Handbook for the sustainable management and long-term conservation of a narrow endemic habitat type in a limited area of occupancy

The case of the habitat type 9590* Cedrus brevifolia forests (*Cedrosetum brevifoliae*)
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*Cedrus brevifolia forests (Cedrosetum brevifoliae)

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1.1 The distribution of wild species

Nature is under the dynamic influence of environmental change and the human impact. Hence, the environments are patchy, leading to the unremarkable assumption that species’ distributions should also be patchy (Ladle and Whittaker 2011). The mapping and inventorying of wild species’ distribution often represent evidence that their distributions are discontinuous, shaping a network of unequal populations (i.e. metapopulation) with different size and areas of occupancy. Under this frame it is acceptable that the populations (and/or subpopulations) of wild species do not remain constant over time and in spatial scale.

For many decades, the researchers from nature sciences have been fascinated by species with restricted geographical range. This interest has turned to urgency, given that an increasing number of species are headed toward extinction, and the central aim has become to prevent biodiversity loss (Gitzendanner and Soltis 2000). Currently, some highly endangered species have such small ranges that they may be considered to consist of a single population.

Nowadays, it is acceptable that different taxa are likely to exhibit different scales of patchiness, reflecting their own individual niche requirements and the way in which they are distributed across landscape and the extent to which the species concerned interact with the environment (Ladle and Whittaker 2011). In general, assessment of the plant species geographical distribution is fundamental in making sense of the ecological requirements, past history, and interpreting their potential future. The geographical distributions of species are constrained within their potential ranges by (Ladle and Whittaker 2011):

i. Biotic interactions
ii. The vagaries of history
iii. The difficulties of dispersing viable propagules to all areas possessing appropriate conditions for their establishment.

In conclusion, the natural geographical range of species is constrained by their fundamental niche requirements (i.e. abiotic factors such climatic, edaphic or other habitat variables and biotic interactions with competitor species).

1.2. Insights on plant species with narrow geographical range

Species with narrow geographical range are often classified as rare and/or endemic species. Rare species could be naturally occurring in a specific location, occupying only one or a few specialised habitats and/or forming only small population(s) in their range (Thomas et al. 2004; Işik 2011). An endemic species grows naturally only in a single geographic spot (i.e. local endemic) where the population size could be either narrow or relatively large (Primack 2006; Işik 2011). On the other hand, as the size of the geographical range increases, an endemic species may be characterised as: provincial endemic (restricted within the borders of a province), national endemic (growing only within the borders of a nation), regional endemic
(growing only in a certain geographical region) and continental endemic (İşik 2011). At this point, it should be clarified that not all endemic species are rare, in the same way that not all rare species need to be endemic. Hence, there are species that inhabit only one or very few localities for historical, ecological or physiological reasons (Major 1988; López-Pujol et al. 2013) and are usually narrowly distributed (restricted geographical distribution), with their size being disturbed and them being stressed by external factors over long time (Ren et al. 2012). In literature the definition of the population with narrow distribution (geographically restricted) is often linked with the term “narrow species”, “narrowly distributed species”, “population of narrow species” or in case where species are also local endemic the term could be “population of narrow endemic species” etc.

Defining the criteria for the term of narrow, rare and endangered plant species has been broad and general, combining information on spatial and temporal patterns of abundance like: (i) restricted distribution ranges; (ii) small population(s) size; (iii) few populations and (iv) high habitat specialization (Kunin and Gaston 1993; Solórzano et al. 2016). Although there do not exist worldwide quantitative criteria to classify a plant species of narrow geographic distribution (Solórzano et al. 2016), illustration attempts have been generated as guidelines:

- “7 forms of rarity” model (Rabinowitz 1981; Rabinowitz et al. 1986) was developed to examine rarity a commonness of species, incorporating data about local population density, geographical range and variety of habitats occupied by species. Based on the model, a modified approach of an “eight-celled” model was developed for species that are dichotomized for each of the original variables (Yu and Dobson 2000). This new “eight-celled” model assumes that species above and below the median for all three characteristics can be thought of as being relatively common and relatively rare, respectively, in comparison with other species (Table 1.1).

### Table 1.1: Table of “eight-celled” model as this was developed by Yu and Dobson (2000)

<table>
<thead>
<tr>
<th>DISTRIBUTION</th>
<th>Large</th>
<th>Small</th>
</tr>
</thead>
<tbody>
<tr>
<td>POPULATION</td>
<td>high</td>
<td>low</td>
</tr>
<tr>
<td>HABITAT</td>
<td></td>
<td></td>
</tr>
<tr>
<td>broad</td>
<td>A (4)</td>
<td>C (3)</td>
</tr>
<tr>
<td>narrow</td>
<td>B (3)</td>
<td>D (2)</td>
</tr>
</tbody>
</table>

- A placement of rare and endemic species in a three-dimensional (3-D) space was presented by İşik (2011), according to three criteria relevant to the conservation status of a species, namely: geographic range (x-axis), population size (z-axis), and habitat demands (z-axis). This illustration shows the relationships among these criteria in 3-D space (Fig. 1.1). Thus, in this illustration rare and endemic species by definition are located in the very low left corner of the 3-D space.
In practice the narrow geographical range could be applied to those plant species that have as distribution areas of 20-100 km$^2$ or those that occupy less than five locations (Rabinowitz 1981; IUCN 2012; López-Pujol et al. 2013; Solórzano et al. 2016).

1.3. The biology and ecology of narrow species

As mentioned in the beginning of this chapter, biological and ecological aspects, as well as human activities are the main factors, which contribute to the narrow distribution (restricted geographical range) of species. In an effort to investigate these parameters and how they are associated with the narrow distribution of plant species, research studies compared different biological and ecological aspects between congeneric narrow endemic and their widespread plants. Such studies detected significant differential for both habitat and inter-specific attributes between congeneric narrow endemic and their widespread plants (see Gitzendanner and Soltis 2000; Lavergne et al. 2003; Lavergne et al. 2004; Thompson 2005; Ma et al. 2013; etc.). Several authors argued that narrowly distributed species (especially of narrow endemic species) may be characterised by ecological tolerance, higher specialized ecological requirements, lower dispersal abilities and lower reproductive investments than their widespread congener species (Gaston 1994; Kunin and Gaston 1997; Debussche and Thompson 2003; Lavergne et al. 2004; Thompson et al. 2005; Becker 2010). In addition, comparison between congeneric narrow endemic and their widespread plants detected significant differential for both habitat and inter-specific attributes between these two groups (see Gitzendanner and Soltis 2000; Lavergne et al. 2003; Lavergne et al. 2004; Thompson 2005; Ma et al. 2013; etc.). In overview, the narrow endemic species are distinguished from their relevant species with broader distribution, because narrow endemic species:

i. Occur in habitats with a steeper slope (at medium to higher altitudes), high bedrock and block cover, fewer coexisting species, open vegetation with lower and sparser vegetation (versus their widespread congensers species).

ii. Are more adapted to stressful habitats and occur on relatively unfertile substrates, while they compete for resources in more productive habitats. In these habitats, competition may be less, owing to a smaller number of existing species and thus less competition between them.

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*Figure 1.1: Three dimensional (3-D) illustration showing the place of rare and/or endemic species in relation to their geographic range (X axis), population size (Y), and habitat demands (Z) (by Işik 2011; Fig.1).*
iii. Are significantly smaller, but have no differences in traits related to resources acquisition (specific leaf area, leaf nitrogen content, maximum photosynthetic rate) or resource conservation (leaf dry matter content).

iv. Produce fewer and smaller flowers with less stigma-anther separation and lower pollen/ovule ratios and produce fewer seeds per plant. Thus, the low investment in pollen transfer and seed production suggest that local persistence is a key feature of their population ecology.

These biological factors are in large responsible for distribution and dispersal of species. Indeed, narrowly distributed species are often also distinguished as rare species. This rarity could be a result of random events associated with dispersal, colonization and extinction, or may be related to reduced chances of movement due to, for example, geographical barriers (Thompson 2005; Ladle and Whittaker 2011).

Thus, ecology plays a significant role in speciation, as the combination of key evolutionary processes on a species, which occurs in certain environmental conditions (microhabitat), such as natural selection, adaptation, geographic, or reproductive isolation. These factors may lead to evolutionary divergence and the origin of a new species, which is adapted to different/new ecological conditions (Wiens 2004) with narrow distinction of occupancy. In addition, natural selection seems to maintain ecological niches and promotes speciation by limiting ecological divergence, meaning that species that are geographically isolated maintain their initial ecological traits, which at the same time prevent dispersal across the barrier (Wiens 2004). Either way, ecology is connected with speciation and plays a significant role especially in narrow endemic species.

Narrow endemic species are widely believed to harbour low genetic diversity mainly due to the small effective population sizes (Ellstrand and Elam 1993; Forrest et al. 2017), as a result of different genetic aspects, such as founder effects and genetic bottlenecks. At intra-population level, the effect of genetic drift, which results in a severe loss of alleles, is linked to high levels of inbreeding and homozygosity within the occurring population (Templeton and Read 1994; Frankham et al. 2002; Nybom 2004; Godt et al. 2005; López-Pujol et al. 2013). Additionally, the genetic structure of narrow endemic plant populations may be analysed within the framework of the isolation by distance model (Rymer et al. 2002; Solórzano et al. 2016). However, as more research is one on population genetics, these arguments seem not to be generated, since new examples of narrow endemic species are revealed, where high genetic diversity is conserved, together with different genetic structure among their (sub-)populations. Comparisons of genetic diversity between narrow and widespread congeneric species for a series of genetic diversity estimators (i.e. percentage of polymorphic loci, mean number of alleles per locus and observed heterozygosity) has shown that the view of rare species lacking genetic variation is an overgeneralization (Gitzendanner and Soltis 2000). Although narrow species have, on average, less genetic variation than their widespread congeners, there is a wide range in values, and levels of diversity are highly correlated within a genus. In addition to these arguments several autonomous studies showed that narrow endemic species could conserve high levels of genetic diversity due to that the species are not characterised by a founder effect, as well as due to the fact that the species population has not been strongly affected by genetic drift and/or inbreeding processes, because their population sizes may never have been reduced below a certain threshold (e.g. Abies equitrojani — Gulbaba et al. 1998; Pinus rzedsowskii — Delgado et al. 1999; Abies nebrodensis — Parducci et al. 2001; Conte et al. 2004; Nothofagus alessandrii — Torres-Díaz et al. 2007; Cedrus brevifolia — Eliades et al. 2011; Cupressus atlantica – Sękiewicz et al. 2020).

1.3.1. From narrow species to narrow habitat type

In ecology, the term "habitat" refers either to area and resources used by a particular species (the habitat of a species) or the combination of certain animals and plants with their abiotic environment (European Environment Agency 2017). A step further, a habitat or a group of related habi-
Habitats can be considered an ecosystem, in which plant and animal communities, together with the non-living environment, interact to form functional units.

Several classification systems have been created in order to describe habitat types accurately and target conservation efforts. At a European level, the most comprehensive hierarchical approach for describing habitats was made by the European Nature Information System (EUNIS) habitat classification, where the habitat is defined as the “…place where plants or animals normally live, characterized primarily by its physical features (topography, plant or animal physiognomy, soil characteristics, climate, water quality etc.) and secondarily by the species of plants and animals that live there” (European Environment Agency 2017). EUNIS classification works in order to create cross linkages with other classifications and habitat mapping initiatives (i.e. Palearctic classification, Corine biotopes and Corine land cover classes, habitats of Community interest listed in Annex I of the Habitats Directive, etc.). These linkages with equivalent habitat types resulting from different classifications (“crosswalks”) enable the EUNIS classification to serve as a common guide that gathers data from various resources into a single database. Under the framework of European Directive 92/43/EEC on the conservation of natural habitat and of wild fauna and flora (recognised as The Habitats Directive), Annex I includes that “… the natural and semi-natural habitats of community interest that are in danger of disappearing, or that have a small natural range, or that present outstanding examples of typical characteristics of one or more of the biogeographical regions of Europe” (EUR-Lex - 31992L0043 – EN). A habitat could be rich in terms of number of species or perhaps host threatened species, other habitats are connected with cultural or historical values and some are appreciated for their high aesthetic value, while there are habitats with different geographical range and size.

While in literature the term narrow plant species is generally understood as a species with restricted geographical range (as mentioned above), the challenge is how to introduce habitat types with restricted geographical range. The terms of narrow habitat and/or narrow endemic habitat seem to be unsolved, with the critical point being how these terms could be interpreted. On one hand, the narrow habitat could be referring to a habitat type with high patchy distribution into small sites/ spots either within specific provincial or specific regional geographical (biomes) boundaries. The main characteristic of the sites/ spots where the habitat occurs is the similar environmental conditions (i.e. soil characteristics, climate etc.). On the other hand, the terms of narrow habitat and/or narrow endemic habitat could be more clearly defined by a narrow plant species (i.e. annual or perennial plant or shrub or tree species), being the keystone species in formulating a discernible geographically narrow area, within which other plant and fauna species also interact, along with abiotic factors. In both cases above, the range of narrow habitat should be strongly associated with specific environmental factors (i.e. ecology, climate, competition with other species etc.) and/or man- induced pressures in the long-term evolution procedure. Narrow habitat could be any habitat type from coastal sand dunes and island dunes, to freshwater habitats, and the sclerophyllous scrub (Matorral) natural habitat, to different forest habitats.

1.4. Threats and pressures on narrow endemic species (habitat)

A narrowly distributed species is obviously under the effect of demographic stochasticity and/or environmental stochasticity, whilst, in some cases, under the threat of genetic stochasticity (e.g. Hattemer 2005; Nentwig et al. 2007). Demographic stochasticity or random fluctuation of population parameters, such as the distribution of age classes or the sex ratio, lead to the imbalanced age structure, responsible for fluctuations in population size. Demographic stochasticity strongly and directly affects population size and biological conservation. However, demographic stochasticity is not by itself a cause of extinction (Menges 1991). On the other hand, environmental stochasticity is induced by temporal changes in survival and reproduction rates. Unanticipated environmental fluctuations lead to population-size fluctuations whereas, in extreme cases, in population extinction. Extinction danger increases with time, as well as with the reduction of population size.
The principal pressures on biodiversity that were identified in a significant number of countries worldwide (parties who signed the Convention on Biological Diversity - United Nations, 1992) are five (Secretariat of the Convention on Biological Diversity 2010, 2014; Noonan-Mooney and Gibb 2013) and refer to:

1. **Loss, degradation and fragmentation of natural habitats**
2. **Climate change**
3. **Excessive nutrient load and other forms of pollution**
4. **Over-exploitation and unsustainable use**
5. **Invasive alien species**

Habitat destruction due to human activities can lead in most of the cases to negative consequences such as habitat loss or habitat fragmentation, as well as degradation of ecosystem services. Although some habitats are naturally patchy in terms of abiotic and biotic conditions (Wu and Loucks 1995), human activities are widely accepted that have been the reason for the fragmentation of landscapes across the world (Haddad et al. 2015), altering the quality and connectivity of habitats. Some of the most important ecosystems across the planet, such as those with exceptionally high levels of biodiversity (e.g. tropical rainforests, mangroves, savannas etc.), have become severely fragmented, threatening the long-term viability of many species, species richness and ecosystem services (Secretariat of the Convention on Biological Diversity 2010). At the same time habitat loss is considered as the most significant cause of biodiversity loss globally (Noonan-Mooney and Gibb 2013; Secretariat of the Convention on Biological Diversity 2014; Haddad et al. 2015) and can lead to rapid extinction of some species especially those with narrow distribution (Millennium Ecosystem Assessment 2005). Anthropogenic activities that can lead to habitat loss and/or fragmentation are deforestation for the creation of pasture and agricultural land, logging, urbanization, road network expansion, aquaculture, tourism, mining (Secretariat of the Convention on Biological Diversity 2014; Corlett 2016) and many others specific to certain areas of the globe. Fragmentation of habitats not only threatens species richness and habitat connectivity but also threatens the ability of ecosystems to adapt to climate change.

Climate change is one of the main threats that biodiversity faces; based on literature it is already having an impact on biodiversity, and is anticipated to progressively become a major driver of biodiversity loss and ecosystem change by 2050 (Secretariat of the Convention on Biological Diversity 2014). Increased urbanization, increased gas emissions and other anthropogenic activities are having a negative impact on global climate with obvious results such as warming temperatures, more frequent extreme weather events, changing patterns of rainfall and drought, and increase in ocean acidification. Such changes have from negative to disastrous consequences to biome, such as alteration in leafing and flowering season, leaf dropping or migration timing, changes in food chain, species extinction due to adaptation problems, extension of distribution range of species towards higher latitudes or higher altitudes and survival of alien species over native species, as well as many more other consequences (Secretariat of the Convention on Biological Diversity 2010; Hijioka et al. 2014; Ge et al. 2015). Forest fires is yet another disastrous consequence seriously affecting the potential resilience and resistance of ecosystems. Indeed such fires are expected to potentially increase in frequency and aggressiveness, owing to climate change.

Pollution is also related to anthropogenic activities and can be nutrient pollution, mainly from phosphorus and nitrogen, or other forms of pollution from chemicals, pesticides and plastics; it threatens not only terrestrial environments but aquatic as well (Noonan-Mooney and Gibb 2013; 1)

1Habitat loss is the transformation or modification of natural environments in order to serve humanity (Noonan-Mooney and Gibb 2013).
Habitat fragmentation is the reduction of a continuous habitat into several smaller spatially isolated remnants (Young et al. 1996; Haddad et al. 2015).
Ecosystem degradation is the persistent decrease in the capacity of an ecosystem to deliver services (Millennium Ecosystem Assessment 2005).
Secretariat of the Convention on Biological Diversity 2014). Accumulation of excess nutrients due to unsustainable agriculture (use of fertilizers) or aquaculture/ fisheries (overfeeding) can lead to eutrophication, meaning increase of algae in water bodies (e.g. wetland, marine, coastal environments, lakes etc.). Air pollution comes from transport and industrial emissions, plastics can cause contamination of large terrestrial and sea areas affecting the living organisms, oil spills are another reason of damage of marine ecosystems, and waste of sewage and industrial waste water affect the quality of water bodies (Secretariat of the Convention on Biological Diversity 2010, 2014).

Overexploitation and unsustainable use of natural resources is another main threat to biodiversity which is however more intense in specific areas of the globe. It is actually the removal of biodiversity faster than it can recover, resulting in extinction of wild species (harvesting exceeds reproduction), either using destructive harvesting practices or not (Secretariat of the Convention on Biological Diversity 2010; Noonan-Mooney and Gibb 2013). Notably, in literature overexploitation is referred as the second most important threat to plant species, following habitat loss (Corlett 2016). At the same time, it is referred as the major pressure being exerted on marine ecosystems (Secretariat of the Convention on Biological Diversity 2010) and it is related mainly to overfishing and unsustainable harvesting of aquatic invertebrates such as sea cucumbers (González-Wangüemert et al. 2018). Apart of these, it is also related to unsustainable forest management, overhunting, destructive and/or over harvesting of certain wild plant species for horticulture and intensive crop monoculture (Noonan-Mooney and Gibb 2013).

Invasive alien species (IAS) are non-native species (animals, plants, fungi and micro-organisms) whose introduction and/or spread outside their natural past or present ranges pose a threat to biodiversity (Kettunen et al. 2008). Increased world trade has been a key indirect driver of the introduction of IAS, as well as transport and tourism, in addition to climate change which can benefit their invasion (Secretariat of the Convention on Biological Diversity 2010). IAS can alter the composition and structure of local ecosystems and their services, as well as the evolutionary pathways of native species, directly or indirectly (Mooney and Cleland 2001). Apart from the environmental damage, IAS are considered drivers for economic damage or can affect human health (Noonan-Mooney and Gibb 2013).

The extent of consequences which a threat or pressure has on a habitat or species depends on several factors, such as the range of its distribution, its status (i.e. endemic, indigenous, rare), as well as the degree of its isolation (e.g. island habitats versus mainland/ inland habitats). For example, species that form island endemic habitats are much more sensitive to anthropogenic disturbances due to their limited range and small population size, than inland or non-endemic species (Frankman 1998). At the same time, based on geographic trends, it is shown that islands not only are regions with significant levels of endemism - “islands have more than 35% of the world’s vascular plant species. There are around 50,000 insular endemic plants of which 20,000 are estimated to be threatened with extinction” (Sharrock et al. 2014) - but also that islands have been the recipients of the largest proportional numbers of invader on the globe (Mooney and Cleland 2001). Thus, insularity and its associated vulnerability could in itself represent a threat to plant species inhabiting islands, especially to endemics with highly restricted distributions (Rumeu et al. 2014).

Focusing on conifers, apart from habitat loss or degradation, IAS, climate change, pollution and overexploitation that have already been mentioned as major threats of biodiversity, other threats specified for conifers are overexploitation through logging, anthropogenic fires combined with overgrazing, conversion of forested ecosystems to pasture, exploitation of other natural resources such as resin, edible seeds or medicines or genetic depletion through selective removal of individuals (Farjon and Page 1999). Paying attention to the mountainous areas of the Mediterranean region which played a significant role during glacial periods since they provided refuge to certain conifer taxa (e.g. Abies, Cedrus, Cupressus, Juniperus and Pinus), these taxa have been under pressure or face specific threats and are now of considerable conservation concern (Esteban et al. 2010).
1.5. The handbook from the project LIFE-KEDROS

The current handbook has been prepared and published by the LIFE-KEDROS project, a project co-funded by LIFE programme. The project aimed to ensure the medium and long-term preservation of the population of endemic coniferous tree species Cedrus brevifolia (Cyprus cedar) in good conservation status. The forest of Cyprus cedar according to the European Directive 92/43/EEC shapes the habitat type of "9590 *Cedrus brevifolia forests (Cedrosetum brevifolia)" that is included in Annex I, where it is characterised as a priority habitat type, as well. The habitat type 9590* has narrow distribution in an area of ~264 ha, and its distribution is patching into five sides (see Chapter 8).

The handbook aims to provide the basis for cost-efficient replication and/or transfer of effective sustainable management of tree species -most preferably coniferous- with narrow distribution, which occur and/or form habitat types in the Mediterranean region. This handbook consisted of two main parts, where the first part presented an integrated conservation vision of narrow habitat types and the second part summarized the outcomes and best practices on cedar forest management (habitat type 9590*), as these will result by the project’s implementation.

The Mediterranean basin is one of the 36 biodiversity hotspots of Planet Earth. These are specific biogeographic terrestrial regions with significant levels of endemism (at least 1500 vascular plant species) which experience exceptional habitat loss (lost at least 70% of their primary native vegetation) (Gorenflo et al. 2012; CEPF - Critical Ecosystem Partnership Fund 2017). The Mediterranean basin is considered one of the most important areas on Earth for endemic plants and the high levels of endemism place the region in the planet’s biodiversity hotspots map. More specifically, the Mediterranean basin is the second largest biodiversity hotspot (CEPF - Critical Ecosystem Partnership Fund 2017) accommodating 30,000 plant species (Fady-Welterlen 2005), 80% of Europe’s endemic plant species (Blondel and Aronson 1999), as well as some of the endemic habitats of Europe out of a total of 233 natural habitat types (Interpretation Manual – EUR28, 2013) which occur in the continent. According to bibliography, around 60% of the native Mediterranean taxa are endemic to the region (Greuter 1991; IUCN 2008) and a very significant remark is that 60% of this endemism refers to narrow endemic species (Thompson et al. 2005).

The high levels of endemism and flora richness in the Mediterranean basin is a consequence of geological, climatic and historical processes for millions of years. The geographic location of the Mediterranean basin between Europe, Asia and Africa, well-known as the continents of Laurasia and Gondwana of past geological periods, the tectonic activity of the African and Eurasian plates, intensive volcanic activity, the Messinian Salinity Crisis, the sea-level fluctuations during glacial and interglacial periods were all associated with the appearance of islands and repeated cycles of connection and isolation from neighboring mainland or insular areas (Sfenthourakis and Triantis 2017; Feliner et al. 2005). These events were on one hand the beginning of the creation of islands and on the other hand the driving force for the creation of refugia during plant migrations and their later isolation, which led to evolutionary events of adaptation and speciation (Greuter 1991; Valdes Castrillon and Hernandez Bermejo 1995).

There are eighteen (18) Mediterranean conifer species together with their related forest habitat types, included in the Red List of Threatened Plant Species proposed by IUCN (Conte et al. 2004). Examples of narrow habitats of conifer species which are endangered are presented in Table 1.2.
### Table 1.2: Examples of taxa with narrow distribution, in Europe and the Mediterranean region.

<table>
<thead>
<tr>
<th>No.</th>
<th>Taxon</th>
<th>Endemism</th>
<th>EU Habitats Directive 92/43/EEC</th>
<th>Threats</th>
<th>AOO (km²)</th>
<th>Global Conserv. status (IUCN criteria)</th>
<th>EU Habitat type</th>
<th>Natura 2000 sites designated for habitat</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Abies nebrodensis</td>
<td>Italy</td>
<td>V (*)</td>
<td>Overexploitation, fire</td>
<td>4</td>
<td>CR</td>
<td>9220 *Apennine beech forests with Abies alba and beech forests with Abies nebrodensis</td>
<td>105</td>
</tr>
<tr>
<td>2</td>
<td>Abies numidica</td>
<td>Algeria</td>
<td></td>
<td>Fire, logging, grazing</td>
<td>8</td>
<td>CR</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>Cedrus brevifolia</td>
<td>Cyprus</td>
<td>V (*)</td>
<td>Fire, climate changes, infestations</td>
<td>8</td>
<td>VU</td>
<td>9590 *Cedrus brevifolia forests (Cedrosetum brevifoliae)</td>
<td>1</td>
</tr>
<tr>
<td>4</td>
<td>Zelkova sicula</td>
<td>Italy</td>
<td></td>
<td>no sexual reproduction, logging, harvesting, overgrazing, fires</td>
<td>8</td>
<td>CR</td>
<td></td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Cytisus aeolicus</td>
<td>Italy</td>
<td>V (*)</td>
<td>changes in land use, fires, volcanoes</td>
<td>12</td>
<td>EN</td>
<td>5330 Thermo-Mediterranean and pre-desert scrub (PAL.CLASS.: 32.26 - Thermo-Mediterranean</td>
<td>5</td>
</tr>
<tr>
<td>Natura 2000 sites designated for habitat&lt;sup&gt;a&lt;/sup&gt;</td>
<td>EU Habitat type&lt;sup&gt;b&lt;/sup&gt;</td>
<td>Global Conserv. status (IUCN criteria)&lt;sup&gt;c&lt;/sup&gt;</td>
<td>AOO&lt;sup&gt;d,e&lt;/sup&gt; (km&lt;sup&gt;2&lt;/sup&gt;)&lt;sup&gt;f&lt;/sup&gt;</td>
<td>Threats&lt;sup&gt;g&lt;/sup&gt;</td>
<td>EU Habitats Directive 92/43/EEC&lt;sup&gt;g&lt;/sup&gt;</td>
<td>Endemism&lt;sup&gt;h&lt;/sup&gt;</td>
<td>Taxon</td>
<td>No.</td>
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</tr>
<tr>
<td>broom fields (retamares)&lt;sup&gt;b&lt;/sup&gt;</td>
<td>CR</td>
<td>14.58</td>
<td>seed collecting, grazing, climate change</td>
<td>Fire, deforestation, forest degrade.</td>
<td>Annex I</td>
<td>Morocco</td>
<td>Cupressus dupreziana var. atlantica</td>
<td>6</td>
</tr>
<tr>
<td></td>
<td>EN</td>
<td>28</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Abies pinsapo var. marocana</td>
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</tr>
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<td></td>
<td>EN</td>
<td>28.7</td>
<td>Wild fire, infestations, climate change</td>
<td>Overgrazing, soil erosion, wild fire, geographical isolation</td>
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<td>Abies pinsapo var. pinsapo</td>
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<tr>
<td></td>
<td>EN</td>
<td>64</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Zelkova abelicea</td>
<td>9</td>
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<tr>
<td></td>
<td>EN</td>
<td>76</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Pinus mugo subsp. rotundata</td>
<td>10</td>
</tr>
<tr>
<td>No.</td>
<td>Taxon</td>
<td>Endemism$^{1,2}$</td>
<td>EU Habitats Directive 92/43/EEC$^3$</td>
<td>Threats$^{1,2}$</td>
<td>AOO (km²)$^{1,2}$</td>
<td>Global Conserv. status (IUCN criteria)$^1$</td>
<td>EU Habitat type$^{3,4}$</td>
<td>Natura 2000 sites designated for habitat$^3$</td>
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<td></td>
<td></td>
<td></td>
<td>Annex I</td>
<td>Annex II</td>
<td>Wild fire, human activities, infestations, logging$^4$</td>
<td>100 – 200</td>
<td>EN</td>
<td></td>
</tr>
<tr>
<td>11</td>
<td><em>Picea omorika</em></td>
<td>Bosnia, Herzegovina, Serbia</td>
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<tr>
<td>12</td>
<td><em>Pinus nigra</em> subsp. <em>dalmatica</em></td>
<td>Croatia</td>
<td>V (*)</td>
<td></td>
<td>habitat degradation, grazing$^3$</td>
<td>300</td>
<td>EN</td>
<td>9530 <em>(Sub-)</em> Mediterranean pine forests with endemic black pines$^4$</td>
</tr>
<tr>
<td>13</td>
<td><em>Cupressus dupreziana</em> var. <em>dupreziana</em></td>
<td>Algeria</td>
<td></td>
<td></td>
<td>climate change, wild fire, logging, tourism, grazing</td>
<td>&lt; 500</td>
<td>CR</td>
<td></td>
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<tr>
<td>14</td>
<td><em>Abies cilicica</em> subsp. <em>isaurica</em></td>
<td>Turkey</td>
<td></td>
<td></td>
<td>Overgrazing, tourism, climate change, infestations</td>
<td>850</td>
<td>VU</td>
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<tr>
<td>15</td>
<td><em>Phoenix theophrasti</em></td>
<td>Greece</td>
<td>V (*)</td>
<td>V</td>
<td>Residential and commercial development, ecosystem degradation,</td>
<td>31</td>
<td>NT</td>
<td>9370 *Palm groves of Phoenix</td>
</tr>
<tr>
<td>No.</td>
<td>Taxon</td>
<td>Endemism(^{1,2})</td>
<td>EU Habitats Directive 92/43/EEC(^1)</td>
<td>Threats(^{1,2})</td>
<td>AOO (km(^2))(^{1,2})</td>
<td>Global Conserv. status (IUCN criteria)(^2)</td>
<td>EU Habitat type(^{3,4})</td>
<td>Natura 2000 sites designated for habitat(^2)</td>
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<td>Annex I</td>
<td>Annex II</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>overgrazing, forest fire</td>
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</tbody>
</table>


(species and/or habitats marked with an asterisk (*) are of priority which means that these are in danger of disappearance)


\(^{6}\) Habitat type 5330 have been designated in 1253 Natura 2000 sites in European Union. However, the species *Cytisus aequalis* participates in subtype "PAL. CLASS. 32.26 Thermo-Mediterranean broom fields (retamares)" which occurs only in five Natura 2000 sites in Italy. (sources: Interpretation Manual of European Union Habitats - EUR28 and [https://eunis.eea.europa.eu/](https://eunis.eea.europa.eu/))

\(^{7}\) The conservation status of the subspecies based on IUCN criteria is Endangered although the species' (*Pinus mugo*) conservation status is LC. In Annex I habitat type with code 4070 *Bushes with Pinus mugo* and *Rhododendron hirsutum* (*Mugo-Rhododendretum hirsuti*), for which 277 Natura 2000 sites have been designated in EU, participates the species and there is no information how many of these sites host the subspecies. (sources: [https://www.iucnredlist.org/](https://www.iucnredlist.org/), Interpretation Manual of European Union Habitats - EUR28 and [https://eunis.eea.europa.eu/](https://eunis.eea.europa.eu/))

The conservation status of the subspecies based on IUCN criteria is Endangered (EN) although the species' (*Pinus nigra*) conservation status is Least Concern (LC). In Annex I habitat type with code 9530 *Sub-)Mediterranean pine forests with endemic black pines, for which 267 Natura 2000 sites have been designated in EU, participates the species and there is no information how many of these sites host the subspecies. (sources: [https://www.iucnredlist.org/](https://www.iucnredlist.org/), Interpretation Manual of European Union Habitats - EUR28 and [https://eunis.eea.europa.eu/](https://eunis.eea.europa.eu/)).

