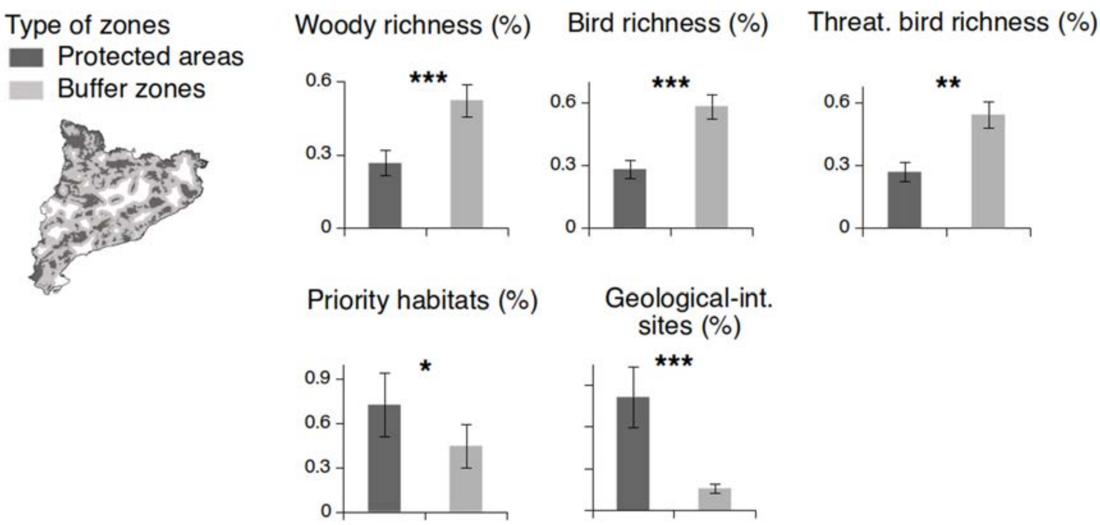

FIGURE 2.3: Plot bars of mean values and standard error of hotspots showing significant differences between the types of zones (protected and buffer). Signification codes: *** < 0.001, ** < 0.01, * < 0.05.

Table 2.1: Results of the linear mixed models (t-value and level of significance) for ES, biodiversity and conservation variables.

	Ecosystem services		Biodiversity		Conservation			
	Carbon stocks (Mg C/ha)	Water provision	Woody richness	Bird richness	Threatened bird richness	Habitats		Geology
						Comm.int habitats (%)	Priority habitats (%)	Geological-interest sites (%)
Intercept	13.1***	15.5***	20.4***	15.0***	20.5***	15.4***	-8.4***	12.8***
Type of zone: buffer zone	-3.8***	1.8	0.7	-1.6	1.0	-4.5***	-3.1**	-6.5***
Protection status: partial	-2.8**	0.8	0.5	-1.1	0.9	-0.2	-0.3	-2.4*
Buffer zone · partial	2.6*	-0.3	1.3	1.0	-0.5	0.1	2.0	1.9

***$P < 0.001$, **$P < 0.01$, *$P < 0.05$.

Table 2.2: Results of the linear mixed models (t-value and level of significance) for hotspots of ES, biodiversity and conservation variables.

	Ecosystem services		Biodiversity		Conservation			
	Carbon stocks (Mg C/ha)	Water provision	Woody richness	Bird richness	Threatened bird richness	Habitats		Geology
						Comm.int habitats (%)	Priority habitats (%)	Geological-interest sites (%)
Intercept	-20.6***	7.1***	2.6*	2.0	1.9	5.7***	1.9	-11.7***
Area of forests and shrubs	8.1***	18.6***	8.3***	12.6***	9.8***	14.3***	3.4**	5.3***
Type of zone: buffer zone	-0.03	1.6	3.7***	4.7***	3.4**	-1.7	-2.1*	-5.3***
Protection status: partial	0.03	0.4	0.5	0.6	1.7	0.2	0.4	-0.7
Buffer zone · partial	0.02	0.3	-0.2	-0.4	0.8	-0.02	1.8	2.1*

***$P < 0.001$, **$P < 0.01$, *$P < 0.05$.

2.4 Discussion

Ecosystem services, biodiversity and conservation variables in protected areas and buffer zones

Overall, some of our findings agreed with our initial hypotheses. As expected in *hypothesis 1.1*, carbon stocks were higher in PAs than in buffer zones (Table 2.1 and Fig. 2.2). We expected more water provision in buffer zones than in PAs (*hypothesis 1.3*), but we did not find significant differences between them. PAs have been shown to be effective in avoiding deforestation and consequently maintaining carbon storage (Andam et al., 2008; Vačkář et al., 2016). But this is not the case of Catalonia as many other European and Mediterranean areas, where forest surface increased since the mid-20th century due to agricultural abandonment (Bielsa et al., 2005; Améztegui et al., 2010). However, the percentage of forests and shrublands in our study is higher in PAs than in buffer zones (51 ± 27% of forests and 22 ± 19% of shrubs in PAs; 36 ± 22% of forests and 14 ± 8% of shrubs in buffer zones). Most PAs are located in mountainous areas, where the highest values of carbon stocks are found (i.e., north of Catalonia, Pyrenees and Pre-Pyrenees, as well as coastal mountainous plains) (Fig. 2.1). Other studies have already showed that location of PAs is biased towards higher elevation, steeper slopes and greater distances to roads and cities (Joppa and Pfaff, 2009), and that forests have higher protection coverage than other land-uses (Hermoso et al., 2018). Besides, contrarily as expected (*hypothesis 1.3*), no significant differences in water provisioning between PAs and buffer zones were found (Table 2.1). In this region, there is more forest percentage in PAs than in buffer zones. As forests use more water and contribute more to the avoidance of water runoff than shrublands, water provision and runoff was expected to be lower in forests than in shrublands (Brauman et al., 2007). But PAs have also more percentage of shrubs than buffer zones, which results in higher levels of water provisioning than expected. Thus, no significant results suggested that other factors operating at local scales can be influencing water provisioning, such as microclimatic effects or the particular type of vegetation (e.g., species with different root depths) (Scott et al., 2000; Brauman et al., 2007).

Although we expected higher biodiversity in PAs than in buffer zones (*hypothesis 1.2*), none of the biodiversity variables showed differences between PAs and buffer zones (Table 2.1 and Fig. A2.1). These results reinforce that PAs exist within broader landscape mosaics that allow or interfere in the movement of species (Wiens, 2009). Additionally, we analyzed the results of a qualitative assessment made by the Catalan Government that looked at ES in PAs (Fig. A2.4), and we found that biodiversity was the predominant supporting ES in Catalan PAs (Fig. A2.6). These results suggested that other biodiversity components can be relevant in PAs, such as other groups of species (plant, invertebrates, other terrestrial vertebrates, etc) or other elements of diversity (functional diversity, endangered or rare species) (Brooks et al., 2006; Maiorano et al., 2015). But even when looking at other species, previous studies have revealed contradictory results. Some studies have showed a positive effect of PAs in birds, vertebrates (mammals, amphibians, reptiles) and arthropods, through reducing extinction risks (Butchart et al., 2012) and resulting in higher biodiversity in PAs than not PAs (Coetzee et al., 2014). But others have indicated that these groups of species are not well covered in PAs (Brooks et al., 2004). Specifically, results vary when looking at specific groups of species, such as migratory birds which are not adequately covered by PAs (Runge et al., 2015). In our study, some PAs in Catalonia were designated to protect species that, in the case of birds, were not considered in our study because their habitat was not forest nor shrubland (e.g., *Aquila fasciata, Neophron percnopterus, Falco peregrinus,* etc). However, the effect on diversity of plants is not conclusive, as some studies have showed high tree diversity in good conservation status areas (Maes et al., 2012), whereas others indicate that plants are not well represented in PAs (Coetzee et al., 2014).

When considering conservation variables, we found higher coverage of community-interest habitats, priority habitats and geological-interest sites in PAs than in buffer zones (Table 2.1 and Fig. 2.2), as expected in *hypothesis 1*. As stated before, most studied PAs are located in mountainous areas, thus higher coverage of forest and shrub community-interest habitats in PAs than in buffer zones could be expected. Even so, these results proved that the Habitats Directive is being effective in having more coverage of these habitats inside PAs than in the buffer zones. But we still do not know if this amount of coverage is sufficient, and an exhaustive evaluation of these habitats regarding their quality and conservation status (e.g., degradation and fragmentation) (Wilson et al., 2016; Sallustio et al., 2017), as well as their changes over time (Bunce et al., 2013) is needed to fully understand if community-interest habitats and priority habitats are achieving conservation goals. We also found more coverage of geological-interest sites inside PAs than in buffer zones, particularly because some specific PAs include geological heritage as an important value of conservation. In fact, the importance of geodiversity is stated in the qualitative assessment of ES in Catalan PAs (made by the Catalan Government) as it is, after biodiversity, the second most frequent supporting ES qualified as very important (Fig. A2.6).

Concerning protection status, moderate PAs have more carbon stocks than partial PAs (Table 2.1 and Fig. 2.2) because partial PAs are mainly Natura 2000 sites which were not being designated for carbon sequestration purposes (*hypothesis 1*). In addition, moderate PAs are located in places with higher water availability (i.e., northern areas and mountainous areas, Fig. 2.1), resulting in higher levels of carbon storage (Zhao and Zhou, 2006; Fischer et al., 2014). As partial PAs were mainly Natura 2000 areas, we expected higher levels of threatened bird richness, community-interest habitats and priority-habitats in partial than moderate PAs (*hypothesis 2.2*). But we found no differences between protection statuses, suggesting that moderate PAs are also contributing to these variables. However, previous studies showed that international conservation policies as Natura 2000 succeeded in covering threatened species stated in the Directive (Donald et al., 2007; Kukkala et al., 2016). In fact, the only conservation variable showing differences between protection status was the percentage of geological-interest sites, that was higher in moderate than partial PAs particularly because some specific moderate PAs were designated, among other reasons, for being geologically singular (e.g., 'Serra del Montsant' or 'Zona volcànica de la Garrotxa').

Hotspots of ES, biodiversity and conservation variables inside and outside PAs

Priority areas for conservation could be defined by identifying areas of high values of ES and biodiversity. Contrarily as expected (*hypothesis 1*), we found more hotspots of woody richness, bird richness and threatened bird richness in buffer zones than inside PAs, but more hotspots of priority habitats and geological-interest sites in PAs than buffer zones (Table 2.2 and Fig. 2.3)(*hypothesis 1*). In fact, the study of Roces-Díaz et al. (2018) showed a negative relationship between bird richness and the existence of Natura 2000 sites in the same study area. However, previous studies have showed high tree species diversity in habitats with good conservation status (Maes et al., 2012). In this sense, although we found more hotspots of priority habitats in PAs, we lack information about the quality of the habitats and their conservation status, which might not be adequate inside PAs, thus resulting in lower levels of biodiversity. In addition, these results highlight the necessity of an effective management and the importance of conserving biodiversity not only inside PAs but also in their surrounding buffer zones (Cox and Underwood, 2011). These buffer zones containing hotspots of biodiversity can be seen as an opportunity to delineate a network of green infrastructure that would enhance connectivity between PAs, already stated in the study of Lanzas et al. (2019). Moreover, the coverage of geological-interest sites was higher inside PAs than buffer zones either considering their whole range or their highest values (i.e., hotspots). Geological sites were one of the reasons of many PAs designation (especially in moderate PAs). The integration of geology in PAs is of great value because geological features and processes contribute to biodiversity and it is considered an integral part of nature conservation. UNESCO Geoparks, which are defined as 'single, unified geographical areas where sites and landscapes of international geological significance are managed with a holistic concept of protection, education and sustainable development' have already included the Central Catalonia UNESCO Geopark with two natural parks that have geogical-interest sites, but the Catalan inventory includes many other sites, thus expanding the geological protection of the territory.

Concerning protection status, contrarily as expected (*hypothesis 2*), none of the hotspots variables showed significant differences between moderate and partial PAs (Table 2.2 and Fig. A2.3), meaning that a high degree of protection was not providing high levels of ES and biodiversity. In fact, partial PAs or non-strict levels of protection are found to be important for preserving biodiversity, but for other groups of species such as terrestrial vertebrates (Maiorano et al., 2015). Previous studies also stated that PAs with non-strict protection are important to maintain ES and biodiversity (Gaston et al., 2008*b*; Bastian, 2013).

2.5 Conclusions

The conservation strategy in Catalonia was only effective at maintaining some of the ES and conservation variables considered. Higher values of carbon stocks were found in PAs than in buffer zones, and more coverage of community-interest habitats, priority-habitats and geological-interest sites in PAs than in buffer zones. PAs with higher degree of protection did not provide higher ecosystem services and biodiversity, or vice versa. We unexpectedly found more hotspots of woody richness, bird richness and threatened bird richness in buffer zones than in PAs. Our study provides a first step on a more in-depth evaluation of ES in PAs that can be applied to other regions, but a detailed analysis of each individual PA including more ES and conservation indicators is needed. Specifically, ES relevant for the specific stakeholders in each PA need to be considered. Cultural ES were not included in this study and they are proved to be important in the study area (Roces-Díaz et al., 2018), especially in PAs (Fig. A2.5). Likewise, other ecosystems like freshwater or farmlands can provide essential ES.

Future scenarios of climate change in the Mediterranean area advert significant and increasing risks during next decades (Cramer et al., 2018). Forest productivity is expected to decline due to increased extreme events such as droughts and fire (Lindner et al., 2010). Global biodiversity indicators have showed declines during past years while, at the same time, pressures on biodiversity have increased (Butchart et al., 2010). Under these circumstances, it is a priority to identify which species, functions, and ecological processes are behind the loss of biodiversity to adequately apply the right conservation strategies. Future landscape configuration for conservation should take into account that species distributions might change under climate change scenarios, thus should not be only focused in PAs as an isolated system. In fact, PAs exist within broader landscape mosaics that can influence the movement of species (Wiens, 2009), thus landscape conservation planning should also include buffer zones and non-PAs. Furthermore, scientific evaluation and monitoring of the impact that landscape management interventions have on particular PAs and their buffer zones is needed and should be done recognizing the dynamic nature of landscapes and their species (i.e., not relying on fixed lists of species) (Hermoso et al., 2017). Landscape ecology and sustainability science need to be integrated to develop comprehensive conservation strategies that consider the dynamic interactions between nature and society (Wu, 2008).

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3 Characterizing forest vulnerability and risk to climate change hazards

Abstract

Wildfires, drought, insect-outbreaks and windstorms are altering forests ecosystem services that are essential for human well-being, and these impacts are likely to increase under ongoing climate change. However, a widely accepted and operational framework to evaluate forest vulnerability and risk to these disturbances is still lacking. Here we propose a general framework to assess forest vulnerability and risk based on the widely used concepts of exposure, hazard magnitude, susceptibility and lack of adaptive capacity as defined by the IPCC. We suggest a standardized procedure to define and combine these components, as well as a list of indicators readily applicable to the main climate change-related hazards to forests. This framework and its methodology constitute a basis for a systematic operationalization of forest risk and vulnerability for policy makers as well as for forest and land managers that can be applied to develop future-oriented policies.

3.1 Introduction

Forest ecosystems provide a wide variety of benefits for human well-being, commonly referred to as ecosystem services (Millennium Ecosystem Assessment, 2005). However, forests are increasingly under pressure from climate change, resulting in changes in disturbance and stress regimes, including forest fires, drought, insect-outbreaks and windstorms (Allen et al., 2010; Kurz et al., 2008; Seidl et al., 2014; Saad et al., 2017). Predicting where these natural hazards will occur in the future and to what extent forest ecosystem services will be affected are fundamental research challenges (Thom and Seidl, 2016). To address them, it is necessary to evaluate the vulnerability of forests to these hazards, as well as the risk of loss of forest ecosystem services that are essential for people's livelihoods (Schröter et al., 2005).

A wide range of disciplines such as medicine, environment and engineering use the concepts of vulnerability and risk. Socioeconomic and climate change studies have a well-established conceptualization of vulnerability and risk, where three components are usually identified: 1) exposure to a hazard; 2) susceptibility or sensitivity; and 3) coping or adaptive capacity (in some cases defined separately) (Welle and Birkmann, 2015; Nguyen et al., 2017). According to the IPCC, vulnerability is 'the propensity or predisposition to be adversely affected by a hazard, including sensitivity or susceptibility to harm and lack of capacity to cope and adapt' (IPCC, 2018). Risk is defined as 'the potential for consequences where something of value is at stake and where the outcome is uncertain, which results from the interaction of vulnerability (of the affected system to a given hazard), its exposure over time to the hazard, as well as the (climate-related) hazard and the likelihood of its occurrence' (IPCC, 2018). These terms have been applied to create global and local indices (e.g., the World Risk Index or the Social Vulnerability Index (Nguyen et al., 2017; Welle and Birkmann, 2015)).

In forests, the natural hazards most likely to increase under climate change are wildfires, drought, insect-outbreaks and windstorms (Thom and Seidl, 2016). These hazards co-occur and interact, but most previous studies on forest vulnerability focus on a single hazard. Furthermore, these studies have defined vulnerability in different ways, and in some cases only considering some of its main components. For wildfires, vulnerability has been assessed by simply considering vegetation recovery (Aretano et al., 2015) or only using indicators such as fuel moisture or frequency of fires (McWethy et al., 2013). Vulnerability to forest fires has also been defined as potential losses to fire, including impacts on properties, people and environmental services; but also just using indicators of adaptive capacity (e.g., forest structure and reproductive strategies) (Román et al., 2013) or fuel characteristics (Schelhaas et al., 2010). Other studies such as Duguy et al. (2012) have assessed vulnerability of forests to wildfires accounting for susceptibility and adaptive capacity, but without including risk and exposure. Concerning drought, a forest vulnerability index (FVI) has been developed by Mildrexler et al. (2016), which is based on forest stress defined by water and energy exchange processes caused by drought and high temperatures. For insect-outbreaks, to our knowledge only resistance and resilience have been previously quantified based on composition and structure of boreal forest inventory plots (Sánchez-Pinillos et al., 2016). Windstorm vulnerability has been based on tree species structural characteristics and composition in relation to its resistance to wind, without including their adaptive capacity (Schelhaas et al., 2010; Anyomi et al., 2017). Yet all of these studies are hazard-specific and the characterization of vulnerability varies depending on the study and the hazard considered, which results in limited applicability of these concepts to other situations and hinders comparability between different study areas. In addition, there is an increasing interest in the study of forest resilience, which is a concept closely related to (the inverse of) vulnerability, typically quantified as the recovery time after a disturbance (Nikinmaa et al., 2020). Again, however, a general framework linking vulnerability and resilience approaches is missing.

Some studies have defined comprehensive indices for forest vulnerability that might be applied to different hazards, but without explicitly considering all the different components of vulnerability and risk. For instance, a Persistence Index (PI) based on ecosystem persistence traits has been defined to quantify the capacity of forest communities to maintain their functions and services after disturbances (Sánchez-Pinillos et al., 2016). Vulnerability of forests to extreme climate events has also been assessed using probabilistic risk analysis based on time series of climate and ecosystem characteristics (e.g., carbon cycle), where vulnerability is defined as the difference in the system performance between hazardous and non-hazardous environmental conditions (Van Oijen et al., 2013). Finally, vulnerability of forests to climate change has been assessed by considering only particular indicators such as mortality, regeneration or productivity (Halofsky et al., 2018). A recent study analyzed vulnerability of southwestern forests in the USA to climate change by considering exposure, sensitivity and adaptive capacity scored subjectively in 10 regional forest types (Thorne et al., 2018). Although this study has considered all of the basic components of vulnerability, it has only been applied to the forest types in a single study region.

To our knowledge, a general framework of forest vulnerability and risk to the main climate change-related hazards that uses the components defined by the IPCC and is applicable to forest ecosystems in different biomes is still lacking. Here we present a framework that accounts for the components of vulnerability adjusted to the particular nature of forest ecosystems, as well as the ecosystem services at risk from four key climate change-related hazards (wildfires, drought, pests and windstorms). Although we acknowledge that other hazards are likely to be important (or even dominant) in specific ecological contexts (e.g., soil flooding and permafrost melting in boreal forests), these four are likely to be the most widespread forest hazards in a climate change context. We also propose an operational methodology to combine the different components and the indicators that can be used to quantify them.

3.2 General framework of forest vulnerability and risk

We use the main concepts of vulnerability and risk as defined in the latest IPCC report (IPCC, 2018), and modify and adapt them to the case of forests (Table A3.1). Vulnerability in forests has two components: susceptibility, related to the immediate effects of the hazard, and adaptive capacity, which measures the mid-term response after hazard occurrence (Fig. 3.1). We focus our analysis on mid-term responses (up to a few decades after hazard occurrence), and hence do not consider long-term processes such as evolutionary adaptation or successional dynamics. We have structured these components in a temporal framework considering whether they refer to the situation of the forest 1) before the hazard; 2) during the hazard or 3) after the hazard, as described in the following paragraphs. Finally, we have characterized each component with factors that are 1) intrinsic, referred to internal characteristics of the forest (e.g., species characteristics); or 2) extrinsic, referred to external factors typically operating at broader spatial scales (e.g., landscape scale)(Fig. 3.1). Note that the framework is designed to be applied at regional scale using local-level information (e.g., data from forest inventory plots, relatively fine-scale pixels from remote sensing surveys), and hence intrinsic refers to stand-level factors and extrinsic to any factor that operates beyond the target stand (e.g., landscape level).

Before the hazard occurs, all forests that can be affected by the hazard are exposed.

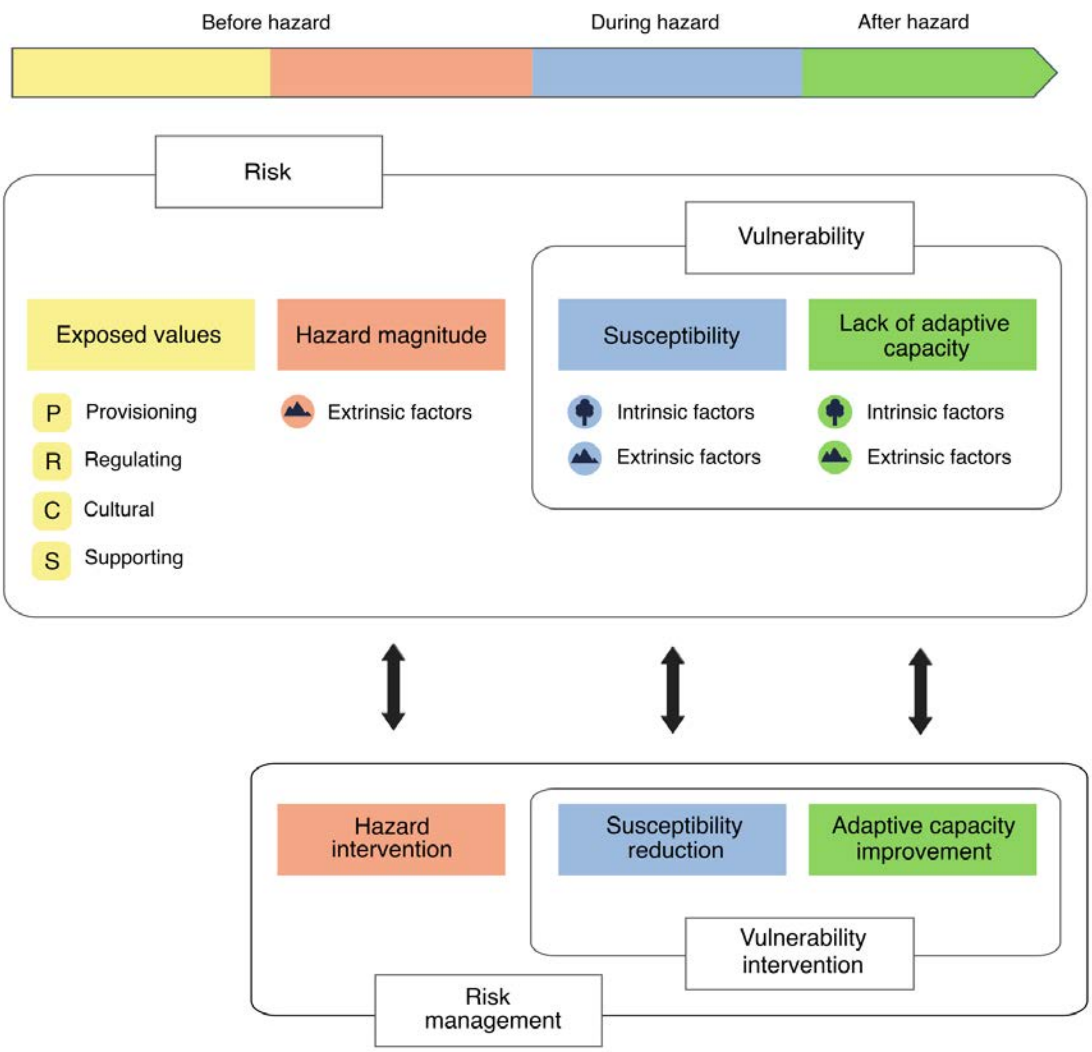

FIGURE 3.1: General framework of forest vulnerability and risk to climate change hazards and its temporal dimension (before, during and after hazard).

However, these forests differ in their 'value', which we quantify here in terms of the ecosystem services they provide and that can be lost if the hazard occurs. We have defined these services as the exposed values, which include provisioning services (e.g, wood), regulating services (e.g, water regulation, climate regulation), cultural services (e.g., recreation, education) and supporting services (or biodiversity) (Millennium Ecosystem Assessment, 2005) (Fig. 3.1). This concept is in line with the IPCC definition of exposure as 'the presence of people, livelihoods, species or ecosystems, environmental functions, services [...] in places and settings that could be adversely affected by a hazard' (IPCC, 2018) (Table A3.1). These ecosystem services could only be lost if the hazard occurs. In a given location (e.g., a forest plot), the magnitude of the hazard and its probability distribution can be quantified using integrative hazard indices that incorporate the most relevant attributes of the hazard (e.g., the Fire Weather Index for wildfires (Van Wagner, 1987)). If needed, hazard indices can be modified using other factors (e.g., forest continuity or human visitation). These factors determine the probability distribution of hazards of different magnitude at each site (or plot), which is normally assessed before the hazard occurs (Fig. 3.1 and Fig. A3.1).

When the hazard occurs, some characteristics of the forest modulate the immediate effects of the hazard. Many of these characteristics are intrinsic, as they correspond to the properties of the tree species (their traits) or to structural attributes, but can also be extrinsic in the case of some hazards (e.g., extinction capacity for wildfires). Both intrinsic and extrinsic factors that have an effect during the hazard contribute to forest susceptibility, defined as the predisposition to be affected by the hazard (Fig. 3.1 and Table A3.1).

After the hazard occurs, forests may recover, which is determined by their adaptive capacity, defined as 'the ability of systems, institutions, humans and other organisms to adjust to potential damage, to take advantage of opportunities, or to respond to consequences' (IPCC, 2018) (Table A3.1). Adaptive capacity (or its lack) depends on intrinsic and extrinsic characteristics, such as regeneration capacity or post-disturbance forest management (Fig. 3.1). Note that other factors may be crucial in determining the adaptive capacity at longer timescales (e.g., evolutionary adaptation, long-term successional dynamics, land-use dynamics at broader spatial scales). However, these longer timescales are not considered here due to the high uncertainties associated to them and the fact that they are unlikely to affect current management decisions.

The risk of losing ecosystem services is thus the combination of exposed values, hazard magnitude and vulnerability (Fig. 3.1 and Fig. A3.1), with the highest risk occurring in forests that provide more ecosystem services, are subjected to the highest hazard magnitude and are the most vulnerable (more susceptible and less able to adapt). Importantly, the concept of vulnerability we adopt here is akin to the concept of resilience as usually used in forest ecology, defined as the product of resistance to disturbance and recovery (e.g., Lloret et al. (2011)).

Risk management and vulnerability intervention

The proposed framework allows for an easy integration of the concepts of risk management and vulnerability intervention (Fig. 3.1). Forest can be managed to be less vulnerable by reducing their susceptibility or increasing their adaptive capacity. In the case of susceptibility, internal factors such as structural and functional characteristics can be improved to promote less susceptible forests. For instance, reducing vertical and horizontal vegetation continuity will make forests less susceptible to fires. Acting on extrinsic factors such as improving the warning system could also reduce susceptibility to wildfires. In the case of adaptive capacity, promoting mixed-species stands that support species with varied recovery mechanisms (i.e., resprouting and seeding) will promote the ability of ecosystems to self-organize and increase their adaptive capacity (Messier et al., 2015). Assisted migration and reforestation using functionally complementary and redundant tree species to those present in the area could also increase adaptive capacity (Janowiak et al., 2014; Messier et al., 2015). Regarding risk management, examples of wilfire risk reduction acting on hazard magnitude include controlling human visitation (e.g., closing forest access roads) and increasing people awareness. Long-term strategies may include land planning (e.g., creating safe areas and defensible spaces in the wildland-urban interface) and zoning land use and development based on fire risk (Fischer et al., 2016). In this way, risk and vulnerability management will determine future risk and vulnerability (Fig. 3.1).

