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Transnational Forest Restoration Strategy Draft

Version 2

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SUMMARY

The transnational strategy for restoring degraded forests of central Europe is a road map for restoring forests of central Europe affected by one or a combination of six major drivers, such as drought, fire, wind, spruce bark beetle, ash dieback, and improper management of riparian forests. Although the RE-ENFORCE project focuses on the effects of these drivers within Central Europe, we realize that the drivers are pan-European and therefore we will present the strategy with a pan-European focus.

The draft strategy of forest restoration includes:

PART 1

- Basic information on forest degradation in Europe, historical context, and why restoration is needed
- Review of forest degradation due to each of the six drivers

In addition, the draft strategy will be further developed by including:

PART 2

- Policy and stakeholder perspectives on forest restoration
- Spatially explicit mapping of forest degradation under current and future climate
- Spatially explicit mapping of the development of ecosystem services
- Spatially explicit mapping of priority areas for restoring degraded forests in Europe
- Perspectives and lessons learned for six pilot actions of the RE-ENFORCE project
- Internal and external review of the draft strategy

The final version of the Transnational strategy for restoring degraded forests of Central Europe will also include **PART 3**, extending to cover a pan-European scale. This part will capitalize on **PART 1** and **PART 2** to formulate a tailored strategy to restore the degraded forests of Europe. This final strategy will incorporate aspects of lessons learned and review of past degradation covered in **PART 1**, and maps of forest degradation, policy issues, and review of forest degradation practices in **PART 2**. The final strategy will be a long-term, chosen plan of action where climate change adaptation, lessons learned, state-of-the-art modelling, and stakeholders' perspectives will be combined.

1. Chapter: Basic information about forest degradation in Europe

This chapter focuses on the historical development and current drivers of forest degradation in Europe, providing the background needed to understand why restoration has become essential.

1.1. Drought

Drought is a major driver of forest degradation in Central Europe (Brun et al., 2020). However, there is still no universally accepted definition or method for quantifying drought, as these depend on the specific processes affected by water availability, detailed knowledge of climatological baselines, and site-specific water balance parameters—many of which are either poorly documented or inherently uncertain. It is therefore important not to focus solely on precipitation deficits while overlooking for example the significant roles of evaporation and transpiration, which jointly determine actual water availability (Lloyd-Hughes, 2014). It is therefore advisable to distinguish between climatic and soil drought, both of which significantly alter the fitness of forest ecosystems.

In general terms, drought is defined as a deficit in water availability relative to long-term average conditions across both spatial and temporal dimensions (Lloyd-Hughes, 2014). Clark et al. (2016) offer a similar perspective, describing drought as a climatic anomaly in a specific region that results in moisture limitations due to low precipitation, elevated temperatures, or a combination of both.

The terminology used to describe drought also varies depending on disciplinary context and the corresponding management objectives. While broad conceptual understandings of drought may align across fields, its numerical representation is considerably more complex. As a result of the diverse impacts and perspectives on drought, over 100 indices have been developed for drought monitoring (Lloyd-Hughes, 2014). These indices differ in focus, aiming to assess drought conditions within meteorological, hydrological, agricultural, or socio-economic domains. The wide range of disciplinary perspectives underscores the multifaceted nature of drought and highlights the difficulty of capturing its full severity through a single metric. Each index reflects specific dimensions of drought, and their relevance is often dependent on regional characteristics and sector-specific needs (Lloyd-Hughes, 2014).

Due to ongoing climate change, the frequency, severity, and duration of drought events are expected to increase significantly in many regions, including Europe (Walker and Van Loon, 2023). When physically accessible soil water is insufficient to meet the physiological demands of trees, such as transpiration for cooling or water uptake for photosynthesis, these processes become impaired, leading to stress and potential damage (Babst et al., 2019; Lloyd-Hughes, 2014).

In addition to climatic inputs such as precipitation and surface or groundwater inflow, human water management practices also play a crucial role in shaping drought conditions and their ecological impacts. Mathematical definitions and quantifications of drought must therefore consider not only water sources, supply, and storage, but also demand and management, framed within clearly defined spatial, temporal, and process-based parameters (Lloyd-Hughes, 2014).

Forest ecosystems are highly sensitive to drought, which can severely impair their ability to provide essential ecosystem services (Raheem et al., 2019). Drought can alter species distributions and biodiversity, influence wildfire regimes, increase stand-level mortality, and alter on both net primary production and overall forest growth (Clark et al., 2016).

While the effects of drought on individual trees are relatively well understood, a significant knowledge gap remains regarding drought responses at the forest stand level. These responses are difficult to predict due to uncertainties in climate projections and the complex interactions between stand characteristics, such as structure and species composition, as well as anthropogenic influences and silvicultural interventions (Clark et al., 2016; Holtmann et al., 2024; Senf et al., 2020).

Drought has increasingly emerged as a major driver of forest degradation in Europe, particularly in recent decades (Allen et al., 2010; Popa et al., 2024). Many countries in Central Europe are now experiencing significantly drier climatic conditions compared to historical period (Ionita & Nagavciuc, 2021). For example, the European heatwave of 2003 caused widespread physiological stress and mortality in various tree species across the continent (Brun et al., 2020). Subsequent drought years, such as 2015 and especially the extended period from 2018 to 2019, were characterized by exceptional spatial extent and severity across Central Europe (Ionita & Nagavciuc, 2021). The extreme drought of 2018 led to large-scale early leaf wilting and tree dieback, especially in central and eastern Germany and the Czech Republic (Brun et al., 2020), and had severe impacts on forest ecosystems in Austria and Switzerland as well (Schuldt et al., 2020).

Given the direct negative impacts of drought on forest ecosystems, as well as secondary effects, such as increased vulnerability to wildfires, pests, and diseases, and the rising likelihood of future drought events, active forest management is crucial. Adaptive strategies aimed at modifying species composition and stand structure are necessary to enhance forest resilience, mitigate adverse effects, and ensure the continued provision of essential ecosystem services (Bolte et al., 2009; Lindner et al., 2014; Seidl et al., 2017). This is particularly important due to the slow pace of forest regeneration and growth, compounded by the rapid mortality of mature trees, which demands urgent adaptation measures (Allen et al., 2010).

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1.2. Fire

Fire is undoubtedly one of the major forest disturbances. In Europe for example, fire is the second most important forest disturbance, responsible for 24% of the total timber volume damage during 1950-2019 (Patacca et al. 2023). The driver will be directly affected by climate change (Seidl et al. 2017), with pronounced effects such as prolonged fire seasons, increasing fire severity, and extreme fire behaviour (Westerling et al. 2006; Duane et al. 2021). Already a decade ago, it was postulated that adapting to coexistence with fire is probably one of the most reasonable strategies for societies and ecosystems worldwide (Stephens et al. 2013; Doerr & Santín 2016).

Central Europe is classified as a non-fire-prone region (Galizia et al. 2022), where – mainly due to the deciduous-dominated forests and generally less flammable vegetation – fire has long been considered a rather uncommon phenomenon in natural ecosystem dynamics (Ellenberg 1996; Leuschner and Ellenberg 2017). However, large parts of the temperate zone in Europe are now covered by anthropogenic conifer monocultures (Timbal et al. 2005; Brus et al. 2012), resulting from over 200 years of timber-oriented forest management (Lowood 1990). Further, the amount of empirical data demonstrating the crucial role of fire in shaping long-term forest dynamics in Central Europe has increased considerably in recent decades (Rösch

2000; Niklasson et al. 2010; Adámek et al. 2015; Feurdean et al. 2020; Spînu et al. 2020; Manton et al. 2022; Zin et al. 2022). In addition, increasing trend in fire disturbance has been recorded in European forests between 1950 and 2019, with peaks of strong individual disturbance years from the 1990s onward resulting from the extreme regional fire years (Patacca et al. 2023), such as, e.g., 2018 (Fernandez-Áñez et al. 2021) - which also applies to Central Europe, including countries such as the Netherlands, Germany, Latvia, UK, etc. (Fernandez-Áñez et al. 2021; San-Miguel-Ayanz et al. 2024; Stoof et al. 2024).

Temperate Central Europe nowadays definitely cannot be considered non-fire-prone anymore (Berčák et al. 2024; Stoof et al. 2024). Climate change greatly increases fuel availability (Duane et al. 2021) and fire weather (Schelhaas et al. 2010), which, along with the high population density and importance of human ignition in the region (Ganteaume et al. 2013; San-Miguel-Ayanz et al. 2024) supports predictions of the increasing fire risk in the future (Patacca et al. 2023) that will likely require post-fire forest restoration. Increased fire activity and its importance for local resource management and policy is being slowly recognized in the region (Berčák et al. 2023, 2024; Stoof et al. 2024), including some suggestions for forest restoration, e.g., switch from conifers to broadleaves (Schelhaas et al. 2010). However, transnational forest fire risk management and post-disturbance strategies and guidelines are still missing in Central Europe.

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1.3. Wind

Wind is a significant disturbance agent in forests and plays a crucial role in the dynamics of forest ecosystems, particularly in temperate regions (Gardiner et al. 2013). However, the frequency of extreme windstorms in temperate forests across Europe has notably increased since 1950 (Gregow et al. 2017; Schelhaas et al. 2003). This can be due to the fact that the first part of the 20th century experienced fewer storms compared to prior centuries. Additionally, forest sensitivity to wind has increased over time due to afforestation, new silvicultural practices such as industrial plantation forestry, and changing economic and environmental conditions (Karjalainen and European Forest Institute 1999; Loustau 2010).

