Disturbances of EU forests caused by biotic agents

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<th>Description</th>
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<tr>
<td>APHIS</td>
<td>Animal and Plant Health Inspection Service</td>
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<tr>
<td>BAWBILT</td>
<td>Bark and Wood Boring Insects in Living Trees</td>
</tr>
<tr>
<td>CLRTAP</td>
<td>Convention on Long-range Transboundary Air Pollution</td>
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<tr>
<td>DAISIE</td>
<td>Delivering Alien Invasive Species Inventories for Europe</td>
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<tr>
<td>DFDE</td>
<td>Database on Forest Disturbances in Europe</td>
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<tr>
<td>EC</td>
<td>European Commission</td>
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<tr>
<td>EEA</td>
<td>European Environment Agency</td>
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<td>EFI</td>
<td>European Forest Institute</td>
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<td>EFSA</td>
<td>European Food safety Authority</td>
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<td>ENFIN</td>
<td>European National Forest Inventory Network</td>
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<td>EPPO</td>
<td>European and Mediterranean Plant Protection Organisation</td>
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<td>EU</td>
<td>European Union</td>
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<td>EWS</td>
<td>Early-Warning System</td>
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<td>FAO</td>
<td>Food and Agriculture Organisation</td>
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<td>FHD</td>
<td>Forest Health Department</td>
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<td>FHP</td>
<td>Forest Health Protection</td>
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<td>FVO</td>
<td>Food and Veterinary Office</td>
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<tr>
<td>ICP-Forests</td>
<td>International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests</td>
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<tr>
<td>IPPC</td>
<td>International Plant Protection Convention</td>
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<tr>
<td>ISPMs</td>
<td>International Standards for Phytosanitary Measures</td>
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<tr>
<td>IUCN</td>
<td>International Union for Conservation of Nature</td>
</tr>
<tr>
<td>IUFRO</td>
<td>International Union of Forestry Research</td>
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<tr>
<td>MCPFE</td>
<td>Ministerial Conference on the Protection of Forests in Europe</td>
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<tr>
<td>MS</td>
<td>Member State</td>
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<tr>
<td>NFI</td>
<td>National Forest Inventories</td>
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<tr>
<td>NFMA</td>
<td>National Forest Monitoring and Assessment</td>
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<tr>
<td>NPPOs</td>
<td>National Plant Protection Organisations</td>
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<td>NWP</td>
<td>Non-wood forest products</td>
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ONF  Office National des Forêts
PCR  Polymerase Chain Reaction
PRA  Pest Risk Analysis
PROLUNP  Programa Nacional de Luta contra o Nemátode do Pinheiro
SFM  Sustainable Forest Management
SLU  Swedish University of Agriculture Sciences
SPHDs  State Plant Health Directors
SPROs  State Plant Regulatory Officials
SRES  Special Report on Emission Scenarios
SSC  Species Survival Commission
UNCED  United Nations Conference on Environment and Development
UNEC  United Nations Economic Commission for Europe
UNFF  United Nations Forum on Forests
USDA  US Department of Agriculture
WSL  Federal Institute for Forest, Snow and Landscape Research
Glossary

In the report, the words defined in the glossary are annotated with an asterix.

**Abiotic agents** include storms, fires, wind, snow, nutrient deficiency, soil condition and stand density (MCPFE 2011).

**Abiotic disturbance** refered to a damage caused by non-living agents (snow, storms, fires, wind) (MCPFE 2011).

**Alien species** refers to species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce (Decision VI/23 of the Conference of the Parties to the CBD)

**Biotic agents** include insects and diseases, wildlife in forests and cattle grazing in wooded land (MCPFE 2011).

**Biotic disturbance** refers to a disturbance caused by living organisms (fungi, insects, etc.) (MCPFE 2011).

**Disturbance** is a damage to the forest which may be caused by biotic or abiotic agents, resulting in death, or a significant loss of vitality, productivity or value of trees and other components of the forest ecosystem (MCPFE 2011).

**Endemic species** is a native species restricted to a particular geographic region owing to factors such as isolation or in response to soil or climatic conditions (CBD 2011)

**Forest available for wood supply** is a forest land where any legal, economic, or specific environmental restrictions do not have a significant impact on the supply of wood. It includes areas where, although there are no such restrictions, harvesting is not taking place, for example areas included in long-term utilisation plans or intentions (MCPFE 2011).

**Forest** is land with tree crown cover (or equivalent stocking level) of more than 10 percent and area of more than 0.5 ha. The trees should be able to reach a minimum height of 5m at maturity in situ. Rubberwood plantations and cork oak stands are included, but this definition excludes land predominantly used for agricultural practices (FAO 1998).

**Forest plantation** is defined as a forest stand established by planting or/and seeding in the process of afforestation or reforestation. It excludes stands which were established as plantations but which have been without intensive management for a significant period of time. These should be considered semi-natural (MCPFE 2011).

**Forest services** are ecosystem services provided by forests and include (1) Protection values (against soil erosion by air or water, avalanches, mud and rock slides, flooding, air pollution, noise, etc.); (2) social and economic values (e.g. hunting and fishing, other leisure activities, including recreation, sport and tourism); (3) Aesthetic, cultural, historical, spiritual and scientific values (including landscape and amenity) (FRA 2000).
Forestry is defined as all the activities related to the management of forests and other wooded land the production and supply of wood and/or other goods and services (TBFRA 2000).

Multivoltine species refers to a species which may have many generations per year.

Native species refers to any plant, animal, fungi, or microorganisms that occurs naturally in a given area or region (FAO 2002).

Other wooded land is land either with a tree crown cover (or equivalent stocking level) of 5-10 percent of trees which are able to reach a height of 5 m at maturity in situ; or with a crown cover (or equivalent stocking level) of more than 10 percent of trees that are not able to reach a height of 5 m at maturity in situ (e.g. dwarf or stunted trees) and shrub or bush cover. It excludes areas having the tree, shrub or bush cover specified above but of less than 0.5 ha and width of 20 m, which are classified under "other land" and predominantly used for agricultural practices (TBFRA 2000).

Pest is defined as any species, strain or biotype of plant, animal or pathogenic agent injurious to plants or plant products (ISPM 2008).

Phytosanitary measure refers to any legislation, regulation or official procedure having the purpose to prevent the introduction and/or spread of quarantine pests, or to limit the economic impact of regulated non-quarantine pests (ISPM 2008).

Quarantine pest is defined as a pest of potential economic importance to the area endangered thereby and not yet present there, or present but not widely distributed and being officially controlled (FAO 1990, revised FAO 1995, IPPC 1997).

Semi-natural forests represent forests, which are neither forests undisturbed by man nor plantations as defined separately (MCPFE 2011).

Sustainable Forest Management definition has been developed by the member countries of FOREST EUROPE as follow: "the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems." (MCPFE, 2011)

Univoltine species refers to a species, which has only one generation per year.
Executive summary

Forest ecosystems are currently facing rapidly expanding new environmental conditions as a result of climatic and anthropogenic changes. While disturbances have always been an important part of the natural forest ecosystems dynamics allowing regeneration, the scale and nature of these disturbances are causing concern. They are mainly challenging traditional forest management practices, often aimed at maintaining or restoring past conditions. Such choices need serious re-assessment, since they could create forests that are ill-adapted to current conditions, and more susceptible to disturbances. The recent endorsement of sustainable forest management in Europe goes in that direction.

This review provides a synthesis of the risks caused by biotic agents to EU forests, with the aim to provide managers and policy-makers with the current evidence-base and tools to fight these risks. Most work so far has usually focused on a specific type of biotic agent, or a restricted geographical extent. This has led to a sometimes segmented view of the problem, and a synthesis of the recent advances is timely.

**EU’s forests and biotic risks**

Forests and other wooded lands form the natural vegetation in most parts of the land base of EU-27 MS, except at high latitudes and altitudes. Even if once cleared for other land uses, forests and woodlands are common land covers that deliver wood and non-wood goods, ecosystem services and landscapes values, all of which are used to generate products and benefits for human well-being and quality of life. The forest sector is thus an important source of economic and social benefits to EU population, with great regional variations. Contemporary policies concerning forests and woodlands increasingly stress their multifunctional role to society.

But while EU forest cover has been increasing over the past 20 years, some concerns can be raised. Several factors may concur to increase the vulnerability of EU forests to disturbances, including the trend towards increased fragmentation and intensively managed semi-natural forests or plantations, often with alien trees. Moreover, climate change and economic globalisation are new factors to take into consideration.

Biotic disturbance agents affecting trees in forests and woodlands include fungi, invertebrates and vertebrates. While disturbances caused by biotic agents are natural elements in forest and woodland ecosystems, they may also be considered as pests and cause negative consequences for the delivery of goods and services. Additionally, biotic disturbance agents may be divided into those that occur naturally in EU’s forests and woodlands (native agents) and those that originate from other continents (alien agents), which are subject to biosecurity.

**Current risks caused by biotic agents**

Around six percent of the EU forest area is estimated to be disturbed by biotic agents. However, surprisingly little is still known about the scale of damages caused by biotic agents in the EU, or about whether they represent a growing cause of concern. No reliable long-term trends are available at EU level, and evidence is mixed. However, biotic agents have clear impacts on ecosystems, sometimes modifying the structure of entire forest communities and landscapes,
Executive summary

and in turn, that of the associated species and ecosystem processes. As a result, both wood quantity and quality may be reduced, as well as other forest values, such as non wood forest products and recreation.

Biotic risk can be seen as depending on (a) the hazard likelihood, (b) the susceptibility to the agent, and (c) the exposure to the forest values. The sheer diversity of agents of biotic disturbances makes it particularly difficult to characterise biotic hazard at EU level. But overall biotic hazard depends on the survival and reproduction conditions of the biotic agent, on its dispersal capacity, as well as on biotic interactions, and mainly on whether it can attack several hosts or not. Biotic outbreaks are generally facilitated by prior stress on forest ecosystems, whether through previous infestations or through other abiotic factors, over-exploitation or fragmentation. Stand susceptibility to pest also typically increases with the age and homogeneity of forest stands. Thus, the trend towards more monocultures, higher average growing stocks and older stands in recent years is putting forests at greater risk of biotic damage.

Moreover, climatic factors are very important drivers of forest biotic outbreaks and favourable weather conditions may increase the severity of biotic outbreaks. It is also likely that changing climatic conditions will open niches for new pest-mediated outbreaks. In particular, as the probability that alien biotic agents enter in the EU increases, largely promoted by global trade, but also by changing climatic conditions, alien biotic agents are likely to cause growing risks for biosecurity.

But the extent to which a particular agent is deemed to have positive or negative consequences to economic, ecological or social dimensions of forests and other wooded lands is highly context-dependent. Aside from the hazard likelihood, susceptibility and exposure, it also depends on the desired product profile of a particular area of forest and woodland. Bark beetles and large herbivores are good examples. Thus, while a bark beetle outbreak in a forest managed for wood production is considered as a pest that should be treated, an outbreak in a large protected area, where the aim is to maintain biodiversity by allowing natural disturbances and their natural range of variability, this agent results in increased amount of habitat for species that require dead wood in different decay stages. Similarly, the sets of values of cultural woodlands may require high grazing or browsing pressures by domestic or wild large herbivores, while the sets of values in natural forests may be threatened by the same biotic disturbance agents.

Lessons learnt from selected biotic agents

Biotic agents, when controlled too late or inappropriately, can cause severe damages. A good example is the Dutch elm disease, a widespread disease of elm trees, caused by two strains of fungi (Ophiostoma ulmi and O. novo-ulmi) that wiped out most the European populations of elms. The environmental and economic impacts of elm dieback have been considerable. None of the management approaches used proved successful. Experience shows that actions must be taken rapidly and applied rigorously for any management attempts to be effective. Accurate and rapid diagnosis is also important.

While more information may be available for the management of native biotic agents, they may also be a cause of concern and spread under climate change. Ips Typographus is native and widely distributed in EU forests, and mostly a secondary biotic agent, affecting trees that are already weakened. As a result, it is possible to assess the level of risk of forest infestation for this species,
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According to the combination of environmental traits that are present, and on the management aims. But Ips Typographus is currently expanding its range, and interestingly, outbreak risks are similarly high outside its natural range than in non-managed, old-growth spruce stands.

It is much more difficult to predict and control outbreaks of alien biotic agents, even when they were well-known in their native range. A good example is the Chestnut canker (or chestnut blight), a disease that caused severe damages in the US over the past century. It appeared in 1938 in the EU and is currently widespread in the distribution range of the chestnut, although a natural event attenuated the virulence of the pathogen (C. parasitica), leading to much lower impacts than in the US. Given that several strains of C. parasitica are known in Europe, it is difficult to predict how these may respond to changing climatic conditions. Research into understanding the development of hypovirulence appears a promising management option.

Climate change may profit both native and alien biotic agents. Thaumetopoea pityocampa, is a native widespread defoliator that seems to be profiting from climate change. It is exhibiting altitudinal shifts in its distribution due to improved winter survival as well as latitudinal shifts. Moreover, it has been able to recruit new host species resulting in new distributions. Bursaphelenchus xylophilus, is an alien invasive nematode attacking pine trees that is currently only present in Portugal, but has the potential to spread northwards and has been listed as a quarantine pest. In this case, management of the pathways (i.e. trade and entry points) appears the most efficient approach to prevent spread in the rest of the EU.

Several examples of emerging or recently emerged risks offer opportunities for early control. Sudden oak death is a recent disease caused by the alien pathogen Phytophthora ramorum that is already present in most of Europe. The pathogen has a wide range of hosts, and climate change is predicted to benefit the spread of the pathogen, making it one of the most significant threats known to forest ecosystems. Control measures in the ornamental nursery stock industry, which is one of the main routes of spread of the pathogen, could form an efficient prevention measure. Similarly, Ash dieback is a rapidly spreading disease caused by the fungus Chalara fraxinea. It is now widespread in central and northern Europe and causes the rapid decline of Ash trees of all ages. Transport of plants from nurseries seems to be the main vector of disease spread, therefore control measures related to trade could help prevent further spread of the disease. Such measures could also be applied for Anoplophora chinensis, a potentially emerging threat to EU forests currently only found in the urban context.

Forest ecosystem management

Because European regions differ markedly in terms of forest types, biotic disturbance agents, profiles of forest values, and power of stakeholders, it is impossible to provide common guidelines concerning the management of biotic disturbances for the entire EU. Rather a decision-tree is recommended, whereby based on the forest profile, and thus on the priorities, specific management actions may be advised.

However, as a general guideline, the review of biotic risks suggests that forest management systems should be adapted to encourage a more natural composition, structure and functioning of forest stands and landscapes. The features of close-to-nature management may need to be adapted for different disturbance regimes.
There is a diversity of forest management and traditional landscape management systems available to improve the match between natural dynamics and management system in the EU. EU forests are naturally very diverse in terms of their natural dynamics. Three main groups of disturbance regimes can be distinguished, using an age gradient: even-aged dynamics, caused by large-scale disturbances, uneven-aged dynamics, where adaptation to low intensity disturbances produces stands with several aged cohorts, and all-aged dynamics, with gap formation formed by small-scale, localised disturbances. Currently, forests are managed using a similar gradient from even-aged to multi-aged and uneven-aged systems, depending on the proportion of trees removed at each treatment, and on the size of the treatment unit. However, there is room for improvement of the match between natural dynamic on the one hand, and forest management and traditional landscape management systems on the other. This is implicitly requested in the conclusions of the report on the State of Europe’s Forests 2011.

Future trends in forest management are expected to diverge, including both increased ‘close-to-nature’ forest management methods to satisfy social and ecological functions, and more intensive methods more similar to agriculture than forestry based on naturally occurring tree species. This will lead to challenges in term of forest governance for an integrated landscape approach to spatial planning across forest land ownerships.

Future trends under climate change

Scenarios regarding the effects on climate change on forests vary among regions, and the relative effects of climate change in relation to contemporary patterns of land use change appear to change more on lower and very high latitudes compared to mid-level latitudes in the EU.

The effects of climate change are going to be different for each host-pathogen system in different locations, so that it is impossible to make generalisations. A combination of all climatic factors, temperature, precipitation, humidity, CO$_2$ will almost certainly play a role in all biotic agents-hosts interactions. Moreover, changes in the associated flora (and fauna) at different trophic levels as climate changes may lead to additional modifications of the host-pathogen interactions.

Thus, as climate changes, modifications in the distributions of both the hosts and the biotic agents, in particular altitudinal expansion of the northern limit of species ranges are likely to be observed, as well as increased frequency and severity of outbreaks. Further, it is important to note that not only alien invasive pathogens will be affected by climate change; the distributions and activities of native pathogens will also change.

But many uncertainties remain as to whether biotic agents will cause a growing threat to forests under climate change. There are clear knowledge gaps regarding the combination of factors that promote biotic outbreak conditions for most species, the ecological impacts of biotic outbreaks on the associated communities and the potentials of adaptation of both hosts and biotic agents. As a result, the predicted distributions of these species are also plagued with uncertainty. The shifts in tree species distributions and of their pathogens, whether native or alien, is however likely to lead to new opportunities for uncontrolled biotic spread.

Given the uncertainties, simple frameworks for helping managers prioritise actions may be suggested. The health of forests should be evaluated in terms of how future climate may influence the overall ecology of forest vegetation and biotic agents. A pro-active approach may
thus have a better chance of reducing or avoiding negative impacts than reactive approaches, although both will be needed. Anticipatory adaptation can help decide which species should be planted today that will do well in a different climate in the future, and given the local site characteristics. As damage increases in a forest type, the forest manager will also need a prioritisation system to help him decide which tree species have a long-term future and are worthy of management, and which species may be lost and should therefore not be promoted. Unforeseen natural disturbances or extreme weather events will always call for reactive responses.

In-line with SFM, actions may be aimed at societal adaptation, in terms of forest management aims, forest adaptation, in terms of management practices, or adaptations to the impacts, by modifying the uses of the forest. Given the above choices, a distinct set of operational and strategic management practices will be needed.

Information, monitoring and early warning systems

Better understanding of biotic risks in the EU is needed to design adapted management strategies. This requires systematic long-term monitoring of disturbance agents, which could also serve as an early warning system to assess whether or not particular agents are increasing.

Two independent EU-wide systems for monitoring of forest health exist: ICP-Forest and the National Forest Inventories (NFI). In both systems, the monitoring of biotic agents or damages is poor, due to a lack of obligation to record this data or to the absence of standard protocols. Moreover, such large-scale fixed sample grids surveys are not designed for monitoring irregular, often rare or localised events, such as biotic outbreaks. As a result, there are currently no long-term trends on biotic occurrence or damage in EU forests.

Efficient monitoring and early warning systems for biotic agents require finding a balance between the density of the survey grid and the frequency of the data collection, so as to be able to detect even small-scale outbreaks, in a cost-efficient way. Long-term monitoring on a sufficient number of sites is essential to obtain time series for estimating trends. However, effective information on biotic outbreaks occurrence and spread can only be obtained through intensive, targeted monitoring. Instead, multi-functional systems adapted to different needs, may be better suited. An example is the Swedish forest health monitoring system, which integrates NFIs but complements it with several other tools, including its Skogs skada database using citizen science for reporting damages throughout the year and throughout the country. Similarly, the early warning system for forest health threats in the US, which is based on a system of four loose functional components, has the flexibility to adapt to specific conditions. Such systems typically combine long-term monitoring with more targeted monitoring of new or rare events.

Multi-functional systems, integrating strategic and operational aims, as well as information, early warning and monitoring functions thus seem a good approach. In sharp contrast to monitoring, currently, hardly any early warning systems on biotic risks are in place in the EU. However, several general information systems are available. Combining these three functions in a modular way tends to be most efficient and reliable. Such systems can be built around existing NFIs and ICP plots, so that the added costs need not be significant. Overall, there is a need to for increased communication, education and public awareness about natural forest dynamic and the effects of
different management systems on the delivery of forest goods and services. There is a need for both (1) trans-disciplinary knowledge production that use landscapes as sampling units (Roux et al. 2010); and (2) encouraging local collaborative learning processes involving researchers, practitioners and other stakeholders on the ground, and in education. Ideally, networks of case study landscapes should be developed, representing different forest use phases and governance arrangements, and different collaborative learning approaches to develop locally adapted SFM solutions.
Chapter 1: Introduction

The health and sustainability of Europe’s forests* are essential in solving the challenges posed by climate change, biodiversity protection and fresh water supplies. Forest vitality is also crucial for fostering a green economy in Europe and across the globe”

FOREST EUROPE, discussions for the launch of a legally-binding instrument

1.1 Aim of this report

The aim of this report is to provide a scientific state of the knowledge regarding forest biotic agents* in the 27 Member States (MS) of the European Union (EU). In particular, the report assesses whether the trends in biotic disturbances are increasing, and whether these trends are linked to climate and forest management. It aims to evaluate the role of forest types and desired forest functions on biotic hazard. Together, this should help provide guidance as to whether an increased diversity of forest management approaches help to create forests that are more resilient to natural disturbances, climate change and alien species* subject to biosecurity.

1.2 Managing forest ecosystems

The earth has entered an era of rapid environmental changes, challenging the traditional paradigms of forestry* and ecological sustainability. Climate variability, both naturally caused and anthropogenic, as well as modern land-use practices and stressors, create novel environmental conditions never before experienced by ecosystems. Maintaining healthy forests in the face of existing and impending changes is necessary to avoid important loss of economic, ecological, social and cultural functions.

Accepting that the future will be different from both the past and the present forces a paradigm shift, to manage forests in new ways. Coping with uncertainty and change requires adaptive capacity and resilience of ecosystems (e.g., Kohm and Franklin 1997, Hunter 1999). Thus, it is essential to view forests not only as areas with trees, but also as ecosystems. This reality is growing amongst policy-makers and managers. The member countries of FOREST EUROPE, which include all EU-27 Member States (MS), have agreed on a joint definition of sustainable forest management* (SFM) that embodies this shift: “the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic

[1] FOREST EUROPE, discussions for the launch of a legally-binding instrument

[2] *All words annotated with an asterix are defined in the glossary.
and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems.” This is echoed by the recently published State of Europe’s Forests report (MCPFE 2011) which identifies four future challenges for SFM:

- to find and deliver the optimum balance among the various forest functions by mitigation and adaptation in the context of changing climate and societal needs.
- to mobilise more wood to meet the targets for bioenergy production, but also to combine this with the other dimensions of SFM.
- to reconcile measures for biodiversity conservation with the more intensive forest management likely to be necessary to meet the expected higher demands for wood, including for renewable energy.
- to achieve the potential of sustainable production and consumption patterns, green building, green jobs in the forest sector, and the supply of renewable energy by strongly developing the “green” features of the sector.

The challenge of SFM is to reconcile multi-functional forests with wood production, biodiversity conservation, and greening of the forest sector.

This more dynamic view of forest management is gradually engendering a paradigm shift, whereby mimicking natural forest disturbance* regimes is becoming a science-based concept, and a societal recognised approach to secure forest functions. But while on the one hand, biotic and abiotic disturbances are inherently a natural element of forest ecosystems, on the other, disturbances can threaten the production of forest goods and services. The EU is thus subjected to conflicting opinions from different stakeholders about the need to take measures for adaptation and mitigation to tackle the growing native* and alien types of disturbances to EU forests caused by biotic agents.

1.3 State of EU forests

Towards multi-functional forests

Forests in the EU27 represent over one third of the land area and they are characterised by a wide variety of climatic, geographic, ecological and socio-economic conditions (Box 1-1). They are vital ecosystems, which have the potential to provide a wide variety of goods and services (i.e. including values) to society. They provide consumption goods in the form of wood, biomass, cork, fruits, berries and game, which have clear economic benefits. In terms of services, they help buffer weather events, regulate local and global climate, regulate the hydrological cycle, protect watersheds, and prevent erosion. In addition, they provide important social values, for recreation and by contributing to human well-being in general.

EU forests are amongst the most intensively managed forests in the world. Indeed, forestry is an important economic driver in the EU, employing about 4.3 million people for over 400 millions m³ of roundwood produced per year (MCPFE 2011). Five MS, Sweden, Finland, France, Spain and Germany, which possess about 60% of the EU forest area, are responsible for most of this wood
production (MCPFE 2011). In contrast, only about 5% of EU forests are undisturbed by anthropogenic activities, and the largest areas of old growth forests can be found in Bulgaria and Romania (Veen et al. 2010, MCPFE 2011). But over the past 20 years, EU forests have become increasingly multi-functional. Over 90% of EU forests offer public access and recreation facilities, while approximately one fifth of forest area offers protective functions. Similarly, one fifth of forest area is protected, mainly as a result of the establishment of the Natura 2000 network (MCPFE 2011).

Towards uniform or degraded forest ecosystems

Although forest area has been increasing in the EU over the past 20 years, there are reasons for concern. Forest cover is increasing partly as a result of afforestation, which often consists in planting alien tree species or conifers over native species. It is also increasing as a result of natural succession on abandoned land, giving rise to significant conversions from grasslands and other semi-natural areas (IEEP 2010). This is a cause of concern as grasslands have a high content of soil organic matter, which plays a crucial role in water, nutrient, and carbon cycles (European Environment Agency - EEA - 2010). Forests also tend to be increasingly fragmented, in particular along a gradient running from South-Western to North-Eastern Europe (Estreguil and Mouton 2009).

In addition, forests are in a poor conservation state overall. Most Natura 2000 forest habitats are in a bad to inadequate conservation status except in the Mediterranean and Mountain forests (ETC/BD 2008). The last old growth forests are threatened to be replaced by intensively managed semi-natural forests* or plantations* (EEA 1998). Finally, annual growth tends to exceed harvest, which results in very densely stocked forest stands in many countries, paralleled by a loss of biodiversity and disfunctioning of some ecosystem services.

Forest management choices are highly variable but leave a legacy on the landscapes. They depend importantly on the ownership structure, with public-owned forests more likely to apply sustainable forest management* aims in the long term. In the EU, over 65% of forests are privately owned (MCPFE 2011), but there is huge variability in ownership structure across MS. In some MS, such as Greece and Ireland, the State owns most of the forest stock, while in others, such as France, Germany, Spain, and Italy, local communities play a considerable role as forest owners. The forest management aims have a lasting effect on the ecosystem structure. Forests managed for wood production emerged at the turn of the century, and mostly led to the establishment of monocultures of conifers, managed as even-aged plantations. This form of management increased after the Second World War, thus many areas of Europe are characterised by relatively young, even-aged stands with few species (European Commission – EC- 2003). Afforestation often made space for fast-growing alien species, which are also managed as even-aged plantations. Thus, about a quarter of the EU forest area has only a single forest type (MCPFE 2011). In contrast, mountains and more isolated areas escaped some of these pressures and still harbour some relatively untouched forest areas and multi-functional woodlands.
Box 1-1 State of EU’s forests
(according to MCPFE (2011), and the forest types presented in Annex 1)

- **Northern Europe (DK, EE, FI, LI, LV, SE)**

North Europe is characterised by a high, stable forest cover. Forests are very important to the society and economy of North Europe and represent up to three quarters of the land area in Finland and Sweden. Their social importance is reflected in the large number of private owners in the North of Europe, where the number of private forest holdings often reaches 20-25% of the rural populations, as well as in the traditional free access to non-wood goods such as berries and mushrooms. The productive functions of forests play an important role, especially for wood, and lead to an intensive use of forests.

Most of the forest is **Boreal forest**. Forest damage is rather low, with a percentage of dead or damaged trees around or below 20%. Most MS have a high proportion of single tree stands, but this is more a reflection of the natural state than indicative of monocultures: most MS have less than 5% plantations. The management practices tend to mimic natural processes and involve natural regeneration.

- **Central-Western Europe (AT, BE, DE, FR, IE, LU, NL, UK)**

Forest cover has been expanding steadily over the last 20 years in most Central West European MS. However, the forest landscape is very fragmented in these often densely populated areas. Accordingly, forest issues are appear of relatively minor socio-economic importance, as reflected by the very low number of private forest owners (less than 2% of the rural population), with the exception of Austria (5%), Luxembourg (16%) and France (23%). In particular, in Ireland, Luxembourg, the Netherlands and the UK, the forest sector (including the forest industries) represents less than 0.7% of the national gross value added, while this share reaches 2% in Austria.

The dominant forest type in this part of Europe is **Atlantic forest**, but some MS like Austria, France and Germany also present large areas of **mountainous forests**. Plantations and **self-sown alien forests** represent 60% of the forest area in Belgium, the Netherlands, UK and Ireland, while it is under 15% in the other MS. Accordingly, the proportion of introduced species is over 40% in Belgium, Ireland and the UK, but much less in the other MS. Defoliation affects less than 35% of the trees in all the MS, in particularly in Ireland and Austria where it is below 15%. The proportion of single tree stands varies widely, ranging from 6 to 56% depending on the sylvicultural choices.

- **Central Eastern Europe (CZ, HU, PO, RO, SK)**

In Central Eastern Europe, the forest cover is relatively low, representing about 30% of the total land area, but slowly expanding, largely as a result of policy objectives. Hungary for example aims to reach 57% of forest cover by 2040. Despite this relatively low cover, the forest sector plays an important role in the economy of all these MS, except Hungary. Most of these MS have a long tradition of public forest ownership, and despite programmes to restore and privatise forests, the number of private owners remains low (about 5% in the Czech Republic and Poland).
Central Eastern Europe is characterised by two main types of forests, Continental forest composed of broadleaved-coniferous trees, and in a smaller proportion Mountainous forest. In Central-East Europe, forest health is an important concern: most of the MS have strong traditional forest sector institutions which monitor forest condition and damage as well as measures to compensate for the damage done. Forest damage is rather low, with approximately 20% of defoliation in most MS, except Slovakia and the Czech Republic where this share reaches up to 50%. Hungary, Poland and Romania report high levels of forest damage from biotic and abiotic causes.

**Southern Europe (BG, CY, ES, GR, IT, MT, PT, SL)**

Forest cover has been expanding in all Southern MS. In Italy for example, forest cover has increased from 31 to 37% over the last 20 years. The region is generally characterised by many small forest holdings and the forest sector is moderately important, except in Portugal. Indeed Portugal developed its forest sector by encouraging large forest plantations for wood production, making it one of the most important EU producers.

South Europe has a distinctive Mediterranean forest on much of its territory. All these MS are severely affected by forest fires. On average, about 0.5 percent of the forest area is burnt every year in Italy and Spain, but over 2% in Portugal. Forest damage by defoliation concerns less than 20% of the forest trees in Portugal and Spain, but is higher in the other MS, peaking at 36% in Italy. Many forests are single stand plantations, in particular in Portugal (72% of the forest area), but also in Bulgaria and Cyprus. However, except in Portugal, few introduced tree species are found. Management practices often involve natural regeneration. Moreover, Italy, Portugal and Spain have monitoring systems in place.

### 1.4 Disturbances in forest ecosystems

**Disturbances, a natural ecosystem process with significant socio-economic impacts**

Several biotic (insects, disease causing fungi, wildlife), abiotic (storms, fires, wind, snow), and human-induced (air pollution) threats affect EU forests. On the one hand, disturbances by biotic or abiotic sources are an essential component of natural ecosystems, because they foster processes such as regeneration, selection, adaptation, and evolution. On the other hand, alien biotic agents, often introduced by man, cannot be considered part of the natural dynamics of the system, and in fact often bring new functions, that can have cascading, not necessarily positive, downstream effects on the ecosystem (see Box 1-2).
Moreover, in managed forest ecosystems, damages caused by disturbances often result in socio-economic losses. Biotic agents form threats to economic interests in the form of rapid increase in insect, fungal and mammal populations that damage forest ecosystems and/or wood. But the maintenance of ecological and socio-cultural values may also require that biotic disturbances are maintained. Grazing wild and domestic herbivores that maintain the structure of forests and wooded other land is one such example (Vera 2000, Saltzman et al. 2011).

Disturbances are part of the natural dynamics of the forest ecosystems. Despite being part of the natural process, biotic agents can cause significant socio-economic losses, which may warrant management or mitigation measures. In particular, alien biotic agents that have not co-evolved with the ecosystem can threaten these natural dynamics, while certain management practices may favour biotic disturbances.

Box 1-2 Alien biotic agents as a phytosanitary issue

The introduction of alien species can result in a cascade of changes to forests, because local trees are not co-adapted to the invading agent, and because they are released from predator control, that would regulate their population. This applies both to introduced species and species expanding their range due to changing environmental conditions.

Introductions can be regulated at the level of the organism, through alert lists, or restrictions can be imposed on the vector or pathway, e.g. through quarantines. One goal of the EU 2020 biodiversity strategy is that “pathways are managed to prevent the introduction and establishment of new invasive alien species”, while priority species are controlled or eradicated.

Two main instruments are already in place regarding the management of alien agents introductions in EU forests.

IPPC - EPPO

At international level, the International Plant Protection Convention (IPPC) manages the development of International Standards for Phytosanitary Measures (ISPMs). The Standards are implemented by National Plant Protection Organisations (NPPOs), which are government agencies within member countries. Collaboration between the national organisations is achieved in Europe through the European Plant Protection Organisation (EPPO).

Founded in 1951 by 15 European countries, EPPO now ensures locally the coordination between 50 European countries including Russia, some countries in the Caucasus and Mediterranean countries. EPPO is primarily a forum for discussion and exchange. While EU legislation is the only binding framework for plant health measures, EPPO plays an advisory role, organises coordination between its Members and contributes to develop standards (non-binding guidelines). For example, EPPO assesses the risk of introduction and spread of potentially harmful organisms, and proposes adapted measures. These measures are then implemented by the NPPOs. The needs for phytosanitary measures are determined on the basis of a pest* risk analysis, during which each pest is assessed for:
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Disturbances of EU forests caused by biotic agents

Box 1-2: Alien biotic agents as a phytosanitary issue

- whether it is present in the exporting and importing countries;
- whether it is associated with the commodity or other pathway;
- whether the pest can enter, find suitable habitats, establish and spread in the importing country;
- whether, and to what extent, it will cause economic damage in the importing country.

According to the risk analysis, the EPPO maintains two lists which recommend that the organisms listed thereunder are regulated as quarantine pests*. Quarantine pests are defined as pests of potential economic importance to the area endangered thereby. A pest named on list A.1 is a quarantine pest that is not present in the EPPO region. Those on list A.2 are quarantine pests that are present in the EPPO region but not widely distributed there and are being officially controlled.

There are two further lists maintained by the EPPO: the Alert List and the Action List44. The Alert List includes pests that possibly present a risk to EPPO member countries (i.e. early warning). The Action List includes the A1 and A2 pests that are recommended for regulation but are not yet included in EPPO member countries’ phytosanitary regulations. Among the top30 biotic agents identified, six are listed on list A.1 and seven on list A.2.

Other measures include regulation of wood packaging material, guidelines for pest eradication, and integrated pest management, involving at least two independent phytosanitary measures that combine their effects. Early-warning actions, related to surveillance and pest reports are also requested.

EU Plant Health Regime

The EU Plant Health Regime was established by Council Directive 2000/29/EC. Its aim is to prevent organisms that are harmful to plants or plant products from being introduced into the Community and spreading within it. The Regime is based on the provisions of the International Plant Protection Convention and of the World Trade Organisation Agreement on Sanitary and Phytosanitary Measures.

Harmful organisms are listed in the annexes of the Directive and are defined as pests of plants or of plant products, which belong to the animal or plant kingdoms, or which are viruses, mycoplasmas or other pathogens. Annex 1 is divided in to two parts. Part A lists harmful organisms whose introduction into and spread within, all Member States shall be banned. Part B lists harmful organisms whose introduction into, and whose spread within, certain protected zones shall be banned. Annex 2 is similarly divided into two parts. Part A lists harmful organisms whose introduction into, and spread within, all Member States, shall be banned if they are present on certain plants or plant products. Part B lists harmful organisms whose introduction into, and whose spread within, certain protected zones shall be banned if they are present on certain plants or plant products.
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Disturbances, in particular biotic ones, are often considered to have been increasing over recent decades, causing a growing risk of damage to EU forests (e.g. Jactel et al. 2009). Climate change and invasive alien species (see Box 1-2) are the two main causes of concern. Forests are particularly sensitive to climate change, because the long life-span of trees does not allow for rapid adaptation to environmental changes. But climate change may not necessarily affect all forests negatively. It has the potential to initiate multiple, interacting processes, some of which affect forests positively (e.g. increased growth rates) and others negatively (e.g. increased water stress). However, extreme weather events (such as storms or droughts) induced by climate change may provide favourable conditions for biotic attack by “secondary” biotic agents and trigger further forest disturbances. Hence, it is in fact particularly difficult to predict the net impact of climate change on forests, and these depend strongly on the regional context and on the management regime.

Similarly, global trade and demand for wood have been growing exponentially over the past 50 years, giving rise to a constantly increasing risk of introducing alien biotic agents. This risk extends beyond the political borders of the country where the agent was introduced, to the geographical region of suitable distribution for the species. But again, it is very difficult to predict which species will establish and cause problems to forests, especially as significant time lags may occur between the introduction and the spread of the alien species, during which it remains innocuous.

**Evaluating the risks of biotic disturbances in forests**

Predicting the risk that forests will be damaged by a disturbance is a complex issue, which depends on the particular forest value (economic, ecological or social) under consideration. A biotic agent that has negative effects on one type of forest value may have positive effect on another. For example, grazing disturbance may help maintain the structure of forests and other wooded land, and thus its ecological and socio-cultural values (Vera 2000, Saltzmann et al. 2011). The risks concerning a particular forest value can theoretically be described as the interaction
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Disturbances of EU forests caused by biotic agents (Kron 2002). Any change in one of these factors will lead to a corresponding change in risk level. In addition, the effects of different forest factors, such as forest types, management traditions, and values, as well as global influences such as climate change and globalised trade need to be considered to cope with biotic disturbances (Figure 1-1). Forest management practices to reach given management goals impact stand composition, structure, and vitality, and consequently affect biotic hazard and the susceptibility of forests to damaging agents. These practices may also include those that aim at enhancing other goods such as non-wood goods, ecosystem services, and values.

Figure 1-1 Relationship between the three components of risk and the management for a particular forest good or service (modified after Jactel et al. 2009)

The different components of biotic risk are difficult to gauge. Biotic hazard is much more difficult to characterise than abiotic hazard. Biotic hazard is mainly influenced by the characteristics of the biotic agent. But the sheer diversity of agents of biotic disturbances, comprising pathogens, fungi, insect and mammals translates in a diversity of life histories that make generalisations difficult. Many insects are favoured by warm and dry conditions (Speight and Wainhouse 1989), but fungi favour more moist conditions (Coakley et al. 1999). Low winter temperatures may reduce insect survival rates (Leather et al. 1993), but may on the other hand be favourable for the synchronisation of life cycles with host plant species (Buse and Good 1996). Overall, biotic hazard depends on the dispersion capacity of the agent, it survival and reproduction conditions, and on biotic interactions (whether negative control by natural enemies, or positive control by mutualism). Clearly thus, biotic hazard will differ between native and alien species, which have not evolved with the native communities and may bring new functions to the forest ecosystem.

Susceptibility, or the vulnerability to biotic risk, depends on the interaction between the biotic agent and forest and other environmental factors. Thus it depends on the host specificity of the agent, regarding tree species, stand structure or habitat characteristics. Trees themselves react differently to damaging agents. For example, broadleaves can react immediately to defoliation by forming new leaves and shoots, whereas defoliation in conifers can remain visible for several
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years. In general, trees exposed to stress, such as drought, wind, extreme temperatures, or that are under a high management intensity are more vulnerable to biotic agents. Under such circumstances, many secondary agents are able to cause severe damage. Different regions, under different climatic conditions, also exhibit different levels or risk. Overall, this vulnerability is what characterises the number, frequency, duration, and spatial extent of the outbreaks.

Finally, exposure to biotic risk depends on the combined climatic and forest changes. The latter are heavily influenced by management aims, which are increasingly multi-functional. As an example, exposure to disturbance is low in forest reserves aiming at the protection of natural processes, in contrary to production forests, where damage results in high costs. The spread of alien biotic agents may be favoured in these new conditions, under climate-matching and predator release hypotheses.

1.5 How to manage biotic risks

Because European regions differ both with respect to forest types and the profiles of natural disturbance agents, as well as in their profile of desired wood and non-wood goods, ecosystem services and landscape values, and power of stakeholders, it would be difficult to develop common guidelines concerning disturbances for the entire EU. Rather, a framework for assessing the risks and prioritising the actions can be recommended (see Box 1-3).

Adaptability of forest systems to change and thus exposure to climate change differ according to bioclimatic zones and forest types, as well as various management regimes. The occurrence of damaging agents and the likelihood of damage in combination with the adaptive capacity of forests in each region account for the vulnerability of forests to climate change. Both forests and biotic agents are sensitive to key factors of climate (change), which allows for an evaluation of potential risks of damage. The intrinsic adaptation potential of a forest system and the planned potential, initiated through management measures, constitute its adaptive capacity.

- Controlling the risks of biotic damage to EU forests

Early warning, frequent monitoring and joint information systems could help prevent and control biotic risk, probably in combination with regulations (e.g. Box 1-2). Periodic assessment is essential to obtain information about the condition of natural resources in space and time, and to assess its relationship with biotic and abiotic factors (Ferretti et al. 1997, 2004). Efficient early warning systems are needed to track the emergence of new biotic agents early enough to be able to control the risk of biotic damage. Although in the EU, data on tree crown condition are collected through ICP forest, there are major concerns about the quality of this data, in terms of reliability, comparability and precision (both regarding the coverage of biotic damages and sampling frequency).
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Box 1-3 Example of a “decision tree” to determine the extent to which a particular biotic agents causes damage to forest goods and services

Question 1. Are trees affected by abiotic disturbance* in the forest I am managing?
- no (=no action)
- yes – single trees: no action because of marginal consequences
- yes – entire stands are affected and may threaten key forest products and functions (2)

Question 2. What are the main products and functions of my forest?
- wood of suitable quality for use as a material (3)
- biomass (4)
- maintain natural dynamic (5)
- protective functions (water, soil, transport infrastructure, settlements, etc.) (6)

Countermeasures:
- 3. Salvage logging
- 4. Adapt by shorter rotations to avoid damage
- 5. Free development
- 6. Short-term (6a) or long-term (6b)
  - Short term (6a): mitigation by removal of affected tress)
  - Long term (6b): adaptation by moving forest management toward more natural dynamic (and trees species composition and structure)

1.6 Structure of this report

In the context of biotic disturbances and contemporary SFM policy, this report contains six chapters, including this first chapter:

- Chapter 1 introduces the aims and logic of this report in a context of sustainable forest management and review the importance and state of EU forests and biotic risk, raising the question as to whether or not natural forest dynamics are being modified by climate change, with increased risks of forest damage.

- Chapter 2 provides (a) a panorama of the current risks of biotic agents on the goods and services forests provide, and (b) review the impacts and trends of selected natural native biotic disturbance agents on forests and woodlands

- Chapter 3 introduces forest ecosystem management as a way to securing resilient ecosystems. This includes overviews of (a) the natural diversity of forest dynamic as a benchmark for composition, structure and function of managed
forests, (b) the systems for forest and other wooded land* management in the EU, and (c) how to improve the match between forest disturbance regimes and forest management systems

- Chapter 4 discusses the future impact of climate change and forestry practices on the risks caused by biotic agents and their relative importance
- Chapter 5 assesses the current state of play and needs for information, monitoring, early warning systems
- Chapter 6 provides conclusions concerning the need for (a) syntheses of knowledge concerning long-term (30+ yrs) changes in biotic disturbance, (b) adaptive management, and (c) adaptive governance to deal with biotic disturbances.

![Figure 1-2 Overview of the different chapters of this report.](image-url)
Chapter 2: Panorama of the current risks caused by biotic agents to the delivery of goods and services

2.1 Key points

- The forest area disturbed annually in the EU27 by insects, fungi, pathogens and mammals reached around seven million hectares in 2005 (6% of the total forest area).

- Outbreaks of native biotic agents can be considered to form part of the natural forest succession dynamics. In contrast, alien biotic agents, which are usually spread by human activity, often cause more severe damage than native biotic agents. This is because the forest ecosystem has not co-evolved defences against them, and in its new environment, the agent is no longer restrained by its natural enemies, or other by physiological or ecological constraints. Therefore, movements of plant material in international trade that could carry invasive alien pests and diseases are regulated worldwide.

- Trends in biotic damage of forests are difficult to establish as no comparable data exists, but indicate a tendency towards increased damage of outbreaks and growing establishment of alien species.

- Biotic agent outbreaks can be either localised or widespread, and rare or recurring. This depend mainly on the life cycle of the agent (in particular, the number of generations per year and its host specificity), its dispersal ability (capacity to move long distances), and the condition and vulnerability of the host (presence of pre-damages).

- Some abiotic factors or pre-damages caused by biotic factors (e.g. previous infestation by a primary pest) can facilitate biotic outbreaks.

- Climatic factors are very important drivers of forest biotic agent outbreaks. Local climate influences host susceptibility to pest and has direct effects on the prevalence of hazards: high temperature and prolonged drought induce physiological stress in trees and at the same time are in general favourable to the development of insect biotic agents (and sometimes to fungi); cool and wet conditions are often beneficial to fungal pathogens. More frequent strong winds increase the risk of damage.

- In a context of climatic change, new pest-mediated diseases may appear either because of a change in tree species composition or because of changes in the geographical range of pests. Moreover, pathogens may encounter new hosts and/or new potential vectors.

- Evaluating the potential risk of damage is very important for forest management. Lessons can be learnt from species that affect wide forest areas (Ips typographus, Ophiostoma novo-ulmi). These can help face emerging risks at the borders or spread in the EU (Phytophthora ramorum, Anoplophora glabripennis, Cryphonectria parasitica) or to limit the risk of spread of localised pests (Bursaphelenchus xylophilus, Thaumetopoea pityocampa).
2.2 Trends in damage to EU forests caused by biotic agents

2.2.1 Biotic damages to EU forests

2.2.1.1 Overview of biotic damages to EU forests
Several biotic (insects, disease causing fungi, wildlife), abiotic (storms, fires, wind, snow) and human-induced (air pollution) threats affect EU forests. In 2005, 6.4% of the total EU forest area was damaged by disturbances (6.99 Mha). This corresponds to about 22 million m³ of growing stock. Forests can be affected by more than one damaging agent at the same time, but biotic causes represent the most important source of damage. Insects and diseases are responsible for 44% of this damage, and wildlife grazing for another third of this damage. The first abiotic cause, storms were responsible for 26% of the damage. However, there is important regional variation in the type and intensity of damages. Portugal, Romania and Italy suffered from particularly severe losses due to insects and diseases, while Sweden was the most exposed to wildlife grazing damages (MCPFE 2011). Compared with Scandinavian MS, fewer problems are noticed with bark beetles in the UK, but instead a number of significant problems with defoliating insects (Day and Leather 1997). Insects seem to have the largest impacts in Atlantic temperate forests and Mediterranean forests (16.3% and 13.4%), but a comparatively small impact in boreal and mountainous forests. In contrast, fungi and wildlife damages are largest in boreal forests (8.2% and 1.2% respectively) (Jactel et al. 2007). In general, biotic hazard seems to be smaller in more extreme climates: boreal and mountain forest biomes were less affected by insect and pathogen damages (Jactel et al. 2007).

Variations in vulnerability to biotic risk according to tree species are also observed. Damages tend to be more prevalent in broadleaved than in coniferous species. Broadleaved tree always showed high levels of damage by pest insects (ca. 20% of the trees) and fungal pathogens (ca 5% of trees), except in boreal forests (Jactel et al. 2007). This can be explained by the fact that broadleaves harbour a higher species diversity (in particular of phytophagous insects) than coniferous (Kelly and Southwood 1999, Brandle and Brandl 2001), and that they last longer and grow more slowly (longer exposure period, especially in conditions of ‘stress’)

Depending on the extent of the disturbances, and the forest management aims, the resulting damages can cause substantial economic, environmental and social impacts. Forest pests can have significant effects on tree growth, mortality, and reproduction, thereby influencing the uses of forest resources for other functions (timber, recreation, wildlife, etc.). They may also impact endangered species or ecosystems.

2.2.1.2 Economic effects
Forests provide multiple services, ranging from the production of timber, cork or berries, to the regulation of climate, the protection of watersheds or landscapes, while also being recreational grounds. The importance of a given service depends both on the forest type, on the forest health
Panorama of the current risks caused by biotic agents to the delivery of goods and services

and on its management aims. For instance, European timber production is mostly concentrated in five MS: Germany (4.5 m$^3$/ha of forest available for wood supply/year), France (4.1 m$^3$/ha of forest available for wood supply/year), Poland (4.1 m$^3$/ha of forest available for wood supply/year), Sweden (3.6 m$^3$/ha of forest available for wood supply/year) and Finland (2.3 m$^3$/ha of forest available for wood supply/year) (MCPFE 2011). In contrast, the distribution of the cork oak (*Quercus suber*) restricts cork production to Mediterranean forests, while Mountain forests are traditionally managed to promote watershed and landscape protection, to prevent natural disasters such as landslides, avalanches or flooding. Outbreaks of biotic agents may affect forest dynamics in a way that modifies or impairs the balance between the different services.

Biotic damages may have very different economic implications, depending on the modes of action of the biotic agent, and on forest management aims. Clearly, production forests will be strongly affected by timber damage, whereas in conservation forests, changes in community structure, and ensuing changes in the ecosystem processes will outweigh timber losses.

Although many forest services* do not have clear markets, it seems that overall services provided by forest ecosystems have a greater value than forest products coming from them (EEA 2009). For instance, wood products only represent 35% of the total economic value of Mediterranean forests, whereas watershed protection, recreation, and other non-use values add up to half of this value (Merlo and Croitoru 2005; Figure 2-1). When accounting for all those services, the loss of one hectare of Mediterranean forest is estimated to amount to about 133 €/ha/yr. In a mountain forest in Romania, this cost has been estimated at 317€ per year for each hectare of forest, half of which for water flow regulation, with all timber and non-timber forest goods responsible for 40% ³. These costs are likely to be underestimates; other estimates may reach up to 4100 €/ha/yr (Holmes et al. 2006, Mavsar and Riera 2007).

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In forests managed for commercial timber production, large scale outbreaks can cause considerable economic losses to forest owners. Biotic damages may result in the deterioration of tree condition, not only in the year of occurrence, but also in later years. Some biotic attacks simply lead to a general loss of vitality and tree growth, but may occur chronically, while others cause the rapid decline and death of the trees, such as the Dutch elm disease caused by the fungic agent *Ophiostoma novo-ulmi*. Thus biotic agents may reduce both wood quantity (through felling, or slower increment growth) and quality (by damaging the wood by making holes, tunnels, or stains; or by causing irregular growth).

In EU forests, losses due to the genus *Heterobasidion* have been variously estimated at between €500 million to over €700 million per annum, sums including timber losses, killing and costs of control, remediation and diagnosis (Woodward et al. 1998, Asiegbu et al. 2005). In the area of the Peninsula of Setubal, south of Lisbon, heavy defoliation of young pine stands by *Thaumetopoea pityocampa* caused a 20% timber loss of the growing stock in the first 20 years after stand establishment, resulting in a timber market value reduction of 12% for technical reasons (Gatto et al. 2009). In the case of *Neodiprion sertifer*, literature data point to only low tree dieback (about 4%) after outbreaks, amounting to an estimated total economic value loss of 28 €/ha after one year of outbreak (Lyytikäinen-Saarenmaa and Tomppo 2002).

The extent of damage by the most damaging wood boring insects was estimated in the Bark and Wood Boring Insects in Living Trees (BAWBILT) project over a 10-year period (Table 2-1). *Ips typographus* and *Tomicus piniperda* together are responsible for an average of approximately 4.5 Mm³/yr of timber damage, which amounts to approximately 1% of the total fellings in Europe (Gregoire and Evans 2004).

**Table 2-1 Damage caused by the top ten wood-borers in Europe (Grégoire and Evans 2004)**

<table>
<thead>
<tr>
<th>Species</th>
<th>Threatened area (ha)</th>
<th>Time period</th>
<th>Volume (m³)</th>
<th>Affected area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Ips typographus</em></td>
<td>7 640 156</td>
<td>1990 – 2001</td>
<td>31 642 535</td>
<td>2 819 055</td>
</tr>
<tr>
<td><em>Hylobius abietis</em></td>
<td>3 418 264</td>
<td>1990 – 2000</td>
<td>88 258</td>
<td></td>
</tr>
<tr>
<td><em>Pityogenes chalcographus</em></td>
<td>8 784 001</td>
<td>1990 – 2000</td>
<td>7 827 506</td>
<td>595 400</td>
</tr>
<tr>
<td><em>Scolytus multistriatus</em> and <em>Scolytus scolytus</em></td>
<td>224 100</td>
<td>1990 – 2000</td>
<td>10 711</td>
<td>524</td>
</tr>
<tr>
<td><em>Tomicus piniperda</em></td>
<td>14 507 433</td>
<td>1990 – 2001</td>
<td>12 933 205</td>
<td>202 235</td>
</tr>
<tr>
<td><em>Phaenops cyanea</em></td>
<td>8 010 001</td>
<td>1990 – 1999</td>
<td>12 839 159</td>
<td></td>
</tr>
<tr>
<td><em>Rhyacionia buoliana</em></td>
<td>327 000</td>
<td>1990 – 2001</td>
<td>20 138</td>
<td></td>
</tr>
</tbody>
</table>

Abiotic disturbances, such as storms and droughts, may increase the risks of outbreaks. Estimations from Sweden suggest that although important, the biotic damages to forests are small in relation to the total value of forestry (Table 2-2).
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Table 2-2 Estimated annual costs of biotic damages to Swedish forest in relation to a severe wind storm event in 2005⁴ (Witzell 2009)

<table>
<thead>
<tr>
<th>Type</th>
<th>Estimated value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forestry in Sweden</td>
<td>Amounts to 2-3% of the Swedish GDP, which is about 300 billion €, i.e. about 6-9 billion €</td>
</tr>
<tr>
<td>The storm “Gudrun” (January 8-9, 2005) in SW Sweden</td>
<td>2.1 million €</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Species</th>
<th>Estimated value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heterobasidion spp.</td>
<td>1.61-3.22 €/ha/yr</td>
</tr>
<tr>
<td>Hylobius abietis</td>
<td>&gt; 0.32 €/ha/yr (which would be 5-10 times higher without protective measures)</td>
</tr>
<tr>
<td>Alces alces</td>
<td>&gt; 0.32 €/ha/yr</td>
</tr>
</tbody>
</table>

When the trees are also a source of fruits, such as chestnut trees, economic losses can also be very significant: in the case of Dryocosmus kuriphilus, the productivity yield reductions range from 50 to 70%.

Invasive biotic agents are often more strongly regulated and costs include not only the lost value from the tree but also control costs, such as the costs of felling infected trees in order to avoid the spread of the infection. In Sweden, the cost to forestry of the Dutch Elm Disease was estimated to range between 0.3-0.6 €/ha/yr (Gren et al. 2009), while in Germany the total annual costs of invasive alien species to forestry have been estimated 0.6 €/ha/yr (Reinhardt et al. 2003). A much higher figure was estimated in the UK, where the total costs of damages caused by mammals, insects and pathogens in forests was recently estimated at 37.5 €/ha/yr (CABI 2010, Table 2-3). In particular, the red band needle blight caused by the fungus Dothistroma septosporum is infecting 70% of the UK Corsican pine stands. Forty-four percent of these are estimated to have a 30% yield loss, resulting in an estimated cost for loss of tree value of 0.27 €/ha/yr. Grey squirrels (Sciurus carolinensis) cause damage to the timber industry through bark stripping (Mayle et al. 2004), which can lead to either a decrease in the quality of timber, or death of the tree. Based on several studies (Mayle 2002, Anon. 2006), the total yield loss was estimated at 0.24 €/ha/yr, for a total economic cost including control costs of 2.19 €/ha/yr (CABI 2010).

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⁴ The estimated values were initially in €/yr and have been converted in €/ha/yr using the Swedish forest area given in Eurostat (2009).
Table 2-3 Annual cost of damage by alien insects and pathogens (estimation given in Euro/ha/year)\textsuperscript{5}

<table>
<thead>
<tr>
<th>Pathogen</th>
<th>England</th>
<th>Scotland</th>
<th>Wales</th>
<th>GB</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Dendroctonus micans</em></td>
<td>0.07</td>
<td>0.01</td>
<td>0.28</td>
<td>0.06</td>
</tr>
<tr>
<td><em>Elatobium abietinum</em></td>
<td>0.46</td>
<td>2.09</td>
<td>1.63</td>
<td>1.34</td>
</tr>
<tr>
<td><em>Phytophthora ssp.</em></td>
<td>0.43</td>
<td>0.37</td>
<td>0.23</td>
<td></td>
</tr>
<tr>
<td><em>Dothistroma septosporum</em></td>
<td>0.60</td>
<td>0.01</td>
<td>0.57</td>
<td>0.28</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>1.56</td>
<td>2.11</td>
<td>2.85</td>
<td>1.91</td>
</tr>
</tbody>
</table>

2.2.1.3 **ENVIRONMENTAL EFFECTS**

The transformation of the original ecosystems by biotic agents can lead to important losses in terms of ecosystem processes and biodiversity value.