3.3 Indicators for the main climate change hazards

The different components of vulnerability and risk to the four main hazards considered here are defined by intrinsic and extrinsic factors that can be quantified using explicit indicators. Specific examples are shown in Figure 3.2, some of which are shared among the four hazards, whereas others are hazard-specific.

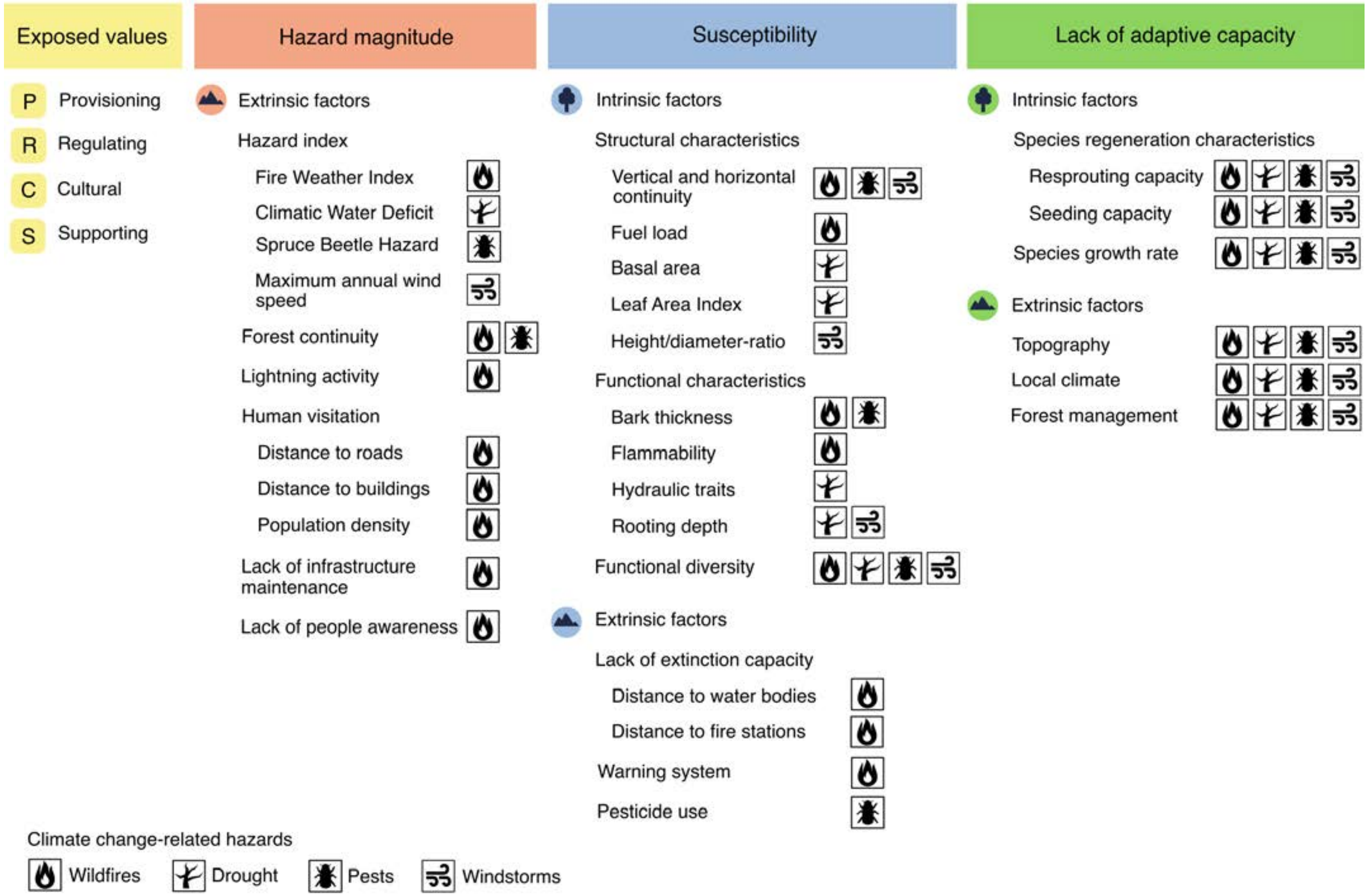

FIGURE 3.2: Indicators for each of the components of risk and vulnerability for the four main climate change hazards considered in this study: wildfires, droughts, pests and windstorms.

Indicators for hazard magnitude

Extrinsic factors

An integrative, hazard-specific index may be used. Suitable indices would be the Fire Weather Index (FWI) for wildfires (Van Wagner, 1987), the Climatic Water Deficit (CWD) for droughts (Anderegg et al., 2015), or maximum annual wind speed for windstorms (Schelhaas et al., 2010). Hazard indices for insect outbreaks are likely to depend on the specific pest under consideration, but several indices are already available for specific pests (e.g, the Spruce Bettle Hazard, (Forests, Lands, Natural Resource Operations and Rural Development, 2014)). In the case of wildfires and pests, forest continuity at the landscape scale can be added as an extrinsic factor modifying hazard magnitude (Fig. 3.2). Anthropogenic factors can also be important determinants of hazard magnitude. In the case of wildfires, for instance, human visitation (e.g., the amount of people visiting a forest), infrastructure maintenance (e.g., maintenance of electrical towers) and the awareness of people can all affect the probability of wildfires (Syphard et al., 2007) (Fig. 3.2).

Indicators for susceptibility

Intrinsic factors

Structural and functional characteristics of the forest can influence its predisposition to be affected by a hazard. For instance, the vertical structure and density of vegetation and the amount of fuel load will affect forest susceptibility to wildfires (Alvarez et al., 2012*b*), whereas leaf area index affects drought susceptibility (Jump et al., 2017) and the height/diameter ratio is key in the case of windstorms (Scott and Mitchell, 2005). Functional characteristics are also important to define susceptibility. For example, rooting depth would be a key determinant of susceptibility to drought and windstorms, and bark thickness influences the resistance of trees to wildfires and pests (Choat et al., 2018; Pausas, 2015). We also include functional diversity and structural diversity as they can modulate the effects of the hazard (e.g., Anderegg et al. (2018) for drought stress).

Extrinsic factors

Some external, human-related factors can influence the susceptibility of forests to wildfires or pests. In the case of wildfires, firefighter capacity to extinguish the fire and their warning system will determine forest predisposition to be affected, whereas the use of pesticides will influence pest attacks.

Indicators for lack of adaptive capacity

Intrinsic factors

The regeneration characteristics of species (e.g., resprouting capacity and presence of a seed bank) will determine forest recovery (Rodrigo et al., 2004). Growth rate (e.g., mean annual increment) and lifespan will also vary by species and will determine the speed of recovery.

Extrinsic factors

They include topography (e.g., northern aspects will typically recover better than southern aspects in xeric ecosystems in the northern hemisphere), local climate (e.g., wetter conditions typically accelerate recovery in relatively dry regions) and forest management (e.g., selective thinning, or planting, to promote regeneration of certain species after a wildfire) (Vallejo and Alloza, 2015).

3.4 Integrating vulnerability and risk components

Here we propose a methodology to combine the components of vulnerability and risk that considers their strong interdependencies in forests. In each location, there is a hazard probability distribution that is hazard-specific. The immediate effect of the hazard will be strongly dependent on susceptibility factors, and we define susceptibility as the slope of the log-log relationship between hazard magnitude and the immediate loss of exposed values (Fig. 3.3b). Although other functions can be applied, we used a power function as it ensures that non-linear impacts, which are common in forest ecosystems subjected to natural hazards, can be easily accommodated. Note also that this formulation accounts also for the possibility of complete forest loss for high hazard magnitudes and/or susceptibilities. Both intrinsic and extrinsic factors that contribute to susceptibility can be modifiers of the slope of the relationship between the magnitude of the hazard and the immediate loss of ecosystem services, and hence different forests will typically have different slopes. For example, s_1 in Figure 3.3b identifies a less susceptible

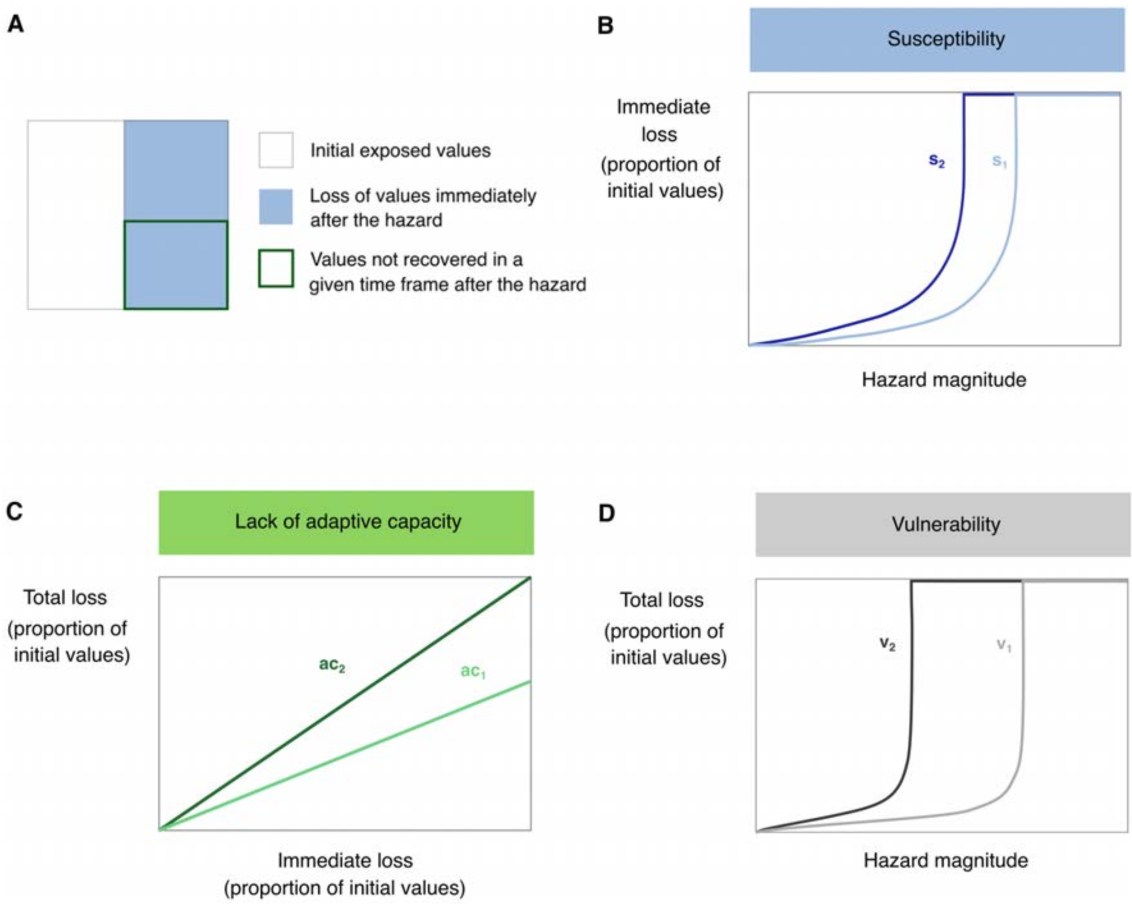

FIGURE 3.3: Graphical representation of the components of vulnerability: (A) example of values lost and recovered in an area affected by a hazard; (B) relationship between hazard magnitude and the immediate loss of values defining susceptibility (s); (C) relationship between the immediate loss and the total loss of values that defines the lack of adaptive capacity (r); and (D) relationship between hazard intensity and the total loss of values to define vulnerability (v). Two different forests with contrasting vulnerabilities (1: low; 2: high) are illustrated in panels B - D.

forest, because it experiences a lower loss for a given hazard magnitude than the forest characterized by s_2.

After a predefined period (in our case, 50 years), a proportion of the values lost may still not be recovered, which is related to the lack of adaptive capacity. This lack of adaptive capacity is formally defined as the proportion of the values initially lost that are not recovered after the predefined time interval or, equivalently, as the slope of the relationship between the immediate loss of values and the loss remaining after this period, assumed to be approximately linear (Fig. 3.3c). Likewise, the indicators defining the lack of adaptive capacity are modifiers of this slope, which varies between 0 (all values are recovered) and 1 (no recovery). Thus, the forest identified by ac_2 in Figure 3.3c has a lower adaptive capacity than the one identified by ac_1. The combination of the relationships depicted in panels 3b and 3c determines the relationship between the magnitude of the hazard and the proportion of total values lost, which defines vulnerability (Fig. 3.3d). In the example of the Figure, the v_2 forest is much more vulnerable than v_1 (Fig. 3.3d).

Combining all the previous relationships, Risk of loss of ecosystem services can be defined as follows:

$$Risk = E \cdot HM^S \cdot LAC \tag{3.1}$$

where E are Exposed values, here measured as the ecosystem services provided by the forest, HM is Hazard Magnitude (weighted by its probability distribution), S is Susceptibility, and LAC is Lack of Adaptive Capacity within a given time frame. Note that both susceptibility and lack of adaptive capacity are determined by intrinsic and extrinsic factors. Because of the non-linear nature of the relationship between hazard magnitude and immediate loss of values (Fig. 3.3b), a convenient way to estimate overall forest risk (a distribution of risk) to a given hazard would be to use the Monte Carlo method. This method relies on 1) conducting repeated random sampling on the probability distribution of the hazard magnitude for the target forest; 2) estimating risk using the equation above for each of the simulated instances to 3) obtain a probability distribution of the risk, and finally 4) aggregating the results as needed.

3.5 Methodological steps to assess forest vulnerability and risk

We detail the different steps of the methodology for the application of the conceptual framework proposed here. Many of these steps are adapted from Nardo et al. (2008). An example of each step for adaptive capacity and how risk is evaluated is provided in Figure A3.1.

Step 1. Selecting the variables meaningful to the framework

Once the framework has been defined, specific variables depending on the climate change hazard under consideration need to be defined and included in the analyses. The procedure to define them should involve experts on the field as well as stakeholders.

Step 2. Assessing data availability

After listing the variables, available indicators need to be selected and the data required to measure them retrieved from available sources. Criteria to select indicators include their measurability, coverage and relevance (Nardo et al., 2008). The use of proxy variables and the imputation of missing data can be alternatives to provide complete datasets (Fig. A3.1).

Step 3. Analyzing the structure of the dataset

Multivariate analyses (e.g., Principal Component Analysis (PCA)) are needed to understand the structure of the dataset and to reduce the number of variables (if necessary). In addition, the statistical structure of the dataset can be afterwards used in step 5 to aggregate indicators (Nardo et al., 2008) (Fig. A3.1).

Step 4. Normalizing the indicators to make them comparable

Different normalization procedures can be used (e.g., ranking, min-max, z-scores) to make indicators comparable (e.g., standardizing to a range from 0 to 1) (Nardo et al., 2008) (Fig. A3.1).

Step 5. Weighting and aggregating the indicators

The indicators are weighted to define each of the components (exposed values, hazard magnitude, susceptibility and lack of adaptive capacity, as needed). The most common approaches to assign weights are: 1) equal weights, when the relative importance of the indicators is unknown; 2) statistical weights, based on the statistical importance of the indicators in an ordination analysis (i.e., step 3); or (3) expert weights, based on expert criteria (Nardo et al., 2008). Then, the weighted indicators are aggregated to define each component. In some cases, the value of a given component may need to be adjusted so that the resulting values are realistic. This is the case of susceptibility, which may need calibration using previous information on the hazard magnitude corresponding to different levels of immediate loss.

Step 6. Aggregating the components and associate them to values at risk

Risk is then calculated using Eq. (3.1) (see also Fig. A3.1). A distribution of risk can be calculated by using the Monte Carlo method (repeated random sampling of the distribution of hazard magnitudes). The measure of risk can be absolute (total amount of values at risk) or relative (percentage of values at risk).

Step 7. Conducting sensitivity analysis

It is recommended to conduct a sensitivity analysis to assess the robustness of the assessment and to estimate its uncertainty (e.g., including or excluding specific indicators, changing weights or changing the aggregation method).

3.6 Applications and future research directions

To our knowledge, the proposed framework is the first assessing the risk and vulnerability of forests to climate change hazards in a comprehensive manner, and proposing an operational means to combine them in simple yet meaningful metrics. Previous studies have been focused on specific hazards or single components of vulnerability and risk (e.g., Aretano et al. (2015); Halofsky et al. (2018)). We also provide a pre-defined set of general indicators that can be used to quantify the different components of forest vulnerability and risk to wildfires, drought, pests and windstorms. This conceptualization can be a basis for systematic risk and vulnerability assessments for policy makers, as well as for resource and land use managers, contributing to an efficient forest hazard management through the identification of the most vulnerable areas and the development of management actions to reduce hazard probability and susceptibility and increase adaptive capacity.

There is no question, however, that important challenges remain. Although our framework considers the four main climate change-related hazards in forests (i.e., wildfires, drought, pests and windstorms), it is a first attempt based in a single-hazard multi-layer approach (i.e., producing one layer - or map - for each hazard separately, without explicitly considering their interactions). Future studies should identify and characterize the interactions between hazards in a multi-hazard approach (e.g., drought can increase the susceptibility of forests to be affected by fire or pests, Anderegg et al. (2015)). Similarly, we do not explicitly provide a means of combining the different indicators of exposure (different ecosystem services, in our case). There is an extensive literature on the means to obtain synthetic measures of ecosystem service values both in economic and biophysical terms (e.g, Häyhä and Franzese (2014)), and reviewing those is beyond the scope of this chapter. Including the temporal dynamics of the hazards and the implications for the associated risk and vulnerability should be also a next step. Our framework takes into account the loses due to a single hazard in a given time frame, without considering that the previous history of hazard occurrence can affect both the probability of hazard and the susceptibility and adaptive capacity of the affected forest, due to legacy effects.

Despite these challenges, the general framework proposed here can be used to improve the performance of vulnerability and risk assessments and contribute to decision making. In addition, this framework can be applied in the context of scenario planning, contributing to develop future-oriented policies by anticipating conditions associated with particularly high risks.

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4 Assessing the risk of losing forest ecosystem services due to wildfires under average and extreme hazard conditions

Abstract

Forest provide multiple ecosystem services (ES) fundamental to society, but these services are highly sensitive to climate related disturbances such as wildfires. As climate change is transforming extreme climate events into more frequent occurrences, the evaluation of forest vulnerability to wildfires and their risk of losing ecosystem services is essential to anticipate and adapt to future conditions. Here, we analyze the spatial patterns of forest vulnerability and risk of losing key ES (i.e., carbon sink, bird richness, hydrological control and erosion control) from wildfires in Catalonia (NE Spain), accounting for the exposed values, hazard magnitude, susceptibility and lack of adaptive capacity. We also determine the effect of historical climate and forest functional types on the risk of losing ES under average and extreme hazard conditions, as well as on the increase in risk between average and extreme hazard conditions. Our results show that hazard magnitude is the most important component defining risk under average conditions. Under extreme conditions, hazard magnitude became less important and exposed values (in particular carbon sink capacity and erosion control) emerged as the most important components determining ES at risk. Climate was the main driving factor of ES at risk under average conditions, but forest functional type - in particular non-Mediterranean conifer forests with low adaptive capacity - gained importance under extreme conditions. The increase in risk between average and extreme conditions was driven by precipitation, with the highest increases in risk in relatively wet forests with currently low average risk. These results have direct implications on the future risk of losing ES due to wildfires in Mediterranean forests but also in other regions, and could contribute to future-oriented policies by anticipating conditions associated with particularly high risks that can be used to guide efficient forest management.

4.1 Introduction

Forest provide multiple functions and ecosystem services that are fundamental to society (Millennium Ecosystem Assessment, 2005). Forests are essential for regulating services since they can mitigate greenhouse gas emissions and regulate water flow (Canadell and Raupach, 2008; Miura et al., 2015), as well as for the provision of services such as timber or food (Millennium Ecosystem Assessment, 2005). Besides, forests support plant and animal habitats that hold terrestrial biodiversity (Pan et al., 2013) and provide cultural services such as recreation and aesthetic values (García-Nieto et al., 2013). However, forests are increasingly affected by climate change related disturbances (e.g., wildfires or insect-outbreaks) (Seidl et al., 2017), which results in changes in the services they provide. Identifying where and to what extent different forest types and ecosystem services will be put at risk by these hazards is still a challenge, yet it can be critical to guide effective management and policy interventions.

Wildfires are one of the most common disturbances in forests worldwide and especially in Mediterranean climate regions, and they can have huge impacts on ES (Moritz et al., 2014). These impacts, however, can be very variable depending on the environmental context and the service considered. Previous studies reported negative effects of wildfires in ES, mainly water provision, erosion control and climate regulation (Roces-Díaz et al., (submitted)). In the case of water, decreases on infiltration and increases in runoff have been reported (Vukomanovic and Steelman, 2019), particularly in water-limited environments (Vieira et al., 2016), as well as effects on water quality for human consumption (Vukomanovic and Steelman, 2019). Erosion control diminished after wildfires, especially during the first post-fire rainstorms (Shakesby, 2011). Regarding climate regulation, previous studies showed a reduction on the carbon sink capacity of forests after wildfires (Seidl et al., 2014). On the contrary, fires can be also beneficial for ES since they can generate open habitats that offer a variety of services for humans (e.g., food, pollination) (Pausas and Keeley, 2019). Given these controversial results, identifying the forest types that can either loose or gain ES due to wildfires and the main causes of these changes constitutes a research priority.

Vulnerability of forests to wildfires and the corresponding risk of losing ES are not easily quantifiable. Here we follow the IPCC to define vulnerability as ‘the propensity or predisposition to be adversely affected by the hazard (e.g., wildfires), including sensitivity or susceptibility to harm and lack of capacity to cope and adapt’ and risk as ‘the potential for consequences where something of value is at stake and where the outcome is uncertain' (IPCC, 2018). Risk therefore results from the interaction of the exposure, the climate-related hazard and vulnerability. Most of the previous studies assessing forest vulnerability and risk to wildfires have not been based on all the IPCC components or only used specific indicators or variables (Román et al., 2013; Ghorbanzadeh et al., 2019; Buotte et al., 2019; Fremout et al., 2020; Duguy et al., 2012; Oliveira et al., 2018). From this research, it emerges that forests subjected to high hazard magnitude (i.e., high wildfire danger or high Fire Weather Index) usually show larger impacts on forest ES such as carbon storage, biodiversity, water quality and soil erosion (Shakesby, 2011; Thom and Seidl, 2016; Harper et al., 2018). Adaptive capacity has been also considered a key component of wildfire forest vulnerability through recovery and the presence of fire-adaptive traits (e.g., post-fire seed bank or resprouting capacity), seed dispersal and seed-longevity (Román et al., 2013; Thorne et al., 2018).

More recently, a general framework including all the IPCC components and being readily applicable to the main climate change-related hazards to forests has been proposed (Lecina-Diaz et al., in press). This framework includes the main components of forest vulnerability and risk (exposed values, hazard magnitude, susceptibility and lack of adaptive capacity), which are defined by intrinsic and extrinsic factors, as well as by explicit indicators depending on the hazard considered. Although a methodology to combine the indicators and the components of vulnerability and risk has been proposed, it has not yet been applied (Lecina-Diaz et al., in press). Hence, the influence of these components on the risk of losing forest ES for different hazard types is still unclear.

Current climate has a great influence on wildfires, especially when climate conditions are extreme (Crockett and Westerling, 2018; Holden et al., 2018). Nevertheless, impacts on wildfires and the corresponding ES at risk also depend on the forest functional type. In fire-prone areas such as the Mediterranean Basin, forest types could differ in the amount of ES exposed, but also in the fire danger or hazard magnitude (e.g., Mediterranean-conifers such as *Pinus halepensis* have greater wildfire danger) (Mitsopoulos and Dimitrakopoulos, 2007). Adaptive capacity (e.g., recovery rate) also varies depending on the forest functional type. While some forest functional types have traits that made them able to survive or re-establish after fire (e.g., seeding or resprouting capacity), others have limited post-fire regeneration capacity (e.g., non-Mediterranean conifers such as several *Pinus* species) (Rodrigo et al., 2004). Thus, current climate and forest functional type are influencing the individual components of vulnerability and risk, yet their consequent effects in the resulting ES at risk are not completely understood. Furthermore, climate change is increasing the frequency of extreme climate events (Seidl et al., 2017). In the case of wildfires, future increases in extreme climate events are expected to increase burn probability, fire size and the length of the fire season (Flannigan et al., 2005; Lozano et al., 2016; Ruffault et al., 2018). In particular, climate change projections in the Mediterranean Basin suggest increases of more than 50% in days leading to extreme wildfire events due to increasing temperature and decreasing humidity, especially during the summer fire season (Bowman et al., 2017). The shift from the current situation to one in which current extremes will become the new normal will certainly have consequences on forest ES. Increases in the geographic extent, duration, intensity and severity of wildfires may change the distribution of forest ES at risk, thus 'new' high risk areas may emerge (Alvarez et al., 2012*b*). In fact, previous studies suggest that wildfires could increasingly affect northern latitudes and higher elevations in mountain ranges in the Mediterranean (Vilà-Cabrera et al., 2012; Duguy et al., 2013).

The general objective of this chapter is to assess the spatial patterns and drivers of the vulnerability of forests to wildfires and the corresponding risk of losing ES, focusing on a climatically diverse region (Catalonia, NE Spain) in the temperate-Mediterranean ecotone. We take advantage of the general framework recently defined by Lecina-Diaz et al. (in press), which we further develop and apply to the specific case of wildfires. Specifically, we address three questions: 1) to what component (exposed values, hazard magnitude, susceptibility, lack of adaptive capacity) is the risk of losing ES most sensitive? Which of these components drives the spatial variation in risk?; 2) is the risk of losing forest ES due to wildfires under average and extreme hazard conditions primarily determined by climate or by forest functional type?; and 3) which climatic factors and forest functional types are associated to higher increases in risk between average and extreme conditions?

Given that fire danger is a relevant driver of burned area, and that climate but also post-fire recovery capacity - which varies with forest functional type - are shown to be influential factors on wildfires, we hypothesize that:

- *Hypothesis 1*: hazard magnitude has the greatest influence on ES at risk, followed by lack of adaptive capacity and exposed values.
- *Hypothesis 2*: ES at risk are largely determined by climate, but the importance of forest functional type increases under extreme hazard conditions.

- *Hypothesis 3*: increases in risk between average and extreme conditions are driven by climate, with the greatest increases in relatively wet areas where wildfires will become common in the future.

4.2 Methods

Study area

The study area is Catalonia (NE Spain), a region located between 40º50' and 42º90' latitude North and 0º20' and 3º32' longitude East. It has a heterogeneous geomorphology and high climatic diversity, encompassing mountainous areas such as the Pyrenees (up to 3,143 m.a.s.l), inland agricultural plains and coastal zones along the Mediterranean Sea. The climate is Mediterranean, with mean annual temperature ranging from 1 to 17.1 ºC and mean annual precipitation ranging from 350 to 1460 mm (Ninyerola et al., 2000). Northern areas are the most humid and coldest of the region (i.e., mean annual precipitation = 1300 mm; mean annual temperature = 5 ºC), whereas southern areas are the hottest and driest (mean annual precipitation = 300 mm with mean annual temperature = 15 ºC), especially during the summer (precipitation < 40 mm and maximum temperature > 30) (Ninyerola et al., 2000). Around 40% of the area is covered by forests (CREAF, 2005), dominated by tree species mainly from the Pinaceae and Fagaceae families. The main species in the Pinaceae family are *Pinus halepensis, Pinus sylvestris, Pinus nigra* and *Pinus uncinata,* whereas the most frequent species in the Fagaceae family are *Quercus ilex, Quercus suber, Quercus humilis/cerrioides* and *Fagus sylvatica.*

Definition of vulnerability and risk

We used the general framework developed by (Lecina-Diaz et al., in press). We applied this framework to wildfires, defining risk as:

$$Risk = E \cdot HM^S \cdot LAC \tag{4.1}$$

where E refers to Exposed values (before the hazard), HM is the Hazard Magnitude, S is susceptibility and LAC is lack of adaptive capacity. We define E as the presence of ES that could be adversely affected by the wildfire, in this case, carbon sink, bird richness, hydrological control and erosion control (Table A4.1). HM is characterized using the probability distribution of the Fire Weather Index (FWI) (Van Wagner, 1974), modified by some additional variables (see below). S is the predisposition to be affected by a wildfire depending on characteristics that modulate the immediate effects of the hazard. Finally, LAC corresponds to the lack of capacity of a forest to recover after a wildfire in the mid-term. Each component is defined by different indicators that are 1) intrinsic, referred to internal characteristics of the forest (e.g., species characteristics) or 2) extrinsic, referred to external factors typically operating at broad spatial scales (e.g., landscape scale) (Lecina-Diaz et al., in press) (Fig. 4.1 and Table A4.1).

Data sources and indicators used

We used different sources of data to define the indicators of the different components of vulnerability and risk, i.e., exposed values, hazard magnitude, susceptibility and lack of adaptive capacity (Fig. 4.1).