Windthrows can cause substantial damage to forest stands, resulting in significant timber loss, as evidenced by past events in northern and central Europe. For example, the windstorms Vivian and Wiebke in 1990 blew down up to 120 M m^3 of timber, Lothar and Martin in 1999 resulted in more than 240 M m^3 of damaged timber, Gudrun in 2005 caused 77.5 M m^3 of damaged timber, Kyrill in 2007 led to 54 M m^3 of damaged timber, and Klaus in 2009 resulted in 44.6 M m^3 of damaged timber (Gardiner et al. 2010; 2013). Lastly, the windstorm Vaia (2018) caused more than 16 M m^3 of timber damage in the Italian Alps (Stefani et al. 2021) *(Figure 1)*. Such high-severity disturbances can have significant economic, environmental, and social implications. Specifically, in managed forests, stand-replacing windstorms can affect management goals and severely impact wood production and timber markets (Gardiner et al. 2013).

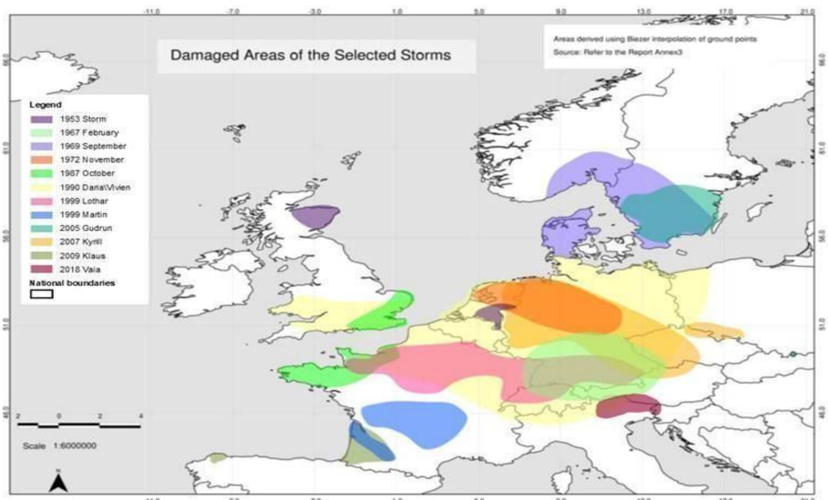

Figure 1: Estimated area of European forests affected by severe storms in the last 70 years (Adapted from Gardiner et al. 2010)

Windthrow is the disturbance that most significantly impacts European forests, and it is expected that wind damage will increase in the years to come (Bebi et al. 2017; Forzieri et al. 2021; Patacca et al. 2023; Seidl et al. 2017). Due to climate change, widespread warming is expected across Europe in the near future. This could lead to an increase in both the intensity and frequency of storms, with their paths shifting northward and eastward (Gardiner et al. 2013; Rahmstorf and Coumou 2011).

Forest ecosystems affected by windstorms may undergo changes in structure and species composition (Schütz et al. 2006; Seidl and Blennow 2012) and a temporary reduction in the provision of ecosystem services (Meyer et al. 2008). Windthrows may also alter the susceptibility of forests to fires (Cannon et al 2017). Additionally, it is important to note that windthrows can trigger biotic disturbances, such as pest outbreaks, especially bark beetles (Bouget and Duelli 2004; Eriksson et al. 2005; Göthlin et al. 2000), which can thrive on the abundant fresh deadwood, posing risks even to healthy trees in the years following the event (Potterf and Bone 2017). The combination of climate change, more intense and frequent storms, along with the interaction between windstorms and bark beetle outbreaks, is likely to increase the vulnerability of forests and forest-related systems (Seidl et al. 2016). This could potentially lead to degradation phenomena of forest ecosystems. In fact, when cascade and compound disturbances (Romagnoli et al. 2023) occur in short succession, before the ecosystem can recover, lasting and unexpected changes can occur (Paine et al. 1998). Therefore, if the resilience of forest stands is insufficient to cope with the increased disturbance regime, forest ecosystems will encounter challenges in achieving recovery, even in the long-term.

In this context, post-event restoration activities are becoming increasingly important in forest management. Specifically, nature-based restoration solutions are essential for promoting the recovery of windthrown stands. These solutions support the creation of microsites for natural regeneration, enhance biodiversity, and improve the resilience of future stands (Lingua et al. 2023; Marangon et al. 2022; Marzano et al. 2013).

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1.4. Spruce Bark Beetle

Bark beetle outbreaks, intensified by climate change and unsustainable forest management, have become a major driver of forest degradation across Europe, threatening biodiversity, ecosystem services, and the timber economy. Addressing this crisis requires a shift from reactive to proactive strategies, including forest diversification, climate-adaptive species selection, and integrated policy frameworks to build long-term ecosystem resilience.

Historical context and current situation

Bark beetles, represented namely by Ips typographus, have emerged as one of the most damaging forest pests across Europe, particularly impacting Norway spruce (Picea abies L. Karst) stands. While outbreaks are a natural part of forest dynamics, their intensity, frequency, and geographical extent have increased dramatically in recent decades due to a combination of climatic stressors and anthropogenic forest management practices (Hlásny et al., 2021). The historical context of bark beetle outbreaks in Europe reveals a long-standing vulnerability rooted in monoculture forest management, particularly the widespread planting of Norway spruce beyond its natural range. These even-aged stands, prioritized for timber production, have become increasingly susceptible to disturbances. Since the early 2000s, Europe has experienced escalating beetle outbreaks fueled by warmer temperatures, prolonged droughts, and storm events. The Czech Republic has been one of the hardest hit, with bark beetles destroying up to 5.4% of spruce stock annually between 2017 and 2019 (Hlásny et al., 2021). Germany has reported over 250 million cubic meters of beetle-damaged timber by 2020, especially in Thuringia and Bavaria, following the extreme droughts of 2018–2019 (Fernández-Carrillo et al., 2024). Austria, Switzerland, Slovakia, and even parts of Scandinavia like southern Sweden and Finland are now seeing intensified outbreaks, as beetles adapt to warmer conditions and longer breeding seasons (Müller et al., 2022; Venäläinen et al., 2020; Jonsson et al., 2012). Forecasts indicate a 60% to 220% increase in bark beetle disturbances by the end of the century, with the potential for outbreaks to spread northward and to higher elevations as thermal suitability increases under climate change scenarios (Seidl et al., 2014; Ruosteenoja et al., 2016; Jaime et al., 2024). This trend threatens not only forest ecosystems and biodiversity but also Europe's carbon balance and timber economy, especially as synchronized cross-border outbreaks overwhelm local management capacities (Forzieri et al., 2021; Grünig et al., 2024).

Needs for restoration

Forest restoration after bark beetle outbreaks is crucial for ecological recovery and long-term forest resilience. Massive tree mortality from beetle infestations significantly reduces forest cover, carbon storage, and biodiversity (Kurz et al., 2008). Restoration helps to reestablish these lost ecosystem functions and mitigate the broader impacts of disturbance (Jactel et al., 2009). In addition to ecological benefits, restoration supports economic and cultural forest values and reduces the risk of long-term ecosystem degradation (Seidl et al., 2017).

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1.5. Ash-Dieback

The European or common ash (Fraxinus excelsior L.) is widespread in Europe (Goberville et al. 2016) and occurs on 1-2 % of the European forest area (Enderle et al. 2019). It has been and continues to be valued for its economic value and timber quality (e.g., furniture, sports equipment, tool handles), its growth performance, the ecological services it provides (e.g. a large number of associated organisms, stream bank stabilisation, high quality litter) and its ability to grow well on wetter sites (Pautasso et al. 2013; Skovsgaard et al. 2017; Vasaitis and Enderle 2017). At least since the early 1990s, dark spots have been observed on the leaves of an increasing number of trees. These spotted leaves fell earlier than usual and twigs or branches died, resulting in a damaged crown (Cleary et al. 2013). After the first dieback symptoms, some trees started to produce epicormic shoots from branches or the trunk to compensate for the losses, giving the trees a "bushy appearance" (Carroll and Boa 2024). Usually ash trees are affected by the disease for several years until it is fatal (Timmermann et al. 2017). This phenomenon became known as "ash dieback" (ADB).

Where it occurs, ADB is able to kill up to 85 % of all ash (particularly F. excelsior and F. angustifolia) and is therefore one of the most destructive diseases within European riparian forests today (Carroll and Boa 2024; Coker et al. 2019). Under 5 % of ash are considered to be resistant (i.e. show no or only minor symptoms) (Enderle et al. 2019; Carroll and Boa 2024). These two figures create a gap of 10 % of ash, which seems to be able to survive under ADB at least for now. However, there are still at least 2-5 million hectares of European forests, which were degraded due to ADB and have to be restored (Enderle et al. 2019; Pautasso et al. 2013). Based on these observations, the search for the causative agent was initiated, full of hope to find a coping mechanism. In the end, it took more than 10 years before the fungus "Chalara fraxinea" was finally isolated (Kowalski and Holdenrieder 2009; Kowalski 2006). After a change in nomenclature, the final and now widely used name for the pathogen is "Hymenoscyphus fraxineus" (Carroll and Boa 2024). Much research has been done on the biology, phylogeny and genetic structure of this species and mitigating factors (Carroll and Boa 2024; Enderle et al. 2019). In Europe however, the pest is still out of control.