- **Impacts on ecosystems and ecosystem processes**

  The widespread mortality of a dominant or co-dominant species constitutes a severe disturbance to forest ecosystems. High mortality of a widespread foundation species, i.e. a single species that defines much of the structure of a community, has much greater ecological effect than the same mortality rate for a less common species (Lovett et al. 2006). But a less common species may have a unique role, providing ecosystem services that are not provided by other species in the ecosystem (Parker et al. 1999). For example, periodic outbreaks of native biotic agents, such as *Zeiraphera improbana*, are known to serve as important disturbances affecting forest nutrient cycling and other ecosystem processes (Lovett et al. 2002). The absence of mass outbreaks of *Z. improbana* since the 1980s is an unprecedented change in this disturbance regime. It can be expected that nutrient cycling and other ecosystem processes operating in the subalpine larch forests of the Alps may be undergoing drastic alterations.

  Long-term impacts of biotic outbreaks can lead to shifts in tree species composition, with subsequent changes in ecosystem processes. As an example, stands of mountain birch along the Scandinavian mountain chain, which commonly recover from periodic infestation by the autumnal moth, *Epirrita autumnata*, suffer at times from dieback at the tree-line, with an eventual conversion of forest into tundra (Haukioja et al. in Berryman 1988, Virtanen and Neuvonen 1998). Similarly, in the boreal forest, long-horned beetles attacks on poplar species, particularly *P. tremuloides*, may result in accelerated succession of mixed stands to conifer stands.

- **Indirect impacts on biotic communities**

  The trophic structure of the ecosystem can change when one or more foundation species are lost, with indirect effects for a large number of species. The decline or disappearance of a tree species can result in the loss of a major food source or habitat. For instance, many mammal and bird

\textsuperscript{5} The estimated values were initially in €/yr and have been converted in €/ha/yr using the GB, England, Scotland and Wales area given in UK forestry statistic report (2011) at (2009).
species depend on the high quality mast\(^6\) produced by pines, spruce or chestnut trees. These effects then have further cascading consequences on the associated predators, parasites, and pathogens, eventually changing the entire community structure. For example, the Dutch elm disease has eliminated elm trees throughout much of Europe (see Section 2.3.7). Healthy elm trees play an important ecological role, as their prolific seed production is a food source for many species of birds and small mammals, the branches provide nesting sites, while deer and rabbits browse on twigs and buds (Waldron 2003). Elm is also considered a soil improving species, as elm litter is rich and decomposes rapidly, thereby supporting an active community of soil organisms. As elm, trees die under the attacks of the pathogen, the landscape of the ecosystem changes. The tree size and age structure is modified, leaving place for increased shrub density in the canopy gaps, and decreased nesting sites for birds (Castello et al. 1995).

In turn, the changes in community structure can lead to modified landscapes processes, by affecting the degree of connectivity between different habitats, and thus the likelihood of spread of other biotic agents (Holdenrieder et al. 2004).

- **Genetic diversity**

When a biotic agent eliminates populations of a host species, gene flow is disrupted. This results in a loss of genetic diversity, with long-term evolutionary effects, such as a reduced capacity of the host trees to respond to environmental changes (Holdenrieder et al. 2004). This may also lead to shifts in forest species composition, that can have ramifying effects through the ecosystem in many ways, because trees have different, often unique properties in terms of nutrient and water uptake, growth rate (thus light), litter quality, soil organic matter, habitats and food quality for animals. Often however, the biotic agent may simply change the local abundance of host species, which may lead to changes in the stand age structure composition, such as evening-out of the age structure, and thus limited genetic impacts.

- **Alien biotic agents**

Alien biotic agents may perform a new function and may lead to significant cascading effects on the ecosystems. For instance, an introduced pathogen may predispose its hosts to biotic attack (Houston 2005) or cause changes to the ecosystem by modifying the habitat, altering the trophic structure and availability or quality of nutrients and physical resources (space, water, light), or by changing disturbance frequency and/or intensity (Loo et al. 2009).

- **Indirect effects of measures to control biotic agents**

The management practices used to control or eradicate an epidemic may also have significant environmental impacts. A number of fungicides or other pesticides are known to have negative side-effects on other species, and affect the nutrients balance in the soil, and therefore soil communities (Turbé et al. 2010). Moreover, when eradication measures involve clear-cutting in large areas, this creates a meta-population pattern, with subsequent indirect effects on the biotic communities. Trees play a strong role for maintaining soil structure through their root system. Clear-cuts have been shown to drastically reduce soil decomposer communities, as well as

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\(^6\) Mast is the edible vegetative or reproductive part produced by trees and shrubs that wildlife species and some domestic animals consume. For example, oak and beech trees produce hard mast in the form of acorns and beechnuts. Other tree and shrub species produce a soft mast, such as leaf buds or berries.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

communities of soil engineers (Baath 1980, Lundqvist 1983). Biological soil processes are thus impaired (even if some deadwood is left on site), which can lead to soil erosion and degradation. This may seriously impair the future regeneration potential of the area (Turbé et al. 2010).

2.2.1.4 ♦ SOCIAL EFFECTS

Across Europe, the social functions that forests can provide are increasingly valued. These include recreation, but also mental health, and quality of life. Forests are often one of the main types of natural landscapes with close-to-home public access for urban dwellers.

► Recreation and wider rural development

European forests offer generous recreational opportunities that are often threatened by biotic outbreaks. In the mid 1990s, approximately 1.4 billion visitors are estimated to have visited Western temperate European forests (Coford 2009). This is equivalent to an average of 6.5 visits per person per year. In Ireland, the benefits from the recreational use of forests have been estimated €97 million or €5.40 per person (for an annual 18 million visits). The total economic activity generated by domestic forest users is estimated at €268 million. Walking tourism, generally by overseas visitors, accounts for €138 million per year (Coford 2009). Mediterranean forests, which are located in a biodiversity and touristic hotspot, showcase a high number of native species (Merlo and Croitoru 2005) and forest recreation generates estimated benefits ranging from 2.5 – 11 € per visit (Bellu and Cistulli 1997). In Mountain forests, tourism is often the largest source of income. For example in Scotland, the Cairngorms National Park receives around 1.4 million visitors a year, each spending on average GBP 69 (EUR 80) a day on accommodation, food, transport and entertainment. Indeed, the recreational activities developed in forests also contribute to support the livelihoods in the entire region, by creating employment and indirect activities to support the touristic demand.

Biotic outbreaks may affect the aesthetics of the forest, resulting in a loss of recreational value. Aesthetics may be spoilt by a range of factors, including the characteristic silky nests of Thaumetopoea pityocampa in Mediterranean pine forests, but also through the sheer destruction of trees or the management practices used to try and control the outbreaks (e.g. clear-cutting). However, the social implications of these losses, in terms of recreation potential or rural livelihood, are seldom quantified. However, in Europe, people seem to prefer spending time in forests that have a moderate management intensity, which ensures a ‘tidy’ look, while providing safety and better accessibility (Sievänen et al. 2008). They also show a clear preference across Europe for more mature forests, with mixed stands (Edwards et al. 2010). Thus dense, even-aged monocultures of conifers, which also tend to be at greater risk of biotic damages, are not favoured for recreation.

► Protection of settlements

The very touristic pressure of forests located close to urban centres or in attractive locations may have a backlash effect that may favour biotic outbreaks, and endanger settlements. For example, urban forests are often the first site for biotic invasions, since they are located close to dense population centres that are often the unintentional vectors of biotic agents. This is illustrated by the case of Anoplophora glabripennis, which was introduced in an urban park in Italy, and for which outbreaks are so far limited to Rome’s region and Lombardy (see Case study in section 2.3.2).
related way, winter sports, such as skiing, create large forest openings that on the one hand increase the risk and damages caused by avalanches (Wipf et al. 2005), and on the other hand, increase the amount of edge habitat, and thus the vulnerability of trees to abiotic disturbances.

### Human health

The hair of some caterpillars may be urticating, or cause allergies when they penetrate the human respiratory system. For example, processionary caterpillars, such as *Thaumetopoea* spp. have caused frequent outbreaks of dermatitis, ocular lesions and allergic reactions (Diaz 2005, Vega et al. 1999). Contact with dead larvae, cocoons, nests and debris from infested pine forests can cause dermatitis throughout the year. In France, media-campaigns during outbreaks advise the public to stay away from affected areas. Similarly, during the severe outbreak of the gypsy moth, *Lymantria dispar*, in Bulgaria in 1996–1997 it was necessary to close off some forest recreation areas to prevent people from coming into contact with the insects. To avoid the loss of recreational value of forests from a high incidence of caterpillars, localised areas are sometimes treated with chemical or biological products, but these in turn may cause problems through spray drift and contamination of ground water as well as through the possible loss of certain non-target species.

### Non wood forest products

Forests have a strong cultural value associated to them. In some MS, recreational activities such as seasonal collection of berries, mushrooms and herbs for personal use, have always been an integral part of rural life. Non-wood forest products (NWFP), such as, cork, fodder, mushrooms, fruits, pharmaceutical and aromatic plants, can contribute significantly to the local or national economies. For example, cork in Portugal accounts for 35% of the estimated total benefits obtained from forests (Merlo and Croitoru 2005).

### Hunting

Some traditional uses of forests may help control some biotic damages. Moose and deer populations (notably *Cervus elaphus* and *Capreolus capreolus*) are causing significant damage to forests across Europe, through browsing and bark stripping. Regulated hunting can help control those populations and avoid or reduce the damages. Moose hunting in Finland represents 32 million €/yr (Finnish Forest Research 2000).

#### 2.2.2 Types of biotic agents

The top-30 biotic agents in EU forests are presented in Table 2-4. Given the multi-dimensionality of the damages caused, it is difficult to define robust criteria for determining importance at EU level. This list was based on expert opinion, policy concerns, and scientific interest, and should thus be considered as indicative. Table 2-4 presents the distribution of the most important biotic agents in the EU, along with the main biotic and environmental characteristics that affect their impact: i.e. the mode of actions of the biotic agent and its host specificity. Further, native biotic agents, which have co-evolved with the forests ecosystems in which they are found, but which may intermittently reach outbreak conditions, cause a different type of risk to forests than alien species. The latter may not necessarily cause any damage in their native range, but once introduced in Europe, they may be released from their biotic controls, or find conditions that promote their growth. This means they have the potential to cause much greater, faster damage.
than native species. However, some important factors, such as host tree characteristics (e.g. vitality, dominance, uniqueness in the ecosystem, influence of management practices) and climatic factors (Lovett et al. 2006) are too variable to be included in this table.

Given the wide range of factors that play a role in the occurrence and severity of biotic outbreaks, the simplest way to classify biotic agents is based on their mode of action. The modes of action of biotic agents are exemplified by the prevalent symptoms: defoliation, wood boring, stem or root damages. Most of the damages caused by insects are defoliation (40%) and bark beetles/wood-boring (37%), whereas fungal pathogens mainly caused damage in the roots (20%) and stem (56%) (Jactel and Vodde 2006). The tactics employed to manage biotic agents are typically tailored to these modes of action (Evans and Speight 2004). The review thus distinguishes between native and alien biotic agents, and is structured according to taxon and mode of action, as this provides the easiest entry for management purposes. Table 2-4 describes the most important biotic agents at EU level according to these characteristics, specifying as well their distribution and host specificity.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Table 2-4 Top-30 biotic agents in EU forests classified by category (from the most to the less damaging)

<table>
<thead>
<tr>
<th>Category</th>
<th>Scientific and common name</th>
<th>Taxon</th>
<th>Native/Alien</th>
<th>Type of disturbance</th>
<th>Distribution in EU27</th>
<th>Trees impacted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Top 1-5</td>
<td><em>Ophiostoma novo-ulmi</em> (Dutch elm disease)</td>
<td>Fungus</td>
<td>Alien (evolution)</td>
<td>Disease</td>
<td>AT, BE, BG, CZ, DK, EE, FR, DE, EL, HU, IT, LT, NL, PL, ES, RO, SE, SK, UK</td>
<td>Elm trees (<em>U. minor, glabra, laevis, americana, rubra, thomasi</em>, and <em>crassifolia</em>)</td>
</tr>
<tr>
<td>Top 1-5</td>
<td><em>Bursaphelenchus xylophilus</em> (pine wood nematode)</td>
<td>Nematode</td>
<td>Alien</td>
<td>Disease</td>
<td>PT and 2 outbreaks in ES under eradication</td>
<td>Pine trees (<em>P. pinaster, silverstris and nigra</em>)</td>
</tr>
<tr>
<td>Top 1-5</td>
<td><em>Ips typographus</em> (European spruce bark beetle)</td>
<td>Insect</td>
<td>Native</td>
<td>Wood boring</td>
<td>AT, BE, BG, CZ, DK, EE, FI, FR, DE, EL, HU, IT, LV, LT, LU, NL, PL, RO, SK, SL, SE, UK</td>
<td>Mainly Norway Spruce trees (<em>Picea abies</em>) but sometimes pine (<em>Pinus sp.</em>) and larch trees (<em>Larix sp.</em>)</td>
</tr>
<tr>
<td>Top 1-5</td>
<td><em>Neodiprion sertifer</em> (European pine sawfly)</td>
<td>Insect</td>
<td>Native</td>
<td>Defoliation</td>
<td>Boreal forests (SE, FI)</td>
<td>Scots pine trees: <em>Pinus sylvestris</em></td>
</tr>
<tr>
<td>Top 1-5</td>
<td><em>Tomicus piniperda</em> (common pine shoot beetle)</td>
<td>Insect</td>
<td>Native</td>
<td>Wood boring</td>
<td>AT, BE, BG, CZ, FI, FR, DE, EL, HU, IT, NL, PL, PT, RO, SE, UK</td>
<td>Scots pine trees (<em>Pinus sylvestris</em>) and occasionally spruce trees (<em>Abies sp.</em>) and larch trees (<em>Larix sp.</em>)</td>
</tr>
<tr>
<td>Top 6-10</td>
<td><em>Chalara fraxinea</em> (ash dieback)</td>
<td>Fungus</td>
<td>Unknown</td>
<td>Disease</td>
<td>AT, CZ, FI, FR, DE, HU, IT, LI, NL, PL, SL, SE and observed on the basis of the symptoms: DK, EE and LV</td>
<td>Ash trees (<em>Fraxinus excelsior</em> and <em>F. angustifolia</em>)</td>
</tr>
<tr>
<td>Top 6-10</td>
<td><em>Heterobasidion annosum</em></td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>AT, DE, EE, FR, IE, FI, LI, LV, PL, SE, UK</td>
<td>Norway spruce and fir trees (<em>Picea abies</em> and <em>Abies alba</em>)</td>
</tr>
</tbody>
</table>
| Top 6-10 | *Phytophthora ramorum* (sudden oak death) | Fungus | Alien | Defoliation | Found only in nursery in: BE, DK, FR, DE, IE, NL, PL, SL, ES, SE, UK In the wild: DE, DK, IE, LU, NL and UK In the wild now eradicated BE and SL | Beech family trees (*Lithocarpus densiflorus, Quercus agrifolia, parvula* and *kelloggi*) and shrub species (*Rhododendron spp.* and *Viburnum sp.*). Bleeding canker due to *P. ramorum* reported on *Quercus rubra* (in NL) as well as *Quercus illex, acuta, falcata* and *cerris* (in UK). Bleeding canker due to *P. ramorum* reported on *Fagus sylvatica* (in ...
### Panorama of the current risks caused by biotic agents to the delivery of goods and services

Disturbances of EU forests caused by biotic agents

<table>
<thead>
<tr>
<th>Category</th>
<th>Scientific and common name</th>
<th>Taxon</th>
<th>Native/Alien</th>
<th>Type of disturbance</th>
<th>Distribution in EU27</th>
<th>Trees impacted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Top 6-10</td>
<td>Lymantria dispar (gypsy moth)</td>
<td>Insect</td>
<td>Native</td>
<td>Defoliation</td>
<td>AT, BG, CZ, DE, EL, FR, HU, IT, PL, PT, RO, SK, SL, ES</td>
<td>Deciduous trees notably oak trees (Quercus sp.)</td>
</tr>
<tr>
<td>Top 6-10</td>
<td>Large herbivores</td>
<td>Mammal</td>
<td>Native</td>
<td>Browsing</td>
<td>Found in all EU27</td>
<td>Deciduous trees notably oak trees (Quercus sp.), hornbeam trees (Carpinus sp.)</td>
</tr>
<tr>
<td>Top 11-15</td>
<td>Phytophthora cinnamomi</td>
<td>Fungus</td>
<td>Alien</td>
<td>Disease</td>
<td>FR, IT, PT, RO, SL, ES, UK</td>
<td>Cork oak trees (Quercus suber), but wide range of hosts like chestnut tress (Castanea sp.) and coniferous trees (Chamaecyparis lawsoniana, Juniperus conferta)</td>
</tr>
<tr>
<td>Top 11-15</td>
<td>Biscogniauxia mediterranea (charcoal disease)</td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>Mediterranean MS (ES, FR, IT, PT) and reported in SL</td>
<td>Oak trees (Quercus ssp.)</td>
</tr>
<tr>
<td>Top 11-15</td>
<td>Dothistroma septospora (red band needle blight)</td>
<td>Fungus</td>
<td>Alien</td>
<td>Disease</td>
<td>AT, BG, DE, ES, EL, FR, IT, PT, RO, UK</td>
<td>Pine trees (Pinus ssp.)</td>
</tr>
<tr>
<td>Top 11-15</td>
<td>Diploida pinea (diploida blight)</td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>From Southern Europe to the north (FR, BE, IT, UK), reported in EE</td>
<td>Pine trees (P. nigra, P. pinaster and P. sylvestris)</td>
</tr>
<tr>
<td>Top 11-15</td>
<td>Thaumetopoea pityocampa (pine processionary moth)</td>
<td>Insect</td>
<td>Native</td>
<td>Defoliation</td>
<td>Central and Southern Europe (AT, BG, CY, FR, EL, HU, IT, PO, ES)</td>
<td>Pine trees (P. nigra var. austriaca, P. sylvestris; P. pinaster, P. pinea, P. canariensis, P. halepensis)</td>
</tr>
<tr>
<td>Top 16-20</td>
<td>Cryphonectria parasitica (chestnut blight)</td>
<td>Fungus</td>
<td>Alien</td>
<td>Disease</td>
<td>AT, BE, FR, DE, EL, HU, IT, PL, PT, SL, SK, ES</td>
<td>Chestnut trees (Castanea dentata and C. mollissima)</td>
</tr>
<tr>
<td>Top 16-20</td>
<td>Ceratocystis fagacearum (oak wilt)</td>
<td>Fungus</td>
<td>Alien</td>
<td>Disease</td>
<td>Wide distribution in North America, not in Europe yet</td>
<td>Oak trees (Quercus sp.)</td>
</tr>
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### Panorama of the current risks caused by biotic agents to the delivery of goods and services

#### Disturbances of EU forests caused by biotic agents

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<tr>
<td>Top 16-20</td>
<td><em>Gremmeniella abietina</em> (brunchorstia disease)</td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>AT, BE, BG, CZ, DK, EE, FI, FR, DE, EL, IT, LI, NL, PL, RO, ES, SE, UK</td>
<td>Spruce and pine trees mainly (<em>Picea abies, P. contorta</em> and <em>Pinus sylvestris</em>)</td>
</tr>
<tr>
<td>Top 16-20</td>
<td><em>Anoplophora glabripennis</em> (Asian Longhorned Beetle)</td>
<td>Insect</td>
<td>Alien</td>
<td>Wood boring</td>
<td>AT, DE, FR</td>
<td>Deciduous trees (<em>Populus sp.</em>, <em>Salix sp.</em>, <em>Ulmus sp.</em> and <em>Acer sp.</em>)</td>
</tr>
<tr>
<td>Top 16-20</td>
<td><em>Anoplophora chinensis</em> (citrus longhorned beetle)</td>
<td>Insect</td>
<td>Alien</td>
<td>Wood boring</td>
<td>Infestation detected mainly in IT where it is spread (Rome, within Milan: mainly in West and North West of the city, and in 30 municipalities North West, West, and South of Milan). In isolation, <em>A. chinensis</em> was detected in FR (but declared eradicated since 2006), in DE in 2008 (but now eradicated) and in NL</td>
<td>Broadleaved trees and shrubs – major concern for Citrus spp.</td>
</tr>
<tr>
<td>Top 21-25</td>
<td><em>Erwinia nimipressuralis</em> and <em>amylovora</em> (fire blight)</td>
<td>Bacteria</td>
<td>Alien</td>
<td>Disease</td>
<td>AT, BE, BG, CY, CZ, DK, FR, DE, EL, HU, IE, IT, LU, NL, PL, RO, SK, ES, SE, UK</td>
<td>Rose family trees (<em>Crataegus sp.</em>), and chestnut trees (<em>Castanea sp.</em>)</td>
</tr>
<tr>
<td>Top 21-25</td>
<td><em>Fusarium circinatum</em> (pitch canker)</td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>ES, IT, FR, PT</td>
<td>Pine trees (<em>Pinus sp.</em>)</td>
</tr>
<tr>
<td>Top 21-25</td>
<td><em>Melampsora allii-populina</em> and other <em>Melampsora</em> (European rust)</td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>BE, ES, FR, PT</td>
<td>Poplar trees (<em>P. balsamifera, P. deltoides, P. nigra var. italic</em> and <em>P. tremuloides</em>)</td>
</tr>
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</tr>
</thead>
<tbody>
<tr>
<td>Top 21-25</td>
<td><em>Armillaria spp.</em> (armillaria root disease)</td>
<td>Fungus</td>
<td>Native</td>
<td>Disease</td>
<td>FR, HU, DE, BG, EL, ES, IT, UK, BE</td>
<td>Wide range of hosts (larch, spruce and pine trees mainly)</td>
</tr>
<tr>
<td>Top 21-25</td>
<td><em>Dryocosmus kuriphilus</em> (Oriental chestnut gall wasp)</td>
<td>Insect</td>
<td>Alien</td>
<td>Disease</td>
<td>IT, FR, SL</td>
<td>Chestnut trees especially (<em>Castanea sativa</em>)</td>
</tr>
<tr>
<td>Top 26-30</td>
<td><em>Phytophthora alni</em></td>
<td>Fungus</td>
<td>Alien</td>
<td>Disease</td>
<td>BE, DE, FR, UK</td>
<td>Alder trees (<em>A. glutinosa</em>)</td>
</tr>
<tr>
<td>Top 26-30</td>
<td><em>Odonta dorsalis</em> (locust leafminer)</td>
<td>Insect</td>
<td>Alien</td>
<td>Defoliation</td>
<td></td>
<td>Robinia trees (<em>Robinia sp.</em>)</td>
</tr>
<tr>
<td>Top 26-30</td>
<td><em>Ips sexdentatus</em> (six-toothed bark beetle)</td>
<td>Insect</td>
<td>Alien</td>
<td>Wood boring</td>
<td>AU, BG, CZ, FR, DE, GR, HU, IT, LI, PO, PT, RO, ES, SE, SO, CH, UK</td>
<td>Pine trees (<em>P. sylvestris</em>, <em>P. pinaster</em>, <em>P. helreichii</em> and <em>P. nigra</em>)</td>
</tr>
<tr>
<td>Top 26-30</td>
<td><em>Phoracantha semipunctata</em> (eucalyptus) (Lonhorned borer)</td>
<td>Insect</td>
<td>Alien</td>
<td>Wood boring</td>
<td>FR, IT, NL, PR, ES</td>
<td>Eucalyptus trees (<em>Eucalyptus sp.</em>)</td>
</tr>
<tr>
<td>Top 26-30</td>
<td><em>Rhynchophorus ferrugineum</em> (Red palm weevil)</td>
<td>Insect</td>
<td>Alien</td>
<td>Wood boring</td>
<td>CY, FR, ES, EL, IT, MT, PT, SL</td>
<td>Palm trees (<em>Phoenix dactylifera</em> and <em>P. canariensis</em>)</td>
</tr>
</tbody>
</table>
2.2.2.1 Native Biotic Agents

Fungi/pathogens, nematodes

Fungi and Oomycota are the best known groups of pathogens affecting trees and forest ecosystems worldwide. Many examples of ‘native’ pathogens in these groups are known in European forests and for convenience may be, classified into those attacking root systems (fine roots or secondarily thickened roots), main stems and branches, twigs and shoots, foliage, and fruit/seed. Most of the damages are observed from tree root and stem pathogens (EFORWOOD, 2007).

Root pathogens

Pathogens attacking fine roots are almost all in the genera *Fusarium* (including the closely related *Cylindrocarpon*), *Rhizoctonia*, *Pythium* and *Phytophthora*. The latter two genera are not fungi, but are more closely related to marine brown algae and are placed in the Oomycota. It is difficult to prove if the range of different species found in all of these genera is native to Europe, as they may have been inadvertently introduced along with translocations of plants (with soil) over a very long period. Within each genus, however, there are species that can cause major losses to productivity in nursery situations, with *Fusarium* and *Rhizoctonia* species tending to be active in sub-optimal soil moisture conditions, whereas *Pythium* and *Phytophthora* cause damage in the presence of excess water (Phillips and Burdekin 1992, Butin 1995). There are few reports of *Fusarium* or *Rhizoctonia* causing damage to trees outside the nursery situation, however, but it must be noted that these fungi do occur, seemingly in species complexes, in the rhizosphere of forest trees (Chavarriaga et al. 2007). However, *Gibberella circinata* (teleomorph of *Fusarium circinatum*) is known to have caused damage in various locations around the world, including in Spain (Iturritxa et al. 2011).

Current research is increasing awareness of the range of *Pythium* and *Phytophthora* species present in European forest ecosystems. Although the genus *Phytophthora* has classically been considered to include plant pathogens of major importance, it is becoming clear that the genus also includes species that process dead or decayed organic matter. The species of *Phytophthora* recognised as causing major plant disease outbreaks, such as *P. cinnamomi*, *P. cambivora*, *P. alni* and *P. ramorum*, are certainly introduced into Europe (or in the case of *P. alni*, hybridised in Europe, Brasier 2008). Recent work, however, has determined that a range of *Phytophthora* species probably occur naturally in European forest ecosystems.

Fungal pathogens attacking secondarily thickened roots of trees are amongst the most destructive agents known in forestry (including urban forestry). The major genera with serious impacts in this category in European forestry are *Heterobasidion* and *Armillaria* (Woodward et al. 1998, Kile and Shaw 1991, Fox 2000), although species such as *Rhizina undulata* and *Inonotus tomentosus* are of local importance. Three species of *Heterobasidion* are present in Europe, and cause serious damage: *H. annosum*, *H. parviporum* and *H. abietinum*, of which *H. annosum* is arguably the most aggressive species (Woodward et al. 1998, Asiegbu et al. 2005). Amongst the seven species of *Armillaria* known in Europe, only two, *A. mellea* and *A. ostoyae* (syn. *A solidipes*), are serious pathogens, the remaining species displaying more saprotrophic life strategies.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

- **Stems and branches pathogens**

  Stems (and main branches) of trees may be affected by heartwood decay (heartrot) or various other dysfunctions, including wilt diseases and cankers. A wide range of fungi, most of which are in the Basidiomycota class Hymenomycetes, may cause stem decay in living trees (Phillips and Burdekin 1992, Butin 1995, Redfern and Gregory 1998). Decay can be divided (roughly) into brown rot, where the fungi selectively degrade the carbohydrate polymers in the cell walls, leaving modified lignin behind, and white rot, in which the fungi produce peroxidase enzymes that remove lignin residues (phenolic compounds) from the cell walls, in addition to other enzymes degrading the cellulose and xylans/arabinogalactans. Brown rot fungi affecting standing trees include *Phaeolus schweinitzii*, *Sparassis crispa* and *Laetiporus sulphureus*. The most common genera associated with white rot are *Heterobasidion* and *Armillaria* described above, along with *Phellinus*, *Inonotus*, *Ganoderma*, *Fomes*, *Fomitopsis*, *Polyporus*, *Pleurotus* and *Stereum*. Members of the genus *Stereum* may also cause pipe rot in dysfunctional sapwood. As with other disease types, the losses to stem decays are very difficult to quantify as the decay is often not reported in any detail during felling, whether in stands managed for timber production or in amenity situations.

  The main native wilt disease in Europe is *Verticillium* wilt (*V. albo-atrum*, *V. dahliae*), which is a problem on a wide range of angiosperm trees, particularly in the genus *Acer* (Butin 1995). It can also cause damage to ash (*Fraxinus*), sweet chestnut (*Castanea sativa*), lime (*Tilia* spp.) and rowan (*Sorbus* spp.), and many ornamental trees. Other species that cause wilt disease are *Ceratocystis fimbriata* sp. *platani*, *Ceratocystis (Ophiostoma) ulmi* and *C. (O.) novo-ulmi*.

- **Twigs and shoots pathogens**

  A number of prominent blights and cankers affecting twigs and young shoots of forest trees are present endemically in Europe. *Diplodia pinea* causes cankerling and shoot dieback (Sphaeropsis canker) on a range of gymnosperms, principally in the *Pinaceae*, the impact of this disease appears to have increased in Mediterranean, Central and Eastern European MS in recent years (Deprez-Loustau et al. 2006).

  Nectria/Neonectria cankers are common on a wide range of host trees, principally angiosperm species, including ash (*Fraxinus* spp.), beech (*Fagus sylvatica*), apple and crab (*Malus* spp.), pear (*Pyrus* spp.), hawthorn (*Crataegus* spp.), poplar (*Populus* spp.) and willow (*Salix*). Other Nectria species are also implicated in various tree diseases, including *N. faginata*, in combination with damage by the scale insect *Cryptococcus fagisuga*, causing beech bark disease and *N. cinnabarina* causing dieback of trees of low vigour. A number of further Nectria species attack conifers, notably *N. fuckeliana* which causes a bark canker of spruce (Butin 1995). *Cryphonectria parasitica* caused local extinction of American chestnut (*Castanea dentata*) in the US in the early 1900s (Redfern and Gregory 1998).

  Other significant canker diseases of gymnosperm trees include *Gremmeniella abietina* (syn. *Ascocalyx abietina, Brunchorstia pinea, Scleroderris abietina*) which causes shoot dieback and cankers on twigs and main stems of a range of Pine family trees and *Lachnellula willkommii* which prevents the planting of European larch (*Larix decidua*) in parts of Europe where the
environmental conditions are conducive to disease development (Butin 1995, Redfern and Gregory 1998).

In parts of Europe, the rust (Uredinales) pathogen *Cronartium flaccidum* causes stem canker ing on two-needle pines, such as *Pinus sylvestris*, *P. pinea* and *P. pinaster*. Within the macrocyclic population of *C. flaccidum*, which requires unrelated alternate host plants, a microcyclic species, *Endocronartium pini* (syn. *Peridermium pini*) occurs (Hantula et al. 2002). *E. pini* does not require alternate hosts, completing its life cycle on two-needle pines. Major outbreaks of these diseases occur from time to time, although the conditions leading to these outbreaks are unknown.

### Foliage pathogens

Foliage diseases include needle casts, mildews, leaf spots, anthracnose and rusts. Needle casts of gymnosperm trees are common, and often cause particular problems in forest nursery situations, resulting in the need to apply fungicides. Examples include *Lophodermium seditiosum* on pine (*Pinus* spp), *Meria laricis* on larch (*Larix* spp.) and *Didymascella thujina* on Western Red Cedar (*Thuja plicata*; Redfern and Gregory 1998). The impacts of these needle casts in forests and plantations vary, although both *Lophodermium* and *Meria* may be quite prominent, given suitable environmental conditions for development. *D. thujina* is considered a problem only on juvenile plants of *T. plicata*.

Many broadleaved trees are susceptible to powdery mildew infections. The most common type of oak mildew, caused by *Erysiphe alphitoides* (Mougou et al. 2008) is described under alien biotic agents, below. Several maple species are highly susceptible to mildew pathogens in the genus *Sawadea* (syn. *Uncinula*), which is not a major problem in terms of forest production, but can result in considerable disfigurement to ornamental varieties in amenity situations (Butin 1995).

Leaf spot diseases are very common, particularly later in the growing season (Phillips and Burdekin 1992, Butin 1995, Redfern and Gregory 1998). Many of these diseases result in what appear to be quite alarming symptoms, but the trees may be relatively little affected, due to late season onset. Leaf spot of horse chestnut, caused by *Guignardia aesculi*, has increased in severity over the last 20-15 years, although the reasons for this change are unknown (Pastirčáková et al. 2009). Striking symptoms are caused on leaves of a number of maple species by tar spot pathogens of the genus *Rhytisma*, although effects on growth appear to be minimal. Anthracnose diseases are also common on foliage of beech (*Fagus*), oak (*Quercus*) and plane (*Platanus*). Moreover, many *Populus* species are subject to regular defoliation following infection by the Marssonina leaf spot pathogen (*Drepanopeziza punctiformis*).

Foliage (and young shoot) rusts in the genus *Melampsora*, for example, are one of the main factors negatively impacting on the use of poplar and willow for biomass production (Royale and Hubbes 1992). Other broadleaved and gymnosperm trees are attacked by leaf and needle rusts in the nursery too: examples include *Melampsoridium betulinum* attacking European birch species, and (at least) three species of *Colesporium* that cause needle rust of pines (Phillips and Burdekin 1992, Butin 1995, Redfern and Gregory 1998). Many rusts are regulated as quarantine pests, such as *M. medusae* and *M. farlowii*, due to the risk that they present to EU forests.
Nematodes

Direct damage can be due to pathogens feeding on plant tissue. This is the case of nematode pathogens including both native and invasive species. Interestingly, a study by Braasch et al. (1998) has revealed severe pathogenicity to pine of two wood nematodes naturally present in Europe (Bursaphelenchus mucronatus, Bursaphelenchus sexdentati). The extent of this phenomenon in field conditions is still under study (Daub 2008). In addition to direct damage, nematodes pathogens can carry dangerous viruses. This is the case of some species of parasitic nematode (e.g. Xiphinema spp. and Longidorus spp.).

Insects

Insects are the main cause of biotic damage to European forests (EFORWOOD 2007). Most of these damages are caused by defoliators and wood-boring species and they can lead to important economic losses. In France for example, insects have been clearly identified as responsible for half of the damaged trees. However, only a small proportion of all forest insects cause serious disturbances: almost two thirds of the damages are caused by only 13 insect species.

Defoliators

The damages caused by insects that feed on leaves can vary substantially, ranging from insects that cause minor defoliation or aesthetic injury to insects that cause extensive tree mortality. The severity of the damage depends on the specificity of feeding, relative to the growth cycle of the tree (Evans and Speight 2004). For instance, Neodiprion sertifer is one of the most serious insect pests in Northern Europe, and repeatedly causes mass outbreaks also in Central and Eastern Europe (Hellrigl and Salvadori 1998). It attacks older foliage on young Scots pines, predominantly leading to significant loss of tree volume growth, up to 38% (Lyytikäinen-Saarenmaa and Tomppo 2002). However, because it does not attack the current growth, trees normally survive repeated episodes of defoliation. Similarly, Elatobium abietinum nearly never kills its host trees, but may cause intense needle and thus increment loss, reducing productivity and thus economical value of Sitka spruce stands for years (Straw et al. 2005, Westgarth-Smith et al. 2007). In contrast, Panolis flammea attacks both the current and older foliage of lodgepole pines in the UK, and can lead to extensive tree mortality (Evans and Speight 2004).

Shoot feeders

Apart, from defoliation, the browning of leaves due to insect larval feeding (e.g. of the alien species Odonta Dorsalis) can also reduce aesthetic and consequently economic values of forests, even when most trees survive pest infestation (Johnson and Lyon 1991).

Tree damage is commonly caused by insects in the larval stage feeding on tree tissue. Rhyacionia ssp. larvae, for instance, are at the origin of three different types of damage occurring on the shoots, depending on the larval stage. Newly hatched larvae kill individual needles around buds; intermediate larvae kill buds in preparation for overwintering; and mature larvae, which cause the most important damage, kill developing buds and elongating shoots (Pointing and Green 1962).
Wood and bark borers

Wood and bark borers damage trees mainly in the larval stage by tunnelling. Some species bore deeply into the bark and larvae mine the sapwood or occasionally the heartwood. Bark borers tunnel just under the bark of trees. The insects develop from egg to adult stage in the tree, but once emerged, they do not usually cause any further damage. The group of wood and bark boring insects includes a variety of beetles and moths, as well as wood wasps.

Bark beetles impair water and nutrient supply of trees by adults and larvae establishing galleries in the phloem of their hosts. Similar pathways of damage are shared among these species. The liberation of volatile pheromones molecules permit the aggregation of male adult individuals which start to prepare a nuptial chamber where mating takes place. The female then burrows out oviposition galleries where larval hatch and start to feed in additional perpendicular galleries. As the main and most aggressive native bark beetle species, the European Spruce Bark Beetle, Ips typographus, together with the small spruce bark beetle, Pityogenes chalcographus, cause dieback of Norway spruce stands in both managed and protected areas, with serious economic, social and ecologic consequences. In contrast, Monochamus sp., which is a secondary biotic agent whose larvae bore into the wood, primarily does not lead to tree mortality, but subsequently causes timber degradation. In Europe, the long-horned beetle is mainly considered dangerous because it is a potential vector for the pine wood nematode, Bursaphelenchus xylophilus, which causes pine wilt.

Mammals

Both wild and domestic mammals cause biotic disturbances in forests. In the boreal forest, moose (Alces alces) have regionally dense populations and cause damage to young Scots pine forests. The economic effect is regionally large. Moose and deer can also restrict the recruitment of adult individuals of deciduous trees such as aspen (Populus tremula) and rowan (Sorbus aucuparia) in boreal, temperate, and mountain forest (Angelstam et al. 2000, Motta 2003). Deer can cause a great deal of damage to forests, under certain conditions, often related to forest management practice, affecting vegetation, from the plant organ to the landscape scale (Weisberg and Bugmann 2003) with cascade mechanisms impacting bird and arthropod communities (Allombert et al. 2005, Barrett and Stiling 2007, Gill and Fuller 2007), ground flora (Kirby 2001), and trees and shrubs (Gill and Beardall 2001).

For example, red deers (Cervus elaphus) are natural components of forest ecosystems in Northern Europe, and cause sporadic browsing and fraying damage in the native forests. Due to extensive deforestation in the UK up to the early 20th Century, red deer became adapted to more open conditions. One consequence was that Cervus elaphus became a big problem on natural regeneration sites, and in reafforestation (Gill 1992). Increasing deer populations (all species) leads to greater amounts of browsing damage. Thus, tree damage caused by deer on plants through feeding is considered as a driver of plant community dynamics of the forest understory (Rooney and Waller 2003) because it affects vegetation composition and structure (Gill and Beardall 2001, Casabon and Pothier 2008, Rooney 2009, Boulanger et al. 2009). Ungulate browsing can then strongly influence the structure, composition, growth and succession of forest stands as a result of the intensity and the selectivity of disturbance via both direct and indirect effects.
Invasive alien mammals species can also displace native species and carry potentially dangerous viruses (Tompkins et al. 2002). This is the case of the grey squirrel, which was introduced into Britain at the end of the 19th Century (Shorten 1959) and into Italy in 1948 (Bertolino et al. 2000). *Sciurus carolinensis*, the grey squirrel, is the major alien damaging agent in broadleaved woodlands. Populations are now out of control in Britain, and spread of the Italian population threatens contiguous countries in the near future (Currado 1998). As a result, this has led to damage to forests and plantations by bark-stripping that degrade wood quality and can facilitate insect and fungi damages.

2.2.2.2 **ALIEN BIOTIC AGENTS**

Alien biotic agents are almost exclusively introduced unintentionally into Europe, inside timber or plant-derived packaging (such as crates, pallets), or inside passengers luggage (Work et al. 2005). Among the exceptions are fungi or insects introduced for biological control. With increasing trade, the number of introductions has grown exponentially with time. However, a characteristic of biological invasions is that only a few biotic agents become serious pests (Williamson and Fitter 1996, DAISIE 2009).

From the ensemble of nonindigenous forest biotic agents that have become established in Europe, a limited number are causing severe economic, environmental and social losses. However, when they do become pests, alien biotic agents are often more likely to cause long-term changes than other disturbances (Loo et al. 2009). Indeed, they tend to bring new functions to the ecosystem, which even if they appear minor may have cascading effects. For example, even if the alien biotic agent does not apparently cause significant tree mortality or damage alone, it may predispose its host tree to attacks by other biotic agents (that may not otherwise have been able to create damage). An additional issue is that once established, alien biotic agents remain a permanent component of the ecosystem.

### Alien fungi/pathogens and nematodes

The number of alien tree pathogens that has become established in Europe is difficult to gauge, as many pathogens are either innocuous or insidious and, in general, no effort is expended to document invasions by microorganisms unless they have measurable impacts (Desprez Loustau 2009). Moreover, the species concept itself is not easily handled in fungi and other microorganisms, and the poor knowledge of their biogeography can make it difficult to determine whether a species is alien. Many fungal species previously defined on the grounds of morphology (or of symptoms on host plant for pathogens) have been shown to be a complex of several cryptic species differing in their ecology, and especially in their geographic range (Pringle et al. 2005). New approaches, including phylogeography and molecular analyses, result both in a surge of newly recorded alien fungi, and in a clarification of the native range of the species. More than 50% of the fungal species identified in Delivering Alien Invasive Species Inventories for Europe (DAISIE 2009), not all of which are found in forests, were described after 1950, and almost 20% only in the 2000s.

Regarding alien species, a classic case providing substantial and useful insight on the invasive potential of fungal pests is the wilt Dutch elm disease, caused early in the 20th Century in Europe by *Ophiostoma ulmi* and from the mid-1960s by the far more aggressive species *Ophiostoma*
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Disturbances of EU forests caused by biotic agents

The first epidemic, caused by *O. ulmi* killed approximately one third (10 – 40%) of infected trees, but by the late 1940s had subsided. In the mid-1960s, however, the ‘new’ species, *O. novo-ulmi*, arrived at ports in the UK on shipments of elm timber from North America. Within 30 years, the vast majority of elms in the UK, some 30 million trees, were dead, and the disease continues to spread northwards. At the same time, a second strain of *O. novo-ulmi* was spreading westwards in Europe, from an origin somewhere in Eurasia, resulting the deaths of most elm trees in Europe as a whole. It is interesting to note that the invasion by *O. novo-ulmi* apparently is leading to the likely extinction of the pathogen causing the original, less aggressive outbreak of Dutch elm disease, *O. ulmi*.

In the 19th Century, timber production in Northern European forestry had been enhanced by the introduction of North American species of 5-needle pines, particularly *Pinus strobus*. After a number of years, however, widespread dieback occurred in these rapidly growing plantations, due to the inadvertent introduction of the stem rust disease pathogen, *Cronartium ribicola*, into the Baltic States from its origin in Wallachia. Within a few years, the disease had prevented the use of the highly susceptible North American 5-needled pine in timber production in Europe (Geils et al. 2010).

Oak mildew, caused by *Erysiphe alphitoides*, is a prominent and common alien invasive pathogen affecting most species of *Quercus* grown in Europe (Mougou et al. 2008). Although it was originally thought that the pathogen was native to North America, and introduced into Europe near the end of the 19th Century, recent evidence suggests that there may be several species within this pathogen complex, with an origin in the tropics (Mougou et al. 2008).

Further highly relevant examples of alien invasive pathogens and the ability of these on-native organisms to cause extensive damage to forest ecosystems are provided by pathogens in the Oomycete genus *Phytophthora*, in which there are many species that have been spread by international trade. *P. cinnamomi*, a particularly virulent pathogen with a recorded host range of over 950 plant species, is possibly the most widespread such invasive organism (Hardham 2005). Invasions of alien areas by *P. cinnamomi* have resulted in ecological devastation, as illustrated by severe eucalypt diebacks in the forests of Western Australian and Victoria. *P. cinnamomi* is also widespread in Europe in chestnut forests in association with *P. cambivora* and in evergreen oak forests characterised by a warm, dry climate (Spain and Portugal). *P. lateralis* emerged in Oregon in the early-mid 20th Century, probably entering on infected material imported from the far east and totally altered forest productivity in that region, seriously reducing the amounts of *Chamaecyparis lawsoniana* timber available to local and international forest industries (Hansen et al. 2000); this pathogen species has also been reported in France (Hansen and Delatour 1999) and very recently in Scotland, where it is also killing *C. lawsoniana* in ornamental plantings and hedgerows. Other examples in Europe include *P. quercina*, frequently a primary factor in the recently emerging problems of oak decline (Jung et al. 1999), the novel heteroploid hybrid *P. alni* (and sub-species thereof) killing riparian alders in northern Europe (Brasier et al. 2004) and the notorious *P. ramorum*, first characterised on ornamental *Viburnum* species (Werres et al. 2001), but particularly noted for the widespread disease it causes on a range of tree species, for example ‘Sudden Oak Death’ in California and Oregon (Rizzo et al. 2005). Recent phylogenetic analyses suggest *P. ramorum* is alien in both Europe and North America (Ivors et al. 2006). *P. ramorum* is commonly intercepted by plant quarantine authorities in European countries, is reported in
forest ecosystems in Europe, especially on important forest species of the Fagaceae and Ericaceae, and is considered a serious threat to European forests. Since 2009, P. ramorum has been killing Japanese larch (Larix kaempferi) in Ireland, Northern Ireland and the western areas of the British mainland (Webber et al. 2010); this larch is killed rapidly by the pathogen, leading to wide-scale felling operations in an attempt to prevent further spread (see Box 2-1).

More recently, another aerial Phytophthora capable of killing Fagaceae and Ericaceae, P. kernoviae was described in the UK and, later, New Zealand. A number of invasive Phytophthora species, including P. citricola, P. cactorum, P. cambivora and P. pseudosyringae are associated with serious beech decline in Europe (Jung et al. 2005). Other newly recognised Phytophthora threats are known, P. kernoviae being one example (Brasier et al. 2005), and many hitherto unrecognised species are ‘emerging’. P. alni (and similarly the presence of the North American ‘type’ of H. annosum in Italy and of different strains/sub-species of Ophiostoma novo-ulmi across Europe) highlights a serious additional risk associated with biological invasion: hybridisation events between species and subspecies of pathogens, both amongst alien pathogen species and between alien and native pathogens (Brasier 2008).

Other important groups of alien pathogen threats targeted in this project include Ceratocystis spp., Mycosphaerella spp., rust pathogens and various bacterial pathogens (Table 1). The genus Ceratocystis, as currently defined, includes a range of cryptic species with very similar morphologies (Wingfield et al. 1997, Harrington and Wingfield 1998, Johnson et al. 2005, van Wyk et al. 2006, Marin et al. 2005, Wingfield et al. 2006), many of which pose considerable threats to a wide range of European plant species. Recorded hosts include tropical crop plants such as sweet potato, cocoa, Colocasia, Ficus, mango, coffee, and temperate trees: Populus, Quercus, Platanus, Prunus and Carya, as well as conifers in the Pinaceae (Picea, Pinus and Larix) (Kile 1993, Harrington and Wingfield 1998). The pathogen attacking Platanus, which has recently become invasive in the native range of Platanus orientalis in southern Europe (Greece), was recently redefined, based on molecular data, and named Ceratocystis platani (Baker Engelbrecht et al. 2005). C. platani is killing large numbers of the highly susceptible host species, Platanus orientalis, in Greece, and may already be present in northern parts of Turkey.

The genus Mycosphaerella spp. is extremely large, including some 10,000 species, many of which are serious pathogens on a range of agricultural and horticultural crops, and on trees. M. pini (Dothistroma septosporum), cause of red band needle blight of pine, for example, has become particularly problematic in an extended range within Europe over the last 15 years.

Major threats are now causing concern for the future of members of the genus Pinus in Europe. Long known in North America, pitch canker, caused by Gibberella (Fusarium circinatum), is found in southern Spain and in Italy, leading to enormous fears that the disease could become widespread in European forestry causing important economic damages. In addition, Bursaphelenchus xylophilus was found in Portugal in the early 1990s (Mota et al. 1999), and two outbreaks recently occurred in Spain, where it is under eradication. Again, this organism is considered a major threat to the long-term survival of many pine species in Europe. It is vectored by European native wood dwelling long-horned beetles of the genus Monochamus, and is known to cause a devastating pine wilt disease. It can develop on various species of conifers, mainly pines (Pinus spp.), and in suitable environmental conditions it may kill even mature trees within only a few weeks. An additional recently emerged problem is Chalara fraxinii (Hymenoscyphus
Panorama of the current risks caused by biotic agents to the delivery of goods and services

*pseudoalbidus*), a recently emerged pathogen, is causing a serious decline and dieback disease of common ash (*Fraxinus excelsior*) in eastern, central and northern Europe, *Fraxinus excelsior* (Kowalski 2006).

### Alien insects

Alien insects are commonly introduced accidentally in Europe with their host plants, as part of horticultural trade (Niemla and Mattson 1996). Polyphagous insects seem particularly successful, with 109 species of phytophagous insects having established populations in European forests (Mattson et al. 2007), and 14 out of 18 of the invasive bark beetles (Kirkendall and Faccoli, 2010). The success of polyphagous insects may be explained by the fact that they are more generalist, and can adapt to a diversity of hosts, although the majority live upon their introduced, native host plants (Mattson et al. 2007). Four orders account for about 84% of the immigrants: Homoptera 39%, Lepidoptera 13%, Coleoptera 19%, and Hymenoptera 13% (Table 2-5). Among forest bark beetle invaders, inbreeding species (11 out of 18) are also over-represented compared to their prevalence in nature (Kirkendall and Faccoli 2010), which may help them circumvent many of the issues linked to inbreeding depression faced by small populations in the establishment phase.

Alien insects may cause more severe damages than native species in outbreak conditions. This is partly due to the fact that native European trees, which have not been able to co-evolve defences against them. For instance, *Bursaphelenchus xylophilis* does little damage in North America, where it is native, but causes severe damage in Asia and in Europe, if allowed to spread.

### Table 2-5 Successful establishment by phytophagous insects in Europe (Mattson et al. 2007)

<table>
<thead>
<tr>
<th>Insect order</th>
<th>From NA to E</th>
<th>From A to E</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Species count</td>
<td>Percent</td>
</tr>
<tr>
<td>Coleoptera</td>
<td>9</td>
<td>15.8</td>
</tr>
<tr>
<td>Diptera</td>
<td>8</td>
<td>14.0</td>
</tr>
<tr>
<td>Hemiptera</td>
<td>4</td>
<td>7.0</td>
</tr>
<tr>
<td>Homoptera</td>
<td>22</td>
<td>38.6</td>
</tr>
<tr>
<td>Hymenoptera</td>
<td>8</td>
<td>14.0</td>
</tr>
<tr>
<td>Lepidoptera</td>
<td>5</td>
<td>8.8</td>
</tr>
<tr>
<td>Thysanoptera</td>
<td>1</td>
<td>1.8</td>
</tr>
<tr>
<td>Total number of species</td>
<td>57</td>
<td></td>
</tr>
</tbody>
</table>

NA, North-America; E, Europe; A, Asia. Species count = number of species in insect order, percent = (Species count/Total number of species)*100

Alien insects often follow their host trees though, and are a cause of damage in alien tree plantations (see Box 2-1). *Phoracantha semipunctata* can cause serious damage to *Eucalyptus spp.*, as larval feeding beneath the bark can spread around the entire circumference of a tree, girdling, or completely removing a strip of bark from, the trunk. Infested trees usually die within a few weeks, although resprouting can occur from the tree base. Recently, huge impacts on palm
trees have been caused by the alien *Rhynchophorus ferrugineum* larvae, which can excavate holes up to a meter long for several weeks in the trunk of palm trees, thereby weakening and eventually killing the host plant. The main symptoms of damage are usually only visible long after the palm has become infested.

**Box 2-1 Biotic damages on alien trees**

In many MS, alien tree species, especially fast growing ones such as Eucalyptus, have been introduced for plantations. They may be more vulnerable to biotic threats, as they have not co-evolved with European pests and pathogens. Moreover, a number of alien biotic agents are accidentally introduced with alien trees, causing them severe damage in the introduced environment.

► **Japanese larch tree disease**

Japanese larch (*Larix kaempferi*) is an important tree in forestry plantations in Northern Europe and particularly in the British Isles. Since 2009, large-scale outbreaks of *Phytophthora ramorum* in the UK and Ireland have been noticed on Japanese larch plantations. Recent outbreaks, which have affected an estimated 1,900 hectares of Japanese larch, or 500,000 trees, in England and Wales alone, are considered to represent a “major step change” in the epidemiology of *P. ramorum*, with associated management issues (European Food safety Authority – EFSA - 2011). These outbreaks raise the concern that the disease will spread to European larch across the EU.

► **Leaf disease in Eucalyptus**

Eucalyptuses were introduced in Spain and Portugal in the middle of the 19th century in order to meet the needs of the growing paper pulp industry. Since 1999, serious damage appeared in young plantations of *Eucalyptus globulus* in the North of Spain and in Portugal (Tejedor et al. 2007, Sylva et al. 2009), where frequent and severe defoliation was observed, causing loss of biomass. The main cause of this disease is suspected to be *Mycosphaerella sp.*, a fungus causing leaf spots and premature defoliation, several strains of which were identified in Spain and Portugal.

*Stand of Eucalyptus globulus not affected and affected by Mycosphaerella leaf disease.*

In short rotation productions of Eucalyptus in Northern Spain, the energy loss from defoliation and reduced growth is estimated to 150000MJ/ha for a defoliation damage higher than 60% (accounting for severity); this amounts to ca. 1200 €/ha.
2.2.3 Spatial and temporal scale of damages

Temporal dynamics

Biotic agents can produce both short and long-term effects on forests, with different periodicity. In particular, both native and alien insect species can affect forests over large spatial and temporal scales (e.g. defoliating sawflies or lepidopteran species). Short-term effects can be understood as those that occur on timescales of weeks to years after the attack of the biotic agent. They include the disturbances directly associated with the action of the pest or pathogen, which may cause defoliation, loss of vigour, or death. Long-term effects in contrast play out over decades or centuries, and are primarily mediated by changes in tree species composition and by the consequent alteration of forest structure, productivity, nutrient uptake, and soil organic matter turnover (Lovett et al. 2006).

Cyclic occurrence is typical for many insect defoliators. Some have been returning periodically since centuries, such as *Zeiraphera diniana*, which displays extraordinary regularity in its cycles (Figure 2-2) (Baltensweiler 1993, Virtanen and Neuvonen 1999, Esper et al. 2007). *Zeiraphera diniana* is considered the most dangerous pest of subalpine larch forests, presumably due to the luxurious feeding of the larvae, leading to desiccation and the typical red-brown discoloration of needles (Baltensweiler et al. 2008). The episodic outbreaks of the moth can actually be interpreted as essential disturbance factor, positively influencing ecosystem functioning via stand composition, forest structure, and succession (Nola et al. 2006, Esper et al. 2007).

Figure 2-2 Characteristics of *Zeiraphera improbana* and *diniana* outbreak events over the past millennium (a) return time record calculated from outbreaks of different intensity (in terms of reduction of wood intensity), and (b) relative frequency distribution of the return time data, averaging 9 years (Esper et al. 2007)
Spatial dynamics

Both population and outbreak dynamics clearly have not only a temporal component, but also a spatial component (Björkman et al. 2011). The spatial dynamics of outbreaks depend first on where the biotic agents can be found, which is related to their host specificity and tolerance to environmental conditions. The distribution of some species is confined to a very narrow geographical range due to adaptation to specific environmental conditions, such as the larch budmoth, which meets its optimal habitat in larch-cembran pine forests between 1700 and 2000 m a.s.l. (Baltensweiler et al. 2008). In contrast, more tolerant species are potentially able to establish populations over a broad geographical range, as in the case of Lymantria dispar (Gray 2004, Karolewski et al. 2007, Pitt et al. 2007).

Most outbreak events are very localised. In 1991, more than 5000 surveyed French forests yielded records of damage by 49 biotic agents, but only three of these (Thaumetopoea pityocampa, Pityogenes chalcographus and Ips typographus) were considered a significant problem in more than 4% of the forests (Anon 1991). Having said this, over the past few decades, many forest pathogen outbreaks have occurred over regional rather than local scales (Castello et al. 1995). Indeed, the spread of a biotic agent interacts with the spatial distribution of its host species and that of the wider landscape, in terms of connectivity and abiotic patterns (Holdenrieder et al. 2004). Improved connectivity has resulted in large-scale outbreaks of the Dendroctonus ponderosae for example (see Box 2-3).