\(^{8}\) The conservation status of the subspecies based on IUCN criteria is Endangered (EN) although the species' (*Pinus nigra*) conservation status is Least Concern (LC). In Annex I habitat type with code 9530 *Sub-)Mediterranean pine forests with endemic black pines, for which 267 Natura 2000 sites have been designated in EU, participates the species and there is no information how many of these sites host the subspecies. (sources: [https://www.iucnredlist.org/](https://www.iucnredlist.org/), Interpretation Manual of European Union Habitats - EUR28 and [https://eunis.eea.europa.eu/](https://eunis.eea.europa.eu/)).
1.6. References


Debussche M., Thompson J.D. (2003) Habitat differentiation between two closely related Mediterranean plant species, the endemic Cyclamen balearicum and the widespread C. repan-


2.1. The Global policies

The United Nations Economic Commission for Europe (UNECE) and the Food and Agriculture Organization of the United Nations (FAO) (UNECE/FAO) provide a definition of the Forest as a “Land with tree crown cover (or equivalent stocking level) of more than 10% and area of more than 0.5 ha. The trees should be able to reach a minimum height of 5 m at maturity in situ. A forest may consist either of closed forest formations where trees of various storeys and undergrowth cover a high proportion of the ground, or open forest formations with a continuous vegetation cover in which tree crown cover exceeds 10%” (Gold 2003).

Forests cover almost 4 billion hectares (ha) or approximately 30% of the world’s land surface, where this cover has been steadily decreasing over the past 20 years (The World Bank 2019a and 2019b).

The way that people and governments perceived forests and their use has changed in the course of history. In the past, governments designed centralized and sectoral policies in order to generate revenue and foreign exchange for national economic development, thus treating forests as reservoirs of new land for cultivation or areas protected as nature reserves. In recent years, society’s requirements and related strategies have shifted, requiring policies to integrate forests in rural development efforts, balancing economic and environmental needs (FAO 1995).

The shift was further documented in 1992, when the three Rio Conventions (the Convention on Biological Diversity (CBD), the United Nations Convention to Combat Desertification (UNCCD) and the United Nations Framework Convention on Climate Change (UNFCCC)) acknowledged the important contribution of forests to their goals (The RIO conventions Actions on Forests 2012). In a nutshell:

- **CBD**: Forests are addressed in various ways with CBD, by programmes/ actions of work elaborated through the Strategic Plan for Biodiversity 2011 – 2020, the Aichi Biodiversity Targets and the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization.

- **UNCCD**: The UNCCD indicates that deforestation and the resultant desertification adversely affect a number of factors, including land productivity, human and livestock health and economic activities; its actions include (amongst others) the National action programmes (NAPs) which contain the national strategies for land and drought-related policies toward combating desertification/ land degradation and drought effects.

- **UNFCCC**: The UNFCCC recognizes the importance of forests in mitigating climate change, as they represent a significant global carbon stock. The policies and methodological guidance developed include: (i) REDD-plus, which aims at reducing emissions from deforestation and forest degradation and promoting the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries, (ii) LULUCF (Land Use, Land-Use Change and Forestry) in developed countries and (iii) Afforestation and reforestation project activities under the clean development mechanism.
2.2. The European policies

In Europe, forests cover 157 million ha and provide a vast array of products, socio-economic benefits and ecosystem services. The legislations for these forests are dynamic and innovative, where practically all European countries, have enacted new forest laws or have significantly amended existing ones. Different aspects of change include adaptation of legislation to changing social demands, multifunctional policy objectives, transfer or delegation of constitutional competencies in forestry matters, new regulative and incentive instruments, increasing emphasis on information and process-steering processes, new strategies to support forest owners, etc. There is however, no common forest policy, which is apparent by the differences in the form and content of the national legislations (e.g. state forest services, degree of detail, role of owners, etc.) (Bauer et al 2004).

In the European Union (EU), forests and other wooded land cover 182 million ha, that consist 5% of world’s forests and represent more than 42% of EU land area1.

Within EU, various legislations have been set up to preserve forests and their genetic diversity. The EU Forestry Strategy2 adopted in 1998 puts forward as its overall principle the application of sustainable forest management and the multifunctional role of forests. In 2005, the Strategy was reviewed and the Commission presented its EU Forest Action Plan in 2006. In 2013, the new EU Forest Strategy for forests and the forest-based sector was adopted, while a Multi-annual implementation plan was adopted in 2015. Table 2.1 presents a timeline of the forest Policy and related decisions and communication in the European Union.

The preservation is further facilitated through instruments such as the European Forest Genetic Resources Programme (EUFORGEN). This is an instrument of international cooperation promoting the conservation and appropriate use of forest genetic resources in Europe, established in 1994, to implement Strasbourg Resolution 2 (adopted by the first Ministerial Conference of the FOREST EUROPE process, held in France in 1990) (de Vries et al 2015).

In addition, the crucial role of forests in biodiversity protection (and provision of ecosystem services) is emphasized by the fact that half of the Natura 2000 network is made by forest areas (20% of EU forest area is included in this network). The Natura 2000 network derives from the implementation of the Birds and Habitats Directives (79/409/EEC and 92/43/EEC, respectively) and their amendments, which have in their turn derived from the implementation of the 1979 Bern Convention (Convention on the Conservation of European Wildlife and Natural Habitats).

Table 2.1: Timeline of forest policy/decisions in the European Union.

<table>
<thead>
<tr>
<th>YEAR</th>
<th>ACTIONS</th>
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<tbody>
<tr>
<td>1998</td>
<td>The EC presents a Communication on a Forestry Strategy for the EU.</td>
</tr>
<tr>
<td>2005</td>
<td>EC presents to the Council and the European Parliament a Communication reporting on the implementation of the EU Forestry Strategy.</td>
</tr>
<tr>
<td>2006</td>
<td>Adoption of the EU Forest Action Plan (builds on the 2005 report and consequent conclusions by the Council).</td>
</tr>
</tbody>
</table>

1 EU Forests and Multi-annual implementation plan: https://ec.europa.eu/environment/forests/index_en.htm
2010 The Commission adopts the Green Paper on forest protection and information.


2013 The Commission adopts a new EU Forest Strategy, accompanied by a Staff Working Document and takes note of a Blueprint for the EU forest-based industries.


2015 The Commission adopts a Multi-annual Implementation Plan of the new EU Forest Strategy.

2018 The Commission adopts the Report on the progress in the implementation of the EU Forest Strategy.

2019 Council Conclusions on the mid-term review of the EU Forest Strategy.

Generally, forests in the EU are affected by many Community policies and initiatives arising from diverse EU sectoral policies. Besides the Birds and Habitats Directives already mentioned, such policies include:

- Council Regulation (EEC) No 3528/86 of 17 November 1986 on the protection of the Community's forests against atmospheric pollution,
- Council Regulation (EEC) No 2080/92 of 30 June 1992 instituting a Community aid scheme for forestry measures in agriculture,
- Council Regulation (EEC) No 2158/92 of 23 July 1992 on protection of the Community's forests against fire,
- The EU 2020 Biodiversity Strategy.

Furthermore, it is noted that the increasing attention to forests relates to the protection of biodiversity, climate change impacts and energy policies, as well as the services they provide, such as public amenities, regulators of climate and local weather, sources of clean water and protection against natural disasters.

2.3. Guidelines on the management of forest protected areas

Even though there are no common policies for the management of forests (as well as for narrow habitats with endemic species), the International Union for Conservation of Nature (IUCN) provides guidelines that centre around five related concepts. These are: (i) Protected areas, (ii) Forests, (iii) Forests as defined for the purposes of Forest Protected Areas, (iv) Forest Protected Areas, and (v) Other conserved forests. These guidelines utilize the IUCN Protected Area Management Category System (Table 2.2) that constitute one instrument among several for the responsible management of forest resources. They should also be supplemented with other types of forest management and protection that are not contained within the IUCN definition of a protected area (Dudley and Phillips 2006).

IUCN also provides the definition for a Forest Protected Area as “a subset of all protected areas that includes a substantial amount of forest as defined for the purposes of Forest Protected Areas. This may be the whole or a part of a protected area”.

Cases like the one of the narrow endemic species of Cedrus brevifolia (and the habitat type 9590 *Cedrus brevifolia forests (Cedrosetum brevifoliae) it creates) can benefit from the use of such guidelines. In this case, the area with the habitat occurs within the boundaries of a larger forest (Dasos Pafou), where the requirements/management needs of its various habitat types could be addressed in a separate manner.
Table 2.2: The six categories of the IUCN Protected Area Management Category System.

<table>
<thead>
<tr>
<th>CATEGORY</th>
<th>DESCRIPTION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ia</td>
<td>Area managed mainly for science or wilderness protection</td>
</tr>
<tr>
<td>Ib</td>
<td>Area managed mainly for wilderness protection</td>
</tr>
<tr>
<td>II</td>
<td>Area managed mainly for ecosystem protection and recreation</td>
</tr>
<tr>
<td>III</td>
<td>Area managed mainly for conservation of specific natural features</td>
</tr>
<tr>
<td>IV</td>
<td>Area managed mainly for conservation through management intervention</td>
</tr>
<tr>
<td>V</td>
<td>Area managed mainly for landscape/seascape conservation or recreation</td>
</tr>
<tr>
<td>VI</td>
<td>Area managed mainly for the sustainable use of natural resources</td>
</tr>
</tbody>
</table>

2.4. References


3.1. Guides on conservation biology

Under the frame of nature conservation, an attempt is being made to protect the biodiversity that has been produced by evolution over the previous 3.5 billion years on Earth. Averting global biodiversity loss has been the target of international conventions, European directives and national strategies. Over 150 national governments have signed a treaty – Convention on Biological Diversity (CBD) – committing themselves to biodiversity conservation (United Nations Environment Programme 1992). In general, the term of biodiversity as a United Nations Environment Programme (1992) includes diversity within species (genes), between species and of ecosystems (which are defined as the three major diversity levels). Sustainable management of nature (wild populations) presupposes understanding their complexity of biodiversity in order to undertake correct decisions and implement robust actions for the conservation of all three fundamental levels of biodiversity (ecosystems, species and genes). In Council Directive 92/43/EEC conservation is defined as a series of measures required to maintain or restore the natural habitats and the populations of species of wild fauna and flora at a favourable status. However, this argumentation could be the baseline for the conservation effort and measures that should be designed and applied in the field.

Relevant literature highlights the need for accumulating multidimensional knowledge regarding the biology and characteristics of both plant species and ecosystems, in order to develop sound measures for their conservation (e.g. Lozano et al. 2005; van Dyke 2008). Thus, any conservation strategy should focus on progress in terms of enhancing the reserve estate, managing the recovery of threatened species and ecological communities, and putting in place frameworks to manage ecological threats to biodiversity, as mentioned in the Australian Capital Territory of 2013. These arguments seem to be more appropriate for ecosystems consisted of tree species with narrow geographical and/or ecological distribution. Such ecosystems/habitats seem to be more prone to demographic processes, supporting the assumption that the two main types of threats causing extinction are deterministic and stochastic (Caughley 1994).

Nowadays, pressures and threats in nature, such as habitat loss, landscape alteration and extinction, as well as global environmental change, at the species, community and even at the ecosystem levels, have resulted in conservation biologists struggling to devise methods and tools for species protection and preservation (Phartyal et al. 2002). The two commonly used methods for conserving the biodiversity (genetic resources, plants and ecosystems) are: (i) in situ conservation method, which allows evolution to continue within the area of natural occurrence, and (ii) ex situ conservation method, providing a higher degree of protection to germplasm compared to in situ conservation (e.g. Frankel and Soule 1981; Burney and Burney 2007; Eliades et al. 2019 etc.). In the last two decades the inter situ conservation method was adopted in several conservation programmes, since it acts as a link between the in situ and ex situ strategies, and aims to bridge the gaps that appear. By definition inter situ conservation is an alternative method focusing on the establishment of species by reintroducing locations outside the current range but within the recent past range of the species (Burney and Burney 2007). In overview, the three conservation
methods should be implemented complementary to each other, since they have advantages and disadvantages; as these were elegantly summarised by Burney and Burney (2009):

**In situ:** Conservation applied on-location in the existing setting and range of species, ensuring the species dynamic conservation within its natural distribution area.

*Advantages:* Species are supposedly adapted to their existing setting. The existence of other species in the setting can affect survival and ecological function.

*Disadvantages:* Possibly difficult access to the site, costly maintenance of site, degraded site. Likely difficulty to address several issues, such as rarity of species, as well as stochastic events (storms, disease outbreaks, and human over-utilization).

**Ex situ:** Man-induced conservation outside species’ natural distribution. Located in controlled environments (botanical gardens, zoos, gene banks, propagation facilities).

*Advantages:* Provides a second line of security for rare species, which can thrive in the absence of whatever threats exist in their natural environment. Easy access to biodiversity by interested parties.

*Disadvantages:* Interferes with the natural course of evolution, either by halting or by diverting it. Species run the risk of becoming domesticated and unable for reintroduction. High cost of *ex situ* maintenance of species may pose restrains on the number of species and/or individuals to be maintained in such settings.

**Inter situ:** Establishing a species, by reintroducing it to locations outside the current range but within the recent past range of this species.

*Advantages:* An intermediate situation between in situ and *ex situ*. Access to species and protection of them can be better than in situ, whereas production/maintenance cost can be lower than *ex situ*. Evolution can be less disrupted, with genetic diversity being better preserved or even enhanced within new populations.

*Disadvantages:* Inadequate knowledge of community and ecosystem function may hinder restructuring of multi-layered networks. Follow up is needed in order for reintroductions to be successful, while long-term monitoring is needed for survival and development of reconstructed populations. Conservation of rarest species may be inhibited by legal complications.

The existing conservation methods should be complementary to each other in species conservation to the wild in a fast-changing world with demographic changes, changes in land use and climate change. Hence, aiming towards sustainable management and a holistic conservation strategy, an integrated approach (conservation measure) for conservation biology purposes should be designed (see Guerrant et al. 2004; Villard and Jonsson 2009). Notably, any conservation measure could be characterized either as a static or as a dynamic conservation measure. The dynamic in situ conservation measures do not disregard evolutionary processes but, rather, allow for adaptation and other biological changes within the natural environment. Thus, forest genetic resources occur within a natural system in which the evolutionary forces that generate and maintain genetic diversity, are allowed to act and alter allele and gene frequencies (Lefèvre et al. 2013). On the other hand, dynamic *ex situ* conservation measures (off-site maintenance) also cause adaptation to the environment during evacuation, thus allowing both the selection and adaptation of collections of individuals to the new environment from the very beginning. In this way, long-term spontaneous mating and reproduction foster evolutionary forces, allowing them to act and modify allele and gene frequencies (Hattemer, 1997; Lefèvre et al. 2013). Therefore, the design and implementation of any conservation strategy for wild organisms should be characterised by an integrated approach (as mentioned above), taking in mind the need for ensuring the dynamic characteristics of conservation strategies (see Fig. 3.1).
The conservation strategy implemented in each case should be in line with the existing knowledge, the identification of pressures or threats that a component of biodiversity faces or will face in the future, as well as international and national legislation or other legal frameworks. The design and implementation of specific conservation strategies (processes and measures) for the conservation of biodiversity are included in international conventions and protocols, as the Convention on Biological Diversity (CBD), the conservation and the monitoring of the genetic resources through the Nagoya Protocol (in 2010), halting biodiversity loss through decisions embodied in European Directives such as Habitats Directive for the conservation of natural habitats and of wild fauna and flora (92/43/EEC) and Birds Directive (2009/147/EC) for the conservation of wild birds, the Sixth Community Environment Action Programme, the European Biodiversity Strategy and the European forest genetic resources programme (EUFORGEN), as well as the International Treaty on Plant Genetic Resources for Food and Agriculture (Cordonier Segger and Phillips 2015; Eliades et al. 2019).

3.2. Setting conservation targets for narrowly distributed habitats

Generally speaking, the goal of “sustainable forestry” is the focus point for the forest managers regardless of the geographical range of the target species (forest/habitat). The goal of “sustainable forestry” dates back to 1713 in Germany when sciences tried to address the problem of sustainable wood production in the context of the local industry’s needs (Schuler 1998). Years later, in 1987, “sustainable development” was defined as a three-legged stool supported by economic, social, and environmental components or “legs” (e.g. Goodland and Daly 1996). About the environmental component, this can be regarded as encompassing the elements of biodiversity, though several biodiversity elements can be argued as belonging to the economic and social components (Angelstam et al. 2004). Sustainable forest management is more easily understood when analysed into several criteria, namely biodiversity, social issues, and economics (e.g. Higman et al. 1999; Duinker 2001, Rametsteiner and Mayer 2004). Of course, the financial issue owing to the harvesting of tree species (keystone) is not a priority when dealing with narrow habitats.

Figure 3.1: A range of conservation measures and the relative ongoing effort on marginal resources needs. (copy by Maunder et al. 2004; Fig.1.1).
For narrow endemic tree species (that are characterised as keystone species for a specific ecosystem -habitat/forest-), the vision on conservation should focus on defining conservation targets (as these are generated for wider forest ecosystems – Villard and Jonsson 2009) that ensure the dynamic ecological processes and ecological functions for the habitat/forest. A conservation target can be defined as any quantitative objective determined by using empirical data or realistic models to adjust management intensity with the purpose of maintaining biodiversity (Villard and Jonsson 2009). In any case, the conservation targets for a narrow habitat should link with ecological processes and demographic parameters of the current target habitat, while any forest management objective will eventually affect the composition and configuration of habitat/forest structure (Baskent 2009). However, in the case of a narrowly distributed habitat/forest, the harvest level should be very low and should be implemented only in an effort to support the natural regeneration and competition of keystone species. In addition, for the conservation of such habitat types, with ecological integrity as a priority, a realistic allowable cut is set to minimize volume loss, including loss owing to wildfire, insects, diebacks due to the impact of climate changes etc.

The identification of threats and pressures that negatively affect the resilience of the targeted habitat is a fundamental issue for developing and designing the conservation strategies and the actions/measures that should be included in each strategy. The design of any conservation targets (strategy) for a narrow habitat/forest type needs to be characterised by different levels of temporal and spatial domains (Jonsson and Villard 2009). Several authors have debated the need for a dynamic characterisation of conservation efforts, which does not only preclude evolutionary processes but also allows for adaptation and other biological changes in natural environments. Jonsson and Villard (2009) illustrated this general argument using four general guide conservation ambitions (Fig. 3.2), where each setting represents different levels of conservation ambition (see also Angelstam et al. 2004).

In order for the conservation ambition for a narrow habitat type to be holistic and at the same time to strengthen the ecological connectivity with the rest elements of biodiversity (abiotic and biotic factors) in the wider ecosystem, adaptability needs to be supported. This argument for the strengthening of ecological connectivity is based on the assumption that ecological connectivity is widely regarded as a critical element in assisting habitat adaptation to environmental change. It is obvious that, for defining conservation goals, specific quantitative conservation targets need to be designed. The conservation targets should combine different levels:

- **Genetic conservation targets**: With the crucial factor being that extinction risk increases when genetic variation is lost (Frankham 2003; Ouborg et al. 2006), these targets should aim to conserve genetic variation within a population, which is a prerequisite for heterozygosity.
that enhances the fitness at the population level. In any case the genetic conservation targets for narrow species should support, in general, the maintenance of high allelic richness within the targeted population (Eliades et al. 2019), and not the specific conservation of rare allele supporting the preservation of a particular trait expression, preservation of maximum variation and preservation of adaptability (Finkeldey and Hattemer 2007, Eliades et al. 2019).

- **Population-level targets:** Focusing on habitat conditions predicting the occurrence of species (via habitat suitability models – used to map the probability of occurrence of particular species in the landscape with fairly limited data) or on the long-term persistence of local populations (population viability model – incorporating temporal aspects of the dynamics of the populations, allowing under specific conditions the putative outcomes of different management scenarios on the central species) (Jonsson and Villard 2009; Fady et al. 2016).

In any case, empirical data support that population sizes that ranged between 500-5000 reproductive individuals, contributed to reducing the risk of inbreeding and genetic drift (also supporting the genetic conservation targets) (e.g. Ewens et al. 1987).

- **Species-level conservation targets:** Setting targets at the level of single species, the detailed biology of individual species needs to be studied, while species must be part of conservation strategies and a monitoring program. Although such conservation targets are ineffective to conserve entire ecosystems, if targets cover the demands of a limited group of species, they may encapsulate the demands of many other species (multi-species approach) (e.g. Roberge and Angelstam 2006).

- **Habitat and ecosystem targets:** Correspond to the tactical and operational tools that can be utilized to achieve conservation goals for patch areas and composition, landscape structure, connectivity and natural disturbances. The conservation of habitats must be sufficiently large and include significant landscape heterogeneity to maintain evolutionary processes not only for the keystone species but also for the rest elements of the ecosystem. In this conservation level the management effort includes: (i) identifying the ecosystem functions that habitats provide in the wider ecosystem, (ii) maintaining population of focal species and the associated critical habitat, (iii) minimizing the loss of soil and protecting water quality, (iv) increasing carbon storage etc. (see van Dyke 2008; Baskent 2009).

- **Community-level targets:** Focus on maintaining the relationship between area and species richness, supporting the target of species composition and ecological succession at the community level (Legendre and Legendre 1998; Økland et al. 2003). The targets under this level support the monitoring of the correlation between species richness and spot size, allowing to predict the number of species lost when a certain fraction of habitat is removed, or to estimate the area needed to retain a certain fraction of keystone species (Jonsson and Villard 2009).

Jonsson and Villard (2009) set the above quantitative targets along with the main approach (see Appendix 3.A).

For narrow habitats the nomination of conservation targets should be all-inclusive, adopting guides from the three levels of biodiversity. Due to the narrow distribution of such type of habitat/forest, and the low or absent financial benefits, often there is limited understanding of needed management measures for acting against their degeneration, thus leaving a gap in data that could be useful towards target-setting and quantitative feasibility assessment. (Armstrong and Wittmer 2009). The conservation targets for a narrow habitat interpreted in biodiversity terms, encompass three main elements: ecosystem composition, structure, and function (Noss 1990; Larson et al. 2004).

### 3.3. Designing a conservation strategy for a narrowly distributed forest habitat

The growing public and scientific concerns about the narrow endemic plant species and especially the tree plants that are the keystone species for unique habitat types, lead to great chal-
challenges in conservation and management strategies. According to the theoretical overview, as presented above, the management of such habitats/forests should integrate the production of multiple values on a sustainable basis without jeopardizing their health and integrity in the long term. Thereby, in any conservation effort, a correct remedy is impossible without correct diagnosis. The section below presents the general guidelines and levels to be addressed for defining conservation targets for a narrow habitat. Elaborating a conservation strategy could encompass the following workflow elements:

i. **Determining the cause of the decline of habitat/forest:** A habitat may decline for many reasons, and for specifying the reasons, available data should be examined, that would allow comparison of the current population to its past, possibly larger, state (van Dyke 2008). Further, the forces causing degradation of this habitat should also be examined. Under this framework the interpretation of the reasons that lead to the decline of habitat and their impact on their resilience should be identified by addressing:

- The alteration of the past and current geographic range of the habitat.
- The use of the habitat (land and keystone species) during the centuries; and how its use diverged in time.
- The identification and characterisation of the pressures and threats (i.e. competitors, predators, parasites, diseases) that affect the habitat nowadays compared to interactions of the past.
- The alteration of environmental conditions (climatic and soil conditions etc.) for the habitat during the centuries and especially during the current period of climate change. Further, comparison of these conditions with past conditions should also be addressed.
- The possible existence of a direct exploitation on the habitat by humans, legally or illegally.
- Whether there is habitat destruction, and whether -particularly in breeding habitats- the population is quantitatively and qualitatively stable.

ii. **Determining the factors relevant to the present conditions:** In addition to any available historical data for the target habitat, there are dimensions of a habitat’s current status that merit examination independent of historic analysis. Namely, the factors below can be analysed:

- Environmental and demographic stochasticity of habitat/forest, namely the level of environmental variation and whether this quantitatively affects populations. Further, current demography of the population, namely age-class of trees, mating system, regeneration, rate of seedlings survival and diebacks.
- Genetic diversity and population genetic structure of keystone species, and definition of the species’ inbreeding and heterozygosity. As any genetic erosion (genetic drift) occurring in a current population of habitat keystone species leads to signs or effects of inbreeding, it is important to include morphological distinctive characteristics, deformities, sterility, or abnormal juvenile mortality.
- The influence of habitat impact under the threat of natural catastrophes and/or any types of natural disasters, together with the identification of the threat kind, and its frequency and severity.

iii. **Determining the conservation goal and targets:** Ultimately, based on the existing knowledge, the recovery goals\(^\text{1}\) will be set, together with the recovery conservation targets through specific conservation actions/ measures focusing to support the increase of

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\(^{1}\) The term of recovery goals corresponds to general goals or policy statements that reflect societal values and political or institutional intent, but lack the detail required for implementation (Armstrong and Wittmer 2009).
the size and persistence of the targeted habitat (van Dyke 2008; Armstrong and Wittmer 2009). The recovery goals could be designed by adopting specific guidelines of habitat representation, by establishing the keystone species across the full array of the potential habitat’s area, by safeguarding its resiliency through protecting a habitat large enough to remain viable, and by ensuring the habitat’s redundancy through saving enough different patches of them (Armstrong and Wittmer 2009). On the other hand, the recovery targets should be aimed at increasing abundance through recovering the remnant habitat and/or establishing new populations (Armstrong and Wittmer 2009). Thus, the management actions in a narrow habitat/forest should focus on increasing the size and persistence of its small range:

- Intensive ecological and environmental management of habitat’s keystone species, should primarily focus on natural reproductive capabilities and adaptations for survivorship of population of targeted species. In such case, the environment should be enhanced in order to maximize favorable environmental conditions, minimize detrimental environmental variation, and optimize population demography toward maximum growth through removing competitors, predators, parasites etc. Thus, managing should first identify the correlations between environmental features and high-density populations, hypothesize about the causes behind these correlations, and then proceed to managing sites in a way to favour these environmental features (James et al. 2001).