Historically, the first serious dieback symptoms appeared in 1995 in north-eastern Poland *(Figure 2)* (Przybył 2002). Nevertheless, there are strong indications that H. fraxineus has been present in Europe for some time before: it usually takes several years for the disease to develop eye-catching signs and DNA sequencing revealed the remnants of the fungus in herbarium samples in Estonia as early as 1978 (Combes et al. 2024). This year coincides with the introduction of seedlings from the Far East into the Baltic States, probably the first introduction of the pathogen into Europe (Drenkhan et al. 2014). Yet, from first reports around 1990 in Poland, most literature estimates spreading distances of 30 to 70 km per year (Enderle et al. 2019; Gross et al. 2014). This figure is assumed to be an interaction of the aerial dispersal of ascospores, as well as plant trade, which is up to this day not forbidden (only not recommended) within the European Union (Carroll and Boa 2024; Combes et al. 2024). However, one must keep in mind that these observational data are highly dependent on written reports of forest pathology experts, their awareness and knowledge about the disease. Hence, the spread history is likely spatiotemporal biased and could have been even faster (Enderle et al. 2019; Carroll and Boa 2024). The dispersal of ascospores purely by air for example is estimated to be on average between 0.2 to 2.6 km distance from the disease front (Cracknell et al. 2023). Other studies found ascospores up to 100 km from the last infected tree (Grosdidier et al. 2018). Today ADB is present over the entire distribution area of F. excelsior on the European continent (Erfmeier et al. 2019). Tests with American ash species revealed that at least five of them (green ash (F. pennsylvanica), white ash (F. americana), black ash (F. nigra), pumpkin ash, (F. profunda), and blue ash (F. quadrangulata) are also susceptible to the disease, posing a potential risk in the future (PPQ 2022; Nielsen et al. 2017).

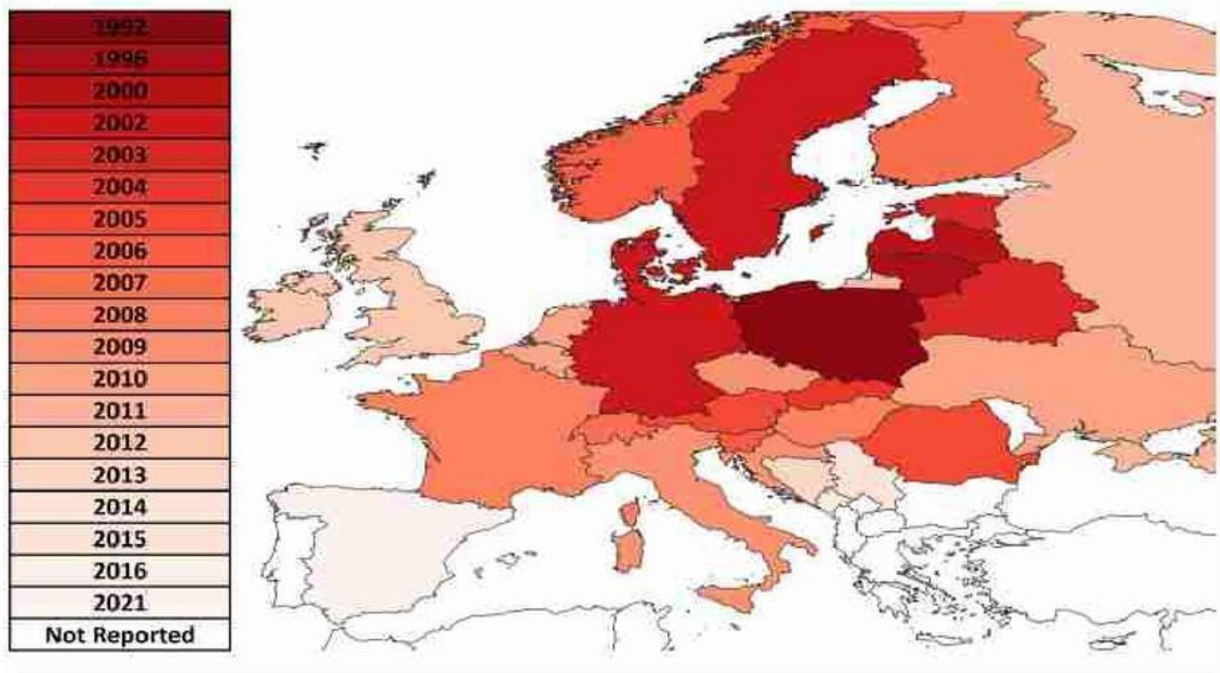

Figure 2: Reported spread of Ash dieback across Europe: a multinational perspective. Source: Carroll and Boa (2024).

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1.6. Improper management of riparian forests

Riparian and floodplain forests are typically located in low-lying areas of river basins, where they are periodically disturbed by flooding events (Horn & Richards, 2006; Hughes et al., 2012). This dynamic hydrological regime makes them unique and ecologically significant ecosystems that provide a wide range of ecosystem services and societal benefits. Among their functions of sequestering carbon, regulating and purifying water, stabilizing slopes, and supporting biodiversity (Dinca et al., 2025), their flood retention and protection roles are particularly important, as they operate through physical, chemical, and biological processes to shield urban areas from natural hazards (Sallmannshofer et al., 2021). By slowing flow velocities during flood events and lowering peak discharges in the hydrograph, floodplain forests mitigate the impacts of flooding on human settlements and infrastructure (Horn & Richards, 2006). Furthermore, riparian forests reduce nutrient runoff and soil erosion from adjacent agricultural land (Peterjohn & Correll, 1984; Dinca et al., 2025) and enhance groundwater recharge by prolonging water retention (Hughes et al., 2012).

Floodplain forests function as ecotones linking aquatic and terrestrial ecosystems and hence play a pivotal role in maintaining ecosystem connectivity and habitat quality. Leaf litter supplies essential nutrients to river-influenced areas, while roots, coarse woody debris, and shading create heterogeneous habitats that are vital for both aquatic and terrestrial biodiversity (Webster et al., 1995). These forests are further characterized by dynamically changing small-scale habitats, which foster structurally complex landscapes with exceptionally high species richness, including invertebrates, birds, and mammals (Hughes et al., 2012; Sallmannshofer et al., 2021). The mosaic of habitats and the emergence of early successional patches following flood disturbances are crucial for the natural regeneration of pioneer species such as Populus nigra and for maintaining overall plant species diversity (Patou & Decamps, 1985, as cited in Hughes et al., 2012). Collectively, this diversity of microhabitats not only establishes floodplain forests as biodiversity hotspots within riverine landscapes but also enhances their resilience and long-term ecological stability.

The diversity of natural habitat types in riparian forests arises from the dynamic interplay between water and land. Along riverine floodplain landscapes, three principal forest communities can be distinguished along the gradient from the watercourse to the outer floodplain boundary. Riverine forests occupy low-lying and frequently flooded sites with unstable soils, where pioneer species such as willows (Salix spp.) and alders (Alnus spp.) dominate. Floodplain forests occur on higher and more stable terrain, often beyond the direct influence of river flow but still within flood-prone areas, and are typically composed of pedunculate oak (Quercus robur), elm (Ulmus spp.), ash (Fraxinus spp.), and hornbeam (Carpinus betulus). Swamp forests, in turn, inhabit depressions with high groundwater tables and prolonged waterlogging, with black alder (Alnus glutinosa) as the characteristic tree species *(Figure 3)*. These forest types – and their respective subtypes – differ subtly in soil drainage, hydrological regime, and species composition, and together they create a rich mosaic of habitats that underpin biodiversity, nutrient cycling, and water regulation (Sallmannshofer et al., 2021).

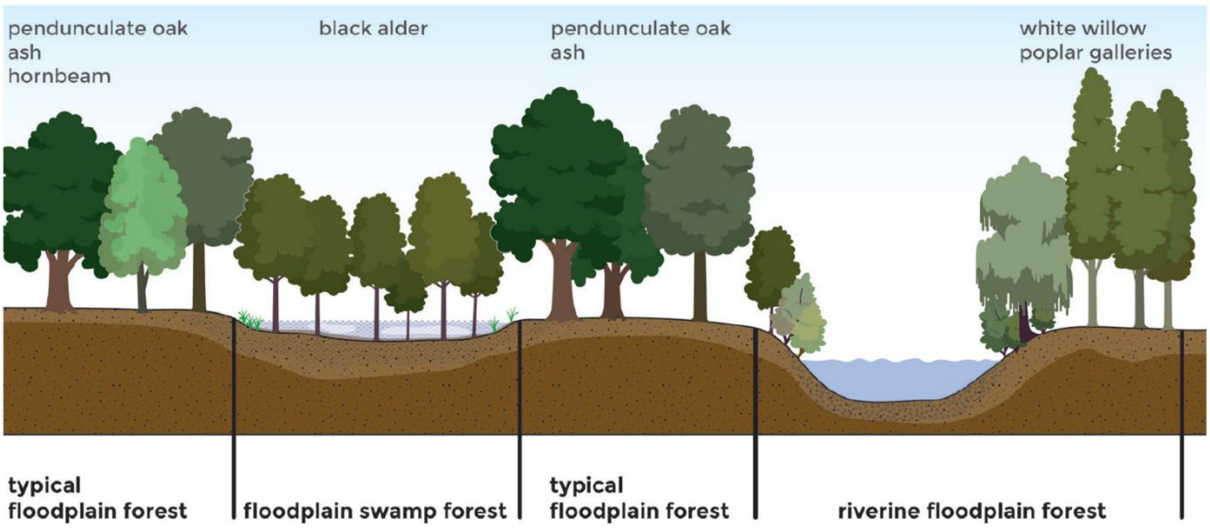

Figure 3: Typical riparian forest types (Sallmannshofer et al., 2021)