Spatial synchrony has also been observed for many outbreaking forest insect species (Liebhold et al. 2000, Klemola et al. 2006). Synchrony may result from the dispersal of the biotic agent during outbreaks, or from weather events that determine the scale of damage.

Spatio-temporal dynamics

Regional importance of insect herbivores may strongly fluctuate in time, such as Cephalcia abietis which defoliated thousands of hectares of spruce forest in Germany, Czech Republic and Austria during the final two decades of the last century, but recently has caused only minor phytosanitary problems in this area (Führer and Nopp 2001). The generally less frequent, related species, C. arvensis, was not reported to cause severe damage, since over 1500 ha of spruce stands were defoliated and 200 ha forest killed by the sawfly end of the 1980s in the Venetian Pre-Alps (Marchisio et al. 1994). The impact of Lymantria dispar on European forest health also changes between severe (e.g. during 1950s and 1960s) and minor and localised damage (Montgomery and Wallner, In Berryman, 1988). Recently, the pest has been playing an important role in oak decline, rendering host trees highly susceptible to attacks by secondary pathogens and insects, e.g. in continental areas of Croatia (Pernek et al. 2008), Italy (Cocco et al. 2010) and Austria (Balci and Halmschlager 2003). In north-western Germany, oak dieback was brought in context with combined effects of severe defoliation by Lymantria dispar or Operophtera brumata, Tortrix viridana, drought stress and winter frost (Thomas et al. 2002).

The temporal and spatial scales of impacts also depend on the insect's and host's life cycle characteristics, dispersal ability, and requirement for pre-damage conditions. For instance, tree mortality caused by the Anoplophora glabripennis is observed three to five years after infestation (Smith 2005) due to a completion of its life cycle lasting up to three years. Outbreaks of I. typographus, on the other hand may occur one to two years after a pre-damage, often leading to
tree or total stand mortality on a large spatial scale, mainly due the huge reproductive potential of this species. Due to adult dispersal by flight and the movement of plants containing eggs and larvae which remain undetected within dormant buds, the average rate of spread of the alien pest *Dryocosmus kuriphilus* is estimated as 8 km/yr. Some insect species prefer to attack trees only after other species have colonised it. For instance, one study showed that the pine shoot beetle *Tomicus minor* did not colonise any of the damaged trees on its own, but only trees already attacked by *Tomicus piniperda* (EFSA 2010).

The scale of impact also depends on the efficiency and timing of pest detection, in particular for alien species. The alien beetle *Agrilus planipennis*, for instance, is very difficult to detect early and damage identification often occurs one year or more after the beginning of infestation. Similarly, the damage symptoms due to *Rhyacionia sp.*, including distorted growth and misshapen trees, may be mis-diagnosed as another shoot borer or as a disease, thus resulting in a delay in the treatment and more widespread, enduring impacts.

### 2.2.4 Trends in biotic agents over the past fifty years

Over the recent years, awareness about biotic agents has increased, as reflected by the exponential growth in the scientific literature dealing with forest biotic agents. This is both a reflection of the growth in ecological concern and improved monitoring, as well as a potential indication of the importance of the issue in EU forests. The two aspects are confounded and long-term comparable records of outbreaks covering large temporal and spatial scales are needed to assess the trends in biotic damages. But no comprehensive statistics exist on biotic agents occurrence and damage in the EU. Although most MS have information available on biotic damage, it is often scattered over many published sources, using different detection methods, and not necessarily regularly collected through time (Box 2-2)

#### 2.2.4.1 Trends in damage occurrence

According to the Database on Forest Disturbances in Europe (DFDE), 16% of the total volume of wood damaged by natural disturbances in the period 1950-2000 is due to biotic agents (8% of which by bark beetles; Schelhaas 2008), whereas the ICP Forest data suggests biotic agents are the main cause of forest disturbance, representing 57% of all disturbances during 1994-2005 period, most of it being due to insects (34% of damages). This large difference may be explained in part by the fact the ICP data underestimates abiotic events (Jactel et al. 2009), in particular storms and fires, which are the most important abiotic sources of damage according to the DFDE.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Box 2-2 Absence of comprehensive data on forest biotic disturbances

There are two main sources of EU data on biotic disturbances: the DFDE and the ICP Forests database. The DFDE incorporates information regarding biotic and abiotic damages to European forests collected through an extensive literature review, regardless of the sources or expected accuracy. The ICP Forests database arguably offers the most comprehensive overview on forest condition in Europe. ICP forest monitoring was launched in 1985 and is based on systematic transnational 16 x 16 km grids through a standardised protocol.

The advantage of the DFDE is that it goes back to 1449 (although data is usable from 1950 approximately) and provides estimates of the intensity of damages (in terms of volume or area affected). But these trends are likely to be biased towards the more catastrophic, large disturbances (which are more likely to be reported). Moreover, they are difficult to interpret since they are blurred by the overall increase in ecological publications and monitoring effort, which increased reporting (regardless of the frequency of biotic damage events). The European ICP Forest data on the other hand is based on standardised protocols, over a number of sites (almost 6000) across the EU, which makes those trends comparable. However, the time series are shorter (ca. 15 years only), not very frequent (once a year in the summer) and based on a small sample of trees (ca. 20) which may underestimate the actual damages. Moreover, while this data may give an idea on the distribution of hazards, it does not provide any information on their intensity.

Over the centuries, periods with low and high frequencies of damage alternate. Looking at the DFDE time series, it appears that biotic damage has increased over time (Figure 2-3 a, b). This parallels, and probably to an extent follows the increase in forest storms damages, although it is no way near as clear as the latter (Figure 2-3 a). Windthrows, resulting in large amounts of dead and dying wood favour outbreak conditions for many biotic agents. For instance bark beetle outbreaks, such as the ones of 1871-1876, and in the early 1990s followed large storms, often in combination with favourable weather conditions in the following summers. An increase in reported volume damaged by biotic agents is particularly observed following the 1990 storms. A similar tendency for increased frequency of events (although again, this may rather be a reporting bias), and greater impacts per event is observed in Germany (Figure 2-3), where very detailed records on the occurrence and degree of damage from insects and fungi are available from 1800 to 2000 (Waldschutzsituation 1999, 2000). The damages are particularly important following the 1990 storm which resulted in 72.4 million m³ of wood blown over.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

2.2.4.2 **DISENTANGLING CAUSES AND EFFECTS**

Assuming these trends towards increased damage are not an artefact of increased reporting in recent years, either hazard frequency or intensity should have increased. Under the hypothesis that the increased volume of damage results from an increased occurrence of outbreaks, a similar trend should be detected from the ICP forest data. But the trend from 1995 to 2005 does not show any increase in biotic hazards (Figure 2-4, Jactel et al. 2005).

Figure 2-3 Volume of wood damaged by (a) storms, (b) biotic agents (except bark beetle), (c) bark beetle, as reported from 1850-2000, and scaled-up at European level for 1950-2000. (d) volume of wood damaged by biotic and abiotic agents in Germany since 1800 (Schelhaas et al. 2003, Schelhaas 2008)
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Disturbances of EU forests caused by biotic agents

Figure 2-4 Evolution in the proportion of damage causes on the 26 most abundant European tree species between 1995 and 2005 (Jactel et al. 2005)

Long-term time series of population dynamics

No clear trends towards an increased frequency of outbreaks, or towards increased damages per outbreak for a given species seem to emerge from long-term data on population dynamics. For instance, a fifteen-year record of *Ips typographus* in four regions of Sweden highlights the variability in outbreaks occurrence. From 1995 to 2000, population increases are observed in all four regions, but with varying importance and timing. The records then indicate low populations levels between 2001 and 2005, the year of the winter storm ‘Gudrun’. Following this storm, a rapid and synchronised increase in population density is observed across the four regions (Figure 2-5, Wulff et al. 2011). Time series between 1995 and 2006 in Finland on defoliating biotic agents indicate similar variability. (Figure 2-5, Nevelainen et al. 2010).

Figure 2-5 Average number of *Ips typographus* caught by pheromone traps in four areas of northern Sweden from 1995 to 2010. Incidence of the most important biotic and abiotic causes of damage on the Level I plots in Finland from 1995 to 2006 (Wulff et al. 2011; Nevelainen et al. 2010)
Changes in outbreak behaviour of several species have been detected in long-term time series. These changes do not necessarily support an increased frequency of outbreaks. For instance, in what is the longest record of biotic agent dynamics, Zeiraphera ssp. populations have exhibited synchronised, stable Alpine-wide nine-year cyclic fluctuations for the past 1200 years. Since the 1980s however, the synchronisation of the fluctuations ceased and outbreaks became more localised, probably as a result of climate change (Esper et al. 2008).

Pan-European trends in alien biotic agents

It is also possible that in recent years, the pressure from a growing number of biotic agents is responsible for increased forest damages. A growth in alien fungi and insects introductions has been documented (DAISIE, 2009) as a result of accelerated trade and travel. But such temporal trends have not yet been demonstrated for forest biotic agents. However, while introductions may have increased, it is much less clear that the rates of establishment of new insect and pathogens are increasing. Most of the studies of establishment over part or all of the century show either no change or an increase in the number of establishments of pathogens (Jones and Baker 2008 in Wage et al. 2007) and insect biotic agents (Smith et al. 2005, Aukema et al. 2010). In the UK however, while reported outbreaks increased from about 150 in 1993 to about 350 in 2002, only three new biotic agents established in that time (NAO 2003). Thus the pressure from new introductions has been increasing and is likely to continue, but the rate of establishment is not increasing as quickly.

The precise date of arrival in Europe is not known for most species, because the introductions of insects and fungi are most often unintentional, and up to several decades can pass before newly established aliens, especially those which do not become invasive, are noticed (Mattson et al. 2007). In particular, because of their microscopic size, there are virtually no estimates of how many fungi, bacteria and pathogens have been introduced and became established but cause no discernible damage. More than 50% of the fungal species identified in DAISIE, not all of which are found in forests, were described after 1950, and almost 20% only in the 2000s.

Alien insects in contrast are more readily detected. A recent study (Kirkendall and Faccoli 2010) re-examined the evidence regarding alien bark beetles introductions in Europe, and identified 18 established species. The first reference of an alien beetle in Europe is the description almost two centuries ago of the East Asian ambrosia beetle, Xyleborus pfeili, as a European species (Ratzeburg 1837). Only nine more alien species were found in the next 115 years, but the successful establishment of aliens seems now to be accelerating (Figure 2-6). This parallels the marked increase in insect alien species establishment over the last thirty years (Hulme et al. 2009).
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Figure 2-6 The accelerating rate of discovery of introduced Scolytinae and Platypodinae in Europe, shown as numbers of new species found in each 30-year period (Kirkendall 2010)

Damage per event

An alternative explanation is that the damaged amounts per event have increased over time. This can be estimated from the DFDE data, as although damage may be reported more often in recent years, the amount of damage per event has no reason to have significantly changed. Figure 2-7 indicates a trend towards an increased volume of wood damage per event in recent years, and a similar but weaker trend for forest area. This may be explained by an increased vulnerability of forest stands, due to trends towards more monocultures, higher average growing stocks and older stands in recent years. Favourable weather conditions may also have increased the severity of biotic outbreaks.

Figure 2-7 Change in the total average damage by biotic outbreak event reported between 1950 and 2000 (a) in terms of volume (m$^3$) and (b) area (ha).
2.2.5 Influence of forest factors on biotic occurrence and damage

2.2.5.1 Fungi and nematodes

As for other types of pests, the current ranges of fungal (and fungus-like) pathogens are related to two principal aspects: host distributions and habitat characteristics. The climatic tolerance of pathogen (and host) is also important.

- **Host distributions**

Pathogens of secondarily thickened roots, such as *Heterobasidion* and *Armillaria*, co-occur with the principal host plants. *Heterobasidion annosum*, for example, is distributed in Europe wherever pines occur naturally; it is the only species of this genus present in the UK, where Scots pine (*Pinus sylvestris*) is the sole native species in the *Pinaceae*. In contrast, *H. parviporum*, which preferentially attacks spruce, occurs in the natural distribution of Norway (*Picea abies*) and Serbian (*P. omorika*) spruce. In recently reforested areas, particularly those planted onto land that had not carried forest in recent times, both *Heterobasidion* and *Armillaria* are absent. *Heterobasidion* first enters recent plantations at the time of thinning, when freshly exposed stumps provide the infection court for spores. *Armillaria* species are generally absent from such young forested areas, unless the infections were maintained in remnant and isolated trees. *Heterobasidion spp.* are problems in managed forests; although they occur in long-standing reserve forested areas, such as Białowieża Reserve in Poland, they are very rare in these unmanaged situations.

- **Habitat characteristics**

Pathogen species in the *Oomycota* are reliant on the presence of free water in the soil (or through waterlogging) for spread and establishment of infection. Most of these organisms produce motile zoospores that require free water for motion and to locate suitable infection courts.

Nutrition is a factor that also influences the development of some diseases (e.g. Blodgett et al. 2005, Woodward et al. 1998). Excess nitrogen is well-known to result in poor maturation of plant tissues, leaving them susceptible to opportunistic pathogens, such as grey mould caused by *Botrytis cinerea*. Lack of major nutrients (N, P, K) can result in poor growth; in pines, needle pathogens may become active in much younger tissues when these nutrients are limiting, leading to early abscission. Generally poor soil nutrient conditions also lead to greater susceptibility to diseases such as *P. cinnamomi*, as illustrated in the little leaf disease phenomenon seen in the former cotton growing areas of the USA in the early-mid 20th Century. Rectification of the nutrient deficits on these lands lead to recovery of the pines from the disease.

For most infections by aerial pathogens, control by antagonistic fungi is not a possibility. For diseases of fine roots, however, the other soil microflora may have a profound influence on whether a disease develops or not. The presence of certain antagonistic fungi, in particular, and possibly bacteria, can prevent pathogens from attacking roots.
2.2.5.2 **INSECTS**

The current distribution of insects affecting forests in a substantial way can largely be explained by the distribution of the host trees and the habitat characteristics. These stand factors interact with the degree of insect-host specialisation and the management regime to produce the observed species distributions.

▲ **Host distribution**

The potential distributions of insect pests specialised on specific plant species or genera are largely defined by host occurrence. For instance, *Agrilus planipennis* (that was discovered in southeastern Michigan near Detroit in the summer of 2002), is rapidly extending its range following the distribution of ash. However, other alien species such as *Anoplophora glabripennis*, even if they have a preferred host genera (*Acer, Populus, and Salix*) can also attack other hosts such as birches (*Betula spp.* and ashes (*Fraxinus spp.*), or more rarely, red oak (*Quercus rubra*) (Morewood et al. 2005). Similarly, the black locust (*Robinia pseudoacacia*) is the preferred host of *Odonta Dorsalis*, but apple, birch, beech, cherry, elm, hawthorn, and oak may also be attacked.

In some cases, the co-evolution of pest and host is also an important factor to determine susceptibility. Alien pests cause particular damage due to the lack of co-evolution between the alien pest and native host, leaving the native host with few defences. For example, *Tomicus minor* beetles are believed to maintain forest health in regions where they are native, though introductions of Scolytids into new regions where beetles and hosts have not co-evolved may result in detrimental environmental impact.

Even for polyphagous species, which may feed on a range of host plants, outbreaks are frequently determined by the dominance of their favoured tree species. Some examples are the regular outbreaks of *E. autumnata* in the northern mountain birch forests (Riihimäki et al. 2004, Yang et al. 2008, Babst et al. 2010), of *T. pityocampa* in stands of *P. nigra* (Hodar and Zamora 2004, Battisti et al. 2005) and *P. pinaster* (Gatto et al. 2009, Samalens and Rossi 2010), or of *L. monacha* in extensive spruce and pine plantations of Eastern Europe (Bejer, In Berryman, 1988, Vanhanen et al. 2007).

▲ **Habitat characteristics**

In addition to forest species composition, stand structure plays an important role in the susceptibility of forests to biotic agents, by affecting micro-climate, providing habitat for insect herbivores and their antagonists, and influencing individual tree physiology and development (Jactel et al. 2009).

Stand susceptibility to pest typically increases with the age and homogeneity of stands. Rising vulnerability can be associated with specific tree characteristics altering with age (e.g. bark thickness, tree height, tree vitality) and with the quality of forest stands as habitat. More homogeneous forests offer optimal conditions for specialised insect pests, but less diverse living environment for natural enemies. A variety of bark beetles predominantly colonises matured forest stands, such as *Tomicus minor* preferring stands of middle age classes, or *I. typographus* being insignificant in young Norway spruce stands (Christiansen and Bakke 1998, Wermelinger 2004). The extended, uniform afforestation of non-native north-American pines in Northern and North-Western MS can be associated with increased occurrence of defoliators, such as *N. sertifer*.
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Disturbances of EU forests caused by biotic agents

on *P. contorta* (Lindelöw and Björkman 2001) or *E. abietinum* on *P. sitchensis* (Carter and Halldórson, In Day et al. 1998, Straw et al. 2005).

Similarly, the forested landscape of many MS, dominated by pure stands of native Norway spruce or Scots pine is highly vulnerable to biotic risks. The presence of *Ips sexdentatus* is for instance tightly connected to the structure of coniferous stands (in particular with the number of coniferous patches).

However, the impact of insects does not necessarily increase with the intensity of forest management. Bark beetle outbreaks also common in national parks (e.g. Schopf and Köhler 1995, Netherer and Nopp-Mayr 2005), and pest species abundance tends to be higher in natural forests compared to managed ones (e.g. Veteli et al. 2006, Wesolowski and Rowinski 2006). This may be explained the management history, but also to the absence of phytosanitary measures in forest reserves, which result in mature, homogeneous forest stands. On the contrary, natural disturbances can be seen as an integral component of the protected ecosystem, maintaining various stages of successional development.

It is widely assumed that mixed stands are less at risk from insect herbivores, through dilution effect (Jactel et al. 2009). This seems to be especially true for certain species of bark beetles and of monophagous defoliators. Diverse studies have shown that when the majority of trees are deciduous or conifers other than the host species, it can markedly reduce the probability of severe stand damage by *Ips typographus* (e.g. Wittek 1984, Moor 1987, Meusberger 1995) or *Neodiprion sertifer* (Kaitaniemi et al. 2007).

With regard to the risk of damage caused by more generalist pests, which can be enhanced in mixed forests (Jactel et al. 2009), landscape rather than stand composition plays an important role. Habitat fragmentation by forest management or in a diverse natural landscape was shown to increase dispersal losses and positively influences the occurrence of antagonists or non-host species, e.g. for *Operophtera brumata* (Wesolowski and Rowinski 2006) or *Thaumetopoea pityocampa* (Robinet et al. 2007, Samalens and Rossi 2010, Barbaro and Battisti 2011). On the other hand, over-exploitation of forests enhances stand susceptibility to pest outbreaks (Cocco et al. 2010).

Forest susceptibility to damage is also linked to tree status. Some alien species are known to preferentially attack stressed or damaged plants (e.g. *Phoracantha semipunctata*), and dead trees (Long-horned beetles). For these species, a well-maintained tree of a susceptible species can be less at risk of infestation than a neglected tree of a resistant species. However, it might happen that some secondary pests, such as *Tomicus minor*, transfer from weakened or dead trees to healthy trees when populations reach epidemic levels.

Site degradation can generally be seen as a factor predisposing forests to pest infestation. Increased damage is evidenced for a variety of species (e.g. *L. dispar, C. abietis, I. typographus*) in stands growing on poor or xeric soils (Worrel 1983, Berryman 1988), exposed to high solar irradiation or wind (Andresen et al. 2001, Führer and Nopp 2001, Baier et al. 2007). At the same time, nitrogen deposition or fertilisation may also enhance susceptibility of trees to attack via improved nutritional quality of needles, as was shown for some sawfly species (Führer and Nopp 2001) or *Elatobium abietinum* (Straw et al. 2005).
2.2.5.3 Mammals

Both wild grazing ungulates and domestic grazing herbivores are clearly affected directly or indirectly by anthropogenic effects. For example, over the past 200 years the abundance of large carnivores as well as wild and domestic herbivores have shown strong changes in European forest landscapes. While the former have generally declined both in range and numbers, domestic grazing herbivores have largely been replaced by wild browsing herbivores.

Damages by mammals and grazing affect 2.2% of the EU27 forest area (MCPFE 2011). Sweden and Italy have the largest areas affected (respectively 6.2% and 3.5%). As an example, at present, three management problems are associated with large mammals in Sweden. These are linked to both reduced grazing disturbance and increased browsing (e.g. Edenius).

- Maintaining the traditionally agricultural wooded grasslands and forests grazed by domestic animals and the associated biodiversity in cultural woodlands.
- Reducing the impact of increased browsing by wild herbivores (Alces alces and Capreolus capreolus) causing forest damage on commercially harvested coniferous trees as well as on deciduous tree species diversity with subsequent negative effects on forest biodiversity.
- Managing the local return of large predators (Lynx lynx, Canis lupus and Ursus arctos) in landscapes where these species became extinct more than 100 years ago. In Scotland for example, the government is studying the possibility to reintroduce the Eurasian lynx (Lynx lynx), which would restore predation on the deer populations (Hetherington, 2006)

Altogether, these dynamics have created an increased number of complicated interactions between large mammals, man, and the landscape (Angelstam 2002).

2.2.6 Drivers of changing biotic agents distribution

Although certain biotic agents may not yet be present in the EU, they may be introduced and spread in the coming years. Biotic risk can thus only be predicted by accounting for the dynamic drivers of biotic agents distribution.

As changes in climate induce changes in biotic agents and/or their host species distributions, biotic risk may increase in some locations. In addition, when climatic changes lead to a desynchronisation between the distribution of host trees and that of their biotic communities, the released biotic agents may be able to reach outbreak conditions. Similar consequences may arise from the climate-induced desynchronisation of the biotic communities. The sensitivity of biotic agents to climate change is discussed below, while the consequences on their distribution will be discussed in Chapter 4.

Biotic risk may also increase simply through a greater number of species reaching outbreak conditions. The expansion of international trade has led to increased movement of goods between countries and continents. Horticultural and wood trade are the main pathways for the introduction of alien forest pests. Although the movement of timber and wood products is controlled, not all types of wood are controlled and not all pests are easily detectable. Moreover,
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Humans are also a significant vector of introduction. Thus, despite these regulations, an increase in alien forest biotic introductions is predicted. It is unclear how much this will affect biotic risk though, as most introduced pathogens or insects do not establish stable populations.

Alternatively, the evolution of biotic agents, through hybridisation or simply through adaptation can increase biotic pressure on forests. Indeed, pathogens, fungi and insects have much shorter generation times than trees (by at least two orders of magnitude), which gives them a chance to be ahead in the arms race against the defences of their hosts.

2.2.6.1 Fungi and Nematodes

Climatic tolerance

Although the presence of host species is clearly a driver for the occurrence of all pathogens, humidity conditions in spring and into flowering time are frequently cited as the main factors influencing development of diseases such as leaf spots and mildews. Humidity at the leaf surface is of great importance for spore germination and subsequent establishment of infection. This is also true for sites presenting low air movement. For example, the most severe damage caused by the pine shoot blight and canker, Sphaeropsis sapinea, are found in protected valley sites (Wingfield and Swart 1994), while spruce stands on slopes or hilltops with a high incidence of fog and westerly winds of high humidity are highly disposed to shoot blight, Sirococcus conigenus (Anglberger and Halmschlager 2003, Jactel 2009).

Temperature also influences pathogen distribution. For instance, limitations to the damage caused by P. cinnamomi in other environments where susceptible host species are present, such as northeast Scotland (Chavarriaga et al. 2007) are probably due to low temperatures; warmer conditions would enable overwintering of the pathogen.

The interactions between climate change and forest pests are well-illustrated by the example of Bursaphelenchus xylophilus. The nematode induces pine wilt in conditions where temperatures are raised during the summer and there is significant seasonal variation in water availability. In conditions that do not result in significant stress to the tree, infestation by the nematode does not result in wilting (Evans et al. 2008). Therefore, given that disease development appears to be linked to temperature and water availability, both of which are expected to change, there is potential for the Bursaphelenchus xylophilus to cause pine wilt across greater geographical areas.

Climatic factors favouring the spread of invasive fungi and nematodes

The major drivers for the occurrence of all invasive alien fungi are similar to those for native species. Humidity is the major environmental factor, although temperature limits the spread of aliens, depending on their origins. Rising temperatures due to climate change will facilitate fungal pest introduced from Mediterranean or tropical countries, such as Sphaeropsis sapinea, Phytophthora spp. and Biscogniauxia mediterranea (Desprez-Loustau et al. 2006). At the same time, there are several fungal pest species that show a preference for lower temperatures and therefore prefer sites of higher altitude and latitude, such as Scleroderris canker (Karlman et al. 1994). With alien species, it is not always possible to determine the potential host range, at least within a family or genus of plants. Phytophthora ramorum, of course, is a generalist pathogen, affecting a wide range of trees, both gymnosperms and angiosperms. The recent host jump to
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*Larix kaempferi*, however, was unanticipated, as it had not previously caused widespread and severe damage to conifers (Webber et al. 2010). The main difficulty in determining the likely host range of an alien invasive pathogen is that the biologies of these organisms are largely unknown until they establish in a ‘new’ environment: within the native ranges, these organisms are in balance, both genetically and environmentally, with the host plant species.

Climate change will alter the range of forest susceptible to alien pests. Changing patterns of rainfall will alter humidity at local and regional scales, leading to increased frequencies of, for example, conditions enabling *Phytophthora* species to cause disease. Increasing mean temperatures will also extend the latitudinal and altitudinal ranges at which alien invasive organisms can cause damage to trees.

### Hybridisation of alien fungi

A further example of a factor impacting on the ability of alien invasive pathogens to cause increasing problems is hybridisation between closely related organisms. *Phytophthora alni* is such an example, it appears that *P. cambivora* and another as yet unidentified *Phytophthora* (possibly closely related to *P. fragariae*, cause of strawberry red-core), were able to mate, resulting in the production of a group of asexual hybrids, designated *P. alni*. Three sub-species are known within this hybrid group. Until this organism arose, the genus *Alnus* was not considered host to *Phytophthora*, yet now, these trees are being killed in riparian ecosystems, where they are of great importance in soil stabilisation and nutrient cycling.

#### 2.2.6.2 INSECTS

The main factors affecting insects distribution are the temperature and habitat requirements specific to each insect herbivore species.

### Climatic tolerance

Climate can influence the geographical distribution of insect species. Moreover, depending on the local climate the same species can present one or more generations per year. This is the case of the alien species *Anoplophora chinensis*, which in tropical and subtropical regions, presents a single generation per year, but the life cycle occasionally takes two years depending on climatic conditions. The bark beetle, *Ips sexdentatus* has one to five generations per year depending on the climatic conditions, whether the north of the Arctic Circle, or the Mediterranean region (EPPO/CABI, 1997).

As ectothermic organisms, insects find optimal developmental conditions within a certain range of temperature limited by specific lower and upper developmental thresholds. Favouring temperature conditions have been associated with mass propagation for a variety of species (e.g. Schwenke 1978, Battisti 2004, Wermelinger 2004, Rouault et al. 2006). The positive effects of temperature increase during the growing season may range from earlier onset and prolonged time for reproduction and accelerated development rates for bark beetles, such as *I. typographus* (Wermelinger and Seifert 1998, Baier et al. 2007) or aphids (*E. abietinum*) (Cannon 1998, Harrington et al. 2007) to enhanced post-diapause development of Lepidoptera (*L. dispar*) (Gray 2004). The alien species *Phoracantha semipunctata* presents a life cycle from egg to adult of 2-3 months in warm conditions and as long as 9 months in cool conditions. *Lymantria dispar* and
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Lymantria monacha are famous examples for high potential of altitudinal and latitudinal range extension, given summer temperatures will not rise above the temperature threshold indicating heat stress (Karolewski et al. 2007, Vanhanen et al. 2007, Hlášny and Turcáni 2008).

The same species on the other hand might show limitations in range expansion or even range contractions due to unsatisfied requirements during hibernation (Gray 2004, Vanhanen et al. 2007). Increased winter temperatures will not be beneficial for insects, which do need a certain amount of low temperature to induce and maintain diapause or to increase frost resistance (Bale et al. 2002). However, species exhibiting obligative winter diapause, such as N. sertifer or webspinning sawflies again will profit from shortened generation development (Hellrigl and Salvadori 1998, Eichhorn and Bogenschütz 2000, Battisti 2004). Milder winters can also positively act upon the survival rate of actively overwintering stages, such as of anholocyclic populations of Elatobium abietinum, which are sensitive to chilling and winter starvation (Saldana et al. 2007, Day et al. 2010). The distribution of the alien Agrilus planipennis (or marcopoli) could potentially be limited by low winter temperatures. However, modeling species range extensions solely based on macroclimate is questionable, since it neglects microclimatic effects and phenotypic plasticity of the organism of interest.

The number of feeding hours limited by certain day- and night-time temperatures also explains population survival of some insect species well, like for instance Thaumetopoea pityocampa (Battisti et al. 2005, Buffo et al. 2007). Reduction of egg mortality likely contributes to range expansion of species at the northernmost limits of distribution, such as Neodiprion sertifer (Virtanen et al. 1996, Veteli et al. 2005). Similarly, Epirrita autumnata is able to tolerate very low temperatures during egg stage, but also exhibits potential for future increased frequency and area of outbreaks in continental regions of Fennoscandia (Virtanen and Neuvonen 1998, Jepsen et al. 2008).

**Biotic interactions**

Tritrophic interactions between pest insects, host plants and control by natural enemies under a changing climate probably play an important role for future forest health and community composition. Yet, these are still insufficiently understood (Netherer and Schopf 2010). Antagonists are seen as main drivers of population cycles, e.g. of the larch budmoth by Turchin et al. (2003) and possibly become more active in a warmer environment (Cannon 1998, Virtanen and Neuvonen 1999). They eventually serve as likely explanation for population outbreaks not explainable by climatic or nutritional factors (e.g. T. pinivora in Gotland, Aimi et al. 2008, Ronnas et al. 2010).
Impact of climate change

*Lymantria dispar* and *Lymantria monacha* are famous examples for high potential of altitudinal and latitudinal range extension, given summer temperatures will now rise above the temperature threshold indicating heat stress (Karolewski et al. 2007, Vanhanen et al. 2007, Hlásny and Turcáni 2008). Indeed, some bark beetles, such as *Dendroctonus micans*, are likely to benefit from an increased frequency of summer drought and climatic warming, although its specific predator (*Rhizophagus grandis*) might benefit to a greater extent, thus potentially reducing the impact.\(^7\)

Temperature influences both insect and plant physiology. Changed patterns of winter temperature will thus strongly influence developmental coincidence between pest and host species. Changes in synchrony between bud-burst and egg hatch have been recorded recently for the *Zeiraphera improbana* and *diniana* (Battisti 2004) and *Operophtera brumata* (Visser and Holleman 2001), where they are generally seen as the key factor causing year-to-year fluctuations in abundance (Watt and Farlane 2002). Its however not clear if synchrony will remain disrupt, given the strong local phenological adaptation of *O. brumata* to its host plants (Tikkanen et al. 2006, Yang et al. 2008).

Trophic interactions are also to be expected with regard to changes in precipitation (e.g. Christiansen and Austara 1996, Huberty and Denno 2004) as well as increased atmospheric concentrations of CO\(_2\) and O\(_3\), with strong species-specific consequences (Hättenschwiler and Schafellner, 2004) ranging from enhanced nutritional quality of host tree needles (Watt et al., in Day et al. 1998), over indifferent (Buse et al. 1998, Dury et al. 1998, Henn and Schopf 2001) to adverse effects for feeding larvae (Hättenschwiler and Schafellner 1999, Wang et al. 2009). Moreover, water stressed trees are in general more susceptible to pest attacks (e.g. drought stressed eucalyptus are more easily infested by *Phoracantha semipunctata*) (Hanks et al. 1999).

2.2.6.1 Mammals

Extensive studies on mammal species indicate that, besides population density, climatic factors can influence juvenile survival, primarily during winter (Milner et al. 1999). We know for instance that increasingly warm winters associated with the North Atlantic Oscillation influence the development and fecundity of the deer *Cervus elaphus* (Ottersen et al. 2001). The mortality of this ungulate species has been observed to be greater during cold winters and at the end of moderate winters. This results from a combination of increased metabolic activity and decreased access to food resources buried in deep snow (Post et al. 1997).

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\(^7\) [www.forestry.gov.uk/fr/INFD-SZGXZ](http://www.forestry.gov.uk/fr/INFD-SZGXZ)
Box 2-3 Emerging risks under climate change – the example of *Dendroctonus ponderosae* in North America

The increased area, frequency or intensity of pest outbreaks is due to improved developmental conditions, such as for *Ips typographus* (Wermelinger 2004, Lange et al. 2006, Baier et al. 2007). There are comparable conditions in North America and Canada regarding recent broadscale eruptions of the spruce beetle, *Dendroctonus rufipennis*, the mountain pine beetle, *Dendroctonus ponderosae*, and the pinyon ips beetle, *Ips confusus* (Raffa et al. 2008).

**Eruptions of the mountain pine beetle, *D. ponderosae* – are similar “worst case” scenarios also probable for European bark beetles?**

The recent epidemics of *Dendroctonus ponderosae* in western North America and Canada most prominently represent the possibility of uncontrollable, large-scale outbreaks of a forest insect pest favoured by improved developmental conditions. Inhabiting a wide geographical range, from high elevations in southern California to sea-level in British Columbia, *Dendroctonus ponderosae* erupted 4 to 5 times during the last century, but the outbreaks never reached the current, vast extents (Williams and Liebhold 2002, Aukema et al. 2008). The persistent mass propagation of the bark beetle during the past decade has been affecting more than 25 million hectares of pine forest throughout British Columbia and the northern and central US Rocky Mountains (Bentz et al., 2010). The lodgepole pine forests in B.C. have been most severely devastated, with outbreak areas escalating from about 2 million ha in 2002, over 7 million ha in 2004, 9 million ha by 2007 to over 13 million ha (Dymond et al. 2006, Robertson et al. 2008, Wulder et al. 2009). In 2009, total cumulative volume losses were estimated at 620 million m³, which amounted to 46% of B.C.s total merchantable pine volume (Wulder et al., 2009). A wood volume of more than 900 million m³ is expected to be killed by 2013, representing more than 80% of mature pine in British Columbia (Stig Larsson, personal communication).

**What are the main causes of these unprecedented pest outbreaks?**

Primarily, *Dendroctonus ponderosae* is profiting from the abundance of highly susceptible host trees. Most vulnerable to attack are mature, intermediate dense stands of *Pinus contorta*, *P. ponderosae* and *P. albicaulis* (whitebark pine), with high proportions of large-diameter as well as stressed or damaged trees (Dymond et al. 2006, Fettig et al. 2007, Robertson et al. 2008). *P. contorta* is the dominating tree species in interior B.C., but was historically underutilised as it was considered to be of minor economic value (S. Larsson, personal communication). *Dendroctonus ponderosae* consequently finds an over mature tree stock, along with high abundance of connected habitat resulting from decades of fire suppression (Robertson et al., 2009). The long-lasting outbreaks and expansion into previously climatically unsuitable geographical regions have finally been triggered by sustained favourable climate for bark beetle survival. The absence of extreme cold events that normally regulate *Dendroctonus ponderosae* populations through winter mortality facilitated altitudinal and northwards range extension (Régnière and Bentz 2007, Régnière 2009, Robertson et al. 2009). Populations have also spread eastwards beyond the climatic isoline that marks winter temperatures of -40°C, lethal to overwintering *Dendroctonus ponderosae* larvae, into the Canadian province of Alberta (S. Larsson, personal comm.).
Box 2-3 Emerging risks under climate change – the example of *Dendroctonus ponderosae* in North America

**Options for control and differences to *I. typographus* in Europe**

Controlling the mass propagation of *Dendroctonus ponderosae* is almost impossible. The large spatial scale of the outbreaks impedes effective direct control measures, such as felling and burning or decortication of attacked trees, removal of the bark on standing trees, or harvest and prescribed burning (Wulder et al., 2009). In the acute epidemic situation, preventive management measures (e.g. alteration of age class distribution or species mix in order to limit the number of host trees) are irrelevant, although there is evidence, that the manipulation of stand density by thinning strongly influences stand susceptibility to outbreaks (Fettig et al. 2007, Wulder et al. 2009).

Due to this very limited human scope of action, recent outbreaks of *Dendroctonus ponderosae* will probably not collapse before host trees are depleted or adverse weather conditions are enduring. In this regard, epidemics of the spruce bark beetle in Europe are still “controllable”, as European forests are much more fragmented and heterogeneous in species composition, structure and age. Contrary to *Dendroctonus ponderosae*, *I. typographus* also attacks felled host trees, so that control and sanitary measures can be more efficiently implemented, also because of the smaller scale of outbreak areas.

### 2.3 Impacts and trends for selected biotic agents

**Box 2-4 Key lessons learnt**

- *Ips Typographus* is probably the biotic agent for which the most detailed and long-term information is available. It is native and widely distributed in EU forests, and mostly a secondary biotic agent, affecting trees that are already weakened (by storms, droughts or other causes). As a result, it is possible to assess the level of risk of forest infestation for this species, according to the combination of environmental traits that are present. Contrasting management strategies should be used depending both on the level of risk, and the management aims. *Ips Typographus* is currently expanding its range, and interestingly, outbreak risks are similarly high outside its natural range than in non-managed, old-growth spruce stands.

- *Anoplophora chinensis* is a potentially emerging threat to EU forests. Currently, this alien insect is only found in the urban context, but given the damages it may cause in its native environment, it constitutes a threat to EU forests.

- *Thaumetopoea pityocampa*, is a native widespread defoliator that seems to be profiting from climate change. It is exhibiting altitudinal shifts in its distribution due to improved winter survival as well as latitudinal shifts. Moreover, it has been able to recruit new host species resulting in new distributions.

- *Bursaphelenchus xylophilus*, is an alien invasive nematode attacking pine trees that is currently only present in Portugal, but has the potential to spread Northwards. It can cause significant economic as well as environmental impacts, and consequently has been listed as
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Box 2-4 Key lessons learnt

- Ash dieback is a rapidly spreading disease caused by the fungus *Chalara fraxinea*. It is now widespread in Central and Northern Europe and causes the rapid decline of Ash trees of all ages. Little knowledge is currently available on its sensitivity to climatic factors. However, transport of plants from nurseries seems to be the main vector of disease spread, therefore control measures related to trade could help prevent further spread of the disease.

- Chestnut canker (or chestnut blight) is a disease that caused severe damages in the US over the past century. It appeared in 1938 in the EU and is currently widespread in the distribution range of the chestnut, although a natural event attenuated the virulence of the pathogen (*C. parasitica*), leading to much lower impacts than in the US. Given that several strains of *C. parasitica* are known in Europe, it is difficult to predict how these may respond to changing climatic conditions. Research into understanding the development of hypovirulence appears a promising management option.

- Dutch elm disease is a widespread disease of elm trees, caused by two strains of fungi (*Ophiostoma ulmi* and *O. novo-ulmi*) that wiped out most the European populations of elms. The environmental and economic impacts of elm dieback have been considerable. None of the management approaches used proved successful. Experience shows that actions must be taken rapidly and applied rigorously for any management attempts to be effective. Accurate and rapid diagnosis is also important.

- Sudden oak death is a recent disease caused by the alien pathogen *Phytophthora ramorum* that is already present in most of Europe. The first outbreaks were reported in 2004 in the UK and resulted in serious landscape-scale damages to the environment, and economic losses. The pathogen has a wide range of hosts, and climate change is predicted to benefit the spread of the pathogen, making it one of the most significant threats to forest ecosystems known. Control measures in the ornamental nursery stock industry, which is one of the main route of spread of the pathogen, could form an efficient prevention measure.
2.3.1 *Ips typographus*

2.3.1.1 **Distribution and Biology**

<table>
<thead>
<tr>
<th>Species</th>
<th><em>Ips typographus</em> (European Spruce Bark Beetle)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Insect</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Native</td>
</tr>
<tr>
<td>Distribution</td>
<td>EU 27: Austria, Belgium, Bulgaria, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Italy, Latvia, Lithuania, Luxembourg, Netherlands, Norway, Poland, Romania, Slovakia, Slovenia, Sweden, UK and Asia</td>
</tr>
<tr>
<td>Host specificity</td>
<td><em>Picea</em> (Norway spruce; main host), <em>Abies</em>, <em>Larix</em> and <em>Pinus</em></td>
</tr>
<tr>
<td>Spatio-temporal scale of impact</td>
<td>Large spatial scale, rapid outbreaks</td>
</tr>
<tr>
<td>Lessons learnt for management</td>
<td>Outbreak risks are similarly high in lowland plantations of Norway spruce outside its natural range and in non-managed, old-growth spruce stands; keeping the balance between interests of nature protection and the control of infestation areas is of major importance.</td>
</tr>
</tbody>
</table>

*Ips typographus* is one of the main and most aggressive bark beetle species causing damage to European and Asian forests over a wide latitudinal and altitudinal range. Being natural part of spruce forest ecosystems, the moderate to large (4.2 to 5.5 mm) scolytid (Figure 2-8) is harmless at low population densities. Under native conditions, spruce bark beetles do not succeed in colonising vigorous trees, which react to the attack by primary and secondary resin flow lethal to beetles (Raffa et al. 2008). Overcoming tree resistance and successful attack is possible only at high population densities. The destructive potential of this beetle thus lies in the eruptive increase of abundance, attack of healthy trees, and expansion of infestation under outbreak conditions, which are commonly triggered by favourable temperature conditions and pre-damages such as storm throw or fire.

Mass outbreaks of *Ips typographus* are not a new phenomenon and date back at least 200 years. During the last century, epidemics were for instance recorded in Central Europe from 1944 to 1951 or in southeastern Norway between 1971 and 1982 (described by Christiansen and Bakke 1988). These epidemics are to a great part due to the intensive cultivation of Norway spruce, which has led to large areas of susceptible, secondary forests on sites naturally supporting deciduous trees, e.g. in the Austrian Pre-alps. (Lindner et al. 2008). Today, not only lowland stands have been affected by severe infestation but also forests at higher altitudes, which often exhibit protective function or belong to nature reserves or national parks (e.g. Bayerischer Wald, Germany; Sumava mountains, Czech Republic; High and Low Tatras, Slovakia; Limestone Alps, Austria).
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Figure 2-8 (a) *Ips typographus* and (b) egg and larval galleries

**Box 2-5 *Ips typographus* biology**

Adult beetles emerge from their overwintering sites and start searching for suitable breeding material as soon as diapause is terminated and temperatures allow for swarming. Males initiate tree attacks by excavating the tree bark to prepare a nuptial chamber. One to four females are attracted by pheromones. After coupling, the female beetles construct galleries, where they lay their eggs and from which the young larvae feed at right angles, producing the typical picture of attack. Since the young beetles feed on the inner bark, sap flow is inhibited and the tree eventually killed. Seven to ten weeks later, at the beginning of summer, the first beetles start emerging through round exit holes. Given favourable climatic conditions, these filial beetles start to establish a second generation, potentially followed by a third one. Reproductive potential may, however, also be high in univoltine populations via the establishment of sister broods (Martinek 1957, Wermelinger and Seifert, 1999). At the end of the summer (facultative) diapause is induced in adult beetles, which hibernate either under the bark or in the forest litter at the base of trees.

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2.3.1.2 Damage significance

Due to the attack of relatively vital host trees and the huge reproductive potential of *Ips typographus*, this species is considered to be one of the most destructive European forest pests (Figure 2-9). In addition, *Ips typographus* can cause indirect damage, by being the vector of fungi harmful to trees, including the primary invader *Ophiostoma polonicum*, a blue-stain fungus.

On a European scale, *Ips typographus*, has been identified as the most damaging species of bark and wood-boring insects, based on quantitative information collected in the BAWBILT database on damage and control (Grégoire and Evans, 2004). The database indicates a potentially threatened area of 7 640 000 ha in ten MS, whereby a minimum of 2 819 000 ha of forests were infested by the pest between 1990 and 2001, amounting to the dieback of 31 643 000 m³ of Norway spruce. The highest losses were indicated for Austria, Poland, Germany and Slovakia (Figure 2-10b).

Concerning Austria, the variability in the amount of timber loss caused by bark beetles and by damaging storm and snow events since the 1950s has been well documented by the national forest research centre (Figure 2-10a). Since the 1990s, bark beetle damage intensity has been increasing, especially in the susceptible, drought-prone Norway spruce stands of the lowlands, and triggered by heavy storm events (e.g. Vivian and Wiebke 1990, Lothar 1999, Kyrill 2007) as well as by climatic conditions favouring insect development.

Outbreaks often lead to tree or total stand mortality on a large spatial scale and thus have enormous socio-economic and ecological impact. The direct economic consequences are the loss of material for timber production. For instance, the last Norwegian mass outbreak, from 1971 to 1981, killed the equivalent of 5 million m³ of spruce timber, which was worth EUR 199 281 600 in

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10 *Ips typographus* damages, Austria: Institute of Forest Entomology, Forest Pathology and Forest Protection, University of Natural Resources and Life Sciences, Vienna

11 COST Action E16 (1998), provides a synthesis on the research work done in Europe on all Bark And Wood Boring Insects In Living Trees (BAWBILT), considering both the biological (trees, insects, associated organisms, and their relationships) and forest management aspects. It developed a database on damage and control methods based on information gathered from the 24 participating countries, see Lieutier et al 2004.
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2006 prices, almost equivalent to the total harvest benefits of Norway spruce timber for one year. In Norway, the average yearly economic loss caused by infestations of *Ips typographus* alone can be estimated to reach EUR 2.615 million (Økland and Skarpaas 2008). In addition, various costs arise from control and sanitary measures, as well as from management consequences, such as harvesting forest stands before maturity, devaluation of timber by blue stain fungi or for the appropriate storage of calamity wood.

The spatial scale of bark beetle infestation is large but difficult to predict. Beetles have a high dispersal capacity and flight distances of up to several kilometres (Botterweg 1982, Forssé and Solbreck 1985, Gries 1989), which means that infestations can occur even beyond the 'high risk'-zone of 500m from the primary attack (e.g. suggested by Lexer 1995, Dutilleul et al. 2000, Wichmann and Ravn 2001). For instance, in the NP Bayerischer Wald, 65% of new infestations occurred within 100m from the previous attack, 95% within 500m, but there was a remaining risk of 5% for distant attacks (Dworschak et al. 2011).

Depending on local conditions, outbreaks will develop differentially, simply extending along borders of a primary infestation (Ravn 1985) or showing patterns of agglomerated or scattered attacks (Nüßlein et al. 2000, Hurling 2002). Present knowledge on outbreak dynamics still does not allow to predict reliably on a regional scale whether the infestations will cease or, in the worst case, exceed all thresholds and become self-sustaining until host trees are exhausted (Malmström and Raffa 2000, Raffa et al. 2008).

### 2.3.1.3 **Factors triggering outbreaks of *Ips typographus***

- **Importance of climatic factors**
  - **Temperature and solar radiation**

As exothermic organisms, insects such as *Ips typographus* are particularly sensitive to temperature. Temperature conditions influence every step of the beetle’s life cycle, ranging from the timing of spring flight and onset of infestation to development rate and number of generations, establishment of sister broods, hibernation as well as beginning and end of diapause (Vité 1952, Annila 1969, Schopf 1985, 1989, Faccoli 2002, Doležal and Sehnal 2007). Microclimatic thermal conditions in the inner bark are particularly crucial for bark beetle development (Wermelinger and Seifert 1998, Baier et al. 2007). For this reason, it is not only increased air
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**temperature** but also the availability of sun-exposed breeding sites and **high solar irradiation** that trigger propagation. As a consequence, any structural feature leading to increased insolation and temperature conditions in the stand favour the development of *Ips typographus*

**| Significance of drought periods**

Incidence and amount of precipitation both impact the vitality status of forest stands and directly influence bark beetle development. Periods of enduring precipitation may inhibit flight and mating or delay larval stage and thus whole juvenile development (Merker 1952, 1956, Schimitschek 1969, Bakke 1983, Reindl 1990).

In particular, fluctuations in water supply have a major impact on the susceptibility of spruce stands to attack by *Ips typographus* (Schwerdtfeger 1955, Merker 1956). Outbreaks are frequently associated with summer droughts (e.g. Bakke 1983, Faccoli 2009, Ogris and Jurc 2010) or site conditions impeding water accumulation and storage (Worrel 1983, Schopf and Köhler 1995). Forests stocking on dry, rocky soils or on soils which have an irregular water supply are more frequently affected by infestations than stands having a constant water supply (Merker 1952, Schwerdtfeger 1955, Vité 1984, Berryman 1988). While moderate water stress is evidenced to rather reduce host suitability for scolytids due to increased production of carbon based secondary metabolites and constitutive resin flow, tree susceptibility to attack increases with rising severity of drought stress, which may inter alia be explained by oleoresin production becoming limited (Ayres 1993, Christiansen and Bakke 1997, Lombardero et al. 2000, Desprez-Loustau et al. 2006). Further research is necessary in order to define critical thresholds of water stress that dispose individual trees or forest stands to bark beetle attack. Yet, reference values of seasonal rainfall amounts are available indicating natural range limits of Norway spruce (Schmidt-Vogt 1989) and highest predisposition to *Ips typographus* (Schwerdtfeger 1955).

**| Pre-damages such as wind-throw or fire**

Storm throw, strong precipitation events (e.g. wet snow accumulation), avalanches or fire can substantially modify stand structure, predisposing forests to further damage such as bark beetle infestation. Such abiotic disasters may provide excess supply of food and breeding material, commonly triggering outbreaks of *Ips typographus* within one or two years after the event (Marchetti et al. 1999, Göthlin et al. 2000, Eriksson et al. 2005). Exposed forest stands on ridges, hilltops or upper slopes, as well as stands suffering from root deterioration (e.g. caused by stagnic soil conditions) are particularly prone to such pre-damages. As a consequence, gaps of different size are created where the sun can penetrate the forest and solar irradiation and temperature increase. Thrown, sun-exposed stems show significantly higher bark temperature than standing trees (Wild 1953), which may result in earlier colonisation and accelerated development of beetles, but also in higher mortality given temperatures above 30.4°C (Baier et al. 2007). At the same time, trees remaining at newly created forest edges may show less resistance to bark beetle attack, because of stress due to root damages or sudden insolation. Bark beetle outbreaks are often initiated from small gaps in the forest, where breeding material deteriorates less rapidly than in large openings with high solar irradiation and evaporation. Large wind-thrown areas however show high attractiveness to beetles, irrespective of size (Göthlin et al. 2000, Eriksson et al. 2005). The timely clearing of damage areas is crucial for avoiding bark beetle mass outbreaks. While infestation of standing trees already peaks one year after the storm incidence and ceases after two years given adequate sanitation measures, epidemic
conditions are to be expected up to six years given no human intervention (Forster and Meier 2010).

**Significance of windthrow extent and intervention-regimes**

The timely clearing of damage areas is commonly regarded as crucial measure to avoid bark beetle mass outbreaks. As Forster and Meier (2010) exemplify in the guidelines for forest practice of WSL (Swiss Federal Institute for Forest, Snow and Landscape Research), epidemic conditions may endure up to 6 years given no human intervention after a storm disaster. In case adequate sanitation measures are implemented, infestation of standing trees usually peaks one year after the windthrow and eventually ceases after 2 years. According to Swiss experience, outbreaks not only last a shorter time, but also are less serious when storm-felled trees are removed (Figure 2-11).

![Figure 2-11: Progress of bark beetle mass outbreaks with and without human intervention (clearing of windthrow areas, sanitary fellings) (modified after Forster and Meier, 2010)](image)

During the last 20 years, several storm events caused significant damage in spruce-dominated forests all over Europe. Most of these disasters were followed by massive propagation of *Ips typographus*. Differences in disturbance management offer the chance to analyse bark beetle development depending on the extent of damage and of human intervention.

In central European national parks such as 'Bayerischer Wald' (Bavarian forest national park) non-intervention led to the almost total loss of mature, montane spruce stands (Heurich, 2001), however also gave rise to impressive natural regeneration of over-aged, highly susceptible forest ecosystems. Information on development of *I. typographus* unhindered by control measures is rare. In a comprehensive review of spruce bark beetle research, Wermelinger (2004) points out the obvious higher abundance of the pest in uncleared windthrow areas. At the same time, he refers to shorter maternal galleries and fewer overwintering progeny found in areas without control measures of the German national park “Harz” compared to zones of integrated control. However, at this time about 70% of spruce stands in the non-intervention area were already killed by *I. typographus*, while sanitary measures allowed for the preservation of adjacent mature forest, which was esteemed economically significant (Niemeyer et al., 1995). Most recently, Nikolov et al. (2011) reported from High Tatra National Park (Tanap) of Slovakia, where the storm Alzbeta destroyed a forest area of about 12,000 ha in 2004. An amount of 165,000m³ of damaged spruce was left uncleared in the core zones, triggering an extended outbreak of *Ips*
Typographus. Infestation patterns were clearly related to windthrows, more than 65% of attacks were recorded within 300m and 96% within 1000m of uncleared areas. However, in stands where sanitary cuttings were implemented, mass propagation could not fully be prevented, a finding that corresponds to previous experience (Forster and Meier 2010, Wermelinger et al. 1999). The timing of maximum beetle abundance obviously does not follow a general trend but seems to depend from the character of phytosanitary measures and climatic conditions. While sanitary cuttings postponed outbreaks by one or two years in the High Tatras (Nikolov et al. 2011), Wermelinger et al. (1999) found population peaks independent from control measures. Recommendations of the WSL again are based on the knowledge that standing trees are attacked a half to one year earlier on cleared windthrow sites compared to damage areas without intervention (Figure 2-11).

It is commonly argued that bark beetle outbreaks tend to initiate from smaller gaps in the forest, where breeding material deteriorates less rapidly than in large openings with high solar irradiation and evaporation (e.g. Forster and Meier 2010). More recent Scandinavian studies however either point to less attractiveness of localised windthrows and the preference for large wind-thrown areas (Eriksson et al. 2005) or to no relationship between gap area and attack (Schröder 2010). Gap area in general seems to be less significant for the intensity of infestation than the specific characteristics of wind-felled trees. Bark beetles prefer broken and thrown trees for colonisation rather than standing tree fragments and more frequently attack stems of high diameter (Wermelinger 2004). Göthlin et al. (2000) found twice as strong infestation of trunks with diameters larger than 46 cm than of smaller stems. Colonisation was found positively correlated with the mean diameter of windfelled trees or basal area of living spruce trees (Schröder 2010), and to be low given small diameter classes even when the number of wind-felled spruce trees is high on damage areas (Eriksson et al. 2005).

Considered that a total clearing of large storm gaps first is often impractical after vast storm events, the recommendations of Swiss WSL to prioritise according to forest structure and function, damage extent as well as outbreak situation seem to be most reasonable. Apart from uncontrolled, protected areas, Forster and Meier (2010) suggest non-intervention also in forests with protective function (e.g. for soil, water, dwellings) given the proportion of spruce is lower than 0.25-0.33%. If the removal of at least 80% of all wind-felled trees in spruce-dominated stands is feasible within one year in lowland or within two years in mountain areas, broken trees are to be processed before thrown ones. In case not all clearings are manageable in time, WSL recommend the preferential clearing of scattered windthrows as well as damage areas smaller than two hectares in forests of high priority function. Supplementary stems adjacent to intact forest stands as well as high-dimension trunks might be cleared first given large damage areas. Already infested standing or lying trees of course are to be removed, best before the emergence of the first generation in order to reduce local attack density.

Silvicultural options and population dynamics

Predisposing stand factors

Apart from climatic conditions and the incidence of extreme, abiotic disturbance events, distribution of Ips typographus is limited mainly by the abundance of its prime host tree, Picea abies. Managed forests dominated by Norway spruce with a high canopy cover offer ideal developmental conditions, presenting high availability of breeding material in combination with
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Low population densities of antagonists (e.g. due to the lack of herbs or shrub layers as important habitats). Silvicultural measures promoting a proportion of deciduous or conifer trees other than Norway spruce of at least 50% can help to markedly reduce the future incidence of bark beetle infestations (Witteke 1984, Moor 1987, Meusberger 1995).

Susceptibility of Norway spruce stands to attack by *I. typographus* is significantly and positively correlated to stand age. The bark of younger trees is too thin for development of this relatively large beetle, so that damage probability clearly increases in stands older than 60 years and attack is most likely for age classes above 100 years. Especially mature and over-mature forests of reduced vitality are disposed to dieback caused by bark beetle infestation (Willmann 1951, Merker 1952, Wild 1953, Schwerdtfeger 1955, Bakke 1983, Vitén 1984, Moor 1987, Schopf and Köhler 1995), a fact that helps explain the vast damage areas in national parks.