- The long-term sound management of the targeted narrow habitat should be secured by conserving its biodiversity and habitat solid structure. An effectively conserved habitat supposes the management and conservation of all the biodiversity elements that directly and indirectly support their resilience and resistance. Any degree of habitat fragmentation usually has a far greater effect at both the keystone species of habitat, and the other species (plant and fauna) living in this. Based on this the argument, it is important to understand the specific effects created and the processes altered when the phenomenon of habitat edge is increased in a landscape; this understanding will lead to sound conservation of a habitat and its species. Owing to habitat fragmentation and area-to-edge ratio on habitat patches, it is not uncommon for the most known and studied aspects of habitat fragmentation to be edge effects (van Dyke 2008). In addition to this assumption, managing should support habitat connectivity, by managing Habitat Corridors: It is Habitat Corridors Percolation theory that sets habitat connectivity as the vital factor for successful conservation of habitats and populations associated with them.

3.4. Conclusion

The general argument that in conservation a correct remedy is impossible without correct diagnosis, seems to be more than suitable for narrow and endemic habitat types. Conservation managers must assess which strategy carries the greatest potential for recovery and the lowest risk of loss of the targeted habitat, as well as its biotic and abiotic elements. Thus, the conservation effort for a such habitats should examine data carefully (existing knowledge), extract additional information, formulate clear hypotheses regarding causes of population decline, and make testable predictions that can be evaluated in management actions if they are to be effective in managing small and declining populations of the habitats’ keystone species.

In overall, an encompassing framework should be established, for setting and implementing quantitative targets for recovering a narrow, and in most cases threatened, habitat. It should be emphasised that such targets should be constantly re-evaluated, according to information flow from such flexible management. A critical point for conservation for a narrow habitat is also the guiding principle that any conservation effort is that of any life form in what it is a niche, shaped as an adaptive fit (Rolston 2004). Wild organisms are subject to processes in a historical lineage.
and are under their populations’ evolution processes. In all cases, this lineage is the outcome of entwined genetic and ecological processes (see Rolston 2004; Dyke 2008; Villard and Jonsson 2009). The implemented conservation measures should ensure the dynamic conservation of habitats at both within their natural sides and outside their natural range. Hence, addressing the causes of a habitat’s decline, researchers should first take into consideration all available data and conditions, and then extract the most informed hypothesis about this decline.

3.5. Reference


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**Appendix 3.A:** The conservation targets (measures) should combine different levels in order to be holistic and at the same time strengthen the ecological connectivity with the rest elements of biodiversity (abiotic and biotic factors). Jonsson and Villard (2009) set the above quantitative targets along with the main approach.

<table>
<thead>
<tr>
<th>Approach used</th>
<th>Type of biodiversity target</th>
<th>Examples</th>
<th>Reference(s)</th>
</tr>
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<tbody>
<tr>
<td><strong>Genetics targets</strong></td>
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<td>Analysis of inbreeding and genetic drift as a function of population size</td>
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<td></td>
<td>Natural level of genetic variation</td>
<td>Variation in allelic diversity as a function of sample size</td>
<td>Blakesley et al. (2004)</td>
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<tr>
<td><strong>Species-level targets</strong></td>
<td>Probability of occurrence</td>
<td>Habitat models – linear regressions with stand attributes; Habitat suitability index</td>
<td>Edenius et al. (2004)</td>
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<td></td>
<td>Minimum viable population</td>
<td>Fitness estimates – plant populations; Populations viability analysis (PVA) - vertebrates</td>
<td>Reed (2005), Reed et al. (2003), Akça kaya (2004)</td>
</tr>
<tr>
<td></td>
<td>Minimum viable metapopulation</td>
<td>Population viability analysis; Incidence function; Simulation modeling</td>
<td>Guittierez (2005), Baguette and Schtickzell (2003), Snäll et al. (2005)</td>
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<td></td>
<td>Metapopulation persistence</td>
<td>Metapopulation capacity of the landscape</td>
<td>Hanski and Ovaskainen (2000)</td>
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<tr>
<td>Community-level targets</td>
<td>Habitat and ecosystem targets</td>
<td>References</td>
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<td>Species richness</td>
<td>Species richness</td>
<td>Desmet and Cowling (2004)</td>
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<td>Community composition</td>
<td>Species-area relationship</td>
<td>Macdonald (2007)</td>
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<td>Community composi-</td>
<td>Indicator species analysis</td>
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<td>Minimum habitat area</td>
<td>Threshold in species occurrences; Species-area</td>
<td>Hayden et al. (1985), Watson et al. (2001)</td>
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<td>habitat, local (stand) level</td>
<td>Threshold in species occurrence – logistic</td>
<td>Guénette and Villard (2005)</td>
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<td></td>
<td>regression and ROC analysis</td>
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<td>Habitat amount (territory level or landscape</td>
<td>Thresholds in patch occupancy – piecewise</td>
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<td>and ROC analysis</td>
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<td>Connectivity</td>
<td>Simulation modeling</td>
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<td>Natural disturbance (frequency, extent,</td>
<td>Incidence function model and spatially explicit</td>
<td>Schultz and Crone (2005)</td>
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<td>intensity)</td>
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<td>Landscape division according to disturbance</td>
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<td></td>
<td>Forest edge structure</td>
<td>Pennanean and Kuuluvainen (202)</td>
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Towards an Integrated Conservation Vision of Narrow Habitat Types - Silvicultural Interventions

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4.1. Introduction

The objective in this chapter is to present and analyze a manual for the development of silvicultural treatments for the conservation of narrow range tree species populations. These species can be endemic or species which are at the limits of their expansion covering a rather small area (see Eliades et al. 2019). The methodology for the assessment and determination of the appropriate silvicultural measures for the conservation of a species or a group of species can also be applied in the populations, of any species, that are under pressure or are in the danger of extinction (see Abeli et al. 2009; Fady et al. 2016; Eliades et al. 2018).

The forest stand analysis referred by Dafis (1992) was used as a starting point for the creation of the manual in which a process of successive stages of analysis of narrow range species ecosystems is developed. This manual also describes basic silvicultural tools and approaches used in the manipulation of forest stand structures. The findings of the successive analysis of the manual will lead in the assessment - determination of the appropriate silvicultural measures for the conservation of the under consideration species, in each different case.

In this analysis of Dafis (1992) the present status of the forest stand and the stand history based on existing data, must be assessed and determined. Also, a forecast of the stand dynamics must be done and the stand structure we want to create (silvicultural objective) must be determined in order to define the treatments which have to be applied in order to achieve this goal.

However, in the case of the narrow distribution range species the above-mentioned requirements of knowledge and needs for determination, are not enough for the construction of the manual which leads to the development of the silvicultural measures that have to be applied for the conservation of the target species. More analytical information is needed and the successive stages of analysis must incorporate specific sub-stages. In this chapter these stages and sub-stages will be presented in detail.

In Figure 4.1 the successive stages of the manual analysis are presented.
4.2 Analysis of the target species ecosystem - formation characteristics and the disturbance regime of the area at the present time

The site conditions where the under consideration species appears, have to be fully analyzed. Site analysis is incorporated in the description of the present condition of under consideration population - species. A very analytical description of the different sites where target species appears is required.

However, the stand structure analysis is the major tool for the correct description of the present status of the forest – population. The stand structure of a formation – stand is a snapshot of the present stand characteristics (see Oliver and Larson 1996). A prequisite for the proper analysis and understanding of the stand structure is the knowledge of the ecological characteristics and biological requirements of the species and the populations, which are under consideration for
conservation and treatment (see Milios et al. 2021). On the other hand, the stand structure analysis itself provides the information needed for the understanding of the ecology of the species present in the stand.

At this point it is very useful to present of analytical definitions of stand structure.

**Stand structure**

The structure of a stand is the physical and temporal distribution of trees as well as of other plants. The expression of this distribution can take place using many characteristics of trees and plants, such as the plant species, the horizontal or vertical spatial patterns, the size of living or dead trees (or plants) or parts of them (including stem cross section, crown volume, stem, leaf area, and others), the age of the plants or combinations of the above-mentioned characteristics (Oliver and Larson 1996). Smith et al. (1997) state that the internal structure of a stand is defined by the distribution of diameter classes, the arrangement of different vegetation stories or layers and variations in age classes and species composition. Smiris (1992) argues that, in ecologically adapted forestry, the stand structure includes the type, degree and form of mixing of forest species, the distribution of ages and dimensions of trees and the spatial horizontal and vertical structure of the forest. The stand structure is the result of the ‘interaction’ of the morphological and physiological patterns of the trees with the physical environment (Oliver 1992).

The description of the parameters of the structure can be achieved through a variety of methods and measurements, such as measuring the diameter (breast height, basal etc.) and height of trees, calculating the basal area by story and forest species, measuring - calculating the beginning (in the bole) and length of the crown of trees, the determination of age-height, age-diameter, diameter-height relationships of trees, the design of vegetation profile of a part of the forest, etc.

The stand structure analysis was used by Milios et al. (2021) in order to acquire basic knowledge regarding the ecological requirements of *Cedrus brevifolia*, an endemic species of Cyprus.

Light requirements of a species determine in a great extent its position in the succession process in an area (Dafis 1986; Dafis 1992; Oliver and Larson 1996). The appearance of a species in the different stories of a forest can give substantial information regarding its ability to endure shade. However, the density of formation in the different stories may affect light availability in each of them. As a result, the height structure of a formation having an adequate density provides information regarding shade tolerance of the goal species. Moreover, the tree density in the different diameter classes in a diameter structure analysis can provide additional useful information.

Along with the shade tolerance of a species another trait that strongly influences competitive ability of a species is its site sensitivity (see Oliver and Larson 1996). Of course, the competitive ability of a species is not constant and depends on site productivity and the characteristics of the rest competitive species (see Dafis 1986; Larson 1992; Oliver 1992). Hence, a height and diameter structure analysis in the areas with different site productivity where the species under consideration appears will strongly contribute to the construction of an adequate amount of knowledge on the target species. Analogous approach of structure analysis in different site productivity areas in order to determine ecological characteristics of *Cedrus brevifolia* like shade tolerance, site sensitivity and competition ability were used by Milios et al. (2021).

It obvious that analogous information regarding ecological behaviour of the competitive species (if any) will be provided through the structure analysis of the under-consideration species formations across the range of sites where the target species appears.

Moreover, the analysis of the target species natural regeneration is a substantially significant process. This analysis provides as much information for the characteristics and sensitivities of a species as the stand structure analysis, and in some cases provides even more information. The determination – description of the micro-environments where the target species seedlings are established will provide essential insight in the sensitivities and endurance of the species against the dominant environmental factors. This analysis must be done in all sites where the species appears.
Is the species established under the facilitation of other plants, or prefer open environments under the full light? The answers in questions like this will increase our knowledge on the target species ecology. On the other hand, the answers in the same question for the other species (if any) which compete the goal species will strongly enhance the understanding of the under study ecosystems as well as the competitive ability of the target species in the various sites where the species appears.

A thorough and detailed stand structure analysis as well as the natural regeneration analysis have not to be applied only in the cases where there is no adequate knowledge on the ecology of a narrow range or of another target species. Even in well studied species in which the ecological characteristics are well known, the above-mentioned analyses are very significant and have to be done. The population(s), of under consideration species, is under pressure or the species is under the danger of extinction and thus the species ecological performance may be different as a result of the harsh ecological conditions created by biotic or abiotic environmental factors which it confronts. Petrou and Milios (2012) in the middle elevations of central Cyprus found that in its first growing season the seedlings of *Pinus brutia* established and survived under the facilitation of other plants, while in the bare ground under full light the newly established seedlings died. *Pinus brutia* is a light demanding, fast growing and site insensitive species (Quezel 2000; Boydak 2004; Kitikidou et al. 2011, 2012; Korakis 2015). However, the severe environment (extreme soil temperatures, summer drought etc.) was possibly the reason for the failure of *P. brutia* to act as a light demanding and site insensitive species and to be established and survive in bare ground under full light (Petrou and Milios 2012).

Site conditions and productivity in combination with the species ecological characteristics and disturbance factors acting in an area can differentiate the behavior of the formations of various species.

*Juniperus excelsa* can be characterized as a narrow distribution range species in Greece and Cyprus, since formations of the species with rather extended area are very rare in both countries (Milios et al. 2007; 2011; Stampoulidis et al. 2013).

In the central part of Nestos valley in northeastern Greece Milios et al. (2007) found that even though there were productivity differences in the different sites, where *J. excelsa* creates (mixed and pure) formations and groups, their impact on the variables of structure of the species formations was small. On the other hand, differences in site productivity and site characteristics combined with grazing, which was one of the main disturbance factors in the area, led to different patterns in regeneration plants establishment between at least two different sites having different productivity and characteristics.

Prespa National Park in northwestern Greece is another area where *J. excelsa* creates stands. In Prespa National Park, as well as in Nestos valley, grazing (through trampling in Prespa) is a crucial factor that affects natural regeneration process (Milios et al. 2007; Stampoulidis et al. 2013). However, facilitation, (in order the *J. excelsa* plants to be protected from grazing) determined the seedling establishment patterns in Nestos valley (Milios et al. 2007), while in Prespa, facilitation is not the dominant factor that influence the process of seedling establishment, even though many regeneration plants were established under facilitation of other plants of the species (Fig. 4.2) (Stampoulidis et al. 2013). According to Stampoulidis et al. (2013) if more livestock were circulating and grazing in Prespa National Park the facilitation process through protection from trampling probably would have been dominant in the regeneration establishment.

According to Milios et al. (2011), in Cyprus, grazing did not influence the regeneration process and facilitation not only did not dominate as a process but probably in many cases the process of competition is the decisive factor that leads the regeneration process of *J. excelsa*.

In Nestos valley apart grazing, a factor that affects the formations of *J. excelsa* is illegal cuttings of small sprouts and of branches, while a litter layer accumulated under the nurse plants, that
belong to the same species, is another factor that facilitate the establishment of the seedlings of the species (Milios et al. 2007). Both grazing and illegal cuttings of small dimension sprouts favor *J. excelsa* in Nestos valley since they suppress the competitive broadleaf vegetation. The reduction of competition imposed on *J. excelsa* enable it to dominate even in some more or less productive sites of the area (Milios et al. 2007). On the contrary, in Cyprus, the absence of factors which favor *J. excelsa* will lead to its replacement in better sites by other more site sensitive species (Milios et al. 2011).

From the above-mentioned example of the narrow distribution range species of *J. excelsa*, it is obvious that the disturbance regime, as well as the site characteristics and productivity, strongly affect the stands – formations of *J. excelsa* leading among others in different patterns in the regeneration establishment.

So, another important factor which has to be thoroughly described and analyzed is the disturbance regime in an area. Some questions have to be answered. Which are the main disturbance factors acting in the area and which are the characteristics of the disturbances taking place (intension, area which affects, periodicity etc.)? Oliver and Larson (1996) provide an excellent analysis of disturbances and their effects in forests. A very important issue is the determination of the specific effect that disturbances have upon the components of the under-concern ecosystems. Which are the exact and net influence of disturbances upon the target species and upon the competitive species (if any)? These are two very significant questions that have to be answered in the analysis of the present status of population of the target species. This type of analysis is going to clarify the risks faced by the under concern species population as the result of the disturbances acting in the study area (see Oliver and Larson 1996; Lindenmayer and Franklin 2002).

**Figure 4.2:** Photo of a *J. excelsa* regeneration plant which have been established under the facilitation of a nurse plant of the same species in Prespa National Park (photo by A. Stampoulidis).
4.3. Past dynamics of the species formations - disturbance regime of the past

Analysis of the past stand structure, and of the forces that influenced and triggered the past formation dynamics of the target species will integrate the knowledge on ecology of the species.

In this context information regarding the structure and characteristics of the species formations in the past have to be gathered from official records, publications, local residents, history records of the area etc. (Milios 2000a; Milios 2000b; Milios et al. 2007; 2009, 2011; Stampoulidis et al. 2013). Data and information can be acquired through stem analysis of trees or by taking increment cores or counting age and measuring the width of annual growth rings in cross sections taken from stumps of cut trees. These techniques will provide information for the time of the tree establishment, events which significantly affected the growth of trees (for example disturbance events) and for the growth rates of the trees in the past (Milios 2000a; Milios 2000b; Milios et al. 2007, 2009). In order to have more accurate and reliable reconstruction of the history of species formations, the data regarding age and growth of trees have to be acquired from a rather large number of trees. This is not easy at most times since it is a very costly and time-consuming process.

The present stand structure of the under-consideration formations can provide an insight in the species formation history, since the present spatial distribution of trees reflects the past establishment patterns (Fig. 4.3). In the example of the J. excelsa in Nestos valley in Greece the present structure in combination to the age data of trees, stem analysis of trees, information from local residents and war as well as socio-economical history of the wider area where the species formations appear, delineated the history and past dynamics of the species formations (Milios et al. 2007, 2009). A period of intense disturbances (grazing and cuttings) acting in the area followed by a period when the intensity of the disturbances was reduced or the disturbances stopped. This disturbance history combined with the ecology and traits of component species of the J. excelsa formations and groups, strongly determined the past dynamics and the present structures of the species formations (Milios et al. 2007, 2009). In the formations of Prespa National Park, tree cuttings during past wars influenced stand structure attributes, while cutting of J. excelsa branches in the past affected growth of J. excelsa trees (Stampoulidis and Milios 2010). Moreover, in Cyprus, according to Milios et al. (2011) past disturbances determined present structure variables of J. excelsa formations.

4.4. Forecast of future stand dynamics of the species formations

After (i) the analysis of the present stand structure of the target species (or population), (ii) the reconstruction of the species formation past dynamics, (iii) the understanding of the regeneration patterns of the species (and of its competitors – if any), (iv) the comprehension of the site characteristic (or productivity) influence in the regeneration, and stand structural patterns of the target species (or population) as well as (e) the understanding of the role played by the past and present disturbance regimes in all the aforementioned issues, an adequate and necessary amount of information on the ecology and competitive ability of the target species and its competitors (if any) will have been obtained. Based on this information the forecast of future stand dynamics is, in most cases, more or less a relatively easy process (see also Larson 1992; Oliver 1992).

However, the forecast has to create various alternative paths of stand dynamics in the different sites of the area where the target species – population occurs. The stand dynamics of the species, in the future, under the current disturbance regime should be assessed first. Also, the stand dynamics, if the predominant disturbances acting in the area cease to occur, should also be assessed. Moreover, the possibility of a new disturbance regime, or of an extreme disturbance event that has never been appeared in the area should be taken into account. A combination of disturbances, extreme disturbance events and novel types of disturbances can create unfavorable conditions for the conservation of the target species or population (see Fady et al. 2016; Sasaki et al. 2015) and a forecast of the target species status after such events and incidents has to be done.
Determination of future stand structure targets

A next step in the process for the assessment and determination of the appropriate silvicultural measures for the conservation of the under-consideration species – population is the determination of future stand structure targets in the various sites where the species appears. The targets to be set should be achievable based on the analysis of the ecology of the species and its competitive ability in the various sites where it appears. It is clear that the setting of the stand structure targets should be based on the available information obtained from the previous sub-stages of the analysis and the forecast of the future species population dynamics. The structural targets are related to quantitative participation of the species in the stand structure and are referred to number of individuals per hectare, to basal area or even to volume of the target species per hectare. This determination has also to be referred to the participation of the species in the various stories of the stands – formations.

The determination of the stand structure targets of the under - consideration species formations will lead to the specification of the necessary silviculture measures to be applied for achieve them. In this process the estimated future population dynamics will assist in specifying the necessary and (in relation to their application) timely correct measures to be taken.

4.5. Assessment and determination of the appropriate silvicultural measures for the conservation of the under consideration species – population

All available information on the ecology and competitive ability of the target species and its competitors (if any) in combination (i) with the future conditions that are expected to prevail in the area, where the species occurs, and (ii) with the stand structure targets, will determine the necessary silvicultural measures to be taken to protect and conserve the target species and its formations.
Of course, any silvicultural treatment that will be developed should be part of a more general conservation and protection plan of the target species (see Lindenmayer and Franklin 2002). For the development of this plan, the above-mentioned estimates of the dynamics of the species formations, should be taken into account, inter alia (see Lindenmayer and Franklin 2002).

A dominant force that determines the dynamics of the species formations is its position in the succession process in the area (see Oliver and Larson 1996), or in the various sites of the area. As a result, if the species that has to be conserved, is an early successional species or there are more competitive species in the area (or in some sites) where the species appears, then the intensity of the manipulations must be higher than in the case the species was a late successional species or it was more competitive than the other species in the area (or in some sites of the area).

As the different species and populations that are under pressure can vary in characteristics, risks they confront and expansion area, the recommended silvicultural treatments may vary in terms of immediacy, implementation timetable and other characteristics. They can vary from a certain package of measures for immediate favor of the species in the short- to medium-term horizon, up to the development of an integrated silvicultural system. This system will include interventions aiming at the species regeneration and to handling the structure of the species formations during the various stages of their development as well as in their different structural forms.

All the proposed silvicultural treatments from the immediate to the long run to be implemented must be described analytically and should be specified for the different site conditions occurring in the area where the target species appears. However, the specification and the analysis of the proposed treatments must not take the form of absolute recipes and oversimplified rules, since all the treatments have to be based up on the basic principles which is the favoring the target species. As a result, the developed silvicultural treatments need to be flexible so that they can be adapted or even to be differentiated depending on the different conditions prevailing in each different group of trees in each formation (Milios et al. 2006). The time of the application of silvicultural treatments should be designated while the periodicity and other characteristics of the treatment applications should be assessed.

In the example of *J. excelsa* formations, Milios et al. (2007) analyzing all the available information regarding the present structure of the species formations, the patterns of the species regeneration establishment, the site sensitivity of *J. excelsa* and the past and present disturbance regime of the area, mentioned that in the central part of Nestos valley, in better sites *J. excelsa* will be replaced by more site sensitive species if the disturbances (illegal cuttings and grazing) which favor *J. excelsa* were to stop. Moreover, in order to prevent this development, they recommend that the forest practice has to imitate the action of the aforementioned disturbances in the area. In the same pattern, for the species formations in Cyprus, in better sites Milios et al. (2011) recommend the suppression through cuttings of the more site sensitive species that compete *J. excelsa* since otherwise they will replace it.

As already mentioned in the present chapter, all the existing knowledge regarding the target species should be used. In this context, the knowledge of the genetic characteristics of the species should be taken into account so that it can be integrated into the silvicultural treatments that will be developed. Based on the existing knowledge both in general aspect and particular for the specific species the silvicultural treatments that will be developed should be designed in order to not have negative genetic implications for the target species (Finkeldey and Ziehe 2004; Koskela et al. 2013; Ratnam et al. 2014; Fady et al. 2016; Aravanopoulos 2018; Eliades et al. 2018, 2019). The previous mentioned bibliography will be very useful to avoid species genetic degradation if in the recommended treatments planting (or sowing) actions of the target species are included.
4.6. Basic silvicultural tools and approaches used in the manipulation of forest stand structures

The basic tool used by silviculture in the manipulation of forests is the redistribution of the growing space. This can be done in most cases through the release of the growing space as a result of tree cuttings or cuttings of tree branches and in other cases via the occupation of free growing space via planting – establishing of new plants (see Dafis 1986; Oliver and Larson 1996; Smith et al. 1997; Nyland 1996; Milios et al. 2006). This redistribution of growing space can alter the competition regime and can favor specific individuals (plants) or species.

However, another significant process for the functioning of many ecosystems is facilitation. It is the main process behind the regeneration of plants in shelterwood systems where the residual trees of the mother stand 'protect' the established seedlings (see Dafis 1986; Smith et al. 1997; Nyland 1996). Its significance is high in the Mediterranean area (Rousset and Lepart 2000; Petrou and Milios 2012), and can be used in the planting - establishment of plants in severe environments (Castro et al. 2004a; Castro et al. 2004b).

A very effective tool in the ecosystem management is the usage of the plant sprouting ability in the various silvicultural interventions (Milios and Papalexandris 2019). Sprouting minimize the effects of disturbances and it is a form of plant ‘persistence’ in an ecosystem leading to a reduction in the turnovers of populations after disturbances (Bond and Midgley 2001). The persistence ability of some species through sprouting is a very useful tool in the development of silvicultural practices for the conservation of broadleaf species that sprout. On the other hand, the suppression of a sprouting species in order to favor a ‘valuable’ species is difficult and requires repeated treatments (see Milios et al. 2011). Seedling sprouts are a category of sprouts having advantages compared to seedlings (Smith et al. 1997). Advance regeneration (see Oliver and Larson 1996; Smith et al. 1997; O’Hara 2014) combined with the ability of seedling sprouts to resprout (Hara 1987) can be used in the regeneration process of species under rather adverse conditions (see Papalexandris and Milios 2010). Milios (2010) presents in detail the facilitation process and the sprouting ability of plants as well as their combination (in silvicultural manipulations) as tools that can be used by forest practice in forest ecosystems in the context of climate change.

Finally, in the example of J. excelsa formations in Nestos valley, in order to preserve them Milios et al. (2006) propose thinning of J. excelsa groups, pruning of the dense lower branches of some J. excelsa trees under which small J. excelsa trees are aggregated as well as plantings of J. excelsa plants. Some of these plantings are proposed to be done under the shade other species plants and afterwards the J. excelsa plants to be gradually released from the competition through successive cuttings of the other species plants.

4.7. References


Considerations Regarding Fire Management in the Frame of Long-term Conservation of Narrow Endemic Habitat Types in a Limited Area of Occupancy

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5.1. Introduction

The principles of forest fire management are well-documented in international scientific literature, covering the aspects of fire prevention, presuppression planning, suppression and post-fire rehabilitation. However, it is well known that application of the principles in reality varies tremendously, as necessarily they must be adapted to existing conditions. These conditions reflect the specific properties of vegetation to be protected, the layout of the land, the people and their perceptions, needs, and activities, the existing infrastructures, the historical and political context, etc. As a result, fire management approaches vary across the globe, from one country to the other, and even within countries, depending on the environmental characteristics and protection needs (Stephens 2005). Thus, the general principle that management is a mix of science and art is necessarily true in the realm of fire management.

One of the conditions that entails special attention regarding wildfire management is the protection of narrow endemic habitat types, in the frame of their long-term conservation, especially when they occupy a relatively limited area. The limited extent of this area results in an obvious risk of near complete elimination of the habitat type and its characteristic species by a single catastrophic fire event. Hence, it is necessary to plan accordingly in order to minimize as much as possible the risk of the occurrence of such an event. On the other hand, emphasis on fire protection may not be enough, because the management of a living ecosystem is very different from the protection of houses or infrastructures. The task is not always easy and straightforward and presents significant challenges to the officers entrusted with the management of such habitat types, as they often have to accommodate various conflicting demands, limitations and restrictions in their decision-making. Some of the most important aspects they have to consider, are presented and discussed in this report.

5.1.1. Fire management considerations

Although, as mentioned above, the principles of forest fire management are well known, the objectives may differ significantly, resulting in quite different planning and course of actions. For example, the fire management plan (FMP) of a forest unit is likely to have some important differences in objectives and hence management priorities from that of a national park, a recreation area or a wildland-urban interface area. For instance, the following are listed as fire management goals1 for the Golden Gate National Recreation Area in California, which belongs to the US National Park Service:

i. Ensure that firefighter and public safety is the highest priority for all fire management activities.

ii. Reduce wildland fire risk to private and public property.

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1 https://www.nps.gov/goga/learn/management/upload/fire_fmp_op_strat_ch2.pdf
iii. Protect natural resources from adverse effects of fire and fire management activities, and use fire management wherever appropriate to sustain and restore natural resources.

iv. Preserve historic structures, landscapes, and archeological resources from adverse effects of fire and fire management activities, and use fire management wherever appropriate to rehabilitate or restore these cultural resources.

v. Refine management practices by improving knowledge and understanding of fire through research and monitoring.

vi. Develop and maintain staff expertise in all aspects of fire management.

vii. Effectively integrate the fire management program into park and park partner activities.

viii. Foster informed public participation in fire management activities.

ix. Foster and maintain interagency fire management partnerships and contribute to the firefighting effort at the local, state, and national level.

x. Minimize smoke generation during prescribed burning through the use of a smoke management plan (SMP) that details best management practices or non-burning alternatives where these options would meet resource management and fuel reduction objectives and also achieve emissions reduction.

Whereas certain management options, such as prescribed burning, may not be available in some regions, making the corresponding objectives irrelevant, the list above is a good reminder of what needs to be addressed in a FMP. As seen, protection of natural resources and of unique elements (historic structures, landscapes, and archeological resources) are high on the list. One such element are "narrow endemic habitat types (NEHT) in a limited area of occupancy" which obviously require special attention, especially when focusing on long-term conservation. Understanding the fire ecology of such habitat types is a prerequisite for their successful long-term management.

5.1.2. Ecological traits of the species of interest

A FMP for a forest unit or park that includes a NEHT, necessarily needs to address both the short term survival and the long-term conservation of the key species. While fire protection through fire suppression may appear as the straightforward answer to short-term survival, long-term conservation is much more complex and requires a good understanding of fire ecology. The ecological traits of each species of interest must be well known and understood, especially concerning its relation with fire.