Nevertheless, human activities increasingly undermine the ecosystem services and resilience of riparian forests. Historically, these distinct site conditions were also reflected in the development of adapted management systems, ranging from coppices on dynamic riverbanks to high forests on more stable floodplains. While such traditional practices often maintained structural and compositional diversity, more recent forms of intensive exploitation and conversion have increasingly disrupted this balance. The high productivity of floodplain stands has additionally promoted intensive management aimed at maximizing timber yields and economic returns (Hughes et al., 2012; Sallmannshofer et al., 2021). River engineering practices have altered stream morphology and modified the hydrological regime within these forests. As a result, natural flood dynamics have been suppressed, groundwater levels have declined, and many flood-influenced stands have been drained, severely compromising ecosystem resilience and stability. Large-scale clearcutting for agriculture and timber production has further reduced their natural distribution, leaving an estimated 10% of riparian forests intact in Europe (Horn & Richards, 2006; Hughes et al., 2012; Sallmannshofer et al., 2021).

Interrupting natural flood dynamics has profoundly negative ecological effects on floodplain forests, rendering them increasingly prone to drought stress. Under climate change, such conditions are projected to intensify, amplifying the risks even further. Altered precipitation patterns, seasonal shifts, rising temperatures, and a growing frequency of extreme weather events pose severe abiotic challenges to these already disturbed ecosystems. At the same time, biotic disturbances – including pest outbreaks, forest pathogens, and invasive plant species – further jeopardize the health and long-term stability of riparian forests (Sallmannshofer et al., 2021).

Given the severe loss of natural riparian forests and the compounded pressures from human disturbance and climate change, restoration has become essential to safeguard their ecological integrity and the ecosystem services they provide. Restoration efforts not only aim to recover biodiversity and habitat heterogeneity but also to re-establish natural flood dynamics, strengthen resilience to drought and extreme events, and secure long-term benefits for both ecosystems and human societies. The unfavourable conservation status of riparian forests has gained international recognition and has been addressed through various policy frameworks and legal instruments, including the Ramsar Convention, the Forest Europe process, the EU Biodiversity Strategy, and the Habitats Directive. These initiatives acknowledge the urgent need for active measures to halt ongoing degradation and to restore healthy riparian ecosystems, thereby supporting biodiversity conservation and human well-being (Sallmannshofer et al., 2021).

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2. Chapter: Review of forest degradation drivers

This chapter focuses on six different drivers of forest degradation in Europe, examining their effects on forest ecosystems, their spatial distribution, and the ways they interact with predisposing factors. It also considers insights from published research on restoration approaches and highlights selected case studies that address these challenges.

2.1. Drought

Extreme drought events can trigger a cascade of physiological and structural damage in trees, including reduced photosynthesis, carbon starvation, and hydraulic failure, which can extend to entire forest ecosystems (McDowell et al., 2008). These effects manifest as early leaf discoloration, premature leaf senescence, partial or complete canopy dieback, and ultimately the mortality of individual trees, tree groups, or entire stands (Schuldt et al., 2020). As a result, tree growth is significantly reduced, forest productivity declines, and the forest's capacity for carbon sequestration is diminished (Treml et al., 2022).

While climatic conditions are the primary drivers of drought stress, the extent of drought-induced tree mortality is influenced by a complex interaction of abiotic and biotic factors. These include site conditions, tree species, age, size class, and the individual tree's vitality, which is often shaped by the legacy effects of previous drought events (Allen et al., 2010; Schuldt et al., 2020).

Drought is a severe driver of forest degradation and tree mortality in Europe. Extraordinarily dry and warm climatic conditions during the late 1990s and early 2000s affected forests across Europe, from various species in the Mediterranean region to pedunculate oak mortality in Poland and Norway spruce dieback in southeastern Norway (Allen et al., 2010). Between 1987 and 2016, approximately 500,000 hectares of forest mortality were observed due to drought conditions. Forest canopy mortality occurred in waves with different hotspots across Europe *(Figure 4)*, with the highest impact in 2000 in southeast Europe, 2003 in central and southern Europe, 2005 on the Iberian Peninsula, and 2007 and 2012 in eastern Europe (Senf et al., 2020).

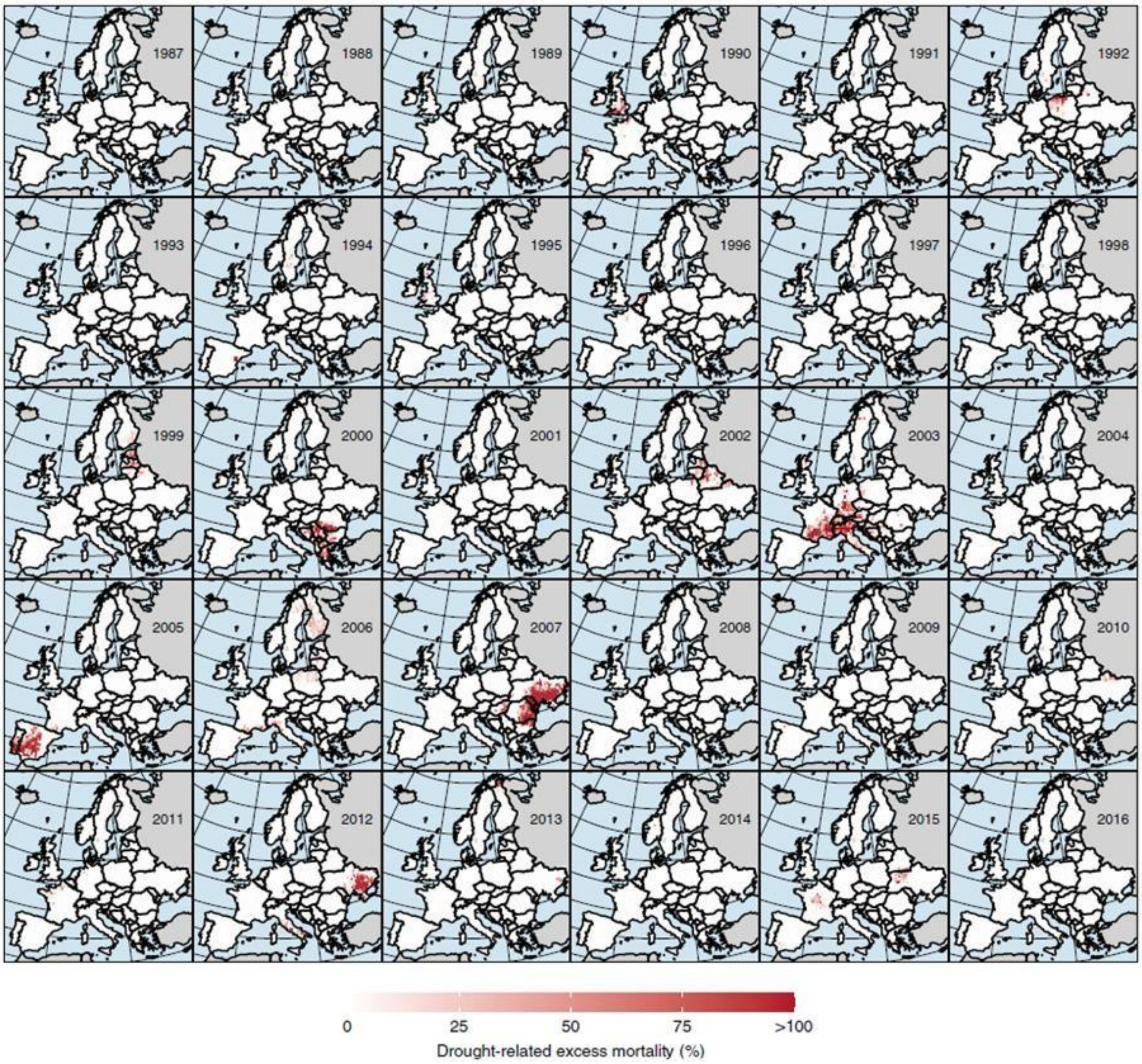

Figure 4: Hotspots of excessive forest canopy mortality due to drought between 1987 and 2016 in Europe (Senf et al., 2020).

Cammalleri et al. (2023) detected about 200 meteorological drought events in Europe using the SPEI-3 (three-month cumulated Standardized Precipitation Index) over the period 1981–2020. They successfully identified severe drought events, including those in 1992 in northeastern Europe, 2003 in Central Europe, 2005 over the Iberian Peninsula, 2007 in southeastern Europe, and 2018 in Central and Northern Europe.