Unlike secondary bark beetle species, which attack weakened trees, *Ips typographus* is able to colonise relatively vital spruce stands. The preference of *I. typographus* for trees of high tree vigour index (according to Münster-Swendsen, 1987 = percentage of the youngest annual ring in relation to the total area of the water-conducting sapwood, used as measure of the vitality of single trees) could be shown by several authors (Mulock and Christiansen 1986, Baier 1993, Lindenthal and Führer 1993). While the tree vigour index proved persistently low for non-infested tree individuals, highly attractive trees featured high tree vigour index values but mostly suffered from acute biotic or abiotic damages.

Stand structural features and management measures influencing risk

Generally, stand structural features leading to increased insolation and temperature conditions may promote bark beetle development rates and survival. Infestation is commonly initiated at forest edges, margins of clear-cuts, small clearings or forest roads (e.g. Grodzki et al. 2003). Uniform age and stand structure, low crown cover or stand density and high gap abundance further enhance stand susceptibility to attack. As a consequence, the risk of bark beetle outbreaks can potentially be reduced by the modification of stand characteristics through silviculture, at least in managed forests.

Management strategies aimed at the establishment of resistant stands with adequate species composition and stand density as well as the implementation of sanitary measures are valued as efficient ways to avoid the build-up of high population densities of *Ips typographus* (Neubeker and Hodges 1983, Berryman 1988, Marchetti et al. 1999). Besides salvage or “clean management” including the clearing of wind-throws and sanitation felling, prophylactic measures should also involve optimised thinning and harvesting regimes. Optimised management has to find the balance between too frequent and intense thinning resulting in opened stands with stressed and susceptible remaining trees on the one hand, and too dense stocking of trees leading to reduced stand vitality due to competition on the other hand (Merker 1952, Moor 1987, Sanders 1987, Schopf and Köhler 1995, Jakus 1998). Intensity of forest management of course varies with financial and human resources available and with timber prices. Besides the economic situation, demands on forest functions, ecological considerations (e.g. promoting dead wood, generating a more natural stand composition or enhancing biodiversity in managed forests) and the primary management goal impact decision-making and the scope of action. As a matter of fact, mass outbreaks of the spruce bark beetle are frequent in inaccessible stands or nature reserves, where
the removal of wind-thrown or otherwise killed and damaged trees from the forest is not possible or restricted (Eidmann 1992).

Forest dynamics during and after outbreaks

Common characteristics of Central and North-European spruce forests are high proportion of Norway spruce and homogeneous forest structure. Uniformity and single-layer is typical, not only in managed spruce stands due to the establishment of age-classes, but also in more natural spruce forest ecosystems (Mayer 1974). Disturbances such as bark beetle eruptions causing large-scale mortality within a comparable short time frame therefore constitute a massive disruption of ecosystem function and structure (see Chapter 3.2). As the majority of European spruce forests are of economic interest, human intervention, e.g. by removal of attacked trees and dead wood or by artificial regeneration, anticipates natural forest dynamics after disturbance events. While the main goal in managed forests is to minimise damage and to fight destructive agents, such as bark beetles, disturbances are regarded as a constitutive element of forest ecosystems under nature protection. Disturbances are important for the initiation of forest succession and regeneration; yet, beyond the theoretical background, there are only limited examples available that illustrate these processes in European spruce stands unaffected by management measures. The Bavarian national park may serve as unique case study, where outbreaks of I. typographus and almost total mortality of mature montane spruce stands have been tolerated, and which today illustrates the natural restoration power of a spruce forest ecosystem.

Regeneration of Norway spruce at mountainous sites of the Bavarian national park did not follow successional processes, as is described by Heurich (2001). Before the calamity in 1991, forests had been composed of 98% Picea abies, 1.2% Fagus sylvatica as well as of 0.6% Sorbus aucuparia together with Acer pseudoplatanus. Forest inventory after massive spruce dieback in 2005 again showed a dominance of Norway spruce with 89.0%, followed by 7.9% rowan and 1.2% beech among seedlings larger than 20cm (Heurich and Rall 2006). Norway spruce was especially able to regenerate at sites where mature spruce forest had already died in the earlier phase of the outbreak (Heurich 2001). Survival of seedlings was not only due to improved light and temperature conditions, but also to an intensive spruce mast year. The massive dieback of the previously 280-300 year old, uniform stands led to a sudden increase of solar irradiation on the forest floor, allowing for an accelerated mineralisation of the organic layer thick of needles fallen down from the dead trees. Accumulation of nutrients from spruce bark and branches also enhanced young tree development. Dead wood left at the sites prevented forest soil from dessication and seedlings from damage by animals and snow. Especially Norway spruce regenerates well on mouldered trunks. Consequently, the density of regenerated plants was more than four times higher in 2005 than at times of closed forest stands in 1991 (Heurich and Rall 2006; Figure 2-12).
The positive restoration of mountain forest stands in the Bavarian forest national park within few years lets expect similar processes in related forest ecosystems, such as the boreal conifer forest, and backs up the protection of natural processes in other forest reserves. Bark beetle infestation can significantly contribute to the transformation of secondary spruce stands into habitat-adapted forests, as is explicitly desired in more recently founded reserve areas\textsuperscript{12}. Spots of bark beetle attack have for instance been recorded at previously managed sites of the Austrian biotope protective forest ‘Wilderness Dürrenstein’ since 2003 (Figure 2-13).

Although the 50-265 year old Norway spruce stands were predisposed to infestation by an avalanche in March 2009, attacked areas till now have remained localised and scattered (Blackwell 2011). In case this trend will persist, forest stands can regenerate and transform gradually in small gaps.

2.3.1.4 **FUTURE TRENDS UNDER CLIMATE CHANGE**

The reproductive potential of *Ips typographus* is strongly related to the number of fully terminated generations within a single year (voltinism), which to a major part depends on spring and summer temperature conditions. Today, the development of two to three beetle generations is possible at warm, lowland forest sites, while one to two beetle generations are observed in the mountainous areas (Baier et al. 2007) of Central Europe. Bivoltine beetle populations are commonly observed as far as Denmark, but not yet prevalent in more northern latitudes. The projected rise of temperature will however support an increase in generation numbers also in Norway or Sweden, so that regional warming can potentially lead to the northward movement of bivoltinism by some 600 kilometres (Lange et al. 2006, Jönsson et al. 2007, 2009).

In central Europe, voltinism is also expected to increase. Hlásny and Turcány (2008) predict a doubling of areas in Slovakia where the spruce bark beetle can fully develop two generations by 2075 and an increased incidence of three generations by 2045. A shift from uni- to bivoltine populations has been observed for many Central European mountain sites, and is here illustrated for the Austrian area ‘Wildalpen’ (Figure 2-14). Rising frequencies of drought periods, which already today commonly promote water stress in Norway spruce stands, will further dispose European forests to bark beetle attack (Bakke 1983, Faccoli 2009, Ogris and Jurc 2010).
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Figure 2-14 Proportion of sites with the potential of none, 1 or 2 fully developed generations of Ips typographus in the area ‘Wildalpen’ (Styria, Austria, 400-1,500m a.s.l) related to stand area damaged by wind throws, avalanches and bark beetle infestations (Bakke 1983, Faccoli 2009, Ogris and Jurc 2010)

2.3.1.5 MANAGEMENT

- Measures to control damage by bark beetles
  - Instruments for monitoring and control

Since damage caused by *I. typographus* is often apparent once the insects have already killed and left the trees, extensive and efficient surveillance is the only way to protect susceptible forests. A range of control and monitoring practices are therefore traditionally implemented in the management of European spruce forests, involving silvicultural and trapping out measures, the application of chemical insecticides and methods of surveying (Box 2-6). According to the BAWBILT database on damage and control (Grégoire and Evans 2004), the procedures most commonly adopted to anticipate bark beetle outbreaks are forest sanitation, clear felling, and selective thinning, the use of pheromone-baited traps for monitoring, trap logs or trees baited or impregnated with insecticides for control. Monitoring is also frequently done by visual surveys; remote sensing is however still underrepresented in MS and, interestingly also damage prophylaxis by silvicultural choice.
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**Box 2-6 Monitoring of population density**

- **Are trap catches representative?**

Although pheromone traps are widely-used for monitoring, trap-catch numbers are to be interpreted with caution when evaluating local population dynamics. The numbers of caught individuals very much depend on trap position, attractiveness of standing trees, weather conditions, quality of pheromone lure, and frequency of trap control. Further, dispersion rates of bark beetles are not implicitly related to attack density (Botterweg, 1982; Hurling, 2002). On a regional scale, Okland and Björnstad (2003, 2006) found positive correlations between trap catches per year and specific characteristics predisposing forests to infestation (e.g. high forest productivity, probability of drought stress, volume of wind-felled trees) by comparing time series patterns. Faccoli and Stergulc (2004, 2005) tried to define “risk thresholds” based on the high correlation of spring trap catches and bark beetle damage at sites of the Italian Alps. Yet, to assess trends of population development for a given area, it is recommended to consider local empirical values in combination with host tree availability.

- **Need for direct evidence**

Eventually, the monitoring of damage areas by visual or aerial surveys is crucial to evaluate trends in population density. Activities including the search for fresh entrance holes and boring dust, or at least the accurate recording of damage volume of course, requires a lot of time and human resources, but is probably the most efficient way to evaluate the actual risk of outbreaks.

- **Risk Assessment**

According to BAWBILT database, more than one third of the MS implement models for rating the risk of spruce bark beetle outbreaks at least on regional scale. For instance, Fahse and Heurich (2003) developed a model with regard to the outbreak situation in the NP Bayerischer Wald (AT). The model is based on two key parameters, the initial beetle numbers and as a proxy for the spread of attacks, the probabilities for finding and removing attacked trees in buffer zones. However, infestation dynamics often seem to follow no rule (e.g. Hurling 2002) and their understanding still requires further research (Wermelinger 2004).

Thus, for these models to be useful tools for decision-making, they should satisfyingly describe the causalities and triggers of epidemics. Such models have successfully been defined for *Dendroctonus ponderosae*, where the risk is commonly defined as a combination of stand susceptibility and beetle pressure (e.g. Dymond et al. 2006, Robertson et al. 2008). Similar models would be needed for *Ips typographus*.

“Susceptibility-models” are based on the hypothesis that dispersal patterns are mainly environmentally determined in the native phase (comp. Berryman 1986, Logan et al. 1998, Wulder et al. 2006). This means that site and forest stand-immanent hazards can be identified, as well as options for prophylactic forest management. A predisposition assessment system following this approach is available for *Ips typographus* (comp. Netherer 2003, “PAS” by Netherer and Nopp-Mayr 2005) and served as a basis for the evaluation table below (Table 2-6). Background information on the factors predisposing forest stands to outbreaks is given in section 2.2.
Disturbances of EU forests caused by biotic agents

Table 2-6 Assessment of the predisposition to bark beetle infestation

<table>
<thead>
<tr>
<th>Age (yrs.)</th>
<th>Predisposition</th>
</tr>
</thead>
<tbody>
<tr>
<td>&gt; 100 (old, over-mature forest)</td>
<td></td>
</tr>
<tr>
<td>80-100 (mature to over-mature forest)</td>
<td></td>
</tr>
<tr>
<td>60-79 (mature forest)</td>
<td></td>
</tr>
<tr>
<td>40-59 (pole wood)</td>
<td></td>
</tr>
<tr>
<td>&lt;40 (young wood, thicket, young pole wood)</td>
<td></td>
</tr>
<tr>
<td>% Spruce</td>
<td>&lt;10 10-24 25-49 50-69 &gt;=70</td>
</tr>
</tbody>
</table>

Table 2-7 indicates the level according to which different environmental traits expose forests to the risk of bark beetle infestation. Three major risk categories are distinguished, and within them, colour intensity further reflects the level of risk. When the risk is high and well-documented, a high colour intensity is used. When the risk is low or poorly documented (hence unreliable), a lighter colour intensity is used. The table thus highlights specific factor constellations, for identifying areas differentially at risk and highlighting the parameters which predominantly contribute to a hazard situation.

Thus, a 40 year old mixed forest with approximately 50% Norway spruce and composed of not too densely planted even-aged, fast growing trees would only be moderately predisposed to bark beetle infestation. However, if in this same forest, some stands are not accessible and located on dry, drought-prone soils, or show a high-risk of pre-damages, then the probability of bark beetle infestation would increase, so that monitoring measures are required. In general, the more medium or high risk factors apply to a forested area or a specific stand, the higher the probability of severe bark beetle infestation. Temperature is the main factor determining spatial variations in potentials of beetle development. Phenological models such as PHENIPS (Baier et al. 2007) explicitly consider the strong effects of regional topography, local air temperature and sun irradiation on microclimatic bark temperature conditions. Allowing for a prediction of bark beetles’ developmental stage, swarming times, start of infestation and potential number of developed generations on local to regional scale, the model may serve as a further, significant component of risk analysis, especially in sensitive areas such as national parks. The model is available in the framework of an online monitoring service, presenting actual bark beetle development based on climate data provided by local weather stations (e.g. Schopf et al. 2010). The monitoring system supports control and timely intervention adjusted to current swarming and attack situations.
### Table 2-7 Environmental traits exposing forests to low, medium or high probability of bark beetle infestation. Darker background colour reflects higher importance (weighting) of the different indicators. Lighter colour reflects smaller importance or smaller reliability.

<table>
<thead>
<tr>
<th>Predisposing Factors / Silviculture</th>
<th>Stand factors / Silviculture</th>
<th>Exploitation / Phytosanitary measures</th>
<th>Site characteristics</th>
<th>Climate</th>
</tr>
</thead>
</table>
| % Norway spruce x Stand age (years)* | young wood, thickets, pole woods (<60 yrs.) with < 25% Norway spruce | high accessibility, "Clean forestry" - timely removal of wind-thrown and infested trees (salvage cutting), no storage of harvested wood in the forest | Topography | Temperature (mean air temperature for the period April to October**)

   5°C: no generation development possible; 
   5°C to <7°C: uni-voltine populations at maximum development of 1 generation + 1 sister brood; 
   7°C to <9°C: possible development of 1 generation + 1 sister brood; 
   9°C to <11°C: bi-voltine populations; 
   11°C to <13°C: 2 generations + 1 sister brood possible; 
   13°C and higher: multi-voltine populations (>=3) |

<table>
<thead>
<tr>
<th>Site structure</th>
<th>Multi-aged and multi-layered forest stands</th>
<th></th>
<th>Potential solar irradiation</th>
<th>Precipitation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Autochthonous tree species</td>
<td></td>
<td></td>
<td>relative low exposure to solar irradiation</td>
<td>2750mm (March-July); 600-800mm = optimal growth conditions for N.spruce in the alpine area</td>
</tr>
<tr>
<td>Stand density</td>
<td>closed forest stands - optimised stand density (regarding competition for light and water)</td>
<td></td>
<td>moderate exposure to solar irradiation</td>
<td>&lt;360mm (April-October); 600mm (annual precipitation) = threshold for natural range of Norway spruce</td>
</tr>
<tr>
<td>Stand vitality and tree sociology</td>
<td></td>
<td></td>
<td>relative high exposure to solar irradiation</td>
<td>Drought periods</td>
</tr>
<tr>
<td>Stand factors / Silviculture</td>
<td>mature to over-mature forest stands with medium to high proportion of spruce (see table)</td>
<td></td>
<td>average high accessibility, &quot;Clean forestry&quot; - timely removal of wind-thrown and infested trees</td>
<td>no drought periods or low incidence of drought, Norway spruce stands sufficiently water supplied</td>
</tr>
<tr>
<td>% Norway spruce x Stand age (years)*</td>
<td>young stands (&lt;60 yrs.) with &gt;= 25% Norway spruce, older stands below 50% spruce (see table)</td>
<td></td>
<td>dry (xeric), but also very wet, waterlogged sites</td>
<td>moderate incidence of drought</td>
</tr>
<tr>
<td>Site water supply</td>
<td>very high stand density (no thinning regime, competition) or slightly opened stands</td>
<td></td>
<td>gleysoils or soils with distinct stagnic properties</td>
<td>high incidence of drought periods leading to critical thresholds of stand water supply</td>
</tr>
<tr>
<td>Soil type</td>
<td></td>
<td></td>
<td>low soil depth, high proportion of skeleton</td>
<td></td>
</tr>
<tr>
<td>Soil depth and skeleton</td>
<td></td>
<td></td>
<td>Site index</td>
<td></td>
</tr>
<tr>
<td>Site index</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Climate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Note:** The table provides a comprehensive overview of environmental traits that can lead to bark beetle infestation in forests. The significance of these traits is indicated by the background colour, with darker shades reflecting higher importance. The table also highlights measures that can mitigate these risks, such as targeted harvesting practices and improved site management.
## Recommendations for bark beetle management

The choice of adequate management strategies to hinder uncontrolled bark beetle outbreaks is not only a question of risk, but also depends on management goals and limitations. Forestry primarily aimed at wood production potentially produces stands highly prone to pre-damages such as storm throw and susceptible to bark beetle infestation, but conventional management of these forests permits the implementation of the whole range of silvicultural options, control and sanitation measures. In contrast, the management of protected forest areas, such as national parks, requires balancing human intervention and interests of nature protection.

Independently of the specific forest management aims, a scrupulous analysis of risks, e.g. on the basis of suggested evaluation and monitoring tools, is strongly recommended as a first step when searching for management solutions. Once initiated, the possible progress of the epidemics is difficult to predict, ranging from stagnation due to natural regulation and limited beetle habitat over repeated flare-ups of outbreaks to epidemic conditions over years. Thresholds determining quality and intensity of intervention are therefore to be defined under consideration of the local hazard situation, possible climatic trends and human demands on the forest.

In the following tables, guidelines are offered that are especially targeted at bark beetle (risk) management in protected forest areas, which encompass strict reserves without management and outer zones where specific intervention measures are requested (Table 2-8). These areas

| Pre-damages | Wind throw | Low predisposition of forest stands to wind damage, few to none actual damage areas | Moderate probability of wind damage, limited areas of actual damage | High probability of wind damage, large areas of actual damage, high proportion of forest gaps and (newly created) stand edges |
| Snow breakage | Low predisposition of forest stands to snow breakage, few to none actual damage areas | Moderate probability of snow breakage, limited areas of actual damage | High probability of snow breakage, large areas of actual damage |
| Other damages | | | High probability of actual presence of fire, avalanches, rock fall or other factors leading to stem or root damages |
| Population dynamics | Abundance (e.g. by number of infested trees or amount of salvage cuttings) | Low, endemic population density | At present low population density, but risk of population growth initiated by triggering factors (need of monitoring) | Increasing population density (initiated mass outbreak), persistent high beetle abundance |
| Situation in adjacent forest stands | Population density in adjacent stands | Low level of infestation and low population density (e.g. monitored by flight barrier traps) in adjacent forest stands | Moderate, controlled level of infestation in adjacent forest stands | High level of infestation and high population density (e.g. monitored by flight barrier traps) in adjacent forest stands |
| Predisposition of adjacent stands to infestation | Low probability of mass outbreaks in adjacent forest stands | Moderate probability of mass outbreaks in adjacent forest stands | High probability of mass outbreaks in adjacent forest stands |

* see % Norway spruce x stand age – evaluation table
**values derived from PHENIPS (Baier et al. 2007)
have high relevance as buffer between zones of natural regulation and adjacent, conventionally managed forest stands. Although of high personnel effort and expense, intensive monitoring and control activities are crucial in the buffer stands, both to preserve the core protected areas and to secure the interests of neighbouring forest managers (Table 2-9).

Table 2-8 Guidelines for bark beetle management

<table>
<thead>
<tr>
<th>FOREST FUNCTION AND MANAGEMENT</th>
<th>MEASURES</th>
</tr>
</thead>
<tbody>
<tr>
<td>Managed forests</td>
<td>Close-to-nature forestry recommended; Monitoring and sanitation measures obligatory</td>
</tr>
<tr>
<td>Protected forests / National parks: Strict reserves, core areas (encompassing &gt;=70% of total forest area)</td>
<td>No intervention, independent from local risk of outbreaks; potential conversion of Norway spruce stands into mixed or deciduous forests tolerated</td>
</tr>
<tr>
<td>Areas outside strict reserves (encompassing &lt;=30% of total area)</td>
<td>Definition of buffer zones of defined width according to risk situation in core zones and adjacent, managed forest stands (minimum extent 500m!) where intensive monitoring and control measures are implemented</td>
</tr>
</tbody>
</table>

Table 2-9 Guidelines for bark beetle management in buffer zones of protected forest areas

<table>
<thead>
<tr>
<th>MONITORING</th>
<th>THRESHOLDS</th>
<th>MEASURES (in buffer zones)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Visual or aerial surveys for Wind throw, Snow damage, other predisposing damages</td>
<td>Single trees to whole damage areas</td>
<td>Clearing of damaged wood, either removal from the forest or decortication</td>
</tr>
<tr>
<td>Bark beetle infestation: Trees attacked the previous year</td>
<td>Red to brown crown colour</td>
<td>Harvesting of infested standing trees, removal from the forest or decortication</td>
</tr>
<tr>
<td>Recently attacked trees</td>
<td>Fresh entrance holes, dust</td>
<td></td>
</tr>
<tr>
<td>Population density –trends: Pheromone traps</td>
<td>Catch numbers increasing or high</td>
<td>Intensive monitoring for newly infested trees and of bark beetle phenology (e.g. by online monitoring systems based on PHENIPS)</td>
</tr>
<tr>
<td>Intensity of infestation during the recent years</td>
<td>Volume of damaged trees increasing or high</td>
<td></td>
</tr>
<tr>
<td>Bark beetle phenology: Online monitoring</td>
<td>Onset of swarming (end of March, April)</td>
<td>Timely deposition of pheromone traps and trap trees</td>
</tr>
<tr>
<td>Modelling of actual bark beetle development (including sister broods) according to local temperature conditions and of potential number of generations</td>
<td>Start of infestation (April, May)</td>
<td>Visual monitoring for recently attacked trees and trap logs (boring dust!) / Removal of attacked trees</td>
</tr>
<tr>
<td>P...parental beetles</td>
<td>P 2 wpi Finished mating and egg deposition</td>
<td>Decortication of trap trees (danger of leaving parental beetles to establish sister broods!)</td>
</tr>
<tr>
<td>F1...first filial generation</td>
<td>P 2-3 wpi Swarming and infestation by parental beetles</td>
<td>Viral monitoring for recently attacked trees and trap logs (boring dust!) / Removal of attacked trees</td>
</tr>
<tr>
<td>F2...second filial generation</td>
<td>F1 5-6 wpi Completed first generation</td>
<td>Timely deposition of pheromone traps and new trap trees for controlling a potential second brood</td>
</tr>
<tr>
<td>wp1...weeks past infestation (depending on temperature conditions and therefore only reference points!)</td>
<td>F1 6-8 wpi Swarming and infestation by filial beetles</td>
<td>Visual monitoring for recently attacked trees and trap logs (boring dust!) / Removal of attacked trees</td>
</tr>
<tr>
<td>wp2i...weeks past second infestation</td>
<td>P 2 wp2i Finished mating and egg deposition</td>
<td>Decortication of trap trees (danger of leaving parental beetles to establish 2nd sister broods!)</td>
</tr>
<tr>
<td>F2...second filial generation</td>
<td>P 2-3 wp2i Swarming and infestation by parental beetles</td>
<td>Visual monitoring for recently attacked trees and trap logs (boring dust!) / Removal of attacked trees</td>
</tr>
<tr>
<td>S 5-6 wp2i Completed second generation</td>
<td>F2 6-14 wp2i Beetles swarming to overwintering sites</td>
<td>Visual monitoring for recently attacked trees / Removal of attacked trees</td>
</tr>
</tbody>
</table>
### 2.3.2 Anoplophora chinensis

#### DISTRIBUTION AND BIOLOGY

<table>
<thead>
<tr>
<th>Species</th>
<th>Anoplophora chinensis (Citrus longhorned beetle)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Insecta</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Alien</td>
</tr>
<tr>
<td>Distribution</td>
<td>Infestation detected mainly in IT where it spread (in Rome, Milan: mainly in West and North West of the city, and in 30 municipalities North West, West, and South of Milan). Isolated cases of A. chinensis were detected in FR (EPPO, 2006; declared eradicated since 2006), in DE in 2008 (but now eradicated) and in NL (Westland and Boskoop, Van der Gaag et al. 2010).</td>
</tr>
<tr>
<td>Spatial and temporal scale of impact</td>
<td>The infestation and spread dynamics can be extremely variable, depending on the location: 10 years to several decades; 50 m to several km of radius. Scale of impact is directly tied to the lag between the accidental introduction(s) and the first eradication efforts; impact increases exponentially with lag.</td>
</tr>
<tr>
<td>Interesting lesson learnt for management</td>
<td>Early eradication efforts are the key point for a successful management. Awareness-raising of the importance of early detection, preventive cutting in 250 m radius around infested trees have proven key elements for eradicating the species. The control measure should last at least 6 years (3 life cycles). It can be further modulated depending on the characteristics of the infestation.</td>
</tr>
</tbody>
</table>

*Anoplophora chinensis* is a polyphagous beetle native to Eastern Asia. Originally from China, Korea, and Japan, the beetle it is also found in Taiwan, the Philippines, Indonesia, Malaysia, Indonesia, Malaysia, Malaysia.

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13 Also called *Anoplophora malasiaca*. However, it is noted that *A. malasiaca* was recognised as a junior synonym of *A. chinensis* by Lingafelter & Hoebke in 2002, hence it can be confusing reading literature pre-dating 2002 that differentiates between *A. chinensis* and *A. malasiaca*.
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Myanmar, and Vietnam. It is well-adapted to European temperate climates, but it can also survive in cooler conditions (e.g. Netherlands). The female beetles lay an average of 194 eggs within the bark tissues near the base of the trunk or on superficial exposed roots (Adachi 1988). The young larvae hatch over a period of 10 days to two weeks (Adachi 1994).

Later in its development the larvae bore tunnels in the wood till the pupation, extruding frass. The amount of frass extruded may depend on the species and the diameter of the plant part being attacked, but also on larval stage, and local climate.

*A. chinensis* needs 1–3 years, depending on the climate to complete its life cycle, both in its native area and in newly invaded areas (Adachi 1994, Haack et al. 2010). This is because larval development is relatively long for this beetle and can last one or two years both in Europe and in its native range, depending in particular on climate (Adachi 1994). At the end of the pupal stage, still two weeks are needed to have a fully mature and active beetle (Figure 2-15) feeding on leaves, petioles, and bark of twigs and capable of mating. Most activity occurs during the day.

It is only recently that *Anoplophora chinensis* has been introduced in Europe, through alien plant trade. A number of interceptions in recent years show that the risk of *A. chinensis* is quite high through import of Acer spp. and others host plants from infested countries (Van der Gaag et al. 2008, Haack et al. 2010). It was first found in the UK (Cooter 2000) and in Italy in 2000 (Colombo and Limonta 2001). It was later found in France in 2003, in Germany in 2008, and in the Netherlands in 2007 (Hérard et al. 2006, Maspero et al. 2007, and Van der Gaag et al. 2008). In all cases the infestation presumably originated from imported bonsais. Today however, the infestations in France and Germany have been declared eradicated (Haack et al. 2010).

Figure 2-15 Anoplophora chinensis

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14 Proceedings of the Second Meeting of Forest Protection and Forest Phytosanitary Specialists, 2007, Austria
2.3.2.2 **Damage Significance**

Unlike many other native wood boring beetles that primarily attack dead trees, *A. chinensis* attacks healthy trees and shrubs of any age, as long as the stems are larger than 2 cm. It is thus common to find citrus longhorn beetle on pot maple trees and bonsais imported from Eastern Asia, with trunks of about 3 cm in diameter only. In addition, *A. chinensis* can be found on a wide range of host plants in Europe, having been listed or reported for more than 70 taxa (see Lingafelter and Hoebeke 2002 for a list of known host genera and EPPO 2010). Initially, the adult feed on the plant tissues, but the most important damage is associated to larval wood boring activity which occurs, of course, later on after the ovoposition step.

The beetle is considered a serious risk since it has the potential to invade most types of habitats, from urban to rural and forest landscapes, as well as orchards. Therefore, new infestations might be expected in the future especially near locations with a history of plant imports (in particular *Acer palmatum*) from China or Japan. These locations pose the highest risk since high numbers of infested plants on the same site increases the probability of successful mating and establishment in the surrounding areas (EPPO 2010). Due to its potential to create serious damages to forests (i.e. based on the Italian experience), *A. chinensis* has been listed as a quarantine pest in the EU\(^{15}\), but also in Canada and in the USA\(^{16}\). However, so far the beetle has not spread beyond the plant nurseries (the majority of cases) or urban parks and semi-urban areas (2 cases). In the long term environmental and economical impacts are expected to be high for EU (Baker and Eyre 2006). The potential future impacts are the following (Van der Gaag et al. 2008):

- loss of trees and shrubs in urban areas, on country sites and possibly also in forests
- loss in biodiversity
- crop losses and loss of export markets for tree nurseries
- potential yield losses in fruit orchards (Citrus, Malus, Pyrus, etc.)\(^{17}\)

However, the risk of spread is limited by the relatively reduced ability of the beetle to actively disperse. As observed in Italy, at early stages of infestation, females generally oviposite on neighbouring trees and move only at a later stage of the infestation, when no more space for ovoposition is available (EPPO 2010). In case of excessive larval density, a competition between larvae is established and a phenomenon of cannibalism can occur. For this reason, we can say that the infestation dynamics are not very rapid and that they are strongly dependent on host plant densities. However, the maximum radius of active dispersal is currently debated and some scientists have observed a dispersal of only few tens of meters (Matteo Maspero, personal communication) while others reported dispersal over a radius of several hundreds of meters (EPPO 2010).

\(^{15}\) www.eppo.org/QUARANTINE/quarantine.htm [Accessed online the 24th of May 2011]

\(^{16}\) Canada and USA are concerned about possible future citrus longhorn beetle introductions, and are proactive to avoid them, but are not currently affected by this pest. In the past (2001), in the USA at Tukwila (WA) citrus longhorn beetle adults escaped from a nursery importing bonsais. Although no infestation was found, an aggressive campaign of eradication was made to eliminate susceptible trees in the neighborhood (F. Hérard personal communication).

\(^{17}\) Never observed in EU
Passive spread, through, for instance, transport of infested plants by trade is more critical. The major contamination comes from import of host plants from China (e.g. it has been assessed that 4 millions of maple plants were imported in EU in 2005-2007). Indeed the risk is limited to imported plants from China and not from EU infested areas (Beniamino Cavagna, personal communication).

A secondary pathway, which is well known for *Anoplophora glabripennis*, but less relevant for *A. chinensis* is the import of solid wood (including wood products, wood packaging material) from Asian infested countries (Franck Hérard et al. 2005). However, one interception of *A. chinensis* in wood packaging material has been reported from Germany in June 2007 (Europhyt 2008).

Finally, a third pathway is the trade of wood chips from infested countries. Infested wood, chipped into pieces larger than 1.5 cm can enable larvae of *Anoplophora spp.* to survive. Chipping infested wood into smaller pieces is an effective way to eliminate *A. glabripennis* (USDA 2008), but there is no data for *A. chinensis*. In any case, wood chips import is not reported from infested areas.

The destructive potential of this beetle lies in the eruptive increase of larvae that weaken the host plant (Figure 2-16). The larval boring activity leads to a structural weakness of the trees that interferes with the transport of water and nutrients to the leaves and contributes to make the trees more vulnerable to secondary pathogens and can in extreme cases lead to tree death. This results in a rapid tree decline and death within a few years. Older trees with bigger trunks are more resistant to damage. For instance in Italy, during the mass outbreak in Lombardy, *A. chinensis* attacked about 10 000 plants in the provinces of Milan and Varese (Cavagna 2011) (semi-urban areas).

The temporal dynamics of outbreaks depend very much on the location. However, it is possible to affirm that in most of Europe, after the introduction, several decades can pass before the pest have a large impacts due to a typical life-cycle duration of 2-3 years and to its limited capacity to spread by natural means. Moreover, according to experiences in Lombardy (Italy) an interval of 5-10 years can occur from the infestation to the tree death (e.g. due to secondary infections). It is worth noticing that in Southern Europe, the pest may already have a large impact within 10 years of its introduction having in this area a life cycle of 1-2 years (Europhyt 2008).

The total cost of *A. chinensis* damage in Europe has been estimated at 0.32 million EUR/yr (Kettunen et al. 2009). This data has been obtained performing an extrapolation from to the cost in the area of impact to the EU known range. However, this methodology does not take into account key environmental factors which can strongly influence the importance of damages, such as the host densities, which of course varies depending on the considered area. For instance in a site of high host density, such as the Lombardy region (Italy), the overall cost of *A. chinensis* damage has been estimated to 10.3 million EUR according to the local authorities (Matteo Maspero, personal communication). In the Netherlands, the eradication cost for the period 2007-2008 was estimated at 1.25 million EUR (Haack et al. 2010).
2.3.2.3 **FUTURE TRENDS UNDER CLIMATE CHANGE**

The reproductive potential of *A. chinensis* is related to the length of the breeding cycle and survival of larvae, both of which are strongly related to climatic factors. As for the establishment, we know now that this species can survive and breed in southern Europe, especially in Italy where this species is considered as established by DAISIE since 1994\(^\text{19}\). The relationship among the degrees day and the life cycle steps is not completely understood for *A. chinensis* (F. Hérard personal communication). However, at lower temperatures, reproduction, oviposition, hatching as well as overall development and activity probably decreases. From experimental observations and degree day calculations, it appears that *A. chinensis*, similar to other longhorned beetles, has a longer life cycle in temperate/mediterranean regions of EU compared to other areas (e.g. UK; Baker and Eyre 2006, Van der Gaag et al. 2008). For instance in England, an adult beetle was observed to emerge from a maple tree imported from China via Netherlands three years before.

This means that the development of *A. chinensis* took 3 years in UK climatic conditions, instead of the 1-2 years observed in the native range and in Italy (EPPO 2010\(^\text{18}\)). To date there is no specific research on the *A. chinensis* trends under climate change. The expected trend is that the northern limit for survival will move North, and ongoing degree-days studies should help specify the extent to which this limit may shift (Franck Hérard, personal communication).

2.3.2.4 **MANAGEMENT**

It is important to stress that to date, *A. chinensis* in Europe has not been detected in natural environments but only on nurseries or urban parks. As a consequence all the management strategies presented hereafter have been applied in such kind of environments and not in forests.

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\(^{18}\) EPPO gallery: photos.eppo.org/index.php/image/720-anolch-03-1024x768/hits/51-anoplophora-chinensis-anolch-

\(^{19}\) Information available in DAISIE’s factsheet: [www.europe-aliens.org/speciesInRegion.do?speciesInRegionId=51319;104839](http://www.europe-aliens.org/speciesInRegion.do?speciesInRegionId=51319;104839) [accessed 25/11/2011]
Panorama of the current risks caused by biotic agents to the delivery of goods and services

**Monitoring**

Since the damage caused by *A. chinensis* is often clearly visible at late stages of infestation when the damage is irreversible, efficient surveillance is critical to detect the beetle as early as possible. Slow or incomplete monitoring or late host tree destruction may lead to outbreak conditions that become difficult to manage, so as was the case in Lombardy (Italy).

The monitoring methods include visual inspection of the trees consisting in looking for:

- exit holes
- larvae (not visible without cutting the plant)
- larval galleries (not visible without cutting the plant)
- frass and debris
- T shaped cracks in the bark
- signs of feeding on twigs (dissection needed)

Such visual inspection can be done before and after the uprooting and dissection (trunk can be cut at different heights, or chipped and stored for long term observations in a safe place in order to avoid eventual uncontrolled spread) of the three. Infested plants dissection of plants can give valuable information about the extent of an infestation (e.g. number of larvae and number and age of exit holes). More sensitive methods, such as PCR\(^{20}\) for larval markers are also possible. New and promising non destructive methods, such as acoustic detection and the use of trained dogs are under development (Farr and Chesmore 2007, EPPO 2010).

**Measures to control damage**

Due to the growing number of interceptions in the EU (1 in 1980 and 72 in 2008 when counting both interceptions at ports of entry and interceptions post entry), emergency measures against *A. chinensis* have been in place since November 2008 (Commission Decision 2008 / 840 /EC\(^{21}\) which ban import of Acer species from China till April 2012). Similarly, in Canada, import of living (Acer spp.) trees from China is prohibited\(^{22}\).

Monitoring and visual inspections are important as preventive measures even if symptoms of *A. chinensis* presence in bonsai trees and wood packaging materials are not easy to detect. The inspections can be done when importing potentially host plants (e.g. notably bonsai from East Asia) or on the site of infestation. In this last case a buffer zone for monitoring of at least 2 km should be demarcated around the infested area. The radius can be decreased depending on the results of systemic surveys in the area (EPPO 2010). Other studies indicate rather 1 km as a sufficient radius (Cavagna et al. 2011). The buffer zone should then be monitored by targeted surveys to detect any new infestations at an early stage.

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\(^{20}\) Polymerase Chain Reaction: a molecular biology method to detect DNA traces


\(^{22}\) Never observed in EU
The mechanical exclusions through wire nettings (Lieu 1945) or spiral guards at the trunk base is another option which might serve as physical barrier for adult outcome (used for instance in Japanese citrus orchards), but the results might be not optimal.

Alternatively, strategies of natural control can be set up. Experiences of larval control with natural enemies (e.g. red ant *Oecophylla smaragdina*) prevented the need for chemical control in China (Lieu 1945, Yang 1984). Similarly, adult mortality was induced by the pathogenic fungi *Beauveria brongniartii* in Japan (CABI 2004). The use of the egg parasitoid *Aprostocetus anoplophorae*, which is strictly specific to citrus longhorn beetle is currently under evaluation (Delvare et al. 2004). The parasitoid was introduced accidentally to Italy in bark of living host plants imported from Eastern Asia (Hérard et al. 2005, Haack et al. 2010). Currently, a population of the egg parasitoid is well established in a part of the major citrus longhorn beetle infestation hotspot around Parabiago (Lombardy, Italy). The parasitoid should soon be introduced in the other areas of the Parabiago infested zone, as well as in the other citrus longhorn beetle infested areas in Northern Italy to help contain the pest. In the native area of citrus longhorn beetle, its use as a biological control agent could help containing the pest, at least for the form *malasiaca*, which mainly occurs in Japan (Franck Hérard, personal communication).

A more classic approach is the chemical control. A promising form of chemical control involves the use of microencapsulated lambda-cyhalothrine, an insecticide to be used in local applications at the base of the trunks. The adults crawling on these microcapsules, in particular females looking for oviposition sites, are killed upon contact (Franck Hérard, personal communication). First tests on citrus orchards in China were successful (Smith, unpublished results, personal communication of F. Hérard). This can be a good strategy to prevent the beginning of pest damage. However, when an infestation is declared the most efficient strategy is entire or partial (sanitation) plants removal and destruction. The scale of action depends on the species present and should be of a minimum of 250 mt (Franck Hérard, personal communication – see following section). Alternative innovative methods are also under development, such as novel control methods to lure adults to traps (e.g. using of pheromones) or the use of *Aprostocetus anoplophorae* a natural control agent (Haack et al. 2010).

### Recommendations for management

The best measures to take will depend on factors such as the size of infestation which will depend on the time elapsed between the entry and the detection of the pest, but also on climate and on its favourite host plant density. First actions to be taken include (EPPO 2010):

- Determining the source of infestation
- Determining the infestation size
- Determining the pest population size

Based on this information, further management actions can be planned. However, the effectiveness of all pest management strategies depends in large part upon proper timing of their implementation to coincide with the targeted life stage. For example, insecticidal control of adult beetles should coincide with adult emergence. Moreover, the best timings could be predicted by degree-day models developed for related species (Knutson et al. 1989, Legg et al. 2002, Kean and Kumarasinghe 2007, Douglass et al. 2008).
Some suggestions to improve the monitoring and eradication efforts are presented below:

- Destroy all susceptible trees (even with no sign) in a 250 m radius around an infested tree. So far, most failures in *Anoplophora chinensis* and *A. glabripennis* eradication were due to omission of preventive cutting. Each time preventive cutting was made, dissection of the trees showed that some trees were in fact infested and would have prolonged the duration and the cost of the eradication efforts.

- Destruction (burn or chip <15 mm) and/or replacement of susceptible trees in the infested area (the tree owner can be compensated); preferably during the winter when beetle activity and dispersal are very low. However, if some beetles are found in the summer on a particular tree, contact insecticide should be sprayed on foliage of that tree to kill roaming citrus longhorn beetle adults, and the tree should be cut with no delay, the stump uprooted, and all plant parts destroyed (chipping and/or incineration).

- Chemical treatment targeting adults in the infested area ('sanitary belt’ to be repeated three times a year), but it’s important to know that the chemical treatment is not effective on larvae.

- Use of sentinel plants, particularly attractive for the pest, to facilitate monitoring.

- Uprooting of infested trees and incineration; if uprooting is not feasible the stump can be chemically treated and covered with a wire mesh (40 cm). The whole stump must be removed in most cases.

- Inform amateur entomologists, gardeners, stakeholders and general public of the importance of an early detection (face to face communication to be preferred).²³

- Targeted preventive survey in locations at risk (hosting *Acer spp.*).

- Consider the activities performed close in the infested area at least in the last 6 years.

- The quarantine should last at least 6 years (3 life cycles). It can be further modulated depending on the characteristics of the infestation.

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²³ See for instance: [www.regione.lombardia.it/cs/Satellite?c=Redazionale_P&childpagename=DG_Agricoltura%2FDetail&cid=1213305544054&packedargs=menu-to-render%3D1213276891208&pagename=DG_AGRWrapper](www.regione.lombardia.it/cs/Satellite?c=Redazionale_P&childpagename=DG_Agricoltura%2FDetail&cid=1213305544054&packedargs=menu-to-render%3D1213276891208&pagename=DG_AGRWrapper)
Further research needs for *A. chinensis* management have been identified (Haack, 2010) and include:

- More detailed molecular studies on the genetic relationships between *Anoplophora* species in order to better target the relevant forms in management strategies (e.g. biocontrol)
- Further information on host plants and geographical range in Asia
- Improvement control and prevention methods in international trade of plants
- Refined models for spread dynamics in different environments (urban, rural, forest)
- Identification of resistant plant species and definition of planting strategies to reduce the impacts of outbreaks
### 2.3.3 *Thaumetopoea pityocampa*

#### 2.3.3.1 DESCRIPTION AND DAMAGE PATHWAY

<table>
<thead>
<tr>
<th>Species</th>
<th><em>Thaumetopoea pityocampa</em> (Pine processionary moth)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Insect</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Native</td>
</tr>
<tr>
<td>Distribution</td>
<td>Central and southern Europe (Bulgaria, Cyprus, Croatia, France, Greece, Hungary, Italy, Romania, Spain, Switzerland, Portugal), Turkey, North Africa (Algeria, Egypt, Marocco, Tunisia), Middle East (Israel, Jordan, Lebanon, Libya, Syria)</td>
</tr>
<tr>
<td>Host specificity</td>
<td><em>Pinus</em> spp., also on <em>Cedrus</em> spp., <em>Larix</em> and <em>Pseudotsuga menziesii</em></td>
</tr>
<tr>
<td>Spatial and temporal scale of impact</td>
<td>Widespread impact, frequent outbreaks</td>
</tr>
<tr>
<td>Interesting lesson</td>
<td>Number of feeding hours in winter strongly influences <em>Thaumetopoea pityocampa</em> distribution</td>
</tr>
</tbody>
</table>

*Thaumetopoea pityocampa* (Figure 2-17 and Figure 2-18) is a prominent defoliator of various pine species in the Mediterranean basin, where it frequently occurs at outbreak density. The insect is well adapted to the Mediterranean climate, passing the typical hot and dry summers in pupal stage (aestivation) so that adults do not emerge before late summer. In more continental or alpine areas, adult flight, mating and egg laying already occur earlier in the season in order to gain time for egg hatch, early larval development, and nest building before the cold period.

*Figure 2-17 Thaumetopoea pityocampa* butterfly (photo: David Element)  
*Figure 2-18 Thaumetopoea pityocampa* caterpillars (photo: Association Plantete Passion)
The silky white nests, preferentially placed on sun-exposed parts of the host trees, are built to protect the larval collective from adverse, low temperatures during the winter season. Larvae leave these nests only at night-time to feed on the pine needles (Figure 2-19). Their capability to endure long periods of starvation enables them to survive also stronger winters with persisting low temperatures. Defoliation is most pronounced between mid-winter and early spring, when larvae are in the fourth and fifth instar. Once they are fully developed, larvae leave their nests and move to the ground in a characteristic procession for pupation in the soil. A variable proportion of individuals does not emerge in the same year and enters extended pupal diapause, which may continue over several years.

An interesting discovery was made in a restricted area of the oldest National Pine Forest of Portugal (Leiria) in September 1997. Striking levels of defoliation were recorded in the highly productive maritime pine stands at a time when egg hatch of *Thaumetopoea pityocampa* should have taken place. It turned out that due to an earlier break of pupal diapause and adult emergence, larval development had happened during the summer, despite conditions of maximum temperature and irradiation (Pimentel et al. 2006). This abnormal behaviour was observed for a single moth population in a limited area of 10x50 km, but has since then reoccurred every year. Genetic analyses showed that the summer population can be seen as a new 'phenological race', which is highly differentiated to the still co-occurring 'normal' population, so that mating between individuals exhibiting different life cycles is not possible (Santos et al. 2011). Though the ecological consequences and any limitations for this shifted behaviour are still not clear, this story underlines the high genetic flexibility of insects and their potential to adapt to (changing) climatic conditions.

### 2.3.3.2 Damage significance

Due to its widespread distribution throughout southern Europe and the Mediterranean MS and its oligophagous behaviour, *Thaumetopoea pityocampa* is rated as one of the most damaging defoliator of pine species. MS, such as Portugal, the forestry of which depends to a large part on pine production and where pines represent about one-third of the forested area (Gatto et al. 2009) are particularly affected. In southern France (Rouault et al. 2006), the outbreaks of
irregular frequency (between three to ten years) are a problem especially in large plantation forests, such as the Massif de Landes de Gascogne with about one million hectares of maritime pine (*Pinus pinaster*) (Samalens and Rossi 2010).

Defoliation by feeding moth larvae can drastically reduce the growth and reproductive capacity of forest stands (Figure 2-20), as was shown for Scots pine (*P. sylvestris nevadensis*) in southern Spain. Hódar et al. (2003) showed that growth increment was on average halved on attacked juvenile trees, and seed production of defoliated adult trees was reduced by 50%, with seeds almost 40% lighter than in unaffected pines. Chronic defoliation of Crimean pine (*P. nigra Arnold*) in Turkey resulted in an average decrease of annual increment by 33% (maximum 40%), with significant impact on radial (20%), height (35%) and volume (61%) growth especially in young plantations (Carus 2010). Still higher losses in biomass increment were calculated by Arnaldo et al. (2010) for young maritime pine stands in north-eastern Portugal, with about 70% for heavily, 50% for moderately and 37% for lightly defoliated trees. Arnaldo and his team estimate economic losing by 100 € per hectare after heavy infestation, which potentially represents up to 12.6 million € for entire Portugal.

Apart from economic demands, Mediterranean forests are of priceless ecological and social value. They play an important role in soil protection, e.g. on sandy, coastal sites, and are highly estimated as places for recreation. Not only that the silky nests of the processionary moth are very visible to the public, impairing aesthetic qualities of pine forests, the poisonous larval hair cause skin irritations (contact dermatitis) and may also lead to an inflammation of the upper respiratory system and asthma like symptoms (Maier et al. 2003; Waldwissen website24). Sanitary threats arise for humans, but also for pets or cattle and may be high also in semi-urban or urban areas, where *Thaumetopoea pityocampa* is increasingly spreading (Robinet et al. 2010).

![Figure 2-20 Left: Severe defoliation of pine trees as a result of feeding by the pine processionary caterpillar in Macedonia.](image)

Right: Pine processionary caterpillar egg pods on pine tree in Macedonia (photos: G. Allard)

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24 website available from: [www.waldwissen.net](http://www.waldwissen.net) (accessed 15/12/2011)
2.3.3.3 IMPROVED PERFORMANCE AND RANGE EXPANSION – THAUMETOPOEA PITYOCAMPA PROFITS FROM CLIMATE CHANGE?

*Thaumetopoea pityocampa* is well adapted to the climatic conditions of its main distribution areas, and winter temperatures and solar irradiation primarily limit altitudinal and latitudinal boundaries of its geographic range (Battisti 2004). Yet, as spatial patterns of regional and local occurrence are rather temperature-driven than depend on biotic factors, *Thaumetopoea pityocampa* is seen as illustrative example for interpreting the potential impact of global warming on future insect pest performance (Netherer and Schopf 2010). Increasingly favourable conditions at higher latitudes or elevations allow for range expansion of the moth into previously unsuitable areas, enhanced by the wide distribution of pine forests and by the insect’s ability to switch host tree species (Battisti 2004). The extension of outbreaks has been recorded in the French and Italian Alps, in the Massif Central and North-Central France, and in Spain, where *Thaumetopoea pityocampa* is used as bioindicator for climate change (Pena et al. 2009). The following examples document causalities and consequences of *Thaumetopoea pityocampa* range expansion: altitudinal range limits in a valley in the Italian Alps as well as in the mountains of Sierra Nevada and Sierra de Baza in southern Spain, and latitudinal range shift of the pest in the Paris basin.

Altitudinal range expansion due to increased winter survival

In the Southern Alps (Northern Italy), where *Thaumetopoea pityocampa* is occurring at its northern range boundary in stands of Austrian black pine (*P. nigra*), the pest has been expanding into higher elevations during the last 35-40 years (Stastny et al. 2006). Between 1975 and 2004, infestations shifted 110 to 230m upwards (Buffo et al. 2007), which raised questions about the mechanisms for this range extension at the very edge of pest distribution. Larval performance during winter plays a key role in population survival and establishment and was therefore given special attention in the EU project *Promoth* (2002-2005).

Effects of winter temperature

The East-West orientated, North Italian Venosta valley ("Vinschgau) served as one focus area in the *Promoth* project to study the different temperature conditions for development on the north- and the south-exposed slopes. Battisti et al. (2005) found that developmental stage of the larvae was already more advanced by the end of the pre-cold period on relatively warmer sites. Enhanced growth due to increased summer temperatures enables the larvae to construct larger and better insulated nests for protection against adverse winter temperature conditions. Inside their nests, larval colonies may survive freezing below -17°C, survival however decreases with the length of exposure to low temperatures (Hoch et al. 2009). In many cases, larval mortality can be related to cumulative chill injuries above the lower lethal temperature. Relatively warm daytime temperatures on the one hand help to repair these injuries, on the other hand are needed to activate larvae for night-time feeding (Buffo et al. 2007, Hoch et al. 2009). Leaving of the nests and food consumption are controlled by the temperature thresholds of 9°C during the day and of at least 0°C during the night (Battisti et al. 2005). Buffo et al. (2007) further showed that a threshold of -6°C mean minimum temperature for the entire cold period well explains *Thaumetopoea pityocampa* distribution in the Italian Alps. Despite the general ability of larvae to endure long periods of starvation (Battisti et al. 2005, Hoch et al. 2009), the number of potential
feeding hours proved as reliable indicator for population survival, and is thus seen as a significant variable in predictive models of range expansion under climate change (Buffo et al. 2007).

▶ Actual expansion and future scenarios

Besides reduced mortality rates in winter, colonisation of trees at higher elevation may be related to increased flight activity and flight distance of female moths during summer emergence. Females are in general bad flyers; experiments however revealed an approximate night temperature threshold of 14°C for flight and pointed to the length of flight increasing with temperature. Battisti et al. (2006) argue that both, increased larval survival and female dispersal might have caused the moth’s rapid altitudinal extension of its previous limit of occurrence at several sites in the Italian Alps in the hot year 2003. At Valle Venosta, shifts in elevation per decade were estimated at 29m for the north-exposed, and 70m for the south-exposed slope (Battisti et al. 2005). In 2003, infestations moved upwards by more than one third of the height distance recorded the previous 25-30 years (Battisti et al. 2006). These new outbreak areas are however not necessarily persistent, as the establishment of populations at gained sites depends on the occurrence of lethal temperatures and availability of feeding times during consecutive winters as well as the effect of prolonged pupal diapause. Nevertheless, model scenarios for Valle Venosta indicate an altitudinal expansion of stable populations by 200m given a further increase of mean winter minimum temperature by 1°C. In case of a future increase of minimum temperature by up to 4°C, all P. nigra stands of Valle Venosta are predicted to be affected by Thaumetopoea pityocampa outbreaks (Pennerstorfer et al. 2005).

Figure 2-21 Occurrence of Pine processionary moth in the north Italian Valle Venosta (Vinschgau) under current climatic conditions (left) and given an increase in long term mean winter temperature of +2°C (right): very likely (red), likely (yellow) and unlikely distribution.

No evaluation for altitudes beyond 1900m
(Promoth, Final Report: WP1 “Core and expansion range”)
Host switch and nature protection issues

*Thaumetopoea pityocampa* is common in pine forests throughout Italy, yet, the Austrian black pine plantations of South Tyrol (Northern Italy) were the first stands where the insect was perceived as threatening pest (Masutti and Battisti, 1990). Severe outbreaks in “new” distribution areas were thus recorded already at the end of the 18th century, when afforestation programs in this region had started to anticipate soil erosion and desiccation of exploited sites. The recently observed altitudinal range shift of the moth happens in black pine stands mainly planted between 1951 and 1965, on natural habitats of deciduous tree species, such as Turkey oak (*Quercus cerris*) or South European Flowering Ash (*Fraxinus ornus*). Today, *P. nigra* is considered the primary host of *Thaumetopoea pityocampa* in Italy, followed by Aleppo pine (*P. halepensis*), maritime pine and Scots pine (Masutti and Battisti, 1990). Host preferences are however region-specific and may be different in other areas (Stastny et al. 2006). In any case, future potential of the moth to expand is not only influenced by climatic factors, but also by its ability to recruit new host species.

Survival on traditional and novel hosts

The recent range expansion in northern Italy gave opportunity to examine the success of *Thaumetopoea pityocampa* development on the previously secondary host species, *P. sylvestris* and novel hosts, the mountain pines (*P. mugo, P. uncinata*) and Swiss pine (*P. cembra*). According to Stastny et al. (2006), behaviour of *Thaumetopoea pityocampa* females strongly impacts patterns of infestation, as they showed clear preferences for oviposition on black pine, but also mountain pine, and were less attracted to Scots pine under both, natural and laboratory conditions. Scots pines were however observed to be equally infested with eggs on southern Spanish pine stands (Hodar et al. 2002). Further, larval growth and mortality rates did not differ significantly between the three pine species in Italy, so that increased acceptance of less favoured hosts and future adaptation of populations in the expanding range is probable (Stastny et al. 2006).

While the mountain pines *P. mugo* and *P. uncinata* both serve as attractive alternative host species in expansion areas, attack of Swiss pine by *Thaumetopoea pityocampa* has not been recorded yet. Future invasion of its high elevation mountain habitat seems to be limited by reduced nutritional quality of *P. cembra* needles, impairing larval performance and delaying phenology (Buffo et al. 2007). Low temperature conditions were not identified as major limiting factors. Yet, as long as winter precipitation falls in terms of snow, covered branches are likely to anticipate larval feeding and constrain population establishment (Petrucco-Toffolo and Battisti 2008).

Outbreak in a relict Mediterranean Scots pine forest

An actual change of *Thaumetopoea pityocampa* to a new local host has been taking place in the National Park of Sierra Nevada, Spain. Hódar et al. (2003) observed first infestations of relict stands of the Andalusian Scots pine (*P. sylvestris nevadensis*), which previously grew outside the natural habitat of the defoliator at elevations between 1600 and 2200m, end of the 1990s. These forests are strictly protected, as they constitute the southernmost distribution of Scots pine (in

25 www.provinz.bz.it/forst/forstdienst/1834.asp
Panorama of the current risks caused by biotic agents to the delivery of goods and services

an native variety) in Europe, and restricted to two small populations and therefore highly endangered. With a changing climate, the areas of optimal growing conditions will move upwards, both for Scots pine and Black pine, which dominates at lower elevations. Currently, an expansion into stands of higher altitude is only recorded for the associated pest insect, but not for the plants (Hódar and Zamora, 2004). The warm winters of 1997 and 1998 triggered defoliation of Scots pine at elevations up to 2000-2100m. Persistent outbreaks will add to the threat of extinction, as growth and reproductive capacity of affected trees are severely impaired, and stand susceptibility to drought or pathogens is increasing (Hódar et al. 2003).