A key term in fire ecology is that of "fire regime" which describes the general pattern under which fires naturally occur in a particular ecosystem over an extended period of time. A fire regime is defined by the physical and biological properties of that ecosystem, and the prevailing meteorological conditions, but also by the spatial, temporal, and behavioral characteristics of the fires that burn in it. Scientists classify fire regimes using a combination of factors including frequency, intensity, size, pattern, season, and severity. Brown and Smith (2000) described four different types of fire regimes based on severity:

**Forests**

- **Understory Fire:** Fires are usually non-lethal to the dominant vegetation and do not change its structure. It is usually a low intensity surface fire and usually 80% or more of the dominant vegetation is able to survive the fire.

- **Mixed severity fire:** Fire causes selective mortality in the dominant vegetation and varying habitat modification depending on the severity of the fire. The type of trees and their susceptibility to fire affect the outcome which can vary between understory and stand replacement. Variation can also occur within a single fire.
Forests, Shrublands and Grasslands

- Stand replacement: Fire kills the above ground parts of the dominant vegetation, changing its structure substantially. It consumes or kills more than 80% of the basal area or more than 90% of the overstory canopy cover.

- Non-fire regime: Little or no occurrence of natural fire. This is considered to be a wet environment where fire is not likely to occur.

Brown and Smith (2000) further developed coarse-scale definitions for natural (historical) fire regimes for the USA and interpreted them for fire and fuels management. The five natural (historical) fire regimes are classified based on average number of years between fires (fire frequency) combined with the severity (amount of replacement) of the fire on the dominant overstory vegetation. These five natural fire regimes include:

i. 0-35 year frequency and low (surface fires most common) to mixed severity (less than 75% of the dominant overstory vegetation replaced).

ii. 0-35 year frequency and high (stand replacement) severity (greater than 75% of the dominant overstory vegetation replaced).

iii. 35-100+ year frequency and mixed severity (less than 75% of the dominant overstory vegetation replaced).

iv. 35-100+ year frequency and high (stand replacement) severity (greater than 75% of the dominant overstory vegetation replaced).

v. 200+ year frequency and high (stand replacement) severity.

Where fire frequency is the average number of years between fires, and severity is the effect of the fire on the dominant overstory vegetation. Knowing the type of fire regime helps to understand the role of fire in a particular habitat and to plan accordingly. Without a good understanding of fire ecology, fire management may be misled. Stephens (2005) summarized some of the problems of this type faced in the history of fire management in the USA, as documented and debated by various authors.

In the case of narrow endemic habitat types in a limited area of occupancy, it is not easy to study directly either the fire frequency or the severity. The small area does not allow assignment of the NEHT to one of the fire regime classes above based on fire history data. For example, a forest of such type, belonging to fire regime V with a 200+ year fire frequency, is quite likely that it has not burned in recent memory. Furthermore, parts of this forest type may have actually burned in the distant past and, if the species is not fire adapted, may have been replaced completely by other vegetation.

Given the difficulties above, managers of such habitat types need to identify and use all the information sources they can get to develop the understanding on which to base their decisions. These include historic information on past small fires in the area with their location and characteristics, examination of the state of floristic recovery after those fires and other disturbances, and forest fuel characterization in mature and recovering stands, and human interventions (logging, grazing, land-use change). The fire tolerance and fire resistance of the key species of interest are of maximum importance and must be clearly characterized.

Fire intolerant plant species are usually very flammable and are destroyed completely by fire. Some of these species are unable to recover naturally after a severe fire while others possess adaptations, such as serotinous cones in many fire adapted pines that ensure the availability of a rich seed bank that is fire activated, that guarantee their natural regeneration (Thanos et al. 1996) (Fig. 5.1). Fire-resistant plants are able to withstand the passage of a light to medium surface fire suffering no or little damage, thanks to traits such as a thick bark, deep rooting system, and for large trees, development of a crown separated by significant distance from ground fuels through shedding lower vulnerable branches.
Figure 5.1: Thick regeneration of Pinus halepensis, a tree with serotinous cones, near the lake of Kaiafas (Ilia, Peloponnese) two years after the stand replacement fire of August 25, 2007.
Managers of NEHT should be aware of these properties regarding the key species they manage and should also be concerned with interactions with other species (competition for sunlight and water, nutrient cycling, etc.), especially invasive ones. They need to consider the potential outcome of long-term fire exclusion in parallel with the capacity of the key species to regenerate after a fire.

5.2. Main goals of fire management

As mentioned earlier, the limited area of occupancy precludes the option of a policy that considers fire as part of the natural circle of a particular NEHT and recognizes the possibility to allow fires to burn under pre-specified conditions. One such example are many wilderness areas of the USA. While the “Wilderness Act” clearly says that certain fires should be allowed to burn in such areas, many wildernesses are relatively small, or in close proximity to values at risk, so managers can’t afford to do that (Blois 2017). Things are even more restrictive to such policies when dealing with a NEHT that occupies a limited area. Even if the key species are fire adapted and regeneration can be assumed as certain, the need to maintain the presence of the habitat makes protection from a stand replacement fire an absolute priority.

Given the above considerations, it can be deduced that the main goals of fire management in the frame of long-term conservation of narrow endemic habitat types in a limited area of occupancy should be:

i. To minimize the probability of destruction of the habitat type by a large intense wildfire, i.e. to apply effective fire protection.

ii. To be prepared for the case of a destructive wildfire that will burn most (or all) of the area of interest, taking measures and planning in advance, i.e. to be prepared for effective post-fire rehabilitation.

iii. To ensure good health, composition and function of the ecosystem while serving the first objective, i.e. to achieve long-term conservation of the particular habitat type.

5.2.1. Fire protection considerations

Given the need to minimize the probability of destruction, fire management has to put a very strong emphasis on fire suppression. However, experience around the world has shown that even the best firefighting resources, under weather extreme conditions, have failed to control some extreme fires, that have run their course unobstructed for hours or even days, until the conditions changed. This type of fires have been called mega-fires, and the conditions that lead to them have been associated with climate change, fuel build-up, and extensive areas of high fire risk (including wildland-urban interfaces and infrastructures). For a NEHT in a limited area, such a fire may mean complete obliteration. Obviously then, strong fire suppression is not enough. In order to reduce the chance of a destructive fire, effective prevention is a key requirement.

5.2.2. Fire prevention and presuppression

Fire prevention needs to be planned very carefully. The particularities of the situation need to be acknowledged and considered in the plan. In order to eliminate or at least reduce fire starts, an in depth analysis of fire causes is needed. The location where fires started, the time (month, day, hour) of ignition, the causes (identified or assumed), the conditions under which the fires erupted, are all very important and need to be plotted and examined carefully. Clusters of fire starts, for example, may mean that a particular person or group of persons may be setting fires in the area. Agricultural activities for example, outside the perimeter of the NEHT, may result in fires that can enter the area of interest. Also, sometimes, NEHT require special protection and reduced human activities. This may affect some people living in the area and they may get hostile to the forest, becoming arsonists. Obviously, good fire cause investigation and systematic record keeping is required to support such analysis.

Contacts with people in the area, including workers, farmers, pensioners, hunters, etc. are key for understanding their feelings, their thinking and interests, and their level of concern for the environment and the particular NEHT. Such contacts also allow effective education and sensitization, and may also give the opportunity for active recruitment of fire prevention volunteers.
Analysis of fire causes should be part of a broader analysis called “threat analysis”. Wildfire threat analysis (WTA) examines and evaluates current fire risks and potential for ignition, intense fire behavior (fire spread, fire intensity and flame length), suppression capability and threats to important values. The threat from a fire coming from elsewhere causing damage to the NEHT, should be identified and analyzed separately from the threat of a fire starting within the NEHT causing damage there and potentially outside its extent.

The WTA needs to analyze the fire potential as related to fire occurrence, the fuels, and the fire behavior (a function of fuels, prevailing weather and topography), the values at risk (lives, assets, non-market values, etc.) and finally the overall threat. Suppression capability (resources, road access, firefighting infrastructures, etc.) must also be considered. Whereas all the elements contributing to the threat can be assessed through rating factors into arbitrary categories, it is better to avoid it, especially when assessing fire severity, fuels and weather (Wilson 2004). Statistical analysis with the help of descriptive statistics in table and graph forms, and development of GIS layers with the help of fire modelling can help in the spatial and temporal quantitative assessment of each of the elements contributing to fire threat. Modelling systems, such as the freely available FlamMap2 system of the US Forest Service (Finney 2006, Salis et al. 2013) can be used for spatial assessment of fire behavior. FlamMap is a fire analysis desktop application that can simulate potential fire behavior characteristics (spread rate, flame length, fireline intensity, etc.), fire growth and spread and conditional burn probabilities under constant environmental conditions (weather and fuel moisture).

The greatest challenge following the assessment of the various elements contributing to fire threat is how to combine them in order to characterize fire threat in space and time. Simply adding together or multiplying measures of risk, hazard and asset values is likely to produce invalid results. The relationships between the factors are complex, and are often non-linear and characterized by thresholds. Because of this, so far, there is not one unique method for weighing the influence on the various factors regarding their contribution to fire threat. The final outcome of the threat analysis should not be only one index mapped spatially. Instead, taking into consideration the purpose of the WTA (in this case the protection of NEHT) and the relationships between the factors, the outcome should include maps and descriptions regarding fire threat in place and time, as well as the sources that lead to it. This allows to develop appropriate fire prevention and fire risk mitigation strategies.

The outcome of the WTA should lead to the necessary preparations for the case of a serious fire event. In the case of NEHT, where fire protection should be maximized, the measures to be applied may include:

- Active fire prevention targeted to the population and to any other identified fire sources (e.g. industries, railways, powerline networks, etc.) in order to reduce potential fire starts.
- Preparation of the landscape through fuel management for fire hazard mitigation.
- Development of necessary infrastructures such as roads, water deposits, fire-hydrants, heliports, etc.
- Ensuring and preparing adequate firefighting resources.
- Developing scenario of potential fire events with corresponding firefighting strategies. This can be done with the help of fire behavior prediction software in combination with analysis of the topography, road accessibility and the network of areas where reduced fire spread is expected (fuelbreaks, firebreaks, water bodies, rocky areas, etc.).
- Ensuring the availability of a credible fire danger prediction system and establishing appropriate staffing levels and alert status according to the predictions.
- Establishing an adequate fire detection network utilizing appropriate resources and technologies, and a Coordination Center for quick and effective dispatching. Scenarios of potential fire locations and fire danger conditions in conjunction with travel times for the resources would allow a well-tuned response from the Center.

2 https://www.firelab.org/project/flammap
5.2.3. **Fire suppression**

In order to minimize damages and the chance of an intense hard-to-control wildfire, emphasis should be given to initial attack. It must be quick and effective. Actually, for the case of a NEHT it is justifiable to “play it safe”, making a type I error in statistics terminology, committing more resources than would normally be required, including aerial resources if the fire danger prediction indicates a possibility for intense fire behavior.

Well-trained and prepared ground resources, with experienced coordinators and well trained firefighting personnel need to be in place and act quickly. They must also be prepared for the case of an escaped fire, with criteria for early identification of such a prospect in order to call early for additional resources, and with large-fire suppression plans developed in advance and activated as soon as the conditions indicate that the initial attack has failed.

It is imperative that a system to call for back-up from all available resources in the area, from the various agencies, volunteers, armed forces, etc. should be in place, ready to be mobilized, with preparations and guidelines on how to get from them the best contribution they can make.

5.3. **Post-fire rehabilitation considerations**

In order to be prepared for the case of a destructive wildfire that will burn most (or all) of the area of interest, managers should plan in advance, taking appropriate measures. The urgency and extent of the measures depend a lot on the degree to which the main species of the NEHT are fire adapted and able to ensure natural regeneration.

In any case, for a NEHT with a limited area of occupancy, it is wise to search for sites with similar environmental characteristics, located at significant distances from the core of the NEHT, and create through artificial reforestation, new stands with similar species composition. Even, if the NEHT is fire adapted, in case it would burn, it would be years before it could produce seed. Furthermore, it would be at immature stage for decades, during which the “copy stands” would offer the opportunity for visiting, studies, seed collection, etc.

In addition to the creation of “copy stands” it is imperative that genetic material of the key species of the NEHT, should be collected and preserved in seed storage banks, in order to be able to facilitate an artificial regeneration effort. Furthermore, it is wise to maintain some production of seedlings of the key species in forest nurseries. This would facilitate the development of more copy stands, would allow quick restoration of small areas that may burn, especially for fires incoming in the NEHT, and would also give a head start of reforestation in case of a major burn. In all the above measures, preservation of the genetic diversity of the key species is an absolute priority.

In case of an extended burn, fire adapted species should be allowed to regenerate naturally. This will ensure optimum biodiversity maintenance and development of a healthy new stand. Where needed, protection from threats such as soil erosion (mainly on steep slopes and specific soil types) and intense grazing is warranted. Obviously, close monitoring of the evolution of regeneration is needed, as well as measures, such as thinning of the new stand, which will reduce fire hazard in the developing vegetation and will ensure good and quick growth. When a NEHT of fire regime V burns, it is highly unlikely that the species are fire adapted and natural regeneration will be ensured in a short time. In such cases, managers should focus on the properties of the particular species and its regeneration difficulties (e.g. need for shade if it is a shade tolerant species, availability of seed and requirements for germination, pests, etc.). All the elements mentioned above (ex situ conservation, seed preservation bank) are even more valid for such species.

5.4. **Long-term conservation considerations**

An effective protection from fire is often inadequate to ensure long-term conservation of a NEHT that occupies a limited area. With fire missing, the NEHT may change in subtle ways that in the long-term may result in trouble for the protected area. Especially for fire regimes with shorter return intervals, problems may compound faster. Managers should be aware of the potential for such developments, should monitor the development of the stands, and should respond accord-
ingly, trying to be preemptive. Some examples are:

- Lack of fire may lead to changes in the structure of stands regarding their composition, density, and competition. For example, shade tolerant species may grow in the understory, creating a dense layer that could prohibit the regeneration of light seeking species.

- Biodiversity may be affected negatively as light seeking plants and fire dependent species may disappear in the long term.

- The understory of shade tolerant species can act as ladder fuels that can facilitate passage of surface fires up to the crowns. This increases steeply the potential for an intense and massive, stand replacement fire.

- As trees grow older, over a long-period of time, diseases and insects are bound to take a toll of trees that grow old. Accumulation of dead material on the ground also affects fire hazard and the potential for an intense fire.

- Accumulation a thick layer of decomposing dead material, as well as lichens, mosses etc. in addition to light limitation may prevent natural regeneration.

Managers should be aware of all the above and should be ready to take measures to avoid negative impacts that may result from fire exclusion. As mentioned earlier, a good understanding of the ecology of the species, and especially the fire ecology, is very important, regarding the needed measures and their timing, being completely different for Mediterranean, central European, alpine, maritime, boreal and other types of ecosystems (Johnson et al. 2001). Examples of measures may include understory fuel management to reduce fire hazard, selective thinning to remove unwanted species, creation of openings in the overstory to allow light and help regeneration, soil scarification (Collins and Schwartz 1998, Hille and Den Ouden 2004, Prévosto et al. 2012, etc.) (Fig. 5.2).

Figure 5.2: A 113-year-old stand of Scots pines (Pinus sylvestris) on Pieria mountain, Greece, facing a regeneration problem due to litter accumulation on the soil and competition by ferns, in need of a disturbance of the soil surface to allow the seeds to germinate and establish roots. Prescribed burning (left) and soil surface scarification (right) were tested experimentally (scientific responsible Dr. Nikolaos Grigoriadis).
5.5. Some examples in support of proposed considerations

5.5.1. The outcome of total fire exclusion

A long bibliographic list exists in the USA where fire exclusion since the early 19th century altered natural fire regimes and resulted in a steep increase in fire hazard especially in the Rocky Mountains region (Arno et al. 1995). Photographic records provide ample documentation (Crotteau et al. 2018). As a result, in the last 2-3 decades, in spite of stronger firefighting resources, fires grow larger, are much more intense and dangerous, much more difficult to fight, ultimately burning larger areas in addition to costing much more (Calkin et al. 2015). The applicability of this widely accepted paradigm over all ecosystems has been questioned (Johnson et al. 2001) reinforcing the statement presented here for the need of managers to gain a good understanding of the ecology and function of NEHT they have to manage.

5.5.2. The need to maintain biodiversity in the long-term

The meadows of the pseudo-alpine zone of the Oiti National Park in Greece (NATURA 2000 priority habitat 6230) have been degraded in terms of biodiversity due to cessation of traditional land uses with the establishment of the Park, and in particular nomadic sheep farming, periodic burning of unwanted plants, and fuelwood cutting. This change has resulted in predominance of competing species and, above all, the invasion of the dwarf juniper (Juniperus communis ssp. nana) at a first stage, paving the way for expansion of true-fir (Abies cephalonica), a shade tolerant species that takes advantage of the shade of the juniper to establish its seedlings. The expansion of the woody plants threatens the existence of the meadows that grow in the openings and the disappearance of certain rare endemic species that grow in the area. It took many decades for these changes to be detected and documented. This led to experimental burns and mechanical treatments in order to mimic the old practices, ultimately leading to the recommendation for introduction of prescribed burning and sheep grazing in order to maintain this habitat type (Mantzanas et al. 2018) (Fig. 5.3).

5.5.3. The fire of the palm forest of Preveli, Crete, Greece

Preveli is a location on the south coast of Crete, Greece, in the prefecture of Rethymno. It is known for its historic monastery, a lagoon and beach located below the monastery and an extensive glade of palm trees that grows behind the beach. The area is an important tourist destination. The palm tree in Preveli, called the Cretan date palm (Phoenix theophrasti), is native to the eastern Mediterranean. It has a very restricted distribution, confined to southern Greece, a few sites on Crete and nearby islands, as well as some places on the Turkish coast.

Most of the palm forest at Preveli burned on the 22nd of August 2010 (Fig. 5.4). The firefighting resources proved unprepared and inadequate to control the blaze. A fire protection system that
had been installed at a cost of 1.2 million euros did not function. On the aftermath of the fire there was a lot of debate about the disaster and the post-fire measures that would be appropriate, but it became evident that there was no true knowledge of what to expect regarding the response of the particular species. However, within a few months, it became evident that the burned trees produced new shoots, while there was significant recruitment of new seedlings from seed. Finally, the only protection measures taken for the next two years had to do with avoidance of grazing and trampling of the new seedlings. Three years later the area was available again for tourist visits (Fig. 5.5).

The example of Preveli, demonstrates the need for very effective fire protection in such a NEHT of limited area of occupancy, since a small area, in this case about 100 ha, can burn in a very short time. It also shows the importance for managers to know the fire ecology of the species they have to manage.
5.5.4. The example of Abies pinsapo

Abies pinsapo is also known as the blue Spanish fir. It is native in Spain and Morocco and is the most drought tolerant of all firs. Its area of occupancy is very limited with only three enclaves in southern Spain and two in northern Morocco. It is a species that is categorized as vulnerable, has suffered by anthropogenic pressure (Linares et al. 2011) and its management requires special attention. The Abies pinsapo forests in Sierra de las Nieves were considered as relict by UNESCO Biosphere Reserve in 1995 and subsequently, they were included in Annex I of the Habitats’ Directive (92/43/EEC) which protects the three main habitats in Spain (Arianoutsou et al. 2012).

Several losses of Abies pinsapo stands due to wildfires have been recorded in the last decades despite the fact that the species is not particularly flammable (Esteban et al. 2010). Due to the sensitivity of the species, the regeneration traits of Abies pinsapo and its conservation have attracted a lot of research attention in the last two decades, in parallel with conservation measures. Currently, all the Abies pinsapo forests in Spain and Morocco are covered by some form of protection, which preserves them from further inappropriate use or exploitation. These forests are now recovering after years of intensive grazing and use of their timber for construction, firewood and charcoal making. However, these relict forests face the new threats of climate change, arson and the appearance of pests (Esteban et al. 2010).

In the frame of protection from fires Cortés-Molino et al. (2020) combined information from aerial LIDAR and hemispherical images taken in the field with ForeStereo—a forest inventory device—to assess the vulnerability and to design conservation strategies for Abies pinsapo stands based on the mapping of fire risk and canopy structure spatial variability. They recognized six fuel models in the largest stand of Abies pinsapo (252 ha), which they investigated, and created a fuel map based on these. Then FlamMap software was used for fire simulation scenarios based on fuel models, stand structure, and terrain data. Additionally, they analyzed canopy structure to assess the status and vulnerability. Their assessment showed a secondary growth forest that has an increasing presence of fuel models with the potential for high fire spread rate fire and burn probability (Cortés-Molino et al. 2020).

5.6. Conclusion

Fire management in the frame of long-term conservation of narrow endemic habitat types in a limited area of occupancy is a very complex task that requires good understanding of the fire ecology of the type as well as a good understanding of the principles of fire protection. Strong fire suppression is not enough. All aspects described above must be considered carefully in order to assure that the measures taken can be applied on the long-term basis, without negative side effects, and that alternatives have been put in place in case an inevitable fire, or other natural hazard, puts the existence of the particular NEHT in jeopardy.

5.7. References


6.1. Introduction

Insects represent three-quarters of the described animal species on the planet with about six million known species. There are currently nearly 300,000 known species of plants and they represent the highest biomass on Earth. Plants and insects have coexisted for some 400 million years and their interactions play a key role in the functioning of ecosystems, and more generally of the biosphere (Stam et al. 2014). These interactions can be of a mutualist type, as in the case of entomophilic pollination or protective associations between trees and ants (Bronstein et al. 2006). They can also be antagonistic since about half of the known insect species are phytophagous, i.e. they consume different parts of the plants (leaves, stems, roots, conductive vessels, flowers, fruits or seeds). They can do so in a more or less selective way and with very different intensities depending on the species. In forest ecosystems, insect phytophagy has consequences for trees from the individual to the community levels. Indeed, insect outbreaks can slow down the growth or kill trees (Kanat et al. 2005), and the consumption of reproductive structures (pollen, flowers, seeds) can interfere with reproductive success and consequently with the regeneration processes of a tree population (Boivin et al. 2019). Interspecific differences in plant exposure to phytophagy can also influence the rate of ecological succession in plant communities (Strauss and Zangerl 2002).

6.2. The concept of entomological risk in forest ecosystems

6.2.1. Interactions between trees and phytophagous insects and forest ecosystem services

Forests produce timber, firewood and a wide variety of non-timber forest products. They also play a protective role against soil erosion, landslides, they help maintaining the water balance at the landscape level, and they have an important recreational role. Forests are increasingly expected to play a mitigating role in the context of global warming through carbon sequestration (in the ecosystem and in wood products) and through substitution for fossil resources. Consequently, phytophagy of forest insects and its consequences for forest health generate strong interferences with human activities and forests ecosystem services. Insect damages can lead to quantifiable economic losses and/or losses in amenity and amenity value associated with decreases in ecological and social services provided by forests (Nagelheisen et al. 2010). In Europe, over the period 1950-2000, damage caused by phytophagous insects accounted for 8% of the total damage caused by forest disturbances (about 2.88 million m³ per year between 1958 and 2001), and in North America, large bark beetle outbreaks over the past 20 years have been the leading cause of major forest disturbances, ahead of hurricanes, tornadoes and fires, with average costs exceeding USD $2 billion (Grégoire et al. 2015). Insects specialized on tree reproductive structures can reduce the supply of high quality seeds for ornamental plants, reforestation, afforestation and conservation in established orchards and selected tree populations (Boivin and Auger-Rozenberg 2016). There are also health risks to humans and domestic animals associated with the expansion and spread of urticating insects (e.g. processionary caterpillars) in visited and exploited forests (Moneo et al. 2015).
6.2.2. **Definition of the risk associated with phytophagous insects in forests**

The risk of phytophagous insects for forests is defined as the combination of a hazard, the vulnerability of tree populations and the ecological and socio-economic challenges associated with insect impacts in forest ecosystems (Fig. 6.1).

The *hazard* is determined by the existence of populations of insects at low abundance (endemic phase) likely to show progressive or eruptive population growth (epidemic phase) in response to favourable environmental factors (e.g. resource abundance, climatic conditions, droughts, disturbances). Hazard can also be determined by the probability of invasion of one or more exotic insect species. Insect hazard analysis can be done by mapping: (i) areas of potential presence by modeling ecological and climatic niches, (ii) areas of presence carried out by observations and monitoring of established populations, or (iii) changing areas of occurrence by observations and modelling of natural or assisted expansion. These approaches are based on knowledge of the distribution and abundance of insect species and their impact on trees (e.g. mortality, dieback, growth loss). Adaptive processes in insect populations, their ecological impacts in new colonized areas, and the role of biotic interactions (e.g. with natural enemies) in modulating insect dynamics can also be investigated (Liebhold and Tobin 2008). The establishment and updating of lists of alien species (Roques et al. 2008), the development of tools for early detection and prediction of invasiveness potential in native areas (Mansfield et al. 2019), and tools for detection and tracing of invasion routes of invasive alien insect species are essential approaches to biological invasion-related hazards (Hulme et al. 2008).

The *vulnerability* is a change in the level of attractiveness of trees for insects and in the ability of insects to exploit these trees. It is primarily related to levels of fruiting, defoliation, dieback, and tree mortality. Vulnerability of forest stands to phytophagous insects is often related to tree size, stand structure and species composition, and stand proximity to human activities. Such characteristics directly influence insect demography and damage intensities to trees. Increased tree diversity can for instance increase the resistance of planted forests to insect pest attacks at different spatial scales through association effects between tree target and non-target species for insects (Jactel and Brockerhoff 2007). Moreover, vulnerability depend on the occurrence of predisposing factors to insect attacks that are generally considered as stress factors for tree. These can be biotic (parasites, pathogens, inter- and intraspecific competition) and abiotic (droughts, heat waves, fires), or a combination of the two (Manion 1981; Jactel et al. 2012).
The challenge is linked to the consequences of insect pest damage at the levels of economy (tree species with market value), ecology (diversity and functioning of the ecosystem) and health (transmission of pathogens to trees and species dangerous to human health). The challenges associated with phytophagous insects in forests are the economic or ecological importance of the species of trees attacked, the extent of damage caused by insects or the possibility of entomophilic transmission of highly problematic pathogens (e.g. the pine wood nematode). Such challenges respond to the alteration of ecosystem services related to the supply of forest reproductive material or timber, to the structuring of habitats of patrimonial or ecological importance, and human health issues in exploited, recreational, urban and peri-urban tree areas. Challenge issues can be addressed through links with forest managers, observation and monitoring networks, municipalities and health services. The ability to develop tools for insect population control (biological or environment-friendly) and decision support in control also define these issues.

6.3. Phytophagous insect-tree interactions and climate change

Changes in the abiotic environment are likely to directly affect various characteristics of tree populations (e.g. phenology, seed production, growth, mortality). Direct effects of climate are also expected on phytophagous insects whose survival, reproduction, dispersal and distribution are closely related to temperature and precipitation conditions (Bale et al. 2002). In addition, there are indirect effects of the environment on insect populations through alterations of tree health and through responses of insect competitors, facilitators or natural enemies to environmental change. Finally, abiotic changes can indirectly affect tree populations through their direct effects on the abundance and distribution of phytophagous insects. Successive episodes of drought can directly limit tree survival through irreversible physiological alterations (Allen et al. 2010), or indirectly when these conditions weaken trees sufficiently to reduce their ability to resist bark beetles (killing-tree insects that feeds on their host’s phloem). This generally limits the resilience of trees, which may prevent them to avoid death before more favourable conditions return (Manion 1981). However, drought may also induce significant changes in the nutritional quality of trees (e.g. water content, carbohydrates, nitrogen) and can trigger the synthesis of antagonistic compounds to phytophagia, particularly in leaves (Jactel et al. 2012). In Spain, a remarkable disruption in the population dynamics of phytophagous insects may have been associated with the impacts of droughts on holm oak (Carnicer et al. 2011).

Climate change also alters the spatial and temporal patterns of interspecific interactions, and in this respect phenological divergences between insects and their host plants can be observed with potentially strong population consequences (Singer and Parmesan 2010). Many forest phytophagous insects affect their host plants only during specific vulnerable periods that often result from drastic changes in the abiotic environment (Rouault et al. 2006). Successive drought episodes can affect directly tree physiology and survival (Allen et al. 2010), or indirectly when higher temperatures and lower tree resistance trigger severe forest insect outbreaks (Durand-Gillmann et al. 2014). However, the interdependence between climate, biotic factors and tree dynamics remains complex to predict. Drought-induced changes in tree nutritional quality (water, carbohydrates and nitrogen contents) or in tree defence mechanisms can limit the development and the damages of parasites (Jactel et al. 2012). Extreme droughts may even be directly involved in the collapse of herbivorous populations at wide scales, but drought can also affect negatively tree physiology and decrease the effectiveness of tree resistance mechanisms to pathogens and parasites (McIntyre et al. 1996).