In the DACH region (Austria, Germany, and Switzerland), the 2018 drought primarily affected Norway spruce (Picea abies) and European beech (Fagus sylvatica). However, other species such as Scots pine (Pinus sylvestris), silver fir (Abies alba), sessile oak (Quercus petraea), and pedunculate oak (Quercus robur) also experienced significant stress, reduced vitality, and, in some cases, increased mortality (Schuldt et al., 2020). Elevated mortality rates in drought-sensitive species like Norway spruce were largely expected due to their shallow root systems and limited drought resilience (Popa et al., 2024; Schuldt et al., 2020). Similarly, Scots pine growing on dry, sandy soils with low water-holding capacity was particularly vulnerable. However, unexpectedly high mortality was also recorded in species like European beech, silver fir, and oak, even on sites previously considered less drought-prone (Schuldt et al., 2020).

Forests across Central Europe were severely impacted by the extreme drought conditions of 2018, which were characterized by anomalies in both precipitation and temperature. Approximately 11% of Central European forests experienced early wilting, resulting in a legacy effect of weakened tree vitality in the following year. The most significant impacts were observed in central and eastern Germany, as well as in the Czech Republic. Regionally, drought-prone sites were primarily found on hilltops, south-facing steep slopes, and areas with low soil moisture. Terrain variables and vegetation characteristics proved to be key factors in predicting early wilting on a regional scale. High-resolution data are crucial for accurately forecasting drought responses such as early wilting. However, this remains challenging at the European scale, as even within the same model, the effects of predictor variables can differ significantly depending on the underlying dataset and spatial scale (Brun et al., 2020).

The climatic water balance (CWB), defined as the difference between monthly precipitation sums and monthly potential evapotranspiration, is also a suitable variable to predict forest canopy mortality. Hotter, drier conditions lead to a decline in the CWB and an increase in canopy mortality (Senf et al., 2020).

Thinning is a highly recommended silvicultural measure to reduce drought sensitivity and increase the resilience of forest stands (Clark et al., 2016; Klos et al., 2009; Manrique-Alba et al., 2022). Manrique-Alba et al. (2022) observed reduced climatic dependence of growth and shorter recovery times after drought events in thinned pine stands. The legacy effect of thinning can last for 15 to 20 years after the intervention.

Establishing mixed forest stands with less drought-prone tree species is also a common strategy to mitigate climate change impacts (Clark et al., 2016; Hereş et al., 2021; Lindner et al., 2014; Popa et al., 2024). Assisted migration of suitable tree species and genotypes adapted to future climate conditions can help maintain forest ecosystem services, such as carbon storage capacity (Chakraborty et al., 2024). It is important to consider local site and stand conditions when planning silvicultural measures (Clark et al., 2016). Forest management can alter forest stands to improve climate sensitivity, reduce disturbance regimes, and play a crucial role in fostering the resilience and recovery of forests for ecosystems and societies (Seidl et al., 2017).

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2.2. Fire

Fire, the second most important forest disturbance in Europe, damaged an average of 12 million m3 of timber per year between 1950 and 2019. However, the majority (82%) of the burned area reported during this period was in the Mediterranean region (Patacca et al. 2023). The year 2023 was one of the worst for fires in Europe, with a significant total area burned (over 500,000 ha) and “megafires” such as the Evros Fire in late August near the city of Alexandroupolis in Greece, which was the largest fire recorded in the European Union since 2000. It affected 81 000 ha and caused over 20 fatalities. Among the countries of the Interreg Central Europe region: Austria, Croatia, Czech Republic, Germany (eastern part), Hungary, Italy (northern part), Poland, Slovakia, and Slovenia, the annual number of fires and the total area burned in 2023 were higher than the average for the period 2013–2022 only in Germany (and in Italy for the second variable) (San-Miguel-Ayanz et al. 2024). In Austria, despite the dominance of anthropogenic fires, natural ignitions caused by lightning are high, especially in summer months – up to 50%. In recent decades, the largest burnt area and average fire size were recorded in 2022. In Croatia, wildland fires are mainly caused by human activities. Notably impactful fire seasons occurred in 2000, 2003, 2007, 2011, 2012, 2017, 2020, and 2022. In the Czech Republic, fires are also mainly anthropogenic and localized in a couple of hotspots in the country. Recently, major fire years were the following: 2003, 2014, and 2022. In Germany, the region of Brandenburg in the eastern part of the country is the current regional hotspot of wildland fires, which was particularly visible in fire seasons of 2003, 2018, and 2023. In Hungary, 99% of forest fires are human - induced. The largest burnt area and average fire size were reported in 2005, 2009, 2012, and 2022. The national forest fire statistics for Italy are not easy to interpret due to different climate conditions in the northern and southern sections of the country. In general, together with Portugal, Spain, and France, Italy belongs to the countries with the highest occurrence of wildland fires in Mediterranean Europe. In the northern regions of the country such as Veneto, Friuli, Trento, and Valle D’Aosta, substantially lower fire occurrence is documented when compared to the southern part of Italy. In the recent decades, 2007, 2017, and 2021 were years with more impactful fire seasons. In Poland, the majority of fires are anthropogenic, with a substantial share of arson greater than 40%. Since 2000, the highest annual number of fires and the area burnt were recorded in 2003, 2015, and 2019. In Slovakia, the most notable fire seasons in the last decades occurred in 2003, 2012, and 2022. In Slovenia, the south-western (i.e., the Mediterranean) section of the country has the highest fire risk. Recently, the largest burnt area and average fire size were reported in 2003, 2006, and 2022, which confirms that some years in the last two decades were major fire years in the region (Fernandez-Áñez et al. 2021; San-Miguel-Ayanz et al. 2024). Nevertheless, several factors are likely to increase the future fire occurrence in the region: climate change (affecting fuel availability and fire weather), a shift in forest management from timber production to multifunctional forest use, including recreational use (and the associated increase in human presence), high population density, the increasing expansion of the wildland-urban interface, and large areas affected by windthrow and bark beetle outbreaks (Schelhaas et al. 2010; Duane et al. 2021; Fernandez-Áñez et al. 2021; Berčák et al. 2023).

The effects of fire on forest ecosystems are complex and depend on the biogeographic region, pre-disturbance forest structure, fire behaviour, and fire severity (Agee 1993; Pyne et al. 1996). Similar to other pine species, such as Pinus ponderosa in North America, Eurasian Pinus sylvestris is a species adapted to a mixed-severity fire regime, where low-severity surface fires are combined with occasional high-severity events of varying magnitudes (often in the form of passive crown fires that create canopy gaps of different sizes) (Keeley 2012). Tree ring fire history reconstructions provide evidence of such a fire regime in Scots pine-dominated forests in Central Europe during the last centuries (Zin et al. 2015; Manton et al. 2022), comparable to the boreal part of the continent (Blanck et al. 2013). Nowadays, in managed forests, stand-replacing fires are definitely associated with the highest economic losses and very often with intensive post-disturbance activities, e.g., salvage logging and tree cover restoration (Piszczek 2007). In temperate Europe, fire-disturbed areas generally have a lower recovery potential compared to the other disturbance agents (Cerioni et al. 2024).

In southern Europe, the framework for post-fire forest restoration was widely discussed over a decade ago (Moreira et al. 2012a). Examples from Central Europe (Ascoli et al. 2013; Kitenberga et al. 2020; Blumroeder et al. 2022) confirm that the same key concepts and activities are generally applied: active restoration (tree planting, seeding), passive restoration (ecological succession, i.e., natural tree regeneration, from seed or resprouting), and assisted restoration (including activities such as thinning, protection from herbivores, control of undesired vegetation, and selection of shoots in coppices) (Moreira et al. 2012b). In addition, they provide evidence that a scientific debate has already been initiated on post-fire forest restoration, including the issue of salvage logging, tree retention and natural forest succession (non-intervention approach) (Kitenberga et al. 2020; Schüle et al. 2023).

Biological legacies, such as remnant living and dead trees, play a crucial role in the resilience of post-fire forest ecosystems by facilitating ecological recovery (Jõgiste et al. 2017). Living remnant trees in these areas offer a seed source and create a favorable microclimate for seedling establishment (Moser et al. 2010; Marzano et al. 2013). However, they can also compete with regenerating trees for light and nutrients, potentially negatively impacting their abundance and height growth (Parro et al. 2015). Conversely, standing and lying dead trees can provide significant microclimatic benefits, e.g. by reducing evapotranspiration, lowering surface and ambient temperatures, etc. (Moser et al. 2010; Blumroeder et al. 2022), and create microsites that promote tree establishment by offering protection from radiation and reducing soil moisture loss (Marzano et al. 2013; Jouy et al. 2025).