Our reference scale is the forest stand (plot) based on the 3rd Spanish National Forest Inventory (IFN3), which was conducted in Catalonia between 2000 and 2001 (Ministerio de Medio Ambiente, 2007*b*). This inventory consisted of a systematic sampling of permanent plots with a sampling density of one plot per km^2 of forest area, where woody species were identified and measured within variable circular size (5 m radius for trees with DBH $\geqslant$ 7.5 cm, 10 m radius for trees with DBH $\geqslant$ 12.5 cm, 15 m radius for trees with DBH $\geqslant$ 22.5 cm, and 25 m radius for trees with DBH $\geqslant$ 42.5 cm). As IFN3 sets the reference scale of the study, all indicators were computed at the plot scale (Fig. 4.1 and Table A4.1).

A complete list and additional details of the indicators (definition, scale, references, etc) is given in Figure A4.1. Further details of each indicator are provided in Appendix 4 - Data sources and indicators used.

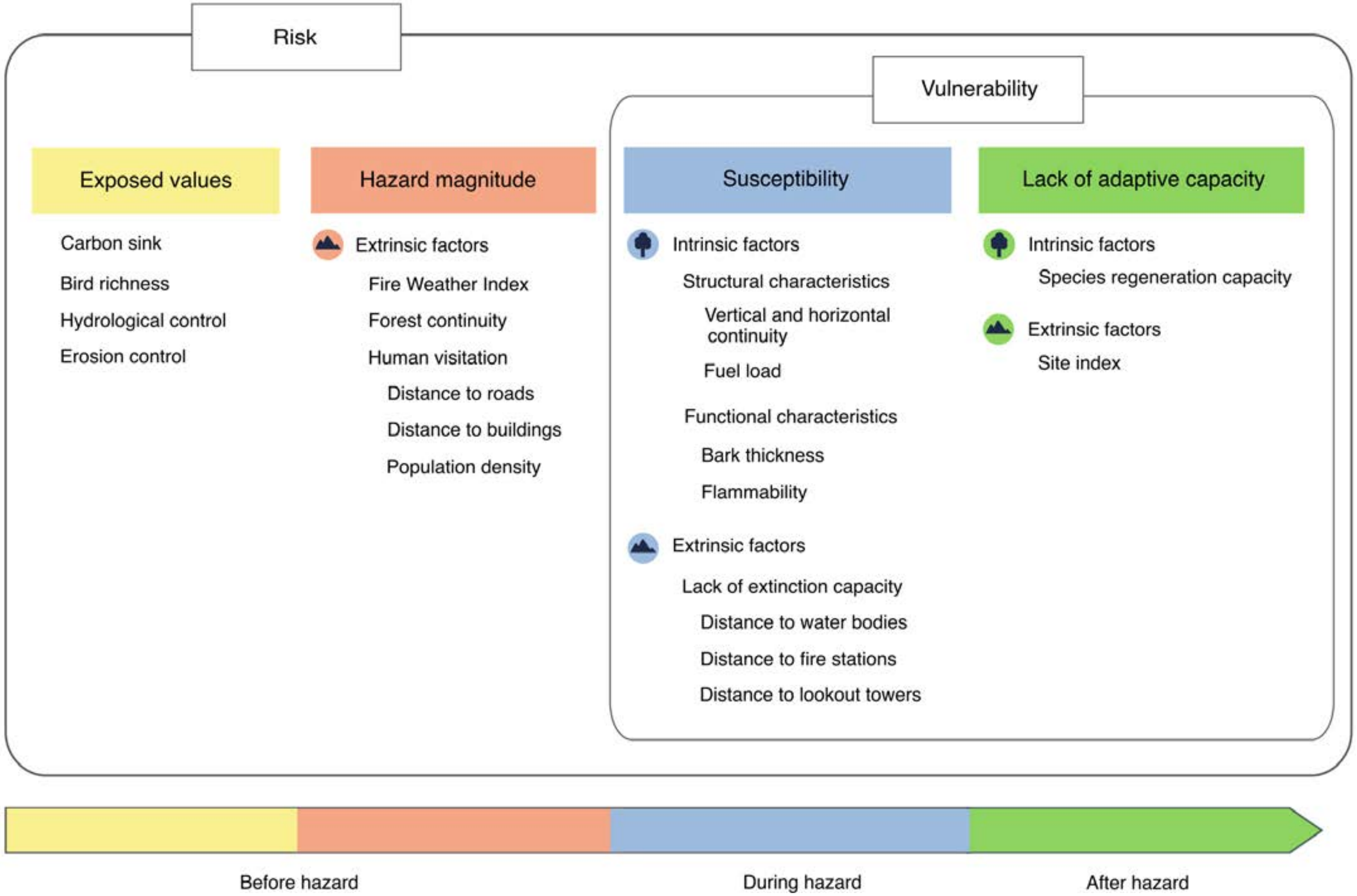

FIGURE 4.1: Components and indicators used in the general framework of forest vulnerability and risk to wildfires and their corresponding temporal dimension (before, during and after hazard).

Exposed values (*E*)

The ES that we have considered are carbon sink, bird richness, hydrological control and erosion control. Carbon sink is the difference of carbon stocks between the 2nd and 3rd National Forest Inventory (IFN), in tons/ha·year. Bird richness has been used as a proxy of biodiversity, and has been assessed using the 2nd Catalan Breeding Bird Atlas (Estrada et al., 2004), counting the total number of species associated to forest habitats (i.e., forest specialist and forest generalist species) present in each 1 x 1 km pixels centered around each IFN3 plot. Hydrological control is the capacity of forests to control water flooding, assessed as (1 - water exported/precipitation), using the average values of the period 1990-2010 predicted by the model of De Cáceres et al. (2015) for each IFN plot. Erosion control has been defined as the percentage erosion avoided by the presence of forests, i.e., the difference of the Revisited Universal Soil Loss Equation (RUSLE) considering soil without vegetation (i.e., $Fcover$ (in $c-factor$) = 0) and the actual forest cover in the plot (Appendix 4 - Data sources and indicators used).

Hazard Magnitude (*HM*)

We have used the distribution of daily Fire Weather Index (FWI) values of June-September obtained from the Joint Research Centre ERA - interim database (Joint Research Centre, 2017). The FWI combines temperature, wind speed, relative humidity, and precipitation on a daily basis (including the cumulative effect of the weather in the previous days) to estimate the fire danger (Van Wagner, 1987). We have used the Monte Carlo method to obtain repeated random sample of the distribution of daily FWI values. We have incorporated forest continuity at the landscape scale and human visitation (defined by a combination of population, distance to buildings and distance to roads) as modifiers of the hazard magnitude (Appendix 4 - Data sources and indicators used).

Susceptibility (*S*)

Susceptibility is defined by intrinsic and extrinsic factors that modulate the immediate effects of the wildfire. Intrinsic factors include structural characteristics (forest vertical and horizontal continuity and fuel load - total shrub biomass and fine biomass from trees -) and functional characteristics (bark thickness and flammability). The extrinsic factors correspond to the firefighter's extinction capacity (distances to water bodies, to fire stations and to fire lookout towers) (Appendix 4 - Data sources and indicators used).

Lack of adaptive capacity (*LAC*)

Lack of Adaptive Capacity is calculated as 1 - adaptive capacity. Adaptive capacity is defined by intrinsic and extrinsic factors. Intrinsic factors are mainly species regeneration characteristics (i.e., resprouting and seeding capacity). Extrinsic factors are the external characteristics that promote species recovery, which are defined by the site index estimated from linear models with tree basal area increment (in cm^2/year) as response variable and radiation, aridity, stoniness and top index as explanatory variables (Appendix 4 - Data sources and indicators used).

Weighting and aggregating the indicators

The indicators listed in Figure 4.1 were standardized (i.e., dividing by their maximum value) so that all indicators had a range from 0 to 1 and were therefore comparable. Afterwards, the standardized indicators need to be combined, yet uncertainties arise when weighting and aggregating the individual indicators (Gan et al., 2017). We applied three of the most widely used weighting methods: 1) equal weights, assigning the same weight for all indicators in a component; 2) statistical weights, using the statistical importance of the indicators based on their variance explained in a Principal Component Analysis; and (3) expert weights, corresponding to the average value of the weights assigned independently by each of the co-authors of the article of this chapter (Table A4.4). To decide which weighting method to use, we conducted Pearson's correlation tests between the estimates of *HM*, *S* and *LAC* resulting from the three different weighting methods. Correlation coefficients were always higher than 0.83, showing that in our study the effect of the weighting method was relatively minor (Table A4.5 - Table A4.7). Thus, we selected the statistical weights to assess risk in all further analyses.

Aggregating the components and associating them to values at risk

We combined the components using Eq. (4.1). As the relationship between hazard magnitude and immediate loss of exposed values that define susceptibility is non-linear, mediated by the exponent *S*, we used FWI data from the literature that corresponded to complete forest loss (immediate losses of values) to adjust the susceptibility coefficient, *S* (Appendix 4 - Aggregating the components). Then, we raised the distribution of hazard magnitudes for each plot to its susceptibility and truncated the results so that the maximum immediate loss was 1 (i.e., 100% of values were lost). By multiplying the result by the lack of adaptive capacity and the exposed values we obtained one distribution of values at risk in each plot for each ecosystem service at risk. From the distribution of ES at risk, we defined two conditions: average and extreme. Average risk conditions were defined by extracting the median value of each distribution. Extreme risk conditions corresponded to the 90th percentile of each distribution. We mapped the ES at risk under average and extreme conditions, generating one map for each ecosystem service and condition. We also mapped the relative changes in risk associated to extreme vs. average hazard conditions using the log-ratio of extreme to average conditions (i.e., log((percentile 90th of the risk)/(median risk))).

Data analysis

To analyze the influence of the components of risk (*E*, *HM*, *S* and *LAC*) on the spatial variability of the risk of losing forest ecosystems services, we initially conducted Pearson's correlation tests between them to assess if the different components of risk were spatially associated among each other.

To analyze the effect of the components of risk (*E*, *HM*, *S* and *LAC*) under average and extreme risk conditions, we conducted a sensitivity analysis using the 'tgp' R package on a random sample of 500 plots. This analysis is based on a fully Bayesian Monte Carlo sensitivity analysis, drawing Random Latin hypercube samples at each Markov Chain Monte Carlo iteration to estimate main effects and first order and total sensitivity indices (Gramacy, 2016).

To determine the effect of climate and forest functional type on the risk of losing forest ES under average and extreme hazard conditions, we conducted regression trees for the four ES at risk (i.e., carbon sink, bird richness, hydrological control and erosion control) and the two situations (i.e., average and extreme). The explanatory variables were forest functional type and climate. For forest functional type, we used four groups depending on the dominant species in the plot: broadleaf evergreen, broadleaf deciduous, Mediterranean conifer and non-Mediterranean conifers (Table A4.9 and Fig. A4.13c). The climate variables used were Mean Annual Temperature (Temp) and Mean Annual Precipitation (Prec) from the Catalan Digital Climatic Atlas (period 1951-1999 at a resolution of approximately 180 m) (Fig. A4.13a and b) (Ninyerola et al., 2000). Regression trees were conducted using the 'caret' R package, and are based on recursive partitioning techniques that repeatedly split the predictor variables into multiple sub-spaces, so that the outcomes in each final sub-space are as homogeneous as possible (Kuhn et al., 2015). To increase the robustness of the regression tree models, we used a random subset of 80% of the data to produce the model – or train it (using repeated cross-validation for control) and the other 20% of the data for testing (cross-validation).

To determine the influence of climate and forest functional type on the increase in risk associated to extreme vs. average hazard conditions, we conducted regression trees for the log-ratio of the risk under extreme and average conditions (log((percentile 90th of the risk) / (median risk))), with forest functional type and climate (Temp and Prec) as explanatory variables (Fig. A4.13).

4.3 Results

Influence of exposed values, hazard magnitude, susceptibility and lack of adaptive capacity on the risk of losing ecosystem services

The spatial distribution of the risk components (exposed values, hazard magnitude, susceptibility and lack of adaptive capacity) showed some common patterns depending on the risk condition (i.e., average or extreme) and the exposed value considered (Fig. 4.2). Under average conditions, the highest hazard magnitude was in southern areas, which corresponded with areas at the highest average risk for all ES, whereas the lowest hazard magnitude and risk were observed in northern areas (Fig. 4.2 and Fig. 4.3). Under extreme conditions, the highest hazard magnitude was in central and southern areas, which corresponded with the highest risk for bird richness and hydrological control (Fig. 4.2 and Fig. 4.3). The highest values of extreme risk of carbon sink and erosion control were more sparsely distributed so that they were not coincident with the highest values of extreme hazard magnitude (Fig. 4.2 and Fig. 4.3).

The sensitivity analysis showed that under average conditions, hazard magnitude was the risk component having the greatest influence for all ES, whereas susceptibility and lack of adaptive capacity were the least influential (Table 4.1). Under extreme conditions, hazard magnitude remained a very influential factor but its importance was lower than under average conditions and, in the case of carbon sink and erosion control, exposed values became more important than hazard magnitude (Table 4.1).

Table 4.1: Mean of total effect sensitivity indices of the components of risk (exposed values (*E*), hazard magnitude (*HM*), susceptibility (*S*) and lack of adaptive capacity (*LAC*)) under average and extreme hazard conditions, estimated from the Markov Chain Monte Carlo iterations conducted in the sensitivity analyses (Fig. A4.5 - Fig. A4.12).

	Carbon sink		Bird richness		Hydrol. control		Erosion control	
	Average	Extreme	Average	Extreme	Average	Extreme	Average	Extreme
E	0.42	0.46	0.38	0.44	0.26	0.23	0.62	0.73
HM	0.53	0.43	0.54	0.46	0.57	0.53	0.69	0.47
S	0.17	0.13	0.18	0.12	0.22	0.17	0.36	0.24
LAC	0.15	0.18	0.19	0.23	0.24	0.26	0.38	0.24

Effect of climate and forest functional type on the risk of losing forest ecosystem services

The regression trees showed that climate and forest functional types (Fig. A4.13) were meaningful factors in defining groups of ES at risk from wildfires under average and extreme conditions. Under average conditions, annual precipitation (hereafter precipitation) was the main factor defining risk groups for all ES except for erosion control. In particular, humid forests (i.e., with precipitation > 697, 733 or 768 mm/yr depending on the ES, see Fig. 4.4) had the lowest risk of losing carbon sink capacity, bird richness and hydrological control capacity in case a wildfire occurred. For these three ES high risk was also associated with warm conditions (temperature > 10 ºC). Functional type was also important, with all forest types except Mediterranean conifers (Carbon sink) or specifically non-Mediterranean conifers (Bird richness being at highest risk). For erosion control, high risk was determined primarily by forest type (higher for non-Mediterranean conifers) and relatively warm temperatures (> 7.8 ºC, Fig. 4.4).

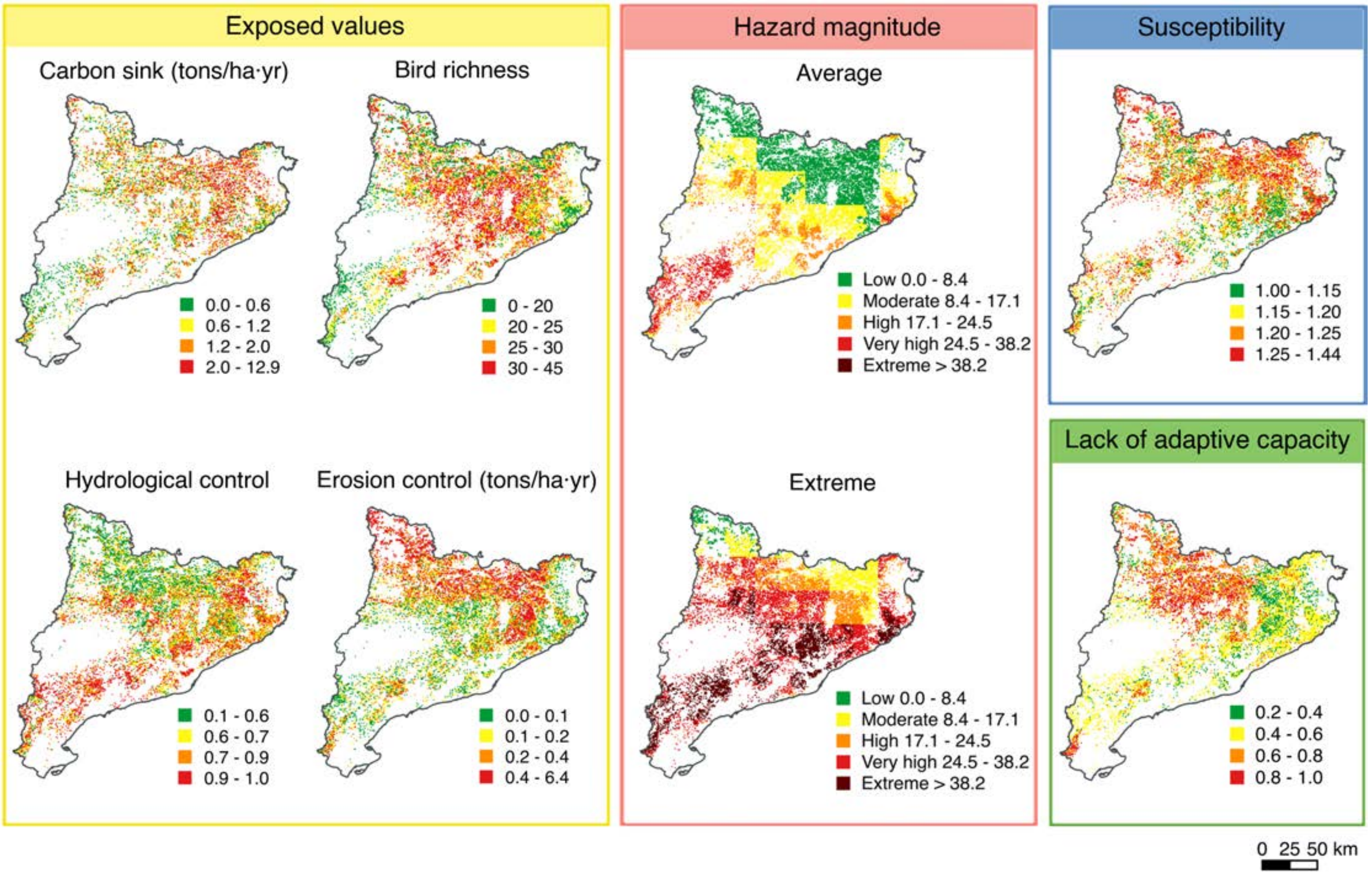

FIGURE 4.2: Spatial distribution of the exposed values (E) (carbon sink, bird richness, hydrological control and erosion control), wildfire hazard magnitude (average and extreme) (HM), susceptibility S and lack of adaptive capacity LAC in the study area (Catalonia, NE Spain).

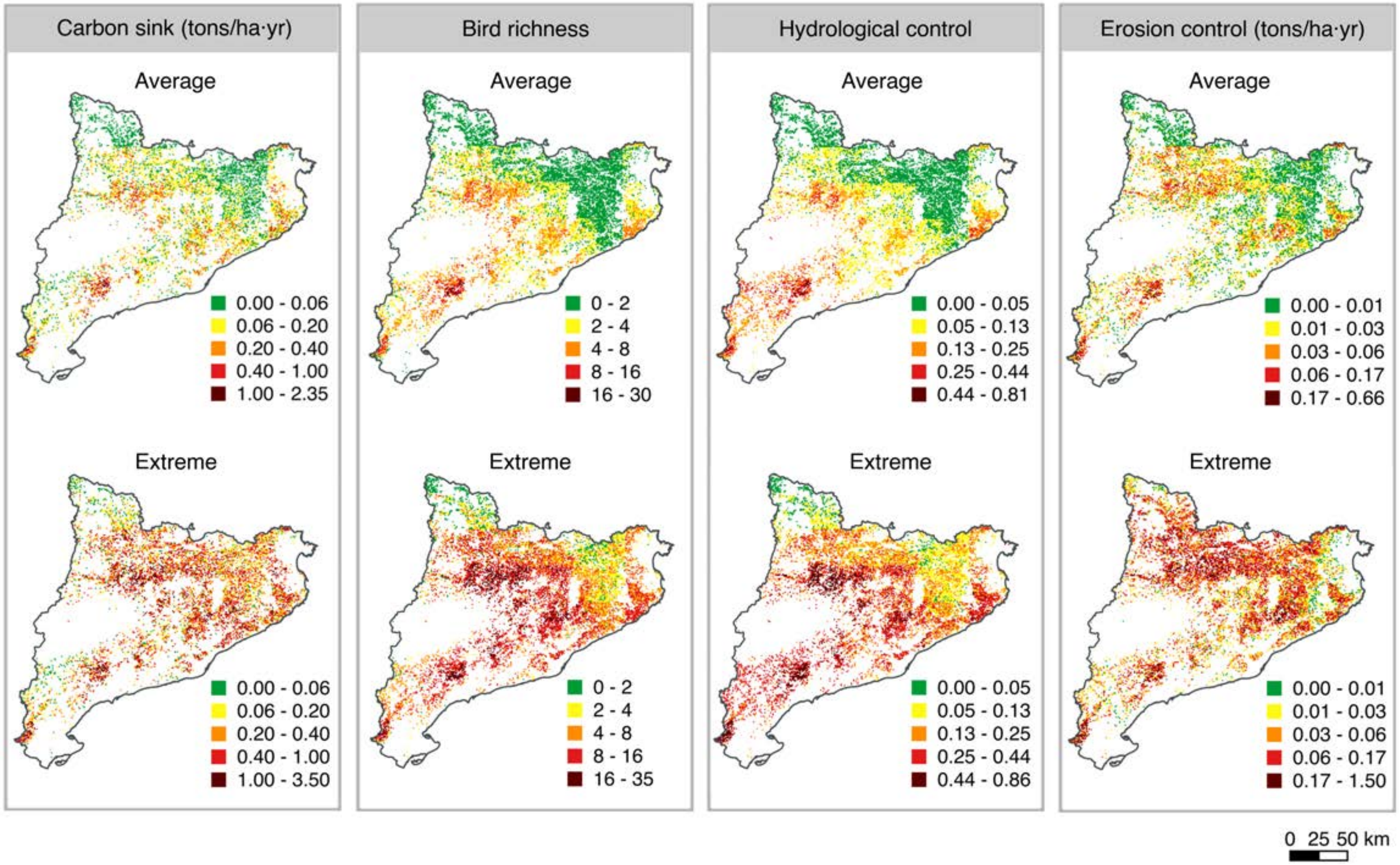

FIGURE 4.3: Spatial distribution of ecosystem services at risk (carbon sink, bird richness, hydrological control and erosion control) in the study area (Catalonia, NE Spain) under average and extreme wildfire hazard conditions.

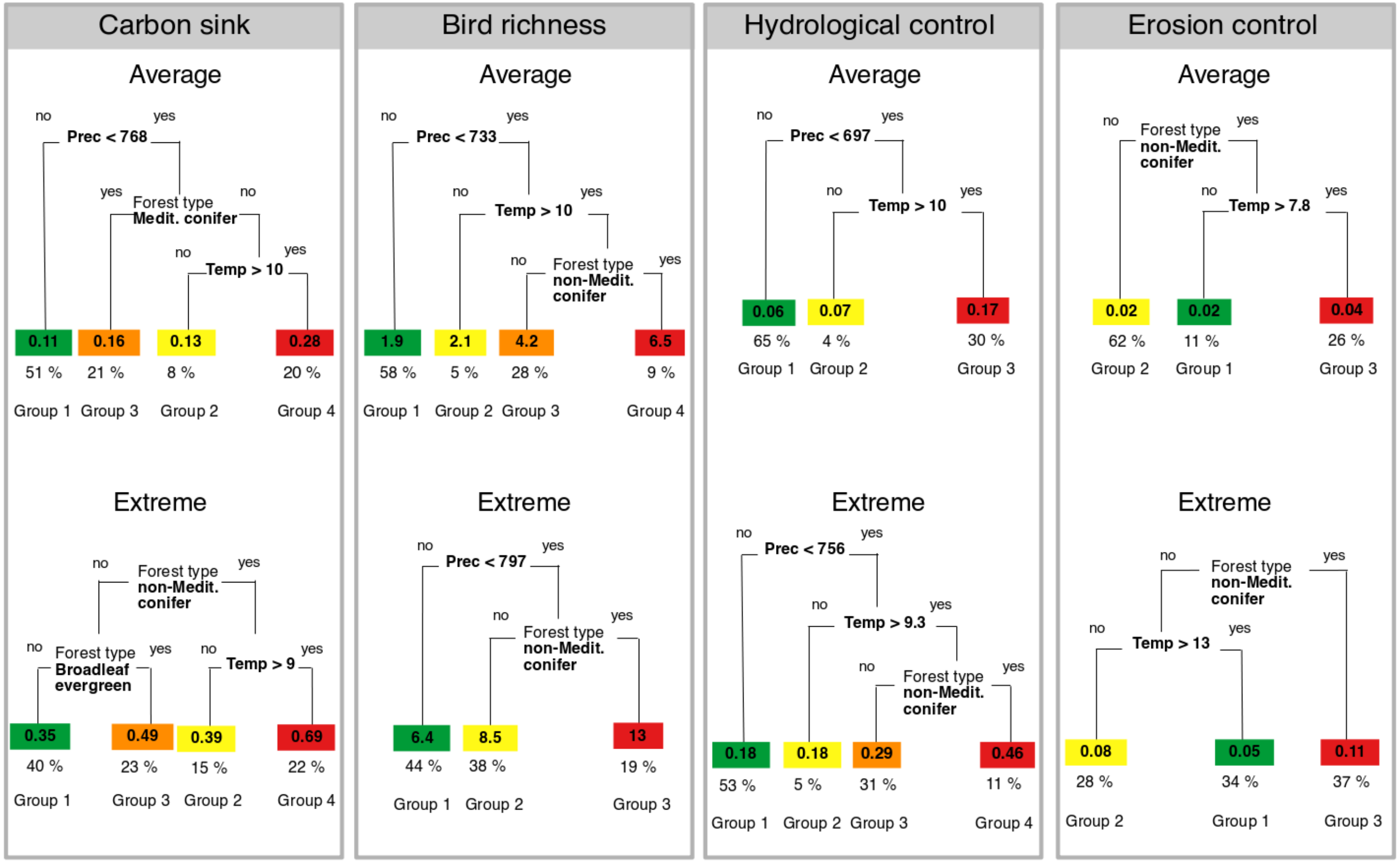

FIGURE 4.4: Regression trees of ecosystem services at risk to wildfires under average and extreme conditions, as a function of climate (Prec: mean annual precipitation, Temp: mean annual temperature) and forest functional type (broadleaf evergreen, broadleaf deciduous, Mediterranean conifer, and non-Mediterranean conifer). Values in color boxes correspond to the mean value of the ES at risk in the group, from green (lowest risk) to red (highest risk), and percentage values indicate the percentage of plots in each group.

The factors defining risk groups were similar under average than extreme conditions in terms of importance and direction. Nevertheless, some differences were observed in the relative importance of the variables and the specific thresholds (Fig. 4.4). In general, the importance of forest type increased and, for all four ES, non-Mediterranean conifers were associated to the highest risks. Precipitation remained the main factor determining the risk of losing bird richness and hydrological control capacity, but with slightly higher thresholds than under average conditions. Warm temperatures also remained associated to high risks for carbon sink and hydrological control, albeit with slightly lower thresholds (around 9 ºC). In contrast, high temperatures (> 13 ºC) were associated with the lowest risk of losing erosion control capacity.

Influence of climate and forest functional type on potential increases in risk

High risk change (i.e., high log-ratios of extreme vs average conditions, Fig. A4.14) is determined by the distribution of hazard magnitudes and hence largely by the FWI distribution at each location. The highest increases in risk were observed in forests with low average risk (i.e., northern areas, Fig. 4.5a), whereas the lowest risk increase was observed in areas where average risk was the highest (i.e., in the southern areas, Fig. 4.5a). The regression tree showed that precipitation was the main factor determining risk change, with two different thresholds depending on the group considered. The lowest increase in risk was observed in forests with less than 606 mm/year of precipitation, whereas the highest increase in risk was observed in forests with more than 815 mm/year of precipitation (Fig. 4.5b). High risk increases from average to extreme conditions were associated with forests with high carbon sink capacity, high erosion control and low hydrological control, whereas the correlation with bird richness was weak (correlations of 0.27, 0.46, -0.30 and 0.03, respectively – Table A4.11).