Fire ecology provides interesting additional examples of the complexity of integrating interdependencies between trees, biotic and abiotic factors. Bark beetle outbreaks and forest fires have jointly increased in extent and severity during the last decades in the USA, raising concerns about their possible interactions (Simard et al. 2011). Bark beetle outbreaks may increase the risk of active crown fire due to the great quantities of dead and ladder fuels that they generate. However, Simard et al. (2011) suggested that risk of active crown fires may not change in the short term but it may rather increase in the decades following an outbreak. Importantly, bark beetles are thus likely to
indirectly affect non-attacked trees through subsequent enhanced fire risks. This clearly illustrates the critical need to integrate the interplay between the abiotic environment, biotic interactions and trees health. This is a major issue with regard to narrow endemic habitats where climate warming is supposed to be a major threat to range-restricted species by shrinking the area of their suitable habitats.

6.4. Interactions between trees, insects and climate in Mediterranean areas

The Mediterranean basin is a major former glacial refuge that is now considered a hot spot for plant and animal diversity at the inter- and intra-specific levels (Lefèvre and Fady 2016). There are nearly 201 species of woody plants, 100 species of trees (compared to 30 species in the rest of Europe for a surface area 4 times larger) and a high level of endemism (60%, Quézel and Médail 2003). Phytophagous forest insects also show particularly high levels of diversity and endemism in many groups. These are most likely the result of tree biodiversity and endemism, the diversity of habitats available for insects on a single tree and the diversification of insect guilds, particularly for specialists (Boivin and Auger-Rozenberg 2016). Mediterranean forests are at the forefront of the global change effects on terrestrial ecosystems due to the increased frequency and intensity of disturbances (fires, droughts, urbanisation and fragmentation, pest epidemics), compared to other European forest ecosystems (Lefèvre and Fady 2016).

Mediterranean phytophagous insect communities show that they are already affected by global change, however, global change effects and their consequences on tree-insect interactions remain much less documented in Mediterranean areas than in temperate or boreal ones. It is also difficult, if not risky, to extrapolate observations made in northern areas to southern areas. For example, temperature increases in temperate and boreal zones are expected to be more favourable to insects in terms of development and winter survival, while the threshold of lethal maximum temperatures can be reached more quickly in Mediterranean areas (Lieutier and Paine 2016). The effects of increased CO2 on tree growth are most often always positive in temperate and boreal regions, but they are already largely negative for some tree species in the Mediterranean due to water limitations (Nageleisen et al. 2010). This is therefore likely to affect different groups of phytophagous insects differently. For instance, defoliators may be negatively affected by climate change effects on their host plants as they generally seek for good quality leaves, while bark beetles may be favoured by weakened tree defenses resulting for successive droughts (Jactel et al. 2012). Increased temperatures and drought frequency are expected to make southern and continental parts of Europe less suitable for heat-sensitive species, which is likely to result not only in range shifts to the north, but also in contractions of distribution areas (Netherer and Schopf, 2010). Many projections therefore remain to be made, and such direct and indirect climate effects on both insects and trees make long-term predictions for sensitive Mediterranean forest ecosystems more complex.

6.5. Seed predators and bark beetles, two potentially important insect threats for narrow forest habitats

Seed predators and bark beetles directly affect tree survival at the embryo and adult stages, respectively. The demographic and evolutionary consequences of their feeding activities in narrow habitats are thus likely to differ from most forms of herbivory which often result in only the partial removal of tissue from individual plants.

6.5.1. Preliminary seed predators

General features: A distinction is usually made between pre-dispersal seed predators that feed on tree reproductive structures (buds, flowers, pollen, fruits, cones and seeds) as long as they remain on their parent plants, and post-dispersal predators that forage for reproductive structures after they have been dispersed. The clumped and conspicuous nature of seeds before dispersal tends to favour specialization in pre-dispersal predators, while post-dispersal seed predators often share generalist habits to forage for an inconspicuous and scattered resource that is often at
low density. Insects are considered the most important animal predators of reproductive structures during the pre-dispersal phase of seed development (Crawley 2014). Seed loss to insects, such as ants, weevils or bruchids, can also reach important levels during the post-dispersal phase (van Klinken and White 2014).

**Damages and impacts:** Conifer predispersal seed predators affect trees’ reproductive output in two, possibly cumulative, ways. First, insects’ feeding activity on the seed storage organs reduces the amount of reserves available for germination and early development of seedlings (Lesieur et al. 2014). Second, consumption of the seed embryo makes further seed germination impossible. Aside from direct seed consumption, insect attacks can also result in the abortion of the seed-bearing structures or selective abortion of co-occurring seeds (Meyer et al. 2014), a constraint to seed release due to resin spill or seed fusion to cone scale following insect attack (Lesieur et al. 2014), or the introduction of tree pathogens (Luchi et al. 2012). Important concerns arose on the potential impacts of seed predators on endangered tree species with poor natural regeneration success (Guido and Roques, 1996). Similar concerns relate to the potential impact of these insects on the adaptation of natural forest ecosystems to climate change as well as adaptation strategies such as assisted migration and the implications of altered seed productivity. Finally, reforestation and afforestation programs require high quality seeds from selected genotypes that represent high value material, for which the establishment and maintenance of high seed outputs relies on low tolerance for cone and seed damage.

**Management options in narrow habitats:** Protection of cones and seeds from insects is generally a complex task, especially in narrow endemic habitats. This is partly due to the cryptic internal feeding habits of many seed predators that makes them difficult to detect and control, and to the spatial heterogeneity of cones at both tree and stand levels. Predispersal seed predators are essentially controlled within seed orchards and tree stands providing seeds or fruits as food, which share some of the features of an agro-ecosystem but also those of a forest ecosystem. This adds significant complexity in establishing effective pest management programs. Seed orchard managers conduct integrated pest management (IPM) programs aiming at combining all suitable direct (e.g. insecticide treatments) and indirect (e.g. use of natural enemies) techniques to maintain populations below a desired economic threshold, but such procedure are nearly impossible to conduct in protected areas such as narrow endemic habitats. In seed orchards, prophylaxy remain the most easily applicable management practice of seed insects and it may also affect insect densities and hinder population growth in seed orchards. The removal of all cones or fruits at harvest is an effective strategy to reduce local seed predator populations (Turgeon et al. 1994). Alternatively, destruction of unharvested cones or fruits left on the soil surface by tilling and discing the seed orchard floor may contribute to reduce overwintering populations of many pest species. Such prophylactic tactics are likely to prevent the formation of local pest inoculums. This can be particularly usefull against species in developmental arrest over several years (i.e. prolonged diapause), and this make new stand infestations only rely on immigration from surrounding areas. When possible, combining the limitation of a local inoculum with the removal of host trees from within a substantial radius of a seed orchard may be thus a major obstacle to seed losses to seed insects. Finally, careful inspection of collected seed lots can prevent or limit long-range dispersal of seed insects at the regional scale, to other countries of the Mediterranean Basin, and beyond.

### 6.5.2. Bark beetles

**General features:** Declines in tree populations are usually triggered by repeated droughts and heat waves, but tree individuals can display resilience abilities, which in some cases allow them to avoid mortality and benefit from the return of favorable conditions. One factor driving resilience opportunities is the presence or the absence of biotic aggravating factors such as bark beetles (Durand-Gillmann et al. 2014). Bark beetles commonly feed on cambial tissues of dead, recently wounded and even heavily defended healthy trees. Dispersing adult bark beetles colonize their host tree by boring an entrance hole through the bark. Females dig a tunnel near the cambium where they
lay their eggs, from where developing broods dig their own tunnel by feeding on the phloem. In conifers, tree defense mechanisms against bark beetle attacks involve a system of toxic resin ducts of variable size and number and the synthesis of defensive chemicals to the entrance site. The likelihood of tree survival to bark beetle attacks is closely linked to the interplay between tree’s ability to mobilize defense mechanisms and the abundance of attacking bark beetles. Stress factors that weaken tree defenses have been usually associated with the start of bark beetle epidemics (Marini et al. 2012). Drought is particularly prone to increase tree susceptibility to insect attacks and to induce changes in tree’s physiology (Netherer et al. 2015). Bark beetles are inherently labile forest insect populations displaying transitions between endemic states, during which they reside in stands at very low densities and kill only a few weakened trees, playing an important role by preparing the substrate for a high diversity of saproxylic organisms. In epidemic states they reach very high densities over large areas and cause high tree mortality at both stand and landscape scales (Kausrud et al. 2011). Knowledge on the impacts of insect epidemics on tree individuals and populations benefited from to extensive research, but less attention has been paid to those associated with endemic states. Global warming increases the risk of damage in the Mediterranean basin and of northward extension of some species in current temperate regions. Life cycles of Mediterranean bark beetles display diverse types of volitism, sister broods and various type of feeding habits (Lieutier et al. 2016)

**Damages and impacts:** During epidemics, bark beetles generally infest trees by a sudden and massive attack, with adverse effects on tree vitality. Attacked trees may initially show no specific symptoms, but the canopy of the trees infested in spring and early summer turn yellow within a few weeks, then reddish and finally dries up completely. In conifers, massive fall of green needles can be observed (Lieutier et al. 2016). Following late summer or autumn attacks, the infested trees can keep green foliage throughout the autumn and winter as canopy symptoms may appear only in the following spring. In these cases, the attack is often discovered months later because absence of symptoms on the canopy does not allow a prompt identification of the infested trees. However, infested plants die within a few months. Insect galleries alter the functionality of the bark tissues (cambium and phloem), which dry out and die in a few months. Tree mortality generally follows exhaustion of tree defenses by associated fungal species and beetle boring activity, and probably involves several factors including various fungal species (Lieutier et al. 2009). Bark beetle epidemics often result from extensive damage both directly, through death of plants, and indirectly through the economic reduction of timber quality due to possible wood discoloration by symbiotic blue -staining fungi. Some species can carry pathogenic fungi to living trees. Severe mountain pine beetles (*Dendroctonus ponderosae*) outbreaks can lead to up to 90% mortality of tree basal area as observed in North America (Harvey et al. 2014) and European forests and plantations are regularly managed to reduce bark beetle impacts (Kausrud et al. 2011). Bark beetles are thus considered as important disturbance agents in forested areas worldwide, affecting critical processes in forest dynamics and forest services with important subsequent societal issues (Weed et al. 2013; Morris et al. 2018).

**Management options in narrow habitats:** The management of bark beetle populations includes two main approaches that are based on the monitoring of forest stands for early detection of attacks and on the control of new tree infestations. The effectiveness of bark beetle management practices in natural and protected areas varies considerably depending on whether they are implemented in a timely and sustained manner, one objective being the anticipation of epidemic phases. The rapid identification of currently infested trees is essential to maximize efficient management response before bark beetle outbreaks. Phytosanitary survey carried out once or twice a year to assess general forest health conditions and damage caused by main bark beetle species is a first step to decide about management strategies and action priorities. Such survey should be conducted by trained field operators in collaboration with entomologists and scientific institutes, with the objective of controlling the initial infestations to prevent more serious damage. More specific survey
may be carried out in case of rapid increase in population density of target bark beetle species, or after climatic events known to trigger outbreaks. Additional field monitoring activities rely on the assessment of bark beetle occurrence and population density using traps baited with attractive lures. Inferences on the flight activity, phenology and volatinism can be made from trapping. Such information may be used as a decision-making tool for control in sensitive areas if mean catches per trap can be correlated with damage, i.e. infested cubic meters or killed trees (Wermelinger et al. 2004). Efficient trapping techniques may provide estimates of spatio-temporal variation in bark beetle population density. Traps may be baited with commercial specific aggregation pheromones of the most common bark beetle species (see www.pherobase.com), or with host volatiles (mainly a-pinene and ethanol) if pheromones are not available.

Control strategies of bark beetle populations in natural forests can be both indirect and direct. Indirect control options primarily rely on silvicultural management including clearing, thinning and pruning. This may help sustainability of forest health and increase forest resiliency of forests to bark beetle infestations and other disturbances. Moreover, all silvicultural practices that reduce excessive tree density likely reduce the vulnerability of individual trees, stands, and forests by strengthening insect resistance mechanisms (Fettig and Hilszczanski 2015). Tree felling and harvesting should be carried out in winter or early spring to reduce the risk of attacks during the spring-summer flight of bark beetles. When felled logs cannot be removed from the forest for several months, log debarking may be efficient in preventing further bark beetle colonization and insect population build-up. Direct control options mainly rely on sanitation felling, i.e. the felling and removal or treatment of infested trees in order to kill bark beetle brood before adult emergence. If economically feasible, transportation of felled trees to mills allows to kill brood during timber processing. Otherwise, felled trees have to be burned or debarked in situ (Fettig and Hilszczanski 2015). The efficiency of sanitation felling depend on its timing relative to the phenology and biology of the target bark beetle species. Tree debarking after adult emergence may be useless and unfavorable to bark beetle predators which usually emerge later than their prey. The target period should be during full larval development, when females have already laid all their eggs. One should however note that, in natural forests, remaining dead or dying trees likely constitute breeding substrate for bark beetle natural enemies. Semiochemicals (aggregation pheromones or host-volatiles) can also be used to employed to attract and concentrate the bark beetle population in a baited trapping device. But such mass-trapping control technique may affect insect population density only with an available efficient aggregation pheromone, which is more likely to ensure high levels of insect captures (several thousands of insects per trap and season). Attractants may also be used on individual trees or small groups of trees or logs (namely trap-trees or trap-logs) to favour and concentrate insect attacks prior sanitation. Trap-trees should be baited and set up vertically to maximize their attractiveness to insects, and then removed or debarked.

6.6. Conclusion

One important outcome of this chapter is that the interdependence between the components of global change, tree populations and forest insect populations results in a number of constraints to prediction. These result from the different directions (favourable/ unfavourable effects) and the expected levels (direct/ indirect effects) of interactions, a multi-trophic dimension in which other actors (predators, competitors, facilitators) can modulate tree-insect interactions, and significant variability in the response of hosts, phytophagous insects and associated communities to environmental change. In addition, from the point of view of entomological risk to forests, global change and forest management practices can both play a major role. Global change will mainly act through its effect on the occurrence of hazards (e.g. pest outbreaks, spread of invasive species, higher frequency of storms) and on forest vulnerability (e.g. water stress reducing insect resistance). Forest management will mostly influence forest vulnerability (e.g. thinning to improve the vigour and therefore the resistance of trees to pests) and the impacts of insect damage (e.g. reducing the standing volume of forests by short rotation).
6.7. References


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& Schiff.) (Lepidoptera: Thaumetopoeidae) on annual diameter increment of *Pinus brutia* Ten. in Turkey. Annals of Forest Science 62: 91–94.


PART Α
CHAPTER 7

7.1. Introduction

In nowadays there are tremendous environmental risks due to environment’s intensive exploitation and extreme centralization of population. Some of the major problems that have been observed are the habitat destruction, the lack of biological diversity, pollution of land and water resources and destruction of cultural heritage (Economou et al. 2015). However, it is obvious that through various (research) projects and actions that have been developed through the years, the environmental projects create many benefits for both their neighboring areas and the wider Cypriot society, too. In general, it can be declared that many environmental projects aim at protecting of habitats, due to their importance for both local communities and the wider society, too. Nevertheless, it should not be underestimated the fact that the establishment of developmental and/or energy projects constitute a challenging process, since they depend on various factors (e.g. fiscal, legal, administrative, etc.), while similar projects become more complicated when they involve local communities and have impact on cultural environments (Poulis and Arabatzis 2015).

Due to the increased complexity of designing and implementing institutionalized interventions in environment, the international literature focuses on economic analysis of the impact caused by these interventions (i.e. Fisher et al. 2008). Even though it is recognized that environment is one of the major factors of human prosperity, the long-term consequences of local interventions are not been directly understood by involved stakeholders. The consistent analysis and quantitative representation of the fiscal consequences on environment based on external interventions consist useful elements for the development of sustainable policies and awareness raising for all involved parties/stakeholders, aiming at improving their cooperation.

In this chapter there are references regarding the benefits and the socio-economic impact of environmental projects to the neighboring areas, in the wider area of Troodos and Cyprus, too. In more details, the benefits can be focused on the following categories: (i) Information and awareness raising, (ii) Active involvement of communities, (iii) Volunteers’ involvement, (iv) Restriction of hazardous activities, and also (v) Sustainable development and green economy. These categories are further described, explored and discussed in this chapter.

7.2. Information and awareness raising

In nowadays, the lack of people’s awareness and education and their wrong believes regarding their personal and collective responsibilities are considered the major reasons for environmental downgrading. The lack of education is probably connected to the medium-term economic consequences, such as in cases in which local communities want to achieve quick benefits, and not being able to realize the long-term consequences. It has to be underlined that in nowadays we have been moved from environmental education towards the education for the environment and sustainability (Flogaiti 2011). In order for citizens to be able to participate in this attempt, it
is necessary to have the appropriate education. The ecological consciousness can be enhanced through the official training process and through popular science papers/reports, which are reader friendly (Modestou 2018). Education should drive people to acquire new knowledge for understanding new concepts and conditions that relate to various environmental issues, in order to realize both their and other organization’s role and responsibility in the making and solution of these problems (Economou et al. 2015). Also, digital games is considered to be a mean for information and awareness raising of the wider audience regarding environmental issues; digital games aim to educate through a gaming environment. International literature states that digital games provoke student’s active participation through exploration, experimentation, competition and collaboration; in addition, they support education through an advanced visualisation, while the gamer develops problem solving strategies and critical thinking, practices that can lead him/her to new learning outcomes (Nari 2018). A community has to move towards this direction, in order to empower wider society’s education and awareness raising, aiming at improving the relation and solidarity between individuals and environment.

7.3. Active involvement of communities

The active and substantial involvement of neighboring communities is considered inevitable and should be ensured. Public participation aims at enhancing citizens’ participation during the formulation of the agenda, the decision-making process and the policy formation (Deligiani 2018). The degree of citizens’ participation in a participatory process varies according to the degree of control they can gain on the information, the decision-making process and its implementation (Stratigea 2015). People’s participation in the decision-making process regarding issues that affect them (directly or indirectly) is considered a democratization element of the process, which is not unfold beyond and without them rather than with them (Makridemitris and Pravita 2012). Of course, that there are cases in which the degree of participation varies between zero (participants are pathetic listeners of the decisions and they just be informed about them) and/or absolute participation, a case in which the full initiative of the process is provided to citizens by giving them access to the information, the decision-making process and its implementation (Stratigea 2015). At a higher level, it can be observed citizens’ enhancement, since they are the main recipients of the decisions. This kind of participation entails that citizens and local communities expand their capacity to determine the problems they face (prioritise) and their solutions, in other words to be able to decide on the issues affect their life (Stratigea 2015). Therefore, local community’s participation has to be enhanced and empowered, in order for citizens to develop active participation in the activities taking place in their region, rather than adopting a passive behaviour and accepting all policies decided by others.

7.4. Volunteers’ involvement

Many studies revealed the significance of educators’ participation in environmental projects. A relevant study shown that educators who consolidated volunteering spirit as an altruistic offer towards the society, they gained specialized knowledge regarding the protection of the environment, they developed collaboration and collective culture with all education stakeholders (e.g. colleagues, parents, students), while they became more extrovert towards the local community at both social and professional level; finally, they improved their teaching methods and communication skills (Kolovos 2019). Also, the significance of volunteering and how educators could be seen as a role model have been recognized. Moreover, close relation and interaction between education-awareness raising-activities regarding environmental issues was stressed, while the ways that volunteers could be involved in similar project was underlined. Finally, it was stated that each society should aim at promoting environmental culture and empowering the relationship between individuals and environment.

7.5. Restriction and prevention of disastrous activities

Human activity in habitats increases the danger of their conservation and rescue. Fires, illegal logging, collection of rare flowers by visitors, water pollution (Lazaridou 2009), environmental
over-exploitation, the lack of environmental friendly consciousness (Charalampous and Tziortzi 2017), waste disposal, poaching, continuous construction activity, uncontrolled grazing and use of pesticides in farming, the developed road network and the lack of environmental education lead these regions/areas to destruction (Mantzinas 2015). Therefore, it is inevitable to design a new policy regarding sustainable development, which should aim towards the development and conservation of the habitats, the flora and fauna, the conservation and improvement of the water and ground quality, while in the same time other disastrous activities should be prevented.

Environmental degradation has enormous economic consequences in human societies and productive activities. If, for example, the environment suddenly stopped the energy production, the vast majority of human activity could not be feasible (NEF 2011). Similarly, any other environmental disastrous activities have serious economic consequences. WWF in collaboration with Global Trade Analysis Project and Natural Capital Project support that if the environmental disastrous is not reversed, there will be tremendous consequences in the global economy, which are calculated to many trillions dollars (Global Futures 2020). According to this, FAO has published at global level particular guidelines regarding environmental impact assessments (EIA) for all environment related projects, in order to prevent from possible negative consequences on the environment (FAO 2011). Similarly, European Union has published guidelines, in order to achieve a reasonable usage of both public and private funds (European Commission 2017). International literature provides great data regarding the extreme cost of fires (i.e. Ramachandran 2002) and the degree of public spending towards their prevention (i.e. Amacher et al. 2005). In the greater region of European Mediterranean, the overall area that has been burnt for the period between 2007-2014 has been doubled (Varela et al. 2014). According to Forest Department data, the cost of fire rehabilitation regarding the latest fires in Cyprus was calculated between €453 per hectare (Saittas region, 2007) and €1149 per hectare (Argaka region, 2016). Moreover, the defaming cost of Cyprus as a touristic destination due to poaching is notably high, either directly – since these conditions can lead to high fines by European Union – or indirectly, through the decrease of touristic flow by conscious tourists (Orountiotis et al. 2011).

7.6. Sustainable development and green economy

Human pressures downgrade habitats and cause serious environmental problems. This condition leads to a gradually loss of forest area through alteration and deforestation. There are examples, e.g. the environment at Prespes Lake and the national forest park at Schinias-Marathonas, which are threatened by activities such as poaching and forbidden fishing and logging. These activities lead towards the disappearance of various species of birds and the continuous reduction of rare animals populations (Ntoba and Tsouri 2008), while it should not be ignored the fact that developmental and protective projects are closely connected to the local community.

At a first glance, the economic activity seems incompatible with the environment protection. In reality, all ecosystem functions of a forest comprise economic functions (Pearce 2001); nevertheless, basic forest functions are not under negotiation in open market and therefore it is difficult to determine their value in advance. In this context, the forest transformation (disaster), rather than conservation, seems to be the most economically rewarding choice. However, the negative consequences of forest loss, e.g. reduction of the quality of drinking water or the degree of flood protection they offer, has actual cost for the community; overall, these conditions are seriously considered only in retrospect.

Emphasis should be given to the natural beauty of a park/region of a special beauty/interest in order to attract visitors (Filippakis 2017) and also to achieve a methodology of coexistence of ecological initiatives and activities such as farming, livestock farming, fishing and tourism (Ntoba and Tsouri 2008). After a period of economic development due to environmental exploitation and the dramatic consequences such as the lack of various kinds of flora and fauna and habitats’ downgrading, there is an attempt to prevent these adverse consequences (Kitsos 2019). Therefore, as it was stated above, the management of habitats has to be administered by the
central government and other stakeholders (e.g. communities, organisations, volunteers), who have direct and/or indirect interest for their protection. It is of great significance to achieve local authorities' direct involvement, after they have been informed and trained. The joint design of a common action framework should consider the concepts of communal and individual responsibility, meaning that it is not feasible for local communities to exclusively take on the responsibility of protecting a habitat, when the indirect benefits are diffused to the overall economy of the country.

The development of the green economy, with direct liquidated economic outcomes, can be the connecting link between business activity and environment protection. The continuous and methodical protection of a habitat can enhance the development of a rural tourism force, which is a mean of economic development. Through green economy, various communities have the chance to achieve economic development by restricting the reduction of natural and cultural resources, utilizing a group of measures such as taxes, subsidies, commercial plans and voluntary approaches (Geronatsios 2017). In addition, in green economy, the increase of incomes and employment that is strengthened by private and public investments, they take advantage of the energy and natural resources and prevent from the disaster of biodiversity and ecological systems (Vavoulides 2017). Green economy, even though contributes to the local and wider economy, ensures the sustainable development of the ecosystem, since this is its main developmental stock. In case that the ecosystem is destroyed, the concept of development is reversed, since the main assets that creates/develops the future flows is destroyed. In Cyprus, there is a great scope for the development of the green economy, which can be achieved through the joint attempts and activities between the central and local level of authority.

7.7. References


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Mantzipas R. (2015) Recording and data collection for a number of wetlands located in the region of Larnaca and free Famagusta in Cyprus, with a view of evaluating their sustainability. Dissertation, Faculty of Geotechnical Sciences and Environmental Management, Cyprus University of Technology.
8.1. The narrow endemic species of Cedrus brevifolia

*Cedrus brevifolia* (Hook. f.) A. Henry (Cyprus cedar) is an endemic coniferous species to Cyprus, belonging to the Pinaceae family (Meikle, 1977). The general taxonomy of the species is (Euro+Med 2006 - 2020):

- Division: Tracheophyta
- Subdivision: Spermatophyta
- Class: Gymnospermae
- Subclass: Coniferae
- Order: Pinales
- Family: Pinaceae
- Subfamily: Abietoideae
- Genus: *Cedrus*

It belongs to the genus of *Cedrus*, which is one of the eleven commonly accepted genera in Pinaceae (Farjón 2001), with a wide but discontinuous distribution in the old world during the pre-Quaternary time, and a high geographically disjunctive distribution in the circum-Mediterranean and west Himalayan (Farjón 1990, 2001; Pons 1998). The genus is represented by four closely related species, with their present natural geographic distribution being limited to the Himalayan Mountains for *Cedrus deodara* (Himalayan cedar); to Lebanon, Syria, and Turkey for *Cedrus libani* (Lebanon cedar); to Morocco and Algeria for *Cedrus atlantica* (Atlas cedar) and to Cyprus for *Cedrus brevifolia* (Cyprus cedar) (Debazac 1964; Vidakovic 1991). The cedar genome is relatively large and was evaluated to be 32.6±0.6 pg per 2C or 15.7 X 10^9 base pairs per 1C; the composition in G/C was calculated to be 40.7% and the diploid chromosome number was found to be 2n=2X=24 (Bou Dagher-Kharrat et al. 2001).

*C. brevifolia* is an evergreen, coniferous, resinous tree up to 30 m high (Fig. 8.1). Its distinguishing character is the shape of its trunk, which can be maintained straight for the whole duration of the tree's life. The crown is at first pyramidal but later becomes broader and tabular in shape (Meikle 1977). Needles are 15 x 1.5-2 mm, spirally positioned on long shoots (macrocladia) and in fascicles on dwarf shoots (Tsintides et al. 2007). Cedar species can be monoecious or dioecious and this can change over time, an observation that is also true for *C. brevifolia* (Tsintides et al. 2002; Krouch et al. 2004; Pattichis and Kyriakou 2013). Male cones are cylindrical, yellowish, on the tops of the young shoots and they release pollen in autumn (September – October) (Fig. 8.2). Female cones are reddish (Fig. 8.2) and mature during August to September, two years after fertilization (Pijut 2000). The mature cones are greyish-brown, barrel-shaped, 8 x 5 cm in size and each ovuliferous scale supports two (2) seeds (Fig. 8.2). Seeds are brownish, pear-shaped and winged (Meikle 1977, Tsintides et al. 2002).
Despite that *C. brevifolia* is classified as a separate species in the *Cedrus* genus and is one of the four cedar species in the world, a debate on its phylogenetic structure exists. Studies on morphological and anatomical needle characteristics argued that *C. brevifolia* conserves the most ancestral type of needle compared to the other three species (Jasińska et al. 2013), leading to supporting the argument of the taxonomic position of *C. brevifolia* as a species (Farjon 2001; Jasińska et al. 2013). However, phylogenetic studies argued a genetic relationship between *C. brevifolia* and *C. libani*, especially with provenances from Turkey (Scaltsoyiannes 1999; Bou Dagher-Kharrat et al. 2001, 2007; Qiao et al. 2007). Molecular clock analysis estimated the divergence time of *C. brevifolia* from *C. libani* about 6.56 (±1.20) million years ago (Qiao et al. 2007), which is approximated to the Messinian Salt Crisis time (Mai 1989; Krijgsman 2002; Hellwig 2004). The long historical presence of *C. brevifolia* on the island of Cyprus has also been mentioned by ancient authors (i.e.}

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**Figure 8.1**: Shapes of *C. brevifolia* trees (Photos: ©Nicolas-George Eliades)

**Figure 8.2**: Male (left) and female (center) flowers of *C. brevifolia*, as well as cone of *C. brevifolia* (right) (Photos: ©Nicolas-George Eliades)
Theophrastus 371-287 B.C.; Plin 23-79 A.D.). Cedrus brevifolia showed the highest stomatal conductance (Ladjal et al. 2005), while it is characterised by the lowest growth rate but was found to be the least drought-sensitive from the other cedar species (Ducrey et al. 2008).