Several case studies from Central Europe on post-fire restoration of Scots pine forests in northern Italy, eastern Germany, Poland, Czech Republic, Estonia, and Latvia, have demonstrated successful tree regeneration of different species (including pine) in a non-intervention scenario (Beghin et al. 2010; Marzano et al. 2013; Parro et al. 2015; Adámek et al. 2016; Dobrowolska & Pawlak 2020; Kitenberga et al. 2020; Blumroeder et al. 2022; Schüle et al. 2023). The crucial role of leaving deadwood on site as a key structure with numerous benefits (such as creation of microsites for tree establishment, reduction of the impact of heat and drought extremes, protection against browsing) (Marzano et al. 2013; Jouy et al. 2025) and substantial share of deciduous tree taxa such as Populus and Betula (Beghin et al. 2010; Parro et al. 2015; Adámek et al. 2016; Dobrowolska & Pawlak 2020; Blumroeder et al. 2022; Schüle et al. 2023; Jouy et al. 2025) have been reported by many of these studies, along with a very diverse dendroflora of naturally establishing taxa, including Salix, Sorbus, Quercus, Fraxinus, Larix, Picea, etc. (Marzano et al. 2013; Adámek et al. 2016; Blumroeder et al. 2022; Schüle et al. 2023), which proves high potential for building mixed, multi-species forests. In addition, a case study from Switzerland provided evidence for the importance of external seed rain for natural tree regeneration of Scots pine after stand replacing fires and showed how short the window of opportunity for tree establishment is (1–2 years) (Moser et al. 2010). Studies, where active restoration of post-fire areas was conducted, documented several challenges such as high mortality of the planted tree seedlings (Blumroeder et al. 2022), lower establishment success of tree regeneration created by seeding, highest tree seedling growth in the scenario with planting and ground coverage by wood chips, lower height of the bare root seedling in comparison with the container seedlings, and substantia l browsing damage in plots with dead standing trees left (Gil 2020), and the importance of timing of the logging operations (and resulting soil disturbance) (Schüle et al. 2023). Interestingly, in contrast to the data from northern Italy, Estonia, and eastern Germany (Beghin et al. 2010; Marzano et al. 2013; Parro et al. 2015; Blumroeder et al. 2022), studies from Latvia and Poland do not bring conclusive evidence for the negative impact of post-fire salvage logging on tree regeneration and growth (Dobrowolska 2008; Kārkliņa et al. 2020; Kitenberga et al. 2020), which was also shown by a study in a beech forest in northern Italy (Ascoli et al. 2013). Although some management suggestions have already been formulated (Ascoli et al. 2013; Schüle et al. 2023), these different results highlight the need for further research and the development of a transnational strategy for post-fire forest restoration in the region.

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2.3. Wind

Windthrow is a natural disturbance that impacts forests in complex and varied ways. It can occur both at a large scale, affecting entire landscapes, and at a small scale, impacting individual trees (Pickett and White 1986). The susceptibility of trees to windthrow is influenced by multiple factors, including meteorological conditions, such as wind speed and gustiness, soil properties, site topography, the structure and composition of the forest stand, and the mechanical and physiological condition of each tree (Coutts 1986; Petty and Swain 1985; Petty and Worrell 1981; Savill 1976). Damage typically occurs when wind gusts exceed the mechanical resistance threshold of trees, leading to uprooting or stem breakage. The peak speed of wind gusts is strongly linked to the potential damage. According to Gardiner et al. (2010), no significant damage should occur at wind speeds below 30 ms-1. Moderate windthrow, affecting up to 2% of the national forest growing stock, can be seen between 30 ms-1 and 40 ms-1. High amounts of downed timber, impacting 2% to 4% of the growing stock, occur at speeds of 40 ms-1 to 45 ms-1. Severe damage, exceeding 4% of the growing stock, is expected for speeds above 45 ms-1. Soil depth and condition are also crucial. Shallow soil (especially in steep montane areas) is critical for tree stability, and the strength of root anchorage increases with soil freezing, but decreases with waterlogging due to heavy rain or poor drainage during storms (Gardiner et al. 2010). Additionally, the topography of a site influences the intensity and distribution of wind damage. In hilly or mountainous terrain, the airflow that descends beyond the hilltops or ridges becomes turbulent, creating wake zones. Forests located in these leeward zones may experience strong and unpredictable winds, increasing their vulnerability to windthrow (Gardiner et al. 2013). Regarding species composition, conifers are generally more susceptible to wind damage than broadleaves, due to the higher drag of evergreen canopies during winter storms, while broadleaved species are leafless. Specifically, Norway spruce, as well as poplar (Populus spp.), are among the most vulnerable species, while silver fir, European black pine (Pinus nigra J.F. Arnold), European beech, and oak (Quercus spp.) are considered among the least vulnerable (Gardiner et al. 2010). Over the past few centuries, forest management practices in Central Europe have changed, leading to a general decline in the resilience of forests to windthrow. Historically, these forests were mainly composed of native deciduous tree species, such as European beech and oaks. However, since the late 18th century, there has been a significant increase in the area planted with coniferous trees, such as Norway spruce and Scots pine. This shift has transformed several deciduous and mixed forests into monospecific, single-layered conifer plantations over large areas (Kirby and Watkins 2015), reducing forest resistance to windthrows.

Post-windthrow forest restoration is a complex issue, impacting ecological, managerial, and societal aspects across various scales. Trade-offs arise from local and landscape factors, including ecosystem effects, operational strategies, costs, and the response of wood-forest value chains (Romagnoli et al. 2023). Usually, post-windthrow management involves rapid salvage logging operations aimed at swiftly removing timber from damaged stands to limit economic loss (Udali et al. 2021). However, the practice of salvage logging is debated, as it may negatively impact forest communities (Leverkus et al. 2018). Several studies refer to salvage logging as a disturbance itself (Hernández-Hernández et al. 2017; Leverkus et al. 2021), or as a management option that results in secondary disturbances (Kleinman et al. 2019; Royo et al. 2016). Indeed, by removing the disturbance legacies created by the windstorm, salvage logging disrupts microsite diversity and limits seedling establishment, which can lead to an overall reduction in biodiversity and accelerate the degradation process (Bouget, 2005; Romagnoli et al. 2023; Waldron et al. 2014). Nevertheless, when windthrow affects large areas of forest and causes significant timber losses, decision-makers become concerned about the potential collapse of the timber market (Pischedda and Stodafor 2004). In such cases, salvage logging is considered the most effective strategy, especially to reduce the risk of bark beetle outbreaks (Nikolov et al. 2014). However, effective sanitation logging requires the rapid removal of a substantial number of fallen trees (Dobor et al. 2020), thereby causing a significant environmental impact.

Alternative approaches, often referred to as nature-based solutions, such as the retention of biological legacies and non-intervention strategies, can effectively promote the recovery of forest stands while maintaining ecological functions (Baker et al. 2023; Morimoto et al. 2021). Biological legacies, defined as the biological materials that have persisted after the disturbance, play a fundamental role in the recovery process of the forest ecosystem (Franklin et al. 2000). Structural legacies, such as deadwood or snags, provide protection and may create optimal microclimatic conditions that promote the establishment of early regeneration (Beghin et al. 2010; Marzano et al. 2013). In protective stands in mountain areas, elements like lying logs, stumps, and snags may also protect against gravity-driven hazards, especially immediately after the windthrow (Costa et al. 2021; Schönenberger et al. 2005; Wohlgemuth et al. 2017). Ecology-based restoration can be a key aspect in post-windthrow management because it enhances the recovery of wind-degraded forest stands by taking advantage of these legacies (Szwagrzyk et al. 2018; Valinger et al. 2014). For instance, deadwood elements can serve as natural shelters for transplanted seedlings (Marangon et al. 2022), and lying logs can act as barriers against rockfalls and avalanches (Costa et al. 2021; Fuhr et al. 2015; Olmedo et al. 2016). Additionally, these strategies can play a crucial role in preventing the establishment of invasive species in the newly created gaps resulting from storm events (Morimoto et al. 2013). However, it is important to consider that the retention of biological legacies can strongly affect existing management targets (Fidej et al. 2018) and may create conditions favorable for other natural disturbances, such as bark beetle outbreaks (Romagnoli et al. 2023). Consequently, strategies for post-disturbance management should be customized to fit each specific situation and context (Sanginés De Cárcer et al. 2021).

The study conducted by Marangon et al. (2022) provides a clear example of a nature-based restoration strategy following windthrow events. The research investigated the impact of coarse woody debris (CWD) on the survival and protection of transplanted tree seedlings in an alpine forest affected by the storm Vaia in 2018. Thirty experimental blocks were established, each containing seedlings of five species: Norway spruce, silver fir, European larch (Larix decidua Mill.), European beech, and rowan (Sorbus aucuparia L.). The results indicated that mortality rates were significantly lower in the microsites adjacent to CWD, particularly on the north side. This reduction in mortality can be attributed to the moderating effects on temperature and shade provided by the CWD. Additionally, the presence of lying logs helps to minimize grazing by obstructing the passage of ungulates, especially when the seedlings are still small. However, this protective effect diminishes as the seedlings grow. The study suggests that selectively retaining deadwood can enhance the effectiveness of restoration efforts in post-disturbance upland environments.

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2.4. Spruce Bark Beetle

Climate change plays a central role in exacerbating bark beetle outbreaks. Warmer temperatures accelerate beetle development, increase voltinism (number of generations per year), and reduce winter mortality, allowing populations to reach outbreak levels more easily (Jaime et al., 2024; Jonsson et al., 2012; Ruosteenoja et al., 2016). The European bark beetle can now produce two to three generations annually in some areas, compared to one in historical norms (Müller et al., 2022). Drought significantly reduces tree defenses (e.g., resin production), making even healthy trees susceptible. During prolonged droughts like the 2018 European heatwave, vast tracts of forest suffered beetle-induced mortality (Venäläinen et al., 2020; Müller et al., 2022). Forest structures such as monocultures, even-aged stands, and extensive clear-cutting contribute to high forest vulnerability. Historically, Norway spruce has been planted extensively due to its economic value, replacing more resilient broadleaf species and creating landscapes with diminished ecological resistance (Hlásny et al., 2021; Jaime et al., 2024).