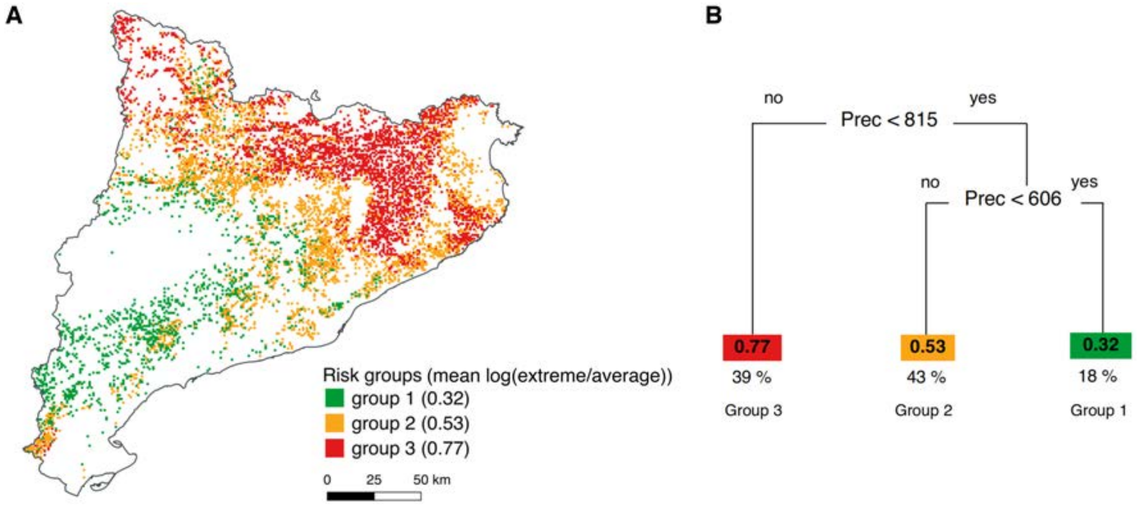

FIGURE 4.5: (A) Spatial distribution of the log-ratio of extreme vs. average hazard conditions; (B) regression tree of the log-ratio as a function of climate (Prec: mean annual precipitation, Temp: mean annual temperature) and forest functional type (broadleaf evergreen, broadleaf deciduous, Mediterranean conifer, and non-Mediterranean conifer). In the regression tree, values in color boxes correspond to the mean value of the log-ratio of the group, from green (lowest risk) to red (highest risk), and percentage values indicate the percentage of plots in each group.

4.4 Discussion

Overall, hazard magnitude was the most important component defining risk under average conditions and, interestingly, exposed values (in particular carbon sink capacity and erosion control) emerged as the most important component of risk when conditions were extreme. As initially hypothesized, climate was the main driving factor of ES at risk under average conditions, but forest functional type - in particular the dominance by non-Mediterranean conifers - gained importance under extreme conditions. Nonetheless, the increase in risk (change from average to extreme ES at risk) was driven by precipitation, with the highest increases in risk in relatively wet forests with low average risk.

Influence of exposed values, hazard magnitude, susceptibility and lack of adaptive capacity on the risk of losing ecosystem services

As initially hypothesized, hazard magnitude was the most important component of risk, especially under average conditions (Table 4.1). The FWI is the main indicator of hazard magnitude as defined here. This metric is one of the most widely used indexes to predict forest fire hazard and has been previously related with wildfire occurrence and area burned in Mediterranean regions (Palheiro et al., 2006; Amatulli et al., 2013; Pérez-

Table 4.2: Mean ± standard error of the components of risk (exposed values (E), hazard magnitude, susceptibility and lack of adaptive capacity) for the different forest functional types (broadleaf evergreen, broadleaf deciduous, Mediterranean conifer and non-Mediterranean conifer).

	Forest functional types			
	Broadleaf evergreen	Broadleaf deciduous	Medit. conifer	non-Medit. conifer
E - carbon sink	1.67 ± 0.03	1.86 ± 0.05	1.17 ± 0.02	1.44 ± 0.02
E - bird richness	24.7 ± 0.14	26.9 ± 0.19	25.3 ± 0.15	25.6 ± 0.14
E - hydrological control	0.77 ± 0.003	0.65 ± 0.005	0.83 ± 0.002	0.66 ± 0.003
E - erosion control	0.30 ± 0.006	0.42 ± 0.01	0.13 ± 0.002	0.36 ± 0.006
Hazard magnitude - average	11.9 ± 0.17	7.77 ± 0.17	18.5 ± 0.16	9.03 ± 0.11
Hazard magnitude - extreme	30.0 ± 0.24	23.2 ± 0.26	37.5 ± 0.18	24.7 ± 0.18
Susceptibility	1.22 ± 0.001	1.21 ± 0.002	1.20 ± 0.001	1.22 ± 0.001
Lack of adaptive capacity	0.49 ± 0.001	0.41 ± 0.003	0.45 ± 0.002	0.79 ± 0.002

Sánchez et al., 2017). Our results are consistent with previous studies that found that areas with high fire occurrence and area burned suffered strong impacts on their forest ES (Thom and Seidl, 2016; Harper et al., 2018; Pausas and Keeley, 2019). Although previous studies showed that adaptive capacity was a relevant component of vulnerability and risk (Román et al., 2013; Thorne et al., 2018), we found that it was among the least influential factors of ES at risk according to the sensitivity analysis (Table 4.1). In our study, however, adaptive capacity is highly dependent on the forest functional characteristics (e.g., Mediterranean conifers have post-fire regeneration strategies whereas non-Mediterranean conifers do not (Rodrigo et al., 2004) and varies strongly in space (see below).

When conditions were extreme, hazard magnitude lost importance, and in the case of carbon sink and erosion control, exposed values became the most important factor (Table 4.1). Under extreme conditions, the extent of high hazard magnitude increased towards central and northern areas (Fig. 4.2), consistent with previous studies suggesting that extreme wildfires could move towards higher latitudes and elevations in the Mediterranean (Vilà-Cabrera et al., 2012; Duguy et al., 2013). These areas are characterized by broadleaf and non-Mediterranean conifer forests that store more carbon than Mediterranean conifers located in southern areas (Table 4.2) (Vayreda et al., 2012). In addition, broadleaf and non-Mediterranean forests are located in humid but also steep areas, thus having higher rain erodibility, vegetation cover and slope-length steepness factors, resulting in high erosion control (Table 4.2). Therefore, the increase in the extent of risk under extreme conditions would lead to more exposed carbon sink and erosion control at risk and, consequently, more ES could be lost if a wildfire occurred.

Effect of climate and forest functional type on the risk of losing forest ecosystem services

As hypothesized, ES at risk were primarily driven by climate. Under average conditions, humid forests had the lowest risk of losing all ES except erosion control. This is an expected result because low precipitation has a strong effect on area burned through decreasing fuel moisture and increasing flammability (Littell et al., 2009; Holden et al., 2018). Less humid and warm conditions put carbon sink at the highest risk for all forest functional types except for Mediterranean conifers (i.e., *Pinus halepensis*, *Pinus pinea*, *Pinus pinaster*) (Fig. 4.4). Low precipitation and warm temperatures increased hazard magnitude which, together with high levels of carbon sink exposed in all forests except Mediterranean conifers (Table 4.2) (Vayreda et al., 2012), resulted in the highest risk of losing carbon sink. In the case of bird richness, non-Mediterranean conifers (e.g., *Pinus sylvestris*, *Pinus nigra*, *Pinus uncinata*, *Abies alba*) and forests growing under warm conditions were at the highest risk (Fig. 4.4). Mean annual temperature was found to negatively affect bird richness in the study area (Lecina-Diaz et al., 2018), as most of the forest birds are cold-dwelling species located in the southern limit of their distribution in Europe (Regos et al., 2017). Although changes on bird communities are common shortly after fire due to changes in habitat and resource availability, bird richness returns to pre-fire levels after few years (Saracco et al., 2018; Zlonis et al., 2019). However, post-fire habitat changes in non-Mediterranean conifer forests could be exacerbated due to the lack of post-fire adaptive capacity of these species (Table 4.2) (e.g., replacement by other tree species such as broadleaves (De Cáceres et al., 2013)), with consequences for forest bird communities. For erosion control, the highest risk was observed in non-Mediterranean conifers with relatively warm temperatures (> 7.8 ºC, Fig. 4.4). The lack of post-fire adaptive capacity of non-Mediterranean conifers compared with the rest of forest functional types (Tapias et al., 2004; Rodrigo et al., 2004) could also result in higher erosion risk due to growth limitations after fire (Maringer et al., 2012; Reyes et al., 2015), but temperature limited risk in these forests through hazard magnitude, at least under average climate conditions (i.e., low FWI in areas with low temperature) (Fig. 4.4).

Under extreme conditions, climate was still a relevant factor for all ES at risk, albeit with different precipitation and temperature thresholds (Fig. 4.4). In particular, more humid and less warm areas than under average conditions were at high risk, which has been already shown in previous studies that related less arid climates that have more biomass with extreme fire severity (Lecina-Diaz et al., 2014). In contrast, the importance of forest type increased, with non-Mediterranean conifers having the highest ES at risk (Fig. 4.4). Under extreme conditions, climate did not characterize risk groups because all forest functional types had very high or extreme hazard magnitude (Fig. 4.3 and Fig. A4.13). Under these conditions, what differentiates the forest functional types is the highest lack of adaptive capacity of non-Mediterranean conifers, which resulted in this forest type being the one with the highest ES at risk. This result is not necessarily obvious, since warming climate is increasing unfavorable post-fire growing conditions regardless of the forest functional type (e.g., lower seedling and resprouting capacity due to unsuitable climate) (Enright et al., 2015; Stevens-Rumann et al., 2018). Nevertheless, non-Mediterranean conifer forests in humid regions (i.e., with higher precipitation thresholds) had been previously affected by extreme wildfires and showed limited regeneration compared with the other forest types (Retana et al., 2002; Rodrigo et al., 2004; Pausas et al., 2008). This is generally consistent with previous studies that described them as vulnerable due to their lack of regeneration capacity, which is tightly linked to seed dispersal from surviving trees (Vilà-Cabrera et al., 2012; Christopoulou et al., 2014). As a consequence, these forests often transitioned into other forest types (mainly dominated by resprouter species) or even into other vegetation types such as shrublands (Retana et al., 2002; Pérez-Cabello et al., 2010), resulting in very high impacts on their ES.

Influence of climate and forest functional type on potential increases in risk

Ongoing climate change is likely to exacerbate wildfire risk in many areas, so that currently extreme hazard conditions become normal. Thus, characterizing forests based on the change from (currently) average to extreme conditions could provide new insights on future ES at risk to wildfires. As hypothesized, the highest increases in ES at risk occurred in the most humid forests of the study area, which currently are under low risk. These relatively wet forests grow under no water limitations, so that they are associated with high carbon sink capacity and erosion control (Table A4.11). Although these forests are not frequently affected by wildfires (Díaz-Delgado et al., 2004; Brotons et al., 2013), previous studies suggested an increase of wildfires in northern latitudes and higher elevations in Mediterranean regions (Duguy et al., 2013). Climate change will increase the severity and intensity of drought events in the Mediterranean, resulting in increases of more than 50% in days favorable for extreme wildfire events (Vilà-Cabrera et al., 2012; Bowman et al., 2017). By the 2080s, future scenarios of 2-4 ºC of temperature increases in southern Europe also involve reduction of precipitation up to 30% (Vautard et al., 2014), and increases in future wildfire activity are expected in other regions of the world (Moritz et al., 2014; Liu et al., 2013; Coogan et al., 2019). Therefore, new areas with high forest ES at risk that appear due to climate change should be considered a priority in management policies directed towards susceptibility reduction or adaptive capacity improvement.

4.5 Conclusions

We have assessed the risk of losing ecosystem services due to wildfires using a set of available indicators that have been applied in a comprehensive and widely applicable framework (Lecina-Diaz et al., in press). This approach could be easily applied to other regions and to other climate- change hazards, establishing a basis for systematic risk assessments that could inform policy makers, resource and land use managers. Our study revealed the current and future risk of losing ES if a wildfire occurred, highlighting the large differences among forest functional types, related with their different adaptive capacity. In this sense, management approaches favoring broadleaf species over non-Mediterranean conifers can be promoted to increase adaptive capacity and consequently decrease the risk of losing ES. However, it is not clear how future climate conditions may change species distributions and fire regimes, and how these changes will affect future hazard magnitude, susceptibility and adaptive capacity, which collectively define risk. In fact, we have approximated future hazard conditions using extreme values of current hazard magnitude, yet a better understanding of the future distribution of hazard magnitude remains a key challenge. Despite these issues, this study constitutes an important advance to the quantification of forest vulnerability and risk in Mediterranean and non-Mediterranean systems, given that increases in forest disturbance regimes in Europe are likely to intensify in the future (Seidl et al., 2011), particularly the increasing vulnerability to fire expected in other regions of the world (Buotte et al., 2019). This study aims at contributing to future-oriented policies by anticipating conditions associated with particularly high risks that can be used to guide efficient forest management (i.e., reducing hazard probability and susceptibility and increasing adaptive capacity).

Acknowledgments

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Conclusions

This thesis has analyzed the spatial distribution of forest ecosystem services, their relevance in conservation and their vulnerability and risk to climate-change hazards. To do so, different ES, levels of analysis and study areas have been considered. In particular, we have analyzed the spatial distribution, relationship and drivers of forest carbon stocks and biodiversity in two regions and five subclimates (Spain and Quebec). We have also determined the role of protected areas in preserving ecosystem services and biodiversity in Catalonia. Then, we have developed a general framework of forest vulnerability and risk, including all the IPCC components and readily applicable to different forest types and hazards. Finally, we have applied this framework to assess the spatial patterns and drivers of ecosystems services at risk from wildfires in Catalonia (NE Spain).

Chapter 1. Carbon stocks - biodiversity relationships

1.1 We have determined the spatial patterns of carbon stocks and biodiversity and the factors that influence them. We have also established the relationships between forest carbon stocks and biodiversity, and we have defined and characterized the areas of high (hotspots) and low (coldspots) values of carbon and biodiversity, as well as their degree of spatial overlap. To do so, we have integrated information of National Forest Inventories and Breeding Bird Atlases across Europe and North America (Spain and Quebec, respectively), covering five subclimates (steppe, dry mediterranean, humid mediterranean, temperate and boreal).

1.2 The highest values of carbon and biodiversity are in northern Spain (humid Mediterranean subclimate) and southern Quebec (temperate subclimate), where there is more carbon as climate conditions are less limiting. High density and structural diversity simultaneously have favored carbon stocks, tree and overall biodiversity, especially in isolated and mountainous areas, often associated with steeper slopes and low accessibility.

1.3 The relationship between carbon stocks and biodiversity is positive in both regions and all subclimates, being stronger in high-contrasting climatic conditions (i.e., steppe and boreal subclimates). Forests with high tree diversity can use resources in a more efficient way through niche partitioning, thus having greater levels of productivity. In addition, competition tends to be reduced in favor of complementarity in more stressful environments, and species interactions improve the availability and efficient use of resources.

1.4 The spatial overlap between hotspots of carbon and biodiversity provides an excellent opportunity for landscape planning to maintain carbon stocks and conserve biodiversity. We have found a high percentage of overlap between hotspots of carbon stocks and biodiversity, especially in the north of Spain and in southern and south-western Quebec. In these areas, maintaining greater carbon stocks will also likely conserve higher levels of biodiversity, and vice versa. Moreover, the variables positively affecting carbon and biodiversity have been also driving the hotspots of both carbon and biodiversity, emphasizing the viability of 'win-win' solutions.

Chapter 2. Ecosystem services and Protected Areas

2.1 We have assessed the spatial distribution of ecosystem services (carbon stocks and water provision), biodiversity (woody and bird richness) and conservation variables (threatened bird richness, habitats and geology) in forests and shrublands of 108 Protected Areas (PAs) from a Mediterranean region (Catalonia, NE Spain). We have quantified the differences in these ecosystem services, biodiversity and conservation variables between the PAs (with different protection status) and buffer zones, considering not only the whole range of values but also the highest values (hotspots).

2.2 We have found higher values of carbon stocks in PAs than in buffer zones. Most PAs are located in mountainous areas, where there are the highest values of carbon stocks (i.e., north of Catalonia, Pyrenees and Pre-Pyrenees, as well as coastal mountainous plains). Although none of the biodiversity variables have showed differences between PAs and buffer zones, we have found more coverage of community-interest habitats, priority-habitats and geological-interest sites in PAs than in buffer zones.

2.3 Overall, PAs with a higher degree of protection (i.e., moderate vs partial) have not provided higher levels of ecosystem services and biodiversity, or vice versa. In particular, moderate PAs have more carbon stocks than partial PAs, because partial PAs are mainly Natura 2000 sites which have not been designated for carbon sequestration purposes. The only conservation variable being significantly higher in moderate than in partial PAs is the percentage of geological-interest sites, since some moderate PAs have been designated, among other reasons, for being geologically singular.

2.4 Concerning hotspots of ES, biodiversity and conservation variables, we have found more hotspots of woody richness, bird richness and threatened bird richness in buffer zones than in PAs. Nonetheless, none of the hotspots variables have showed significant differences between moderate and partial PAs, meaning that a high degree of protection does not provide high levels of ES and biodiversity.

2.5 These results highlight the importance of maintaining biodiversity not only inside PAs but also in their surrounding buffer zones. These buffer zones containing hotspots of biodiversity can be seen as an opportunity to delineate a network of green infrastructure that would enhance connectivity between PAs.

Chapter 3. Characterizing forest vulnerability and risk

3.1 We have proposed a general framework to assess forest vulnerability and risk based on the concepts of exposure, hazard magnitude, susceptibility and lack of adaptive capacity as defined by the IPCC. Vulnerability in forests has two components: susceptibility, related to the immediate effects of the hazard, and adaptive capacity, which measures the mid-term response after hazard occurrence. The risk of losing ecosystem services is the combination of exposed values, hazard magnitude and vulnerability.

3.2 The different components of vulnerability and risk to the four main hazards (i.e., wildfires, drought, pests and windstorms) have been defined by intrinsic and extrinsic factors that can be quantified using explicit indicators. We have suggested specific examples of indicators readily applicable to the main climate change-related hazards to forests.

3.3 We have also proposed a methodology to combine the components of vulnerability and risk that considers their strong interdependencies in forests, as well as the methodological steps for the application of this conceptual framework.

3.4 This framework and its methodology constitute a basis for a systematic operationalization of forest risk and vulnerability for policy makers as well as for forest and land managers that can be applied to develop future-oriented policies.

Chapter 4. Ecosystem services at risk to wildfires

4.1 We have applied the conceptual framework defined in chapter 3 to assess the risk of losing ecosystem services due to wildfires. In particular, we have combined a set of available indicators to quantify the different components of vulnerability and risk (i.e., exposed values, hazard magnitude, susceptibility and lack of adaptive capacity). Afterwards, we have assessed the risk of losing four ES (i.e., carbon sink, bird richness, hydrological control and erosion control) under average and extreme hazard conditions.

4.2 We have shown that hazard magnitude is the most important factor defining risk under average conditions. Under extreme conditions, hazard magnitude loses importance and exposed values (in particular carbon sink capacity and erosion control) emerges as the most important factor of ES at risk.

4.3 Climate is the main driving factor of ES at risk under average conditions, but forest functional type - in particular non-Mediterranean conifers with low adaptive capacity - gains importance under extreme conditions.

4.4 The increase in risk between average and extreme conditions has been driven by precipitation, with the highest increases in risk in relatively wet forests with currently low average risk, which according to climate trends will become common in the future.

4.5 These results have relevant implications on the future risk of losing ES due to wildfires in Mediterranean forests but also in other regions, and could contribute to future-oriented policies by anticipating conditions associated with particularly high risks that can be used to guide efficient forest management.

Appendix 1

Supplementary Results

Supplementary Tables

Table A1.1: Correlation coefficients (pearson at p<0.001) of the computed variables. Numbers in bold indicate the variables excluded from the GLM analysis.

	Sqrt C (C/-forest ha)	Overall bio (B)	Bird richness (Bb)	Tree richness (Bt)	Basal Area	Density	Struct. div (Hd)	Slope	% forest	Forest type	Annual Mean Temp
Overall bio (B)	0.66	-	-	-	-	-	-	-	-	-	-
Bird richness (Bb)	0.7	-	-	-	-	-	-	-	-	-	-
Tree richness (Bt)	0.65	-	0.65	-	-	-	-	-	-	-	-
Basal Area	**0.84**	0.62	0.69	0.62	-	-	-	-	-	-	-
Density	0.6	0.52	0.6	0.42	**0.72**	-	-	-	-	-	-
Structural div (Hd)	0.7	0.57	0.52	0.62	**0.76**	0.33	-	-	-	-	-
Slope	0.06	0.25	-0.04	0.07	-0.04	-0.11	0.1	-	-	-	-
% forest	**0.78**	0.41	0.43	0.42	0.42	0.36	0.37	0.11	-	-	-
Forest type	0.23	0.15	0.26	0.24	0.19	0.18	0.15	0.03	0.14	-	-
Annual Mean Temp	-0.65	-0.54	-0.81	-0.42	-0.64	-0.73	-0.36	0.24	-0.49	0.29	-
Annual Precip	0.73	0.58	0.65	0.53	0.69	0.62	0.54	0.07	0.49	0.2	-0.71

Table A1.2: Correlation tests (pearson at p<0.001) of carbon (C) (without accounting for forest area, i.e., in Mg C/ha) and biodiversity (Bb, Bt and B). Significant values are showed in bold.

	Tree carbon stocks (Mg C/ ha)						
	Regions		Subclimates				
	Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Temperate	Boreal
Forest bird richness (Bb)	**0.42** **p <0.001**	**0.22** **p <0.001**	**0.26** **p <0.001**	**0.17** **p <0.001**	**0.23** **p <0.001**	0.09 p = 0.001	**0.24** **p <0.001**
Tree species richness (Bt)	**0.39** **p <0.001**	**0.62** **p <0.001**	**0.25** **p <0.001**	**0.23** **p <0.001**	**0.34** **p <0.001**	**0.57** **p <0.001**	**0.51** **p <0.001**
Overall biodiversity (B)	**0.48** **p <0.001**	**0.47** **p <0.001**	**0.31** **p <0.001**	**0.24** **p <0.001**	**0.35** **p <0.001**	**0.36** **p <0.001**	**0.48** **p <0.001**

Table A1.3: Percentage of overlap between hotspots and coldspots of carbon (C) and overall biodiversity (B); carbon (C) and bird richness (Bb); and carbon (C) and tree richness (Bt), in the regional and subclimate classification. Consider that the highest percentage of overlap would be 100% (e.g., when the 20% highest C values overlapped with the 20% highest B values)

		Regions		Subclimates				
		Spain	Quebec	Steppe	Dry Medit.	Humid Medit.	Temperate	Boreal
C and Bb	C + Bb +	75.5	33	89.5	65	56.5	29.5	41.5
	C + Bb -	3	19	5	14	17	27	9
	C - Bb +	4	24	12.5	7	8.5	25	29.5
	C - Bb -	81	44.5	84.5	64	73	40.5	53.5
C and Bt	C + Bt +	72.5	68.5	59	54	58	57	87.5
	C + Bt -	5.5	3.5	4.5	14.5	6.5	3	3
	C - Bt +	7.5	13	14	15	4.5	21.5	9
	C - Bt -	57	87.5	90	50.5	71.5	78	99
C and B	C + B +	85	35	80.5	68	68.5	29.5	69
	C + B -	2.5	11	5	11	7	17	3
	C - B +	2.5	23.5	12	8	5.5	26	19
	C - B -	81	54.5	80.5	59.5	91.5	58	78

Table A1.4: Multinomial Logistic Models of carbon (C) + bird richness (Bb) for the regional and the subclimate classification, with C- Bb- areas as the reference level. Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1. Intercept is the forest type Broadleaf. Note that the boreal subclimate was not included due to lack of data.

				Regions				Subclimates							
				Spain		Quebec		Steppe		Dry Med.		Humid Med.		Temperate	
C + Bb +	Forest stand variables	Density		1.9	.	2.6	*	-1.5		0.3		3.7	***	0.1	
		Structural div (Hd)		6.3	***	4.2	***	2.5	*	6.3	***	4.6	**	2.8	**
	Landscape variables	Slope		5.8	***	4.2	***	2.7	**	7.9	***	3.2	**	5.3	***
		Forest type	Conifer	-0.7		-2.7	**	-1.2		-3.8	***	-0.2		0	
			Broadleaf	-3	**	-3.9	***	0.5		1.7	.	-3.8	***	-3.2	**
			Mixed	1.1		1.6		-0.1		-0.7		0.5		1.3	
	Climate variables	Mean Annual Temp		-6.6	***	-0.5		-2.3	*	-6.6	***	-1.7	.	2.3	*
		Mean Annual Prec		5.4	***	-0.3		1.5		3.2	**	2.4	*	0	
C + Bb -	Forest stand variables	Density		1.5		2.3	*	-1.7	.	0.5		4.3	***	0.1	
		Structural div (Hd)		3.8	***	4.2	***	1.4		3.7	***	4.5	***	2.7	**
	Landscape variables	Slope		2.8	**	4	***	2.6	*	2.8	**	2.4	*	5.3	***
		Forest type	Conifer	-1.4		-2.3	*	0		-2.8	**	-0.5		0.1	
			Broadleaf	-4.7	***	-3.5	***	0		0.7		-5.7	***	-2.1	*
			Mixed	2	*	1.1		0		-1.4		0.9		0.2	
	Climate variables	Mean Annual Temp		-0.7		-2.1	*	-2.2	*	-3.9	***	1.3		-0.8	
		Mean Annual Prec		4.7	***	-0.7		1.6		2.7	**	2.8	**	-1	
C - Bb +	Forest stand variables	Density		-2.7	**	-1.7	.	-1.1		0		-0.8		-2.9	**
		Structural div (Hd)		1.1		-0.1		0.5		-0.4		0.6		-2	*
	Landscape variables	Slope		0.1		1.2		0.4		1.3		-0.7		3.8	***
		Forest type	Conifer	-1.2		-0.9		0.3		0		-0.8		0	
			Broadleaf	1		0.9		1.6		1.8	.	-0.5		1.2	
			Mixed	0		1.9	.	0		0.3		3.1	**	1.3	
	Climate variables	Mean Annual Temp		-5.4	***	1.5		-2.1	*	-3.7	***	-1		2.8	**
		Mean Annual Prec		5.4	***	-1.1		-0.7		2.1	*	2.5	*	-0.7	

Table A1.5: Multinomial Logistic Models of carbon (C) + tree richness (Bt) for the regional and the sub-climate classification, with C - Bt - areas as the reference level. Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1. Intercept is the forest type Broadleaf.

				Regions				Subclimates									
				Spain		Quebec		Steppe		Dry Med.		Humid Med.		Temperate		Boreal	
C + Bt +	Forest stand var.	Density		3.6	***	3.1	**	2	*	2.8	**	4.4	***	2.1	*	2.2	*
		Structural div (Hd)		7.4	***	5.6	***	2.4	*	4.9	***	4.6	***	6.4	***	2	*
	Landscape var.	Slope		5.8	***	4.6	***	2.7	**	5.9	***	2.1	*	5.6	***	-0.9	
		Forest type	Conifer	3.1	**	0		1.4		3.9	***	1.3		0		-1.7	.
			Broadleaf	-7.5	***	-4.8	***	-2.2	*	-3.8	***	-5.8	***	-4.6	***	NA	
			Mixed	3.3	**	1.9	.	0.3		2.6	**	0.4		0.2		-0.3	
	Climate var.	Mean Annual Temp		-1.1		2.1	*	0		-4.4	***	2.1	*	-1.7	.	2	*
		Mean Annual Prec		5.2	***	-0.4		2.5	*	5.1	***	3.4	***	-2.3	*	0.6	
C + Bt -	Forest stand var.	Density		0.5		3.1	**	2.3	*	-3	**	4.3	***	0.6			
		Structural div (Hd)		5.2	***	4.2	***	1.5		5	***	3.1	**	4.5	***		
	Landscape var.	Slope		0.6		-1		0.8		0.9		0.4		1.7	.		
		Forest type	Conifer	0.7		-1.9	.	-0.4		0.1		0.8		-2	*	NA	
			Broadleaf	-3.7	***	-3.7	***	-2.4	*	-1.1		-4.5	***	-4	***		
			Mixed	0.3		1		0		0		0		-0.5			
	Climate var.	Mean Annual Temp		-1.7	.	-2.7	**	1.4		-2.6	**	1.2		-3.3	**		
		Mean Annual Prec		3.2	**	1.5		2.4	*	2.6	*	3.3	***	1			
C - Bt +	Forest stand var.	Density		0.5		2.8	**	0.7		0.3		4.2	***	3.7	***	1.6	
		Structural div (Hd)		3.8	***	3	**	1.8	.	2.4	*	2.8	**	4.4	***	1.2	
	Landscape var.	Slope		1.5		1.7	.	1.6		4	***	-1.7	.	-1.3		0.2	
		Forest type	Conifer	2.3	*	0		-0.4		3.3	***	0.2		0		-1	
			Broadleaf	-5.3	***	-3.1	**	-1.3		-2.1	*	-3.2	**	-5	***	NA	
			Mixed	2.6	**	2	*	0		1.8	.	-0.1		2	*	1.3	
	Climate var.	Mean Annual Temp		2.2	*	2.9	**	1		-1.3		1.1		3.2	**	2	*
		Mean Annual Prec		2.6	*	0		0.4		1.8	.	2.3	*	1		0	

Table A1.6: Multinomial Logistic Models of carbon (C) + overall biodiversity (B) for the regional and the sub-climate classification, with C - B - areas as the reference level. Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1. Intercept is the forest type Broadleaf. Note that the boreal subclimate was not included due to lack of data.