Studies on genetic variability revealed a high level of genetic diversity, while the species is not characterized by a founder effect, appearing not to have experienced severe bottleneck events or extensive genetic drift (Eliades et al. 2011). The assumption is that the high genetic diversity of the species is most likely due to its long-term presence in the mountains of Cyprus and that the species originated from a widespread congener species (Eliades et al. 2011). In addition, the admixture of several C. libani provenances in the island during species migration in Cyprus (Bou Dagher-Kharrat et al. 2007; Eliades et al. 2011) most likely during the Messinian period (7–5 Mya) (Eliades et al. 2011), could be another significant factor that influenced the high genetic diversity of Cedrus brevifolia, despite its narrow distribution. The unique population of Cyprus cedar is divided into remaining sites (see below) that showed low but significant genetic structure, while significant isolation-by-distance was observed among the identified sites (Eliades et al. 2011).

In the last two decades the Department of Forests (Cyprus) has established an inventory system in the Cyprus cedar area, with the most recent forest inventory having estimated that the natural growing trees (in natural stands) are 22,418 (±3626) with a diameter at breast height over 12 cm (Department of Forests 2012). The conservation status of the species has been evaluated based on the criteria of the International Union for Conservation of Nature (IUCN) and this is characterized as Vulnerable (VU) and it is included in The Red Data Book of the Flora of Cyprus (Tsintides et al. 2007).

8.2. The habitat type 9590 *Cedrus brevifolia forests (Cedrosetum brevifoliae)

The endemic woody species of C. brevifolia consisted the keystone species of “C. brevifolia forest”, which occurs only in a small area of ~263 ha on the top hills of Pafos Forest (Fig. 8.3). The forest is characterized by limited altitudinal distribution from the upper limits of the meso-Mediterranean to the mid supra-Mediterranean zone (altitude of 900–1400 m above sea level – Department of Forests 2005). Cyprus cedar forest represents 1.66% of the island's total forested area and at the same time corresponds to less than 0.2% of high vegetation in Cyprus forests. It forms small relic sites within its natural distribution range; the largest subpopulation is found on Tripylos mountain (209 ha), while smaller subpopulations are found on surrounding areas such as Mavroi Gremoi (13.4 ha), Selladi tis Ellias (8.8 ha) and Throni (6.5 ha) (Tsintides et al. 2007; Eliades et al. 2019).

The ecological importance of C. brevifolia forest was recognized at a European level, since it has been coded and included in Annex I of European Directive 92/43/EEC (Habitats Directive) as a habitat type, namely “9590 *Cedrus brevifolia forests (Cedrosetum brevifoliae)”. This habitat type was classified as a priority habitat type in Annex I of the Habitat Directive, which means that sustainable management practices need to be implemented in order to improve its conservation status. Habitat type 9590* Cedrus brevifolia forests (Cedrosetum brevifoliae) is one of the eleven forest habitat types that occur in Cyprus and one of five endemic habitat types found on the island (PAF 2013). The distribution area of habitat “9590 *Cedrus brevifolia forests (Cedrosetum brevifoliae)” is included within the Natura 2000 network, under the Natura 2000 site SCI “Koilada Kedron – Kamos” (CY 2000006), a site which was upgraded in 2019 (see Chapter 10). Following recent evaluation regarding its conservation status and based on Article 17 of the Directive, this is characterized as “Favourable”.

The habitat type 9590* develops on rocky mountain slopes made out of diabase (igneous rock derived from erosion of Troodos ophiolite) (Eliades et al. 2018), where relatively shallow and eroded rocks of great and steep slopes are found, with acidic pH (5-6.75) (Eliades 2015). The sites where the habitat type occurs are characterized by Mediterranean climate with lower average annual temperatures than the rest of the island (Tsintidis et al. 2007).
The habitat type 9590* shapes pure formations appearing scattered within a wider area of 106 ha, while the mixed stands (formations) appear in an area of 184 ha (Eliades et al. 2019). The mixed formations include mixed habitats (forests) with other habitats such as habitat 9540 Mediterranean pine forests with endemic Mesogean pines (keystone species: *Pinus brutia*) and/or habitat 9390*Scrub and low forest vegetation with *Quercus alnifolia*.

Apart from *C. brevifolia* which is the keystone species of habitat type 9590*, other flora species are involved in the composition and structure of the habitat. Other tree species that occur in the...
canopy layer of the habitat is *P. brutia*, while on the understory layer grow evergreen and sclerophyllus shrubs and subshrubs of vegetation class Quercetalia ilicis: Quercion alnifoliae. More specifically, on the understory layer of habitat type 9590* are found tall shrubs of the endemic species *Q. alnifolia* and, less frequently, individuals of the species *Arbutus dracaneae*. Relatively frequent is the *Cistus creticus* species and in more humid locations the species *Rubus sanctus* and *Pteridium aquilinum* are found. Other species found mostly on the floor of the forest are *Crepis fraasii*, Crucianella imbricata, *Cerastium brachypetalum* subsp. roeseri, *Stellaria ciliaca*, as well as the endemics *Anthemis plutonia*, *Arrhenatherum album* subsp. cypricum, *Cyclamen cyprium* and *Lactuca cyprica* (= *Cephalorrhynchus cypricus*) (Delipetrou and Christodoulou 2010; Andreou et al. 2017).

8.2.1. **The habitat 9590* functioning as a biodiversity hotspot**

The habitat type 9590* provides significant shelter for a great number of species of Cyprus flora and fauna. Thus, within the distribution boundaries of habitat type 9590* are found the two plant species *Arabis kennedyae* and *Ranunculus kykkoensis* (Fig. 8.4) which are included in Annex II of Habitats Directive (*A. kennedyae* is a priority species) (Eliades et al. 2018). In addition, the shape of habitat 9590* on the high elevation hills of Pafos Forest, has as a consequence the conservation and management of a significant number of animal and plant species. It is important to be noted that within habitat type 9590* are found 22 taxa of mammals out of which five (5) are endemic to Cyprus and 13 are bats (Gatzogiannis et al. 2010). Cyprus' mouflon (*Ovis gmelini ophion*) (Fig. 8.4) and bats are included in the Annexes of Habitats Directive while some of the mammals are included in the Annexes of Bern Convention and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Of great importance is also the avifauna of habitat type 9590*, since at the wider area of habitat 9590* (boundaries of ex-side of Natura 2000 “Koilada Kedron – Kamps”) occur 97 bird species, 22 out of which are listed in Annex I of Birds Directive (2009/147/EC) (Iezekiel and Christodoulou 2005). Other important organisms of the fauna of the island found in habitat 9590* are reptiles and amphibians. In Pafos forest there occur 10 taxa of lizards (five out of which are endemic subspecies) and 6 species of snakes (one endemic). The most important representatives of reptile fauna (Fig. 8.4) are *Acanthodactylus schreiberi schreiberi* and

![Endemic plant and fauna species observed in habitat type 9590*](image)

**Figure 8.4** Endemic plant and fauna species observed in habitat type 9590* (Plant: *Arabis kennedyae*, Photo: ©Marios Andreou; Fauna: *Otus cyprius* & *Ovis gmelini ophion*, Photos ©Harris Nikolaou)
Hierophis cypriensis which have their conservation status characterized as "Endangered" based on evaluation criteria of IUCN (Eliades et al. 2018). Regarding the amphibians, all three species of the island are found in Pafos forest. Out of these, the *Bufo viridis* is in Annex IV of Habitats Directive.

The invertebrates of the habitat are also of high importance. All six (6) Cyprus endemic species of butterflies, while 40% of endemic species and subspecies of Orthoptera occur within the habitat’s distribution range. Other important invertebrates are the endemic beetle *Propomacrus cypriacus* which is listed in Annexes II and IV of Habitats Directive 92/43/EEC and its conservation status is characterized as "Critically Endangered" according to the criteria of IUCN; and the moth *Euplagia (Callimorpha) quandripunctata* which is a priority species of Annex II of the same Directive (Eliades et al. 2018).

Systematic efforts for protection and conservation of the natural population of *C. brevifolia* in Cyprus began in 1879, when the population came under degradation and reached a critical point for its sustainability, due to environmental pressure and human activity (i.e. over-grazing and overlogging). In this time, the cedar forest was declared as a Forest Reserve (Wild 1879). In the beginning of the 20th century, the Cyprus Department of Forests established, in Tripylos mountain, a system of fire-paths, together with access roads and a fire-watch station, which could contribute significantly to the protection of cedar population against fire (Thirgood 1987; Christou 1997). In 1984, the Cyprus Council of Ministers, in an effort to preserve the cedar forest, declared an area of 823 ha at Tripylos mountain as a Natural Reserve, while as mentioned above, since 1999 the Cyprus Government proposed the Tripylos area for inclusion in the Natura 2000 network of natural protection areas of the European Union (Eliades et al. 2019). In addition, since the establishment of Cyprus Republic Government (in 1960), the Department of Forests launched numerous conservation measures which are being periodically applied up to today, focusing on expansion of the Cyprus cedar area, through *inter situs* and *ex situs* plantations. The Department of Forests established plantations (reforestation programme) at the periphery (boundaries) of the existing natural distribution of habitat 9590*; these plantations today cover an area of ~130 ha (*inter situs* conservation activity). Plantations of *C. brevifolia* were also conserved and in locations outside the natural distribution of habitat 9590* (i.e. in Madari forest and Amiantos asbestos mine).

### 8.3 Threats and pressures on habitat 9590*

The narrow distribution of habitat 9590* renders it extremely vulnerable to different threats (pressures and impacts; Fig. 8.5). The stochastic threats of forest fire and climate change are nowadays real and the most severe; thus they pose significant pressure on this small-area, unique habitat type, and could be potentially catastrophic. Further pressures and threats are increasingly high temperatures together with fluctuations in precipitation patterns occurring in the Mediterranean basin, which have caused the dieback of *C. brevifolia* stands (Christou et al. 2001), insect attacks due to several stress factors, damage from extreme weather conditions, severe competition by *Pinus brutia*, and the utmost danger of a potential forest fire, as the species has limited regeneration ability after fire.

The species’ and hence the habitat’s long-term preservation are realistically considered as very uncertain. Assessment of the data collected as well as the long-term experience of the partners of LIFE-KEDROS project showed the main pressures that habitat type 9590* faces or will potentially face, here presented in Table 8.1.
## Table 8.1: Identified threats and pressures for habitat 9590*

<table>
<thead>
<tr>
<th>No.</th>
<th>Threat/Pressure</th>
<th>Pressure (P)/Threat (T) and Assessment*</th>
<th>SDP&lt;sup&gt;a&lt;/sup&gt;</th>
<th>PAF&lt;sup&gt;b&lt;/sup&gt;</th>
<th>Manag. Plan Pa-fos&lt;sup&gt;c&lt;/sup&gt;</th>
<th>Manag. Plan SPA&lt;sup&gt;d&lt;/sup&gt;</th>
<th>Annex D&lt;sup&gt;e&lt;/sup&gt;</th>
<th>LIFE-KEDROS&lt;sup&gt;f&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Forest road network (non-paved forest roads)</td>
<td>P, T (L)</td>
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<td>✓</td>
<td>✓</td>
<td></td>
<td></td>
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<tr>
<td>2</td>
<td>Main road network (paved/tarred roads)</td>
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<td>3</td>
<td>Forest fire</td>
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<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
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<td>Establishment of electricity and phone lines</td>
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<td>5</td>
<td>Use of motor vehicles</td>
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<td>6</td>
<td>Human induced changes in hydraulic conditions (in drain basin)</td>
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<td>Competitive species</td>
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<td>8</td>
<td>Invasive non-native species</td>
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<td>9</td>
<td>Harmful insects</td>
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<td>10</td>
<td>Other harmful animals e.g. rat</td>
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<td>✓</td>
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<tr>
<td>11</td>
<td>Extreme climate phenomena (e.g. rise of temperature and extremes, long droughts)</td>
<td>T (L)</td>
<td></td>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
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<tr>
<td>12</td>
<td>Water flow changes</td>
<td>T (L)</td>
<td></td>
<td></td>
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<td>13</td>
<td>Soil erosion</td>
<td>P (M)</td>
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<td>✓</td>
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<td>✓</td>
</tr>
<tr>
<td>No.</td>
<td>Threat/Pressure</td>
<td>Pressure (P)/Threat (T) and Assessment*</td>
<td>SDF</td>
<td>PAF</td>
<td>Manag. Plan Pafos</td>
<td>Manag. Plan SPA</td>
<td>Annex D</td>
<td>LIFE-KEDROS</td>
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<td>Habitat fragmentation</td>
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<td>Illegal logging</td>
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</tbody>
</table>

*In the case of SDF and Annex D, there is an assessment of the significance of each pressure or threat and these are named as of low (L), medium (M) and high (H) significance.


b Prioritized Action Framework for Natura 2000 (PAF), which the Republic of Cyprus reported to EU for the period 2014 – 2020. (PAF)


f Field visits at habitat type 9590* distribution area, during the project (LIFE-KEDROS) by Project Management Team and project’s Scientific Committee. (LIFE-KEDROS)

Figure 8.5: Threats and pressures that negatively affect habitat 9590 *Cedrus brevifolia forests
8.4 References


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part b

chapter 8


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Wild A.E. (1879) Report on the forest in the south and west of the island of the Cyprus (Cyprus No 10). Harrison and Sons, London.
9.1. Setting the favourable reference values

The Habitats Directive and Birds Directive ensure the conservation of important habitat types, as well as a wide range of rare, threatened or endemic plant and animal species in Europe. The Habitats Directive considers explicit favourable reference values while the Birds Directive requires to maintain bird populations at a level which corresponds to their ecological, scientific and cultural requirements (Bijlsma et al. 2018).

In order to assess the conservation status (a key concept in European nature conservation laws and policy) for habitat types under the Habitats Directive, it is necessary to determine Favourable Reference Values (FRVs) for the range of habitat types (Favourable Reference Range - FRR) and for area of habitat types (Favourable Reference Area - FRA). FRVs are key reference levels to define when Favourable Conservation Status (FCS) is being achieved for habitat types (Bijlsma et al. 2018).

Specifically:

- FRVs derived from definitions in the Directive, particularly the definition of Favourable Conservation Status (FCS) that relates to the ‘long-term natural distribution, structure and functions as well as the long-term survival of its typical species’ for habitat types in their natural range (European Environment Agency 2017).

- FRR is defined as: “favourable reference value of range must be at least the range (in size and configuration) when the Habitats Directive came into force; if the range was insufficient to support a favourable status, the reference for favourable range should take account of that and should be larger (in such a case information on historic distribution may be found useful when defining the favourable reference range); ‘best expert judgement’ may be used to define it in absence of other data” (Chobot 2010). By definition, Range and FRR are the same if the Range is sufficient to support the population in favourable status. Range itself is rather theoretical concept, derived from the distribution (population) map. Range covers actual distribution and suitable and/or potential localities within the area of included gaps (Chobot 2010).

- FRA is defined as: the “total surface area in a given biogeographical region considered the minimum necessary to ensure the long-term viability of the habitat type; this should include necessary areas for restoration or development for those habitat types for which the present coverage is not sufficient to ensure long-term viability; favourable reference value (of area) must be at least the surface area when the Habitats Directive came into force; information on historic distribution may be found useful when defining the favourable reference area; ‘best expert judgement’ may be used to define it in absence of other data” (Chobot 2010).

Members states have the obligation to maintain habitat types and their typical species in favourable conservation status on a long-term basis, meeting the objectives of the Article 17 of the Habitats Directive. Despite the fact that FRVs are essential elements to determine the distance to
FCS, reporting has shown that they are still poorly developed and often inconsistently applied across Member States (Bijlsma et al. 2018).

Setting FRVs is a stepwise approach, starting with the collection of all relevant information about a habitat type in order to understand its ecological and historical context (Bijlsma et al. 2018). The following data is of utmost importance when estimating FRVs for habitat types (European Environment Agency 2017):

- Current situation and assessment of deficiencies i.e. any pressures/problems
- Trends (short-term, long-term, historic i.e. well before the directive came into force)
- Natural ecological and geographical variation (including genetic variation, inter- and intra-species interactions, variation in conditions in which habitats and species occur, variation of ecosystems)
- Ecological potential (potential extent of range taking into account physical and ecological conditions, contemporary potential natural vegetation)
- Natural range, historic distribution and abundances and causes of change, including trends
- Connectivity and fragmentation
- Dynamics of the habitat type
- Requirements of its typical species.

There are two basic approaches for setting FRVs: reference-based approach and model-based approach (Bijlsma et al. 2018). The reference-based approach considers the historical distribution/area of a habitat type in a period when the habitat type was supposed to be in a (stable) favourable condition. Model-based approaches use habitat type-specific features such as habitat suitability or required area for proper functioning. Clearly, semi-natural habitats (meso-habitats) require reference-based methods whereas habitats with extensive natural (macro-habitats) can be assessed using a minimum-area method (model-based method).

Table 9.1 below helps to assess conservation status of habitat types, based on range, area, structure and functions and future prospects of a given habitat type.

**Table 9.1:** General evaluation matrix for habitat types (from the Report format for the period 2013-2018: [http://cdr.eionet.europa.eu/help/habitats_art17](http://cdr.eionet.europa.eu/help/habitats_art17)).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Conservation Status</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Range</strong> (within the biogeographical region concerned)</td>
<td>Favourable (‘green’)</td>
</tr>
<tr>
<td>Stable (loss and expansion in balance) or increasing AND not smaller than the ‘favourable reference range’</td>
<td>Large decrease: Equivalent to a loss of more than 1% per year within period specified by MS OR More than 10% below ‘favourable reference range’</td>
</tr>
<tr>
<td>Parameter</td>
<td>Conservation Status</td>
</tr>
<tr>
<td>-----------------------------------------------</td>
<td>--------------------------------------</td>
</tr>
<tr>
<td><strong>Area covered by habitat type within range</strong></td>
<td><strong>Favourable</strong> ('green') <strong>Unfavourable – Inadequate</strong> ('amber') <strong>Unfavourable - Bad</strong> ('red') **Unknown (insufficient information to make an assessment)'</td>
</tr>
<tr>
<td>Stable (loss and expansion in balance) or increasing AND not smaller than the favourable reference area AND without significant changes in distribution pattern within range (if data available)</td>
<td>Large decrease in surface area: Equivalent to a loss of more than 1% per year (indicative value MS may deviate from if duly justified) within period specified by MS OR With major losses in distribution pattern within range OR More than 10% below favourable reference area</td>
</tr>
<tr>
<td>Any other combination</td>
<td>No or insufficient reliable information available</td>
</tr>
<tr>
<td><strong>Specific structure and functions (including typical Species)</strong></td>
<td>Structures and functions (including typical species) in good condition and no significant deteriorations / pressures</td>
</tr>
<tr>
<td>Any other combination</td>
<td>No or insufficient reliable information available</td>
</tr>
<tr>
<td><strong>Future prospects (as regards range, area covered and specific structures and functions)</strong></td>
<td>The habitats prospects for its future are excellent / good, no significant impact from threats expected; long-term viability assured</td>
</tr>
<tr>
<td>Any other combination</td>
<td>No or insufficient reliable information available</td>
</tr>
<tr>
<td><strong>Overall assessment of CS</strong></td>
<td>All ‘green’ OR three ‘green’ and one ‘unknown’</td>
</tr>
</tbody>
</table>
Step 1.1 – Ecology of the habitat 9590*

The habitat type 9590* Cedrus brevifolia forests (Cedrosetum brevifoliae) occurs only in Cyprus, in a small area of 263.4 ha on the top hills of Pafos Forest. The targeted habitat type is fragmented since it is found at five distinct areas. The largest site is located at Tripylos mountain (209 ha), while smaller sites are found on surrounding areas. This fragmentation is supported by the observed significant isolation-by-distance among the sites, probably due to insufficient gene flow (Eliades et al. 2011). Pure stands with C. brevifolia, forming habitat type 9590*, cover 103 ha. Mixed formations (i.e. mixed 9590* habitat type with 9540 Mediterranean pine forests with endemic Mesogean pines (keystone species: Pinus brutia) and/or habitat 9390 *Scrub and low forest vegetation with Quercus alnifolia) cover 160.4 ha (Eliades et al. 2019).

The habitat type 9590* occurs on rocky mountain slopes made out of diabase (igneous rock derived from erosion of Troodos ophiolite) (Eliades et al. 2018), where relatively shallow and eroded rocks of great and steep slopes are found, with acidic pH (5-6.75) (Eliades 2015). The climate where the habitat type 9590* occurs is characterized by Mediterranean climate with lower average annual temperatures than the rest of the island (Tsintidis et al. 2007).

Within the framework of LIFE-KEDROS project, 33 random sample stratified plots were installed for vegetation analysis, in pure stands of C. brevifolia, in mixed stands of C. brevifolia and Q. alnifolia, in mixed stands of C. brevifolia and P. brutia and in reforestation areas of C. brevifolia. The study (Andreou et al. 2017) demonstrated the significant differences that occur between the different communities of the targeted habitat type (e.g. pure stands, mixed stands). Cluster analysis of 33 plots data revealed the existence of four clusters:

i. Cluster 1 which represents mostly pure and undisturbed stands of C. brevifolia (9590*),

ii. Cluster 2 which represents mixed stands of C. brevifolia and Q. alnifolia (9590* + 9390*),

iii. Cluster 3 which represents an intermediate stage both in terms of succession and environmental conditions between dry and disturbed forests and moister and undisturbed ones.

iv. Cluster 4 represents mainly mixed stands of P. brutia and C. brevifolia (9540+ 9590*), which are dry and/or disturbed.

Typical species of the targeted habitat type (Cluster 1), which are found on the understory layer of C. brevifolia stands (keystone species), are tall shrubs of the endemic species Q. alnifolia and, less frequently, A. adrachne. Relatively frequent is the C. creticus species and in more humid locations the species R. sanctus and P. aquilinum are found. Other species found are Crepis fraasii, Crucianella imbricata, Cerastium brachypetalum subsp. roeseri, Lecokia cretica, Stellaria cilicica, as well as the endemics Anthemis plutonia, Arrhenatherum album subsp. cyricum, Cyclamen cyprium and Lactuca cyprica (Delipetrou and Christodoulou 2016; Andreou et al. 2017; European Environment Agency 2019).

In Cluster 2, C. brevifolia cover is generally almost 50% (in some plots the species has 100% cover) and Q. alnifolia is relatively abundant in the canopy (50-75% per plot). In Cluster 4, C. brevifolia cover is in most of the plots between 12.5 – 50% and the cover of P. brutia is less than 12.5%. In addition, the most common and abundant species in this Cluster are Asphodelus ramosus, Q. alnifolia (shrub layer), C. brevifolia (shrub layer), Cistus creticus, Crepis fraasii, Aira elegans and Crucianella imbricata. It is important to note that regeneration of P. brutia is limited in contrast to the regeneration of C. brevifolia, which is much higher (around 5% in some plots). This is indicative that the environmental conditions are more favorable for C. brevifolia than for P. brutia. In Cluster 3, the cover of tree layer of C. brevifolia in most of the plots is around 75%. The most common and abundant species are C. creticus, Q. alnifolia (shrub layer), C. fraasii, C. brachypetalum subsp. roeseri and Valantia hispida.
**Step 1.2 – Spatial scale of functioning**

*Cedrus brevifolia* forests are found in ecosystems with broad abiotic range and natural dynamics, comprising quite high diversity of different vegetation types (mixed habitat types or neighboring other pure habitat types) and successional stages. The floristic composition (including typical species) and structure of 9590* habitat type is based on the ability of *C. brevifolia* individuals to form pure stands, or to coexist with other habitat types (i.e. 9540 and/or 9390*). This heterogeneity and dynamics are important for characteristic species and can only occur and develop in more or less continuous areas of forest habitat from 10s to 100s of ha.

**Conclusions**

The habitat type functions at the macro-habitat level (category 1a).

**Step 1.3 – Historical perspective**

No quantitative data on historical range and area are known. Nevertheless, cedar trees seem to be protected from the ancient times. Theophrastus mentioned in his book ‘Historia Plantarum’ (written some time between c. 350 BC and c. 287 BC) that cedar trees were not used for timber in Cyprus since they were protected by the Kings and they were very difficult to be transferred (due to their large size).

At late 1800s, the Colonial Government of Great Britain found severe habitat degradation. During this period, the 9590*habitat was characterized by overripe trees and low regeneration. The main causes of habitat degradation were illegal logging, uncontrolled fires and forest overgrazing (Thirgood 1987, Christou 1991). However, no map and quantitative data is available. The Government created the Department of Forests and since 1879 the natural habitat type of *C. brevifolia* was designated as a Forest Reserve, where logging and grazing were prohibited (in situ measures).

Since the establishment of the Republic of Cyprus in 1960, the first ex situ measures of the targeted habitat started, i.e. collection of Cedar seeds for the production of seedlings and their subsequent planting or for their dispersal in specific locations (within natural range or in other locations). In 1984, the Council of Ministers, at the suggestion of the Forest Department, declared an area of 823 ha within the boundaries of Tripylos site and Mavri Gremos site, as a ‘Natural Reserve,’ based on the provisions of Forest Law.

After Cyprus' accession to the EU, *C. brevifolia* forests were proposed to be included in Annex I of Habitats Directive, as a priority habitat type (9590* habitat type). The whole range of the targeted habitat was included in the Natura 2000 network in 2008. Since then, several management and conservation measures carried out towards the improvement of conservation status of 9590* habitat type (see Andreou et. al 2017). The culmination of efforts to conserve this habitat type was the designation of an area within Tripylos site as a Natural Micro-Reserve of *C. brevifolia* forests, based on the provisions of Forest Law.

**Step 1.4 – Analysis of distribution and trends**

The current range of 9590* habitat type is 28 km² (see Fig. 9.1). The current and future trend is supposed to be stable. Historical trends remain unknown. Since none of the characteristic species or structures are in bad condition, there is no indication that the habitat is not viable in the long term regarding its function and structure.

The most important threats identified for the targeted habitat is the low regeneration of *C. brevifolia* only in forest openings (seedlings and/or plantlets die due to high summer temperatures; climate change effect) and the possibility of a future forest fire. The species has no post-fire mechanism for regeneration and the neighbouring *P. brutia* forest is expected to colonize the available space.
**Conclusions**

Based on experts’ opinion, the range is likely to be stable over several decades, if the ongoing measures and legislation continue in the future. The successful establishment of *C. brevifolia* in other locations (outside natural range) and the reforestation efforts that took place in the past, secure the sound management and restoration of a site in the unlikely event of a forest fire.

**Figure 9.1:** Distribution and range of 9590* habitat type: Cedrus brevifolia forests (Cedrosetum brevifolii).
Step 2.1 – FRA assessment

The current value (CV) is considered as FRA, because of the absence of indicators of non-viable functioning or structure of the targeted habitat type. Despite the fragmented character of this habitat type, no change has been recorded (through monitoring) the last years to the habitat’s composition and structure. For this, the only available quantitative estimate is chosen, based on detailed mapping of 9590* habitat type.

Conclusions
FRA can be considered equal to 2.63 km².

Step 2.2 – FRR assessment

The current range is large enough to contain the current FRA and therefore is considered as the FRR. Enlargement of the range is not required for a viable structure and function.

Conclusions
FRR= CV (28 Km²).

9.2. References


10.1. Introduction

The forests in Cyprus came under pressure of climate change since the beginning of 21st century, resulting in the die-back of trees, excessive loss of foliage, insect attacks, reduction of vitality and productivity of forests and, in certain cases, loss of ecological stability due to wild fires. Thus, the Department of Forests, as well as the academic community (scientists) of the island of Cyprus adopted the vision of maintaining the forests in the best possible condition through sustainable management and the improvement of the conservation status of the environment. This vision, aimed towards achieving the greatest possible environmental, social and financial benefits that will be balanced, will be based on the principle of sustainability and will fulfil the expectations of society.

Based on this general vision, as well as on the global environmental change, several projects focusing on the conservation and monitoring of nature have been running or are under implementation in Cyprus. The Department of Forests, the wider scientific community and the civil society are recognizing the great ecological importance of biodiversity on the island, and especially the need for the conservation of the species with unique and narrow (local) occupancy. These biodiversity elements have witnessed the island’s history throughout the ages and consist the natural inheritance of the island between succeeding generations. These elements of nature (fauna and flora) should be preserved based on the existing knowledge from international literature and the practical experience from the field in Mediterranean ecosystems.

The forest of *Cedrus brevifolia* in Cyprus is one such element with important ecological and environmental impact on the forestry of the island (see Chapter 8). For the sustainable management and conservation of the narrow habitat of 9590 *Cedrus brevifolia* forest (*Cedrosetum brevifoliae*) all guidelines mentioned in previous chapters of this book need to be adopted, in order to secure the protection and the sound management of this priority habitat type of Cyprus occurring in a Natura 2000 site. Based on the step-by-step diagram (Fig. 10.1) the LIFE-KEDROS project was developed.
The conservation effort through the project LIFE-KEDROS

The project entitled “Integrated conservation management of priority habitat type 9590* in the Natura 2000 site Koilada Kedron – Kampo” (LIFE15 NAT/CY/000850) and with the acronym LIFE-KEDROS, has been implemented during the period from September 2016 to January 2021. In order to specify the objectives of this project, the beneficiaries followed adopting the guides from previous chapters.