North-central France – accelerated shift of latitudinal range boundaries

The trend of latitudinal range expansion of *Thaumetopoea pityocampa* and its potential to conquer new habitats is most striking in north-central France. Between 1972 and 2004, the pest moved northward by 87 km, coinciding with an increase of minimum winter temperature by 0.9-1.1°C (Robinet et al. 2007). Until the last century, further expansion was prevented by a geographic area located about 50-70 km south of Paris and with climatic conditions unfavourable for winter feeding. This barrier was lost with rising winter temperatures in the early 2000s, so that dispersion into the Paris Basin (by about 5.6 km/ year) is now solely influenced by the availability of host trees (mainly Black and Scots pine) and flight capability of female moths (Robinet et al. 2010).

Assuming mean female dispersal distances of 3 km per year and an average temperature increase by 3°C, Roques et al. (2011) predict the arrival of *Thaumetopoea pityocampa* in downtown Paris by 2025. This prognosis does however not take into account human-mediated long-distance movement of the insect, which was proved to occur quite frequently by Robinet et al. (2011). Pioneer colonies were discovered far from the actual range limits, in the surroundings of Paris and in eastern France at the border to Germany. The new populations most likely originate from transportation of plants together with *Thaumetopoea pityocampa* pupae in the soil. Triggered by human translocation activities, colonisation of pine trees in the city of Paris is probable already in 2013 (Figure 2-22, Robinet et al. 2010).

![Figure 2-22 Distribution of Thaumetopoea pityocampa in France and range expansion from the period 1960-1980 (left) to the winter of 2005-2006 (right) (Robinet et al. 2010)](image-url)
2.3.3.4 Management of Thaumetopoea pityocampa on stand and landscape scale

- Measures to control Thaumetopoea pityocampa damage
  - Mechanical, chemical and biological control methods

High population densities of Thaumetopoea pityocampa, reoccurring in cycles of about 6 years (Cayuela et al. 2011) throughout its main distribution range, warrant the need for pest control and the development of various fighting measures. In the Mediterranean MS, mechanical methods, such as the destruction of nests by cutting, burning or shooting, have a long tradition (Masutti and Battisti 1990). These strategies are quite laborious; however control options for private forest owners are limited, sometimes also including the placement of insecticide-impregnated adhesive bands around pine trunks to capture larvae in their procession to the ground (Gatto et al. 2009). On a larger scale, aerial applications of biological insecticides have been widely used. Stands are usually sprayed in late summer or early autumn, e.g. with the biochemical substance Diflubenzuron which inhibits larval ecdysis (Cayuela et al. 2011), to prevent winter defoliation. All over the Mediterranean basin, the pine forest area controlled by insect growth regulators or with Bacillus thuringiensis kurstaki (Btk) preparations amounts to 2 500 000 ha. Although these applications have significant influence on population density (Cebeci et al. 2010), they are in many cases not cost-efficient (Gatto et al. 2009). Cayuela et al. (2011) showed that spraying is mostly applied at the time of calamity peaks, when natural enemies are highly efficient and population numbers would break down anyway. The scientists therefore question the general necessity of intense control measures, which are based on high annual investments, such as 1.0-1.5 million Euros solely spent for the region of Andalusia. Eventually, negative environmental effects cannot be excluded, particularly on the occurrence of insect parasitoids, bird predators or non-target Lepidoptera (Barbaro and Battisti 2011).

- Silvicultural strategies

Occurrence of Thaumetopoea pityocampa, seriousness of forest damage resulting from defoliation and sanitary problems are most significantly associated with the presence of extensive pine plantations in the Mediterranean MS, which are mainly due to afforestation projects of the early 20th century. Monocultures of maritime pine for instance dominate the Peninsula of Setúbal, south of Lisbon (Gatto et al. 2009), or the Landes forest in South-western France (Dulaurent et al. 2011). Also the homogeneous, even-aged stands, mainly of Black pine, in Mediterranean mountains offer environmental traits that facilitate host-finding and reproduction of the pest insect (Hódar and Zamora, 2004). Although Thaumetopoea pityocampa damage prophylaxis by silvicultural measures is no common strategy, various studies point to the efficiency of forest and land-use management aimed at species and landscape diversity in anticipating Thaumetopoea pityocampa calamities. Management options to reduce predisposing stand and landscape characteristics are summarised in Table 2-10 Stand and landscape characteristics featuring Thaumetopoea pityocampa outbreaks and prophylactic management options.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Table 2-10 Stand and landscape characteristics featuring *Thaumetopoea pityocampa* outbreaks and prophylactic management options

<table>
<thead>
<tr>
<th>Predisposing Factors</th>
<th>Forest Management Implications: Stand level</th>
<th>Landscape level</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pure pine stands</td>
<td><strong>Species composition:</strong> promotion of broadleaved and non-host conifer species</td>
<td><strong>Biodiversity:</strong> increase of species diversity, fostering of resources for natural enemies</td>
</tr>
<tr>
<td>Even-aged pine stands of regular spacing</td>
<td><strong>Stand structure:</strong> reduction of stand density, increase of spatial heterogeneity</td>
<td></td>
</tr>
<tr>
<td>Stand edges and open areas</td>
<td><strong>Edge management:</strong> promotion of broadleaved trees and shrubs especially at forest edges, creation of herbaceous edges</td>
<td><strong>Landscape structure:</strong> alternating broadleaved and pine stands, promotion of broadleaved hedgerows adjacent to pine stands, general reduction of open areas</td>
</tr>
<tr>
<td>Homogeneous, extensive stands of planted pine</td>
<td><strong>Promotion of spatial heterogeneity:</strong> diversification of the landscape, maintenance of diverse habitat mosaics, keeping fragments or patches of broadleaved or other non-host trees</td>
<td></td>
</tr>
</tbody>
</table>

The mixture of pine stands with non-host conifers and broadleaved trees influences habitat availability of both the insect pest and its antagonists. Jactel et al. (2011) could show the repellent effect of birch branches for *Thaumetopoea pityocampa* females, leading to reduced numbers of larval nests and subsequent lower defoliation of pine trees. Males were as well less attracted to traps baited with a combination of sex pheromone and non-host volatiles released by cut birch branches. Reduced damaged in mixed pine and beech forests was earlier recorded by Geri (1983), also pointing to species diversity lowering stand susceptibility to infestation via ‘associational resistance’ (Jactel et al. 2011). In order to mitigate *Thaumetopoea pityocampa* outbreaks in Mediterranean mountain stands, Hódar and Zamora (2004) for instance propose the composition of pine, together with maple, oak and yew species.

Increasing biodiversity in pine plantations by promoting broadleaved trees in order to reinforce insect parasitoids and bird predators instead of *Thaumetopoea pityocampa* control through insecticides is also recommended by Cayuela et al. (2011). Special attention should be laid on the management of forest edges and stand borders, where nest density is usually very high, as females preferentially oviposit on trees with a distinct silhouette (Samalens and Rossi 2010). Micro-sites at the fringe of pine forests, featuring herbal and shrub vegetation, are favoured both as habitats for pupation and for foraging birds. Hoopoes are for example most efficient in preying on larvae and pupae, and may significantly contribute to pest regulation under non-outbreak conditions (Barbaro and Battisti 2011).

Insectivorous birds need habitat resources complementary to homogeneous pine forests, such as deciduous woodlands or forest fragments. Diversified landscapes are generally estimated to offer a multitude of habitats for different natural enemy species of insect pests. The maintenance of a mosaic of units of different land use on landscape scale is consequently postulated for viable...
biologic control of *Thaumetopoea pityocampa*. Landscape composition and fragmentation not only impacts population densities of antagonists, but also host tree finding and dispersal success of *Thaumetopoea pityocampa* (Samalens and Rossi 2010, Barbaro and Battisti 2011). The alteration of pine forests with broadleaved stands may particularly contribute to pest control, as latter are indeed preferred as pupation habitats, but proved to impair survival rates of pupae. On the other hand, open areas turned out to promote pupae development, probably because of improved temperature conditions compared to sites under forest cover. Low rates of mortality and prolonged pupal diapause lead to a high proportion of emerging adults at open sites (Dulaurent et al. 2011).
2.3.4 *Bursaphelenchus xylophilus*

2.3.4.1 **DESCRIPTION AND DAMAGE PATHWAY**

<table>
<thead>
<tr>
<th>Species</th>
<th><em>Bursaphelenchus xylophilus</em> (pine wood nematode)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Taxon</strong></td>
<td>Nematoda</td>
</tr>
<tr>
<td><strong>Native/Alien</strong></td>
<td>Alien</td>
</tr>
<tr>
<td><strong>Distribution</strong></td>
<td>EU27: Portugal, 2 outbreaks under eradication in Spain; rest of the world: Japan, China, Korea, Vietnam, Taiwan, North America</td>
</tr>
<tr>
<td><strong>Host specificity</strong></td>
<td>Many species of pine; not virulent on North American pines Larch</td>
</tr>
<tr>
<td><strong>Spatial and temporal scale of impact</strong></td>
<td>Entered Portugal in 1999; spread to Spain 2008</td>
</tr>
<tr>
<td><strong>Interesting lesson learnt for management</strong></td>
<td></td>
</tr>
</tbody>
</table>

Generally, wilt diseases have been considered a problem associated with angiosperm plants (e.g. Dutch elm disease); in gymnosperms, however, it has been recognised that this type of disease is caused by ascomycetes in the genus *Leptographium* and by the nematode *Bursaphelenchus xylophilus*. Although native to and widespread in North America, including the USA, Canada and Mexico, the damage resulting from *Bursaphelenchus xylophilus* attack was first noted on pines in Nagasaki, Japan in 1905 (Mamiya 1983), although it was not until 1971 that *B. xylophilus* was recognised as the causal agent (Kiyohara and Tokushige 1971). Since 1905, the problem in Japan has spread widely in that archipelago, probably via the transport of infected logs.

In China, this nematode was first found in Jiansu Province in the early 1980s and has since affected over 80,000 hectares of pine forest, initially infecting *Pinus thunbergii* and *P. massoniana* (Zhao 2008). Work in China in the 1980s demonstrated that a range of native pines were very susceptible to the infection (Baojun and Qouli 1989). In North America, the first damage was reported in 1979 in plantations of pines alien to that continent (Dropkin and Foudin 1979). Initially, it was thought that the nematode was introduced from China or Japan, but it became clear that *B. xylophilus* was in fact endemic in North America (Dropkin et al. 1981, Robbins 1982). Further research demonstrated that *B. xylophilus* is a normal component in the saprotrophic community present in dead and dying wood of pines and other conifers in its native range. Furthermore, the nematode was previously discovered and described from wood of *Pinus palustris* in the 1930s (Stenier and Buhrer 1934).

*Bursaphelenchus xylophilus* is also now present in Taiwan, Vietnam and South Korea (Mota & Vieira 2004). In these countries, along with China and Japan, it is the main problem impacting on the growth and production of pine (Vieira and Mota 2006).
In Europe, *B. xylophilus* was reported in shipments of wood and pulp in Finland, Norway and Sweden in the 1980s (Rautapaa 1986, Magnusson and Schroeder 1989) and from dead and dying pines France in 1979 (Baujard et al. 1979). *B. mucronatus* has also been reported in Austria. These reports led to the implementation of EU Directive 77/93 EEC on *Bursaphelenchus xylophilus*. The nematode isolated from these reports were later re-examined using microscopic and biochemical techniques, and proved to be the related species *B. mucronatus*. This second species may be associated with pines, but does not cause mortality.

Despite the precautions put in place after the mistaken identifications, however, in 1999 *B. xylophilus* was reported in dead and dying *Pinus pinaster* in Portugal (Mota et al. 1999); the affected trees also showed symptoms of attack by *Phaenops cyanea*. Molecular evidence strongly supported the diagnosis. These first findings were at two sites, approximately 3 km from each other, Herdade Moinho Novo and Herdade Vale de Landeira in the Marateca/Pegoes region, approximately 40 km south-east of Lisbon. Since that time, the affected range in Portugal has expanded to include an area of 510,000 hectares, as far south as Sines and Santiago do Cacem, east to Vendas Novas and north/northeast of Lisbon, beyond Alcochete (Rodrigues 2008).

Moreover, in November 2008, the nematode was found in a single *Pinus pinaster* tree in Villanueva de la Sierra, Cáceres, Extremadura, Spain (EPPO 2010). A further seven trees of the same species were found infected in 2010, in the As Neves province of Galicia, Spain. All of these affected trees were destroyed by felling and burning. Further locations where *B. xylophilus* was found were discovered in Portugal in 2008 (Evans, personal communication).

Molecular evidence suggests that *Bursaphelenchus xylophilus* occurring in Portugal are very closely related to isolates from China, Japan and South Korea, rather than to those from the native range in North America, indicating that introduction to Portugal was from the Far East (Evans 2007).

**Life Cycle**

*B. xylophilus* is transmitted locally by the wood boring longhorn beetles in the genus *Monochamus* (Coleoptera; Cerambycidae, Figure 2-23) during maturation feeding in the crowns of living pine trees (Futai and Furuno 1979, Rutherford and Webster 1987), or during oviposition of female Monochamus sp. on already weakened or dead trees of most coniferous species. In the early detection of *B. xylophilus* in Portugal, *Monochamus galloprovincialis* appeared to be the vector (Sousa et al. 2001). Survey work in Portugal and Spain within the EU-funded PHRAME project demonstrated that *M. galloprovincialis* was the sole pine-associated insect species that demonstrably carried *B. xylophilus* (Naves et al. 2007, Evans 2007). Other Monochamus species present in Europe, and potential vectors of the nematode are listed in Table 2-11.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Disturbances of EU forests caused by biotic agents

![Figure 2-23 Adult Monochamus beetle (photo: Dr. Jim Appleby)](image)

Table 2-11 Monochamus species present in European forests and potential vectors of *Bursaphelenchus xylophilus*: geographical distribution in Europe

<table>
<thead>
<tr>
<th>Species</th>
<th>Geographic distribution</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Monochamus galloprovincialis</em></td>
<td>Spain, Portugal</td>
</tr>
<tr>
<td><em>Monochamus galloprovincialis</em></td>
<td>Central-Eastern Europe (Alps included)</td>
</tr>
<tr>
<td><em>Monochamus saltuariis</em></td>
<td>Middle (Alps included) and Eastern Europe</td>
</tr>
<tr>
<td><em>Monochamus urossovii</em></td>
<td>Northern and Eastern Europe</td>
</tr>
<tr>
<td><em>Monochamus sartor</em></td>
<td>Albania, Austria, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, France, Germany, Hungary, Italy, Lithuania, Poland, Romania, Serbia and Montenegro, Slovakia, Slovenia, Switzerland and the Ukraine</td>
</tr>
<tr>
<td><em>Monochamus sutor</em></td>
<td>Western (Pyrenees), Middle (Alps included) and North-eastern Europe, Romania, Bulgaria, Georgia</td>
</tr>
</tbody>
</table>

*a*: recent evidence suggests insufficient distinction to warrant two sub-species of *M. galloprovincialis*

*b*: genetic diversity between *M. sartor* and *M. urossovii* is extremely low

In Portugal, the adult *M. galloprovincialis* emerge from pupation between mid-May and early September each year (Naves et al. 2008). The adult beetles lived for approximately 2 months and female beetles laid approximately 65-70 eggs during this period. There is only a single generation of beetles each year (univoltine). Nematodes are transmitted by beetles for approximately six weeks from emergence.

Inoculation of pine with *B. xylophilus* leads to the development of embolisms in the xylem within hours. Within a few days (to weeks), the foliage begin to show signs of discoloration (Figure 2-24), following disturbance to xylem flow (Kuroda and Kuroda 2004). The ability of *B. xylophilus* to reproduce quickly and the fast migration of nematodes from the point of infection each contribute to the rapid development of symptoms in infected host trees. These infected trees then become attractive to *Monochamus*, which lay eggs in them; the resulting adult insects then transmit the nematode to previously unaffected trees.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

The highest dispersal distance for marked *M. galloprovincialis* from a release point was approximately 240 m over a 6 week period; these marked insects had dispersed an average of 12.7 m in the first week after release (Evans 2007).

In tests of the susceptibility to *B. xylophilus* of European conifers, many native and important tree species proved highly susceptible, including *Pinus nigra, P. sylvestris, P. cembra* and *Larix decidua* (Table 2). In contrast, *P. pinea* and *P. halepensis*, showed low susceptibility to *B. xylophilus*. There were variations in susceptibility to *B. xylophilus*, however, depending on the source of the *B. xylophilus* used in inoculations (Evans 2007).

Table 2-12 Relative susceptibility of European *Pinaceae* to killing by Portuguese isolates of *Bursaphelenchus xylophilus* (adapted from Evans 2007)

<table>
<thead>
<tr>
<th>Host tree species</th>
<th>Low</th>
<th>Moderate</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Pinus nigra</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pinus sylvestris</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pinus cembra</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Larix decidua</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pinus pinaster</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pinus mugo</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pinus pinea</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pinus halepensis</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Abies alba</em></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Picea abies</em></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

### 2.3.4.2 Damage significance

The initial problems noted on pines in China in the early 20th Century lead to an epidemic of deaths of *Pinus densiflora* and *P. thunbergii*, with the outbreak eventually controlled by the
burning of all dead trees (Kobayashi 1981). Outbreaks in the Japanese regions of Nagasaki, Kyushu and Shikoku from 1925 onwards produced annual timber losses of approximately 400,000 m³ (Mamiya 1988). The affected areas expanded greatly during and after the Second World War, when all attempts at control were suspended, resulting in annual timber losses by 1979 of some 2.4 million m³ and a spend of US$35 million on control measures in 1980 (Mamiya 1984). By the 1990s, the only region in Japan where *B. xylophilus* did not occur was Hokkaido (Evans et al. 1993).

Rapid death of the trees is characteristic for pine wilt disease. Tree death usually occurs during the same vegetation period after infestation if the climatic conditions are suitable. Moreover, the risk is long lasting, as *B. xylophilus* can be found up to three years after the trees have died from pine wilt disease (Malek and Appleby 1984).

The discovery of *B. xylophilus* killing pines in Portugal in the late 1990s has lead to great concern in Europe about the distribution of the pest, the possibility for further spread and the potential impacts its presence has on timber movement within and exports from the EU (Table 2-12).

**2.3.4.3 IMPORTANCE OF CLIMATIC FACTORS**

*B. xylophilus* causes rapid wilting and death of infected trees when summer temperatures exceed 20°C and drought conditions occur (Evans et al. 2008). In regions where the average summer temperatures are below 20°C, the disease is rare (Rutherford and Webster 1987). Under conditions prevalent in affected regions of China, severity of pine wilt is positively correlated with mean temperature and with winter and spring precipitation; in contrast, severity is negatively correlated with precipitation in June (Xi and Niu 2008, Zhao 2008).

Within the PHRAME project, experimental evidence was obtained showing that pine wilt did not develop in inoculated Scots pine plants maintained at temperatures of 15°C; in contrast, trees incubated at 20 or 25 °C developed wilt symptoms within 61 days of inoculation (Evans 2007). Similar results were also obtained for European larch (*Larix decidua*), but no wilting developed in Norway spruce (*Picea abies*) at any of the incubation temperatures. Symptoms and mortality developed more rapidly at 25 than at 20°C. Moreover, populations of *B. xylophilus* were higher in inoculated plants following incubation at 25 than at 20°C. In larch, however, there was little apparent effect of temperature on numbers of *B. xylophilus* found in inoculated plants. During this work, it was also demonstrated that relative water content of needles, but not stem wood, could be used to predict pine wood disease.

**2.3.4.4 FUTURE TRENDS UNDER CLIMATE CHANGE**

*B. xylophilus* is present in approximately half of China, due to the climatic restrictions in the unaffected regions of this country. The affected regions include Yunnan and Beijing, where warm summer temperatures and high late winter-early spring precipitation, coupled with summer droughts, are conducive to *B. xylophilus* development (Zhao 2008).

In the EU, the PHRAME project (Evans 2007) developed models for the prediction of suitable climatic envelopes for *B. xylophilus* to survive and cause pine wilt, based on data gathered in the
Panorama of the current risks caused by biotic agents to the delivery of goods and services

EU within the project, and pre-existing data from Asia (Evans et al. 2008). These models demonstrated the potential risks posed to regions of the EU other than Portugal by *B. xylophilus*, and indicated that the nematode will survive in any area where *Monochamus* spp. are present, but that climatic factors determine the risk of disease expression even when the nematode is present.

The modelling tools developed in PHRAME are currently being further developed in the recently instigated EU-funded Research programme REPHRAME, coordinated by Dr Hugh Evans (Head of Forest Research, Wales), which began in 2011.

**2.3.4.5 MANAGEMENT OF PORTUGUESE OUTBREAK**

Immediately on detection of *B. xylophilus* in Portugal, the EU authorities were alerted and immediate measures put in place, including isolation of the affected area, felling and burning of all infested trees and the launch of a national survey in Portugal to determine the distribution of *B. xylophilus* (Programa Nacional de Luta contra o Nemátoide do Pinheiro: PROLUMP). A pest risk analysis for the EU area was published in 1996 (Evans et al. 1996) and further recommendations for the management of spread of *B. xylophilus* (EPPO 2003). This latter document describes the formal methods for survey and delimitation of affected areas, and for the application of control measures within the quarantine zone.

As with any pest organisms, a thorough understanding of the life cycle is required in order to develop efficient control and management options. In Portugal, an eradication strategy was adopted as soon as the presence of *B. xylophilus* in the dying pines was confirmed. The Portuguese phytosanitary authorities instigated these measures despite a lack of knowledge of the responses of pines in that region, or of the local vectors of the nematode. Sites were surveyed intensively for the presence of *B. xylophilus*, showing the infestation to be centred to the South-East of Lisbon, in the Setúbal region. After the first discovery, a 20km wide buffer zone surrounding the affected area was delimited and annual surveys for *B. xylophilus* conducted. If any trees with *B. xylophilus* symptoms were discovered, these were felled and burned, in an attempt to eliminate the nematode and the potential vectors. Trees were destroyed before the likely emergence of the next generation of vectors.

The strategy of PROLUMP (National Eradication Programme), a Portugal-wide programme, is to prevent further spread of the nematode from its current locations and to eradicate it through felling and destruction of all breeding material within a designated zone (Rodrigues 2008). The area affected by *B. xylophilus* (510,000 hectares) was demarcated and a buffer zone of a further 500,000 hectares of land was mapped around the affected area (see maps in Rodrigues 2008). All forestry activities within the affected area and buffer zone are very strictly controlled. Surveys are carried out within the affected and buffer zones each year, in the autumn and winter. Moreover, controls were applied during spring and summer each year in an attempt to prevent dispersal via the vector insect, *M. galloprovincialis*. Despite these measures, large increases in the numbers of trees showing symptoms of *B. xylophilus* infection were noted in the 2005/6 season, leading to an expansion of the affected and buffer zones. A clear-cut belt, 3 km in width, was then established beyond the buffer zone, and all pines and spruce within this belt were felled and burned on site. These actions are all carried out to minimise disease spread, in accordance with Article 2 of the Commission Decision 2006/133/EC (revised July 2006) based on the opinion of the
Standing Committee on Plant Health. Tree owners were compensated for timber losses in 2006/7, according to Commission Decision 923/2006/CE (Dec. 13th). Surveys are also made in the B. xylophilus -free zone, beyond the clear cut zone, in order to monitor for break-outs from the various demarcated areas.

Additional potential control measures tested in Portugal include the injection of nematicides or insecticides into trees. These tests had no effects on the host trees, but were not effective against B. xylophilus or M. galloprovincialis (Evans 2007). Several types of traps successfully caught the vector beetles. A range of potential lures (kairomones) was also tested, the most effective of which proved to be a mixture of turpentine and ethanol.

Conclusions from the EU-funded PHRAME project (Evans 2007), based on the process model developed in that work, suggested that it was possible to reduce the risk of B. xylophilus spread through heat treatment or fumigation of round wood or sawn timber, both of which are considered to be high risk pathways for spread. Wood chips and other wood products are considered low risk for transmission, even if the nematode is present, although it was still recommended that these products be heat treated or fumigated, and that such materials be kept away from potential host trees.

### 2.3.4.6 Survey and Control Requirements

Since 2006, the EU legislation requires the MS to take measures against the dissemination the B. xylophilus (Commission Decision of the 13th February 2006). This means each MS should verify annually that the pest is not present on its territory. EPPO issues some recommendations on the survey strategy. The surveys should preferentially be conducted in a time of year when both pine wilt disease symptoms and Monochamus activity are likely to be high. Surveys should be pathways based, focusing on entry points or exit points (e.g. points of import of wood products, wood processing facilities) and on the known host range and distribution of Monochamus spp. Thus by order of importance it should focus on:

- Roundwood with bark in recently logged areas of commercial forests, as weakened trees can become breeding grounds for Monochamus
- Wood packaging/sawn wood in wood processing facilities, as Monochamus or B. Xylophylus may still be present after wood processing
- Conifer plants for planting, in particular larger trees that can be used for maturation feeding or breeding

If the presence of B. xylophilus is detected, set containment and eradication measures should be taken, involving: destruction of the infested tree and a 500m to 3km radius clear cut zone around the infested tree, and an adjacent 3 km area that should be monitored intensively.

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2.3.5 Ash dieback

2.3.5.1 DESCRIPTION AND DAMAGE PATHWAY

<table>
<thead>
<tr>
<th>Species</th>
<th>Hymenoscyphus pseudoalbidus (anamorph: Chalara fraxinea)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Fungus: Ascomycota</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Uncertain: probably alien</td>
</tr>
<tr>
<td>Distribution</td>
<td>Europe (Austria, Czech Republic, Denmark, Estonia, Finland, Germany, Hungary, Croatia, Latvia, Lithuania, Netherlands, Poland, Romania, Slovakia, Slovenia, Sweden)</td>
</tr>
<tr>
<td>Host specificity</td>
<td>Ash (Fraxinus excelsior, Fraxinus angustifolia)</td>
</tr>
<tr>
<td>Spatial and temporal scale of impact</td>
<td>First reports in Poland, early 1990s; now widespread in Central and Northern Europe</td>
</tr>
<tr>
<td>Interesting lesson learnt for management</td>
<td>No effective management methods known</td>
</tr>
</tbody>
</table>

The phenomenon now named ash dieback was first noted in Poland in the 1990s (Kowalski 2001), but has since spread widely in Central and Northern Europe, and into the north Balkan Peninsula. Currently (2011), the disease has been recorded in Poland (Kowalski 2001; Przybył 2002), Sweden (Barklund 2005), Norway (Timmerman et al. 2011), Denmark (Thomsen et al. 2007), Finland (PATHCAR 2007, Rytkonen et al. 2010), France (Husson et al. 2011), Germany (Schumacher et al. 2007), The Netherlands, Italy, Austria (Cech 2006, Halmschlager and Kirisits 2008), Switzerland (Engesser et al. 2009), Czech Republic (Jankovsky and Holdenrieder, 2009), Hungary, Slovenia, Croatia, Lithuania (Juodvalki and Vasiliauskas 2002), Latvia and Estonia (Rykönen et al. 2010) and the west of Russia (Kirisits, Woodward and Vaisatis, unpublished).

This disease affects trees of all ages, resulting in severe dieback and death (Figure 2-25). During the early stages of infection, symptoms include wilting, necrotic lesions on the foliage, petioles and bark leading to the formation of shoot cankers (Figure 2-26, Heydeck et al. 2005, Cech 2006, Kowalski 2006, Schumacher et al. 2007, Kowalski and Holdenrieder 2008, Halmschlager and Kirisits 2008, Bakys et al. 2009).

The causal organism remained unknown for several years after the dieback was first observed, but was ultimately identified as the hyphomycete Chalara fraxinea (Kowalski 2006). This pathogen was consistently demonstrated to cause symptoms of ash dieback in inoculations (Bakys et al. 2009, Kowalski and Holdenrieder 2009a). Subsequently, molecular work suggested that C. fraxinea was the anamorph of the long-known Helotiales species Hymenoscyphus albidus (Holdenrieder and Kowalski 2009b). H. albidus however, was first recorded in 1850 (Rehm 1896), and is widespread in Europe; it was not hitherto recognised as a pathogen, being endophytic and saprotrophic on the petioles of Fraxinus species. Subsequently, more stringent molecular
analyses based on multigene sequencing indicated that the teleomorph of the causal agent was a
cryptic species closely related to *H. albidus*; this newly recognised species was named
*Hymenoscyphus pseudoalbidus* (Queloz et al. 2011). *H. pseudoalbidus* has only been found in ash
stands showing dieback symptoms.

As suggested by the spread of the ash dieback problem from Eastern to Central Europe over the
last 15–20 years, the pathogen is probably invasive, although the origins are unknown at present.
*H. pseudoalbidus* attacks both *F. excelsior* and *F. angustifolia*, though the behaviour of the
pathogen on other European native ash is unknown at present. Current observations suggest
that the manna ash, *F. ornus*, is not host to the disease.

The life cycle of *H. pseudoalbidus* is, as yet, not entirely clear. Infection appears to be through
leaves (Kraj et al. 2010, Rytkönen et al. 2010) from which it spreads into the shoots. The cankers
form on infected shoots, eventually girdling the tissues, causing death of distal parts
(Schumacher 2011). Fruiting bodies are rarely formed on cankers, however (Figure 2-27). In the
summer following abscission, the apothecia form on petioles of infected, abscised leaves,
releasing ascospores that go on to infect fresh foliage (Engesser et al. 2009, Kirisits et al. 2010,
Kowalski and Holdenrieder 2009b, Schumacher et al. 2007, Schumacher et al. 2010).

Optimum growth of *H. pseudoalbidus* is between 20 and 25°C *in vitro*, although rates of growth
and temperature tolerance varied between isolates (Kowalski and Bartnik 2010). Some isolates
still grow at 30°C. Variations in growth optima between isolates appeared to be related to the
geographic origin of the cultures.
2.3.5.2 DAMAGE SIGNIFICANCE

The pathogen attacks *Fraxinus* species in both forest and amenity situations, causing lesions on the leaves, canker like symptoms on twigs and shoots and serious dieback (Figure 2-28 and Figure 2-29; in severe cases, death of the host trees occurs. In Poland, damage was recorded over several hectares of forest (Zachara et al. 2007). The extent of damage in other regions where the disease is recorded has not been reported.

Several other fungi have been isolated from declining ash, including *Cytospora* sp., *Diplodia* sp., *Fusarium* sp., *Phomopsis* sp. (Przybył 2002, Kowalski and Łukomska 2005, Cech 2006) and *Armillaria* spp. (Bakys et al. 2005). However, *H. pseudoalbidus* is the species most consistently associated with declining ash trees in all states where the disease occurs to date. Although it is still unknown whether other pathogens might contribute to ash decline, possibly predisposing trees to infection (Queloz et al. 2011), some recent evidence suggested from Poland and Denmark that species *Phytophthora*, including *P. cactorum*, *P. plurivora*, *Phytophthora* taxon *Salix* soil and *P. gonapodyides* may also be involved in the problem, following work (Orlikowski et al. 2011).

2.3.5.3 IMPORTANCE OF CLIMATIC FACTORS

With such recent emergence of the problem and few description of the pathogen, there is little information available on the possible impacts of climatic factors on disease etiology. Based on the current distribution range of the disease, however, in climatic zones with warm to hot summers and cold winters, it is highly likely that the causal agent will tolerate a wide range of temperatures and humidities. Anecdotal evidence from Denmark suggests that the pathogen is able to survive very cold winters, as occurred in 2009, and cause severe disease symptoms in the spring immediately after the extreme cold (McKinney et al. 2011). Anecdotal evidence suggests that there is a relationship between disease severity and moisture. Ash dieback is less severe in the drier eastern areas of Lower Austria than in the more humid western areas, although it was not clear if host stress was a major contributing factor in these differences (Kessler et al. 2011).
2.3.5.4 **FUTURE TRENDS UNDER CLIMATE CHANGE**

No data are available on climatic tolerance of *H. pseudoalbidus*, or the likely impacts of the pathogen on host trees under changing climatic conditions. The main limiting factor on occurrence of the disease is probably presence of suitable host trees.

2.3.5.5 **MANAGEMENT**

No effective management protocols are known for the prevention or reduction of damage caused by *H. pseudoalbidus*. It is highly likely, however, that the spread of the pathogen from the regions where it was first noted, into more westerly and southerly regions is via transport of infected plants from nurseries. Recent work carried out in Denmark (McKinney et al. 2011a,b) suggests that some resistance to infection may be present in populations of *Fraxinus excelsior*. This resistance was associated with clonal genotypes showing early leaf senescence and abscission in field trials, although other genetic components of these clones also contributed to the resistance.

Efficient detection techniques, based on quantitative PCR\(^{27}\), have been developed (loos et al. 2009, Chandelier et al. 2010) and it is likely that in the near future, these improved methods, coupled with the potential to detect resistance in native provenances of ash, will enable detailed management protocols to be developed.

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\(^{27}\) Polymerase Chain Reaction
2.3.6 Chestnut Canker

2.3.6.1 DESCRIPTION AND DAMAGE PATHWAY

<table>
<thead>
<tr>
<th>Species</th>
<th>Cryphonectria parasitica</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Fungi (Ascomycota)</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Alien</td>
</tr>
<tr>
<td>Distribution</td>
<td>Europe (Albania, Czech Republic, FYR Macedonia, France, Greece, Hungary, Italy, Slovakia, Slovenia, Switzerland) and North America</td>
</tr>
<tr>
<td>Host specificity</td>
<td>Castanea species: C. dentata (North America); C. sativa (Europe)</td>
</tr>
<tr>
<td>Spatial and temporal scale of impact</td>
<td>Widespread in the distribution range of the chestnut, and also northern France, Germany and UK; slower spread than in the US.</td>
</tr>
<tr>
<td>Interesting lesson learnt for management</td>
<td>The potential use of ‘virus diseases of fungi’ (hypovirulence) to control major disease problems.</td>
</tr>
</tbody>
</table>

Before the turn between the 19th – 20th Centuries, the American chestnut (Castanea dentata) was arguably the most important broadleaved tree in the forests of the eastern USA, comprising some 25% of the individual trees present in these ecosystems which covered some 81 million hectares. Estimates suggest that over 4 billion chestnut trees were present in the eastern broadleaved forests at the end of the 19th Century. Some of the largest angiosperm trees ever recorded were chestnuts present in these forests. Moreover, the trees were heavily exploited. The fine, hard and durable timber was in great demand for all manner of applications, from carpentry to make furniture, to construction, to railway sleepers (Anon. 1995a). Chestnuts produced reliable seed crops each year and were gathered for sale by many people in the native range of C. dentata. The chestnuts were also an important food source for many wild animals.

In 1904, however, the first reports of chestnut blight emerged in the Zoological Gardens in New York City (Merkel 1905, Anagnostakis 1996). Within less than 40 years, the disease went on to reduce the American chestnut from the dominant broadleaved tree it had been, to a small shrubby species with a short life expectancy (Milgroom and Cortesi 2004), having spread at the alarming rate of almost 40 km per year. The pathogen had been inadvertently imported into the New York area on shipments of chestnut timber from Asia and an expedition to the suspected origins in 1913 were successful (Anagnostakis 1996a, 1996b). Symptoms caused by infection were reported very soon after the first infections were described in New York, and include the formation of sunken cankers on the bark (Figure 2-30), wilting of foliage distal to the cankers, and, following death of the distal parts of the shoots, development of epicormic sprouts in the living tissues immediately proximal to the cankers (Murrill 1906a, Metcalf 1912, Anderson 1914). Fruiting structures of the pathogen are orange in colour and
emerge from infected tissues at the canker margins. Both pycnidia (asexual) and perithecia (sexual) occur in these structures. In conditions of high humidity, conidia are forced from the pycnidia as sticky orange tendrils (Rankin 1914, Shear et al. 1917). Cankers are most readily visible on bark of juvenile trees during wet conditions because of the orange colour of the infected tissues (Figure 2-31); on mature trees, the most obvious symptoms of infection are the presence of dieback and, eventually, flushing of epicormic shoots below the dead tissues. The downward growth of the pathogen in the main trunk of the tree leads to death of the whole of the tree, although the root system may remain alive for many years, repeatedly producing coppice shoots in the understory. These shoots succumb to the disease before reaching any great height.

In 1938, chestnut blight was found in the area around Genoa in Italy (Biraghi 1950), killing the European sweet chestnut, *Castanea sativa*, another important tree providing a wide range of benefits matching those of the American chestnut. The disease then spread rapidly throughout the Italian peninsula and into neighbouring chestnut producing countries, including North into France (by 1946), Switzerland (by 1951) and through the Balkan states into Greece (by 1964) and Turkey (by 1967; Anagnostakis 1996, Biraghi 1950, 1951, Bazzigher 1957a, Kailides 1967, Delen 1975, Grente 1981, Turchetti 1978). *C. sativa* proved rather more resistant than *C. dentata* to chestnut blight, however, and the rate of spread in Europe was slower than in North America (Mittempergher 1978, Hebard 1982).
Currently, there is a considerable amount of research focused on determining genetic variations within populations of *Cryphonectria parasitica* and in the ds-RNA occurring in these populations (Milgroom and Lipari 1995a, Milgroom et al. 1996, Hoegger et al. 2000, Marra et al. 2004, Bruellin et al. 2006). This work has suggested that the North American and European isolates of *C. parasitica* arose from single or low numbers of original introductions of the pathogen.

**Life Cycle**

Because of the immediately obvious importance of the disease on chestnut at the time of the first outbreaks in early 20th Century, a great deal of research has been carried out on the life cycle of *C. parasitica*. Early work on spore dissemination (Heald et al. 1915) showed that the (sexual) ascospores were released in huge quantities from perithecia for up to 14 hours after rather small amounts of rainfall (4.5 mm). The asexual conidia, however, were not found in the atmosphere around the cankers, but were found to adhere in large numbers to potential vector animals, including insects, birds and arboreal mammals (Craighead 1912, Studhalter 1914, Studhalter and Ruggles 1915). In contrast, ascospores were rarely found on insects. Further work over subsequent years (Anagnostakis 1982a, Wendt et al. 1983, Russin et al. 1984) demonstrated that several insect species that wound bark tissues may vector the pathogen, including *Strophiona nitens* Forster and *Ectoedema phleophaga*. Folivorous species such as *Leptostylus maculata* also carry conidia. Compared with insects, the roles of birds and arboreal mammals in dissemination is less clear, although conidia are known to attach to both groups of animals from sporulating cankers (Heald and Studhalter 1914, Scharf and DePalma 1981).

The pathogen requires wounds for entry, but once in host tissues, mycelia proliferation is rapid, leading to the development of the symptoms described above.

2.3.6.2 *Damage significance*

The destruction of the American chestnut as a forest tree species in the first half of the 20th Century is a very striking illustration of the potential an alien invasive pathogen has for completely altering ecosystems. This disease is considered, along with white pine blister rust caused by *Cronartium ribicola*, as a textbook case for the study of the dangers posed by alien invasive pests and pathogens (Agrios 2005). In Europe, a natural event attenuated the virulence of the pathogen, leading to much lower impacts on *Castanea sativa* populations (see below).

2.3.6.3 *Importance of climatic factors*

Various environmental and climatic factors are implicated in pathogen growth and in disease development. Light affects the ability of the pathogen to sporulate; an alternating light and dark cycle is required for the formation of abundant pycnidia and the maturation of conidia (Anderson 1914). Copious pycnidia also form on the edges of wounds into infected bark tissues.

The temperature optimum for *in vitro* growth of *C. parasitica* is broad, from 21 to 32°C, regardless of light quality or photoperiod. Moreover, changing incubation temperatures from cool to within the optimum range has an immediate effect on growth rate (Anagnostakis and Aylor 1984). Growth will occur at temperatures above 5°C, but ceases at approximately 38°C. In trees, growth will occur at the same temperatures as those determined *in vitro*, but the growth rate is
approximately half of that occurring on agar under controlled conditions. Cankers expand at a rate of 1.0 – 1.3 mm per day, under ambient conditions in the field (Anagnostakis and Aylor 1984).

2.3.6.4 **FUTURE TRENDS UNDER CLIMATE CHANGE**

In Europe, the pathogen is present throughout much of the range where chestnut is grown for commercial production of the nuts. For timber production, however, the host range extends further North into northern France, Germany, and the UK. Moreover, the tree is present as an ornament in many botanic gardens and arboreta. To date these northern areas are not affected by canker.

Several different strains of *C. parasitica* are known in Europe (Bruellin et al. 2006) and, despite these strains being closely related, they may respond differently under climate change conditions. The presence of hypovirulence in the pathogen population, for example, has been shown to alter the host-pathogen interaction to variations in temperature (Byrer and Rigling 2011), favouring certain pathogen genotypes over others.

Changes in humidity may alter viability of pathogen spores over distance, but is unlikely to impact greatly on growth with host trees, as moisture conditions in tissues of *Castanea* are usually sufficient for *C. parasitica* to grow and disease to develop.

2.3.6.5 **MANAGEMENT**

**Fungicide treatments**

After the initial recognition of the chestnut canker problem in New York, Murrill (1906b) suggested pruning out the affected branches and spraying with Bordeaux mixture. These measures, however, failed to have an impact on disease incidence and spread. It was quickly realised, as the epidemic built up rapidly and spread throughout the native range of *C. dentate*, from Maine in the north-east to Alabama, that the disease was largely unstoppable using any disease management methods known at the time.

**Chestnut breeding**

A breeding programme, aimed at incorporating resistance into *C. dentata* was initiated by the US Department of Agriculture (USDA) at the Connecticut Agricultural Experimental Station in 1930. The aim was to create hybrids between resistant Chinese species of *Castanea* into *C. dentata*, but was success was limited; the USDA discontinued their programme in 1960, although work at the Connecticut Agricultural Experimental Station continued for some time (Anon. 1995b, Anagnostakis 1996d). Further breeding work started in the early 1980s, when the American Chestnut Foundation was established (Anon 1995b), with the aim of restoring a chestnut with the characteristics of *C. dentata* to the forests of the eastern states within 30 to 50 years. Variations in susceptibility of chestnut are known both within and between species, and in hybrids between species (Clapper 1952, Graves 1950, Anagnostakis 1992). Chinese and Japanese chestnut (*Castanea mollissima* and *C. crenata*, respectively) were far less susceptible than American chestnut to growth of the pathogen in inoculation tests (Anagnostakis 1992), whereas hybrids between American chestnut and the Asian species showed intermediate susceptibility.
In Europe, a chestnut breeding programme began in Italy in 1951 (Bazzigher 1981). By the end of the 1990s, over 40,000 *C. sativa* selections with medium to high resistance to *C. parasitica* were produced.

**Hypovirulence**

Fifteen years after chestnut blight was first found in Europe, it was noted that infected trees near Genoa in Italy were recovering from the disease (Biraghi 1951, 1953, Mittempergher 1978). Trees that before 1951 had few shoots older than 4 or 5 years were growing well, despite showing rates of infection as high as 85% (Biraghi 1953). Symptoms on the recovered shoots were atypical, with the pathogen restricted to the outer bark tissues and the cankers showing signs of healing. This striking discovery lead to the Food and Agriculture Organisation (FAO) sponsoring further work by a group of French scientists lead by Grente from Clermont-Ferrand. Isolates of *C. parasitica* Samples of the healing cankers showed atypical growth in culture, maintaining a white colony appearance, as opposed to the orange colours developing in aggressive isolates; moreover, the white isolates were far less aggressive in inoculation tests on chestnuts (Grente 1965, Figure 2-32). Inoculation of these ‘hypovirulent’ isolates into cankers caused by aggressive isolates cured the existing disease and it was shown that the aggressive also became non-aggressive in the presence of hypovirulent strains (MacDonald and Fulbright 1991). Hypovirulence spread naturally in stands of *Castanea sativa* and within 25 years, the impact of chestnut canker had reduced considerably in Europe. Hypovirulent isolates of the pathogen were later found in Michigan and, subsequently, in other states of the USA (Fulbright et al. 1983).

Since the discovery of hypovirulence, much research has gone into trying to develop an understanding of the phenomenon, so that it could be utilised to control chestnut canker, particularly in the badly damaged forests of the eastern USA. Hypovirulent isolates of the pathogen contain double-stranded ribonucleic acid (ds-RNA), similar to a virus infection (Milgroom and Cortesi 2004). This apparent infectivity of the ds-RNA suggests that, with suitable manipulation, the virulence of the destructive isolates of *C. parasitica* in North America may be attenuated if the ds-RNA was released into forested areas with coppice chestnut growth in the understory (Milgroom and Cortesi 2004). Attempts to generate these infections in the forest, however, have led to problems, as the fast growth of the coppice shoots of chestnuts favours development of rapidly growing aggressive strains of the pathogen, as opposed to the much slower growing ds-RNA infected strains. Hence, the absence of large chestnut trees in these forests prevents the hypovirulence from controlling aggressive isolates of the pathogen (Griffith 1989).

![Figure 2-32 Attenuated canker on Castanea sativa, showing evidence of hypovirulence](image)
2.3.7 Dutch elm disease

2.3.7.1 DESCRIPTION AND DAMAGE PATHWAY

<table>
<thead>
<tr>
<th>Species</th>
<th>Ophiostoma ulmi and O. novo-ulmi (Dutch elm disease)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Fungus: Ascomycota</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Alien</td>
</tr>
<tr>
<td>Distribution</td>
<td>Europe (Austria, Belgium, Bulgaria, Czech Republic,</td>
</tr>
<tr>
<td></td>
<td>Denmark, Estonia, Finland, France, Germany, Greece,</td>
</tr>
<tr>
<td></td>
<td>Hungary, Italy, Latvia, Lithuania, Luxembourg,</td>
</tr>
<tr>
<td></td>
<td>Netherlands, Norway, Poland, Romania, Slovakia,</td>
</tr>
<tr>
<td></td>
<td>Slovenia, Sweden, UK) and America</td>
</tr>
<tr>
<td>Host specificity</td>
<td>Elms</td>
</tr>
<tr>
<td>Spatial and temporal scale of impact</td>
<td>Widespread impacts, infrequent but long outbreaks</td>
</tr>
<tr>
<td>Interesting lesson learnt for management</td>
<td>Action must be taken rapidly and applied rigorously for any management attempts to be at all effective. Research has to be ready to examine and identify any potential threat, so that diagnosis can be highly accurate; it should not be constrained by previous assumptions or legislation.</td>
</tr>
</tbody>
</table>

The current outbreak of Dutch elm disease is one of the most significant diseases of trees known and has caused extensive losses in Europe, North America, and parts of western Asia. It has been described as the most destructive plant disease ever recorded (Brasier 1996). Previous outbreaks of relatively non-aggressive types of Dutch elm disease were well documented from 1918 to the mid 1960's; this outbreak led to the death of approximately 30% of infected trees, but many of them recovered from infection. In the UK, the recorded incidence of the disease had fallen by 1960, due to an apparent reduction in the virulence of the pathogen. From the early 1930's in the USA, however, a severe outbreak of Dutch elm disease was in progress, with the highly susceptible American species, Ulmus americana, suffering enormous losses. It is thought that the disease had entered North America on logs imported from Eastern Europe or Asia (Gibbs 1978a).

In the mid 1960's a severe outbreak began in southern England, and the strain of O. ulmi responsible was shown to be identical to that found in North America (Gibbs 1978b). The disease was imported into England on logs of elm from North America: an investigation showed that the pathogen was present in logs of U. thomasii brought into the UK from Ontario. By the end of the 1970's, over 75% of the elm population of southern England was dead or dying, approximately 17.5 million trees (Gibbs 1978b). Since that time, most trees in the UK have been killed as the disease moved northwards (Figure 2-33).
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Figure 2-33 Devastation caused by Dutch elm disease (Photo courtesy: USDA Forest Service via Dr. R. Jay Stipes, Virginia Polytechnic Institute and State University)

Elsewhere in Europe, Dutch elm disease has caused problems similar to those found previously in North America and in the UK, although there are still mature elms remaining in some isolated areas. The pathogen is present in the elm populations of southern Sweden and Norway, and is found in all countries further south in Europe (Brasier 1996).

It became apparent that the ‘strain’ of *O. ulmi* causing this second outbreak of Dutch elm disease differed in several respects from that causing the outbreak in the early 20th Century; sufficient evidence was found to raise the strain from North America to separate species status (Brasier 1991), and the name *Ophiostoma novo-ulmi* was adopted for the more aggressive strain. A second highly virulent strain of *O. novo-ulmi* exists and is spreading westwards in Europe, whereas the North American strain is moving eastwards. In some areas, both of these two strains are present and mating occurs, giving the possibility of further highly virulent strains emerging. To compound matters further, another related species, given the name *O. himal-ulmi* was isolated from a beetle breeding galleries in a single *Ulmus wallachiana* tree in the Western Indian Himalayas during a survey in that area for a possible source of *O. novo-ulmi* (Brasier and Mehortra 1995).

The disease is transmitted from tree to tree by various species of elm bark beetles, principally in the genus *Scolytus* (Webber and Brasier 1984), but the major factor in long-distance spread is the transport of infected logs (and firewood) around the world and within individual countries by humans.

**Life Cycle**

The life cycle of Dutch elm disease reflects the frequent associations that occur between bark beetles and fungal pathogens of tree. Several species of *Scolytus* may vector the pathogen in Europe: *S. scolytus* (Figure 2-34 a), *S. multistriatus* (Figure 2-34 b), *S. kirschii*, *S. pygmaeus* (Figure 2-34 c) and *S. laevis* (Figure 2-34 d; Webber and Brasier 1984, Webber 1990, Favaro and Battisti 1993). Elm bark beetles require declining elm trees for breeding purposes. Following mating, female beetles excavate a brood chamber in the secondary phloem tissues (inner bark) of
Panorama of the current risks caused by biotic agents to the delivery of goods and services

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Disturbances of EU forests caused by biotic agents

Declining elms, laying eggs at regular intervals during the excavation process. The eggs hatch and the grubs feed on the secondary phloem tissues for a period before pupating. The beetles overwinter as larvae in breeding galleries (Figure 2-35) in the bark of killed trees or in logs on the ground.

Figure 2-34 The principal Coleoptera vectors of the Dutch elm disease pathogen in Europe: (a) Scolytus scolytus (bar = 2 mm); (b) S. multistriatus (bar = 1 mm); (c) S. pygmaeus (bar = 0.5 mm); (d) S. laevis (bar = 1 mm). (Photos were from Bugwood: www.invasive.org)

Figure 2-35 Elm bark beetle larvae (photo: Nature conservation, Somerset County council)

Adult beetles may emerge from the bark when the air temperatures exceed approximately 16°C (Fransen 1939). Most flights occur, however, when mean daily air temperatures exceed approx.
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Disturbances of EU forests caused by biotic agents

21°C for *S. scolytus* or *S. laevis* (Anderbrant and Schlyter 1987). Until such conditions arise, the adults remain within the bark tissues. Although bark may be of equal thickness on northern and southern sides of a tree, emergence of beetles from southern aspects of the tree may occur up to one week earlier in spring, due to increased temperatures inside the bark facing the sun (Segonca and Liesse 1984, Trockenbrodt 1994). In poor weather conditions for flight, such as rain or cool temperatures, the beetles tend to remain on or in the brood tree, until conditions improve.

Several generations of beetles may emerge each year, depending on weather conditions. *S. scolytus* may produce 2 – 4 generations per year, *S. laevis* 1, *S. multistriatus* 1-3 and *S. pygmaeus* 2 (Lekander et al. 1977, Hansen and Somme 1994). The main factor influencing number of generations is the temperature.

Speed of flight for *S. scolytus* was reported to be 1.6 m per second, although flights rarely last longer than 2 hours. Wind speed affects distances travelled, however, and *S. scolytus* adults are rarely recovered more than a few hundred metres from their brood tree (Karnosky 1979). Beetles remain on the bark of the tree if wind speed exceeds 2 m per second. *S. multistriatus* will fly 400 m in a single flight (Lanier 1989) and *S. pygmaeus* may fly even several hundreds of metres in a single day (Minks and Van Deventer 1978).

Being highly humid, the beetle galleries provide ideal conditions for sporulation of the pathogen; *O. novo-ulmi* develops saprotrophically in the bark, producing (imperfect) coremial stages from the sides of the beetle galleries. During emergence of the adult beetles in the spring, coremiospores from the sides of the galleries adhere to the beetle elytra and are transported from the gallery. Immediately after emergence, the beetles must feed on the bark of shoots of nearby elm trees (maturation feeding; Figure 2-36); this feeding is essential for sexual maturation to be achieved. A proportion of the coremiospores on the beetles infects these wounds, germinate, and form hyphae in the exposed xylem vessels.

![Figure 2-36 Longitudinal cut through elm twig showing the wound resulting from maturation feeding and brown streaking characteristic of a wilt infection in the exposed xylem (photo: D.K. Barrett)](image)

Once inside the xylem vessels of the host tree, yeast-like budding of the pathogen occurs, and these bud cells (oidia) are swept upwards in the xylem sap. The pathogen produces toxins which are translocated rapidly, causing the symptoms of wilting in the foliage and death of affected shoots. One of the most recognisable symptoms of Dutch elm disease is the brown streaking produced in the affected xylem vessels (Figure 2-37); in addition to this browning resulting from
oxidation of phenolic compounds in the xylem companion cells, tyloses form in the affected vessels (Figure 2-38). Within weeks, dieback of the affected tree occurs, leading to death. The bark of the killed branches and the main stems then provides the breeding substrate for the next generation of beetles.

Figure 2-37 Cross-section through a Dutch elm disease-infected elm, showing the browning reaction surrounding affected xylem vessels (photo: D.K. Barrett)

Figure 2-38: Occlusion of xylem vessels by tyloses in elm infected with Dutch elm disease (photo: D.K. Barrett)

2.3.7.2 Damage Significance

Dutch elm disease infections result in death of almost all affected trees. Hence, the disease has had major impacts on landscapes and ecosystems in all affected areas. Moreover, populations of any organisms dependent on mature elms for the provision of food or habitat have reduced as a consequence of the enormous losses in elm numbers in Europe over the last 40 years. Elm timber was also highly valued, being hard and durable; supplies of the timber were very high for a number of years as the disease developed in Europe, but there is a shortage of this timber now.
In the UK, the less aggressive disease outbreak was first recorded in 1927, though ring analysis suggests trees were affected perhaps as early as 1920. The disease intensity peaked in 1936-37, after which numbers of affected trees recorded declined (Peace 1960). As the virulent outbreak of Dutch elm disease began in the UK in the 1960s (Brasier and Gibbs 1973a), the development of the epidemic and the destructive potential of the disease are best illustrated using data from that state. The outbreak of Dutch elm disease that spread rapidly in Europe in the early 20th Century led to losses of between 20 and 40% of elms (Brasier 1996).

Survey work on the outbreak showed very clearly that the new outbreaks occurred around major sea ports in England and Wales: London, Chatham, Southampton, Portsmouth and Avonmouth. Further data on points of introduction, which included all the major ports in the UK, are given in Harwood et al. (2011). Gibbs and Brasier (1973) demonstrated that the ‘new’, aggressive strain of the pathogen differed greatly in culture from the fungus causing the earlier outbreak in the UK; these conclusions were later substantiated using molecular methods, and the new species, Ophiostoma novo-ulmi was raised (Brasier 1991).

Modelling work on the spread of O. novo-ulmi in the elm populations of the UK has suggested introduction dates of either late 1964 (Swinton and Gilligan 1996), or November 1962 (Harwood et al. 2011). In the absence of a large scale management approach, the disease spread rapidly throughout southern Britain, and by the mid-1980s there were probably fewer than 10% of the original elm population left alive. Given the biology of the causal organism in relation to elm regrowth and phenology, it is likely that fluctuations in numbers of elm trees will continue in the foreseeable future: many elm root systems live on and re-grow to a point where the bark is once again suitable for beetle feeding. At this point, the trees become reinfected and died back again.

It has been argued (Tomlinson and Potter 2010) that the response of the UK authorities to the second and far more aggressive outbreak of Dutch elm disease in the mid-1960s was somewhat slow, based on the conclusions given by Peace (1960), that, by then, the disease was of relatively minor importance, with local outbreaks. It should be noted, however, that Peace (1962) did state that "it is impossible … to say with certainty that the decline [in disease levels] will continue, so that the future outlook remains rather uncertain.”

A massive and diverse group of organisms is associated with elm. For example, there are at least fifteen species of aphid associated with elm in Europe, some sucking sap directly from shoots, leaves and roots, others inducing the formation of galls on foliage or shoots. Not all of these species are threatened with extinction as the elm populations declined. Some species are not restricted to elm, and can persist on other plants. Obligate organisms may persist on the phenologically young elm that persist in hedgerows and other isolated areas. Populations of these associated organisms, however, have reduced drastically. Beyond the organisms that require elm for feeding, there are other species with a secondary need for elm, feeding on the organisms directly associated with the trees: the nematode and insect parasites of elm bark beetles described below fall into this category.