				Regions				Subclimates							
				Spain		Quebec		Steppe		Dry Med.		Humid Med.		Temperate	
C + B +	Forest stand variables	Density		3.2	**	2.7	**	0.7		2.7	**	4.4	***	1.1	
		Structural div (Hd)		6.3	***	4.9	***	1.3		5.3	***	4.8	***	3	**
	Landscape variables	Slope		5.2	***	4.8	***	1.4		7.2	***	2.9	**	5.5	***
		Forest type	Conifer	0.4		0		0.8		0.7		1.6		0	
			Broadleaf	-4.4	***	-3.5	***	1		-2.3	*	-5.9	***	-2.4	*
			Mixed	1.9	.	0.8		0		1.6		1.4		0.8	
	Climate variables	Mean Annual Temp		-5.6	***	0.9		-1.1		-4.2	***	1.3		2.5	*
		Mean Annual Prec		5.3	***	-1.1		-1		3.8	***	3.7	***	-1.6	
C + B -	Forest stand variables	Density		-1.4		3.1	**	-0.4		-0.3		4.4	***	0.1	
		Structural div (Hd)		4.2	***	5	***	1.2		3.1	**	1.7	.	2.3	*
	Landscape variables	Slope		2.1	*	3.6	***	1.4		0.9		-1.5		4.5	***
		Forest type	Conifer	-0.1		-1.9	.	0		-1.3		0.9		-0.2	
			Broadleaf	-4.3	***	-3.2	**	0		-2.1	*	-5.9	***	-1.8	.
			Mixed	0.9		-0.3		0		0		0.9		0.3	
	Climate variables	Mean Annual Temp		-0.9		-2.9	**	-1.4		-0.8		2.6	**	-1.3	
		Mean Annual Prec		4.7	***	-0.1		-0.9		2.3	*	4.8	***	-0.6	
C - B +	Forest stand variables	Density		0.5		1.3		-0.7		-0.7		3.5	***	0.4	
		Structural div (Hd)		1.6	.	2.5	*	1.2		2.9	**	0.6		1.8	.
	Landscape variables	Slope		2.3	*	0.8		1.3		3.8	***	-2	.	3.1	**
		Forest type	Conifer	0.1		0		0.2		1.3		0.4		0	
			Broadleaf	-3	**	-1.5		1.4		-0.5		-3.8	***	-1.3	
			Mixed	1.7	.	1		0		1.2		1.4		1.1	
	Climate variables	Mean Annual Temp		-1.5		1.8	.	-1.9	.	-2.7	**	1.8	.	2.7	**
		Mean Annual Prec		3.5	***	-1		-0.9		1.8	.	3.6	***	-1.4	

Supplementary Figures

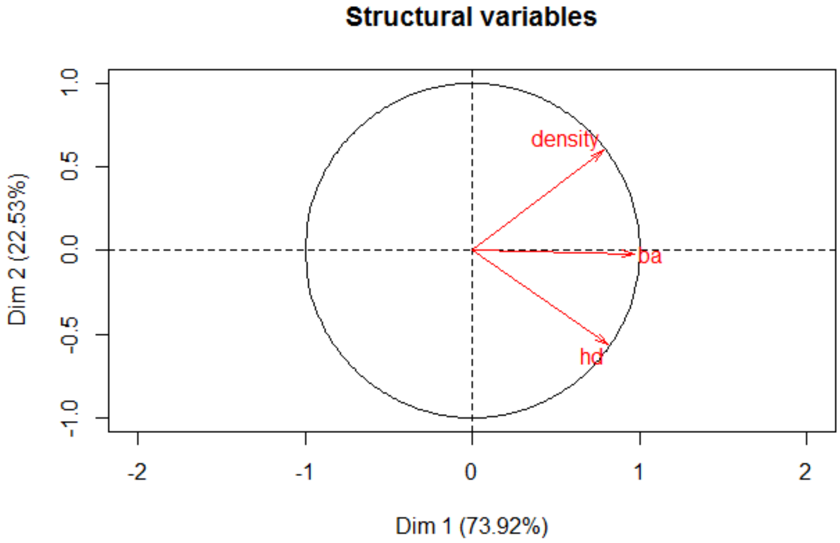


FIGURE A1.1: Principal Component Analysis of the structural variables: density, ba (Basal Area) and structural diversity (Hd).

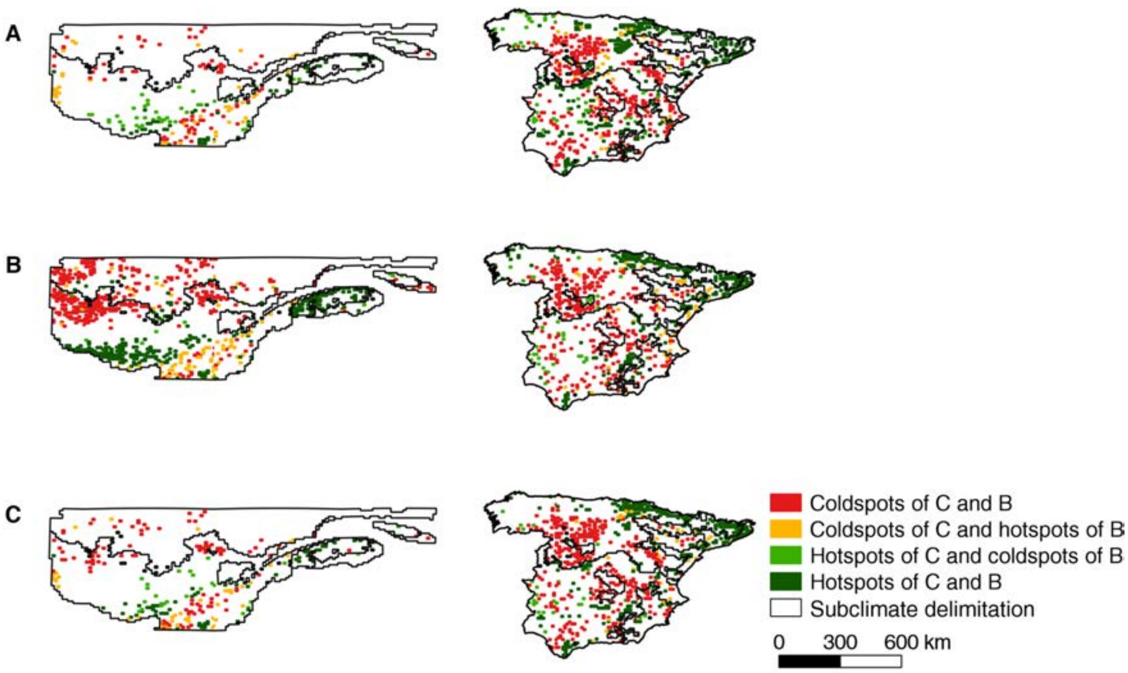


FIGURE A1.2: Synergies and trade-offs between C and B, i.e., overlap of the highest 20% (hotspots) and lowest 20% (coldspots) values of (A) Carbon (*C*) + bird richness (*Bb*); (B) Carbon (*C*) + tree richness (*Bt*) and (C) Carbon (*C*) + overall biodiversity (*B*) for the subclimate delimitation.

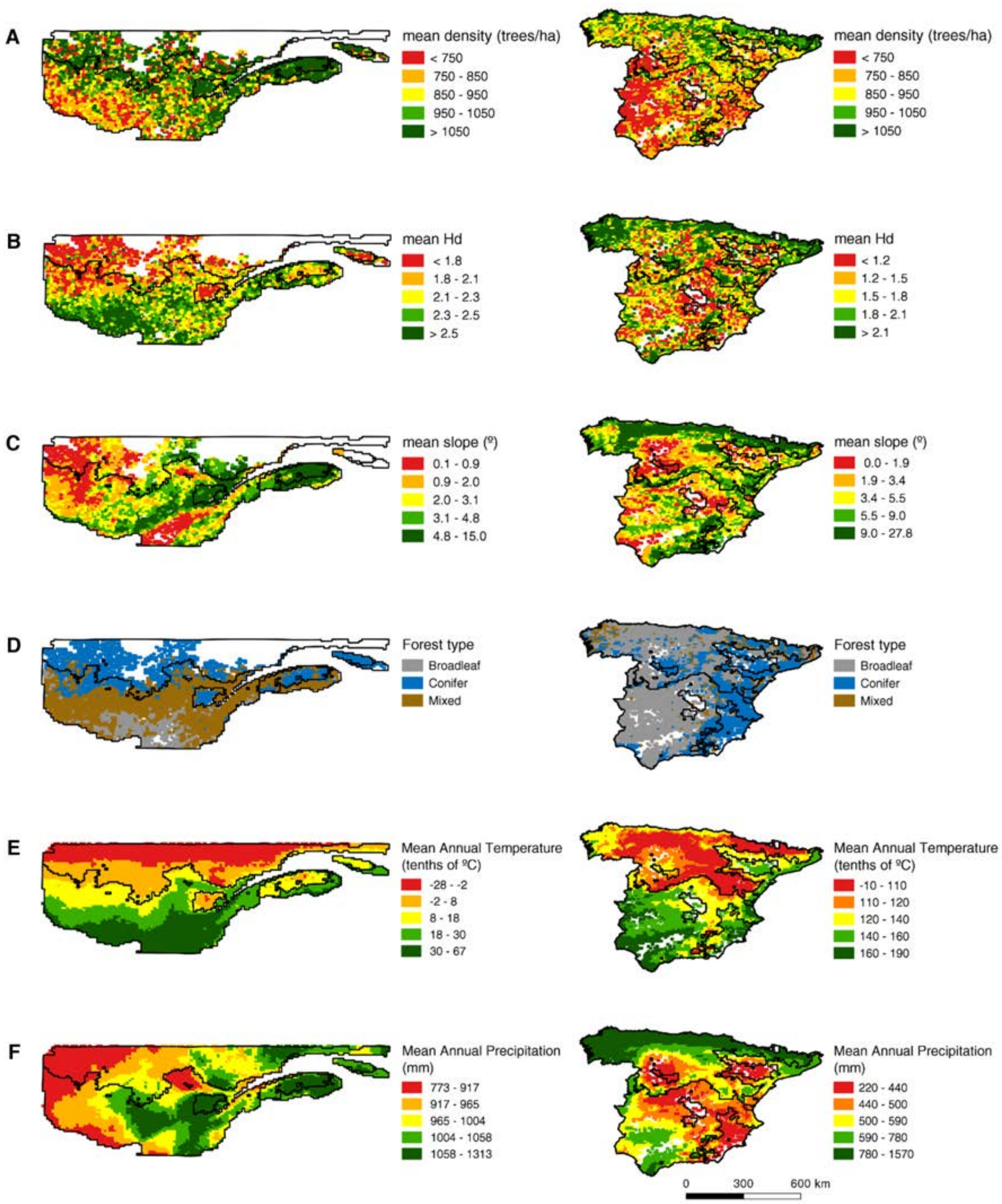


Figure A1.3: Spatial distribution of (A) mean density (trees/ha); (B) mean Hd; (C) mean slope (º); (D) forest type; (E) Annual Mean Temperature (tenths of ºC) and (F) Annual Precipitation (mm), in the two regions and the five subclimates.

Appendix 2

Supplementary Results

Supplementary Tables

Table A2.1: Name, source of information, data type and scaling methods of the variables used in the study.

Variable		Source of information	Data type, resolution	Scaling to protected area/ buffer zone	Scaling to 1 x 1 km cell (for hotspots definition)
Carbon stocks (Mg C/ha)	Forest with inventory data (10,016 nodes)	3rd Spanish National Forest Inventory (2000-2001)	Circular plots (diameter=5-25 m)	Average (Mg C/unit analyzed), weighted by forest and shrub surface	Value of the plot located at the bottom-left (Mg C/ha)
	Forest without inventory data (3,307 nodes)	LIDAR (2009)	2 km grid cell		
	Shrubs (1,753 nodes)	Open forest inventory plots (2000-2001), Catalan habitats map (2005) and photointerpretation (2009)	Circular plots (diameter=5-25 m), shapefile 1:50000		
Water provisioning		Water balance model (de Cáceres et al 2015) (1993-2002)	1 x 1 km points	Average weighted by forest and shrub surface	Value of the plot located at the bottom-left
Biodiversity	Woody richness	3rd Spanish National Forest Inventory (direct data + linear models application) (2000-2001)	Circular plots (diameter=5-25 m)	Average weighted by forest and shrub surface	Value of the plot located at the bottom-left
	Bird richness	2nd Catalan Breeding Bird Atlas (filtered by forest and shrub species) (1999-2002)	1 km grid cell	Average weighted by forest and shrub surface	-
	Threatened bird richness	2nd Catalan Breeding Bird Atlas (filtered by forest and shrub species of conservation interest in the Birds Directive) (1999-2002)	1 km grid cell	Average weighted by forest and shrub surface	-
Habitats (%)	Of community interest	Catalan Government	Shapefile 1:50000	% of coverage in each protected area/buffer zone	% of coverage in each cell
	Priority habitats of community interest	Catalan Government	Shapefile 1:50000	% of coverage in each protected area/buffer zone	% of coverage in each cell
Geological-int. sites (%)	Catalan Government	Shapefile 1:50000	% of coverage in each protected area/buffer zone	% of coverage in each cell	

Table A2.2: Mean ± standard error of the variables of the study in function of their type of zone (protected areas with moderate and partial degree of protection, buffer zone).

		Carbon stocks (Mg C/ unit ha)	Water	Woody richness	Bird richness	Threatened bird richness	Comm. int. habitats (%)	Priority habitats (%)	Geological int. sites (%)
PAs	Moderate	45.9 ± 4.9	38.2 ± 3.2	9.1 ± 0.5	18.2 ± 1.2	1.5 ± 0.1	0.3 ± 0.05	0.02 ± 0.01	0.15 ± 0.02
	Partial	34.2 ± 2.5	40.3 ± 1.7	9.3 ± 0.3	16.9 ± 0.9	1.7 ± 0.1	0.3 ± 0.03	0.05 ± 0.01	0.12 ± 0.02
Buffer zone		33.0 ± 1.6	43.9 ± 1.2	9.6 ± 0.2	16.5 ± 0.6	1.7 ± 0.04	0.2 ± 0.01	0.01 ± 0.00	0.03 ± 0.00

Table A2.3: Percentage of hotspots surface in the different type of zones relative to the overall surface of each type of zone (protected areas, buffer zones and other unprotected areas).

		PAs - protection status			Buffer zone	Other unprotected areas	Total
		Moderate (2)	Partial (3)	Total PA			
Total % surface		10.3	19.6	29.9	54.4	15.6	99.9
Hotspots	Carbon stocks	0.18	0.13	0.31	0.08	0.05	0.44
	Water	0.15	0.13	0.27	0.08	0.07	0.42
	Woody richness	0.08	0.12	0.2	0.09	0.09	0.38
	Bird richness	0.17	0.17	0.34	0.21	0.23	0.78
	Threatened bird richness	0.11	0.27	0.38	0.14	0.11	0.64
	Comm. interest habitats	0.29	0.28	0.56	0.13	0.07	0.77
	Priority habitats	0.04	0.07	0.11	0.03	0.02	0.16
	Geological-interest sites	0.05	0.05	0.11	0.01	0.00	0.12

Supplementary Figures

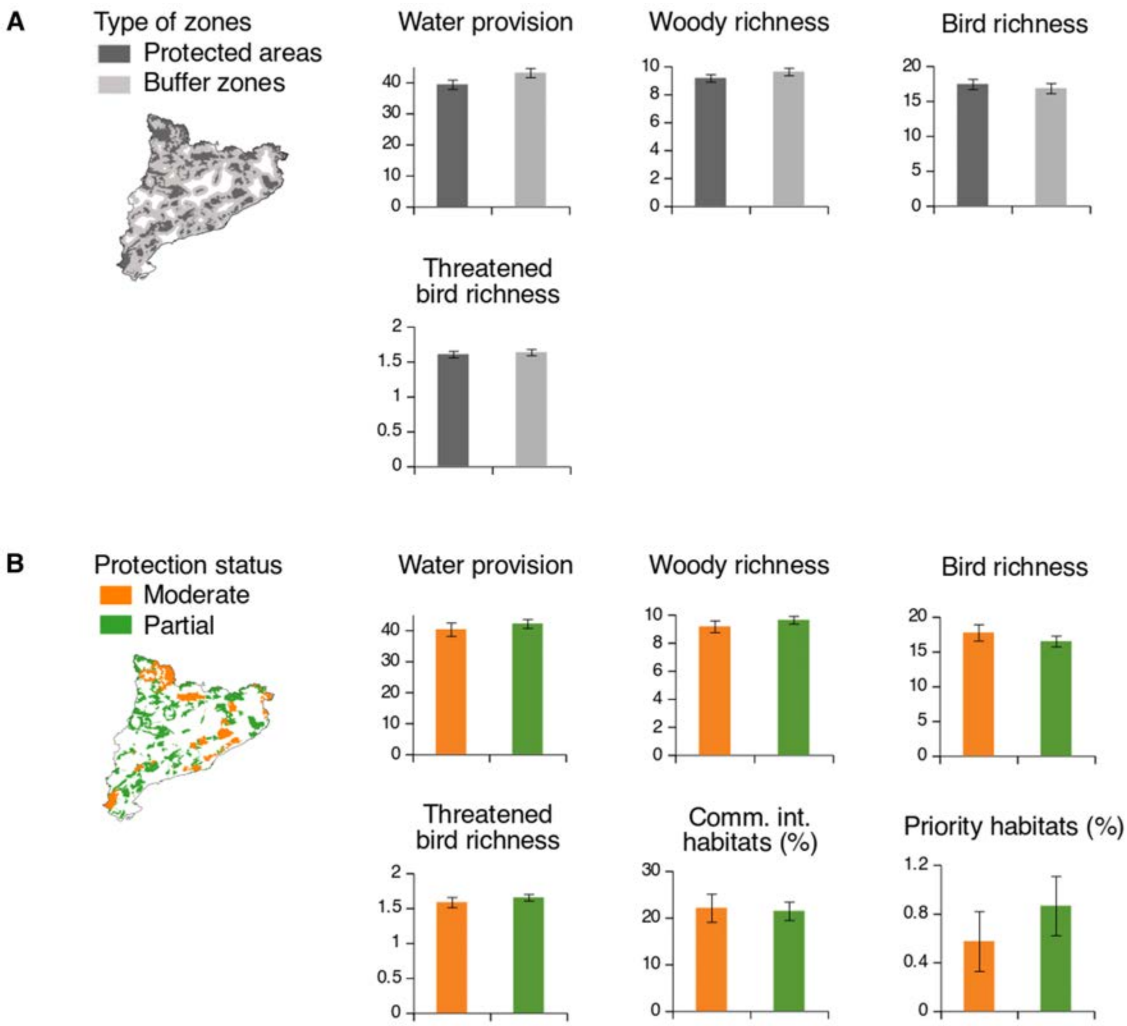


FIGURE A2.1: Plot bars of mean values and standard error of the studied variables showing not significant differences between (A) the types of zones (protected and buffer) and (B) the protection status (moderate and partial).

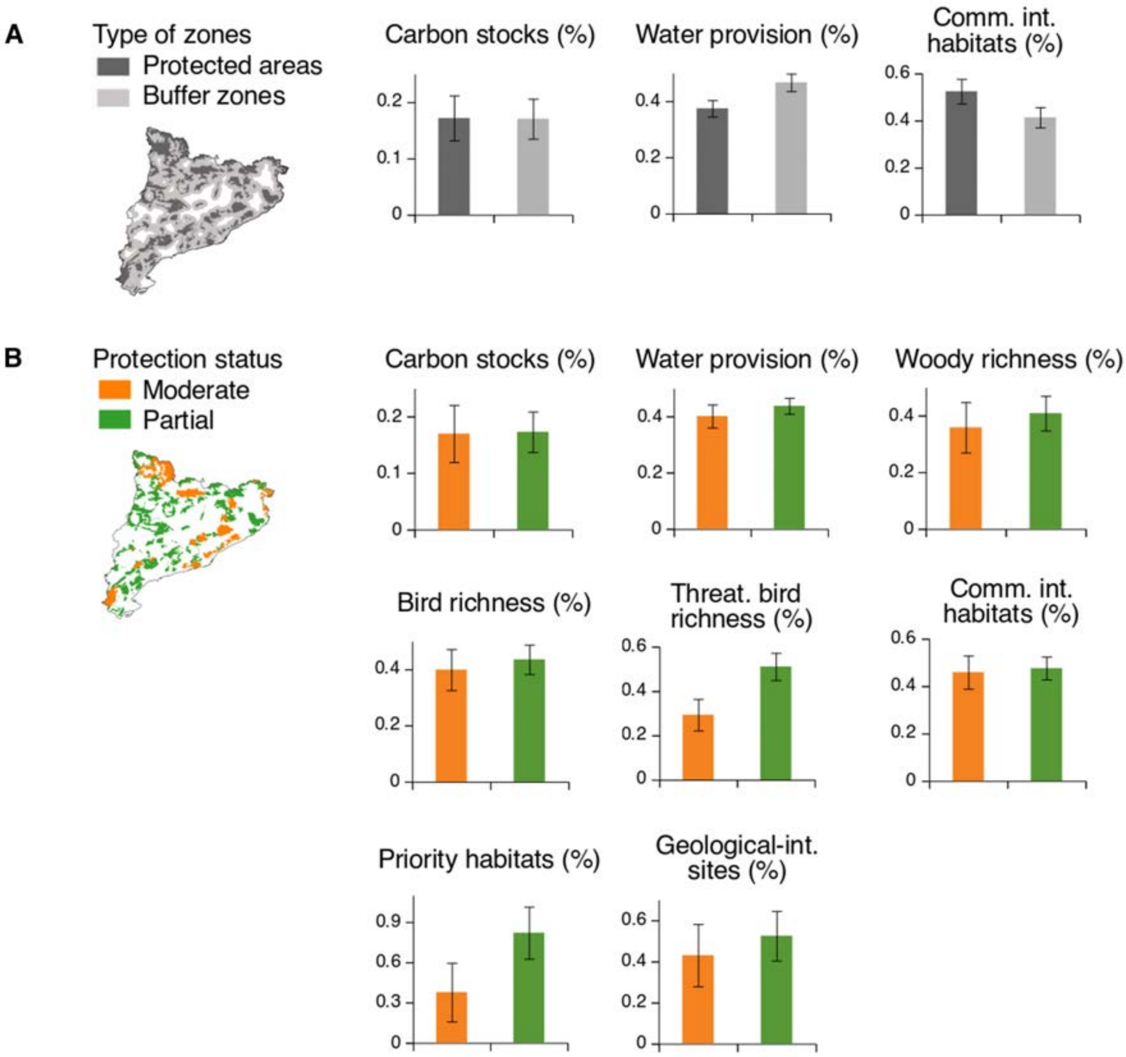


Figure A2.2: Plot bars of mean values and standard error of the hotspots of the studied variables showing not significant differences between (A) the types of zones (protected and buffer) and (B) the protection status (moderate and partial).

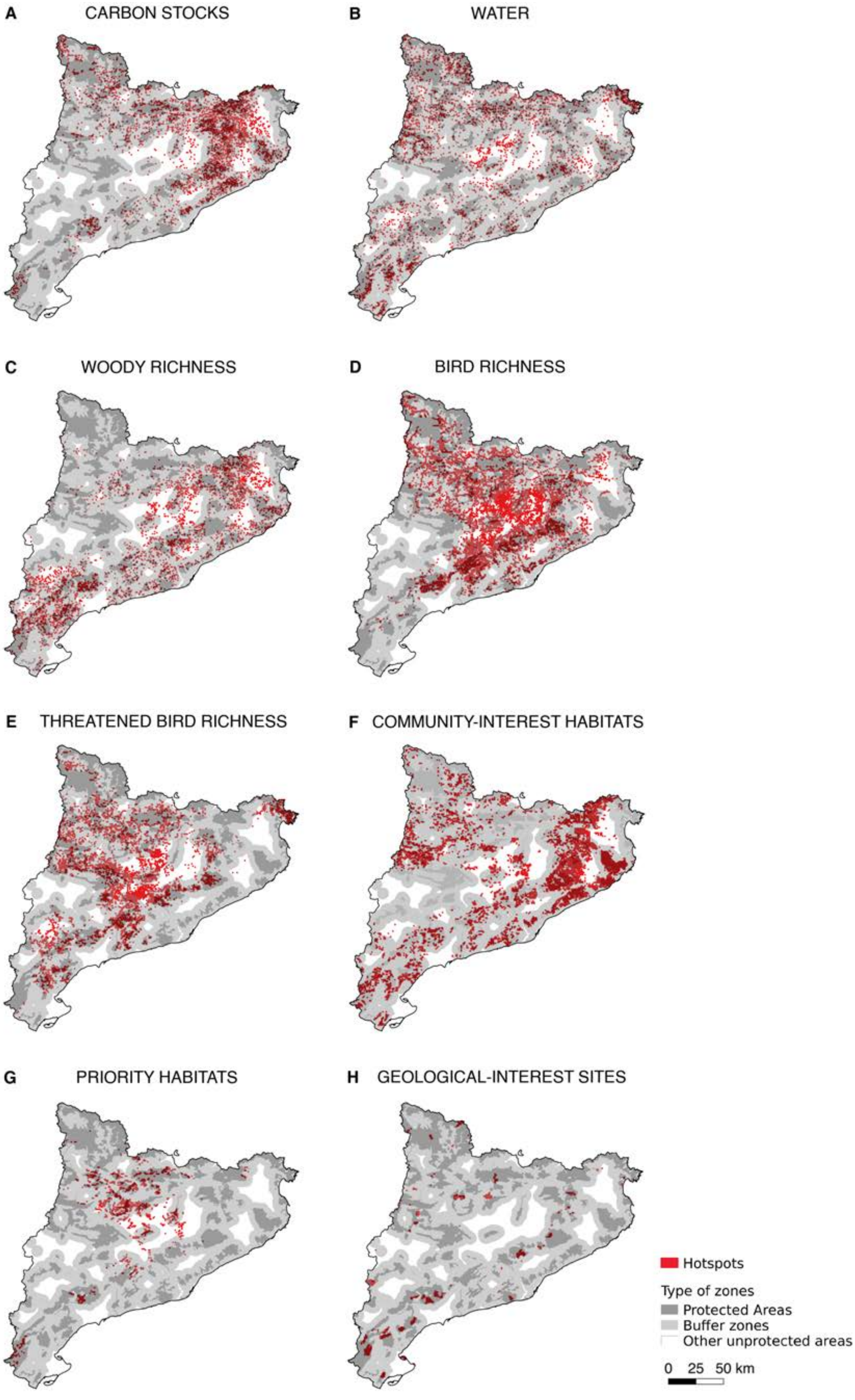


FIGURE A2.3: Spatial distribution of hotspots maps (20th percentile) of the studied variables within the type of zones (protected areas, buffer zones and other unprotected areas).

Carbon stocks - Methodology

Objective

To assess carbon stocks in forest and shrubland areas for the 2000-2001 period in order to have data availability as the other ecosystem services and conservation variables.

Grid nodes

Grid of 1 km resolution over Catalonia, with three different point categories used in the analysis:

1. Node on a 1 km^2 cell that includes a forest inventory plot (11225 points).
2. Node on a 1 km^2 cell without forest inventory plot and including trees, according to the Catalan Land Cover Map MCSC05 (CREAF, 2005) (4041 points).
3. Node on a 1 km^2 cell without forest inventory plot and corresponding to shrubland, according to the Catalan Land Cover Map MCSC05 (CREAF, 2005) (3409 points).

IFN3 plots (category '1')

Aboveground and belowground biomass of trees was derived for each species from allometric equations from Gracia et al. (2004).

1km grid cells category '2'

Tree data

Dominant species were identified from the Catalan Land Cover Map MCSC09 (CREAF, 2009).

Tree canopy cover photointerpreted at each point.

Tree aboveground biomass (Mg/ha) at each point derived from LIDAR.

Tree belowground biomass from species-specific allometric equations from Gracia et al. (2004).

Shrub data Average shrub biomass (Mg C/ha) of the tree dominant species according to the Catalan Land Cover Map MCSC09 and forest inventory plots.

1km grid cells category '3'

In this category, we used the vegetation map to determine the shrub land species (as the Catalan Land Cover Map did not have information regarding the shrub species).

Tree data

If classified as shrubland in the vegetation map, tree data was missing.

If classified as forest in the vegetation map:

- Tree dominant species identity was drawn from the vegetation map.
- Tree foliar biomass (Mg/ha) at each point (resolution 20 m) was derived from LIDAR.

Shrub data

Percentage of shrub cover photointerpreted at each point (25 m buffer).

Shrub dominant species identity (one or two species) derived from vegetation map and grouped following Pasalodos-Tato et al. (2015).

Biomass: combination of shrub cover (photointerpreted) with mean biomass of shrubs (Mg/ha) from IFN3 plots with basal area lower of 5 m^2/ha from the dominant species identity (above-mentioned).

Water balance model

Objective

To describe the water balance modeling on both forest inventory plots (IFN3) and additional grid nodes.

Grid nodes

Grid of 1 km resolution over Catalonia, with three different point categories used in the analysis:

1. Node on a 1 km^2 cell that includes a forest inventory plot (11225 points).
2. Node on a 1 km^2 cell without forest inventory plot and including trees, according to the Catalan Land Cover Map MCSC05 (CREAF, 2005) (4041 points).
3. Node on a 1 km^2 cell without forest inventory plot and corresponding to shrubland, according to the Catalan Land Cover Map MCSC05 (CREAF, 2005) (3409 points).