The project’s general aim was to ensure the medium and long-term preservation of the priority habitat type 9590* Cedrus brevifolia forests in good conservation status, at the only Natura 2000 site that the habitat is found. To address this aim, specific objectives (conservation targets) were achieved through the adoption of particular conservation actions, both within (in situ) and outside (ex situ) its natural range, namely:

i. Reducing fire danger and the possibility of habitat loss or even its complete destruction as a result of a single large forest fire incident.

ii. The enhancement of the habitat’s resilience and adaptation capacity to climate change, and to competition by other forest trees and shrubs.

iii. The restoration and expansion of the habitat within the project site and the enhancement of the natural regeneration capacity of Cedrus brevifolia stands.

iv. The improvement of other biotic and abiotic factors that are important for the health and vigorosity of Cedrus brevifolia stands/trees and the stability of local ecosystems.

v. The preservation of the genetic material for the core species of habitat 9590*, namely Cedrus brevifolia, through the implementation of ex situ conservation measures, including

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2 The initial duration of the project was 48 months with the expected end date to be on 31 August 2020. However, due to the Covid-19 pandemic, the project was extended for five months.
storage of seeds in a seed bank and creating a new population of cedar outside its natural environment.

vi. The implementation of public awareness activities and the dissemination of project results to local and overseas managers and scientists.

The project LIFE-KEDROS was implemented within the framework of the LIFE program of the European Union (EU). The total project budget was € 1,413,304, of which € 968,330 (68.6% of its total eligible budget) were co-funded by EU. The project was carried out by a well-organised consortium, which included: (i) Department of Forests (Ministry of Agriculture, Rural Development and Environment), which is the competent authority of Cyprus on forests conservation, (ii) Frederick University, through the Nature Conservation Unit (NCU); NCU focuses on biodiversity conservation, natural resources management and conservation, and environmental education and awareness in Cyprus; it is the first unit in the Cypriot academic system to have ever dealt with issues related to nature conservation. and (iii) Cyprus Forest Association, one of the largest non-governmental non-profit organizations in Cyprus, which has as its main objective to contribute to nature conservation on the island in general, and of forest resources in particular.

10.2.1. Project structure and project innovation

The successful implementation of the project’s objectives was achieved through designing particular actions, where in each action specific activities (deliverables & milestones) were set. The actions were closely related to each other (Fig. 10.2), all aimed toward the common goal of meeting conservation targets, both qualitatively as well as quantitatively. The project activities that aim towards addressing the threats on the habitat are inspired by nature and correspond to the ecological requirements of habitat 9590*.

![Figure 10.2: Flowchart of LIFE-KEDROS project](image-url)
10.3. The LIFE-KEDROS implementation

10.3.1. Preparatory actions, elaboration of management plans and/or of action plans

This preparatory part of the project was divided into five (5) preparatory actions, which were implemented during the 1st – 18th month of the project’s duration. These actions focused on collecting all necessary data, information and knowledge, resulting in the preparation of relevant reports, which defined the guidelines and timelines for the implementation of the concrete conservation actions. The preparatory actions were:

i. Description of composition and structure of habitat 9590*
ii. Elaboration of a fire protection plan for the habitat 9590*
iii. Assessment of health and vitality of habitat 9590*
iv. Assessment of the impact of specific concrete conservation activities
v. Preparation of Action Plan for the implementation of conservation actions

Implementation of these actions led to addressing specific gaps on the ecology and silviculture of the keystone species, *C. brevifolia*, and at the same time led to elaborating specific plans, such as:

*Marking guidelines for silvicultural treatments of habitat 9590*

This guideline presents the principles for silvicultural treatments of natural and artificial stands of *Cedrus brevifolia*. The keystone species of habitat 9590*, *C. brevifolia*, is characterised by competitive advantages against the main competitive tree species of *Pinus brutia*. These competitive advantages are that *C. brevifolia* has higher resistance to shade conditions and an increased ability to cope with more unfavourable soil conditions (Milios et al. 2021). In fact, analysis of field data supports that *C. brevifolia* can be considered as a semi-shade tolerant and site-insensitive species (Milios et al. 2021) which can survive and grow at different rates in various site qualities. The silvicultural structure of *C. brevifolia* artificial stands is approximately even-aged, with dominant and codominant trees constituting the most important structural features of these stands (Petrou et al. 2018). However, *P. brutia* appears with different density in each site type of *C. brevifolia* artificial stands (Petrou et al. 2018).

*Fire Protection Plan of habitat 9590*

The elaboration of a fire protection plan consists of three main parts (Christodoulou et al. 2017). These parts can be summarised as follows: (i) description and assessment of the most important forest fire causes, based on an analysis of forest fire statistical data, which were then used for the establishment of five forest fire scenarios in areas located around habitat type 9590*, (ii) analysis of the existing situation and evaluation of the forest fire infrastructure and other forest fire protection measures implemented and (iii) the elaboration of recommendations for infrastructure and for other actions which need to be applied towards the integrated protection of the habitat against its most vital threat, forest fire. This plan provided a holistic approach for forest fire fighting in the wider cedar forest area in Cyprus (see Chapter 101).

*Action Plan for the sustainable management and conservation of habitat type 9590*

The Action Plan provides general information about habitat 9590* (Eliades et al. 2018), the biology and ecology of its core species *C. brevifolia* and the actions taken in Cyprus for the sustainable management of this habitat, as well as pressures and threats that it faces. This information is derived both from both existing knowledge, as well as from outcomes and results of the preparatory actions of the LIFE-KEDROS project. In addition, the Action Plan describes the activities planned to be undertaken for the management and conservation of habitat 9590*. The activities proposed are divided into seven measures as followed: (i) silvicultural interventions, (ii) restoration and expansion of habitat 9590*, (iii) protection of habitat 9590* against forest fires, (iv) mea-
sures for improving the resilience of habitat 9590*, (v) ex situ conservation of habitat 9590*, (vi) conservation and enhancement of biodiversity in the Natura 2000 site "Koilada Kedron – Kampos", (vii) communication, information and raising public awareness for habitat 9590*. The Action Plan acts as a guiding tool for the implementation of specific concrete conservation activities, since it provides guidelines on the methodology that should be followed, the selected locations where the activities should take place and the timeframe for their implementation. Further, the Action Plan sets the future objectives of each activity together with an estimated budget for each of these activities.

- **Key points**: A critical point before any conservation activities for narrow habitat types (and/or narrow endemic species) is to collect and summarise all the relevant existing knowledge and at the same time to identify and evaluate the threats and pressures that nowadays negatively affect the targeted habitat or species. Thus, the conservation experts, by identify the gaps on this knowledge, can aim towards enhancing this knowledge through further research. A specific Management Plan and Action Plan should be elaborated for the targeted habitat or species, presenting the conservation targets and conservation measures. Both Plans could be revised according to the new threats and pressures affecting the habitat, which could lead to setting new conservation targets.

10.3.2. **Concrete conservation actions**

The key for implementing any project with nature conservation objectives is designing and completing concrete conservation actions. These conservation actions must aim towards improving the conservation status and ecological condition of the targeted habitat. Their impact must be measurable and must be monitored and evaluated during the project (see #3.3). In all cases, these actions should be designed for addressing specific conservation purposes for a specific habitat, and, hence, general guidelines could be set for transferability and replicability in other habitat types (or keystone species).

In LIFE-KEDROS project, the concrete conservation actions were the main tool for the direct improvement and/or reversal of the decline of the ecological condition and conservation status of the targeted habitat 9590*. They consisted of sound forestry management techniques and infrastructure investments, mainly within (in situ) but also outside (ex situ) of the habitat 9590*. The conservation actions were implemented from the beginning of the project until its end. The project involved five concrete conservation actions, which were divided into:

*Silvicultural interventions in Cedrus brevifolia stands*

This Action aimed to favour the keystone species of the targeted habitat 9590* (*C. brevifolia*) over its competitors, through: (i) thinning of dense *C. brevifolia* young stands, pure or mixed with *Pinus brutia* (Fig. 10.3), (ii) removal of *P. brutia* competing with mature cedar or suppressing cedar regeneration and (iii) pruning *Quercus alnifolia* suppressing young *C. brevifolia* trees (Fig. 10.3). The silvicultural interventions focused on reducing competition and providing adequate growing space and increased resources (soil nutrients and water) to selected *C. brevifolia* trees. In LIFE-KEDROS project, the silvicultural interventions were carried out within ~200 ha of habitat 9590* (natural and artificial stands) and benefitted more than 10,000 *C. brevifolia* individuals (of which 99% were young trees).

- **Key points**: For narrow endemic habitats silvicultural interventions need to identify and characterise all structure types, by combining the species composition of the formation and the productivity of the site stands. For each structure type, specific silvicultural goals and treatments should be proposed, while the silvicultural manipulations – treatments should include redistributing of growing space, in order to create certain conditions, together with favouring specific individuals or species. Thus, silvicultural manipulations...
- treatments in narrow endemic habitats should focus on multi-functionality and on meeting the demands of the habitat, and not of wood production, by ensuring the intra-environmental ecological condition and without disturbing the ecological procedures and ecological niches under the canopy of habitat for other important species (fauna and flora). The achievement of even a relatively simple goal, such as reducing the intensity of competition faced by individual trees of the keystone species through the removal of neighbouring competitors belonging to other species, in most cases cannot be accomplished by a single intervention.

Figure 10.3a: Thinning/removal of *P. brutia* competing with mature cedars or suppressing cedar regeneration (Photos: © Petros Petrou)

Figure 10.3b: Pruning of *Q. alnifolia* suppressing young *C. brevifolia* trees (Photos: © Petros Petrou)
Restoration and expansion of the habitat type 9590*

This action aimed towards restoration of low-density and degraded natural regeneration groups of *C. brevifolia* stands by increasing their density and expanding the targeted habitat’s area by establishing new regeneration groups in existing gaps that are bare or covered by phrygana (mostly *Cistus* spp.). In addition, a long-term goal for this action is the improvement of connectivity within patches of geographically and genetically isolated stands of *C. brevifolia*. For the successful restoration and expansion of habitat 9590*, the LIFE-KEDROS project implemented specific measures such as cedar seed dispersal (Fig. 10.4) and/or planting of cedar plantlets (Fig. 10.4) into seven different spots. This conservation activity resulted in both: (i) expansion of habitat type 9590* at an area of 9.37 ha, by planting 2403 plantlets in several spots (in a total area of 3.29 ha) and by dispersal of 5.85 kg of cedar seeds (87,750 seeds), and (ii) restoration of habitat 9590* within its natural distribution covering an area of 12.31 ha by planting 1355 plantlets and by dispersal of 10.10 kg of cedar seeds (151,500 seeds).

- **Key points**: The success of any effort of restoration and/or expansion of a targeted habitat, presupposes that selected spots (for planting or sowing) fully correspond to the ecological niche of the keystone species of the targeted habitat. The observation of previous natural regeneration of the keystone species is a strong evidence that the spot could be used for planting or sowing of seeds. The strategy of using plantlets seems to be more successful than that of seed dispersal (sowing), despite that the cost of maintaining plantlets for a period of three years is higher than the sowing. The plantlets’ maintenance includes digging and watering (irrigation) during the summer season (dry period of the year – irrigation every 15-days). For the irrigation of plantlets, an irrigation system (including water tanks and an irrigation network of water pipes) could be installed. In spots where irrigation is not possible, a new planting approach could be used (such as water-boxes), without, however, this alternative approach having such/particularly successful outcomes in the case of low quality (extreme) stands.

![Figure 10.4: Cedrus brevifolia seeds dispersal (left) and planting of cedar plantlets (right) at seven locations for the purposes of LIFE-KEDROS (Photos: © Nicolas-George Eliades & George A. Constantinou)](image-url)
In all cases, particular effort should be given during the sampling of seeds for the production of plantlets, as well as during their production in the nursery. For these steps, specific international protocols and methodologies could be adopted, such as a seeds sampling methodology (i.e. random sampling from the whole distribution area of the targeted habitat) and production of seedlings and plantlets in a nursery. However, the use of existing scientific information for the targeted habitat, and particularly the knowledge regarding the population genetics (variation and structure) of the keystone species of the habitat type, ensure the long-term effectiveness of conservation activities. Thus, sampled seeds should be healthy and correspond to the high genetic variability, while plantlets should be produced in a nursery with similar ecological (climatic) conditions as those in the natural distribution area of the habitat. The plantlets should be cultivated in a nursery for at least two years before being established in the field.

Protection of the habitat type 9590* against forest fires

The goal of this action was to set a series of measures to prevent or minimize the risk of a forest fire event. A random stochastic catastrophic event such as fire could have an irreversibly destructive impact on the entire narrow-distribution habitat, but also on other protected elements of the site. In the case of habitat 9590*, a critical point was to design and apply particular fire protection measures (Fig. 10.5) focusing on preventing or minimizing the risk of forest fires within or near the project site and on enabling faster and more effective intervention of the fire fighting forces. Prevention reduces the risk of fires while it is a viable and cost-effective way to avoid irreversible losses directly affecting the habitat and numerous other national assets.

Thus, LIFE-KEDROS implemented the following measures in order to protect habitat 9590* against forest fire:

i. Enhancement of Department of Forests patrolling, from June to September of each year.
ii. Installation of six (6) fire risk warning signs.
iii. Creation of a new fire break (300 m).
iv. Closure of selected forest roads of total length of ~32 Km by using heavy metallic bars and cultivating 13 km of old forest roads for enhancing natural vegetation.
v. Implementation of silvicultural interventions along the forest road network (1 Km), in order to facilitate prompt access of fire vehicles.

vi. Removal of dry herbaceous vegetation along roads of a total length of 8 Km.

vii. Construction of two (2) 90-ton water tanks for supplying fire vehicles in case of fire.

viii. Control of flammable dry ground herbaceous vegetation through grazing by animals.

This is the first time that a project sought to take advantage of grazing by wild animals for the control of dry flammable vegetation, within the boundaries of habitat 9590*. For this purpose, the project implemented the following:

i. Construction of six (6) water guzzlers and three (3) small technical pools within the habitat’s area, in order to attract wild fauna species.

**Key points:** Before the implementation of any fire protection measures, a Fire protection plan should be elaborated, including evaluation of possible fire causes in the study area and of the various likely scenarios of fire expansion in case of a large fire and its effects. This methodology revealed the needs and deficiencies, in order to propose specific prevention, pre-suppression and suppression measures. These types of measures should be applied as complementary to each other (according to the available budget). In addition, more ecological friendly approaches, such as grazing by domestic and wild animals could be used, in order to reduce the dry ground annual and perennial herbaceous vegetation. Finally, based on the scenarios and the simulation test, the fire-fighting task force and the relevant infrastructure will be better employed.

**Measures for improving the resilience of habitat type 9590**

In the current project the improvement of the habitat’s resilience, especially in the context of climate change, was implemented by measures for controlling the biotic threats described (insects and rats) and by improving the abiotic environment with erosion control measures. The control of both biotic and abiotic pressure in habitat 9590* was done by adopting ecologically friendly measures, without extending in the whole distribution area of the habitat.

The soil erosion control measures include establishing of: (i) dry stone terraces (176 m in total), (ii) gabions (156 m in total) within the flow path of the main stream that flows through habitat 9590* and (iii) log-erosion barriers (100 m in total). All these anti-erosion measures (Fig. 10.6) showed effective results from the first year that they were installed, since they controlled and minimized soil erosion on the locations where they were established. Also, these measures indirectly benefit various species regeneration dynamics, including *C. brevifolia*, as they improve microclimatic conditions, while at the same time they create specific micro-habitat (micro-environment) conditions for fauna species, supporting the conservation of biodiversity in this area.

![Figure 10.6: Anti-erosion measures applied in habitat 9590* (dry stone terraces, gabions, log-erosion barriers) (Photos: © Nicolas-George Eliades)](image-url)
As regards to controlling harmful species, this focused on increasing the productive potential of *C. brevifolia*, and therefore the health and viability of habitat 9590*. For this, an eco-friendly approach was used, by supporting the stability of a specific food web (at its highest level) within the habitat boundaries, and by trying to control bark beetles using the mass trapping method, which involves the use of pheromone/kairomone-baited black window-slot traps (see Chapter 10II). For the purposes of managing the population of insects (e.g. *Megastigmus schimitscheki*) and rats within the habitat, artificial nests were established. The artificial nests encouraged the presence of owls (10 artificial nests for owls) and bats (20 bat boxes), supporting the food web in the study area by specific predators for insects and rats (based on this food web) (Fig. 10.7). The artificial nests were inhabited during the project’s implementation as follows: (i) five nests were inhabited by Cyprus scops owl (*Otus cyprius*), (ii) one nest by Barn owl (*Titus alba*), while during the project’s monitoring ~9500 rats were predated and (iii) four bat boxes were inhabited by *Pipistrellus pipistrellus* and *P. kuhlii* (bats), which led to the predation of 32 million of different insects.

**Figure 10.7:** Artificial nests encouraged the presence of owls (Photos: © Erodotos Kakouris)

- **Key points:** Any effort on improving habitat resilience, in the context of climate change, by applying measures to control/ manage biotic and abiotic threats and pressures, should be inspired by nature and correspond to the ecological requirements of the targeted habitat. Based on this assumption the use of artificial stocks for infrastructure in the environment should be avoided (i.e. concrete, sheet metal etc.), while any construction should be carried out by handwork and minimize the use of engines (i.e. trucks, bulldozers etc.). In addition, any activity for the management of the population size for harmful species, should be designed as to follow eco-friendly approaches, supporting the identifying and use of natural predators within existent food web. A mass-trapping approach for bark beetles (i.e. outbreak beetles, wood beetles) could be adopted during a season with high temperature and low precipitation and the year after this season, by using a network of Multi Wit bark beetle slit traps baited with both ethanol and the aggregation pheromone, in order to minimise the insects’ population diffusion.

**Ex situ conservation of the targeted habitat type**

The long-term survival of the targeted habitat 9590* outside its natural environment is a complementary conservation measure on the holistic sustainable management of habitat 9590*. Preserving plants in their native habitat (*in situ*) is clearly a first step, but *ex situ* preservation is also necessary to secure the plants against catastrophic events and human impact and to facilitate reintroduction in the future, when appropriate habitat and favourable conditions become available. In LIFE-KEDROS, two conservation measures, complementary to each other, were implemented by:

i. storage of seedlots (Fig. 10.8) in a seed bank (200 kg of seeds = 3 million of seeds); this makes available suitable and adequate propagation material (seeds) which will help re-
establish cedar in the site in case its stands are destroyed completely or to a large extent, by a future forest fire,

ii. establishing a new population of 6500 *C. brevifolia* plantlets (Fig. 10.9), in an area of 8.5 ha in Amiantos asbestos mine, at an altitude above 1350 m (within the Natura 2000 site of Troodos National Forest Park).

- **Key points**: *Ex situ* conservation is recognised by the Convention on Biological Diversity and by International Bodies such as IUCN, FAO and IPBGR, as one of the most important methods for conserving wild plants. Together with *in situ* preservation, *ex situ* preservation ensures resilience of the species against human impact and possible catastrophic events, while easing potential reintroduction of the species. A critical point for success on *ex situ* conservation efforts is the balance between cost and benefits, therefore the seedlots should represent all the geographical distribution of the species and be of the highest genetic diversity (based on existing specific knowledge or by adopting international guidelines). For the seed collections and the further processing and manipulation (cleaning, drying, storage and general management) of seedlots, international standards and recommendations should be adopted (i.e. according to the ENSCONET Seed Collecting Manual for Wild Species & ENSCONET Curation Protocols and Recommendations). Further, new plantations could be established
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at higher altitudes than the natural distribution of the targeted habitat and/or with similar ecological conditions. This strategy will enhance the possibility for the targeted habitat’s preservation in the case that re-establishment is not possible or that climate changes will be so severe that the species will not be able to persist in its current altitudinal zone.

10.3.3. Monitoring of the impact of the project actions

This group of actions aimed towards developing the methodology and the implementation of protocols for monitoring the impact of the project’s actions on socio-economic and ecological (ecosystems functions) aspects. These actions ran during the project’s period and were clustered into:

i. Monitoring and evaluation of the project’s performance, findings and outcomes.

ii. Assessment of socio-economic impact.

iii. Assessment of impacts on ecosystem functions.

- **Key points:** The monitoring of concrete conservation actions is important, since it allows identifying and managing any mistakes and lack of best practices for the sustainable management of the targeted habitat. In addition, the monitoring and evaluation of the project’s conservation actions ensure the success of the project’s objectives, as well as the ability to generate conservation measures that could be replicated and/or transferred onto effective sustainable management practices for this specific habitat and/or habitats with similar ecological conditions.

10.3.4. Actions on public awareness and dissemination of results

The implementation of raising awareness and disseminating activities for the promotion of the project’s objectives, is a complementary tool for the sustainable management of the targeted habitat type. These awareness and dissemination activities should generally contain an appropriate amount of communication and dissemination activities, typically including:

*Information and awareness raising activities*

The activities focus on promoting the project’s objectives and actions, to the general public and stakeholders, aimed at facilitating the implementation of the project, and particularly at increasing awareness on the targeted habitat type (forest). For the general public, the awareness should promote historical and ecological information about the targeted habitat, together with emphasising on the uniqueness of the habitat for nature and as an element of natural inheritance for the whole country. The threats and pressures that currently affect the habitat negatively should also be presented in an awareness campaign. Finally, for the stakeholders, such campaign could emphasise the need for identifying such unique habitat as a “Flagship forest” for specific sites (areas or regions), leading to direct and indirect socioeconomic benefits for local communities. This linkage between local communities and the project’s goals promotes a positive background for the long-term conservation of the targeted habitat. The information and awareness raising activities (Fig. 10.10) could include:

i. Promotion materials with the project’s logo and an attractive slogan.

ii. Leaflets or newsletters on an annual basis, providing information about the targeted habitat (forest), the project’s objectives and the project’s progress.

iii. Installation of notice boards in strategic places (i.e. rural areas, neighbouring to the forest’s communities) providing information about the habitat type. Particular emphasis should be given on ecological importance, the uniqueness of the habitat and the threats and pressures that affect this habitat nowadays.

iv. The creation of a social media account (which appears to be more user-friendly and more visitable than a website) in order to frequently inform the general public about the
The project's implementation and objectives, as well as the ecology and importance of the targeted habitat.

v. The creation of a project's film for promoting the project, the conservation targets and the conservation measures implemented. However, special emphasis should be given on the uniqueness of the habitat and of the general site (ecosystem).

vi. The publication of a press release and articles in newspapers and particularly in daily news portals in regular intervals (i.e. every six-months).

vii. The organizing of rural workshops at least in the beginning and before the end of the project, in order to inform the local communities and local stakeholders about the threats concerning the targeted habitat (keystone species / forest), the management measures and the conservation actions applied.

Technical dissemination activities

These activities aim to transfer the results, best practices, and scientific data to those stakeholders that could usefully benefit from the project's experience and implement themselves the actions demonstrated in the project, as well as to the scientific community. Technical dissemination should be more focused on the transferability of scientific knowledge (Fig. 10.11) received by the project's implementation. Such dissemination could include:

i. Participation and presentation of the project's scientific outcomes in international scientific conferences.

ii. Publication of a technical guide (short reports) during the project's implementation, in order to provide the conservation methodology and/or scientific knowledge on the targeted habitat.

iii. Organising of thematic workshops (or conferences) where scientists from relevant projects and policy makers can discuss, on a scientific basis, about the design and implement of conservation measures for ecosystems and targeted habitat types.

iv. Organising of events that enhance public participation in conservation initiatives and overcoming external, political, administrative, and management issues that may arise.

Figure 10.10: Information and awareness raising activities that were carried out through the LIFE-KEDROS project (Photos: © Nicolas-George Eliades & George A. Constantinou)
10.3.5. Project management actions

To ensure the project’s effectiveness and its successful implementation, both project management and project quality control actions should be included. Through these actions the proper implementation of the project can be ensured at a technical, scientific and financial level. Thus, a Project management action should be defined, for effective administrative and financial coordination of the project, timely implementation of the project, together with risk management related to the project’s objectives and implementation. For the better management of the project, a Project Management Team (PMT) should be set, comprising of nominated scientific experts on topics relevant to the project’s objectives and management. In addition, from the beginning of the project, numerous monitoring indicators should be specified, in order to monitor the progress of the project and to evaluate the timely and proper implementation of the project’s actions. The Project monitoring action is key in ensuring that the project is implemented according to its proposal and timeframe. Finally, a post-project plan should be elaborated and run after the project’s completion. This plan should include details regarding the conservation targets that will be carried out after the end of the project, indicating the timeline (with optimum duration being three to seven years after the end of the project). Thus, maintaining the investments made through the project’s actions must be ensured in the long-term, following the project’s completion.

10.4. Benefits of the project and its actions

In general, any conservation project should bear best-and demonstration characteristics, in order to ensure the successful implementation of its objectives. LIFE-KEDROS project is characterized as a best-practice and demonstration project owing to the fact that it interferes and implements, for the first time within a designated reserve such as the cedar valley, actions and integrated management measures (see #10.3).

As mentioned, the LIFE-KEDROS project aims to address the threats and pressures that negatively affect the habitat’s resilience, as an act being inspired by nature and corresponding to the ecological requirements of habitat 9590*. Thus, the project is to support the mid-term and long-term preservation of the priority habitat 9590* in good conservation status, at the only Natura 2000 site that the habitat is found. In addition to this, the Department of Forest proceeded with updating the Standard Data Form (SDF) for the Natura 2000 site in Pafos Forest in general, and, hence, updated the whole distribution area of habitat 9590*, which is now within a new Natura 2000 site (Dasos...
Pafou – CY2000016) also including the *Exo Milos* patch (Fig. 10.12). These conservation efforts support the biodiversity and the ecological succession of habitat 9590*. The conservation actions of the current project led to both mitigation and adaptation to climate change. Although the conservation measures were designed and applied for this specific habitat, the general principle behind them could be adopted for other narrow forest habitat types. The project’s implementation also focuses on public awareness, with the general public, tourist guides, scientific groups and policy implementers on Natura 2000 networks being the audience and stakeholders with the highest interest. In addition, the project promotes the natural inheritance and ecological importance of 9590* by producing specific dissemination outcomes. Thus, as mentioned in previous reporting of the project, there is indirect support of the agro-tourism and walk tours, together with the socio-economic sector of communities neighboring the targeted Natura 2000 site. The project’s activities also contribute to the implementation of national and European legislation (i.e. Aichi’s targets 5 and 7; EU 2020 Biodiversity Strategy - Targets 1, 3, 6; Prioritised Action Framework –PAF).

Under the framework of LIFE-KEDROS project, the Department of Forests, according to the provisions and procedures set in Article 15 of the Forest Law of 2012, nominated five *C. brevifolia* trees (Table 10.1; Fig. 10.13), within the distribution area of habitat 9590*, as “nature’s monuments”. The oldest cedar tree is approximately 350 years old and has a stem diameter of 3.70 m. These nature’s monuments (aged trees) have managed to survive over the centuries, their existence running alongside many important historical events. Taking this into account, together with the respect, which the past generations have shown to these trees, there arises the need to protect and preserve these trees as elements of natural and cultural heritage.

**Table 10.1:** Nature’s monuments: aged trees of *C. brevifolia* within the distribution area of habitat 9590*.

<table>
<thead>
<tr>
<th>No</th>
<th>Location</th>
<th>DBH (in m)</th>
<th>High (in m)</th>
<th>Age</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Limnitis Forest (Limnitis valley – Beat 1; Compartment 29)</td>
<td>3.40</td>
<td>30</td>
<td>290</td>
</tr>
<tr>
<td>2</td>
<td>Panagia Forest (Routhia valley – Beat 6; Compartment 159)</td>
<td>2.30</td>
<td>10</td>
<td>288</td>
</tr>
<tr>
<td>3</td>
<td>Panagia Forest (Routhia valley – Beat 6; Compartment 159)</td>
<td>2.10</td>
<td>10</td>
<td>240</td>
</tr>
<tr>
<td>4</td>
<td>Panagia Forest (Routhia valley – Beat 6; Compartment 159)</td>
<td>2.79</td>
<td>12</td>
<td>310</td>
</tr>
<tr>
<td>5</td>
<td>Panagia Forest (Routhia valley – Beat 6; Compartment 21)</td>
<td>3.70</td>
<td>10</td>
<td>350</td>
</tr>
</tbody>
</table>
The LIFE-KEDROS, focusing on the holistic conservation of habitat 9590*, proposed two spots that could be declared as "Nature Reserves" (Fig. 10.14). With the enactment of the Cyprus Forest Law in 2012, a Nature Reserve is defined as a natural area that is designated solely for scientific research and no human intervention or even access is allowed. The distribution area and size of habitat 9590*

Figure 10.13: Trees of C. brevifolia that was nominated as "nature's monument" (Photos: © Nicolas-George Eliades)

Figure 10.14: Map of the proposed Nature Reserves in habitat 9590*
could not allow zonation management that would adopt a protected area and surrounding areas. The proposed Nature Reserves for habitat 9590* are located in two different subpopulations (sites) of the distribution area of habitat 9590*. The first proposed Nature Reserve is located in the core site of C. brevifolia forest, in the toponym of Gremos tis Pellis. This Nature Reserve occupies an area of 40.55 ha, including a pure stand of habitat 9590* (21.50 ha), the habitat 9540* Mediterranean pine forests with endemic Mesogean pines (13.25 ha) and 9390 *Scrub and low forest vegetation with Quercus alnifolia (5.80 ha). The second Nature Reserve is located in the site of Konizi and expands in an area of 23.21 ha, in which habitat 9590* covers an area of 5.72 ha, while the remaining area of the Reserve comprises the habitat 9450 (12.69 ha) and the habitat 9390* (4.80 ha).