An insect that has been noted for the population decline with loss of elms is the white letter hair streak butterfly, Satyrium w-album, which requires large elm trees for successful completion of its life cycle. The recently available resistant cultivar of elm Morfeo (see below) provides suitable habitat for this species.
Of the large numbers of fungi associated with elm, few are dependent on elms alone. There are no known obligate mycorrhizal fungi on elms, for example. The decay-causing *Fomes ulmarius* appears confined to species of *Ulmus* in Britain, but is known from other woody hosts elsewhere in Europe. There are numerous Ascomycota found on elm, but it is unclear how many are solely found on this tree.

### 2.3.7.3 *IMPACT OF CLIMATIC FACTORS*

The main factor restricting the rate of range expansion of Dutch elm disease, is the minimum temperature required for the vector beetles to fly (see above). Optimum growth of *O. novo-ulmi* occurs at a wide range of temperatures, however, and the pathogen itself is unlikely to be restricted by climate change.

### 2.3.7.4 *FUTURE TRENDS UNDER CLIMATE CHANGE*

The most likely problem with climate change is the increasing latitude at which vector beetles will be active. Two factors arise: (1) elm bark beetles associated with more southerly latitudes, such as *Scolytus scolytus*, will migrate northwards; (2) as daytime temperatures increase, the number of days per year on which beetles can fly will increase. Hence, it is likely that the rate of spread of Dutch elm disease northwards will be faster in the near future.

### 2.3.7.5 *MANAGEMENT*

Several strategies were adopted in an attempt to prevent infections and reduce the impact of Dutch elm disease. A detailed knowledge of the life cycle of the pathogen lead to various approaches to management.

#### Control of Beetles

The elm bark beetles *Scolytus scolytus*, *S. multistriatus* and *S. laevis* (and several others) are responsible for vectoring the disease locally. A number of insect management methods were attempted to prevent vector-related spread of Dutch elm disease.

i) **Destruction of beetle breeding grounds**: sanitation felling of elms in an affected area, plus burning of felled material or careful removal of all bark and burning was recommended to reduce bark beetle populations. This method had many difficulties, particularly under the epidemic conditions, where large numbers of declining and dead elms of considerable size were available for the beetles to colonise. In addition, large numbers of beetles, particularly of *S. scolytus*, may emerge from a single log.

ii) **Chemical control**: some success was achieved in areas of the USA through the application of insecticides, although the technique is generally difficult and expensive, involving repeated spraying in the early part of the growing season. Systemic insecticides may give better beetle control than contact chemicals, but do tend to be more toxic to mammals. Overall there are strong environmental objections to the use of insecticides. Moreover, applications of chemicals would certainly miss many broods, and the (potentially) hundreds of beetles arising from these escapes would vector a new wave of disease.
iii) **Biological control**: several potential biological control organisms exist for controlling the beetles, including parasitic wasps, *(e.g. Dendrossiter and Celoides spp.)* and parasitic nematodes. Pheromone traps are used in management of Dutch elm disease, usually in order to monitor beetle populations.

One of the main problems with most biological control agents is that they achieve a balance in populations between the pest and the control agent, rather than resulting in complete eradication. With Dutch elm disease, a single beetle may carry many coremiospores and cause several infections.

▶ **Control of the Pathogen**

i) **Chemical control**: Protection to elms given by fungicide injection is at best 30 to 40 %, depending on the size of the tree. On trees over 25 metres in height, effectiveness is much reduced. This type of treatment is very expensive, and is required each year; hence it is only recommended for trees with high amenity values.

ii) **Biological control**: An intriguing possibility for the biological control of *O. novo-ulmi* is the use of mycoviruses. Several viruses have been characterised from the three Dutch elm disease pathogens *(Hong et al. 1998, 1999, Crawford et al. 2006)* /factors in *O. novo-ulmi*.

▶ **Breeding for Resistance**

Large elm resistance breeding programmes were instigated in the USA (e.g. in 1920s) and the Netherlands (in 1928, ending in 1994), mainly attempting to exploit the natural resistance found in the Asian species of *Ulmus*. Some European species of elm also appear to be more resistant to the disease, for example *Ulmus laevis*, which is found naturally in eastern parts of Europe.

In 1936, the Dutch breeding programme released a cultivar named ‘Christine Buisman’, a selection of *U. carpinifolia*, which showed good resistance to *O. ulmi*. However, the cultivar had an increased susceptibility to *Nectria cinnabarina*, which caused severe cankering and dieback. Other cultivars selected in this programme, for example ‘Commelin’ and ‘Groenveld’, were not very resistant to the aggressive species, *O. novo-ulmi*.

In 1975, three clones were released by the Dutch breeding programme, which have shown moderate resistance to *O. novo-ulmi*:

- **Lobel** (*U. glabra ‘Exoniensis’ x U. wallachiana) x U. hollandica ‘Bea Schwarz’*
- **Dodoens** (*U. glabra ‘Exoniensis’ x U. wallachiana) F₁ hybrid
- **Plantjin** (*U. glabra ‘Exoniensis’ x U. wallachiana) x *U. carpinifolia*

A fourth cultivar, **Clusius** (*U. glabra ‘Exoniensis’ x U. wallachiana) x *U. hollandica* is available. ‘Bea Schwarz’, another selection, was released in the 1980s, and has greater resistance to *O. novo-ulmi*. Recently, a new resistant clone, Morfeo, from the Istituto per la Protezione delle Pianta, CNR, Firenze, has been tested in Mediterranean areas, and in southern Britain, showing high levels of resistance to the pathogen *(Santini et al. 2011)*.

Several resistant cultivars of elm were released over many years from the North American breeding programmes:
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- **Sapporo Autumn Gold** (F₁ hybrid of *U. pumila* x *U. japonica*)
- **Regal** (*Commelin* x *U. pumila* x *U. carpinifolia*)
- **American Liberty** (selected from survivors of 60,000 seedlings of *U. americana* collected from throughout the natural range of this tree species)
- **Urban elm** (*U. hollandica* ‘Vegeta’ x *U. carpinifolia*) x *U. pumila*
- **Homestead** (*U. pumila* x (*(*U. hollandica* ‘Vegeta’ x *carpinifolia*) x *U. pumila*) x *U. carpinifolia*)
- **Pioneer** (*U. glabra* x *U. carpinifolia*)
- **Valley Forge** and **New Harmony** (from *U. americana* screening programme)
- **Jacan** (Canadian selection of *U. japonica*)
- **Thompson** (Canadian selection of *U. japonica*)
- **Mitsui Centennial** (Canadian selection of *U. japonica*)

However, hybrids bred under other conditions may not prove particularly resistant to Dutch elm disease in all areas of Europe, where the vectors and climatic conditions differ from those in the regions where breeding took place. Moreover, many of these resistant cultivars do not have the classical elm form required to replace the elm as the former significant component of the northern European landscape.

**Genetically engineered** elms, resistant to Dutch elm disease were developed by Professor Kevan Gartland and the University of Abertay, Dundee in the early 21st Century, but were never planted outside highly controlled conditions due to public opposition to the use of genetic modification (Anon. 2001).

**Legislation**

In the UK, a number of elm disease orders were in force from early in the outbreak of the *O. novo-ulmi* epidemic. Regulations aimed to restrict the spread of the disease through felling of infected trees and destruction of the infected material on site, usually by burning. Where regulations were enforced most stringently, there was some success in control, e.g. in East Sussex (Brighton, Eastbourne, Hove). In this area of England, there is some isolation from outlying elm populations on three sides due to the sea, the Pevensey Levels and the South Downs. The UK Dutch elm disease Orders were revoked in 1996, having failed in the attempt to prevent spread of the disease.

Other MS had also applied legislation in an attempt to reduce the rate of spread of the pathogen; these attempts have also largely failed.

**Conclusions and recommendations**

Ultimately, however, none of the management methods trialled or put in place has prevented rapid spread of the disease and loss of the vast majority of elms in Europe.
Large elms can still be found in a few localities, however, including Amsterdam and The Hague, and in and around the city of Brighton and Hove on the southern coast of Britain. The number of elms in Amsterdam was estimated at 75,000 in 2007 (Anon. 2007). In each of these cities, the local authorities have maintained high vigilance against the disease, with sanitation fellings of any trees showing symptoms and monitoring of beetle populations using kairomones (Anon, no date, a; Greenland no date). Many trees also receive regular fungicide injections to try to eradicate infections. Trees that are removed are replaced with resistant varieties of elm. Brighton and Hove designated an ‘Elm Disease Management Area’ within which the rigorous sanitisation policy was implemented.

The National Collection of Elm for the UK is held in Brighton and Hove (Anon. no date, b). In 2010, however, there was a large increase in the population of elm bark beetles in the city, and the collection is now considered as under threat.

The best hope for the continuation of elm, as mature trees, lies in breeding for resistance. Successes in this particular field of research continue to be released and are undergoing long-term trials in many parts of Europe.
2.3.8 Phytophtora ramorum

2.3.8.1 DESCRIPTION AND DAMAGE PATHWAY

<table>
<thead>
<tr>
<th>Species</th>
<th>Phytophthora ramorum (Sudden oak death)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Taxon</td>
<td>Oomycota</td>
</tr>
<tr>
<td>Native/Alien</td>
<td>Alien</td>
</tr>
<tr>
<td>Distribution</td>
<td>Europe (Austria, Belgium, Bulgaria, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Italy, Latvia, Lithuania, Luxembourg, Netherlands, Norway, Poland, Romania, Slovakia, Slovenia, Sweden, UK) and North America</td>
</tr>
<tr>
<td>Host specificity</td>
<td>Very broad host range, particularly Ericaceae and Fagaceae. Recent host jump to Larix kaempferi in western UK.</td>
</tr>
<tr>
<td>Spatial and temporal scale of impact</td>
<td>Widespread impacts</td>
</tr>
<tr>
<td>Interesting lesson learnt for management</td>
<td>The very high level of importance that should be attached to the ornamental nursery stock industry in terms of providing an effective route for dispersal of alien invasive pests and pathogens.</td>
</tr>
</tbody>
</table>

Sudden Oak Death was a common name applied to killing of trees in the Fagaceae, mainly tan oak (Notholithocarpus densiflorus syn. Lithocarpus densiflorus) in the coastal strip of mid-northern California in the mid-1990s when the causal agent was not known (Figure 2-39). The killing resulted from infections by the Oomycete Phytophthora ramorum, which was first recorded, though not formally described, causing disease in nurseries and gardens in 1993. The disease on garden shrubs was largely ignored until the major outbreaks began in the native forests in California (Rizzo et al. 2002). Despite the developing importance of the problem in California, the causal organism was first formally described and named in Europe in the early 21st Century (Werres et al. 2001), where it was causing problems in plant nurseries, principally on cultivars of Rhododendron and Viburnum. Molecular evidence, however, suggests that the introductions of P. ramorum into North America and Europe were separate events, as the isolates found in the two continents are distantly related within the species (Ivors et al. 2004, 2006). Since the first discovery of P. ramorum in Europe, it has spread to at least 21 European MS (Walters et al. 2010). The first outbreaks on European forest trees were reported from the UK by Brasier et al. (2004).
Panorama of the current risks caused by biotic agents to the delivery of goods and services

Disturbances of EU forests caused by biotic agents

In California, *P. ramorum* kills tanoak (*Lithocarpus densiflorus*), coastal live oak (*Quercus agrifolia*), California black oak (*Quercus kelloggii*) and Shreve oak (*Quercus parvula var. shrevei*). Symptoms on these oaks include bleeding cankers on the stem exuding red liquid (Figure 2-40). Foliage on infected trees has a characteristic darkening of the main vein (Figure 2-41). Although *P. ramorum* infections on these oak family species are lethal, the pathogen does not sporulate on these ‘dead-end hosts’.
In European forests, *P. ramorum* has mainly been recorded on trees in *Fagaceae*, both *Fagus* and *Quercus* spp. (Brasier et al. 2004) and, more recently, *Nothofagus* (Webber et al. 2010). Understory shrubs, particularly *Rhododendron ponticum* and *Vaccinium* spp. have, until very recently, been the recognised sporulating hosts in Europe. In 2009, however, an unpredicted host jump was reported, when serious mortality was observed on Japanese larch (*Larix kaempferi*) trees in plantations in the south-west of England (Brasier and Webber 2010). It is possible that this host jump had occurred one or two years earlier in Eire (Ireland) but had gone unrecognised. Over the three years since the first reports of *P. ramorum* on Japanese larch, the disease has spread throughout the western seaboard of Britain and Ireland, killing this particular tree. Moreover, the pathogen has also been found killing European larch (*L. decidua*) and attacking other tree species, including Sitka spruce (*Picea sitchensis*) and grand fir (*Abies grandis*) under conditions of high spore loads coupled with high relative humidities. Clearly, *P. ramorum* poses a highly significant threat to forest ecosystems in high humidity environments.

*P. ramorum* has a very wide host range, on which it may cause either fatal or non-fatal infections including stem cankers, twig dieback and leaf lesions. Apart from forest trees, many popular ornamentals are affected, although the main hosts in this sector are species and cultivars of *Rhododendron*, *Camelia* and *Viburnum* (Werres et al. 2001, Brasier et al. 2004). A current list of known host trees and shrubs is given in Table 2-13. Many of these shrubs are sporulating hosts, and serve as major sources of spores for disease spread and, through human transport in the ornamentals trade, long distance transfer of the pathogen.

<table>
<thead>
<tr>
<th>Scientific Name</th>
<th>Common Name</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Abies grandis</em></td>
<td>Grand fir</td>
</tr>
<tr>
<td><em>Acer macrophyllum</em></td>
<td>Bigleaf maple</td>
</tr>
<tr>
<td><em>Acer pseudoplatanus</em></td>
<td>Sycamore maple</td>
</tr>
<tr>
<td><em>Adiantum aleuticum</em></td>
<td>Western maidenhair fern</td>
</tr>
<tr>
<td><em>Adiantum jordanii</em></td>
<td>California maidenhair fern</td>
</tr>
<tr>
<td><em>Aesculus californica</em></td>
<td>California buckeye</td>
</tr>
<tr>
<td><em>Aesculus hippocastanum</em></td>
<td>Horse chestnut</td>
</tr>
<tr>
<td><em>Arbutus menziesii</em></td>
<td>Madrone</td>
</tr>
<tr>
<td><em>Arctostaphylos manzanita</em></td>
<td>Manzanita</td>
</tr>
<tr>
<td><em>Calluna vulgaris</em></td>
<td>Heather</td>
</tr>
<tr>
<td><em>Camellia</em> spp.</td>
<td>Camellia—all species</td>
</tr>
</tbody>
</table>

28 For updated lists visit: www.aphis.usda.gov/plant_health/plant_pest_info/pram/ or rapra.csl.gov.uk/objectives/wp1/naturalhostsearch.cfm
### Scientific Name | Common Name
--- | ---
Castanea sativa | Sweet chestnut
Castanopsis orthacantha | Castanopsis
Erica carnea & varieties | Bell heather
Fagus sylvatica | European beech
Frangula purshiana | Cascara
Fraxinus excelsior | European ash
Griselina littoralis | Griselinia
Hamamelis virginiana | Witch hazel
Heteromeles arbutifolia | Toyon
Kalmia spp. | Mountain laurel—all species
Lithocarpus densiflorus | Tanoak
Lonicera hispidula | California honeysuckle
Larix decidua | European larch
Larix kaempferi | Japanese larch
Laurus nobilis | Bay laurel
Magnolia doltsopa | Michelia
Maianthemum racemosum | False Solomon’s seal
Michelia dolosopha | Michelia
Nothofagus oblique | Nothofagus
Osmanthus delavayi | Osmanthus
Parrotia persica | Persian ironwood
Photinia fraseri | Red tip photinia
Pieris spp. | Pieris—all species
Pseudotsuga menziesii var. menziesii | Douglas fir
Quercus agrifolia | Coast live oak
Quercus cerris | European turkey oak
Quercus chrysolepis | Canyon live oak
Quercus falcata | Southern red oak
Quercus ilex | Holm oak
Quercus kelloggi | California black oak
Quercus parvula var. shrevei | Shreve oak
Panorama of the current risks caused by biotic agents to the delivery of goods and services

<table>
<thead>
<tr>
<th>Scientific Name</th>
<th>Common Name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quercus petraea</td>
<td>Sessile oak</td>
</tr>
<tr>
<td>Rhododendron spp.</td>
<td>Rhododendron—all species</td>
</tr>
<tr>
<td>Rosa gymnocarpa</td>
<td>Wood rose</td>
</tr>
<tr>
<td>Salix caprea</td>
<td>Goat willow</td>
</tr>
<tr>
<td>Sequoia sempervirens</td>
<td>Coast redwood</td>
</tr>
<tr>
<td>Vaccinium myrtillus</td>
<td>Bilberry; blaeberry</td>
</tr>
<tr>
<td>Viburnum spp.</td>
<td>Viburnum</td>
</tr>
</tbody>
</table>

The most consistent and diagnostic symptom of the disease on larger trees is the development of cankers that have brown or black discoloured outer bark on the lower trunk and that seep dark red sap (Garbelotto et al. 2001). These cankers develop before foliage symptoms become evident. Trees die rapidly after infection.

Symptoms on many shrubs (Table 2-13) are usually dieback of infected shoots and blight on the foliage. Although sometimes fatal, many species tolerate infection for considerable periods of time.

Life cycle

*Phytophthora ramorum* is unusual amongst species in the genus occurring in temperate forest ecosystems in having an important aerial component of its life cycle (Davidson et al. 2005, 2008), although more recently aerial life cycles have been noted in a number of other *Phytophthora* hitherto assumed to be entirely soil-borne, such as *P. lateralis* (Robin et al. 2010, Webber et al. 2011), *P. plurivora* and *P. cambivora* (Nechwatal et al. 2011).

In California, sporulation of the pathogen is at a maximum between the months of December and May when climate is cool and humid (Davidson et al. 2005). The main sporulating hosts in Californian forests are the California bay laurel (*Umbellularia californica*; Davidson et al. 2005). Infections on oak family trees are non-sporulating, giving rise to the phrase ‘dead-end hosts’ for these trees (Garbelotto et al. 2003).

Numbers of sporangia produced by *P. ramorum* vary with host and (probably) with climatic conditions prevalent at the time of sporulation. On excised shoots of Japanese larch, for example, numbers of sporangia varied from 0 to 2685 per needle, although more typically 12 – 612 sporangia per needle were recorded (Webber et al. 2010); this latter figure equates to approximately 7.25 million and 360 million sporangia per square metre of larch foliage (Woodward, unpublished report).

*P. ramorum* exists in two mating types, A1 and A2; most isolates obtained in Europe to date have been of the A1 mating type, whereas most isolates in North America are A2 (Kleijunas 2001). In laboratory work, however, it has been difficult to obtain mating between A1 and A2 isolates; moreover, the majority of oospores arising from these matings are abnormal in structure and have failed to germinate (Brasier et al. 2007). The additional threat of matings between A1 and
A2 genotypes remains, however, possibly leading to further changes in behaviour of this destructive pathogen (Garbelotto et al. 2005).

Molecular evidence indicated that genetic variation in *P. ramorum* isolates was considerably higher in nursery populations than in forests (Ivors et al. 2004, 2006). The same analyses indicated that isolates obtained in North America were less diverse than those from Europe; those from forests of the USA appeared to be near clonal. The presence of three clades of the pathogen in US nurseries strongly supported the suggestion that *P. ramorum* is transported in the plants for planting pathway. Further molecular work has demonstrate that the transport of plants for planting in the nursery industry was the major route for dissemination of the pathogen in North America (Goss et al. 2009). Given frequent isolations of *P. ramorum* in nursery plants in the trade in regions hypothetically unsuitable for the pathogen (e.g. Greece; Tsopelas and Soulioti, personal communication), it is clear that the same dissemination route operates in Europe (Xu et al. 2009).

Although considered primarily a foliar/shoot pathogen, there is considerable evidence that *P. ramorum* may also be transported from nurseries in soil, or in the roots of plants (Fichtner et al. 2007, Park and Lewis 2007, Shishkoff 2007). Hence, this transport process may enable infected but symptomless plants to remain undetected in the nursery trade for some time, promoting the further unwitting dissemination of the pathogen.

### 2.3.8.2 Damage Significance

*Phytophthora ramorum* causes serious landscape-scale damage to environments (Figure 2-39), ecological losses and also major financial losses in the valuable plant nursery industry.

Coast live oak (*Quercus agrifolia*) is a keystone species of California’s coastal woodlands, providing habitat for a wide range of insects, birds and mammals. It forms a vegetation type in central and southern California that is primarily single species (78% of basal area) and is relatively dense compared with other oak species in California (Waddell and Barrett, 2005). Being such a dominant species in the forested areas, the loss of coast live oak presents a major ecological disaster in the region.

Modelling work focused on several birds associated with cast live oak in California (Monahan and Koenig 2006) suggested that the disease could have major impacts on populations of species such as the acorn woodpecker (*Melanerpes formicivorus*), Nuttall’s woodpecker (*Picoides nuttallii*), Hutton’s vireo (*Vireo huttoni*), western scrub-jay (*Aphelocoma californica*), and oak titmouse (*Baeolophus inornatus*), with predicted reductions in numbers as drastic as 68%, and variability of 49%.

After *P. ramorum* was first reported in Oregon, total eradication treatments were applied to affected forest areas. Between 2001 and 2008, approximately 960 ha of forest were treated, costing US$4.3 million (Kanaskie et al. 2009).

In 2009, the regulating authority for England and Wales (The Department for Food and Rural Affairs) allocated £4 million per year for the next 3 years towards management measures for *P. ramorum*, in a bid to reduce the rate of spread.
In 2010, the first reports of *P. ramorum* killing Japanese larch in the UK suggested that over 2,000 ha of plantations were affected, principally in the south-west of England and south Wales (Brasier and Webber 2010). Since that time, however, the significant damage to larch plantations in Eire and Northern Ireland has been recognised, and the problem has spread as far north as the west coast of Scotland. The disease clearly, therefore, poses a highly significant threat to the 120,000 ha of Japanese larch planted in the UK, and the significant plantations of the same species in other MS, where *Larix kaempferi* because of the disease problems impacting on European larch (*L. decidua*). Moreover, in areas with extremely high spore loads, as occurs with *P. ramorum* on *L. kaempferi* (Webber et al. 2010), other tree species also become infected, including birch (*Betula pendula*), sweet chestnut (*Castanea sativa*), grand fir (*Abies grandis*), Douglas fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*). In affected areas of Ireland, the pathogen has been observed on the key UK and Ireland timber-producing species, Sitka spruce (*Picea sitchensis*), and on European larch (Hamilton, A-M., personal communication, July 2011).

As *P. ramorum* is prevalent in the nursery industry, a large economic burden has been imposed on the nursery sector in an attempt to reduce the chances of further widespread transport of the pathogen. Data from North America amply illustrate the devastating effect the pathogen has had on trade (Frankel 2008). During the National Nursery Survey for the pathogen in 2004, several nurseries in Southern California were found to be affected and tracing of stock indicated infection in a large nursery in Washington State. Millions of plants had been shipped from these nurseries to 1200 other nurseries in 39 other states of the USA; this news led to 15 states placing an outright ban on nursery stock from California, with consequent losses estimated at US $4.3 million to CA’s nursery industry in March alone. The California Department of Food and Agriculture and US Department of Agriculture have cooperative agreements in place for the detection and treatment of infections, which in the 6 years from July 2002 to 30 June 2007 cost some c. US$25 million.

Costs of *P. ramorum* detection alone in Oregon nurseries (55 of 6067 proved positive) between 2001 and 2005 were estimated at US$ 3.2 million. The Canadian government rapidly recognised the threat posed by plants imported from the Pacific Coast states, and in 2001 suspended imports from Oregon and California. It was estimated that Oregon nurseries have lost approximately US$15–20 million per annum as a result of this ban.

Nurseries also face many other economic impacts, if found positive for *P. ramorum*: the costs of removal and destruction of affected plants, fumigation of affected areas, loss of customers, the ban on re-use of materials and additional sanitation and cleansing protocols that must be implemented.

In the USA, nurseries producing susceptible stock have altered management protocols, for example, through interspersing rows of susceptible plants with rows of non-suspects (Suslow 2005). Prophylactic applications of pesticides may also be used. Where the stock has come in from a nursery subsequently shown to have the disease, further management methods are used. All of these management methods increase costs to the nursery owner.
2.3.8.3 **IMPORTANCE OF CLIMATIC FACTORS**

*P. ramorum* has optimal growth in vitro at 18 - 22°C, similar to other aerial *Phytophthora* species. The propagules (sporangia) are produced on leaf and young shoot surfaces of sporulating are dispersed by rain splash to neighbouring plants, or are windblown in the rain (Davidson et al. 2005). Very high humidity is required for both survival and spore production. In the coastal forests of California, winter rains provide the conditions required for the pathogen to survive and flourish.

To date, the areas where *P. ramorum* has caused most damage in forests in the world are the coastal strip of California, in the south-west of the UK (counties: Cornwall, Devon and Somerset) and more recently in Eire and Northern Ireland. Hence the climatic types typified in these regions, mild and very humid, have been used as the basis for modelling the potential for disease spread in Europe. Not surprisingly, regions on the Atlantic seaboard present the most suitable climatic zones for further proliferation of *P. ramorum*. On a broad basis, these regions include Portugal, Galicia, Western France, the Western part of the British Isles, South-West Norway, and Mediterranean areas such as Dalmatia (Kluza et al. 2007, Sundheim et al. 2009, Kliejunas 2010).

In California, more detailed modelling aiming to determine the importance of the disease in terms of landscape and at the level of the forest stand (Holdenrieder et al. 2004, Rizzo et al. 2005, Frankel 2008), has demonstrated that many years of fire suppression was almost certainly a factor in the rapid spread of the disease (Meentemeyer et al. 2008). Reductions in the numbers of fire events lead to an increase in the abundance of understorey species capable of acting as reservoirs of infection and sporulating hosts, leading to more rapid and intensive spread of the pathogen. Recent attempts to prepare parallel models for *P. ramorum* in Europe have again strongly emphasised the importance of human transport in disease spread (Chadfield and Pautasso 2011). Although this work, focused on England and Wales, did not include data from the more recent host jump of the pathogen to Japanese larch, it indicated that disease incidence was likely to reduce from west to east (wetter to dryer climatic conditions). The volume of trade in susceptible host plants was another major factor in the model accounting for disease spread.

2.3.8.4 **FUTURE TRENDS UNDER CLIMATE CHANGE**

Predictions of increases in rainfall with climate change in Western Europe will strongly favour diseases caused by pathogens in the genus *Phytophthora*, because of the importance of free water in the life cycle, for the spread and infectivity of zoospores (Deprez-Loustau et al. 2007). Moreover, increasing temperatures will enable greater numbers of propagules to survive, whether in those *Phytophthora* species that form oospores, or those relying on survival in host tissues or as chlamydospores. *P. ramorum* requires very high humidity atmospheric conditions to expedite the aerial dispersal of the sporangia (spores). It has been predicted, based on CLIMEX-modelling in the USA (Venette and Cohen 2006) that increases in rainfall and occult precipitation in currently unsuitable areas will lead to further *P. ramorum* outbreaks outside areas of present suitability. Specifically, the areas favourable to *P. ramorum* infection in the eastern USA will reduce, but further areas will become more suitable for disease development and spread in the western states of California, Oregon and Washington.
2.3.8.5 MANAGEMENT

With such a potentially serious pathogen, a large number of control and management techniques have been suggested and tested. Very strict quarantine regulations are in place worldwide, in an attempt to prevent further spread of the pathogen through trade in ornamental plants (Anon. 2006, 2008). Despite these measures, the pathogen continues to spread in suitable environmental conditions and is one of the most significant threats to forest ecosystems known. Moreover, spread in the nursery trade continues, particularly through the transport of symptomless hosts.

- **Use of chemical control measures**

Application of chemical control measures have had some success in nurseries and private and municipal gardens. Based on the successful results obtained against *P. cinnamomi* in Western Australia (Guest et al. 1995, Hardy et al. 2001), phosphite was quickly registered for use against *P. ramorum* in California. When combined with the organosilicate surfactant ‘PentraBark’ (Agrichem, Medina, OH), phosphite can be painted or sprayed onto the main stem, resulting in good control of infections without wounding (Garbelotto et al. 2007).

Other fungicides have more mixed effects on the pathogen; moreover, different isolates of *P. ramorum* show differing sensitivity to chemicals including copper hydroxide, copper sulphate, phosphonates (phosphite preparations), maneb, and metalaxyl (Garbelotto 2004).

- **Prevention**

In nurseries, prevention of *P. ramorum* damage should be possible (Kleczewski et al. 2008). As with any problem, prevention is the most economically appropriate technique to avoid damage. Ideally, nurseries should trade in disease-free stock, although this laudable aim is, in reality, extremely difficult to achieve. In depth training of nursery staff, so that they are aware of the symptoms of infection, and of methods for avoid infection transfer, must be implemented. Moreover, if the problem is discovered, it must be reported to the phytosanitary authorities immediately. Similar processes should be applied to forests, although the symptoms and impacts may differ considerably in these situations.

- **Cultural Practices**

All equipment used in handling and processing the plants should be sanitised appropriately to avoid spreading pathogen propagules (Kleczewski et al. 2008). Any infected plant material (live plants, plant debris) must be removed and burned as soon after the pathogen is discovered as is possible (Turner et al. 2006). Irrigation systems in the nurseries should also be monitored for infective propagules of the pathogen; sand filters or other suitable sterilisation systems should be installed to eliminate pathogen propagules in recycled irrigation waters (Themann et al. 2002, Tjosvold et al. 2008).

Machinery used in affected forests and plantations should be thoroughly cleaned before moving to other sites to reduce the chances of transferring infection. Timber extracted from affected forests should be processed by ‘quarantine sawmills’, which have been inspected and instructed in how sanitize the sawmill lines (e.g. Anon. 2011).
Landscape

Infections may commonly be found in plants used for landscaping, resulting from transport of infected individuals from nurseries to planting situations. Such plants should be removed and destroyed by burning; it is also advisable to remove and destroy susceptible plant species and varieties from the vicinity of the confirmed infections.

Forest

In Oregon, the sudden oak death containment plan now comprises early detection, delimitation of infested areas, treatment of affected plants, research and monitoring for further outbreaks and reduction in the density of host trees in areas of high risk of disease outbreak and spread (Kanaskie et al. 2009). Treatment comprises cutting and burning all infected hosts and any potential hosts within 30 – 100 m of the infected plants. This strategy is unlikely to give long-term control, however, as the range of hosts which may produce sporangia is so wide. Chemical treatments are not feasible in natural areas, due to the environmental damage they may cause. Recent work jointly between University of California and Ohio State University has suggested that some host resistance may be present, at least in coast live oak, which may be exploited in the future (McPherson et al. unpublished).

Legislation

The potential for *P. ramorum* to cause massive damage was recognised soon after the causal agent was described. Quarantine regulations were deployed rapidly in both North America and Europe, to reduce the risk of spread in the ornamental nurseries industry. The Pest Risk Analysis (PRA) for Europe was published online in 2009 (Sanford 2009). Despite the legislation, however, susceptible host plants continue to be transported internationally and nationally, leading to further spread of the pathogen.
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Chapter 3: Towards ecosystem management for adaptation and mitigation

3.1 Key points

- EU forests are naturally very diverse in terms of their natural dynamics. Disturbance regimes vary along a continuum from large-scale disturbances (fire, wind, floods and insect outbreaks) to small-scale or localised disturbances (gap formation caused by fungi, insects, and single uprootings of trees).

- Three main groups of disturbance regimes can be distinguished as: successional dynamic (even-aged) caused by large scale disturbance agents such as fire, wind or large-scale insect outbreaks which initiate succession and allow forests to regenerate simultaneously, with a single cohort of trees over large areas; cohort dynamic (uneven-aged), where adaptations to low intensity disturbances produce stands with several aged cohorts; and gap dynamic (all-aged), in the absence of large external disturbances the death of single tree or groups of trees maintains the formation of gaps in which more or less shade tolerant trees can regenerate.

- Given that the occurrence of biotic disturbances is more common in forest stands that (1) form monocultures, (2) are fertilised, (3) grow on degraded sites, or (4) have a low degree of naturalness, forest management systems should be adapted to encourage a natural composition, structure and functioning of forest stands and landscapes.

- There is a diversity of forest management and traditional landscape management systems available to improve the match between natural dynamics and management system in the EU. These will likely lead to future challenges in terms of forest governance aiming for an integrated landscape approach to spatial planning across forest land ownerships.

3.2 Natural forest dynamics as a benchmark

As stated after the Forest Europe summit in June 2011 in the context of launching negotiations for a legally binding agreement on forests, “the health and sustainability of Europe’s forests are essential in solving the challenges posed by climate change, biodiversity protection and fresh water supplies. Forest vitality is also crucial for fostering a green economy in Europe and across the globe.” Logically, understanding of and considerations to the diversity of forest ecosystem dynamics...
and their composition, structure and function is crucially important.

The diversity of forest types in a landscape is determined by the interaction between non-biotic and biotic disturbance factors. Any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment is considered as a disturbance (Pickett and White 1985). Soils, topography, climate and access to nutrients and water form the basis of the biophysical landscape and determine the range of possible compositions of tree species (e.g. Ellenberg 1996). Finally, the composition and structure of forests are modified by different kinds of interactions and disturbances. These range from non-biotic (e.g. fire, wind, water) to biotic (e.g. grazing, browsing, predation) and anthropogenic (e.g. clearing, livestock grazing) (see Pickett and White 1985, Peterken 1996, Ellenberg 1996, Angelstam 1996, 1998, Esseen et al. 1997, Kirby and Watkins 1998, Engelmark 1999, Engelmark and Hytteborn 1999).

As a consequence, different combinations of disturbances create characteristic disturbance regimes (sensu Pyne 1984, Pickett and White 1985). Disturbance regimes vary along a continuum from large-scale disturbances (fire, wind, floods, and insect outbreaks) to small-scale or localised disturbances (gap formation caused by fungi, insects, and single uprootings of trees).

In an attempt to introduce the importance to understand the diversity of natural disturbance regimes in EU’s forest, the considerable variation of the role of interacting biotic and abiotic forces in forest and woodland systems need to be understood. We distinguish three groups of disturbance regimes following the logic presented by Dyrenkov (1984), who distinguished the following main types of the stand age structures: even-aged, uneven-aged, and all-aged. The trees that develop after a single disturbance event, whether minor or major, represent a cohort or a generation (Smith et al. 1997). With this terminology even-aged stands are stands that contain a single cohort that develops through successional development. Similarly, uneven-aged stands have two or just a few cohorts, and all-aged stands have many cohorts that develop simultaneously in the stand. To avoid this confusion of terms, and stress the dynamic characteristics of each forest type we use the words successional, cohort and gap dynamics to describe the three types of forest dynamics (Table 3-1). Equivalence between the EEA categories of forests and the more general classification used throughout the report is available in Annex 1.
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Table 3-1 Tentative summary of the profile of different natural disturbance regimes according to forest type (see Annex 1 and EEA 2008)

<table>
<thead>
<tr>
<th>Categories</th>
<th>DISTURBANCE REGIME</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>SUCCESION</td>
</tr>
<tr>
<td>1. Boreal forest</td>
<td>+</td>
</tr>
<tr>
<td>2. Hemiboreal forest and nemoral coniferous and mixed broadleaved-coniferous forest</td>
<td>+</td>
</tr>
<tr>
<td>3. Alpine coniferous forest</td>
<td>(+)</td>
</tr>
<tr>
<td>4. Acidophilous oak and oak-birch forest</td>
<td>+</td>
</tr>
<tr>
<td>5. Mesophytic deciduous forest</td>
<td>(+)</td>
</tr>
<tr>
<td>6. Beech forest</td>
<td>+</td>
</tr>
<tr>
<td>7. Mountainous beech forest</td>
<td>+</td>
</tr>
<tr>
<td>8. Thermophilous deciduous forest</td>
<td>+</td>
</tr>
<tr>
<td>9. Broadleaved evergreen forest</td>
<td></td>
</tr>
<tr>
<td>10. Coniferous forests of the Mediterranean, Anatolian and Macaronesian regions</td>
<td>(+)</td>
</tr>
<tr>
<td>11. Mire and swamp forest</td>
<td></td>
</tr>
<tr>
<td>12. Floodplain forest</td>
<td>+</td>
</tr>
<tr>
<td>13. Non riverine alder, birch, or aspen forest</td>
<td>+</td>
</tr>
<tr>
<td>14. Plantations and self-sown alien forest</td>
<td>+</td>
</tr>
</tbody>
</table>

3.2.1 Successional dynamics – six development stages

Without natural disturbance or human intervention, a given set of trees of a single age class, or cohort, proceeds from life to death through a series of developmental stages (Oliver and Larson 1996). Large-scale disturbance agents such as fire, wind or large-scale insect outbreaks initiate succession and allow forests to regenerate simultaneously over large areas. In boreal forest examples of different successional stages are recent burns, young stands of mixed coniferous and/or deciduous trees, and old-growth forest stands (e.g. Furyaev and Kireev 1979, Angelstam 1998). Due to spatial and temporal heterogeneity of disturbances the structural complexity of age classes within a landscape increases with age (Johnson 1992). However, if viewed over longer time spans, successional stages are usually ephemeral at a particular site. To persist in the landscape, species specialising in a particular stage must be able to disperse from areas with suitable but degrading habitat in order to colonise new sites where the habitat conditions are good or improving. A critical requirement of many species is therefore that a stable patch dynamics is maintained within the landscape (Picket and White 1985). Typically, successional dynamics are initially dominated by shade-intolerant tree species, but later in the succession shade-tolerant species may become established.
At the scale of a stand, succession following a large-scale disturbance event produces a cohort with a more or less narrow bell-shaped tree age distribution, the average age of which increases with the time after disturbance. This pattern is usually most distinct at intermediate points in time after the disturbance. The reason is that in the early phases a considerable number of trees remain from the past stand, either as dead standing, fallen or surviving trees (e.g. Östlund et al. 1997). In the later part of the succession young shade-tolerant trees may become established and produce a more uneven age class distribution again.

The development of a single cohort of trees following a disturbance event can be divided into several distinct stages. Oliver and Larson (1996) describe four stages of stand development (initiation, stem exclusion, understorey reinitiation and old-growth stage). However, both from a silvicultural and wild life (sensu Hunter 1990) point-of-view more than these four stages are needed to capture the structural and compositional variation among different successional stages. In managed forest cutting classes, the terms harvested, young, thinning, and final felling are useful as they link the development of the stand to the silvicultural operations (e.g. Smith et al. 1997).

However, this division does not include later developmental stages of particular importance for forest biodiversity. Angelstam’s (1999) distinction of six stages provides a compromise between simplicity and detail and is adopted here (Box 3-1).

It is, however, rare that the development after a stand-replacing disturbance in an area is a linear sequence passing through each step in the successional development described above. Instead, there are several pathways through which successions may proceed. In principle, the disturbance can be initiated in any of the different stages, albeit with different probabilities. But there are often some restrictions, both with respect to fire and wind. In mesic boreal forest a new fire is unlikely to occur due to low fuel loads before a stand age of 20 years. During the first 3 to 5 decades after a disturbance episode the fire risk increasing, probably due to fuel accumulation (Schimmel 1993, Niklasson and Granström 2000). Similarly, all factors being equal, a stand's susceptibility to wind varies with age (Gardiner and Quine 2000).
Box 3-1 Presentation of the six stages of forest development in a successional dynamic (Thomas 1979, Angelstam 1999)

The four first developmental stages in the succession will have their equivalents in most managed forests.

- **Succ-1 (stand initiation):** just after extensive disturbances (fire, windthrow, large-scale insect outbreaks, clearcutting), the environmental conditions are often unique. The specific conditions of the site, and the surrounding matrix determining the further forest forming process are created at this stage.

- **Succ-2 (young stand):** the typical herb, shrub and tree layers vegetation has recovered, often after a phase of herb-rich pioneer ground vegetation. There are often still living trees and large amounts of coarse woody debris left from the previous stand. Remnants of the previous stand that died will start to decay.

- **Succ-3 (middle aged stand):** self-thinning and gradual replacement of light-demanding species (e.g. Betula and Populus) with shade-tolerant species (e.g. Picea, Abies, Fagus) take place. Towards the end of this stage trees start to compete with each other and some trees die from lack of light or soil moisture, a process called suppression, which leads to stem exclusion (Smith et al. 1997).

- **Succ-4 (harvestable stand):** the light-demanding deciduous broad-leaved phase fades away and shade-tolerant take over. The forest gradually acquires a multi-storey vegetation structure and the herb layer vegetation changes towards having more shade-tolerant species. Scattered trees (previously successful) begin become damaged or die due to insects, fungi, snow-break, wind, falling trees or other factors. These small gaps in the canopy allow more light and moisture to reach the forest floor. As a consequence, there is an advance regeneration of shade-tolerant species.

- **Succ-5 (ageing stand):** this stage is usually not allowed to develop in a managed forest. Shade-tolerant species are becoming older and start to develop diameters of interest for the largest primary nest excavators, bark texture suitable for different specialised lichens and canopies that can carry the nests of large raptors. Dead wood is accumulating and the vertical and horizontal vegetation structure is becoming more complex.

- **Succ-6 (old growth stand):** after well over a century or two without a completely or partially stand replacing disturbance the stand is gradually opening up with the formation of gaps in the canopy as large trees or groups of trees fall down. Coarse woody debris is abundant and vegetation structure is complex. As in the young forest, the tree age distribution is usually bimodal (e.g. Oliver and Larson 1996), now dominated by old trees but with appearing young cohorts both in gaps and as an additional vegetation layer. The relationship between the size and the age of the trees is becoming less and less obvious.
### 3.2.2 Cohort dynamics

Several tree species show clear adaptations to low intensity disturbances. Scots pine and fire is a good example. In the boreal zone, natural Scots pine forests on dry sites are characterised by frequent low-intensity fires that produce stands with several age cohorts of trees (Sannikov and Goldammer 1996, Angelstam 1998). Due to its thick bark, and long distance between the ground and canopy, a Scots pine tree becomes less sensitive to fire damage with increasing time. As a consequence, a typical natural dry site Scots pine forest has several distinct age cohorts of living trees, standing snags, both of which eventually produce a continuous supply of dead wood on the ground in different stages of decay (e.g. Sannikov and Goldammer 1996). Such forest has a park-like appearance. Burned and/or grazed oak forests of the old cultural landscape with a more or less dense canopy can show similar dynamics (e.g. Ellenberg 1996). When grazing animals are at low densities, the regeneration of trees is possible and a new cohort of trees is formed. According to Dyrenkov (1984) and Fedorchuk et al. (1998) this type of disturbance regime occurs also in Norway spruce forests on mesic well-drained sites in association with windthrow events that remove a portion of the canopy. Dyrenkov (1984) distinguished three different types of uneven-aged cohort dynamics (Box 3-2) ranging from a dominance of young trees (regeneration) to a dominance of old cohorts (digression). Following Dyrenkov (1984), the variants regeneration, intermediate and digression are used.

<table>
<thead>
<tr>
<th>Box 3-2 Presentation of the three stages of forest development in a cohort dynamic (Dyrenkov, 1984) (Thomas, 1979 and Angelstam’s, 1999)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>COHORT-1:</strong> Stands are dominated by younger trees but with an overstorey of old and very old trees as well as snags and coarse woody debris.</td>
</tr>
<tr>
<td><strong>COHORT-2:</strong> The different age cohorts are evenly distributed within the stand. In Scots pine forests on sediments there are typically 3-5 distinct cohorts that range over at least 200-300 years of age (e.g. Sannikov and Goldammer 1996).</td>
</tr>
<tr>
<td><strong>COHORT-3:</strong> Cohorts of old and very old trees dominate.</td>
</tr>
</tbody>
</table>

Sometimes, due to absence of fire for longer time, and an associated accumulation of nutrients, the site type may develop towards more productive ones.

The three types are related to the relative frequency of occurrence of disturbances with different intensities and/or return intervals. For example, in Scots pine forest on dry soils the first type should prevail after intensive disturbance such as crown fires, or after frequent fire disturbance not allowing recruitment of trees into older less fire damage-prone age cohorts. If fire disturbances are of lower intensity the second type should prevail. Finally, with infrequent fire disturbance the third type would prevail.

### 3.2.3 Gap dynamics

In the absence of large external disturbances, the death of a single tree or a group of trees maintains the formation of gaps in which more or less shade tolerant trees can regenerate.

In naturally dynamic landscapes such stands often form corridors, networks, or clusters in the wet and moist parts of the landscape. Typically, these forests have a relatively moist and stable...
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microclimate and a continuous supply of dead decaying wood in different stages of decay. This type of dynamics occurs also in large extensive areas where the climate is moist and where fires are uncommon (Angelstam 1998, Ohlson and Tryterud 1999). These areas include large parts of Europe where European beech (*Fagus sylvatica*) is the dominant shade tolerant species. As a result of these dynamics, a fine-grained mosaic structure develops in naturally dynamic beech forests. The forest cycle has been described as series of developmental phases by several Central European authors (e.g. Standovár and Kenderes 2003). The tree species involved include Norway spruce and *Abies* spp. in boreal and mountainous forests, beech and several broad-leaved tree species in lowland temperate forests as well as riparian forests (Mayer 1984, Giorgievskij 1992, Drobyshev 2000, Angelstam 1996). Following Dyrenkov (1994), we distinguish two sub-types: with even or patchy spatial tree distribution within the stand, respectively (Box 3-3).

**Box 3-3 Presentation of the two sub-types of the gap dynamic (Dyrenkov, 1994, Dyrenkov, 1984, Thomas, 1979 and Angelstam’s, 1999)**

- **Gap 1**: the first type is characterised by a spatially even distribution of different tree ages in the stand. This is associated with smaller gap sizes including one or a few trees.
- **Gap 2**: The second type is characterised by a patchy distribution of different tree ages in the stand. This is associated with larger gap sizes.

3.3 Forest and other wooded land management in Europe

Forests undisturbed by man have been cleared for cultivation and grazing or managed for wood production for millennia. As a result, today forests undisturbed by man remnants are scarce and most European forests are managed. Additionally forest management systems have evolved through the years, thus changing forest landscapes.

Forestry often began as a crusade (Davis and Johnson 1987). At first, forests were rapidly cut with little thought of providing a heritage for the future. Later, after continuous forest decline and a resource crisis in Europe at the end of the 18th century, the forestry profession grew to protect, plant, and manage the forest as a source of wood, or for protection against flooding and snow (e.g. Schuler 1998). In fact, the concept of sustainability was established in Europe by Venetians (Caniato 2006), in France (Colbert in 1669) and Germany (Carlowitz in 1713) (see Wiersum 1995, Floyd 2002) in response to high-grading and other forms of forest exploitation to ensure the long-term wood supply (Bode and von Hohnhurst 1995, Schuler 1998, Bachmann et al. 1998).

Silviculture involves the methods for establishing and maintaining communities of trees and other vegetation that have value to people (Nyland 1996). The value can either be the trees themselves, but as recently recognised, also biodiversity, clean water, protective functions as well as intangible benefits such as recreation and beauty (Salem and Ullsten 1999).

From the 1980s environmental degradation and unsustainable use of natural resources was widely recognised among political leaders. This resulted in new policies regarding the use of forests, water and seas, and the addition of development to the sustainability concept. As a
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consequence the focus shifted from the state of ecosystems to steering of society (Baker 2006), i.e. sustainable development in the sense of the Brundtland Commission (1987). The broad evolving vision of satisfying economic, ecological and social values by good governance was also adopted regarding forest management. The ‘Forest Principles’ adopted at The United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro in 1992 captured the general international understanding of sustainable forest management but also reflected the failure to adopt an initially planned Forest Convention alongside the three other Rio MEAs. This triggered a wide range of international and national processes to define SFM under a UN discussion process that led to the current UNFF. In the pan-European area, the forest policy discussion was developed through a series of ‘Ministerial Conferences on the Protection of Forests in Europe’ which became the ‘Forest Europe’ process in 2008 and recently started proceedings towards a legally binding agreement of forests in Europe. In the EU, there is no formal forest policy and MS have explicitly categorised forestry as a subsidiary issue.

Following the emergence of the SFM discourse, the role of silviculture has been challenged. For example, Puettmann et al. (2008) describes how forest managers are under increasing pressure to develop practices that sustain the full function and dynamics of forested ecosystems and maintain ecosystem diversity and resilience while still providing needed wood products. Thus, the best way to manage a forest has been, and will be, subject to continuous discussions in response to societal preferences (e.g. Wallmo 1897, Arpi 1959, Leibundgut 1993). There is, however, a general agreement that forest management somehow should adapt the choice of species based on site types and natural disturbance regimes (Sukachev 1951, Pogrebnyak 1954, Ilvessalo and Ilvessalo 1975, Hägglund and Lundmark 1986). Thus, the selection of trees species is often related to the requirements of the species at different site types (e.g. Mayer 1984, 1992).

Following the results of the previous sections, the occurrence of biotic disturbances is more common if forest stands (1) form monocultures, (2) are fertilised, (3) grow on degraded sites or (4) have a low degree of naturalness. These are arguments in favour of adapting forest management systems that encourage a natural composition, structure and function of forest stands and landscapes (see also Table 3-1). Consequently, the remainder of this section focuses on (1) a review of management systems for forests and other wooded land, and (2) a discussion about the extent to which the main disturbance regimes of 14 main forest types (Table 2-1) are matched by the current silvicultural systems and by free development or the methods employed in the ancient cultural landscape.

### 3.3.1 Silvicultural systems

Throughout most parts of the world, different silvicultural systems are defined using a gradient from even-aged to multi-aged and uneven-aged systems (Matthews 1989, Muelder 1996, Puettmann et al. 2008). There are thus three general types of age-class stand structures that are managed for: even-aged, stands with two age classes, and uneven-aged (Figure 4). Even-aged systems include clear-cutting or seed tree systems, the intermediate double-cohort systems include shelterwood systems, and finally uneven-aged systems include single tree and group
selection (Table 2). The different systems can be understood better if considered as located in a continuum of proportion trees removed at each treatment and the size of the treatment unit.

**Even-aged management with clearcutting systems**

In even-aged stands all trees germinate, sprout or are planted at about the same time. While the tree size will increasingly vary as the stand ages, the calendar age of the trees is about the same when it is time to regenerate the stand. Because of a definite beginning and end, it is often relatively easy to characterise stands just by the stand age (similar to or slightly shorter than time since disturbance), as well as to guide decisions about treatments and harvesting of the stand. It is also easy to manipulate tree species composition and the genetic makeup by planting or seeding. The smallest stand size ranges from less than one hectare to tens of hectares, but clear-cutting units may also cover thousands of hectares (e.g. Devall 1993). The following three methods are commonly used.

- **Clearcutting/coppice**: This is the most common silvicultural regime for even-aged management. It implies the removal of the entire stand in one cutting. Coppice can be viewed as a special case of clearcutting, which is dependent on vegetative regeneration of sprouting broad-leafed species such as *Populus*, *Corylus*, *Carpinus* and *Fraxinus* (e.g. Jahn 1991).

- **Seed tree method**: This method involves removal of the old stand in one cutting, except for a small number of seed trees left dispersed or in small groups to ensure the recruitment of a future tree generation. When a new tree generation has been secured, the seed trees are removed, usually with a considerable addition of valuable diameter growth (Smith et al. 1997).

- **Variable retention**: In response to criticisms to clearcutting systems, considerations to both visuals and biodiversity maintenance have been developed. Retention of trees during harvest may include both dispersed trees and patches or clumps of trees. Compared with the seed tree method the aim is, however, different. For visual reasons the shape and manner of harvesting can be designed to avoid the impression of almost complete harvesting (Lucas 1991). For biodiversity, the retention trees usually include forest components typical of old-growth stands (e.g. large/old trees, snags, coarse woody debris).

**Double-cohort management with shelterwood systems**

In this system, the removal of the old stand takes place in a series of cuttings, which extend over a relatively short part of the successional development. The establishment of the next cohort takes place under the partial shelter of the seed trees, or under trees of other species. These double-cohorts represent an intermediate category in which the presence of both cohort stands can be temporary or continuous (Smith et al. 1997).

- **Shelterwood cutting**: Shade-tolerant species are often regenerated under a protecting canopy of remnant trees from the previous stand. The aim is to reserve the best trees, which have large crowns and strong boles. Such trees are considered both windfirm and with the health and vigour that lead to good seed production (Smith et al. 1997).
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- **Coppice with standards**: This system includes the combination, in the same area, of short-rotation coppice growth with scattered older trees (standards). The standards are often species that may be of seedling origin. While the generation time of the coppice is only a few decades the standards may be harvested at ages exceeding a century or more. A common combination in Central Europe is Fraxinus/Carpinus/Corylus as coppice and Quercus as standards (Kirby and Watkins 1998).

- **Uneven-aged management with selection systems**

  In uneven-aged stands there is no definite beginning or end of a cycle in time. Trees in a given area vary by age as well as size, and are frequently of different species. Throughout their lives the trees compete for light or moisture with larger or taller trees nearby. Uneven-aged stands are in principle composed of small even-aged cohorts of different ages located in gaps of different size in the stand. As each little even-aged group becomes older competition reduces the number of trees, first rapidly and later more slowly. Therefore, if each age class occupies the same area, the diameter distribution for a balanced uneven-aged stand follows an asymptotic relationship usually referred to as reverse J-shaped. Control of the genetic origin of trees is difficult because new trees regenerate from seeds. The system favours shade-tolerant species such as Picea, Abies and several broad-leaved tree species (Kuuluvainen 1994, Oliver and Larson 1996).

  For management purposes, a distinction is made between balanced and irregular uneven-aged stands (Smith et al. 1997). In a balanced uneven-aged stand there are three or more age classes, each occupying an equal area. Irregular uneven-aged stands do not contain all age classes and will therefore not ensure that trees will arrive at a certain time at an even rate. Here the distribution will look like a series of humps on the diameter distribution curve. However, in particular the contemporary Central European and old Scandinavian, literature and terminology on uneven-aged silvicultural systems is very diverse. Following Mayer (1992), Börset (1986) and Smith et al. (1997) we include only two types, the single-tree and group-selection systems.

  - **Single-tree selection**: In the classic form of the selection system every even-aged cohort of the uneven-aged stand occupies a space about equal to that created by the removal of a single tree. Stands, or whole forests, are characterised by these scattered gaps with small even-aged groups of trees, which are thinned as they grow. The types tree species that are perpetuated with this system are shade-tolerant (e.g. Fagus, Abies, and Picea on productive sites).

  - **Group-selection system**: If the regeneration openings are made larger, the ecological requirements of a range of other tree species will be satisfied. A special case of the group-selection system is strip-selection (Saumschlag in German). Here the components of the uneven-aged stand are created in slowly advancing strips. Advantages are that the harvested wood can be transported through the old stand and the strips can be arranged so the front of progressively taller trees faces the prevailing winds.

- **High-grading**

  Although not a silvicultural system, harvesting of the most valuable trees only has often been part of the first phases of forest usage, where typically all specimen above a certain diameter are
cut and which is still the state of tropical forest use at its best in many places. As a consequence the average diameter of the remaining trees is gradually being reduced. Finally there is usually a need to restore the sustainable wood production by introducing some forest management.

3.3.2 Traditional cultural landscape management regimes

Clearing and cultivation of forested land, a major impact on forests for millennia, has caused a dramatic reduction and fragmentation of the once naturally dynamic primeval forests (Mayer 1984, Mantel 1990). Nevertheless, in some regions, forest functions such as biodiversity conservation and cultural landscape values in what forms other wooded land have to some extent been rescued by management methods practised in the old cultural landscape (Mantel 1990, Tucker and Evans 1997, Agnoletti 2000). To maintain summer and winter fodder for domestic animals, land was managed using fire, mowing, clearing, tree and water management (Kirby and Watkins 1998). This range of cultural disturbances often resulted in the presence of large and special trees in a landscape dominated by grazing and/or agriculture (e.g. Tucker and Evans 1997, Kirby and Watkins 1998). Today such habitats usually remain as small isolated patches in a managed matrix. But in parts of Europe the old management regimes are still in use. This applies to remote valleys in mountains, which have not yet been reached by the agricultural revolution and its intensive management (Angelstam 2006), and existing silvopastural systems such as Portuguese montados and Spanish dehesas (Joffre et al. 1999). The ancient practice of pollarding and lopping whereby branches of wild trees are cut but the tree is not, does maintain large trees that are growing slowly. Coarse woody debris on the ground was often limited but dead wood was available in the crowns of large trees, which were left to shade the ground.