Grid spatial reference was assumed to be '+proj=utm +zone=31 +ellps='. We considered simulating soil water balance (SWB) on vegetation of grid cells corresponding to categories '2' and '3' points). SWB in forest inventory plots (category '1') was simulated separately.

Climate data

Climatic daily temperature, precipitation and relative humidity for the 1981-2016 period was interpolated on 1 km grid cells (categories '2' and '3') or on forest plot locations (category '1') from available weather station data of the Spanish and Catalan networks (AEMET and SMC). Radiation was calculated according to Thornton and Running (1999), and potential evapotranspiration (PET) was calculated following Penman's formula. Calculations were performed using the R package meteoland (De Cáceres et al., 2018).

Soil parameters

Soil data (i.e. soil texture, organic matter and bulk density for different layers) was obtained for 1 km grid cells (categories '2' and '3') or on forest plot locations (category '1') from SoilGrids (Hengl et al., 2017). Soil grids provides estimates, at 250 m resolution, of soil texture (% sand, % silt and % clay), organic matter content (%) and bulk density (kg/m3) at seven soil depths (0, 5, 15, 30, 60, 100 and 200 cm) for all terrestrial areas. Soil grids also offers an estimation of soil depth to the R horizon, modelled from soil profile data (Hengl et al., 2017), and two additional variables modeled in a companion paper (Shangguan et al., 2017): absolute soil depth (estimated using soil drilling data) and probability of R horizon within the first 2 m. Soil layer macroporosity was calculated from bulk density and sand percentage following Stolf et al. (2011). Three soil layers were considered (0-30 cm; 30-100 cm; 100-200 cm), but they were trimmed according to the estimated depth to the R horizon. An additional rocky layer (to a maximum of 400 cm) with 97.5% rock fragment content was added if absolute soil depth was larger than depth to the R horizon. For forest plots (category '1'), rock fragment content of soil layers was estimated from the percentage of rocks in the surface, using the following table:

Surface class	Surface %	Layer 1	Layer 2	Layer 3
1	0.00 %	0.00 %	12.50 %	70.00 %
2	1-25 %	12.50 %	37.50 %	80.00 %
3	25-50 %	37.50 %	50.00 %	80.00 %
4	50-75 %	50.00 %	50.00 %	87.50 %
5	75-100 %	50.00 %	50.00 %	97.50 %

For 1 km grid cells (categories '2' and '3'), surface percent of rocks was obtained, taking the surface rock content value from the nearest IFN3 plot.

Vegetation parameters

IFN3 plots (category '1')

Foliar biomass of trees was derived for each species from allometries based their DBH and the total Basal Area of Larger trees in the plot (BAL), to account for competition effects. Equations were calibrated using data from Gracia et al. (2004). Biomass values were transformed to leaf area index, using specific leaf area coefficients (SLA in m^2/kg; from the same source). Foliar biomass and leaf area index for shrubs was derived from allometries calibrated at the individual level (details in De Cáceres et al. (2019)). Tree heights and shrub mean heights were directly taken from forest inventory data. Fine root distribution of trees was assumed to follow the linear dose response model with Z50 = 30 cm (50% of roots between 0 and 30 cm depth) and Z95 = 300 cm (fine roots reaching 3 m depth). Root distribution of shrubs is assumed conical and reaching 1 m depth.

1km grid cells category '2'

Tree data

Tree foliar biomass (Mg/ha) at each point (resolution 20 m) derived from LIDAR.

Mean tree height (m) at each point (resolution 20 m) derived from LIDAR.

Tree dominant species identity was drawn from the the Catalan Land Cover Map MCSC09 (one or two species).

Tree leaf area index (LAI) derived from foliar biomass using SLA values (m^2/kg).

Shrub data

Projected forest cover (PFC) of trees and shrubs photointerpreted on each point.

Shrub dominant species identity (one or two species) derived from vegetation map (Mapa d'Habitats).

Mean height of dominant shrub species in the point obtained from IFN3 data.

Average shrub cover (of any species) under trees assumed to be equal to that outside trees.

Foliar biomass and leaf area index (LAI) for shrubs was derived from allometries calibrated at the individual level (details in De Cáceres et al. (2019)).

1km grid cells category '3'

Tree data

If classified as shrubland in the vegetation map, tree data was missing.

If classified as forest in the vegetation map:

- Tree dominant species identity was drawn from the vegetation map (one or two species).
- Tree foliar biomass (Mg/ha) at each point (resolution 20 m) derived from LIDAR.
- Mean tree height (m) at each point (resolution 20 m) derived from LIDAR.
- Tree leaf area index (LAI) derived from foliar biomass using SLA values.

Shrub data

Projected forest cover (PFC) of shrubs photointerpreted on each point.

Shrub dominant species identity (one or two species) derived from vegetation map.

Mean height of dominant shrub species in the point obtained from IFN3 data.

If classified as forest in the vegetation map, average shrub cover (of any species) under trees assumed to be equal to that outside trees.

Foliar biomass and leaf area index (LAI) for shrubs was derived from allometries calibrated at the individual level (details in De Cáceres et al. (2019)).

Soil water balance simulations

Model description

The water balance model described in De Cáceres et al. (2015) (implemented in the R package 'medfate') was used to simulate water fluxes for each forest plot and 1 km cell. Stand structure is represented by the plant's height and LAI. Plants of a same species and of similar size are lumped into cohorts, although the definition of cohort is flexible in the model. The model follows the design principles from BILJOU (Granier et al., 1999) and SIERRA water balance submodel (Mouillot et al., 2001). The model runs start at the beginning of the year with the SWC at full capacity and performs daily updates of SWC as a function of the stand structure and daily weather (radiation, temperature and precipitation). The soil water balance is the difference between infiltration (i.e., precipitation minus canopy interception and surface run-off) and the different water outputs: deep drainage, bare soil evaporation and plant transpiration. Maximum daily transpiration (Emax) is a function of LAI and PET defined by Granier et al. (1999) and actual daily transpiration (E) is the product between Emax and the whole-plant relative hydraulic conductance (G). Further details on the formulation of each of these processes and species parameter values are given in De Cáceres et al. (2015).

Blue water

Blue water (mm = L/m^2) was defined as the sum of water exported daily via runoff and deep drainage. Annual sums of daily blue water were calculated for 1 km cells and for IFN3 plots. The proportion of annual blue water over annual precipitation (0-1) was also calculated.

Simulation period

Daily soil water balance was performed for:

Period 1981-2016 for 1 km cells (categories ‘2’ and ‘3’).

Period 1991-2009 for IFN3 plots (category ‘1’).

Period selection

We selected the 10-year period 1993-2002 to account for the interanual variability in the water balance and to adjust to the periods of the rest of the data (Breeding Bird Atlas 1999-2002 and IFN3 in Catalonia 2000-2001). We therefore tested for the correlation between the average of blue water in two 10-year periods (1993-2002 and 1991-2000) and two other periods (2003-2004 and 2001-2002, respectively). As the correlation was good ($R^2 = 0.96$, $p<0.005$ and $R^2 = 0.97$, $p<0.005$, respectively), we concluded that the average of blue water for the period 1993-2002 was adequate for our analysis.

Tree and shrub richness - Linear Models

Objective

To develop three linear models to complement those nodes with direct available data from forest inventory data (10,016 points). Therefore, we estimate tree and shrub richness in 1) in nodes without forest inventory plot but including trees (tree + shrub model) (3,307 points) and 2) in nodes without forest inventory plot and corresponding to shrubland (shrub model) (1,753 points).

Model description

The models have the following data, response and explanatory variables:

1. Using all available forest inventory data (3rd National Forest Inventory), to be applied to nodes without forest inventory plot but including trees, according to the the Catalan Land Cover Map (CREAF, 2005).

   a) Model with tree species richness as response variable (**Model 1a**).
   b) Model with shrub species richness as response variable (**Model 1b**).

   Where the explanatory variables are:

   Location: x coordinates.
   Location: y coordinates.
   Shrub carbon stocks (Mg C/ha), from forest inventory plots.
   Tree carbon stocks (Mg C/ha), from forest inventory plots.
   Slope (º), from a Digital Elevation Model (20 m resolution).
   Mean annual temperature (Hijmans et al., 2005).
   Mean annual precipitation (Hijmans et al., 2005).
   Main forest species: the predominant species in the forest inventory plot (considering 6 categories: *Pinus halepensis*, *Pinus nigra*, *Pinus sylvestris*, *Quercus ilex*, other conifers, other broadleafs).

2. Using forest inventory data corresponding to open forest inventory plots (i.e. plots with basal area < 5 m^2/ha, the most similar to shrubs), to be applied to nodes without forest inventory plot and corresponding to shrubland, according to the Catalan Land Cover Map (CREAF, 2005).

a) Model with shrub species richness as response variable (**Model 2a**).

Where the explanatory variables are:

Location: x coordinates.

Location: y coordinates.

Shrub carbon stocks (Mg C/ha), from forest open inventory plots.

Slope (º), from a Digital Elevation Model (20 m resolution).

Mean annual temperature (Hijmans et al., 2005).

Mean annual precipitation (Hijmans et al., 2005).

Model results

Table A2.4: Results of the linear models (*t* value and level of significance) for model 1a (tree richness in forests with IFN3 data); model 1b (shrub richness in forests with IFN3 data); and model 2a (shrub richness in open forests with IFN3 data). Signification codes: *** < 0.001, ** < 0.01, * < 0.05.

	Model 1a		Model 1b		Model 2a	
(Intercept)	-4.98	***	7.39	***	8.08	***
Coordinate x	-8	***	-1.21		-0.88	
Coordinate y	6.33	***	-6.30	***	-7.58	***
Shrub carbon stocks	28.35	***	24.20	***	7.95	***
Tree carbon stocks	14.68	***	-14.51	***	-	
Slope	2.74	**	1.58		2.09	*
Mean Annual Precipitation	12.45	***	0.97		-2.80	*
Mean Annual Temperature	14.71	***	26.99	***	10.04	***
Main species (other broadleaf)	9.98	***	-8.38	***	-	
Main species (*Pinus halepensis*)	6.80	***	12.93	***	-	
Main species (*Pinus nigra*)	24.58	***	-1.39		-	
Main species (*Pinus sylvestris*)	27.21	***	-2.67	**	-	
Main species (*Quercus ilex*)	4.78	***	0.04		-	
df	9,780		9,582		1,308	
R^2	0.24		0.51		0.49	

The results of the models shown in Table A2.4 were applied to calculate tree and shrub richness where IFN3 data was not available. In particular, model 1a was applied to calculate tree richness in nodes without IFN3 plots but including trees, model 1b was applied for shrub richness in nodes without IFN3 plots but including trees, and model 2a was applied for shrub richness in nodes without IFN3 plots and corresponding to shrublands.

Qualitative evaluation of ES in PAs

Objective

To analyze the qualitative assessments of ecosystem services (ES) in Protected Areas (PAs) made by the Catalan Government in order to determine the overall importance of provisioning, regulating, cultural and supporting ecosystem services in the PAs of Catalonia.

Data description

The data is located in the website of the Territory and Sustainability Department of the Catalan Government:

`http://mediambient.gencat.cat/ca/05_ambits_dactuacio/patrimoni_natural/senp_catalunya/espais_sistema/` (accessed 29th November 2018).

In the website, there is a list of the Catalan protected areas. In each protected area, there is a description of the elements of the area (e.g., location, socioeconomic factors, photo gallery, etc). One of the elements are the ecosystem services, and includes a downloadable table with a brief description and evaluation of the ecosystem services in the PA (an example is shown in Fig. A2.4).

These tables provide a qualitative approach with the aim to introduce the concept of ecosystem services in PAs, and are based in expert criteria (natural park technicians, government technicians, etc) and some statistical information. The ES evaluated are classified in provisioning, regulating, cultural and supporting services, as well as other subtypes of ES (see example in Fig. A2.4). Each ES was qualitatively evaluated based on their importance within the protected area, distinguishing three main categories of importance: very important, important and present. There is also a brief description of the reasons for including the ES in the category.

There are 71 PAs of our study with this information, which represents a 65.7% of the total PAs included in our study. It contains the most relevant PAs that were included in our study (e.g., ‘Montserrat’, ‘Montseny’, ‘Alt Pirineu’).

Serveis dels ecosistemes de l'espai natural protegit SERRA DEL MONTSANT

Molt significatiu · *Significatiu* · *Present*

DE PROVEÏMENT	Subtipus	Valor del servei	Breu descripció
Provisió de recursos alimentaris	Fruita seca		A l'Espai hi ha petites àrees de conreus d'ametller, noguer i avellaner
	Oli		L'olivera és un conreu important en alguns sectors de l'Espai, especialment l'occidental
	Vinya		Algunes superfícies de vinya prosperen a l'Espai, especialment a la seva cara sud, tant dins la DOQ Priorat com la DO Montsant
	Bolets		A l'Espai es produeixen importants quantitats de bolets
	Mel		A l'Espai i els voltants hi ha diverses explotacions apícoles
	Productes del bosc		A l'Espai es fa la recol·lecció d'alguns productes com diversos fruits del bosc
	Caça		L'Espai s'incorpora en diverses àrees privades de caça
	Ramaderia		A l'Espai es practica la ramaderia de cabrum
Medicines naturals i recursos actius			Les comunitats arbustives de l'Espai tenen algunes espècies d'interès en aquest sentit
Provisió de materials	Fusta		A l'Espai hi ha unes poques explotacions silvícoles, especialment de pi roig
	Llenya		L'Espai proveeix de llenya als habitants locals
Recursos hídrics	Recàrrega d'aqüífers		L'Espai està conformat per materials permeables que tenen una gran importància en la recàrrega dels aqüífers subjacents
	Proveïment d'aigua per a consum humà, incloent aigua embotellada		A l'espai hi ha diverses fonts
	Proveïment d'aigua per a regadiu		L'embassament de Margalef, situat dins l'Espai, abasteix diverses zones de regadiu aigües avall. També hi ha diversos assuts i rescloses per a regadius locals als cursos fluvials de l'Espai
Patrimoni genètic	Races i varietats autòctones		Els conreus de vinya de l'Espai incorporen les varietats dominants de garnatxa i carinyena

DE REGULACIÓ	Subtipus	Valor del servei	Breu descripció
Protecció	Contra l'erosió		Les comunitats forestals i arbustives de l'Espai tenen una gran importància per a la protecció contra l'erosió
Millora de la qualitat de l'aigua	Depuració		L'Espai incorpora diversos trams de rius i torrents (Montsant, riu de Prades, riuet d'Escaladei, Siurana) que tenen una bona capacitat autodepurativa, com també les tenen embassaments com el de Margalef
Mitigació contra els gasos d'efecte hivernacle			La gran massa forestal de l'Espai té un important efecte embornal
Pol·linització			Les espècies de flora existents a l'Espai, especialment les brolles, proporcionen floracions molt interessants per als insectes pol·linitzadors i les zones agrícoles circumdants

FIGURE A2.4: Example of the ES reported by the Catalan Government in the PA 'Serra del Montsant'.

CULTURALS	Subtipus	Valor del servei	Breu descripció
Gaudi del Paisatge			La serra del Montsant té un especial valor paisatgístic a causa dels seus relleus de cingles verticals de conglomerats
Desenvolupament d'activitats de coneixement del medi	Educació i sensibilització ambiental		A l'Espai hi ha diverses iniciatives locals de centres i activitats d'educació ambiental. El Parc Natural de la Serra del Montsant hi desenvolupa i promou aquestes activitats
	Lleure i turisme de natura		Aquest Espai reuneix gran part de l'activitat esportiva i de lleure de muntanya del camp de Tarragona; s'hi practiquen tot tipus d'esports d'aire lliure i hi ha gran quantitat de senderistes i visitants, a més d'escaladors
	Activitats esportives		
	Desenvolupament d'activitats de gestió, recerca i innovació		El Parc Natural de la Serra del Montsant gestiona l'Espai, on es realitzen nombroses activitats de gestió i recerca sobre àmbits molt diversos, incloent el seguiment d'espècies protegides i d'interès, la restauració d'hàbitats, el control de l'ús públic, la recuperació del patrimoni cultural.
Patrimoni històric i cultural			Els elements més remarcables del patrimoni cultural són les nombroses ermites existents, algunes d'origen romànic com la de Sant Salvador a Fraguerau. Hi ha també algunes pintures rupestres de gran valor, masos, balmes i elements relacionats amb l'ús del territori, així com la situació propera de la cartoixa d'Scala Dei
Gaudi Espiritual i religiós			Com el seu nom indica, el caràcter espiritual i sagrat de la serra de Montsant està determinat per l'existència de les ermites i encara d'ermitans.
Identitat cultural i sentit de pertinença			La serra del Montsant és l'element identitari principal de la comarca del Priorat i un dels elements de l'imaginari espacial del migjorn català

DE SUPORT	Subtipus	Valor del servei	Breu descripció
Biodiversitat			Hi cal subratllar una variadíssima flora, amb elements boreals i també de flora de caràcter rupícola, fauna invertebrada d'interès i comunitat vertebrada d'ambients forestals, rupícoles i arbustius de gran valor
Geodiversitat			La serra de Montsant incorpora fins a 3 geozones i és una mostra molt singular dels relleus de conglomerats recolzats sobre el sòcol paleozoic
Connectivitat i complementarietat ecològica			Es tracta d'un espai bàsic estructurador de la serralada Prelitoral tarragonina, amb connexions a l'oest amb el Pas de l'Ase i a l'est amb les muntanyes de Prades

FIGURE A2.4: (continued)

Results of the qualitative evaluation

We counted the number of distinct PAs that have at least one type of ES (provisioning, regulating, cultural and supporting) in one of the categories of importance (very important, important or present). Results are shown in Figure A2.5. We also analyzed the percentage of PAs that have provisioning, regulating, cultural and supporting ES classified as very important and the relative percentage of each subtype within the main type of ES (Fig. A2.6).

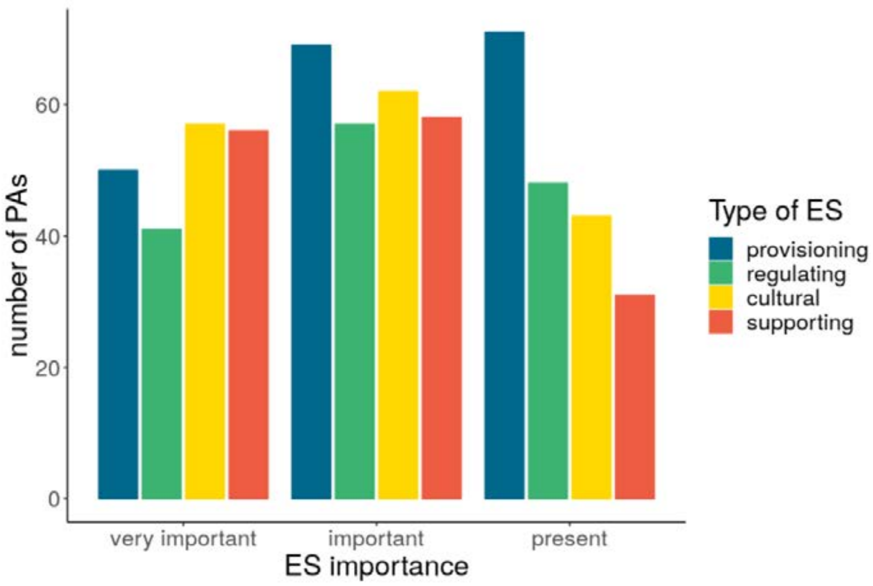


FIGURE A2.5: Number of distinct PAs containing at least one of the ES types in the categories of importance.

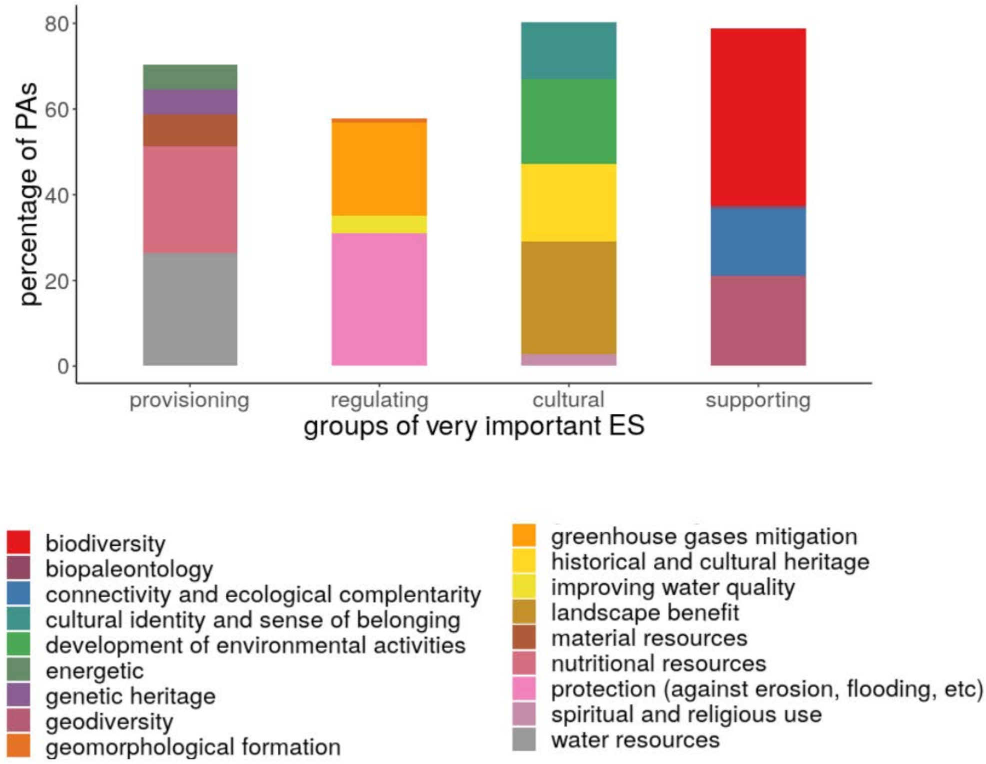


FIGURE A2.6: Percentage of PAs having ES valuated as very important in each group and subtype of ES.

Appendix 3

Table A3.1: Key terms of forest vulnerability and risk. IPCC definitions are referred to the latest IPCC report (IPCC (2018), but see IPCC (2012) for susceptibility).

Term	IPCC definition and specific use in our framework
Exposure, Exposed values (E)	'The presence of [...] environmental functions, services, and resources [...] in places and settings that could be adversely affected' (exposure). In our case, we use exposed values, referring to the ecosystem services that could be affected by the hazard.
Hazard magnitude (HM)	'The potential occurrence of a natural or human-induced physical event or trend that may cause loss of [...] service provision, ecosystems and environmental resources.' In our case, the potential occurrence of climate change hazards that could cause loss of forest ecosystem services.
Susceptibility (S)	'Physical predisposition of human beings, infrastructure, and environment to be affected by a dangerous phenomenon due to lack of resistance and predisposition of society and ecosystems to suffer harm as a consequence of intrinsic and context conditions [...]. In our case, predisposition to be affected by the climate change hazard (i.e., lack of resistance).
Adaptive Capacity, Lack of Adaptive Capacity (LAC)	'The ability of systems, institutions, humans and other organisms to adjust to potential damage, to take advantage of opportunities, or to respond to consequences.' In our case, it refers to the ability to forests to respond to climate change hazards within a predefined timeframe. We use its complementary: lack of adaptive capacity.
Vulnerability (V)	'The propensity or predisposition to be adversely affected.' The combination of susceptibility and (lack of) adaptive capacity.
Risk	'The potential for adverse consequences where something of value is at stake and where the occurrence and degree of an outcome is uncertain. In the context of the assessment of climate impacts, the term risk is often used to refer to the potential for adverse consequences of a climate-related hazard [...]. Risk results from the interaction of vulnerability (of the affected system), its exposure over time (to the hazard), as well as the (climate-related) hazard and the likelihood of its occurrence.' $Risk = E \cdot HM^S \cdot LAC$

Step 1. Selecting the variables meaningful to the framework

We list the variables that can influence the capacity of forests to recover after wildfires.

Lack of adaptive capacity

- Intrinsic factors
  - Species regeneration characteristics
    - Resprouting capacity
    - Seeding capacity
  - Species growth rate
- Extrinsic factors
  - Topography
  - Climate
  - Forest management

List of variables:

- Species regeneration characteristics
  - Species resprouting capacity
  - Species seeding capacity
    - Seedings in the plot
    - Seed dispersal from outside the plot
- Species mean growth rate
- Topography
  - Topographic index
  - Stoniness
- Climate
  - Annual Solar Radiation
  - Precipitation
- Forest management
  - Tree thinning

Step 2. Assessing data availability

From the list in step 1, we select the indicators and the data required to measure them retrieved from available sources.

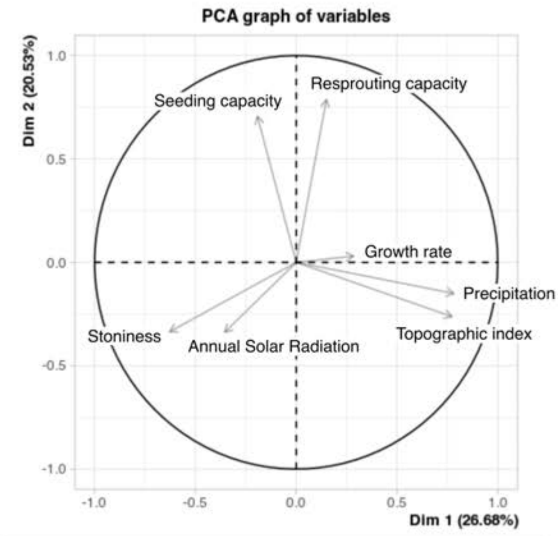
List of variables and data sources:

- Species regeneration characteristics
  - Species resprouting capacity: BROT database (Tavşanoğlu *et al.* 2018).
  - Species seeding capacity: BROT database (Tavşanoğlu *et al.* 2018).
- Species mean growth rate: difference in basal area from two National Forest Inventories.
- Topography
  - Topographic index: Digital Elevation Model.
  - Stoniness: Information in National Forest Inventory.
- Climate
  - Annual Solar Radiation: Climatic Atlas (e.g., Fick and Hijmans 2017).
  - Precipitation: Climatic Atlas (e.g., Fick and Hijmans 2017).

FIGURE A3.1: Example of the methodological steps to apply the conceptual framework of forest vulnerability and risk to wildfires. Example of the component Lack of Adaptive Capacity.

Step 3. Analizing the structure of the dataset

We conduct a Principal Component Analysis (PCA) to analyze the structure of the dataset.



Step 4. Normalizing the indicators to make them comparable

We select the min-max method to normalize the indicators so that they will have an identical range [0 – 1] by subtracting the minimum value and dividing by the range of the indicator values (Nardo *et al.* 2008).

Step 5. Weighting and aggregating the indicators

Different weighting methods can be used: 1) equal weights; 2) statistical weights; or 3) expert weights. In this example, we apply the same weight to all the indicators: 1/7 for each indicator. Afterwards, we aggregate the indicators by summing them.