10.5. Reference


Guides-Deliverable_A.1.2.pdf; access date: 11/01/2021)
Protection of Habitat Type 9590* Against Forest Fires

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101.1. Introduction

The elaboration of a fire protection plan for the habitat type 9590* aimed to the effective reduction of forest fire risk. The occurrence of even a single forest fire incident near or within the habitat, may threaten its viability due to partial or, even worse, total destruction.

The main parts of the fire protection plan are mentioned below:

i. Description and assessment of the most likely forest fire causes, based on an analysis of forest fire statistical data which afterwards were used for the establishment of five forest fire scenarios in areas located around the habitat type 9590*,

ii. The existing situation analysis and evaluation of the forest fire infrastructure and other forest fire protection measures implemented, and

iii. The elaboration of recommendations in infrastructure and other actions which need to be applied for an integrated protection of the habitat against its most vital thread, the forest fire.

101.2. Forest fire risk

The forest fire causes within and around the study area were listed and analyzed in order to improve the forest fire protection status of the habitat, by applying the most efficient forest fire protection measures for the area. Forest vegetation, topography and climate consist the natural parameters which affect the forest fire risk. On the other hand, demographics and especially agriculture activities and forest recreation activities consisted the human induced parameters of the forest fire risk in the study area.

101.3. Description and assessment of the most prominent forest fire causes

A detailed description of the possible forest fire causes was conducted, covering an area geographically located within and around the habitat type 9590*. The assessment of the forest fire causes followed, based on data extracted from the Department of Forests’ Database. The data which covered the period 1960 – 2016 were analyzed statistically (Fig. 101.1). The results were used for the establishment of five forest fire scenarios (Fig.101.2). The selected areas for the for-

Figure 101.1: Map of starting points of forest fires and burned areas in the Natura 2000 “Koilada Kedron-Kampos” site for the period of 1960 – 2016
est fire scenarios, are located around the habitat type 9590*, covering the whole Natura 2000 site “Koilada Kedron – Kampos”.

![Maps with forest fire scenarios in the study area: onset and development of forest fire](image)

**Figure 10I.2:** Maps with forest fire scenarios in the study area: onset and development of forest fire

### 10I.4. Existing situation analysis and evaluation of the forest fire infrastructure

In detailed existing situation analysis and evaluation of the forest fire infrastructure and other forest fire protection measures implemented by the Department of Forests was given. The evaluation revealed the needs in the areas of prevention, pre-suppression and suppression of forest fires (i.e. patrolling, construction of water tanks, construction of fire track, installation of warning signs, controlled grazing, removal of dry vegetation, facilitating quick access to fire vehicles etc.). Final, summarizing the actions needed and infrastructure for effective reduction of risk of forest fires in habitat type 9590* were recommended. These recommendations are located within and around the habitat type 9590*, covering the whole Natura 2000 site “Koilada Kedron – Kampos” where the actual risk lies due to higher human activity. All the recommended measures which have also been costed and mapped, are presented in the annex.
Concrete Conservation Techniques and their Effectiveness
Based on the Results Acquired - Monitoring and Management of Phytophagous Insects in the Paphos Forest

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10II.1. Introduction
The biotic factors altering the survival and the reproductive success of trees are key drivers of the ecological and dynamic processes that affect vulnerable and narrow forest habitats. The feeding activity of diverse forest insect species may limit growth, reproductive success and survival of trees with both direct and indirect consequences on tree species demography and forest sustainability. The aim of this work in LIFE-KEDROS was to clarify how two critical phytophagous insect groups, namely seed feeders and bark beetles, can constitute a threat to the Cedrus brevifolia habitat in the Pafos forest. Insect seed feeders have a direct influence on the reproductive success of their host trees and subsequently on their regeneration and colonization potentials in natural stands. They can have both demographic and genetic impacts on tree populations by directly eradicating individuals (i.e. embryos within seeds) (Crawley 2014; Boivin et al. 2019). Bark beetles are generally considered as secondary pests that target stressed and weakened trees facing climate, soil, water or pathogen stresses, but some species (primary pests) or some outbreaking populations of secondary pests species can attack and kill healthy trees (Raffa et al. 2015). Most of bark beetles reproduce and spend most of their life cycle in the subcortical region of their host trees, feeding on the phloem, which results in the death of the tree or some tree parts such as branches. One objective here was to determine the seasonal and interannual dynamics of both seed feeders and bark beetles using specific field and laboratory monitoring methods. This was done in order to establish the basic requirements for implementing appropriate tools for both preventive and effective habitat protection against undesirable insect pest outbreaks in the C. brevifolia narrow and vulnerable habitat.

10II.2. Seed predation by insects on Cedrus brevifolia in Cyprus.
Seeds directly influence both population dynamics and genetics of trees as they are key to the local increase of a population, to the replacement of individuals that die in a population, and to the colonization of new areas. Forest insects that specifically feed on tree seeds, namely seed insects, constitute worldwide a major threat for high quality seed supply for ornamental, reforestation, afforestation and conservation purposes when infesting established seed orchards, ex situ plantations and both seed selected and natural stands (Boivin et al. 2019).

In the Mediterranean Basin, cedar (Cedrus spp.) seeds are targeted by highly specialized seed wasps (Megastigmus spp., Hymenoptera: Torymidae) that lay their eggs within the ovules in
young growing cones, then larvae consume both the embryo and the reserve organs during the 2 or 3 followings months. This prevents any possibility of germination of the attacked seeds when released at the end of the cones’ maturation period (usually at fall). For instance, an important pest species, *Megastigmus schimitscheki* Novitzki, has become a particularly successful invader of southern French Atlas cedar (*C. atlantica*) stands since the mid-1990’s, where it generates up to 90% seed loss to seed suppliers in selected stands (Fig. 10II.1). *Megastigmus schimitscheki* is native to Turkey, Lebanon and Syria where it feeds on *C. libani*, and to Cyprus, where it feeds on *C. brevifolia* (Auger-Rozenberg and Boivin 2016). The ecology and the ecological impacts of the invasive populations of *M. schimitscheki* have been intensively studied in France in the last decade (Suez et al. 2013; Boivin et al. 2015; Gidoin et al. 2015), but how this pest behave and interferes with cedars in its native areas remains a black box. Seed damages of *M. schimitscheki*, or other potential seed pest species, to *C. brevifolia* in Cyprus has been poorly studied and documented, and there was a crucial need in LIFE-KEDROS to estimate whether this seed wasp is likely to threaten the natural regeneration potential of *C. brevifolia* stands.

The project LIFE-KEDROS raised the opportunity to assess the quantitative impact of *M. schimitscheki* on *C. brevifolia* seeds across the Pafos forest. It aimed at providing a detailed and wide-range knowledge of the pre-dispersal seed predation risk across the whole habitat. For this purpose, the quality of seeds has been estimated in seed lots collected in the Pafos forest in fall 2016. Seeds were analysed from cones collected randomly on trees before cone disarticulation, so these estimations of seed quality reflected the potential for natural regeneration seed output. This potential corresponds to the proportion of viable seeds in seed samples.

10II.2.1. *Methods*

In the Pafos forest, the cedar cone sampling area consisted in four zones (A, B, C and D) within which three cone sampling plots were defined. This resulted in a total of 12 cone-sampling plots (e.g. plots A1-A3…D1-D3), with five cones being collected randomly in each plot. Cedar cones contain both small- and normal-sized seeds. Small-sized seeds consist only in a seed envelope and a wing, and they result from early seed abortion that is mainly due to pollination failure. Normal-sized seeds reflect both pollination and early seed development successes, and they include viable (with full embryo and reserve tissues), empty (naturally aborted) and wasp-infested winged seeds. After cone collection and manual disarticulation, normal-sized seeds were separated from small-sized ones by the Forest (Cyprus Department Forests) Department of Cyprus and sent to INRAE (Thomas Boivin, France) for the estimations of the relative proportions of viable, empty and seed wasp-infested seeds using numerical X-ray radiography (Faxitron-MX20, 20 kV, 0.3 mA, 1045° with an EZ20 digital scanner).

Numerical X-ray radiography is a useful non-destructive technique that allows the unambiguous assessment of seed contain (Fig. 10II.1). Viable seeds are identified by visualizing the cedar embryo and the reserve storage organs that will feed it during development. Empty seeds do not contain any embryo nor seed storage organs, they result from ovule fertilization failure by pollen or early embryo abortion during embryogenesis. Seed wasp-infested seeds contain one unique mature diapausing larva of *M. schimitscheki* that consumed both the embryo and the reserve storage organs during its development.
10II.2.2 Seed predation rates and field trapping of seed wasps

A total number of 4694 seeds (average number of normal-sized seeds per cone: 78.2 ± 10.3) from the 12 plots were X-rayed (Fig. 10II.2). Percentages of viable seeds displayed no significant variation between sampling areas A-D (P=0.432). These percentages were found relatively high (range: 71.6-91%), which may be explained by statistically similar low percentages of both empty and infested seeds between areas (P=0.887 and P=0.456, respectively). Seed predation however showed inter-
individual variation within sampling plots. It corroborates previous work on *C. atlantica* in France and it suggests that trees might not contribute equally to the seed rain and subsequent regeneration dynamics (Doublet et al. 2019). However, pluri-annual estimates of seed predation rates are still needed in the Pafos forest to test such hypothesis on *C. brevifolia* in Cyprus. Surveys of yellow sticky insect traps installed in each zone throughout 2019 indicated that the presence of *M. schimitscheki* was limited to May-June, which corroborated its previously known oviposition phenology with regard to the early development of cedar cones. Their abundance was low in all these traps (a total number of <100 individuals), which suggests that either populations were at a low level or that these traps were not efficient to successfully mass-trap this species during its short flight period. However, these traps are usually used for qualitative presence/absence information rather than for population density estimates. No parasitoid wasps of *M. schimitscheki* (e.g. possibly Hymenopteras of the *Eupelmus* or *Mesoplobus* genera) were captured in these traps nor emerged from infested seeds kept under natural conditions for subsequent insect emergence. Such difficulty to sample *M. schimitscheki* parasitoids in the field supports either the low abundance of natural predators of the seed wasp in the Pafos forest, or the non-attractivity of sticky traps for such species and parasitism of wasp larvae during the seed post-dispersal phase (i.e. when seeds lay on the ground).

![Figure 10II.2: Relative proportions of viable, empty and seed wasp-infested seeds of Cedrus brevifolia (seed quality) in four study sites (A – D) in the Pafos forest. Seed quality has been characterized by numerical X-ray radiography at INRAE, France. Only viable seeds are available for natural regeneration. N: average number (+ standard error) of normal-sized seeds per cone.](image)

**10II.2.3. Conclusions on the outcomes of LIFE-KEDROS on *C. brevifolia* seed predation**

The low overall predation rate of *M. schimitscheki* on *C. brevifolia* seeds indicated that this insect species may currently not constitute a major threat for tree seed outputs during the pre-dispersal phase as they seem to occur at rather low abundance in the Pafos forest. Interestingly, this was in clear contrast with the situation observed in French cedar stands where seed damages can reach 50-90%. During the project, cedar cones were collected in a year of high cone production (2016), which was followed by consecutive years of null or low cone production and which prevented repeated estimations of seed predation rates. It is commonly observed that seed predation rate vary inter-annually as a function of cone production, with low seed predation rates during years of high cone production (mast years) and high seed predation rates during years of low cone production (non-mast years). The latter results from larger abundance of emerging seed predators that has
been favoured by large seed amounts during previous mast years and the occurrence of a seed resource at lower abundance as a non-mast year. One consequence of such resource limitation during non-mast years is an overall lower number of infested seeds in the population, which leads to lower abundance of emerging seed predators and lower seed infestation rates during subsequent mast years. This refers to as the predator satiation hypothesis that predicts alternating periods of seed predator satiation and starvation due to overabundance and shortage of seeds during mast and non-mast years, respectively (Janzen 1971). Little is documented about seed production cycles of *C. brevifolia* in Cyprus, but both high fragmentation levels in tree populations and adverse environmental conditions (e.g. drought periods) are thought to generate longer intervals between mast years, which apparently occur every four to seven years in *C. brevifolia* (Cyprus Forestry Department 2005). Such several consecutive years of starvation for *M. schimitscheki* larvae between each masting period might contribute to maintaining their populations at low levels, despite a fraction of larval cohorts displays temporal dispersal abilities (i.e. prolonged diapause) as an evolutionary response to temporal unpredictability of the seed resource (Suez et al. 2013). Moreover, it is likely that the seed wasp possesses natural enemies in its native area that regulate its population growth, although we were not able here to sample and identify any parasitoid candidate in the Pafos forest. The difficulty to sample such natural enemies in the study sites may reflect their low overall abundance, which might be linked to the actual low abundance of their prey.

As masting trees such as *C. brevifolia* basically rely on seed production during mast years for natural regeneration, it is thus unlikely that seed wasp such as *M. schimitscheki* impede this process in the Pafos forest. Thus, further efforts to understand or predict natural regeneration issues in this narrow forest habitat should probably address factors affecting cone/seed production cycles, as well the post-dispersal limiting factors of regeneration, e.g. seed and seedling survival or seed predation on the ground by birds, insects and mammals.

The project LIFE-KEDROS has important outcomes for seed cedar conservation in Cyprus and beyond the island boundaries (e.g. France, Northern Africa) where natural forests and plantations of closely related *Cedrus* species occur in Mediterranean landscapes. In Cyprus, despite seed predation rates are low in the natural stands of *C. brevifolia*, *M. schimitscheki* remains a potential risk for seed outputs in *ex situ* plantations aiming at producing seeds for further plantations. In such plantations, trees may be more synchronized for fructification than in natural stands including different age classes and micro-environmental factors affecting seed production. This may be a favourable factor for the growth of seed wasp populations due to less variable resource availability, which would result in lower seed productivity of *ex situ* plantations such as that observed in seed orchards of other conifers in Europe and the USA (Boivin and Auger-Rozenberg 2016). One important recommendation is to collect before cone disarticulation every cone of every tree in the plantations every year, and then destroy all cones or seeds that will not be used for conservation purposes. This is expected to prevent wasps to build up local populations in the plantations.

The project LIFE-KEDROS also stimulated further research and collaboration opportunities on the natural enemies of *M. schimitscheki* in its native area, which became a critical issue for the management of its invasive populations in France. This seed wasp indeed leads to important seed loss in southern French stands of *C. atlantica* selected for seed provisioning for reforestation and ornamental programs. In France, this appears as a potential constraint in the current perspectives of establishing seed orchards for this tree species, which is considered as a response of French Mediterranean forest to climate change.

10II.3. Bark beetles (Coleoptera: Curculionidae) in the Pafos forest

Biotic aggravating factors of forest tree health include numerous phytophagous insect species, such as bark beetles, which commonly feed on the cambial tissues of dead, recently damaged or even heavily-defended healthy trees (Paine et al. 1997). A majority of bark beetle species are secondary pests that target stressed and weakened trees facing pathogenic, climate and water stresses (Raffa et al. 2015). Most of them reproduce and spend most of their life cycle in the
subcortical region of their host trees, feeding on the phloem, which results in the death of the
Tree or some of its parts (e.g. branches). Bark beetle populations display transitions between
Endemic states in which they reside in stands at very low densities and are expected to kill only
A few weakened trees, and epidemic states in which they reach very high densities over large
Areas causing high tree mortality at both the stand and landscape scales (Kausrud et al. 2011).
Consequently, the potential for bark beetle outbreaks represents a serious threat for natural C.
brevifolia stands, which are restricted to only one area in Europe.

In LIFE-KEDROS, there was a crucial need to identify the main bark beetle species occurring in
Both pure and mixed (with P. brutia) stands of C. brevifolia as bark beetle communities remain
Poorly documented in Cyprus. The field monitoring of bark beetle populations in the Pafos forest
Has been carried out from spring to fall across the same four study zones (A-D) as for seed wasps
during three consecutive years. This aimed at defining the seasonal activity of the main bark
Beetle species in this narrow habitat.

10II.3.1. Methods

Two Multi Wit bark beetle slit traps (Witasek, Austria) have been baited with both ethanol and
The aggregation pheromone lure Erosowit® (Witasek, Austria) (Fig. 10II.3). Traps have been set in
each of the four study sites (A-D) implemented in the Pafos forest. Such baited-traps are glob-
ally known to enhance bark beetle detection probability (no repellent effect, attraction of many
Individuals) and are extensively used in programs of early-warning surveillance, monitoring and
Control of bark beetles, wood borers, and their natural enemies. Insect phenology and abun-
dance have been analysed using sequential and cumulated insect counts in each baited traps
Between May and October 2017, 2018 and 2019. Traps have been collected every two weeks and
All trapped insect specimens have been identified and counted (Fig. 10II.3).

Figure 10II.3: Monitoring of bark beetle populations in the Pafos forest. (a) Multi Wit bark beetle slit trap baited with
Both ethanol and an aggregation pheromone lure for Orthotomicus erosus. (b, c) Orthotomicus erosus adult collected
In the Pafos forest, on dorsal and ventral views, respectively. This species was by far the most abundant found in the traps,
it can be a threat for cedars during population outbreaks favoured by drought or fire. (d) Bi-weekly trap surveys at the
Four study sites resulted in a large number of samples to be processed by the Department of Forests, only fraction here.
(Photos: ©LIFE-KEDROS)
10II.3.2. *Species collected in the Pafos forest and their basic ecological features*

The 3-year of trap survey in the Pafos forest allowed the detection of two main bark beetle species, *Orthotomicus erosus* Wollaston (Coleoptera: Curculionidae) and *Hylurgus ligniperda* Fabricius (Coleoptera: Curculionidae), and one species of a generalist predator of bark beetles, *Aulonium* sp. (Coleoptera: Colydiidae).

**Orthotomicus erosus** Wollaston (size: 2.5–3.5 mm): Native populations of this bark beetle are widely distributed throughout central and southern Europe, North Africa, Western and central Asia, and it has been introduced in Northern Europe, South Africa, Chile, Fiji Islands and the United States. It occurs in a wide range of Mediterranean countries, including Turkey, Syria, Lebanon and Cyprus. To reproduce and develop, *O. erosus* can apparently use all *Pinus* species (*P. halepensis*, *P. radiata*, *P. pinea*, *P. pinaster*, *P. brutia*, *P. canariensis*, *P. sylvestris*, and *P. nigra*), but it has also been found to attack *C. libani* and *C. atlantica* (Lieutier et al. 2016). Host colonization generally occurs from March to October, essentially on weakened trees, which the adults attack on main branches and trunk by an irregular multiramous system consisting of 3-5 irregular egg galleries, longitudinal, and with no clear pattern. Larvae have three instars, which galleries strongly affect sapwood, and young adults feed on phloem to reach sexual maturation before emergence. Volutinism may expand from 1 to 7 generations per year, depending on the geographical area and climatic conditions. For instance, Mediterranean populations infesting *P. halepensis* may reach four generations per year (Lieutier et al. 2016).

*Orthotomicus erosus* is considered as a secondary species, common in recently felled trees and non-debarked logs or in trees attacked and killed by more aggressive bark beetle species. Living trees affected by some kind of stress (fire and drought) may also be attacked, causing their death during epidemic phases from massive attacks. Adults are the vector of blue stain fungi such as *Ophiostoma* and *Leptographium* (Kirisits 2004), and of the phoretic fungus *Fusarium culmorum* (Romón et al. 2007).

**Hylurgus ligniperda** Fabricius (size: 5–5.5 mm): Native populations of this bark beetle are widely distributed in Central, Eastern and Southern Europe, Crimea and Caucasus, and Northern Africa, and it has been introduced in the United States (California), South Africa, Australia, New Zealand, Chile, Brazil, Uruguay, Sri Lanka and Japan. It is common across the Mediterranean basin, including Greece, Turkey and Cyprus. *Hylurgus ligniperda* is oligophagous on both Mediterranean (*P. halepensis*, *P. brutia*, *P. pinaster*, *P. pinea*) and continental (*P. sylvestris*, *P. nigra*) pine species. It is still not recorded on the *Cedrus* genus (Lieutier et al. 2016). Adult flights start in late May and in early June to mate and infest the host trees. Adults usually infest the lower part of trunk, root collar and emerging roots of large recently dead, weakened or cut trees. The female bores a very irregular, large and long egg gallery, often bifurcated without a precise pattern. Larval galleries may reach 8 cm in an irregular and sinuous pattern (Faccoli 2015). Conversely to *O. erosus*, voltinism is globally restricted to two generations per year.

*Hylurgus ligniperda* is considered as a very common secondary and aggressive species in pine forests, infesting only strongly decaying trees or fresh stumps. This bark beetle is apparently of no economic importance, except in stressed and dying pine plantations (Lieutier et al. 2016). Literature suggests that *H. ligniperda* is unlikely to threaten natural stands of *C. brevifolia*, and its occurrence in traps of the survey is most probably associated with the presence of pines in mixed cedar stands or in the surroundings of pure cedar stands.

**Aulonium** sp. Predatory beetles of this genus are usually most abundant in trees attacked by *O. erosus*, *Pityoktenes calcarius* or *Tomicus destructus* (Lieutier et al. 2016). Interestingly, *A. ruficorne* adults are generally captured in large numbers by traps baited with pheromone lures of *O. erosus* or *Ips typographus* (e.g. Mendel and Opatowski 1997). The main activity of *Aulonium* occurs...
during the early development stage of their preys, as it feeds mainly on eggs and young larvae, with *A. ruficorne* leading for instance to important mortality rates on immature *O. erosus* in Israel during spring and late summer (Podoler et al. 1990). *Aulanus* sp. Usually leave the tree soon after the emergence of its bark beetle prey (Lieutier et al. 2016).

10II.3.3. **Temporal and spatial distributions of bark beetles and predators in the Pafos forest**

The total abundances of *O. erosus*, *H. ligniperda* and *Aulonium* sp. per trap over the study period (2017-2019) were used to test the fixed effects of species, year, site and their interactions (year X site) using a generalized linear model (GLM with a Poisson family). The GLM indicated that there were significant effects of each factor (species, year and site) on insect abundance (all P-values<0.0001). This analysis indicated in particular that: (i) one species (*O. erosus*) was predominantly represented in insect counts compared to the others, (ii) insects were significantly more abundant in sites C and D than in A and B ones, and (iii) there was a decreasing trend in abundance for all species from 2017 to 2019. These trends were also supported by a significant interaction between year and site factors (P<0.0001). Higher abundance in sites C and D may be explained by the presence of pines within and in the surroundings (mixed stands). The decrease in overall insect abundance during the study period (Fig. 10II.4) may be explained by factors including: i) increased individual tree health within sites, which decreased the success of insect attacks, ii) the mass-trapping effects of baited traps during an endemic phase of insect population (e.g. in 2017), which affected insects’ demography. It would be interesting to assess the health status of both cedars and pines in the C and D sites, where the decrease in insect abundance was more pronounced than in the other study sites.

In 2017, the temporal distribution of trap captures in the Pafos forest, i.e. pooling the four study sites, suggested two flight periods between early June and early November (Fig. 10II.4). This might reflect at least two consecutive generations for these three species, with possible more overlapping generations for *O. erosus*, as there is currently no evidence of sister generations in the literature for each of them (Lieutier et al. 2016). High insect counts at the first date of survey (1st June 2017) suggested that the first flight had begun earlier in the area, possibly in early or mid-May. In each species, the first flight ended by late July and the second flight occurred between early September and early November. It is noteworthy that both *O. erosus* and *H. ligniperda* shared a similar seasonality and that they may both benefit from similar environmental conditions in the Pafos forest. The data however suggested that *O. erosus* may have a slightly earlier phenology than *H. ligniperda* during both first and second flights (Fig. 10II.4). Moreover, the seasonal occurrence of *Aulonium* sp. did not show any significant divergence with that of *O. erosus* (Mann-Whitney-Wilcoxon test, P=0.105), while it was significantly divergent from that of *H. ligniperda* (Mann-Whitney-Wilcoxon test, P=0.043). One possible explanation may rely on a stronger link between the abundance of this predator and the abundance of *O. erosus* rather than with that of *H. ligniperda* (Lieutier et al. 2016). Patterns of temporal occurrences differed between 2017 and both 2018 and 2019, which was likely due to the substantially lower insect abundances in traps during the latter years. *Orthotomicus erosus* remained the most abundant species in traps, with a continuous occurrence from May to September, suggesting several overlapping generations during this period. Interestingly, a very low total number of insects have been trapped in 2019, suggesting that the *O. erosus* population had entered an endemic state in the Pafos forest. This is in clear contrast with trapping data in 2017 and might suggest a change in the population dynamics of this species (Fig. 10II.4).
10II.4. Conclusions on the outcomes of LIFE-KEDROS on bark beetle monitoring and risk management in the Pafos forest

Field monitoring of bark beetles in study sites A-D of the Pafos forest confirmed the presence of a species that is already known to be potentially harmful to weakened cedar trees, namely *O. erosus*. This species is however known to be oligophagous, i.e. it can be associated to both *Cedrus* and *Pinus* genera. Trapping data also revealed the presence of *Aulonium* sp., which is already known to co-occur with *O. erosus* as an efficient predator. There was an effect of site on abundances in the traps, i.e. bark beetle populations were more abundant in both C and D sites, which could be related to local population increase within cedars and/or higher frequency of occurrence of pines. Overall, *O. erosus* was by far the most abundant species trapped across the area throughout this 3-year study period. This is very likely to result from the use of the specific aggregation pheromone of this species in addition to alcohol in the trapping design. Although *H. ligniperda* is oligophagous, it is preferentially observed on *Pinus* species rather on *Cedrus*, which may potentially explain lower abundances in the traps in Pafos’ cedar area. The use of the *O. erosus* aggregation pheromone may also explain the significant association found between *O. erosus* and *Aulonium* sp. captures, as the latter uses the pheromone to locate its preys when foraging, parallelly to volatile compounds (e.g. terpenes) produced by weakened and attacked trees.

Interception baited traps as those implemented here are widely acknowledged monitoring tools of bark beetle populations in forest ecosystems, but one should keep in mind that the use of attractants does not fully ascertain that the individuals/species caught in the trap would have necessarily been found attacking the trees. This may be particularly true regarding *H. ligniperda*,

Figure 10II.4: Temporal variation in abundance of *O. erosus*, *H. ligniperda* and *Aulonium* sp. in baited traps implemented in the Pafos forest during 2017, 2018 and 2019. Traps were monitored every two weeks for insect identifications and counts. Data of the four study sites A-D have been pooled here to represent overall abundance in traps in the habitat. Note that abundance is presented on a logarithmic scale in 2017 to allow the graphical representation of both high and low abundance values.
which may have been attracted by alcohol baits from surrounding pine areas and this might also result in the lower abundances recorded in this species across the study area. In a next future, direct observations of signs of bark beetle attacks on different tree parts (e.g. branches, trunk) would help clarifying the prevalence of *O. erosus* and *H. ligniperda* as potentially harmful species to *C. brevifolia* in the study area. Maintaining annual monitoring of insect populations by this type of trapping remains a necessity to track possible variations in bark beetle population levels and to determine potential thresholds for transitions from endemic to epidemic stages. LIFE-KEDROS stimulates similar investigations and monitoring purposes in planted and natural stands of *C. atlantica* in France and other Mediterranean hosting the *Cedrus* genus, where bark beetle populations remain often poorly documented despite their potential as emerging forest pest in the future, especially in the context of climate change and environmental disturbance (e.g. drought and fire). With regard to *O. erosus*, it has been shown that development still accelerates until 36°C, a temperature at which both oviposition and egg development still occur (Lieutier et al. 2016). This demonstrates that such worldwide bark bettle species is adapted to high temperatures and thus still constitute a potential harmful forest insect for cedar populations in the following decades.

Another important outcome of LIFE-KEDROS is the confirmation that the presence of *O. erosus* may interfere with silvicultural actions implying to cut selected trees, which may not be removed from the habitat for practical and financial reasons. Such remnant cut trees constitute a resource for the latest flights of bark beetles (early September-late October) and this likely increases further risks of insect local outbreaks. However, such interference between silvicultural actions and bark beetle risk management can be bypassed by two alternative means:

i. Cut trees are debarked as soon as they are cut, so that bark beetles can not use them for both feeding and reproduction, which substantially impede subsequent population outbreaks and damages to living trees.

ii. Cut trees are debarked in early winter (e.g. late November-December), i.e. after bark beetles have infested the cut trees, so that these trees constitute trap trees for insects that will not survive debarkment after infestation. This can prevent further massive attacks on living trees and impede local insect population growth.

Additional bated insect traps (the same as those used for bark beetle monitoring) can also be implemented in late August on sites where tree cuts are expected in order to intercept and mass trap flying adults of *O. erosus* before they reproduce on cut trees. The management of bark beetle populations is generally a complex issue in natural forest ecosystems, and especially in vulnerable narrow habitats. The implementation of a such combination of effective prophylactic methods against insects during endemic stages remains the most efficient and parsimonious option.

### 10II.5. References


PART B • CHAPTER 10


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