As a consequence, suitable substrate both on the outside of the trees, as well as inside if hollow, will provide habitat for many forest species ranging from shade-intolerant vascular plants, lichens and insects to large birds (Mikusinski and Angelstam 1998, Nilsson et al. 2001). Additionally, recreation and cultural values are maintained. Both biodiversity and cultural heritage are important to consider as their role as infrastructures for tourism and recreation products are becoming increasingly important (Parrotta et al. 2006). Additionally, traditional cultural landscape management has positive effects on soils. Similar trends in the development of the composition and structure of domesticated trees (e.g. olive and fruit trees) have occurred both in Central Europe and in the Mediterranean (e.g. Tucker and Evans 1997, Ispikoudis et al 1993). It may also include area-demanding raptors needing large trees for breeding as well as large carnivores in areas where shepherds protect livestock.
3.4 How forest disturbance regimes and forest management systems match

The review of biotic disturbances shows that the occurrence of biotic disturbances is more common if forest stands (1) form monocultures, (2) are fertilised, (3) grow on degraded sites, or (4) have a low degree of naturalness. These are arguments in favour of adapting forest management systems that encourage development toward a natural composition, structure, and functioning of forest stands and landscapes (also Table 3-1). However, close-to-nature silviculture is different for different forest disturbance regimes.

There are mismatches between the natural disturbance regimes in different forest types, and how they are managed (Table 4-1). As an example, Axelsson et al. (2007) reported that 96% of Swedish forests are management by even-aged silvicultural systems while the proportion of natural disturbance regimes characterised as succession was much lower. Thus, to counteract uniformisation and monoculture, a broader suite of silvicultural systems is needed (see Tahvonen 2009, Tahvonen et al. 2010).

Table 3-2 Overview of the range of disturbance regimes in different forest types (see Annex 1) and the main forest management systems used. The aim is to illustrate that in some forest types (e.g. boreal and hemiboreal forest) one management system dominates (even-aged), while the range of disturbance regimes is large. This does not counteract uniformisation and monoculture, two factors that favour biotic disturbances.

<table>
<thead>
<tr>
<th>Categories</th>
<th>Disturbance regime</th>
<th>Forest management system</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Boreal forest</td>
<td>Suc, Coh, Gap</td>
<td>Even-aged</td>
</tr>
<tr>
<td>2. Hemiboreal forest and nemoral coniferous and mixed broadleaved - coniferous forest</td>
<td>Suc, Coh, Gap</td>
<td>Even-aged</td>
</tr>
<tr>
<td>3. Alpine coniferous forest</td>
<td>Suc, Coh, Gap</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>4. Acidophilous oak and oak - birch forest</td>
<td>Coh, Gap</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>5. Mesophytic deciduous forest</td>
<td>Coh, Gap</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>6. Beech forest</td>
<td>Coh, Gap</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>7. Mountainous beech forest</td>
<td>Coh, Gap</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>8. Thermophilous deciduous forest</td>
<td>Coh, Gap</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>9. Broadleaved evergreen forest</td>
<td>Coh, Gap</td>
<td>Even-aged</td>
</tr>
<tr>
<td>10. Coniferous forests of the Mediterranean, Anatolian and Macaronesian regions</td>
<td>Suc, Coh, Gap</td>
<td>Even-aged</td>
</tr>
<tr>
<td>11. Mire and swamp forest</td>
<td>Coh, Gap</td>
<td>Uneven-aged</td>
</tr>
<tr>
<td>12. Floodplain forest</td>
<td>Suc, Coh, Gap</td>
<td>Uneven-aged</td>
</tr>
<tr>
<td>13. Non riverine alder, birch, or aspen forest</td>
<td>Suc, Coh</td>
<td>Even-aged, Uneven-aged</td>
</tr>
<tr>
<td>14. Plantations and self sown alien forest</td>
<td>Not relevant</td>
<td>Even-aged</td>
</tr>
</tbody>
</table>

In the following sub-sections, we discuss how the match between silvicultural systems on the one hand, with natural disturbance regimes and traditional cultural landscape systems on the other, can be improved.

- **Even-aged management vs. natural disturbance regimes**

  *Succession*: Because silviculture aims at maintaining a high yield, stands are as a rule harvested at a point before the increase in volume growth starts to diminish. Consequently, the amounts of ageing and old-growth stands will decline with increasing time of application of even-aged forest...
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management under the sustained yield paradigm in a once naturally dynamic landscape. The absolute age for final harvesting varies among tree species. Hence, if sustained yield even-aged systems are applied systematically, the later successional stages will not be perpetuated. This is the main reason for the loss of forest species, reduced structural diversity, and altered processes in most managed forests. To favour natural structure different systems of variable retention cutting have been developed. For example, in Sweden the amount of residual trees left after final harvesting increased from 2% in the mid 1980s to 13% in the 1990s (Angelstam and Pettersson 1997).

- **Cohort-dynamics**: Clearcutting with residual seed-trees are superficially similar to cohort dynamic where the stand is dominated by younger cohorts. The obvious difference is the lack of dead wood, both snags and coarse woody debris, in different dimensions and decay stages. However, with long-term consistent retention of trees at every harvest event, a certain amount of dead wood in different stages of decay, and several cohorts of different living trees could be restored over long time.

- **Gap dynamics**: The even-aged silvicultural systems are not appropriate for gap dynamics.

► **Cohort management vs. natural disturbance regimes**

- **Succession**: Following large-scale disturbances associated with surviving remnants of the previous stand the establishment of the new stand can lead to a structure similar to a shelterwood system. This could for example be a combination of oak with a turnover time of 100+ years and coppice; e.g. the typical Central European Mittelwald.

- **Cohort dynamics**: This kind of dynamics can to a large extent be emulated in forest management with repeated retention of trees and dead wood, and the use of fire as a biodiversity management tool (Fries et al. 1997). The restoration of a dry pine forest where clear-cutting with seed tree regeneration could be achieved in the following steps: instead of harvesting the seed trees when the regeneration of a new stand has been secured, a proportion of the seed trees would be left to develop into a cohort of old trees. During the following rotation the procedure should be repeated. The restoration of dead wood could either be active by leaving high stumps, or passive by leaving some of the wind-blown trees.

- **Gap dynamics**: Not appropriate.

► **Uneven-aged management systems vs. natural disturbance regimes**

- **Succession**: Opening gaps for generating a new tree generation is in principle the beginning of succession. However, this approach would not ensure a sufficiently wide distribution in patch sizes to satisfy the area requirement of many of the species. Gap formation would also often be a too weak disturbance, and therefore exclude many species. Uneven-aged management systems could, however, to some extent maintain the composition and structure of the final old-
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growth phase in the succession. However, to succeed, the restoration and maintenance of dead wood would also be needed.

- **Cohort dynamics:** Uneven-aged group-selection management systems could potentially be useful for stands, which naturally have a range of age classes. One example is Pinus sylvestris forests that naturally are subject to repeated low-intensity fire episodes (Fries et al. 1997).

- **Gap dynamics:** The continuous application of single and group selection harvesting in forests with shade-tolerant tree species such as *Fagus sylvatica*, *Abies alba*, and *Picea abies* on fertile soils leads to a multi-layered vegetation structure. Typically, an inverse J-shaped diameter distribution is the long-term result. Near-to-nature forestry (Naturnahe Waldwirtschaft) was developed in Germany and Switzerland with the aim of maintaining this forest structure. In spite of this apparent good match between the silvicultural system and the natural disturbance regime, traditional uneven-aged near-to-nature silviculture does not emulate all components of the natural gap phase dynamics. The absence of dead wood is the most striking difference. But also the absence of large old trees is evident.

**Exploitation**

Often the early phases of the utilisation of forest resources can be described as tree mining. Only certain species, and/or dimensions are harvested per unit area, and the timber resource is viewed as infinite. From a biodiversity point-of-view, this is not necessarily a problem unless the repetition of this practice (called ‘high-grading’) leads to forest degradation and ultimately deforestation, as it is still the case in the larger part of tropical forestry operations.

**Old cultural landscape management systems**

Because of the long history of anthropogenic impact on Europe’s forests and woodlands there are land management systems that maintain forest qualities such as old and large trees, dead wood and trees with substrates otherwise confined to old-growth forests. This motivates individual analysis for the ancient agroforestry systems based on animal husbandry, once widespread in most of Europe.

- **Succession:** A characteristic feature of agriculture is land abandonment. This may take place at all spatial scales. Small fields or whole land holdings may be abandoned due individual preferences, family events and disputes. Villages, and whole regions, may be deserted as a consequence of famine, disease and changed economic conditions for the current land use. As a result, in spite of active management, certain successional stages in the development from abandoned pastures, meadows and fields have a chance of being dynamically present in the landscape. Large herbivores may once have had this role (Vera 2000).

- **Cohort dynamics:** The maintenance of wooded meadows and pastures can allow the long-term presence of large old trees in a landscape for a very long time (Kirby and Watkins 1998) even if that is not considered a forest. However, for the
maintenance of forest biodiversity, such a landscape can provide a crucial rescue effect for a number of species. A large-scale example of this can be found in the Extremadura region in Spain. In combination with sheep, pig and cork production, the area host both large raptors and mammalian carnivores (e.g. Tucker and Evans 1997). Similarly, old-fashioned apple orchards with large old trees maintained, a rich bird fauna until recently, when they were replaced by young trees (Weitnauer and Bruderer 1987).

- **Gap dynamics**: The low tree density and limited number of age classes will not host gap phase dynamics unless the area is converted to forest again.
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Chapter 4: Future trends in biotic disturbances

4.1 Key points

- Environment has a marked influence on plant diseases, impacting the host, the pathogen, and the interaction between these players, as illustrated in the diagram below. The thin arrows highlight the direct impacts of climate change, the thick arrows indicate the indirect impacts of climate change on risk of biotic outbreaks.

- As climate changes, the climatic regions suitable for both hosts and pathogens will alter, leading to changes in the distributions of both. Altitudinal and expansion of the northern limit of species ranges are the most commonly observed patterns.

- Increased frequency or severity of outbreaks may also be observed. Certain pathogens, particularly *Phytophthora* spp. causing root infections, are likely to increase in importance as climate change benefits their reproductive potential. Other biotic agents such as scolytids and needle blights will also be favoured indirectly by climatic changes, as their host trees are increasingly under stress.

- Most pathogens have short sexual life cycles, enabling faster adaptation to changing climatic conditions than found with trees. Understanding the adaptative potential of both the host and the biotic agent populations may be one of the key predictors of the level of climate change impacts.

- Natural dispersal of pathogens can be regional or even intercontinental, meaning that potentially, they may rapidly reach new regions with suitable climates; this problem is further exacerbated by human transport of pathogens.

- Uncertainty pervades predictions of future impacts of climate change, and future impacts of biotic outbreaks, as the interactions between hosts and biotic agents are complex, and still poorly understood. The effects of climate change are going to be different for each system, in different locations.

- Simple frameworks for helping managers prioritise actions, based on a diagnosis of the problem and of their aims can them help match forests to the changing disturbance regimes.
4.2 Climate change scenarios and potential impacts on the EU forests

Since 1900, the global average temperature has already increased by 0.8°C and the twelve hottest years observed since 1880 all occurred between 1990 and 2005. These climatic disturbances are already affecting European forests: growth rate has increased in northern latitudes, while some species have declined on dry sites close to their southern distribution limits.

Different scenarios exist for the prediction of future climate. The IPCC fourth assessment report applied different scenarios based on the Special Report on Emission Scenarios (SRES; Nakicenovic et al. 2000). This SRES scenarii are grouped into four scenario families (A1, A2, B1, and B2) that explore alternative development pathways, covering a wide range of demographic, economic, and technological driving forces and resulting GHG emissions. The emission projections are widely used in the assessments of future climate change, and their underlying assumptions with respect to socioeconomic, demographic and technological change serve as inputs to many recent climate change vulnerability and impact assessments.

The outcomes of these different climate change scenarios showed regional variability of climate change. The potential impacts of climate change on biotic events, forest sensitivity and tree species distribution are presented below per forest type, after Lindner et al. (2008). For simplification, these forests have been categorised in five main types, boreal forest, Atlantic forest, continental forest, Mediterranean forest and mountainous forest, which encompass the 13 EEA forest types (the correspondence between both classifications is detailed in Annex 1).

4.2.1 Boreal forests

Significant increases in mean annual temperature (by 3.5-5°C) and precipitation (up to 40%) are predicted, resulting in wetter and warmer winters. The foreseen increase in temperature would lead to a prolongation of the growing season and an enhancement of soil organic matter decomposition (Kirschbaum 1995). Nevertheless, forest damage by wind and snow are also projected to increase (Peltola et al. 2000) as milder winters would reduce winter hardening in trees, increasing vulnerability to frost (Hänninen 2006) and to violent climatic events. In addition, tree species distribution is expected to change: broadleaves deciduous trees are expected to migrate northwards and the woody vegetation is expected to spread into tundra at higher latitudes and higher elevations (Grace et al. 2002).

4.2.2 Atlantic forests

Mean annual temperature in the temperate oceanic region is predicted to increase by 2-3.5°C, with the summers sometimes becoming up to 4°C warmer. This increase in temperature would have a positive impact on forest growth and wood production in northern and western parts, but a negative impact in southern and eastern parts, since it would intensify the water scarcity concerns. As in boreal forests, winters would be accompanied by more frequent extreme events, such as storms and floods. The expected change in tree species distribution would mainly result in an improvement of the development conditions for thermophilic plant species which will
expand their area of distribution, while the area of distribution of the cold-tolerant species would decrease (Reid, 2006) as a consequence of the temperature increase. As an example, natural conifer areas would be progressively replaced by deciduous trees.

### 4.2.3 Continental forests

The annual mean temperature increase is projected to be in the order of 3–4.5°C in continental regions. In this bioclimatic zone, forest production is mainly constrained by water availability (Maracchi et al. 2005) and decreasing annual precipitation or changes in inter- and intra-annual distribution would result in stronger water limitations. The effects of climate change on forest production would vary: production would decrease at sites vulnerable to water stress, and increase in the areas that would benefit from an increase in precipitation. Impacts on individual species can be either positive or negative, depending on site conditions and regional climatic changes. *F. sylvatica* is projected to face severe problems under increasing temperatures (Geßler et al. 2007) and as in boreal forests, milder winters may increase their vulnerability to frost. In addition, fire risk is likely to increase (Moriondo et al. 2006). In Slovenia, at the northern climatic limit of olive trees, frost occurrence is projected to further restrict the distribution range of the species (Ogrin 2007).

### 4.2.4 Mediterranean forests

Annual mean temperatures are projected to increase in the order of 3–4°C and yearly rainfall is expected to drop by up to 20%, mainly in summer (reduction by up to 50%), whereas winter precipitation is expected to increase. Changes in frequency, intensity, and duration of extreme events are supposed likely to result in more hot days, heat waves, heavy precipitation events, and fewer cold days. The rising temperatures and the projected decrease in rainfall are predicted to magnify drought risk and in case of prolonged droughts and hot spells will further aggravate forest fire risks. These forest fires, already frequent in Mediterranean forests would become an even larger threat (Moriondo et al. 2006) and they would participate in the degradation of the ecosystem by increasing soil erosion due to enhanced hydrophobicity (Certini 2005) and reducing plant regeneration (Delitti et al. 2005). Over the last 50 years, a temperature increase of 1.4°C has already resulted in progressive replacement of European beech (*F. sylvatica*) by Holm oak (*Quercus ilex*) in the higher elevations of the Pyrenees due to a reduced recruitment and an increasing defoliation of beech (Penuelas and Boada 2003). In addition, the foreseen global warming and fire increase may trigger irreversible biodiversity losses and shifts in vegetational composition within a few decades or centuries at most. Fire and drought-sensitive vegetation types seem particularly threatened by large-scale displacement (Colombaroli et al. 2007).

### 4.2.5 Mountainous forests

European mountain regions are more sensitive to increases in temperature than surrounding regions. In the Alps, for instance, temperature increases twice as high as the global average were observed over the last century (Auer et al. 2007). Forest fires, particularly difficult to combat in complex mountainous terrain, are likely to increase in the Pyrenees and become a relevant disturbance factor also in temperate mountain ranges (Schumacher and Bugmann 2006).
Mountain forests are especially sensitive to climate change and remarkable shifts are projected in the potential distribution of herbaceous, dwarf shrub alpine plants and even tree species (Skvarenina et al. 2004).

### 4.2.6 Summary of likely climate change impacts on EU forests

Below the line, climate change appears to induce more detrimental than positive effects to the various forest types in different bioclimatic regions of Europe. Except for the benefit of increased biomass production in Boreal forests or in sub-alpine areas, the projected rise in temperature and changes in precipitation will very likely enhance the vulnerability of European forests to different biotic and abiotic threats (Table 4-1). Most at stake are Mediterranean forest ecosystems, which are particularly threatened by drought and fire. Under more extreme environmental conditions, natural disturbances may result in the permanent loss of forest vegetation, soil loss and eventually desertification of the landscape. Forest fires will gain importance also in mountainous areas, especially in southern regions such as the Pyrenees, but also in the Alps and the Carpathians (Lindner et al., 2008).

Climate change scenarios predict a change in the occurrence of tree species ranging from altered distribution of forest types over shifting tree lines, to regression or expansion of different species. In several regions of Europe temperature increase is likely associated with a regression of conifer forests. Species richness might regionally be promoted when oak and beech-dominated forest types are going to replace former coniferous forest ecosystems. Yet, also broadleaved trees are sensitive to climatic changes, and especially beech is at risk in drought-prone areas (Lindner et al., 2008). In addition, the changing environmental conditions are likely to affect tree resistance, making all tree species more vulnerable even to minor abiotic events as well as to insect or pathogen attacks.

The analysis of climate change impacts has been focused more on wood than on NWFP. Severe implications are however to be expected also on a social perspective, for example through the loss or changing distribution of fruit trees or impaired protective or recreative functioning of forests. More research is needed in these fields to assess future (negative or positive) developments under a changing climate.
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### Table 4-1 Synthesis of the expected consequences of the climate change on the different forest types

<table>
<thead>
<tr>
<th>Forest type</th>
<th>Climate change expected</th>
<th>Change in tree species distribution</th>
<th>Change in abiotic factors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Boreal</td>
<td>4-5°C of temperature increase and A 40% increase in yearly precipitation</td>
<td>Expansion of the broadleaves deciduous trees area northwards</td>
<td>Foreseen increase of damage by wind and storms</td>
</tr>
<tr>
<td>Atlantic</td>
<td>2°C of temperature increase</td>
<td>Decrease of the conifers distribution area</td>
<td>more frequent extreme events like storms and floods in winter mainly</td>
</tr>
<tr>
<td>Continental</td>
<td>3-4.5°C of temperature increase and global decrease in precipitation with changes in inter- and intra-annual distribution</td>
<td>Decrease in the number of F. sylvatica</td>
<td>- Increase of the vulnerability to water stress in some areas</td>
</tr>
<tr>
<td>Mediterranean</td>
<td>3-4°C of temperature increase and A 20% decrease in yearly precipitation</td>
<td>Progressive extinction of the European Beech</td>
<td>- Increase of the vulnerability to frost for some tree species</td>
</tr>
<tr>
<td>Mountainous</td>
<td>Rapid increase of the temperature compared to the other forest types</td>
<td>Predicted shifts in distribution of herbaceous, dwarf shrub alpine plants and even tree species</td>
<td>Forest fire threats in the Pyrenees and in temperate mountain range</td>
</tr>
</tbody>
</table>

### 4.3 Influence of climatic factors on biotic agents

In addition to management and landscape change, biotic agents are influenced by weather both directly and indirectly. Changes in temperature, precipitation, frequency and intensity of storm events as well as increased concentrations of atmospheric CO$_2$ can directly impact biotic agents and their hosts by modifying their:

- Phenology: e.g. changes in development rates, quiescence, etc.
- Physiology (nutrition, plant resistance, etc.)
- Synchrony and spatial distribution

In turn, changes in host species can modify community interactions, and their nutritional quality and resistance, making them more vulnerable to biotic infection.
As these difficulties accumulate, trees may develop symptoms of dieback, eventually leading to death.

4.3.1 Effects of temperature increase

Direct effects on species phenology – positive and adverse implications

Climatic conditions strongly influence the development of ectothermic organisms. Changes in surrounding temperature regimes involve alterations in development rates, voltinism, and mortality of biotic agents. These subsequently affect the reproductive potential, and thus the size, density and genetic composition of populations, as well as their extent of damage (Bale et al. 2002). As long as temperature does not increase beyond a species optima for development, positive implications are likely to predominate, such as enhanced reproductive potential. Acceleration of development by shortened egg and larval periods increases chances of survival, as the times most susceptible to predation and parasitism are reduced (Rouault et al. 2006). High sensitivity towards changes in temperature is especially shown by aphids, so that future outbreaks of increased severity are to be expected (Cannon 1998, Bale et al. 2002, Harrington et al. 2007). Fungal sporulation and rates of infection are also highly temperature-dependent. Warmer temperatures have been shown to be highly correlated with sporulation rate and incidence of Heterobasidion (La Porta et al. 2001, Gonthier et al. 2005). Generally, earlier swarming of insects is driven by warmer winters and increased spring temperatures. Parmesan (2006) for instance refers to several European and North American studies documenting the recent advance of first appearance for a multitude of butterfly species. However, when ambient temperatures go beyond biotic agent’s optima, for instance during heat waves, biotic agents populations may be affected in a negative way. This is typically reflected by decreased growth rates, reduced fecundity, and increased rates of mortality (Rouault et al. 2006). For instance, optimal growth and infection conditions of most cankers are around 25°C, with maximum tolerance in the range of 30°C to 35°C (Graniti 1998, Mancini and Scapin 1981). Thus severe heat waves, such as those that occurred in 2003, could reduce their viability and populations. Furthermore, the timing and maintenance of diapause, which is essential for many insect species of temperate regions to complete their life-cycles, will be modified by increased temperature during autumn, winter and spring (Buse and Good 1996).

Increased winter temperatures can either favour or delay insect development, depending on the species (Battisti, 2004). Fast growing, multivoltine, frost-sensitive species that do not undergo diapause or are not dependent on low temperatures to manifest diapause will probably benefit from increased winter temperatures according to Bale et al. (2002). In contrast, adverse and detrimental effects respectively are to be expected for fast and for slow growing species, which do need low temperatures to manifest diapause or to increase their frost resistance. Mild winters are also known to enhance the growth of cankers, although they are not sufficient to cause outbreak conditions (La Porta et al. 2008). Warmer winter temperatures will also tend to decrease the food requirements of browsing mammals, such as deers and moose. However, the net impact on forests would probably still be negative, as warmer winter temperatures also improves winter survival, a major factor regulating their population sizes (Bartmann et al. 1992).
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Indirect effects through plant nutritional quality, host resistance and community interactions

Temperature increase affects the development of both insect herbivores and plant tissue. Given insect herbivores are more responsive to changes in temperature than their hosts, as was shown in a range of experimental studies, even slight temperature increases are likely to enhance epidemics of various biotic agents (Ayres 1993). Although there is abundant evidence that phenotypic changes in plant physiology can affect herbivores, comparable studies with pathogens are few, and results are equivocal (Ayres and Lombardero 2000). However, reliable projections are difficult due to the considerable lack of knowledge on the relative temperature sensitivity of various insect species and their host plants, the effects of temperature changes on plant allocation patterns and host resistance to herbivory. As an example, temperature-induced changes in plant physiological processes may as well have adverse effects on foliage quality and performance of defoliators (Rouault et al. 2006, Ayres 1993).

Many host-pathogen interactions are well-adapted to host phenology. This is particularly the case also for the numerous disease syndromes that involve both insects and pathogens, such as the Dutch elm disease, which is transported by elm bark beetles (see section 2.3.7). Many host-specific insect herbivores require exact timing with their host, for instance between bud burst and egg deposition or larval development. In this case, differential effects of temperature change on the growth rates of insects and host plants are likely to disrupt phenological synchrony, at least for some time (Cannon 1998, Bale et al. 2002, Rouault et al. 2006). In the long term, phenological coincidence could be restored again, given gradual warming allows for selection pressures bringing insect populations back to synchrony (Cannon 1998, Tikkanen et al. 2006). Not only synchrony between herbivores and host plants, but also interactions with biological control agents are influenced by a warmer environment. On the one hand, the effectiveness of certain species of predators (e.g. coccinelids) and parasitoids is supposed to be enhanced; on the other hand, drought-induced changes in host plant quality can also support herbivore populations to escape natural enemy regulation (Cannon 1998, Parmesan 2006). It is yet uncertain whether climate affects tritrophic interactions directly or parasitoids and predators are simply responding to the population sizes and preferences of their hosts (e.g. Roy et al. 2004, Price et al. 2006). Many herbivores produce cyclical outbreaks as a result of delayed density-dependence with other species, such as Lymantria dispar populations which may depend on mice that prey upon moth pupae (Jones et al. 1998). Any climate-related change affecting any of the species or processes that produce delayed density-dependence could make outbreak conditions either more or less likely (Ayres and Lombardero 2000).

4.3.2 Changed patterns of precipitation and humidity

Direct effects (of reduced winter precipitation) on insect development

Many pathogens are sensitive to precipitation and humidity, and their rate of reproduction, spread and infection tends to be higher when conditions are moist (Harvell et al. 2002). Floods and droughts are generally accepted as triggers of epidemics of Phytophthora. cinnamomi, while moisture is essential for survival and sporulation of P. ramorum, where the duration, frequency and timing or rain events during winter play a key role in inoculums production (Sturrock et al.
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Distinct waves of *P. alni* infection are also associated with flooding episodes (Webber et al. 2004). **Improved food supply and extended growing seasons** in response to decreasing snow precipitation coupled with temperature increase was described by Roy et al. (2004) or Petrucco-Toffolo and Battisti (2008).

In contrast, reduced snowfall and earlier snow melt might **reduce overwinter survival** of species that hibernate in the forest litter or are commonly insulated by snow cover (Ayres and Lombardero 2000). Distributional shifts and even local extinction of butterfly populations were associated with the negative impact of low snowpack years and extreme winter droughts by Parmesan (2006). The incidence of *Armillaria* species decrease under low precipitation (Shaw and Kyle 1993).

**Host nutritional quality and resistance influenced by water stress**

Severity as well as phenology of water stress are key factors in the carbohydrate and protein metabolism of plants. Nutritional compounds (nitrogen) but also allelochemicals (secondary metabolites) tend to be elevated in water stressed trees, while turgor pressure and water content of plant tissue decrease, and plant growth is reduced (Huberty and Denno 2004, Desprez-Loustau et al. 2006). Based on the ‘Growth-Differentiation Balance’ hypothesis (Herms and Mattson 1992), tree defence and resistance to pest attacks is likely to be stimulated by moderate water stress, but to be impaired by severe water deficiency. Plant water stress is however experienced differentially depending on the feeding guild and the species. But separating the role of drought *per se* from the direct role of several species of pathogens and insects is not always possible (Cattaneo 1992). This is in particular the case of root diseases, since their effects interact directly with the plant’s water use.

Moderately water-stressed trees show improved levels of resistance to fungi associated with scolytids (e.g. Dutch elm disease), in some cases counterbalancing increased intensities of attack by the bark beetles. Insect defoliation combined with drought clearly promote fungal colonisation in case of moderate water stress (Capretti and Battisti 2007). Elevated levels of allelochemicals were also shown to anticipate performance of free-living chewers on intermittently stressed host plants. The development of sap-feeding aphids was also shown to be impaired on water-stressed host plants, as the extraction of plant nitrogen is impeded due to enduring negative turgor pressure. Other feeding guilds such as leaf miners, are however able to profit from elevated levels of nitrogen by avoiding compartmentalised secondary metabolites (Huberty and Denno 2004, Desprez-Loustau et al. 2006, Rouault et al. 2006).

Many biotic agents benefit from the stress on trees induced by droughts, as with increasing intensity or frequency of water stress, tree resistance commonly decreases. In any case, drought conditions appear to be one of the most significant climatic factors leading to stress-induced disease development for many tree-pathogen interactions (Deprez-Loustau et al. 2006), such as *Biscogniauxia mediterranea* which favours drought-stressed oak species, The severity of *Sphaeropsis* (Diplodia) shoot blight, caused on many species of pine by *Sphaeropsis sapinea* (*Diplodia pinea*) is associated with water stress (Blodgett et al. 1997a,b, Paolletti et al. 2001). When pines are affected by drought prior to flushing, susceptibility to this opportunistic pathogen increases (Desprez-Loustau et al. 2006, La Porta et al. 2008). Scolytids are also likely to benefit from severe water deficiency of host plants (Ayres 1993, Christiansen and Bakke 1997, Lombardero et al. 2000, Desprez-Loustau et al. 2006). Explanations are the loss of defense
mechanisms of trees against primary attack and fungal transfer due to limited oleoresin production. More frequent drought events are thus likely to lead to future outbreaks of bark beetle populations on a larger scale.

Yet, not all insect herbivores are influenced positively by water deficits, as observed for various species of Lepidoptera. Nutritional quality of host plant tissue is also reduced for chewing insects (defoliators) in response to decreased water content, as the foliage is tougher due to reduced nitrogen availability (Huberty and Denno 2004).

Complex effects of climate change through community interactions

The complex interactions between host plant water supply and resistance as well as development of herbivores and their natural enemies are still insufficiently understood, and documented only by examples. Both induced resistance of host plants to insect attack occurring later in the season and increased vulnerability to secondary pest attacks the year after the drought is possible given initial impairment of trees by water deficiency. Another example are reduced development rates of defoliators due to drought-induced low nutritional quality of leaf tissue, leading to prolonged exposure to parasitoid attacks and higher mortality (Rouault et al. 2006).

4.3.3 Frequency and intensity of storms, droughts or fire events

In forest ecosystems damaged by windthrow, abiotic conditions (e.g. temperature, humidity) are severely changed and, depending on the dimension of the disturbance, local to regional dynamics of insect populations and species composition are strongly affected (Bouget and Duelli 2004). The consequences for insect species may range from habitat loss to a gain of nutrition and living space. Several species of bark beetles and cerambycids (e.g. the long horned beetle, *Monochamus sutor*) benefit from the increase of attractive micro-habitats on sun-exposed deadwood and weakened standing trees in forest gaps, areas or stand edges newly created by windthrow. Thus, higher frequencies of catastrophic storm events in combination with temperature increase and periods of drought will trigger mass propagation, especially of significant pest organisms such as *Ips typographus* (Wermelinger 2004, Netherer and Schopf 2010).

4.3.4 Increase of atmospheric CO$_2$

Diverse and often unclear consequences of changes in plant nutritional quality and resistance

Plants react on increased concentrations of atmospheric CO$_2$ by enhanced productivity due to direct fertilisation effects (e.g. Lindroth et al. 1993). Commonly, carbohydrate levels in plant tissues rise under increased ambient CO$_2$ concentrations, leading to increased C: N ratios and decreased nitrogen concentrations. At the same time, production of carbon-based secondary metabolites, e.g. condensed tannins, may be enhanced (Ayres 1993, Cannon 1998, Ayres and Lombardero 2000, Hunter 2001).

CO$_2$ induced climate change may improve plant-host resistance against biotic agents. For example, an experimentally induced doubling of atmospheric concentrations of CO$_2$ lead to decreases in intensities of fusiform rust (causal agent: *Cronartium quercuum* f.sp. *fusiforme*) and...
pine pitch canker (causal agent: *Fusarium circinatum*) on *Pinus taeda* (Runion et al. 2010) and increased the latent period for *C. quercuum* f.sp. *fusiforme* on the alternate host, *Quercus rubra*. Insects also suffer, as they are often not able to fully compensate for reductions in plant quality by enhanced feeding, as was shown for *Lymantria* by Hättenschwiler and Schafellner (1999, 2004) and Schafellner and Schopf (2008). Larvae exhibited reduced survival and growth rates, as well as increased development times and reduced weights of pupae. Leaf-mining insects are generally expected to suffer from declined pupal weights, while some phloem-sap feeding insects, such as aphids, show improved performance by enhanced development and reproduction. Eventual levels of subsequent damage or defoliation may however be mediated by CO₂-induced increases in plant biomass or alterations in population densities (Cannon 1998, Hunter 2001, Battisti 2004).

The physiological effects of increased CO₂ levels are highly variable among different plant and insect species. Induced changes in leaf chemistry differ with tree species (Cannon 1998, Hunter 2001, Battisti 2004), which consequently leads to unequal responses of polyphagous insects depending on the host (Lindroth et al. 1993, Williams et al. 2000, Hättenschwiler and Schafellner 2004, Wang et al. 2009). Due to the wide scope of different reactions, predictions of future developments are of high uncertainty. Moreover, plants and herbivores are exposed to complex interactions among elevated levels of CO₂ and changes in temperature, precipitation, and nutrient availability, leading to synergistic effects or mitigation.

**Positive effects for natural enemies?**

CO₂-mediated changes in plant chemistry will have impact across all trophic levels and also influence the nutritional quality of herbivore larvae for natural enemies. There is evidence for beneficial effects for certain parasitoids (Schafellner and Schopf 2008), so that a combination of increased rates of parasitoid attack and reduced host plant quality might result in increased mortality rates and consequently decreased population densities of some herbivore species (Hunter 2001).

### 4.4 Impact of climate change on risk of outbreak and species distributions

The effects of climate change are going to be different for each host-pathogen system in different locations, so that it is impossible to draw general conclusions. A combination of all climatic factors, temperature, precipitation, humidity, CO₂ will almost certainly play a role in all biotic agents-hosts interactions. Moreover, changes in the associated flora (and fauna) at different trophic levels as climate changes may lead to additional modifications to the host-pathogen interaction. Further, it is important to note that not only alien invasive pathogens will be affected by climate change; the distributions and activities of native pathogens will also change.
4.4.1 Possible consequences of climate change with regard to the risk of pest outbreaks

- **Increased area, frequency, or intensity of outbreaks**

Outbreaks of forest biotic agents caused by native or introduced species are predicted to become more frequent and intense, as drought and other abiotic stressors are amplified under climate change, and as climate favours improved developmental conditions for the biotic agents. For example, *Ips typographus* populations may increase as more generations per year become possible (Section 0, Wermelinger 2004, Lange et al. 2006, Baier et al. 2007). Comparable conditions in North America and Canada led to recent broadscale eruptions of *Dendroctonus rufipennis*, *Dendroctonus ponderosae*, and *Ips confusus* (see Box 2-3, Wermelinger 2004, Lange et al. 2006, Baier et al. 2007). Other tree diseases, both native and alien, for which potential effects of climate change may lead to increased outbreaks include various stem and shoot cankers, blights, rusts, and foliage diseases (La Porta et al. 2008, Sturrock et al. 2010). Dothistroma needle blight, caused by *Dothistroma pini* and *D. septosporum*, has in recent years spread widely in Europe, causing severe defoliation and incremental damage to several important timber-producing gymnosperm species, principally pines (Watt et al. 2009). In the first decade of the current century, Dothistroma has increased so rapidly that it has lead to moratoria on the planting of *Pinus nigra* var. *laricio* and *P. contorta* var. *latifolia* in the UK (Sturrock et al. 2010). A large increase in damage has also been reported in France (Villebonne and Maugard 1999) and in several Central MS. Similar problems are present in the Pacific North West (Woods et al. 2005, Welsh et al. 2009). The pathogen has been shown to survive in very cold (arctic) to sub-tropical environmental conditions (Watt et al. 2009), with infection reliant on periods of needle wetness and temperatures in the range of 5 – 25°C (Gadgil 1974, Bulman 1993). The distribution of Scleroderris canker (brunchorstia dieback), caused by *Gremmeniella abietina* (syn. *Brunchorstia pinea*, *Scleroderris lagerbergii*), is likely to be markedly affected by climate changes, as the disease is particularly impacted by wet spring conditions and cool summers with high rainfall and relative humidities (Butin and Hackelberg 1978, Uotila 1988, Venier et al. 1998). Although the disease is already present on many pine species from the Mediterranean region, northwards into Scandinavia, the predicted strong fluctuations in climate in the future will lead to increased outbreaks of epidemic events throughout the range of the pathogen.

- **Establishment of more stable populations of the biotic agents at range limits**

Usually, populations at the limit of a species range tend to be smaller or subject to greater fluctuations than populations well inside the species range, as conditions are already less suitable, and competition with other, better-adapted species becomes greater. Climate change, may make the northern fringes of a species distribution more suitable, resulting in extensions of biotic agents outbreak zones. In the boreal zone, areas disturbed both by fire and insect-induced mortality may extend (Malmström and Raffa 2000, Volney and Fleming 2000). Another possible consequence would be the loss of the control factors limiting outbreak areas, such as synchrony with host development or winter mortality.

- **Negative feedback of altered environmental conditions on the biotic agent**
As discussed earlier, when climatic changes go beyond the tolerance range of a given species, this may negatively impact its population viability. Similarly, a reduction in the nutritional quality of host plants may lead to decreased growth rates, reduced fecundity and survival of the biotic agents. Finally, climatic changes may result in lost synchronisation between the biotic agent and its host plant, resulting in less risk of Zeiraphera diniana (Baltensweiler 1993, Esper et al. 2007).

**Intensification of background herbivory**

Temperature increase has been observed to be associated with enhanced damage of forest stands by defoliating insects even under low population levels (Wolf et al., 2008). These low-level damages have the potential to totally alter the nature of forest ecosystems, and might for instance promote a change of birch-dominated forests into coniferous forests in northern Europe in the long-term.

### 4.4.2 Changes in species distribution

**Altitudinal and latitudinal range expansions**

The distributional ranges of species are determined by the availability of suitable habitats, in interaction with favourable climatic conditions. Host tree distributions are generally less limited than the geographical occurrence of most biotic agents, so that the range of a biotic agent could change rapidly in response to changing environmental parameters. Typical examples of such shifts are, *T. pityocampa* (see section 2.3.3; Hodar and Zamora 2004, Battisti et al. 2005) or *I. typographus* (see section 0; Jönsson et al. 2009). There is also already some evidence that root-rot species in the genus Phytophthora are spreading with climate change (Jung 2009). The potential for range expansion under climate change of the most damaging root-rot causing Phytophthora species, *P. cinnamomi* has been modelled several times over the last 15 – 20 years (Brasier and Scott 1994, Bergot et al. 2004, Marçais et al. 2004, Deprez-Loustau 2007). The predicted changes in temperature and humidity used in most climate modelling indicate that activity of the pathogen will increase in some areas as the temperature and rainfall increase, but will decrease in other areas, due to a likely reduction in rainfall. The most detailed modelling for *P. cinnamomi* in Europe was carried out in France (Bergot et al. 2004, Marçais et al. 2004, Deprez-Loustau 2007); the pathogen survives more readily in infected host tissues than suggested by the original temperature survival work carried out using in vitro methods (Benson 1980), explaining the occurrence and apparent survival of *P. cinnamomi* in regions outside what has been hitherto considered the climate range of the pathogen (e.g. Chavarriaga et al. 2007). Moreover, increases in temperature will enable the pathogen to cause disease at higher latitudes and higher elevations.

**Northward range extension**

Northward range extensions are likely in response to increased summer temperatures and extended growing seasons (Williams and Liebhold, 1995, Ayres and Lombardero, 2000). Indeed, Northern range boundaries of many insects are frequently defined by the degree of frost resistance of overwintering stages (Cannon 1998, Rouault et al. 2006). Non-diapausating, but freezing-sensitive species, such as anholocyclic aphids, may also show less mortality during increasingly warm winters (Bale et al. 2002). Canker pathogens, such as Biscogniauxia charcoal diseases of oaks and beech, Seiridium canker of cypress and Sphaeropsis shoot blight of pines...
have clear associations with climatic factors, that are translating in northward expansions. *Biscogniauxia mediterranea*, cause of charcoal canker in Mediterranean oak woodlands, is first noticeable in the Mediterranean after severe drought events coupled with high temperatures (Vannini and Valentini 1994, Jurc and Ogris 2006, Vannini et al. 2009). The geographic range of this pathogen is expected to extend northwards in Europe as mean summer temperatures increase (Vannini and Valentini 1994, Desprez-Loustau et al. 2006, Vannini et al. 2009). Similarly, chestnut blight, caused by *Cryphonectria parasitica*, which killed or caused the decline of many chestnut plantations and amenity trees is currently spreading Northward e.g. in France and in other central MS.

Climate change already appears to be altering the geographic range of *Sphaeropsis sapinea* (*Diplodia pinea*; La Porta et al. 2008, van Staden et al. 2004). In Europe, the disease was more or less confined to the Mediterranean basin, but in the late 20th Century, damage was also noted in Central Europe (Jankovsky and Palovcikova 2003, Hanisch et al. 2006). Spores of *S. sapinea* are tolerant of temperatures as high as 55°C (Jacobs 2005). *Seiridium cardinale*, the cause of canker and dieback on Cupressus spp., was introduced into Europe from its native range in California, and is now widespread in the Mediterranean area, from Turkey to Portugal (Neves et al. 1986). The pathogen is very tolerant of high temperatures (Graniti 1998) and, with climate change, is likely to cause serious problems on Cupressus species planted in more Northern MS.

**Range shifts and/or (Southern) range contractions:**

As species move northward with changing climate, corresponding range shifts or contractions may be expected in the southern parts of their ranges. When temperature conditions in southern or warmer edges of a species occurrence consistently exceed the thresholds for heat stress, or the threshold necessary for the manifestation of diapause (Bale et al. 2002), habitat losses may be observed. For instance under current climate conditions, the ranges of *Lymantria monacha* and *Lymnantria dispar* (Karolewski et al. 2007, Vanhanen et al. 2007) are predicted to shift northwards by 100-900 km. However, the potential for fast adaptation of the biotic agents highlight the difficulty of making predictions. Adaptation by heat-tolerant genotypes is possible. Alternatively, in response to habitat loss, shifts in host preferences may enable species to settle in alternative habitats (Parmesan 2006).

**Establishment of alien biotic agents**

As the climate changes, it is likely that both native innocuous and alien invasive species will increasingly cause forest disturbances. Moreover, the potential for damaging alien biotic agents to become established in the future will be enhanced owing to increased niche availability (Ward and Masters 2007), combined with greater numbers of introductions as trade increases. The main risk is that ecosystems are not co-evolutionarily adapted to invaders by adequate defense mechanisms (Cannon 1998, Ayres and Lombardero 2000).

### 4.5 Conclusions for managing forest biotic risk as climate changes

Forests have a growing economic, social and ecological value, and play an important role in mitigating climate change. Given the complexity and multiple interactions between forest biotic
agents and climate change, it is important to assess management options for forests. Although trees in un-managed forests may have to adapt to climate change without human intervention, most EU forested areas can be managed to try to minimise forest dieback and biotic damages.

Forest management has traditionally assumed that current conditions will continue, and has often not taken into account climatic or ecological uncertainty to a significant extent (Johnston and Williamson 2007). Accounting for future climatic conditions should also be done before setting aside forest areas for conservation or other purposes, to ensure the long-term relevance of these areas.

The health of forests should be evaluated in terms of how future climate may influence the overall ecology of forest vegetation and biotic agents. A pro-active approach may have a better chance of reducing or avoiding negative impacts than reactive approaches, although both will be needed. Anticipatory adaptation can help decide which species should be planted today that will do well in a different climate in the future, and given the local site characteristics. As damage increases in a forest type, the forest manager will also need a prioritisation system to help him decide which tree species have a long-term future and are worthy of management, and which species may be lost and should therefore not be promoted. Unforeseen natural disturbances or extreme weather events will always call for reactive responses.

Three main types of adaptation can be distinguished, in line with sustainable forest management aims (Spittlehouse 2005):

1. **Actions aimed at societal adaptation**

   These actions would tend to be strategic ones, related to changes of forest management aims, or changes of public expectations. They typically address large areas and long-term time horizons, and promote objectives related to the desired future forest state.

2. **Actions aimed at forest adaptation**

   These actions in contrast are more operational and relate to smaller areas, and shorter timeframes. They involve changes in management practices, such as species selection or breeding programs.

3. **Actions aimed at adaptation to the impacts on forests**

   These may involve changes in how the forest is used, so as to adapt the use to the potential changes in forest productivity and management aims.

Determining the optimal set of actions requires a systematic approach to adaptation decision-making. No common guidelines concerning management of forest disturbances can be given at the EU level, but a framework for assessment may be recommended.

- **What is the adaptation problem and the main product of my forest?**

   It is important to clarify the forest current, but also the future management aims (wood production, climate mitigation, recreation, etc.). The current objectives may indeed need to be re-evaluated owing to climate change. Moreover, the characteristics of the site, trees and the main issue at stake also need to be known or defined.
Future trends in biotic disturbances

What are the current and future risks?

This includes the adaptive capacity of the forest sector, determined by the socio-economic and other characteristics of the sector and its participants. In practice, this means the degree of financial resources available, the human capital, and the perceptions of risk (Johnston and Williamson 2007).

Forecasting can be a useful tool in this step, as forest managers cannot rely on historical observations and experiences to forecast the future. A variety of modelling tools exist today (Beukma et al. 2007), which can be used to guide management of forests under a changing climate. They can for instance be used to relate current tree and biotic agent distributions, to future climate factors, to project future distributions, based on the physiological responses of the species to environmental factors (Pearson and Dawson 2003). However, these predictions remain plagued with high degrees of uncertainty, often resulting in large predicted ranges that are not useful for on-the-ground decision-making. Moreover, it is difficult to account for the adaptation potential of both the host trees and the biotic agents (Scherm 2004). Such forecasting can however provide qualitative insights of the magnitude and direction of change.

Choice of adaptation strategies

Given the answers to the above, a distinct set of operational and strategic management practices will be needed. Strategic management actions require planning, and mainstreaming into different policy areas, to ensure forest health issues can be responded to quickly and effectively. Operational management choices may involve facilitated or assisted migration of tree species with narrow resource requirements or poor dispersal ability, promoting tree species diversity in planting and or through breeding programmes that promote tree genetic diversity, and therefore resistance and tolerance to environmental stresses.

Evaluation of the adaptation strategies

Evaluation is important to assess how successful the measures were in meeting their objectives. Multi-criteria assessments, focusing on economic, social and environmental criteria will enable to get a balanced view on whether the sustainable forest management objectives have been met.

Economic criteria may involve efficiency and impact on competitiveness, social criteria the public perceptions and awareness about forest values and risks, and the environmental criteria the impacts on biodiversity. Other relevant criteria given the large uncertainty of these decisions may be the level of risk of severe outcomes and the degree of flexibility the chosen strategy provides.
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Chapter 5: Information, monitoring, early-warning systems

The information presented in this section is based on data collected through a desk study and a questionnaire sent to ten forest experts across the EU. The questionnaire and the list of experts can be found in Annex 2 and 3.

5.1 Key points

- The preoccupations about forest health have changed over the last 30 years. In the 1980s the concerns were mainly acid rain and atmospheric pollution. Currently, the main concerns are the decline of biodiversity and the risks linked to climate change.

- The Forest-Focus regulation (2003-2006) was key to ensure and fund the monitoring of forest health in each MS.

- Two independent EU-wide systems for monitoring of forest health exist: ICP-Forest and the NFI. In both systems, the monitoring of biotic agents or damages is poor, due to a lack of obligation to record this data or to the absence of standard protocols. Moreover, NFI and ICP-Forest are not designed to monitor rapid changes, such as biotic outbreaks. As a result, there are currently no long-term trends on biotic occurrence or damage in EU forests.

- There is little potential for homogenising the European forest data into one coherent dataset, whether between NFIs and ICP-forest monitoring systems, or among NFIs. However, some harmonisation of the NFIs across the EU MS may be possible, which would allow EU-wide data series on forest health.

- In sharp contrast to monitoring, hardly any early warning systems on biotic risks are currently in place in the EU. However, several general information systems are available. The case studies highlight that systems combining these three functions in a modular way tend to be most efficient and reliable.

- Large-scale, fixed grid surveys may not be adequate to capture rare, irregular biotic outbreaks. Efficient monitoring and EWS for biotic agents may require finding a balance between the density of the survey grid and the frequency of the data collection, so as to be able to detect even small-scale events, in a cost-efficient way.

- The Swedish forest health assessment system, through the Skogs skada database, is a good example of a participative approach, which allows the integration of the early warning, monitoring and information components in a flexible way. The French Forest Health Department (FHD) is particularly active by coordinating all the monitoring systems related to forest health through a well-trained team responsible for the detection of biotic damage.
5.2 Monitoring systems

5.2.1 Monitoring and inventory systems with a European coverage

Two main types of monitoring systems exist at European level, aimed at monitoring forest condition. National forest monitoring systems have been developed throughout Europe for almost a century, with the aim to assess the condition of forests and some parameters related to forest health (e.g. Wulff et al. 2010). NFI have been implemented since the beginning of the 20th century with the aim to produce and report timely and accurate estimates of forest resources. Over time, they have been expanded to cover more variables.

Another type of forest monitoring system was set up in response to several cases of forest decline in different areas of Europe that occurred in the late 1970s and the beginning of the 1980s. Air pollution was largely incriminated for this decline originally, which triggered the setting up in 1985 of the International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests), under the Convention on Long-range Transboundary Air Pollution (CLRTAP) of the United Nations Economic Commission for Europe (UN/ECE).

5.2.1.1 National forest inventories of the Member States

Traditionally, NFIs have been designed to provide country-based data of different forest variables of interest to the forest managers. They aim to describe the main features of forests in terms of size, condition and change by measuring parameters like forest area, timber volumes and tree species composition. They were thus more concerned with the productive features of forests than in an extensive description of the forests (Rego et al. 2004).

An increasing demand for information on non-productive functions of forests in recent years has led NFIs to “develop as more comprehensive natural resources surveys, broadening their scope in two major directions” (Kleinn et al. 2001):

- including additional variables (such as attributes related to biodiversity, climate change, sustainability), and
- expanding the target population towards non-traditional objects, like non-wood forest products and trees outside forests (Corona and Marchetti, 2007).

These changes led to more sophisticated sampling techniques, such that sample based inventories progressively replaced the woodland surveys. Today, most NFIs use a systematic sampling grid. NFIs now provide information about:

- forest area and land cover,
- resource management (growing stock and the balance between increment and felling),
- forestry methods and land use (felling systems, regeneration methods, road network density, specific methods such as ditching of swamp forests and soil scarification),
- forest dynamics with regard to different disturbance factors (fire, storms, insects, browsing), and
- forest state (tree species composition, age distribution, dimension of living trees, tree mortality and deadwood), and partly also about conservation measures, i.e. protected forest areas (Stokland et al. 2003).

The FAO published a guidance for NFI, the National Forest Monitoring and Assessment (NFMA) (FAO 2009). This manual provides guidelines and descriptions of the field data collection methodology and procedures used to inventory and monitor forestry and management practices. The methodology has been applied in several countries worldwide since 2000, including Sweden and Denmark.

At the national scale, each country should have a guidance to homogenise the data collection across each sampling campaign. Some of these guidances are publicly available on the internet, as in the case of Belgium34, Ireland35, Italy36, Luxembourg37 and Sweden38.

The type of data surveyed and the survey methods are not homogeneous among MS. While deadwood volume is always monitored, only some MS measure variables related to forest health, but not necessarily the same ones or in the same way (Table 5-1). Differences also remain between MS in the sampling design (e.g. permanent vs. temporary plots), the survey period and its frequency (Gabler et al. 2007). In addition, the NFI design makes it difficult to monitor rapid changes of biotic agents (e.g. the insect populations during the vegetation period). Also, in order to maintain high quality assessments there is a limit to the number of damaging agents that can be measured in this monitoring system.

The organisations in charge of the implementation of the NFI are usually relatively independent but funded by a Ministry. For instance, in France, the ‘Inventaire Forestier Naional’ is an independent organisation, but depends mainly on the Ministry of Agriculture for funding. In the UK, the Forestry Commission is also autonomous but officially merged with the DEFRA, while in Finland the NFI depends from the Finnish forest research institute, which is supported by the Ministry of Agriculture and Forestry. In many countries, the unit in charge of the NFI is supported by the Ministry of Environment.

38 Methodology available from: www.slu.se/PageFiles/26356/RIS_F%c3%a4ltinstruktion_2010_hela.pdf [accessed 22/9/2011]
Table 5-1 Indicators of the forest health measures by some MS during the NFIs. This information is based on a questionnaire sent by EU Cost Action 43 to each MS.

<table>
<thead>
<tr>
<th>Country</th>
<th>Forest health data measured</th>
</tr>
</thead>
<tbody>
<tr>
<td>Belgium Walloon Region</td>
<td>Damage and disease on the trees (syndrome, time and intensity)</td>
</tr>
<tr>
<td>Estonia</td>
<td>Prevalence of <em>Polyporus</em> sp. (this fungi are parasites and take advantage of the weakness of the trees to proliferate)</td>
</tr>
<tr>
<td>France</td>
<td>Prevalence of a limited number of relatively perennial biotic agents (<em>mistletoe, Melampsorella caryophyllacearum</em>) as well as symptoms (cankers, rot, etc.).</td>
</tr>
<tr>
<td>Hungary</td>
<td>Damage types and occurrence</td>
</tr>
<tr>
<td>Ireland</td>
<td>Negative factors limiting regeneration, peeling intensity, stem rot, tree root damages, defoliation type and intensity</td>
</tr>
<tr>
<td>Italy</td>
<td>Disease and damage</td>
</tr>
<tr>
<td>Lithuania</td>
<td>Damage (type, cause and intensity) and defoliation</td>
</tr>
<tr>
<td>Portugal</td>
<td>Evaluation of the forest health conditions (no more information available in the questionnaire)</td>
</tr>
<tr>
<td>Slovak Republic</td>
<td>Damage from pests and insects</td>
</tr>
<tr>
<td>Slovenia</td>
<td>Occurrence of leaf/needle injuries (disease and insect), determination of stem injury (canker, fungi, insects, wildlife) and freshness of the injuries</td>
</tr>
<tr>
<td>Sweden</td>
<td>Wounds due to mammals, signs of fungi (Resin top disease; rot or cancer; <em>Gremmeniella abietina</em>; needle cast; <em>Melampsora pinitorqua</em>, unspecified rust fungi; other fungi), signs of insect (notably <em>Tomicus</em> sp. and <em>Ips typographus</em>).</td>
</tr>
<tr>
<td>UK</td>
<td>Tree health and damages (no more precision were given in the questionnaire)</td>
</tr>
</tbody>
</table>

5.2.1.2 **Harmonisation of the NFIs: ENFIN and COST Action E43**

The need to develop common strategies and methodologies across NFIs, so as to have comparable and ‘user friendly’ data across Europe has become evident. At Vienna, in 2003, a network was established by representatives of the European NFI: the European National Forest Inventory Network (ENFIN), grouping 27 European countries (but not all from the EU). ENFIN members got funding from COST to provide comparable forest resource information across MS, through an improvement and a harmonisation of the concepts and definitions of the existing NFIs in Europe (COST 2009). The table in Annex 4 provides some examples of the heterogeneity observed between ten NFIs before the implementation of the cost action E43.

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39 Results of the questionnaire available from: [www.springerlink.com/content/jr8758/back-matter.pdf](http://www.springerlink.com/content/jr8758/back-matter.pdf) [Accessed 22/9/2011]
One major achievement of COST Action E43 was to collect and compare for the first time large amounts of data on NFIs. The information was collected through questionnaires directed to a wide group of NFI data producers in the participating countries. This comparison revealed the enormous variations among countries, and served as a catalyst for harmonising European definitions. The most comprehensive summary of how the NFIs are implemented in the different MS is available in Tomppo et al. (2010). Regarding the potential for harmonisation of the NFIs, COST Action E43 recognised that standardisation was not a realistic option. Some NFI features such as sampling designs and plot configurations for individual countries have been developed over time to accommodate the unique topography, climates, forest types and commercial interests of a given country. Further, the NFI are often required for many different local, regional and national agreements, which makes it difficult to elaborate a standard that would fulfil all these constraints. This means that no standard protocol for the data collection can easily be implemented in the 27 MS. Furthermore, there would be a risk of losing some of the longest forest data series available if the sampling method changed. Therefore, harmonisation appears a better option since it focuses on developing methods for producing comparable data despite the lack of standardisation.

COST Action E43 thus worked on the construction of common definitions and the development of procedures for producing harmonised estimates. All COST Action E43 participants agreed on key definitions related to forests. They agreed to use the forest definition provided in the FAO forest Resources Assessment (FAO 2005) as a reference definition for the purpose of harmonisation. This definition fixes the threshold crown cover at 10% (this threshold initially varied from 10 to 50% among national definitions) and the minimum tree cover at 0.5 ha (this threshold varied between 0.4 and 0.5 ha among national definitions). Since changes in practice, to adopt new definitions tend to be slow, tools for converting forest estimates corresponding to the non-reference definition to estimates corresponding to the reference definition have been defined. These “bridges” may involve procedures that are statistical in nature or not, depending on whether the national data is more extensive (reductive bridges), less than required (expansive bridges) or adequate (neutral bridges) compared to the reference definition (Stahl et al. 2010).