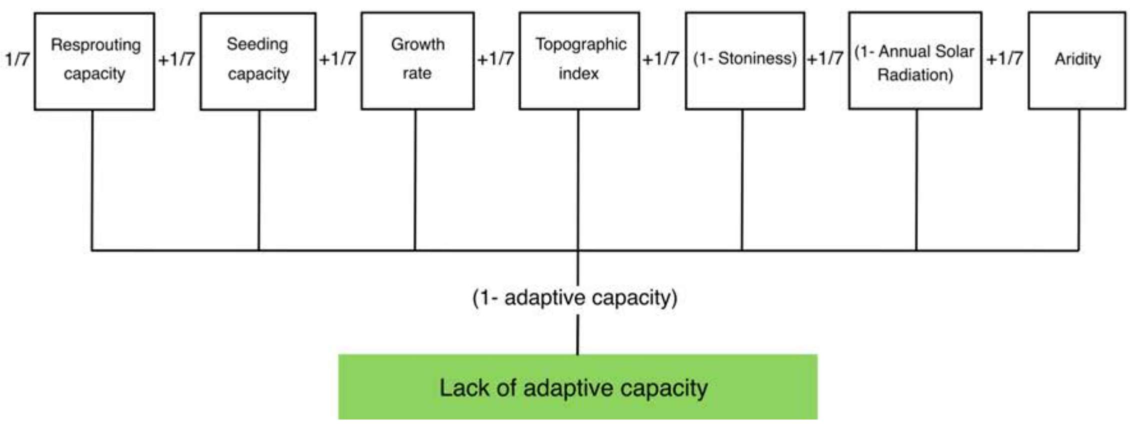
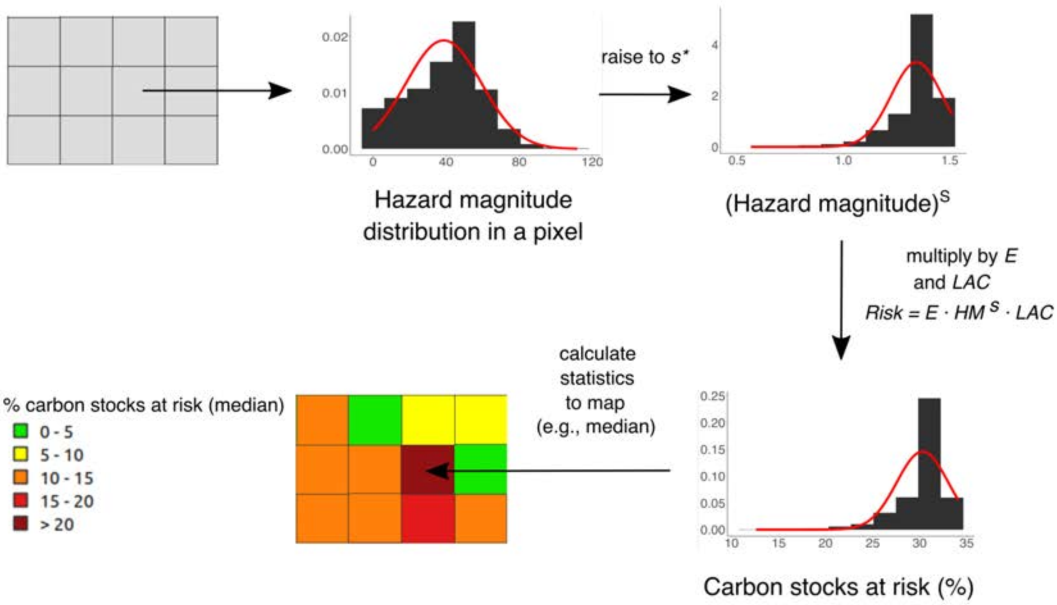


FIGURE A3.1: (continued)

Step 6. Aggregating the components and associate them to values at risk

We repeat the previous steps for all components: Susceptibility and Hazard magnitude. In the case of Hazard magnitude, we have a distribution of the hazard index (e.g., daily values of Fire Weather Index) in each pixel or plot that was modified by the indicators defined in Hazard magnitude. Then, Hazard magnitude is raised to susceptibility so that the distribution at each pixel or plot changes. Afterwards, we select the exposed values (in this case carbon stocks) and calculate the percentage of carbon at risk.

Example of a given pixel or plot:



* If minimum and maximum values of Hazard Magnitude (or the FWI) to immediate loss of all values are known, then *s* can be adjusted by applying the graphical representation in Figure 3b.

Step 7. Conducting a sensitivity analysis

We conduct a sensitivity analysis for the indicators of each component (Hazard Magnitude, Susceptibility and Lack of adaptive capacity). We make changes if required.

FIGURE A3.1: (continued)

Appendix 4

Data sources and indicators used

The indicators used are shown in Table A4.1. The IPCC components of vulnerability and risk are exposed values, hazard magnitude, susceptibility and lack of adaptive capacity (Lecina-Diaz et al., in press). Each component is defined by different indicators that are 1) intrinsic, referred to internal characteristics of the forest (e.g., species characteristics); or 2) extrinsic, referred to external factors typically operating at broad spatial scales (e.g., landscape scale) (Lecina-Diaz et al., in press).

Exposed values

Exposed values are the presence of ecosystem services that could be adversely affected by the wildfire (Lecina-Diaz et al., in press). We used two main datasets: National Forest Inventories (second and third) and Second Breeding Bird Atlas.

The second Spanish National Forest Inventory (hereafter IFN2) was conducted between 1986-1996 (Ministerio de Medio Ambiente, 1996) whereas the third Spanish National Forest Inventory (hereafter IFN3) was conducted between 1997-2007 (Ministerio de Medio Ambiente, 2007*b*). The data consisted of a systematic sampling of permanent plots with a sampling density of one plot in every 1 km^2 of forest area, where trees above 7.5 cm were identified and measured within variable circular size plots (5 m radius for trees with DBH $\geqslant$ 7.5 cm, 10 m radius for trees with DBH $\geqslant$ 12.5 cm, 15 m radius for trees with DBH $\geqslant$ 22.5 cm and 25 m radius for trees with DBH $\geqslant$ 42.5).

The second Catalan Breeding Bird Atlas (Estrada et al., 2004) was conducted between 1999-2002 at a 10 x 10 km cell and downscaled to 1 x 1 km cell.

Regulating services

Carbon sink The capacity of forests to absorbe carbon is defined as the change in carbon stocks from the second to the third IFN (in tons/ha·year). Carbon stocks is computed from tree biomass (above-ground + below-ground, in tons/ha) of each live tree in each IFN2 and 3 plots, that were computed from DBH using species-specific allometric equations developed by Gracia et al. (2004) and Montero et al. (2005) for Spain. We then applied the widely established relationship of 1:0.5 between tree biomass and carbon (McGroddy et al., 2004).

Table A4.1: Indicators (definition, units, data sources and references) for each of the components of forest vulnerability and risk to wildfires used in the study. Abbreviations: IFN2, Second Spanish National Forest Inventory; IFN3, Third Spanish National Forest Inventory; JRC, Joint Research Center; MCSC, Catalan Land Cover Map; ICGC, Catalan Geographical and Cartographical Institute; IEFC, Forest Inventory of Catalonia.

Components and indicators			**Definition**	**Units**	**Data sources**	**References**
Exposed values						
Ecosystem services	Carbon sink		Change in carbon stocks from IFN2 to IFN3.	tons/ha^{-1}yr	IFN2 and IFN3	Ministerio de Medio Ambiente 2007
	Hydrological control		1 - (water exported/precipitation), average for the 1990-2010 period.	-	De Cáceres et al. 2015	De Cáceres et al. 2015
	Erosion control		Max. erosion - real erosion using RUSLE equation.	tons/ha^{-1}yr	JRC maps, 'meteoland' model	Panagos et al. 2015, Panagos et al. 2014, De Cáceres et al. 2018
Biodiversity	Bird richness		Number of bird species(forest generalists + forest specialists species).	-	2nd Catalan Breeding Bird Atlas	Estrada et al. 2004
Hazard magnitude						
Extrinsic factors	Fire Weather Index (FWI)		Daily values for the fire extinction period (months 6-9).	-	JRC maps	JRC 2017
	Forest continuity		Percentage of forest around 5 km of each pixel of the MCSC.	-	MCSC	MCSC 2009
	Human visitation	Population	Total amount of population in a 25 km buffer from the IFN3 plots.	Inhabitants	European grid (geostat)	Geostat population grid
		Distance to buildings	Distance from IFN3 plots to the nearest cell of the Geostat grid.	Meters	European grid (geostat)	Geostat population grid
		Distance to roads	Distance from IFN3 plots to the nearest communication route.	Meters	Topographic Map (ICGC)	ICGC 2019
Susceptibility						
Intrinsic factors	Struct. charact.	Vertical and horiz. continuity	A coefficient (from 0 to 1) from the forest structures defined in Alvarez et al. 2012a,b.	-	IFN3	Alvarez et al. 2012a, b
		Fuel load	Total amount of shrub biomass in IFN3 plots and fine biomass from trees (leaves and branches with a diameter up to 6.35 mm).	tons/ha	IFN3	Ministerio de Medio Ambiente 2007, Sanchez-Pinillos et al. 2019
	Funct. charact.	Bark thickness	Community Weigthed Means (CWM) of bark thickness calculated using species allometries (IEFC).	mm	IFN3 and IEFC	Ministerio de Medio Ambiente 2007, Gracia et al. 2004
		Flammability	Flammability model that defines 10 models of flammability depending on the percentage of FCC having flammable species.	-	CREAF Flammability model	http://www.creaf.uab.cat/ mmci/descarrega.htm
Extrinsic factors	Extinction capacity	Distance to water bodies	Distance from each IFN3 plot to the nearest water body.	Meters	Firefighter's maps	Firefighter's maps
		Distance to fire stations	Distance from each IFN3 plot to the nearest fire station.	Meters	Firefighter's maps	Firefighter's maps
		Distance to lookout towers	Distance from each IFN3 plot to the nearest lookout tower (towers where government workers watch for fire or smoke).	Meters	Firefighter's maps	Firefighter's maps
Lack of adaptive capacity						
Intrinsic factors	Species regeneration charact.		Community Weigthed Means (CWM) of resprouting capacity + seeding capacity of the species in IFN3 plots.	-	BROT database, literature search	Tavsanoglu and Pausas 2018
Extrinsic factors	Site index		Linear model with growth (in cm^2/year) as response variable. Explanatory variables: 1) Annual solar radiation (10 kJ/m^2 day); 2) Aridity (Annual precip./Annual Potential Evapotransp.); 3) Stoniness and 4) Top index.	cm^2/yr	IFN3 and climatic data	Ministerio de Medio Ambiente 2007

Hydrological control Hydrological control is the capacity of forests to control water flooding, i.e., the amount of water that is not exported due to forests, which depends on vegetation cover and precipitation. The water balance model in De Cáceres et al. (2015) was applied in each forest inventory plot of the third Spanish National Forest Inventory (Ministerio de Medio Ambiente, 2007*b*) to obtain the amount of water exported for the 1990-2010 period ($L/m^2 \cdot yr$). Then, it was divided by precipitation for the same period of data. Finally, hydrological control is defined as 1 - (water exported/precipitation).

Erosion control We measured the erosion avoided by forests, i.e., the amount of soil that is not eroded because of forests. It is the difference between the potential maximum erosion (without vegetation) and the erosion rate with a certain amount of forest. This indicator is based on the Revisited Universal Soil Loss Equation (RUSLE) (Equation A4.1).

$$RUSLE = L \cdot S \cdot K \cdot R \cdot C \tag{A4.1}$$

where L is the slope length, S is the steepness factor, K is the type of soil, R is the rain erodibility and C is the proportion of soil without vegetation. L, S and K are constant and were obtained from the Joint Research Center European database (Panagos et al., 2014, 2015*a*). R and C varied with time. Average values of R for the 1991-2009 period were obtained using the 'meteoland' R package (De Cáceres et al., 2018). C for non-arable lands (i.e., forests) is defined in Panagos et al. (2015*b*) as

$$C = Min(Clanduse) + Range(Clanduse) \cdot (1 - Fcover) \tag{A4.2}$$

where min and max $Clanduse$ for forests are 0.0001 and 0.003, respectively (range is therefore 0.003 - 0.0001 = 0.0029). $Fcover$ is the vegetation cover (range 0 - 1).

Biodiversity - Bird richness

We computed biodiversity from birds. Birds represent the group of vertebrates where we have an in- depth knowledge of their relationships with forest age, structure and composition with global coverage (Drapeau et al., 2000; Gil-Tena et al., 2007; Drever et al., 2008). Furthermore, bird assemblages and trees can be considered as complementary when used as indicators of biodiversity (Kati et al., 2004), thus increasing biodiversity surrogacy (Larsen et al., 2012).

We used the accumulative number of species detected in each 1 km cell of teh second Catalan Breeding Bird Atlas (Estrada et al., 2004) to compute the species richness. Given that the majority of the indicators were assessed in forested areas, we only considered forest bird species. Two groups of bird species were defined according to their degree of specialization in forest habitats: 1) forest specialists, species tightly linked to forested habitats both for feeding and nesting; and 2) forest generalists, species with a preference for forest habitats but can thrive in a wide range of environmental conditions and use a variety of different resources, e.g., feed in set-asides and grassland patches.

Hazard Magnitude

The magnitude of the hazard and its probability distribution can be quantified using integrative hazard indices that incorporate the most relevant variables of the hazard (Lecina-Diaz et al., in press). In the case of wildfires, we used the Fire Weather Index (FWI) (Van Wagner, 1987). We incorporate effects of extrinsic factors (i.e., factors that operate at broad scales such as landscape scale) (Lecina-Diaz et al., in press): forest continuity and human visitation. Both factors were standardized by centering them to one, so that Hazard Magnitude has the same scale as the FWI.

Extrinsic factors

Fire Weather Index We used The Fire Weather Index (Van Wagner, 1987), which is commonly used as a general index of fire danger. 'The FWI System is based on the moisture content and the effect of wind of three classes of forest fuels on fire behavior. It consists of six components: three fuel moisture codes (Fire Fuel Moisture Code, Duff Moisture Code, Drought Code), and three fire behavior indexes representing rate of spread (Initial Spread Index), fuel consumption (Buildup Index), and fire intensity (Fire Weather Index). The FWI System outputs are determined from daily noon weather observations: temperature, relative humidity, wind speed, and 24-hour rainfall' (Van Wagner, 1987). We used daily values of FWI from the Join Research Centre for the fire extinction period (months 6-9) for years 2000-2018 at a spatial resolution of 0.28 degrees (Joint Research Centre, 2017) and generated repeated random distributions using Monte Carlo simulations.

Forest Continuity Forest continuity at landscape scale can increase the magnitude of wildfires, whereas heterogeneity of landscapes can give opportunities for reduction of wildfire magnitude (e.g., landscapes with less fuel available such as croplands can prevent the spreading of fire and, as a consequence, reduce wildfire magnitude). Thus, we calculated the proportion of forest around 5 km of each pixel (300 m) of the Catalan Land Cover Map (CREAF, 2009) as a measure of forest continuity. The range of values is 0 to 100 (as percentage).

Human visitation The presence of people in forests or close to them increases the probability of fire ignicion and, as a consequence, its hazard magnitude. Therefore, we included three variables related with human visitation.

Population We used the European Geostat 1 km^2 population grid (European Comission, 2012). We defined a buffer of 25 km from each IFN3 plot (Ministerio de Medio Ambiente, 2007*b*) and summed the value of population of each grid that overlaps with the buffer.

Population distance We measured the distance from each IFN3 plot (Ministerio de Medio Ambiente, 2007*b*) to the nearest cell of the European Geostat 1 km^2 population grid (European Comission, 2012).

Communication route distance We measured the distance from each IFN3 plot (Ministerio de Medio Ambiente, 2007*b*) to the nearest communication route from the Catalan Topographic Map (1:25000) (Catalan Cartograhical and Geological Institute, 2019). Communication routes include highways of preferential paths and rails, other paved roads and not paved roads (e.g., paths).

Susceptibility

The characteristics of the forest that modulate the immediate effects of the hazard define susceptibility (Lecina-Diaz et al., in press). It includes intrinsic characteristics (i.e., internal characteristics of the forest) and extrinsic factors (i.e., factors operating at broader scales) (Lecina-Diaz et al., in press).

Intrinsic factors

Structural characteristics

Table A4.2: Coefficients of susceptibility for forest structures.

Forest structures	Coefficient of susceptibility
1	0.59
2	0.21
3	0.45
4	1.00
5	0.54
6	0.61
7	0.54
8	0.81
9	0.64
10	0.54
11	0.74
12	0.25
13	0.72
14	0.86
15	0.54
16	0.81
17	0.81
18	0.54
19	0.54
20	0.81

- Vertical and horizontal continuity. We classified IFN3 plots (Ministerio de Medio Ambiente, 2007*b*) into forest structures based on vertical and horizontal continuity following Alvarez et al. (2012*b*,*a*). For each forest structure, we established a coefficient (from 0 to 1) based on the percentage of plots of this forest structure that can be burned with high intensity, i.e., including passive and active crown fires, which can spread with high intensity and speed (Table A4.2).
- Fuel load. We calculate fuel load as the sum of total aerial shrub biomass (from IFN3 plots) and tree fine biomass, i.e. leaves and branches with a diameter up to 6.35 mm (Sánchez-Pinillos et al., 2019*a*).

Functional characteristics Functional characteristics of trees that influence their predisposition to be affected by wildfires includes bark thickness and flammability of species.

- Bark Thickness. Species with thick bark are more resistant to fire than species with thin bark (Pausas, 2015). We applied the specific bark thickness allometries from the IEFC database (Gracia et al., 2004) to the mean DBH of each species in each IFN3 plot (Equation A4.3). In cases where the species allometry was missing, we applied the genus allometry or used the most similar species in terms of bark thickness. Then, we weighted the specific values by the percentage of basal area of each species in each IFN3 plot (i.e., applying Community Weighted Means (CWM) using basal area) to obtain a value of bark thickness for each plot. In cases without information of the species, the species was not considered in the plot when the percentage of basal area in the plot was < 20%. When basal area of the species lacking information was > 20%, the whole plot was deleted.

$$BT = a \cdot DBH^b \tag{A4.3}$$

where BT is the specific bark thickness (in mm), DBH is the diameter at breast height (in cm) and a and b are the parameters of the allometry that depend on the species (Gracia et al., 2004). As bark thickness reduces susceptibility to wildfires, we computed 1-barkthickness, so that thin bark increases susceptibility.

- Flammability. The flammability models were defined using the particular flammability of the most abundant shrub and tree species and its degree of abundance (CREAF, 2003). They comprise 10 models of flammability depending on the percentage of fraction of vegetation cover having flammable species:
  - Model 0: 0 to 9 % of very flammable species.
  - Model 1: 10 to 19 % of very flammable species.
  - Model 2: 20 to 29 % of very flammable species.
  - Model 3: 30 to 39 % of very flammable species.
  - Model 4: 40 to 49 % of very flammable species.
  - Model 5: 50 to 59 % of very flammable species.
  - Model 6: 60 to 69 % of very flammable species.
  - Model 7: 70 to 79 % of very flammable species.
  - Model 8: 80 to 89 % of very flammable species.
  - Model 9: 90 to 99 % of very flammable species.
  - Model 10: more than 100% of very flammable species.

Extrinsic factors

Extinction capacity The capacity of firefigthers to extinguish fire in IFN3 plots was defined by measuring the distance from each IFN3 plot to the nearest water body, firefigther's station and lookout tower.

Water bodies distance We measured the distance (in m) from each IFN3 plot to the nearest water body from the oficial Firefigther's Map, which includes all inland waterbodies used to extinguish fire (e.g., pools, ponds or dams). We add the sea as a waterbody than can be used in extinction.

Fire stations distance We measured the distance (in m) from each IFN3 plot to the nearest fire station from the oficial Firefigther's Map.

Lookout towers distance To detect a wildfire as soon as possible, lookout towers are spread throughout the territory from where government workers can detect fire or smoke. We measured the distance (in m) from each IFN3 plot to the nearest fire station from the oficial Firefigther's Map.

Lack of Adaptive Capacity

The capacity of forests to recover depends on intrinsic characteristics of the forest (regeneration capacity) as well as on extrinsic factors (the quality of the site). We aggregated regeneration capacity and site index, which both constitute adaptive capacity. Then, we calculated lack of adaptive capacity as 1 - adaptive capacity.

Intrinsic factors

Species regeneration characteristics The capacity of forests to recover after a wildfire depends on their regeneration capacity, which is based in resprouting capacity and seeding capacity. These characteristics depend on the species. Resprouting capacity is defined as the average proportion of adult plants that resprout as percentage, comprising 6 categories (0; 0.2; 0.4; 0.6; 0.8 and 1). We used the variable *resprouting capacity* from BROT database (Tavşanoğlu and Pausas, 2018) and conducted a literature search to complete the species without information (Table A4.3). Seeding capacity is the number of postfire seedlings per ha, grouped in 6 categories from 0 to 1 (Table A4.3) depending on the values of seedlings/ha obtained from a literature search (Table A4.3).

The overall capacity of species to regenerate is the sum of resprouting capacity and seeding capacity, with a maximum value of 1 (corresponding to 100%). For instance, if a species has a resprouting capacity of 1, overall regeneration capacity of the species will be 1 (i.e., no matter what are the values of seeding capacity). As with bark thickness, we weighted the specific values by the percentage of basal area of each species in each

Table A4.3: Categorization of seeding capacity.

Seeding capacity	Seedlings/ha
0	< 100
0.2	100 - 500
0.4	500 - 1500
0.6	1500 - 3000
0.8	3000 - 5000
1	> 5000

IFN3 plot (i.e., applying Community Weighted Means (CWM) using basal area), being an estimate of the regeneration capacity at the plot level. In cases without information of the species, the species was not considered in the plot when the percentage of basal area in the plot was < 20%. When basal area of the species lacking information was > 20%, the whole plot was deleted.

Extrinsic factors

Site index To characterize each site where plots are located, we developed a linear model with growth (in cm^2/yr) as response variable, previously transformed using square root to meet the assumptions of normality of residuals. The explanatory variables were: 1) (Annual solar radiation)^2 (10 kJ/m^2 day); 2) Aridity (Annual precipitation/Annual Potential Evapotranspiration); 3) Stoniness and 4) Top index = ln(a/tan(beta)) where a is the area of the hillslope per unit contour length that drains through any point and beta is the local surface topographic slope (delta vertical) / (delta horizontal). We used the r.topidx function from QGIS (`https://grass.osgeo.org/grass78/manuals/r.topidx.html`) using elevation raster (30 m) as input.

Weighting and aggregating the indicators

We applied three of the most used weighting methods: 1) equal weights; 2) statistical weights; and 3) expert weights (Table A4.4). Therefore, we conducted pearson's correlation analysis between hazard magnitude, susceptibility and lack of adaptive capacity for the three weighting methods (section 5.1).

Table A4.4: Weights (equal, PCA and expert weights) assigned to the individual indicators in each component (hazard magnitude, susceptibility and lack of adaptive capacity).

Component	Indicator	Equal weights	PCA weights	Expert weights
Hazard magnitude	Forest continuity	0.25	0.27	0.50
	Population	0.25	0.26	0.10
	Distance to population	0.25	0.20	0.15
	Distance to communication routes	0.25	0.27	0.25
Susceptibility	Forest structure	0.14	0.12	0.25
	Fuel load	0.14	0.15	0.20
	(1 - Bark thickness)	0.14	0.13	0.10
	Flammability	0.14	0.17	0.15
	Distance to water	0.14	0.13	0.10
	Distance to fire stations	0.14	0.16	0.10
	Distance to lookout towers	0.14	0.13	0.10
Lack of adaptive capacity	Regeneration capacity	0.50	0.50	0.60
	Site index	0.50	0.50	0.40

Equal weights

Equal weights are used when there is no information on statistical or ecological importance of the indicators and all variables are given the same weight (i.e., all variables have the same importance) (Nardo et al., 2008). In this case, we applied the same weight to all indicators in a component. For example, hazard magnitude has four indicators, thus we applied a weight of 0.25 to each indicator of hazard magnitude (and the same procedure was applied for the rest of indicators)(Table A4.4).

Statistical weights

Statistical techniques such are used to capture the information common to individual indicators. We conducted a Principal Component Analysis (PCA) for each component. The specific weights (w_i) for each indicator were obtained from the matrix of factor loadings following

$$w_i = r_j \cdot (l_{ij}^2 / E_j) \tag{A4.4}$$

where r_j is the proportion of the explained variance of the factor j in the component, l_{ij}^2 the factor loading of the i^{th} indicator on factor j and E_j the variance explained by the factor j (Nardo et al., 2008; Gan et al., 2017). Results of the weights for PCA method are given in Table A4.4.

Expert weights

Expert weights are used to decide the importance of the indicators based on expert criteria. In this case, each co-autor of the manuscript individually proposed a weight for each indicator, and the averaged value was used to determine expert weights for each indicator (Table A4.4).

Correlation analysis and maps of the risk components for the three weighting methods

We conducted correlation analysis between the components that have weights: 1) hazard magnitude (modifiers of the FWI, i.e., the combination of forest continuity and human visitation)(Table A4.5); 2) susceptibility (Table A4.6) and 3) lack of adaptive capacity (Table A4.7). We also mapped the results of the three weighting methods for hazard magnitude (Fig. A4.1, in this case considering the median and percentile 90th of the distribution of FWI), susceptibility (Fig. A4.2) and lack of adaptive capacity (Fig. A4.3).

Table A4.5: Pearson's correlation analysis of hazard magnitude for the three weighting methods.

	Equal	PCA
PCA	0.99	
Expert	0.83	0.85

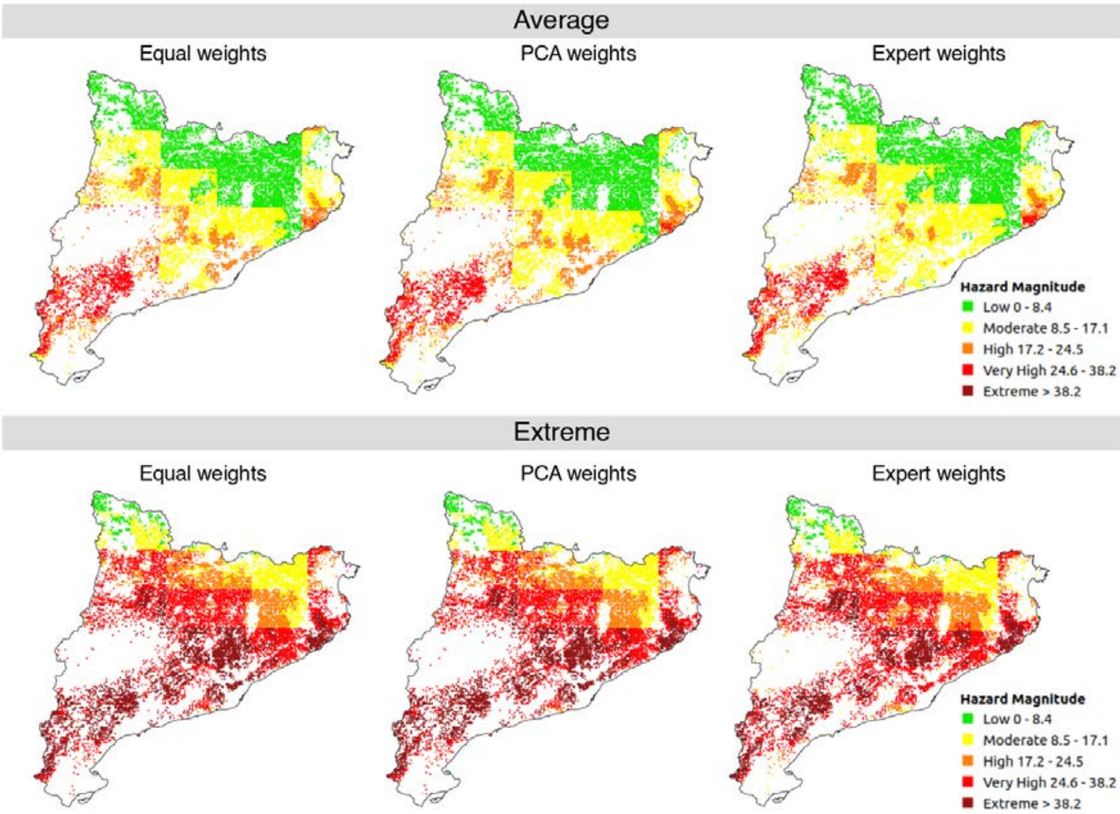


FIGURE A4.1: Hazard magnitude (average and extreme) for the three weighting methods (equal, PCA and expert weights).

Table A4.6: Pearson's correlation analysis of susceptibility for the three weighting methods.

	Equal	PCA
PCA	0.99	
Expert	0.93	0.91

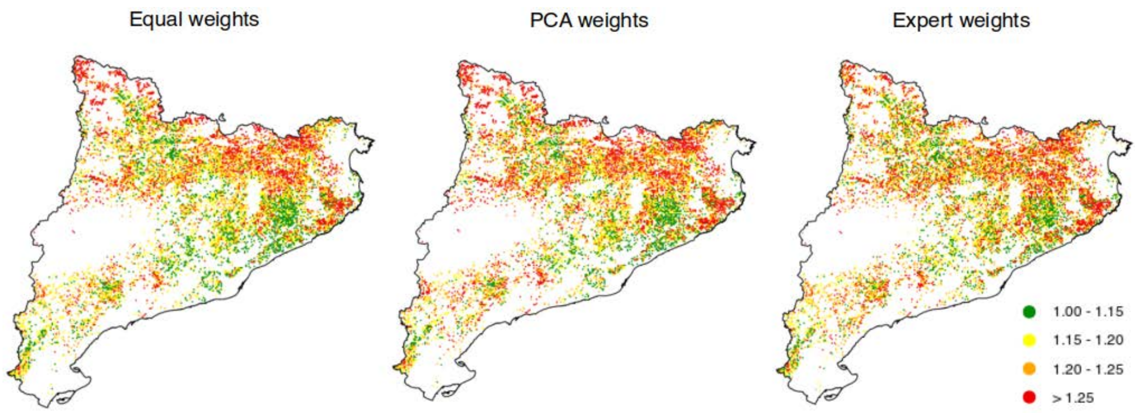


FIGURE A4.2: Susceptibility for the three weighting methods (equal, PCA and expert weights).

Table A4.7: Pearson's correlation analysis of lack of adaptive capacity for the three weighting methods (equal, PCA and expert weights). Note that weights for equal and pca methods are the same.