Bark beetles disperse short distances to neighboring trees but can also travel several kilometers under favorable wind conditions. Proximity to previously infested stands increases the likelihood of attack within a 500 m radius (Müller et al., 2022). Clear-cutting and storm damage provide additional breeding substrate, further amplifying outbreaks (Jonsson et al., 2012).

Bark beetle outbreaks fundamentally alter forest ecosystems in a way of changing forest structure, biodiversity, and ecosystem functioning, as they a) shift species composition, b) reduce carbon sequestration, c) increase fire risk due to dead biomass accumulation, e) cause cascading ecological and economic damage (Jaime et al., 2024; Venäläinen et al., 2020).

Current restoration and management approaches

a) salvage logging and sanitation felling

These are short-term containment measures aimed at removing infested trees before beetle emergence. However, such practices can have limited long-term effect and may even exacerbate disturbances by altering microclimates and removing predator habitats (Hlásny et al., 2021; Jonsson et al., 2012).

b) spatial risk mapping and decision support

Advancements in remote sensing and machine learning (e.g., fuzzy-AHP and Bayesian belief networks) are helping to create spatially explicit risk maps, allowing forest managers to prioritize vulnerable areas for proactive treatment (Tahri et al., 2022).

c) diversification and proactive forest design

Long-term restoration is shifting toward planting mixed-species and structurally complex forests, which have shown higher resistance to pests and climate variability. Species selection is being adapted to future climate projections, emphasizing ecological suitability over timber yield (Jaime et al., 2024; Müller et al., 2022).

d) policy integration and adaptive management

Restoration requires aligning forest policy with climate adaptation strategies, integrating conservation, economic use, and ecological resilience. Promoting continuous cover forestry and avoiding monocultures are core principles of modern restoration policy (Hlásny et al., 2021; Tahri et al., 2022).

Restoration case studies

Case studies from Central Europe highlight the importance of active forest management in mitigating the impacts of bark beetle outbreaks and supporting effective forest recovery. Hlásny et al. (2017) emphasize that climate change is intensifying disturbance regimes, making it essential to adopt adaptive management strategies, particularly in production forests where salvage logging and targeted interventions can help reduce future vulnerability. Falt'an et al. (2021) acknowledge that while unmanaged areas may show structural diversity, active measures are crucial to maintain landscape stability and prevent excessive fragmentation that can hinder long-term forest resilience. Similarly, Nováková and Edwards-Jonášová (2015) observed natural regeneration in non-intervention zones, but their findings also underscore the slow and unpredictable nature of spontaneous recovery, which may not align with management goals in multifunctional or economically valuable forests.

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2.5. Ash Dieback

From the time the ascospore lands on an ash (usually in spring and early summer), it could invite the tree through lenticels, leaf stomata or by penetrating the epidermis (Schumacher et al. 2010). Once inside the plant, the fungus enters the xylem, where it preferentially spreads longitudinal and through the starch-rich paratracheal parenchyma (Combes et al. 2024). Initially, the fungus grows in the intercellular space, then penetrates the cells, which die as a result, causing discolouration of the leaves and bark (Gross et al. 2014; Haňáčková et al. 2017). In autumn and winter, H. fraxineus behaves as saprophyte and mates, developing small fruiting bodies on the previous year’s infected leaf litter, which can release ascospores in the next years (Gross et al. 2012; Chandelier et al. 2014; Mansfield et al. 2019). Kirisits (2015) found that the fungus is able to survive on the ground at least up to 5 years and still produce vital ascospores. Additionally, it is also able to infect ash trees through the root system (Fones et al. 2016).

Table 1: Factors associated with symptom severity of ADB according to multiple references (Erfmeier et al. 2019; Carroll and Boa 2024; Combes et al. 2024; Enderle et al. 2019; Cracknell et al. 2023; Fuchs et al. 2024). Modified after Enderle et al. (2019).

Predisposing factor	ADB severity increases on or at
Stand or tree characteristics	
Tree size (tree height, DBH, Crown projection area, age)	smaller and younger trees
Productivity	less
Density	high
Species mixture	low
Location (open land, closed forests)	within forests
Leaf flushing	depends on the reference
Tree Gender	depends on the reference
Cultivars or provenances	*NA*
Regeneration method	artificial
Infested with …	*Armillaria* spec.
Host species (which *Fraxinus* species)	*F. excelsior* and *F. angustifolia*
Site characteristics	
Temperature	low to moderate temperature
Precipitation (soil type, moisture, humidity, distance to a river, drought)	moist, humid, wet sites
Solar radiation, light availability	lower
Slopes	steeper
Altitude	lower
Air humidity	lower
Management	
Thinning regime	unthinned stands
Time of pathogen arrival	

A number of factors have been identified that may exacerbate the disease ("predisposing factors"). (1) First of all, taller trees (i.e. increasing measures like height, DBH or crown projection area) seem to increase ADB, presumably because older specimens can better cope with losses in their crown. (2) The same applies for individuals that are more productive. (3) An increase in species mixture and forest density, however, facilitates higher air humidity, less light availability and less wind speed, which might be the cause, why ADB severity is commonly reported to increase under such conditions. (4) Moreover, trees in open landscapes seem to be less affected than individuals in dense forests. (5) The time of leaf flushing and tree gender influence the course of the disease probably because both factors account for leaf area exposed to ascosporic contact. (6) Different provenances or cultivars respond differently to the disease, as well as trees regenerated by different methods. Both factors may influence genetic diversity and productivity and therefore the ability to defend themselves. (7) Stands infected with the Armillaria root rot show stronger dieback symptoms, although it is often not entirely clear which pathogen came first. (8) Finally, some species of the Fraxinus genus, for example F. excelsior and F. angustifolia, have a higher susceptibility to ADB (see also Table 1) (Erfmeier et al. 2019; Carroll and Boa 2024; Combes et al. 2024; Enderle et al. 2019; Cracknell et al. 2023; Fuchs et al. 2024).

Thus, what management strategies or coping mechanisms do we know of to reduce the impact of ADB? Starting with the detection of the disease: in addition to (a) visual assessments (e.g., mortality rates, defoliation percentages or classes, degree of dieback, presence of epicormic shoots or necrosis), which are to some degree subjective and differ between scoring systems, there are several objective measures available (Enderle et al. 2019). One relatively slow method is a (b) PCR (Grosdidier et al. 2017; EPPO 2013). Faster would be (c) qPCR and feasible on site or (d) a kit developed by Fera Science Ltd and OptiGene (Chandelier et al. 2010; Harrison et al. 2017). Alternatively, (e) affected individuals can also be recognised by remote sensing (Gašparović et al. 2023).

More interesting however, are methods to find the 1-5 % of ash, which are resistant. In addition to the aforementioned (i) visual assessments (i.e., seeking trees, displaying no or minor signs of dieback), we found two more procedures in the literature: (ii) fourier transformed infrared spectroscopy of bark tissue extracts (Villari et al. 2018) and (iii) genomic predictions using 200 single nucleotide polymorphisms (Stocks et al. 2019).

Generally, the restoration strategy of ash in Europe can be divided into two time periods: short-term mitigation and a long-term strategy. In the short-term, it is recommended to use silvicultural practises like

(1) retaining tolerant-appearing ash as seed trees, (2) destruction of material which could promote disease spread (e.g. leaves by pruning, removing leaf litter), (3) restrict planting of ash trees (if considered, local material on non-hydromorphic soils), (4) early thinning and support of vital (female) specimens and (5) tree species selection towards mixed forests with site-adapted species (Enderle et al. 2019; Carroll and Boa 2024; Combes et al. 2024; Skovsgaard et al. 2017; Cracknell et al. 2023). Possible alternative species include for example sycamore (Acer pseudoplatanus) and beech (Fagus sylvatica), as well as sessile oak (Quercus petraea) or other ash species (Fraxinus spec.) (Lévesque et al. 2023). Another option in the future, could be the introduction of mycoviruses, but more research needs to be done in this field (Combes et al. 2024). Overall, legal restrictions on the trade of live and dead ash between countries within the EU could have helped in early stages of disease spread, but were never introduced. As H. fraxineus has already reached the entire distribution area within Europe, this also seems no longernecessary (EPPO 2014). In fact, national legislation to prevent further spread of the disease has only been changed in the UK (Enderle et al. 2019; Forestry Commission 2012). However, the EU enacted the Plant Health Law in 2016, which prohibits the import of live ash into EU member states until a risk assessment has been carried out (Carroll and Boa 2024; EU Commission; European Parliament and the Council). Almost all of these short-term solutions rely on preserving ash genetic diversity to set up breeding programs for ADB resistant ash in the future, although the underlying defense mechanisms are still not understood (McKinney et al. 2014; Enderle et al. 2019). It is estimated that it will take at least 20 years to get a first generation of ADB resistant seedlings (Carroll and Boa 2024). The distribution, planting and restoring of formally ash-dominated ecosystems will certainly take decades up to centuries. Yet, the genetic resistance will likely be passed down to the next generation and may even enhance growth potential, as well as the defence against the emerald ash borer (Agrilus planipennis), a potential next great threat for European ash species (Gossner et al. 2023; Eisen et al. 2024).

Studies of post-ADB forest restoration tend to focus on ash regeneration and how it is affected by the disease and how dominant compared to other species. Reported results usually show varying infection rates between 10-60 % and a shift to other tree species (Jochner-Oette et al. 2021; Lygis et al. 2014; Matisone et al. 2025).