	Equal	PCA
PCA	1	
Expert	0.99	0.99

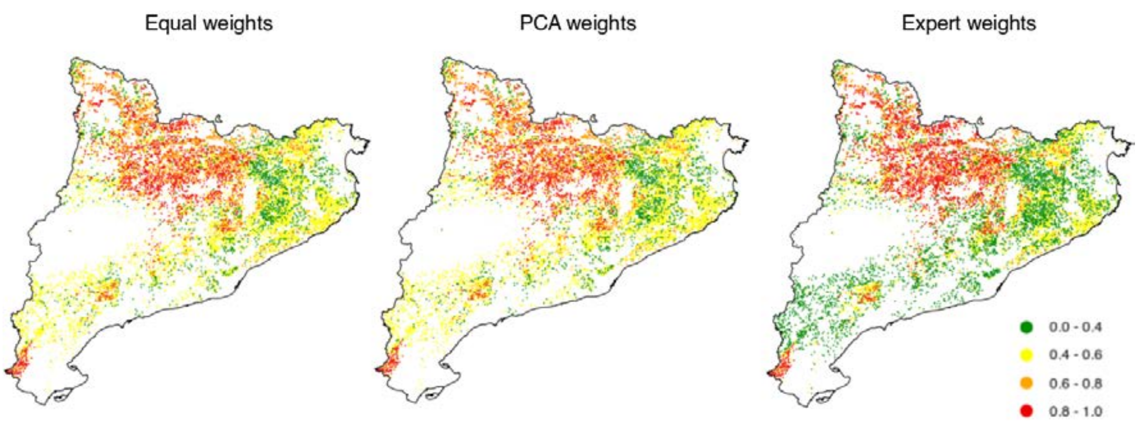


FIGURE A4.3: Lack of adaptive capacity for the three weighting methods (equal, PCA and expert weights).

Aggregating the components

We combined the components by following the Eq. 4.1 (main chapter). As the relationship between hazard magnitude and immediate loss of values that define susceptibility is non-linear (Fig. A4.4), we used data of the FWI from the literature that correspond to complete immediate losses of values to adjust susceptibility values. In Figure A4.4, s_2 is the maximum susceptibility value, which corresponds to the FWI (or hazard magnitude) where immediate loss of values is 100 (100%) following

$$Immediate\ loss = HM^S \tag{A4.5}$$

where HM is the FWI and S is susceptibility. Maximum immediate loss (100) corresponds to FWI of 24.6 (Palheiro et al., 2006; Tedim et al., 2018). Then, maximum susceptibility is 1.44. Minimum susceptibility is 1 and, therefore, susceptibility ranges from 1 to 1.44. Thus, we applied statistical weights to calculate susceptibility, and reescaled the results to have a range from 1 to 1.44. Afterwards, we raised hazard magnitude to susceptibility and truncate the results so that the maximum was 100 (i.e., 100% of values were lost). We scaled HM^S so that values were between 0 and 1 and multiplied them by lack of adaptive capacity and exposure, producing one distribution of values at risk in each plot for the PCA weighting method: carbon sink, bird richness, hydrological control and erosion control at risk from wildfires.

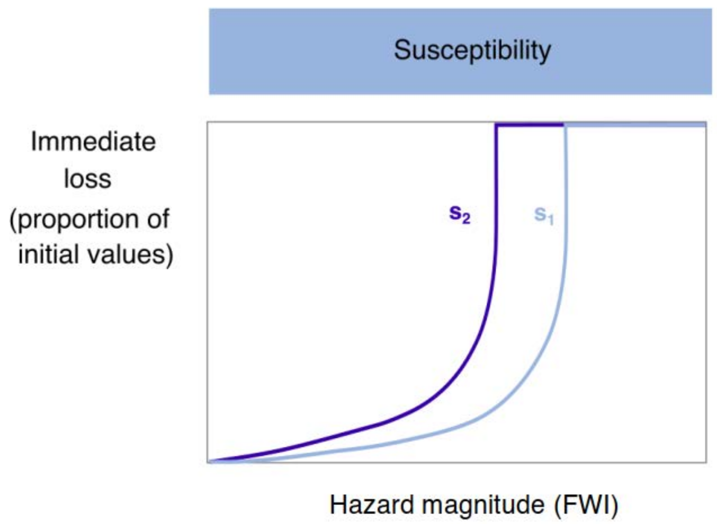


FIGURE A4.4: Relationship between hazard magnitude and immediate loss of values that defines susceptibility.

To obtain average and extreme values at risk in each plot, we extracted the median and percentile 90th values of each distribution in each plot. We obtained a map of the median and a map of the percentile 90th for each value at risk. Then, we mapped the relative changes in risk associated to extreme vs. average hazard conditions using the log-ratio of extreme to average conditions (i.e., log((percentile 90th at risk)/(median at risk))).

Supplementary results

Correlation analysis between the components of risk

Table A4.8: Pearson's correlation between the components of risk for Exposed values (*E*), Hazard Magnitude (*HM*), Susceptibility (*S*) and Lack of Adaptive Capacity (*LAC*) (* p < 0.05, ** p < 0.01, *** p < 0.001).

	E - bird richness	*E* - hydr. control	*E* - erosion control	*HM* - average	*HM* - extreme	*S*	*LAC*
E - carbon sink	0.11 ***	0.19 ***	0.17 ***	-0.26 ***	-0.21 ***	0.06 ***	-0.14 ***
E - bird richness		0.04 ***	-0.05 ***	-0.08 ***	0.08 ***	-0.12 ***	-0.02 *
E - hydr. control			-0.21 ***	0.39 ***	0.39 ***	0.08 ***	-0.23 ***
E - erosion control				-0.39 ***	-0.49 ***	0.13 ***	0.05 ***
HM - average					0.91 ***	-0.14 ***	-0.16 ***
HM - extreme						-0.22 ***	-0.17 ***
S							0.03 ***

Sensitivity analysis

To determine the effect of the components of risk to average and extreme ES at risk, we conducted sensitivity analysis applying the 'tgp' R package (Gramacy, 2016) to a random sample of 500 plots. Results are shown in Figures A4.5 - A4.12.

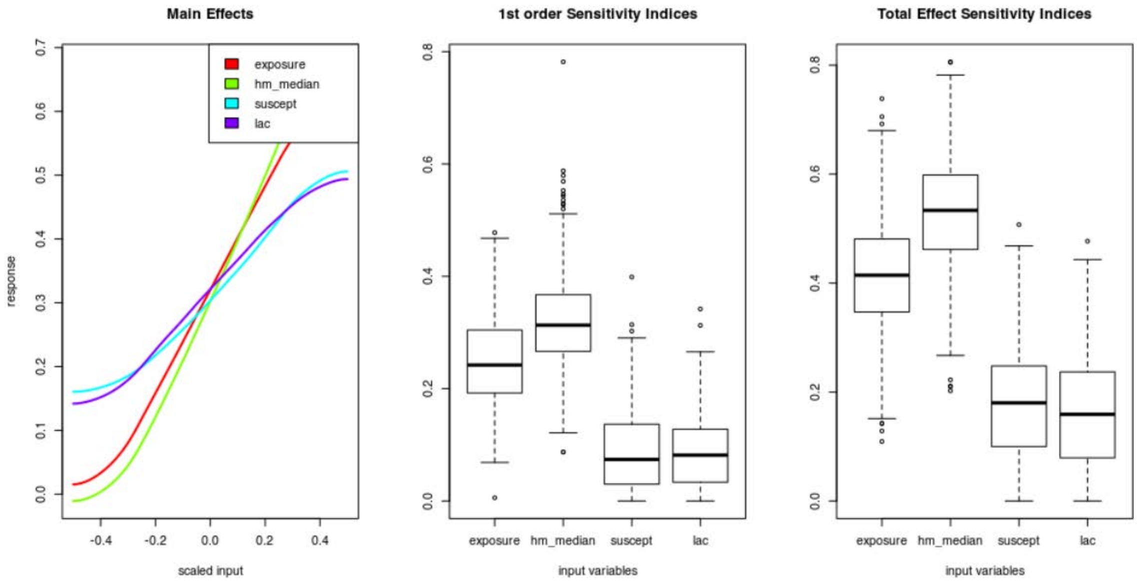


FIGURE A4.5: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, average hazard magnitude (hm_median), susceptibility (suscept) and lack of adaptive capacity (lac)) to carbon sink at risk under average conditions.

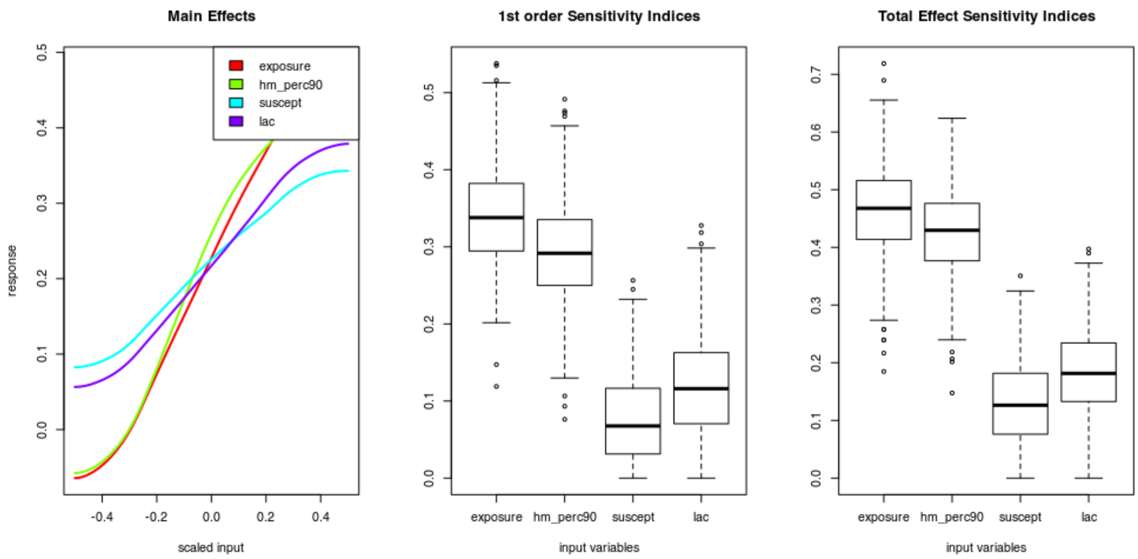


FIGURE A4.6: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, extreme hazard magnitude (hm_perc90), susceptibility (suscept) and lack of adaptive capacity (lac)) to carbon sink at risk under extreme conditions.

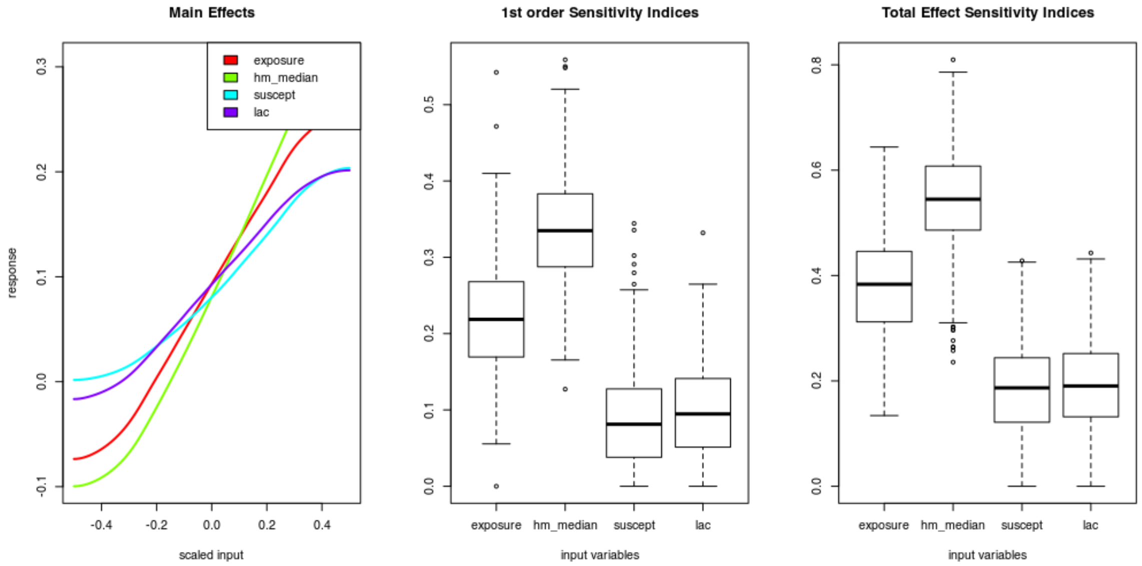


FIGURE A4.7: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, average hazard magnitude (hm_median), susceptibility (suscept) and lack of adaptive capacity (lac)) to bird richness at risk under average conditions.

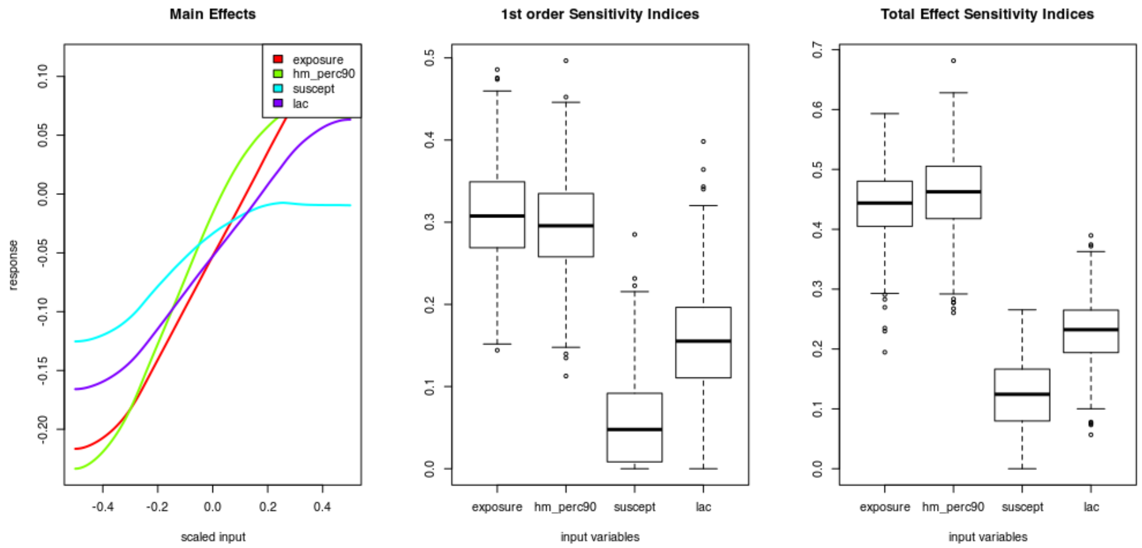


FIGURE A4.8: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, extreme hazard magnitude (hm_perc90), susceptibility (suscept) and lack of adaptive capacity (lac)) to bird richness at risk under extreme conditions.

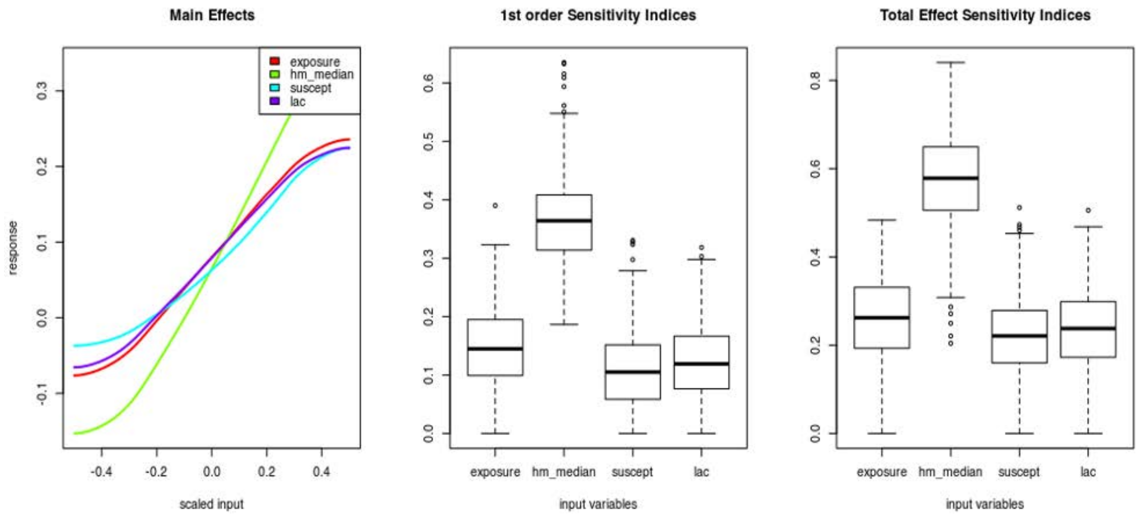


FIGURE A4.9: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, average hazard magnitude (hm_median), susceptibility (suscept) and lack of adaptive capacity (lac)) to hydrological control at risk under average conditions.

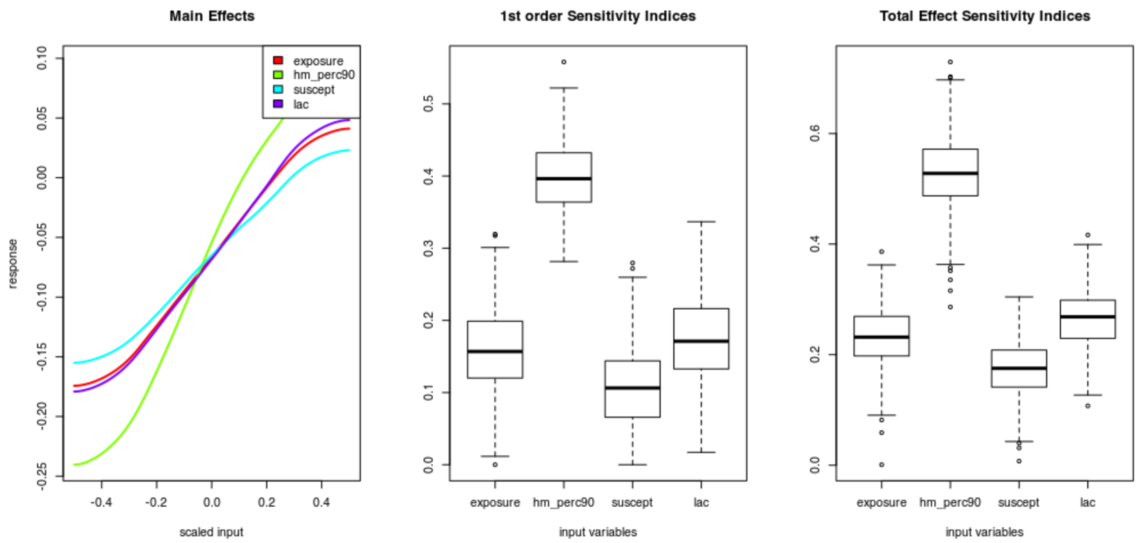


FIGURE A4.10: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, extreme hazard magnitude (hm_perc90), susceptibility (suscept) and lack of adaptive capacity (lac)) to hydrological control at risk under extreme conditions.

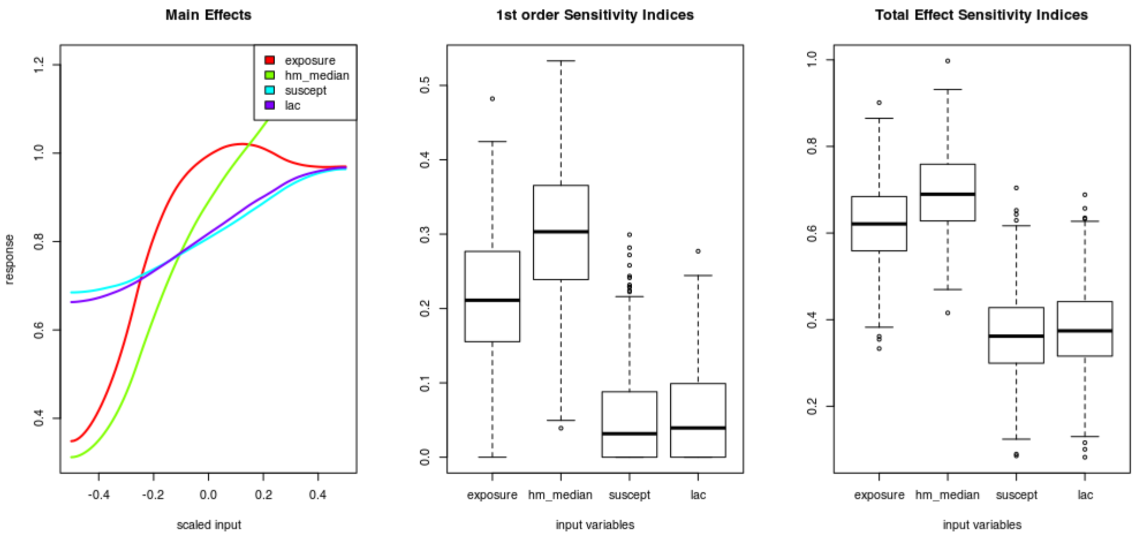


FIGURE A4.11: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, average hazard magnitude (hm_median), susceptibility (suscept) and lack of adaptive capacity (lac)) to erosion control at risk under average conditions.

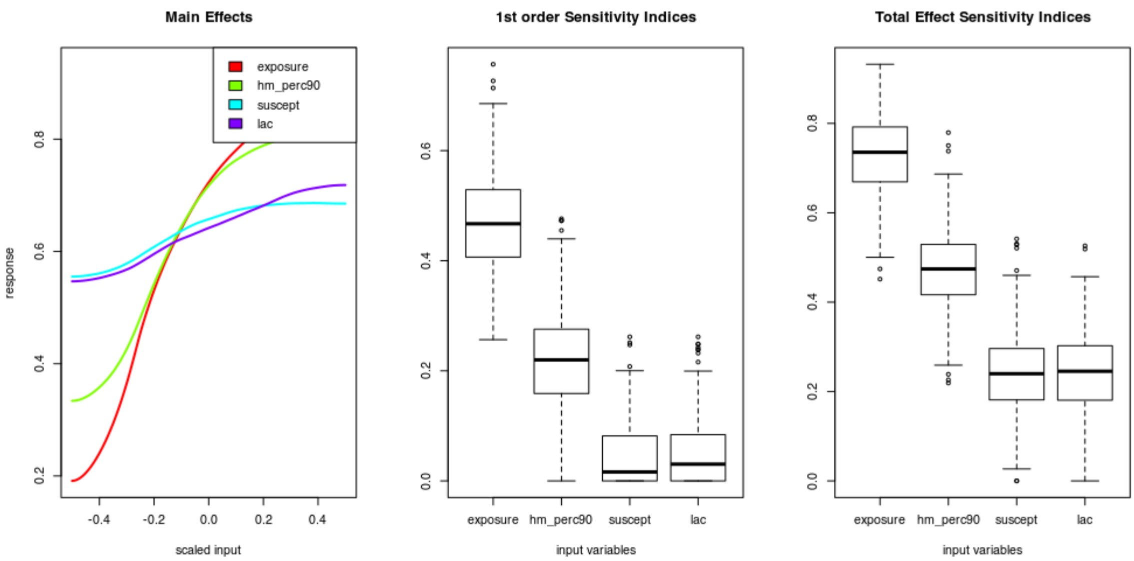


FIGURE A4.12: Main effects (first pannel), first order sensitivity indices (second pannel) and total effect sensitivity indices (third pannel) of the components of risk (exposure, extreme hazard magnitude (hm_perc90), susceptibility (suscept) and lack of adaptive capacity (lac)) to erosion control at risk under extreme conditions.

Regression trees

Table A4.9: Dominant species considered in the forest functional types (broadleaf evergreen, broadleaf deciduous, Mediterranean conifer and non-Mediterranean conifer).

Forest functional type	Dominant species
Broadleaf evergreen	*Quercus ilex* *Quercus suber* *Olea europaea* *Arbutus unedo*
Broadleaf deciduous	*Quercus humilis/cerrioides* *Fagus sylvatica* *Quercus petraea* *Quercus faginea* *Castanea sativa* *Fraxinus excelsior* *Populus canadensis* *Betula pendula* *Populus nigra* *Populus tremula* *Alnus glutinosa* *Acer campestre* *Platanus hispanica* *Corylus avellana* *Fraxinus angustifolia* *Quercus robur* *Robinia pseudacacia*
Mediterranean conifer	*Pinus halepensis* *Pinus pinea* *Pinus pinaster*
Non-Mediterranean conifer	*Pinus sylvestris* *Pinus nigra* *Pinus uncinata* *Abies alba* *Pseudotsuga menziesii* *Pinus radiata*

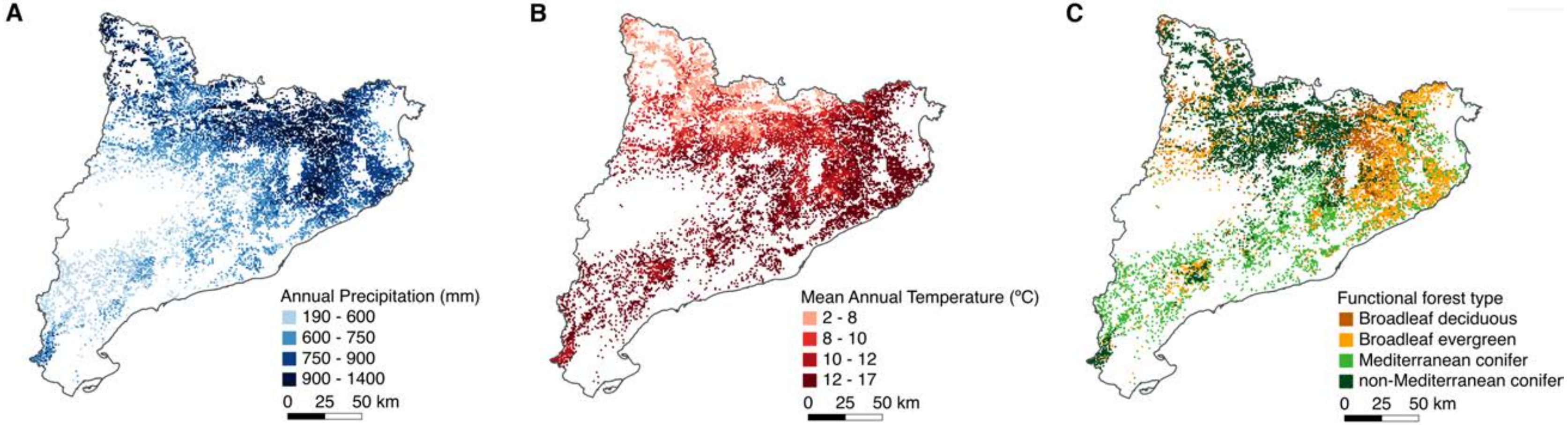


FIGURE A4.13: Maps of the variables used in the regression trees: (A) Annual Precipitation (mm), (B) Mean Annual Temperature (ºC) and (C) Functional forest types.

Table A4.10: Parameters of the regression trees for ES at risk (carbon sink, bird richness, hydrological control and erosion control) under average and extreme conditions. Complexity parameter (cp), R squared and Pearson's correlation tests between train and test data (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$).

Variable	Conditions	cp	R squared	test-train data correlation
Carbon sink	Average	0.017	0.12	0.34 ***
	Extreme	0.011	0.10	0.34 ***
Bird richness	Average	0.027	0.32	0.55 ***
	Extreme	0.090	0.25	0.49 ***
Hydrological control	Average	0.037	0.33	0.56 ***
	Extreme	0.038	0.36	0.62 ***
Erosion control	Average	0.013	0.04	0.17 ***
	Extreme	0.019	0.11	0.31 ***

Correlation analysis between the log-ratio(extreme/average) and the components of risk

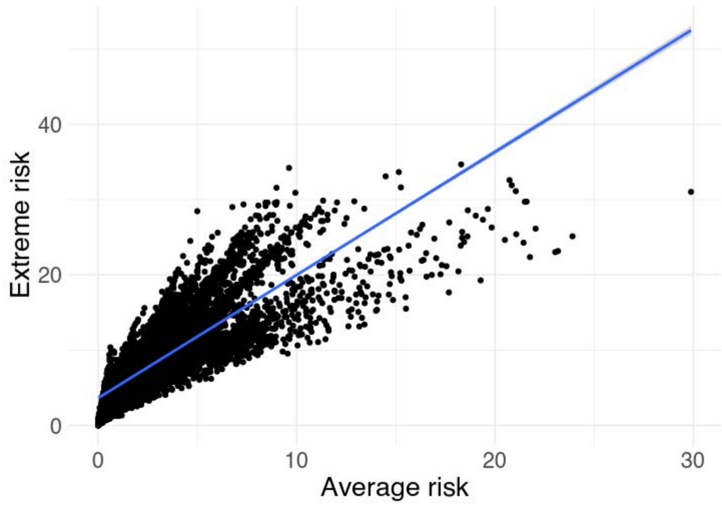


FIGURE A4.14: Scatter plot of extreme vs average risk conditions.

Table A4.11: Pearson's correlation between the log-ratio of extreme vs. average hazard conditions and the components of risk. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

	Correlation
Log(extreme/average) - Exposed values (carbon sink)	0.27 ***
Log(extreme/average) - Exposed values (bird richness)	0.03 ***
Log(extreme/average) - Exposed values (hydrological control)	- 0.30 ***
Log(extreme/average) - Exposed values (erosion control)	0.46 ***
Log(extreme/average) - Susceptibility	0.24 ***
Log(extreme/average) - Lack of adaptive capacity	0.08 ***

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