However, ash still regenerates and utilizing the natural regeneration could be a good complementary strategy to organised breeding programmes (Enderle et al. 2017).

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2.6. Improper management of riparian forests

Riparian and floodplain forests are typically located in low-lying areas of river basins, where they are periodically disturbed by flooding events (Horn & Richards, 2006; Hughes et al., 2012). This dynamic hydrological regime makes them unique and ecologically significant ecosystems that provide a wide range of ecosystem services and societal benefits. Among their functions of sequestering carbon, regulating and purifying water, stabilizing slopes, and supporting biodiversity (Dinca et al., 2025), their flood retention and protection roles are particularly important, as they operate through physical, chemical, and biological processes to shield urban areas from natural hazards (Sallmannshofer et al., 2021). By slowing flow velocities during flood events and lowering peak discharges in the hydrograph, floodplain forests mitigate the impacts of flooding on human settlements and infrastructure (Horn & Richards, 2006). Furthermore, riparian forests reduce nutrient runoff and soil erosion from adjacent agricultural land (Peterjohn & Correll, 1984; Dinca et al., 2025) and enhance groundwater recharge by prolonging water retention (Hughes et al., 2012).

Sustainable management of riparian forests requires approaches that move beyond traditional compartment-based forestry. Two key concepts in this regard are the landscape perspective and ecological integrity. From a landscape perspective, riparian forests are understood as heterogeneous mosaics of patches and corridors, whose ecological value increases with size, compactness, and connectivity. Ecological integrity, in turn, refers to the capacity of riparian forests to maintain their structural, compositional, and functional diversity. Together, these concepts emphasize management at broader spatial scales, favoring practices such as spatial planning, selection systems, and structural thinning over uniform clearcutting, thereby enhancing the persistence and resilience of diverse habitat types (Sallmannshofer et al., 2021).

Riparian forests are also shaped by natural disturbance regimes, which play a crucial role in sustaining biodiversity and ecological function. The intermediate disturbance hypothesis explains how species richness is maximized under moderate disturbance levels, as illustrated by successional gradients along river-floodplain transects – from pioneer stages near watercourses to mature oak-dominated stands further inland. The insurance hypothesis further highlights the importance of species redundancy, as subordinate taxa can take over ecological functions when dominant species decline due to stress or disease. These insights underscore the dynamic and adaptive nature of riparian forest ecosystems. Consequently, management must integrate ecological knowledge with risk awareness – acknowledging threats from altered river dynamics, climate change, pests, and invasive species – in order to sustain both ecological integrity and the long-term socio-economic value of riparian forests (Sallmannshofer et al., 2021).

Traditional management systems co-evolved with the habitat types of riparian forests and include coppices, high forests, and coppice-with-standards, each adapted to different site conditions and disturbance regimes. Coppices, often found on cutbanks, islands, and water-adjacent terraces, rely on vegetative regeneration and maintain dense, low forests dominated by willow and alder. High forests, established through seed planting or natural regeneration, produce tall, mature stands of oak, elm, ash, and hornbeam. Coppice-with-standards combines both systems, with a lower coppiced layer and an upper canopy of tall standards, creating high structural diversity and multiple timber assortments. Historically, all three forms contributed to biodiversity conservation, the persistence of light-demanding species, and local livelihoods. Today, they remain both valuable cultural heritage and practical management options for sustainable wood production (Sallmannshofer et al., 2021).

Beyond these traditional small-scale forms, destruction and inappropriate forest management have caused significant biodiversity losses and weakened the stability of riparian forests and their ecosystem services (Hughes et al., 2012). Due to high soil fertility and economic demand, naturally regenerating riparian forests in temperate Europe were widely converted into hybrid poplar plantations (Klimo & Hager, 2001, as cited in Hughes et al., 2012). Following the Second World War, the urgent need for timber accelerated breeding programs and the establishment of poplar hybrids. Even today, economic pressures often push managers to integrate fast-growing, high-value species into their practices (Sallmannshofer et al., 2021). Meanwhile, river regulation has suppressed natural flooding, increasing tree mortality. Between 1930 and 1980, hydraulic engineering and land-use change reduced floodplain forests along the Austrian Danube from 33,000 to just 8,000 ha (Hager & Schume, 2001, as cited in Hughes et al., 2012).

Ecological restoration of riparian forests is only effective if it re-establishes the natural geomorphological heterogeneity and dynamics of floodplain systems (Brown et al., 1997). Brown et al. (1997) argued that large-scale restoration at the catchment level is unlikely in Europe due to complex land ownership and dense human populations. However, local-scale measures, such as removing river embankments to reconnect cut-off floodplains, are widely applied. In contrast, the EU-funded MERLIN project (Mainstreaming Ecological Restoration of freshwater-related ecosystems in a Landscape context; https://project-merlin.eu/) demonstrates how large-scale riparian and floodplain forest restoration can be implemented. Through 18 best-practice case studies, MERLIN showcases nature-based solutions that enhance biodiversity, resilience, and ecosystem services while addressing societal needs. For example, in Hungary, a 200 ha restoration near Nagykörü on the Tisza River combined ecological restoration with modernization of traditional floodplain farming, delivering benefits such as drought risk reduction, reconnection of wetlands, biodiversity enhancement, and increased opportunities for eco-tourism. In Austria, Danube restoration east of Vienna addressed ecological deficits, riverbed deepening, navigation safety, and flood protection, ultimately improving 10.8 ha of floodplain habitat. In Romania, 400 ha of agricultural land along the Lower Danube at Gârla Mare were re-converted into wetlands to improve biodiversity, mitigate flood risks, and restore lateral connectivity. Collectively, the MERLIN case studies illustrate how restoration can balance ecological and socio-economic objectives by combining practical interventions with adaptive management and stakeholder engagement (Gerner et al., 2023).

As part of the Interreg project REFOCuS (https://dtp.interreg-danube.eu/approved-projects/refocus), a stakeholder workshop was organized to strengthen dialogue between scientists and practitioners. It also created a platform for conservationists, forest managers, landowners, researchers, and local residents to exchange perspectives and articulate needs. Discussions highlighted socio-economic challenges in riparian forest management, including policy constraints and conflicting stakeholder interests. Ecological challenges were also noted, such as pest outbreaks, groundwater decline, and difficulties in natural regeneration caused by dense ground vegetation and browsing. Overall, the workshop emphasized the importance of inclusive communication in managing riparian forests within the Mura–Drava–Danube Biosphere Reserve. Its outcomes underline negotiation and stakeholder engagement as prerequisites for balancing interests, improving management practices, and building the social foundations for successful conservation and restoration (Sallmannshofer et al., 2021).

Forest management plays a central role in reconciling ecosystem services with ecological restoration by enhancing biodiversity, resilience, and stability – though achieving this at landscape scales remains challenging. A common strategy is the removal of non-native species combined with the planting of native trees, thereby transforming stands towards a more natural state (Hughes et al., 2012). Special emphasis lies on forest regeneration, as this critical phase determines future species composition and sets initial conditions for climate-adapted habitats. Whenever possible, natural regeneration should be promoted, as it harnesses natural selection, relies on autochthonous seed sources, and reduces costs. Where species composition has been heavily altered, however, artificial regeneration - or a combination of both - may be necessary (Sallmannshofer et al., 2021). Regardless of the strategy, complementary measures such as fencing to prevent browsing, controlling competing vegetation, and maintaining seedlings are often essential for successful restoration (Hughes et al., 2012; Sallmannshofer et al., 2021).

Converting poplar plantations into native riparian forests is another widely applied restoration strategy to enhance biodiversity and ecosystem resilience. Poplar monocultures often reduce habitat quality and compromise ecosystem integrity, for example by offering fewer suitable breeding sites for birds (Porro et al., 2021) or by altering understory vegetation. Although Martín-García et al. (2016) found no significant differences in overall species richness between poplar plantations and native forests, the plantations lacked vascular plants typical of riparian habitats, underlining the higher ecological value of close-to-nature forests. Moreover, intensive practices such as harrowing for weed control strongly altered species composition, illustrating trade-offs between wood production and biodiversity goals (Martín-García et al., 2016). González et al. (2016) further showed that abandonment of plantations – with or without tree harvesting – induces major vegetation shifts. However, these communities still diverge markedly from natural riparian forests, indicating that passive restoration alone cannot recreate the habitat quality of native ecosystems (González et al., 2016).

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The following chapters will be included in the final version of the transnational strategy to restore degraded forests of Central Europe:

3. Chapter: Policy and stakeholder perspectives

This chapter will be based on:

- Input from Stakeholders Workshop (Deliverable 1.1.2)
- Input from Stakeholder Survey (Deliverable 1.1.3)
- Input from Regional Policy Analysis (Deliverable 1.2.1)
- Input from review of economic instruments for restoration (Deliverable 1.2.3)

4. Chapter: Mapping and modelling inputs

This chapter will include:

- Maps of drivers of forest degradation under current and future climate scenarios (Deliverable 2.1.1)
- Maps of ecosystem services (Deliverable 2.1.2)
- Map of priority areas for restoring degraded forests in Europe (Deliverable 2.2.1)

5. Chapter: Review of the draft strategy

This chapter will include internal and external reviews of the draft strategy, specifically:

- Review by Project Partners and Associated Partners (Deliverable 1.4.2)
- Review by internal and external experts and stakeholders, combined with the Joint stakeholders' workshop after the implementation of Pilot actions (Deliverable 2.3.2)