While COST Action E43 did not focus on biotic agents, it investigated the potential for a common approach to carbon and biodiversity reporting. These examples are illustrative for highlighting the nature of the harmonisation problems. The NFI have much of the data required by international reporting obligations on GHG emissions (including aboveground biomass, deadwood, forest area and also sometimes litter and soil) (Cienciala et al. 2008). However, the definitions of carbon pool change differ across MS, resulting in large discrepancies in the estimations (Cienciala et al. 2008). Regarding biodiversity, it seems encouraging that most NFIs already monitor the most ecologically relevant and technically feasible variables. However, the methods for assessing these variables in the field varied considerably among MS, as well as the field expertise (Winter et al. 2009). In both cases, guidelines for revising NFI methods, with tools for making estimates based on different definitions comparable were developed (Tomppo et al. 2010). Similar approaches could be developed for harmonising reporting on forest damages.

The countries participating in the EU COST Action E43 were strongly encouraged to adopt the common definitions or at least to implement procedures that would permit reporting according to the common definitions.
5.2.1.3 ICP – Forests

Presentation and aims of ICP - Forest

ICP Forest was implemented in 1985 in response to the concern about the alleged progressive deterioration of forests in Europe and elsewhere (Innes et al. 1993). Since 1986, around a systematic network of 6000 plots across the EU, comprising over 135,000 trees, have been assessed annually. This programme monitors forest condition using harmonised methods and criteria (UN/ECE and EC, 2000) and is organised in two monitoring intensity levels, corresponding to different networks, aims and features.

The level I (systematic level) provides an annual overview of forest condition based on 6000 fixed plots located on nominal 16x16 km grid (Figure 5-2). It covers all the EU MS except Malta. Three main types of data are collected: vital assessment of crown condition and damaging agents (defoliation, tree mortality, specification of affected parts, symptoms and causal agents), soil condition assessment (measure of air pollutants deposition) and foliar survey (to monitor tree nutrition). This assessment aims to produce information on the status and trends of forest condition across Europe, and contribute to a Europe-wide early warning system for diseases in forest ecosystems.

Despite a series of meetings, workshops, and cross calibration courses, the sampling schemes remain different among countries in terms of target populations as not all countries use the same definition of forest, grid density (the 16 x 16km grid is sometimes considered the average density of the network rather than a strict rule), sampling scheme (probabilistic vs. fixed selection of sites and trees) and plots design (e.g. fixed area vs. fixed number of trees). Moreover, the tree condition assessment criteria (reference standard, assessable crown, defoliation vs. crown transparency) are not harmonised and have changed over time in some countries (Ferretti 2010).
The Level II (arbitrary selection of plots) has been established since 1994 and consists in a research-oriented intensive monitoring system on a reduced number of sites chosen to represent major forest types. This system covered around 800 monitoring plots across Europe. Since the revision of the level II by the FUTMON project in 2009, only 200 Standards Plots and 100 Core plots have been maintained (FUTMON 2011). As the repartition of plots is heterogeneous compared to Level I plots, no conclusion at transboundary level can be drawn from its operation. No plots are established in Malta and Cyprus and three only are established in Greece. The aim of this part of the programme is to follow trends in stresses and responses over a period of at least 15-20 years. The core activities are the assessment of crown condition (annually), increment (every five years), and the chemical composition of foliage (at least every two years) and soil (every 10 years) on all plots. Additional continuous measurements on a sub-sample of the plots.

Figure 5-2 Plots distribution of the large-scale condition monitoring (Level I)

40 Map available from: www.icp-forests.org/DocsMonitoring/Level1-2.pdf [accessed 01/06/2011]
plots include soil solution (amount of the main cations and anions), ambient air quality, meteorology and atmospheric deposition. Compared to Level I, the Level II work goes into details of the causes and effects of stress factors: the observed symptoms, the tree crown location, and the age of the damage are specified.

During the Forest Focus regulation, the EC gave a budget of 10 to 15 millions EUR to ICP Forest each year. This fund was distributed between the ICP organisation itself (for the running costs and the expert meetings) and the MS involved in the network. As an example, Sweden received 320 000 EUR/yr and Spain 350 000 EUR/yr to carry out the Level I monitoring. The level II monitoring has a higher cost, and in Spain cost 500 000 EUR/yr. Since the cancellation of the Forest Focus regulation, the budget has decreased.

Assessment of damaging agents

Regardless the level of assessment, the assessment of damage causes consists of three major steps:

- symptom description,
- determination of the cause and,
- quantification of symptoms.

The symptom description is restricted to causal agents which may influence crown condition but the description is not restricted to foliage symptom and has also to describe damage to the other organs (branch, stem, etc.). Symptoms are classified in broad categories (woods, necrosis, deformation, etc.). In addition, the observation of nests of caterpillars, fungal fruit bodies, etc are interpreted as 'signs' of insects, fungi, etc. This symptom description is the first step in the identification of the causal agent. It indicates which part of the tree is affected and the type of symptoms it shows. An overview of the existing symptoms is given in the guidelines on 'Visual Assessment of Crown Condition'\(^1\). In order to reduce the time needed for the symptom description, countries have developed a national standard list with a complete symptom description for well-known and frequently occurring damage factors for their field teams.

Once the symptom description done, the causal agent has to be identified. However, the list of the damage symptoms is not always enough to determine the causal agent and in many case, further examination will be necessary. This step is carried out by trained observers and, when it is feasible and needed, confirmed by an expert phytopathologist.

Finally, the symptoms are quantified by referring to the assessable crown. The affected leaf area is expressed as a percentage of affected leaves/ needles. All these data collections have to be supervised by a national expert on tree condition assessment. To ensure a high level of quality, an independent check should measure again at least 5% of the sample plots (for level I and II).

All the data collected during the monitoring phase are sent to the National Focal Centres of ICP Forest, which are responsible for data processing, storage and submission. However, in practice, ,

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it appears that very few countries collect this information as since the end of the regulation Forest Focus (presented below in section 5.2.1.1) the MS have no more obligation to conduct such data collection.

5.2.1.1 Forest Focus (2003-2006)

Forest focus (Regulation (EC) No 2152/2003) Forest Focus was implemented as a follow-up to regulations on the protection of forests against atmospheric pollution and fire (EEC No 3528/86 and Regulation (EEC) No 2158/92), both of which expired on 31 December 2002. The Forest focus regulation, implemented for a limited duration of four years, was a key regulation for the assessment of forest health as well as a financial instrument to implement this assessment. This Directive obliged each country to monitor forest health and to report the data to the JRC, which was managing these data (JRC 2011). It also promoted the monitoring of biotic factors, which until then was not a subject of concern.

Indeed, in addition to the forest threats related to air pollution and forest fires, this scheme aims also to monitor wider issues such as biodiversity, carbon sequestration, the impacts of climate change, soil and the protective function of the forests (point 12 of the regulation). In particular, it aims to assess the requirements for monitoring other agents that have an impact on forests, such as biotic and abiotic factors, and factors of anthropogenic origin (article 1).

This scheme also provides for measures to (article 2):

- promote the harmonised collection, management and assessment of data,
- improve data evaluation at European Community level,
- improve quality of data and information,
- enhance the understanding of forests and,
- develop indicators and methodologies for the risk assessments.

The budget for the implementation of Forest Focus was 65 million Euros for the four-year period. This included 9 million Euros for fire prevention measures. The Commission financed part (50-75%) of the national programmes and the rest was funded by the MS themselves.

A detailed review of the actions undertaken under the EC Forest Focus was realised at the end of the Regulation. The Forest Focus monitoring network has been mainly successful in monitoring air pollutant and other damages/impacts to forests. The high spatial coverage and long time series required by the regulation make the data valuable for scientists and policy makers alike. In 2005, the assessment of biotic agents (like fungi and insect) were also included into the data collection.

The ending of the Forest Focus Regulation (December 2006) removed the obligation from MS to conduct coordinated, EU-level forest monitoring and the co-funding of the network. This led to a slippage back into uncoordinated monitoring of the ICP forest plots. Since 2007 a new financial

instrument, Life+, was implemented. This instrument has a much broader scope, as it can be used to fund any environmental project. However, it is not suitable for funding long-term monitoring activities, since recurrent programs are not eligible.

Thus today, all the data required in the crown condition guidelines are assessed in some countries only, which thanks to their own funds still continue this monitoring, such as France, Italy, and the Nordic countries.

5.2.1.2  **EUROPEAN AND MEDITERRANEAN PLANT PROTECTION ORGANISATION**

EPPO’s organisation has been described earlier in the report in the Box 1-2. EPPO is not a monitoring system but aims to register risks for the pests and diseases. Thus, the organisation has to keep updated and ensure the publication of these alert lists through the European countries.

The EPPO maintains two lists, which recommend that the organisms listed there are regulated as quarantine pests\(^{43}\). Quarantine pests are defined as pests of potential economic importance to the area endangered thereby, and not yet present there, or present but not widely distributed and being officially controlled. A pest named on list A.1 is a quarantine pest that is not present in the EPPO region. Those on list A.2 are quarantine pests that are present in the EPPO region but not widely distributed there and are being officially controlled. For the organisms mentioned in EPPO quarantine list, each European country has the obligation to develop a specific monitoring system to ensure the non-propagation of the organism. In the case for example of *Bursaphelenchus xylophilus*, Portugal and Spain have both implemented a national action plan to fight against this threat. In Portugal, a monitoring of the bark beetle *Monochamus* is realised each year between May and October; in case of presence of *Bursaphelenchus xylophilus*, the eradication campaign occurs between November and April. In addition, all the wood products are controlled before to be exported. In Spain, the plan is also focused on the monitoring and control for the presence of *Bursaphelenchus xylophilus*. A systematic survey is realised in all the high risk and medium risk areas.

There are two further lists maintained by the EPPO: the Alert List and the Action List\(^{43}\). The Alert List includes pests that possibly present a risk to EPPO member countries (i.e. early warning). The Action List includes the A1 and A2 pests that are recommended for regulation but are not yet included in EPPO member countries’ phytosanitary regulations. Among the top30 biotic agents identified (see Table 2-4), six are listed on list A.1\(^{44}\) and eight on list A.2\(^{45}\).

The main weak point of this pest monitoring system concerns the methodology used in each country to monitor the presence of pests. There are no guidelines or good practices provided by the EPPO. Each country is free to use the method of its choice. In addition, there is also an heterogeneity regarding the national institution in charge of the reporting: the monitoring can be

\(^{43}\) Available from: [www.eppo.org/QUARANTINE/quarantine.htm](http://www.eppo.org/QUARANTINE/quarantine.htm) [Accessed 30/6/2011]

\(^{44}\) Species listed in A.1: *Anoplophora glabripennis*, *Melampsora farlowii*, *Mycosphaerella gibsonii*, *Mycosphaerella larici-leptolepidis*, *Ceratocystis fagacearum*, and *Cronartium coelesporioides*.

\(^{45}\) Species listed in A.2: *Anoplophora chinensis*, *Agrilus planipennis*, *Melampsora medusae*, *Erwinia amylovora*, *Mycosphaerella dearmessii*, *Dryocosmus kuriphilus*, *Bursaphelenchus xylophilus*, *Gibberella (Fusarium) cincinata*
realised under either the authority of the Agriculture or the Forest Institutions. The former may be less competent to deal with forest pests.

5.2.2 Monitoring systems with a national, regional or local coverage

Besides the Level I and II ICP-Forest monitoring and the NFIs, 21 forest health monitoring systems were identified at national or sub-national level. Five of these were not further analysed as they had little relevance to biotic agents, and focused more on polluting agents, climate change and forest ecosystems. The other sixteen systems focused partially or totally on biotic agents (see Annex 5).

The Dutch monitoring system on insect pests is the longest running, with uninterrupted data series going back to 1946. Otherwise, the longest running systems tend to originate from countries from the Central Europe (Czech Republic, Estonia, Lithuania and Poland), which had a tradition of forest monitoring. Accordingly, half of the monitoring systems identified were located in Eastern Europe, and more must have existed as many of the former systems have been abandoned today due to a lack of funds. Most systems focus on insects, fungi or both, reflecting the importance of the impact of these organisms on forests (Figure 5-3).

The monitoring systems are usually implemented by national forest authorities, research institutions or national parks and tend to run annually. One exception in this list is the national Czech system which aims to report data for Plant health. All EU Member States have this reporting obligation.

Figure 5-3 Type of biotic agents targeted by the 16 reviewed EU monitoring systems

Sometimes, these monitoring systems are used as a decision aid for forest management. For example, the three very different Swedish systems identified provide the Swedish Forest Agency with a decision tool to apply specific mitigation schemes against biotic risk:

- The Swedish Target tailored inventory of forest damages, provides information on the less common and irregularly occurring forest pest and diseases
The spruce bark beetle flight activity monitoring provides annual risk assessments for tree mortality due to bark beetle attacks.

The Pine weevil monitoring system provides information about geographical and temporal variations in the extent of damages caused by pine weevil.

**Staff qualifications**

All monitoring systems identified relied on skilled and well trained staff to carry out land surveys. The systems were all supported by researchers or forest engineers. Volunteers may be involved in the field inventories, but always under supervision from a professional.

In the case of ICP-Forest monitoring systems, a guidebook on crown condition assessment explicitly specifies that the person in charge of the assessment should have the relevant skills and train the rest of the team which will be involved in the project.

Concerning the degree of qualification required for performing NFIs, a survey from Winter et al. (2008) reveals that most of the MS surveyed estimate that a medium to high level of expertise is required. A third of the responding MS estimated that the realisation of a NFI requires highly skilled staff with specialised training. Only Germany estimated that a typical forest staff member without specialised skills or training should be able to assess most of the key variables asked for the NFI.

**Cost of the systems**

This survey indicates that systems focused on insect detection, using traps (barber’s traps or pheromone traps) are the less expensive systems. The overall cost for such projects ranges between 55 000 and 77 000 EUR/yr. On average between 100 and 1250 EUR is spent each year on material and equipment. They require 40 to 80 person-day each year which represents two equivalent full time for one month.

Monitoring systems focused on general forest health present a higher cost (from 300 000 to several millions euro) since they require the collection of various data and skilled staff from different disciplines.

### 5.2.2.1 Transferability of the systems

It is very hard to evaluate the degree of transferability of each system since the information available is often scarce.

The density of the survey grid is key to ensure cost efficiency while providing frequent enough data to detect even small-scale outbreaks. Some system have developed very dense grids, such as the 'local' monitoring system of wood insects and *Bursaphelenchus xylophilus* observed in the Polish Roztocze National Park, which monitors 147 plots per year over 84 km² (1.75/km²) of forest area. On the other hand, some systems present a very low density grid, like the Hungarian Biodiversity Monitoring System for which 124 plots cover 93 000 km² (0.001/km²).
5.3 Information systems

The capacity to identify, prevent and stop the spread of harmful biotic agents depends on the availability to collect rapidly appropriate and updated information at the appropriate scale. Different types of information systems have been developed in the EU that can be useful to identify biotic agents, assess the risks of introduction or spread, and identify proper management responses to prevent or control outbreaks. They comprise general information websites, databases, tools, good practice guidance, expert networks, or information repositories. Efficient information systems would provide all of the above, with free and unrestricted access, and constantly updated information. In addition, expert registers on forest health in each MS would facilitate information transfer in case of emergency. The main stakeholders and information systems are described below.

Several experts group contribute to gather data or provide good practices on harmful biotic agents. At the international scale, the Invasive Species Specialist Group (ISSG), which is part of the Species Survival Commission (SSC) of the International Union for Conservation of Nature (IUCN) is a global network of scientific and policy experts on invasive species. The International Union of Forestry Research (IUFRO) has also developed a forum to exchange information on methods, techniques, research results and monitoring of forest insects and diseases. It covers mainly Central Europe but also worldwide.

The FAO also contributes to increase the knowledge about forest pests and diseases by publishing several reports on this concern. The institution published in 2009 a large report entitled ‘Global review of forest pest and disease’ which presented the main biotic agents per continent. In addition, the institution also proposes a specific website ‘Forest Health’ with news on biotic and abiotic concerns, advices on forest pest management strategies, a country pest overview per country as well as some training materials related to forest health and pest management.

5.3.1 Information systems with a European coverage

Few information systems with EU coverage are focused on forest biotic agents, and most of them result from COST actions (Table 5-2). However, several information systems that are focused on forests or invasive species may deal with forest biotic agents, at least in part (Table 5-2). In particular, identification tools and good practice guidances have been developed. For example, the Euroforest portal refers to an American publication for the identification and the management of the Dutch elm disease. In DAISIE, each alien species is presented in a factsheet. In addition to the general information about species, habitat and ecology description, the factsheet also provides some chemical, mechanical and biological management methods as well as the name of one or two key experts. Some databases also exist that either include or are focused on forest biotic agents. These are described in more detail below.
### Table 5-2 Information systems identified with European coverage

<table>
<thead>
<tr>
<th>General information websites</th>
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<tbody>
<tr>
<td><strong>Global Forest Information Service</strong>[^46]: provides the framework to share forest-related data and information through a single gateway. It promotes the dissemination and sharing of forest and tree-related information and knowledge among the global forestry community by developing common information exchange standards, building capacity and enhancing partnerships among forestry information providers and users. It also possesses a collection of forest databases including forest pests,</td>
</tr>
<tr>
<td><strong>IUFRO</strong>[^47]: promotes global cooperation in forest-related research and enhances the understanding of the ecological, economic and social aspects of forests and trees. It disseminates scientific knowledge to stakeholders and decision-makers and contributes to forest policy and on-the-ground forest management.</td>
</tr>
<tr>
<td><strong>Forest Europe</strong>[^48]: (The Ministerial Conference on the Protection of Forests in Europe) develops common strategies for its 46 member countries and the EU on how to protect and sustainably manage forests. It provides general information and publications on forest health and management.</td>
</tr>
<tr>
<td><strong>Euroforest Portal</strong>[^49]: provides maps and information about forest resources, forest policy and legislation, forest ecology and ecosystems, forest management and planning, forest protection, research, forest products industries, wood science, wood preservation, bioenergy, and non-wood forest products.</td>
</tr>
<tr>
<td><strong>Forest Data and Information System (led by the JRC)</strong>[^50]: has established a European Forest Data Centre which is the central point of forest information at European level in support of relevant EU policies, and as the basis of the European Forest Monitoring System proposed in the EU Forest Action Plan.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Databases covering forest biotic agents</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>DFDE 2003 – to date)</strong>[^52] database providing historical information about natural disturbances in the forests of Europe</td>
</tr>
<tr>
<td><strong>DAISIE</strong>[^53] database on the alien species found in the EU covering all taxa (inventory, species description, ecology and habitat, distribution, impact and management, experts)</td>
</tr>
</tbody>
</table>

[^46]: [www.gfis.net/gfis/search/datasets+and+databases/](http://www.gfis.net/gfis/search/datasets+and+databases/)
[^47]: [www.iufro.org/](http://www.iufro.org/)
[^48]: [www.foresteurope.org/](http://www.foresteurope.org/)
[^49]: [forestportal.efi.int/](http://forestportal.efi.int/)
[^50]: [forest.jrc.ec.europa.eu/forest-mapping/forest-cover-map/2006-forest-cover-map](http://forest.jrc.ec.europa.eu/forest-mapping/forest-cover-map/2006-forest-cover-map)
[^52]: [www.efi.int/portal/virtual_library/databases/](http://www.efi.int/portal/virtual_library/databases/)
[^53]: [www.europe-aliens.org/](http://www.europe-aliens.org/)
Databases covering forest biotic agents

FAO - country pest overviews\textsuperscript{54} includes information about the pests and diseases found in naturally regenerating and planted forests in the country, and also methods of forest protection.

Global Invasive Species Database\textsuperscript{55} database on alien invasive species that threaten native biodiversity, covering all taxonomic groups Species information is either supplied by expert contributors from around the world and by the Invasive Species Specialist Group (ISSG) of the IUCN Species Survival Commission.

EPPO database on quarantine pests\textsuperscript{56} quarantine lists and species information to draw the attention of EPPO member countries to certain pests possibly presenting a risk to them and achieve early warning.

Identification tools

EPPO database on diagnostic expertise\textsuperscript{57} provides an inventory of the diagnostic expertise available in the EPPO region. Its aim is to cover the expertise on regulated pests pests possibly presenting a risk to EPPO member countries and plants of the EPPO List of invasive alien plants. This database does not include common pests which are widely distributed in the EPPO region.

ICP Forest manual on Visual Assessment of Crown Condition and Damaging Agents\textsuperscript{58} aims at providing a consistent methodology to collect high quality, harmonised and comparable tree condition data about crown condition (biotic and abiotic damages).

Forestry images\textsuperscript{59} provides image of forestry pests insects, diseases and other damage agents.

Distribution Maps of Plant Diseases\textsuperscript{60} covers important disease affecting agriculture, horticulture and forestry.

Forest and shade tree pathology\textsuperscript{61} contains information about major diseases that affect trees and provides an aid to learn forest and shade tree pathology.

Information repositories

NOBANIS - European Network on Invasive Alien Species\textsuperscript{62} provides database on species introduced in the region, a literature database and factsheets on the most invasive aliens. This network mainly acts in North and central Europe.

CIRCA SANCO-EUROPHYT (2002 to date)\textsuperscript{63} has been implemented to protect the EU territory from introduction and spread of harmful organisms that pose phytosanitary risk.

\textsuperscript{54} www.fao.org/forestry/38536/en/
\textsuperscript{55} http://www.issg.org/database/welcome/
\textsuperscript{56} www.eppo.org/DATABASES/pqr/pqr.htm
\textsuperscript{57} dc.eppo.org/
\textsuperscript{58} www.icp-forests.org/pdf/FINAL_Crown.pdf
\textsuperscript{59} www.forestryimages.org/
\textsuperscript{60} www.cabi.org/default.aspx?site=170&page=1016&pid=2210
\textsuperscript{61} www.forestpathology.org/
\textsuperscript{62} www.nobanis.org/
\textsuperscript{63} ec.europa.eu/idabc/servlets/Doc3ee9.pdf?id=1935
Information repositories

Network for a European Forest Information Service (2003-2005) explored an overall information system architecture based on existing data reporting structures at national, EU and international levels.

Creafor⁶⁴ is a website listing all projects and programmes dedicated to forest adaptation to climate change.

Good practice guidance

Methodology of Forest Insect and Disease Survey in Central Europe (IUFRO)⁶⁵ has been established to exchange information on methods, techniques, research results and monitoring of forest Insects and Diseases in European forest protection Europe-wide. It covers mainly Central Europe but also world-wide.

PRATIQUE - Enhancements of Pest Risk Analysis (PRA) techniques⁶⁶ this recently finished FP7 project aimed at assembling PRA datasets valid for the EU, improve PRA techniques and robustness, and improve the user-friendliness and efficiency of PRA decision support tools.

COST Action FP0801- Established and Emerging Phytophthora⁶⁷ unites scientists and disease control experts working on Phytophthora in forest ecosystems with the overall aim of increasing understanding of the biology and ecology of Phytophthora species with potential to cause damage to European forestry.

DFDE (2003 – to date)

The DFDE is hosted by the European Forest Institute (EFI). The access for the data consultation is free but the users need first to register. The DFDE provides pan-European historical data on biotic (mainly insects and diseases) and abiotic disturbances in the forests of Europe. Some data even go back to 1449 and are based on an extensive literature review of published disturbance events, and their impacts on forests in terms of volume or area affected. Currently, the database contains approximately 27 500 records distributed in 34 countries, 26 from the EU (just Malta is missing) (see Figure 5-4), covering an affected volume of almost 4.4 billion m³ and an area of more than 172 million ha (www.efi.fi/projects/dfde). However, this data is biased towards catastrophic events, and confounded by the increased publication and monitoring effort in recent years, which make it hard to interpret the historical trends. Moreover, although the database has been designed with the aim to be easily updatable, no records have been added after 2001, limiting its ability to provide updated information on biotic damages. Nevertheless, this database is unique in bringing together all available information on biotic damages across Europe.

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⁶⁴ creafor.gip-ecofor.org/index.php?sujet=mission&ssmenu=realisations
⁶⁵ www.iufro.org/science/divisions/division-7/70000/70300/70310/activities/
⁶⁶ https://secure.fera.defra.gov.uk/pratique/
⁶⁷ www.cost.esf.org/domains_actions/fps/Actions/FP0801
Figure 5.4 Countries covered by the DFDE and number of records in the database per country

The data are presented by country. The user can select a time period from 1449 to 2001 as well as the damage type he wants (biotic, abiotic or both). Figure 5.5 shows how the data are presented.

Figure 5.5 Presentation of the data in the DFDE database


The Action brings together scientists and disease control experts working on *Phytophthora* in forest ecosystems. The overall aim is to increase the understanding of the biology and ecology of *Phytophthora* species with potential to cause damage to European forestry. This knowledge can be used for the development of effective control and management protocols for the problems caused by *Phytophthora* in European forests and woodlands. The Action is organised into four working groups, researching the following aspects:

- Invasive potential and pathology,
Host-pathogen interactions,
- Diagnostics, and
- Management and control.


The general aim of BAWBILT (Bark and Wood Boring Insects in Living Trees) COST Action was to provide a synthesis of the European research on bark and wood-boring organisms, so as to consolidate the knowledge base that would support improvement in forest conservation and protection in future. A database on damage and control methods was developed based on information gathered from the 24 participating countries (Figure 5-6). It includes all recent European references used during the project and some older ones. Unfortunately, the access of the database is restricted and needs to be purchased.

![Figure 5-6 European countries having participated in the BAWBILT network for the data collection](image)

**5.3.2 Information systems with a national, regional or local coverage**

Ten national information systems focused on forest biotic agents were identified, only six of which were in the EU27 (see Table 5-3). In the EU, the information systems identified do not target a specific agent, but rather give information on the main biotic agents types, in particular insect pathogens and fungi. In the UK, the two information systems identified are focused on species listed in the quarantine list by EPPO. An information system devoted to a single species was only found in France, for the saproxylic beetle. The Skogs Skada information system developed in Sweden stands out as a best-practice and is further described in the section 5.5.2.
### Table 5.3 National information systems identified, and biotic agents targeted

<table>
<thead>
<tr>
<th>Name of the information system</th>
<th>Biotic agents targeted</th>
<th>Start and end date</th>
<th>Description of the system</th>
</tr>
</thead>
<tbody>
<tr>
<td>French Information System on Saproxylic Beetle Ecology</td>
<td>Insects</td>
<td>2000 – to date</td>
<td>Aims at assessing and comparing the conservation status of stands, forests or reserves, by cumulating the species patrimonial indices of saproxylic beetle assemblages sampled in standardised conditions, and (carrying out functional analyses of the relationship between local dead wood profiles and ecological groups defined on the basis of ecological variables.</td>
</tr>
<tr>
<td>Protection of Slovenian Forests</td>
<td>Pathogens, Fungi, Nematodes, Insects</td>
<td>Unknown – to date</td>
<td>Aims to recognise and register the harmful biotic and abiotic factors found in Slovenian forests, accompanied by their appearance and a collection of samples.</td>
</tr>
<tr>
<td>Spanish Forest Damages Inventory</td>
<td>Pathogens, Fungi, Nematodes, Mammals, Insects, Plants</td>
<td>1987 – to date</td>
<td>Has been implemented in order to have better acknowledgement about health condition of Spanish forests.</td>
</tr>
<tr>
<td>Skogs Skada (Forest Damages) database (Sweden)</td>
<td>Fungi, Insect, Mammals</td>
<td>1990 – to date</td>
<td>See section 0 below.</td>
</tr>
<tr>
<td>Quarantine identification cards (UK)</td>
<td>Pathogens, Fungi, Nematodes, Insects, Plant</td>
<td>Unknown – to date</td>
<td>Contain images and information on hosts, distribution, symptoms, sources, development and disease status on various pests and diseases listed in EC plant health legislation.</td>
</tr>
</tbody>
</table>

**Examples outside EU**

<table>
<thead>
<tr>
<th>Name of the information system</th>
<th>Biotic agents targeted</th>
<th>Start and end date</th>
<th>Description of the system</th>
</tr>
</thead>
<tbody>
<tr>
<td>National Forestry Database Programme (Canada)</td>
<td>Insects</td>
<td>1990 – to date</td>
<td>Provides information on forestry inventory, forest fires, forest insects and diseases and pest control use.</td>
</tr>
<tr>
<td>Pests and diseases of forestry in New Zealand</td>
<td>Pathogens, Fungi</td>
<td>Unknown – to date</td>
<td>Provides information on: forestry pests, forestry diseases, pests of timber,</td>
</tr>
</tbody>
</table>
**5.4 Early warning systems**

It is now widely acknowledged that, together with preventing unwanted introductions, early warning and rapid response are crucial for mitigating epidemiological risks (Genovesi and Shine 2004, Wittenberg and Cock 2001).

Indeed, at the ninth meeting of the Conference of the Parties to the Convention on Biological Diversity, Parties were invited ‘to collaborate on the development and use of early warning systems, including networks of focal points, and on the development and use of rapid response mechanisms’ (Decision IX/4). In-depth review of ongoing work on alien species that threaten ecosystems, habitats or species. The need to develop effective global early warning and rapid response systems has also been stated as a priority action in the ‘Charter of Syracuse’ on biodiversity, adopted at the G8 Environment Ministers Meeting (22–24 April 2009, Syracuse, Italy).

The aim of an early warning system is to detect rare or new events dispersed in space and/or time. This kind of system thus needs frequent sampling over large forest areas. Indeed, the timing for an outbreak can be very short and very localised, making it challenging to monitor these episodic events. The ideal implementation of such early warning systems (EWS) would thus involve huge financial costs and manpower. EWS focused on the main supposed entry points and pathway for key species for which the risk expected is important may thus be a more realistic and effective strategy. Furthermore, the indicators to look for need to be carefully defined, since by definition this system will often have to target new organisms, which are not necessarily known by the technical staff, or even to science (e.g. in the case of fungi or pathogens).

No EU-wide early warning system currently exists that can monitor forest biotic agents. The ICP forest monitoring cannot be used as an early warning system since the sampling is too infrequent and the plot distribution is not representative. However, in the context of the invasive species, the EEA has already published a report in which it proposed a structure for a pan European early warning system.

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**Name of the information system** | **Biotic agents targeted** | **Start and end date** | **Description of the system**
---|---|---|---
The Exotic Forest Pest Information System for North America (US) | Insects, Nematodes, Fungi, Pathogens, Plants | Unknown – to date | Provides information on alien insects, mites and pathogenic organisms with potential to cause significant damage to North American forest resources.
How to identify and manage Dutch elm disease (US) | Fungi | Unknown – to date | Provides information about symptoms, disease cycle, and how to manage this pest.

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68 Decision available from the website of the CBD: [www.cbd.int/decision/cop/?id=11647](http://www.cbd.int/decision/cop/?id=11647) [accessed 5/12/2011]
warning system (EEA 2010). It suggests that to be able to respond adequately to the threat of alien species, an early warning system should cover the following functions:

- detection (surveillance and monitoring);
- diagnosis and data processing;
- risk assessment (or quick screening);
- reporting to competent authorities and circulation of information;
- response action (e.g. eradication, control);
- follow-up.

To date in Europe, no early warning system focused on forest health threats integrates these six points. However, the Early Warning System for Forest Health threats in the US has a structure close to the one proposed by the EEA: the American system includes all the steps from the identification of a potential new threat until the eradication and control. This system is presented in more detail in section 5.5.3.

Early warning is usually a component of existing information or monitoring systems. For example, the Swedish information system SkogsSkada, can serve as an EWS thanks to the continuous reporting activity done by professionals and volunteers (see Section 5.5.2 for more details). Similarly, the official surveillance based on phytosanitary legislation of the Czech Republic acts as an early warning system. The system of ‘Correspondents Observers’ in France, which aims firstly to monitor the French forests also allows for the quick detection of new, potentially harmful, organisms (see section 5.5.1).

In Sweden, a stand-alone EWS was identified for *Bursaphelenchus xylophilus*. This system was implemented in 2000, in order to prevent the spread of the nematodes from Portugal. It is led by the Swedish University of Agricultural Sciences and costs approximately 210,000 EUR/yr. It consists mainly in sampling of wood products and wood packaging and in clear cutting areas nearby risk areas. Samplings are performed regularly (the precise frequency is unknown to the authors of this report).

### 5.5 Best practices for information, monitoring and early warning of biotic agents

Three case studies have been developed in this section:

- the French Forest Health Department monitoring network,
- the Swedish forest health monitoring system, and
- the early warning system for forest health threats in the US
5.5.1 Case study: the French Forest Health Department monitoring French network

5.5.1.1 Organisation

The Forest Health Department (FHD) was created in 1989 in response to the crisis caused by acid rains and oak decay at the beginning of the 1990s. The FHD coordinates the ICP forest monitoring system and manages the resulting databases. It is the national contact point for the French forest organisations (Inventaire Forestier National, Office National des Forêts – ONF, the National Organisation of the Forestry Town, CEMAGREF) as well as for the European and international organisations, and spreads information down to the regional level. Five forest health poles (for the regions Aquitaine, Auvergne, Centre, Lorraine and Provence-Alpes-Côtes-d’Azur) ensure the contact with the regional forest organisations. The FHD is also supported by several research laboratories, in particular to help draw diagnosis. Figure 5-7 below summarises the interactions that the FHD has with other institutions, organisations and forest monitoring systems. The FHD team is currently composed of 25 permanent staff (11 engineers, 8 technicians and 6 administrative assistants, Nageleisen et al. 2010).

The two main activities of the FHD are the management of the ICP-Forest monitoring in France and the training and coordination of a network of trained correspondents. The FHD coordinates an important network of 220 trained correspondents (technicians and forest engineers) throughout the French territory, which allows a global monitoring of the forests all year-round. The correspondents belong to public (e.g. French National Forest Board) and private forest managers, and receive specific trainings from the FHD (presentation of the future outbreak risks of some biotic agents, training about damage identification, etc.). They then devote 15 to 25% of their time to FHD activities. More than 500 harmful biotic agents have been identified thanks to this network over 20 years of observation (Nageleisen et al 2010).

In 2006, the FHD convinced the NFI to quantify the prevalence of a limited number of relatively perennial biotic agents (mistletoe, Melampsorella caryophyllacearum) as well as symptoms (cankers, rot, etc). Previously, the NFI did not measure any biotic factors and measured only productive features. As the FHD has the skills to realise this assessment, they are officially in charge of that part of monitoring during the NFI. Accordingly, all forest health monitoring in France is performed by the FHD.

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69 Website of this departement available from: agriculture.gouv.fr/departement-de-la-sante-des-forets
5.5.1.2 The monitoring activity of the FHD

With the correspondent-observers network, the FHD manages a permanent monitoring of the French forest for all the entomologic, pathologic or abiotic issues. It follows the spread of the harmful biotic agents already present in the French territory. At least 50 different harmful biotic agents were monitored in 2010. Most of these were already known, but some new species proliferations were also noticed. As an example:

- an outbreak of the Pine-tree Lappet (*Dendrolimus pini*) was noticed in the Aube department, whereas no outbreak of this species had been observed on the French territory since the first world war, and

- *Lachnellula willkommii* (an native species in France but rather in the maritime regions) was identified on old plantations in the regions Puy-de-Dôme, Haute-Loire and Lozère.

Anomaly reporting

Any forest field worker that notices an anomaly should report it to the trained observers who have the skills to determine the nature of the damage. The observers then record their observations in the DSF database. Each year, an average of 10 000 damage descriptions is recorded in the database. The database is accessible to all the forest managers and to anyone upon demand. When a new threat is detected, this system allows a quick mobilisation of the forest practitioners since the FHD works in close cooperation with the regional authorities.

Figure 5-7: Presentation of the central place of the FHD into the forest monitoring network
(Modified after Nageleisen et al. 2010)
Quality insurance

The correspondent-observers have a robust background in insect and fungi identification, as well as in phytopathology in general. They have the obligation to check all the observations made by their non-trained colleagues. In addition, the Forest Health Department is well-connected with several laboratories. For unusual cases, tissue samples are sent to pathology laboratories so as to obtain a reliable diagnosis.

5.5.1.3 INFORMATION AND TRAINING ACTIVITIES

Two newsletters are published each year (in June and December) by the FHD on the website of the French Ministry of Agriculture. They are mainly designed for the correspondents observers and for the forest managers. These newsletters spread technical, scientific and policy (international negotiations) information about forest phytosanitary concerns.

Each year, the FHD also releases a forest health assessment, which brings together several articles that give an overview on the current events encountered during the year. For example, the main issues discussed in the 2010 forest health assessment were:

- chestnut gall wasp,
- the indicators of forest health,
- broad-leaved tree defoliators,
- Phytophtora lateralis, and
- the results of the national ICP-forest monitoring campaign 2009.

At the local scale, the FHD publishes some interregional or regional reports in order to give as precise and updated an overview as possible on the health of the French forests. These reports are based on the observations made by the correspondent-observers network. In addition, exceptional releases may occur to cover emerging issues (e.g. Phytophtora lateralis), the identification of new pathogens like Meria needle cast, caused by the fungus Meria laricis, or to present a new technique of analysis.

Since its creation, the FHD also manages a comprehensive database collecting all the observations of the correspondents-observers. All together, 69 814 entries were recorded from 1989 to 2006. This information source is used by the forest managers for the assessment of the health of their forest (in particular by measuring the recurrence of some pests and diseases), but also by researchers to carry out impact assessments or to draw hypotheses concerning the potential impacts of global warming.

The FHD has also an important function of training and advice towards the forest managers.

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70 Example of newsletter available from: [agriculture.gouv.fr/IMG/pdf/lettre_du_ds42.pdf](http://agriculture.gouv.fr/IMG/pdf/lettre_du_ds42.pdf) [Accessed 20/9/2011]
5.5.1.1 **ADVANTAGES AND SHORTCOMINGS**

This system is particularly efficient, by including at the beginning of the chain all the people working in the forest sector. Through this network, the system allows a global monitoring of the territory all year-round.

However, this system can be hard to implement in all the EU MS, since it needs a large budget and a good regional distribution of the officers in the country. In addition, being an independent organisation, the system needs to develop good collaborations with the different agencies or stakeholders in charge of the management of forests in the country. Moreover, despite its important number of correspondents-observers, this system fails to cover short-term outbreaks. More people would be needed to cover such episodic and localised outbreaks.

Currently, the researchers involved in this joint monitoring, information and early warning system are establishing training sessions for forest managers in Belgium and in Morocco, in order to teach them how to develop similar systems. This knowledge transfer is particularly interesting in the case of Belgium, since this MS will also use the French database for its data collection. France and Belgium will thus be able to share their information.

5.5.2 **Case study of the Swedish forest health monitoring system – a multi-dimensional system with an early warning function**

5.5.2.1 **THE SWEDISH FOREST HEALTH MONITORING SYSTEM**

**Organisation**

The Swedish forest health monitoring system has been entirely revised recently (Wulff et al. 2011). The new system is built around the idea that forest health information is primarily needed to support decision-making, for example to recommend new silvicultural policies at the national scale, or changes in management strategies to limit disease outbreaks for the landowners (Davis et al. 2000). Such a system thus needs to be multi-dimensional, with different components targeting different needs.

The comprehensive information, monitoring and EWS is divided into three main cooperating components (see Figure 5-8, Wulff et al. 2011):

- strategic monitoring, which provides long term data (Barth 2007),
- knowledge development with research projects focused on biotic and abiotic agents, to increase the knowledge base, and
- operational inventories of forest damage, including the Skogs Skada database, to provide reliable information on the less common and irregularly occurring biotic outbreaks.
Strategic monitoring in sparse large-scale networks provides information on long-term changes over time. These trends allow forestry professionals to be better prepared to adapt to changes in damage patterns. The NFIs are the primarily source of data in this component as in Sweden the measurements on the ICP Forests level I plots are now completely integrated in the NFIs. The measurements are carried out annually, which only allows coverage of the rather common and easily-detectable damages. This system may be complemented by long-term monitoring of insect populations in separate plots (e.g. *Ips Typographus*) in some regions, to provide a better understanding of the population dynamics.

The knowledge development component aims to answer the growing demand for further, reliable and up-to-date, knowledge about forest pests, diseases, and damages. Research projects aim to increase the understanding of the biotic and abiotic processes related to forest damage. For example, the monitoring data on *Ips Typographus* is used in different research projects for developing population models related to a changing climate, forest structure and management (Jönnsson et al. 2009). In addition, there are many damage symptoms for which the causes are poorly known, and where further research is warranted. Knowledge development is also necessary in a context of climate change, where new types of diseases or damages may occur.

The operational inventory, and in particular the “Skogs skada” database71 play both an early warning and information role, by giving early indications on biotic or abiotic damage, as well as an assessment of the extent, cause and consequences of damage.

**The Skogs Skada database**

A core part of the Swedish forest health system, and also the most original, is the ‘Skogs skada’ database. This participatory science database, implemented in 1990, is an openly accessible internet information and early warning system. Its contents are updated via the internet-based

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71 [www-skogsskada.slu.se/SkSkPub/MiPub/Sida/SkSk/About/About.jsp](http://www-skogsskada.slu.se/SkSkPub/MiPub/Sida/SkSk/About/About.jsp)
reporting and assessments filled in through active public participation. The database covers the damages caused by fungi, insects and wildlife on forest trees across Sweden, and also provides information on abiotic damages caused by frost or drought. It is organised by both damage type and harmful biotic agents: the user can access directly the information about the harmful biotic agent he has identified. When the user is not able to identify the biotic agent himself, a helping tool based on the damages observed guides him in the identification of the harmful agent. The cost of maintaining and running this information system has been estimated circa 20 000 EUR/yr.

The salient features of this database in terms of information contents and as an EWS are explained in more details below.

5.5.2.2 THE INFORMATION FUNCTION OF THE SKOGS SKADA DATABASE

The Skogs Skada database addresses the wide public as well as practitioners, decision-makers or scientists who are interested in finding out more about damage to trees and forests.

► Detailed information on biotic agents

The database comprises detailed information on approximately 200 species of insects, fungi, and vertebrates, allowing users to make a rather accurate diagnosis using the system. For each biotic agent identified, there is a map presenting its distribution throughout Sweden as well as information concerning its biology, its importance to forests and the factors that influence the spread of the species. The information is available as reports that describe the damages observed by a given agent in the MS.

► Damage diagnosis function

Skogs Skjada provides a tool to help practitioners diagnose the observed injuries. A drop-down menu allows the user to identify the reason of the observed injury based on the kind of injury observed, the tree species affected, the age of the tree, the damaged part, and the proportion of the stock presenting these injuries. Based on these parameters, the user obtains a list of potential damaging agents and can access the corresponding injury reports.

Each injury report is presented as a factsheet. For each damaging agent, the factsheet gives information on:

- injury description (symptoms),
- occurrence (maps of distribution area are available),
- history of the harmful biotic agent (date of apparition in Sweden for example),
  - biology of the harmful biotic agent,
  - risk for forests, and
  - the main harmful biotic agent with which it can be confused.

► News

This system keeps the users informed of the new outbreaks, pest apparitions and pest risks on the Swedish territory. The five more recent reports are presented as well as information
concerning seasonal pest activities. This webpage allow the users to be aware of the current potential risks.

5.5.2.3 **The early warning function of the Skogs skada database**

A key innovative aspect of this system, which ensures it is up-to-date and locally specific, is that is participative. It therefore allows early detection or a more general monitoring of biotic threats in Swedish forests. Any user, regardless of his prior knowledge on biotic agents, is invited to report the damages he notices. Data are thus available at the national, regional and local scale.

**Reporting methodology**

Reported injuries are collected continuously in the database, but only the five most recent reports are available on the website. The entire dataset does not appear to be openly accessible. However, the information provided in the reports is used to improve the sections on “diagnosis”, ‘injury description’ and to update the different maps.

The reporting is performed directly on the website of the information system, via an online questionnaire. Each observer (volunteer, practitioner, researcher, etc.) can report an observed injury. The following information is collected using a drop down menu:

- date when the damage was noticed,
- cause of the damage (more than 200 causes are mentions: harmful biotic agents, climate causes, nutrient starvation, etc.),
- tree species damaged,
- age of the tree (or the stock),
- extent of the damage (damage part of the tree, propagation in stock, orientation in stock, propagation in the vicinity) and,
- damage location (GPS coordinates, name of the closest municipality).

The last section of the form allows the observer to give additional data, such as the forest management measures used and the weather conditions. The observer is also invited to send samples or pictures of the damages to one of the three units (insect damage, fungal and abiotic damage and wildlife damage) in charge of the database.

**Validation by an expert team**

These reports are the starting point for researchers to determine the extent of biotic damage. Based on these data, an injury follow-up is performed by SLU's experts at a later date.

Each report is systematically reviewed by SLU's experts before it is considered acceptable and included in the database. The audit checks the plausibility of the report, and may contact the person who sent the report if needed. The reports that contain information that is unreliable or cannot be confirmed are deleted from the database. Else, if the damage is already listed in the database, it will be immediately published. If the injury reported is unspecified or refers to a ‘new’ pest, the publication of the report will be done only after the verification and the validation by an expert committee from SLU. The structure of the reporting, with its drop-down menus, clear instructions and systematic verification, ensures harmonised, coherent data are collected.
5.5.2.4 **ADVANTAGES AND SHORTCOMINGS**

The Swedish forest health monitoring system provides a comprehensive, reliable, and flexible source of information. It integrates more traditional building blocks, such as the NFIs and ICP monitoring systems, to novel approaches, such as the Skogs skada database. Moreover, the system combines monitoring with research at different levels, to ensure the reliability of the data collected. Overall, given its multi-functional nature, the Swedish forest health monitoring system has many aspects that would be easily transferable to other MS.

In particular, the main originality of the system is the Skogs skada operational information and EWS. Its main advantage is that it is open-access and uses participatory science, so that anyone can help to fill in the database by reporting an observed damage, regardless of his prior knowledge of forests or biotic agents. This system thus manages to have a large scale coverage, and data collection throughout the year, which allows early detection and precise mapping of the spread of the harmful biotic agents. The success of such participative systems depends on the fact that the reporting is simple and easily-accessible, and that the data can be scientifically useful. The supervision of each report by an expert group from the SLU ensures the quality of the information provided. The researchers can also raise the alert through the website very rapidly.

The relatively low cost of the maintenance of the database and its management by researchers from the Swedish University of Agricultural Sciences is also a considerable advantage: the salary of the staff is supported by the university and contributes to the reduction of the running cost of the system. Currently, this system only exists in Swedish and does not explicitly consider nematodes. But Sweden has a dedicated EWS in place for *Bursaphelenchus xylophilus*, as part of its obligation to ensure the nematode is not present on its territory (see Section 2.3.4).

5.5.3 **Case study: the EWS for forest health threats in the US**

- EWS overview and its application to insect and disease threats

The USDA Forest Service, working together with the Federal State, tribal and private partners, has developed the EWS for early detection and response to environmental threats to forests in the United States. The EWS consists of an ordered series of individual functional components that are grouped and arranged in loose chronological sequence according to four key steps of early detection and response. Each step contains four functional components, arranged as follows:

- Identify potential threat (nature of specific threat, mode of spread, environmental influences and vulnerable ecosystems),

- Detect actual threats (surveillance and reporting, systematic detection surveys, special detection surveys, verification and notification)

- Assess impacts

- Respond (consult and coordinate actions, implement appropriate treatments, monitor treatment effectiveness and restore affected area).

National coordination of the EWS for insects and diseases is provided by the USDA Forest Service Forest Health Protection (FHP) program. The Forest Service also organises an EWS insect and
disease steering committee comprised of representatives from key organisations and agencies, which has the task to strengthen the functioning of the early warning system.

Initial financial support of the EWS was provided by base program funds of each EWS agency or organisation. The EWS steering committee developed a strategy and action plan to identify future directions and resource needs.

Components and resources of the EWS for insect and disease threats

The early warning system is divided in four main steps (Figure 5-9):

■ The first step is to identify potential and existing threats to forest health so that detection activities may be planned and implemented. Identification of potential threats entails gathering, analysing, and organising pertinent information in ways that facilitate awareness and detection of potential threats.

■ The second step is the detection of actual threats. The goal of this step is early detection of
  - 1) invasive organisms before they become established, or if already established, before populations expand to previously unoccupied areas or build to harmful levels,
  - 2) native pests before populations build to harmful levels, or
  - 3) any disturbance phenomenon that threatens forest ecosystem function or sustainability.

■ The third step is impact assessment. This step encompasses all processes and programs related to the evaluation of potential impacts caused by environmental threats, including developing quarantine restrictions for invasive organisms and treatment options. Careful impact assessment is an essential prelude to appropriate response actions.

■ Finally, the fourth step is the response to environmental threats. Timely response and follow up is crucial to prevent or ameliorate unacceptable environmental damage from environmental threats. When new infestations of regulated invasive species occur, responses are coordinated by State Plant Regulatory Officials (SPROs) and APHIS State Plant Health Directors (SPHDs).
Information, monitoring, early-warning systems

Figure 5-9 EWS for insect and disease threats in the United States.

Example of species detected thanks to this system

Native to parts of Asia, the Asian gypsy moth (*Lymantria dispar*), was identified as a potential invasive pest of forest ecosystems in the United States during the pathway assessment for logs from Russia. Logs, ships and cargo from the Russian Far East have been considered as a significant pathway for the introduction of the species.

In 1991, port inspectors on a Russian ship docked in Vancouver, British Columbia, discovered hatching egg masses of *Lymantria dispar*. Because it was feared that young larvae might have been blown onshore at this and other ports visited by Russian ships, special detection surveys using pheromone traps were immediately initiated at port sites in North-Western North America. That year, *Lymantria dispar* were captured at three trapping locations in Oregon, Washington, and British Columbia, with confirmed identification using genetic techniques. Annual special detection surveys continued, and the Animal and Plant Health Inspection Service (APHIS) and the USDA Forest Service completed a pest risk assessment for this insect. A cooperative monitoring and suppression program was initiated in Russia in 1993.

In fall 2000, a single gypsy moth captured near shipping terminals in Portland, Oregon, was verified as *Lymantria dispar*. APHIS, Oregon State Department of Agriculture, and FHP cooperated to evaluate treatment options and assess potential response actions. The following
spring, over 900 acres were sprayed with insecticide in the vicinity of the capture site to eradicate any remains.

**Advantages and shortcomings**

This early warning system aids in ensuring that a comprehensive approach is taken when dealing with environmental threats, and that all relevant resources are brought to bear upon the problem at hand. The pathway through the EWS is not prescribed; rather it responds to the diverse nature of environmental threats, varying according to the particular circumstances of each application. An drawback of this system is that it requires the collaboration of a large number of organisations and authorities. This collaboration is already difficult at the scale of the US, and it can be assumed it would be similarly or more so at the EU scale.

5.6 Conclusions

5.6.1 **Key characteristics for an efficient monitoring of forest biotic threats**

**Building a multi-functional flexible system**

Biotic outbreaks are irregular, and their occurrence or location difficult to predict. Therefore, it is questionable whether large scale monitoring networks based on sparse, fixed grids offer the best strategy to acquire information about forest damages. Multi-functional systems adapted to different needs, such as the Swedish or US best practices, may be better suited. Such systems can combine long-term monitoring with more targeted monitoring of new or rare events. Long-term monitoring on a sufficient number of sites is essential to obtain time series for estimating trends. However, real information on biotic outbreaks occurrence and spread can only be obtained through intensive, targeted monitoring. In practice, these are usually limited to a small number of biotic agents, to maintain high quality assessments, and limit costs (financial and staff). However, the Skogs skada example highlights the potential of citizen science to help detect damages.

Given that the monitoring, information and early warning functions are highly interrelated, it is most efficient when these functions are combined within a single system. Then, early warning can, for instance, be used to launch targeted monitoring, and control actions can be implemented based on the knowledge provided by the information system. When these three functions are regrouped, economies of staff, costs, time and knowledge can be gained. Otherwise, multiple collaborations are necessary, with the different forest institutions, or decision-makers. The French and Swedish best practices are examples of different and successful integration of these functions. Both systems have a tight connection with research, as well as with the ICP and NFI monitoring networks. However, while in Sweden most of the early-detection is performed by the wide public, in France a well-trained team is responsible of the detection of biotic damage. The detection capacity of the French system is less sensitive than the Swedish one, but the latter may be difficult to implement in MS which have a much larger pool of biotic agents on their territory. This may indeed require the development of more complex online protocols to assign damage causes, and possibly a wider expert network to check the
observations. However, successful citizen science projects have been launched on similarly complex issues (e.g., pollinators surveys in France, BIO 2011).

From an organisational point of view, monitoring, information and EWS need to combine stability with sufficient flexibility. This can be a challenge, as both have quite different needs, but can help ensure the cost-efficiency of the system. Data provision for strategic decisions requires a long-term stable monitoring environment and a skilled workforce, i.e. typically full-time staff. On the other hand, data provision for operational decisions must be flexible, as it might be needed in localised places for restricted time periods only. The workforce needs not necessarily be skilled, as long as specific training can be provided. Such flexibility can be acquired locally, e.g. by using park wardens to also conduct monitoring as part of their normal activities, by recruiting a team of volunteers, or by using the general public, as in the case of the Swedish system.

Identification of the key characteristics for an early warning system for forest biotic agents

Based on the EWS reviewed during the project, a set of key characteristics can be identified that would ensure the system is efficient:

- Good knowledge about the vectors and biology of the biotic agent in order to ensure a good allocation or resources (e.g. staff and equipment) in the regions where it is the most needed. In particular, monitoring of the entry points and commodities can help prevent entrance or spread of biotic agents in the wild, as in the case of *Lymantria dispar* in the US, which mainly enters on logs from Russia. Alternatively, areas most suitable for, or at risk of, certain biotic agents can be subject to more frequent monitoring than others.

- Regular exchange of information between existing information, monitoring and EWS, so as to report the outbreaks and the spread of the harmful organisms in regions where their presence has been already assessed, provides a better overview of the potential risks.

- Close cooperation between the different authorities in charge of forests, and the quarantine pest authorities.

5.6.2 What can the EU do to improve forest health monitoring?

Promoting further harmonisation to obtain EU-wide comparable data on forest health

Different projects tried without success to homogenise the data collection protocols across the MS both between NFIs, and also between the level I of ICP-Forest monitoring and the NFIs. This objective is now unanimously considered unrealistic, as it would entail loosing precious data series. Therefore, procedures to harmonise the different data appear a more sensible option, as has been started under Cost Action E43 for carbon and biodiversity monitoring.
A new European regulation and funding system for forest health monitoring

Retrospectively, the Forest Focus regulation appeared very helpful by obliging MS to monitor health parameters in their forests, and by awarding funds to carry out this monitoring. Although LIFE+ was supposed serve as a substitute source of funding for the MS, it failed in this for two main reasons. Life+ funding cannot be used to support administrative staff, nor for recurrent activities, such as long-term monitoring programs. Simultaneously to the lack of funding, the removal of the reporting obligation has led many MS to reduce the quantity of data collected during the forest health monitoring programmes.
Disturbances are an important part of the natural forest ecosystems dynamics allowing regeneration. However, when these disturbances go beyond the natural range intensity, they can lead to forest dieback. Most work so far has usually focused on a specific type of biotic agent, or a restricted geographical extent. This has led to a sometimes segmented view of the problem, and a synthesis of the recent advances is timely.

Surprisingly, little is still known about the scale of damages caused by biotic agents in the EU, or about whether they represent a growing cause of concern. No reliable long-term trends are available at EU level, and evidence is mixed. However, regarding biosecurity, the probability that alien biotic agents will appear in the EU will increase, largely promoted by global trade. Thus the probabilities of establishment and spread of alien biotic agent is likely to increase, in particular as climate is changing and opening niches for these species.

Many uncertainties remain as to whether biotic agents will cause a growing threat to forests under climate change. There are clear knowledge gaps regarding the combination of factors that promote biotic outbreak conditions for most species, the ecological impacts of biotic outbreaks on the associated communities and the potentials of adaptation of both hosts and biotic agents. As a result, the predicted distribution of these species is also plagued with uncertainty. The shifts in tree species distribution and of their pathogens, whether native or alien, is however likely to lead to new opportunities for uncontrolled biotic spread.

Concerning the application of forest ecosystem management as way to securing resilient ecosystems we conclude that (a) EU’s forests are naturally very diverse in terms of their natural dynamic, (b) there is indeed a diversity of forest management and traditional landscape management systems available to improve the match between natural dynamic and management system in the EU, and (3) that there is room for improvement of the match between natural dynamic on the one hand, and forest management and traditional landscape management systems on the other. This is implicitly requested in the conclusions of the report on the State of Europe’s Forests 2011.

Thus, even in the face of uncertainty, actions can be taken to try and prevent future problems. Some lessons can be learnt from some of the better documented examples of biotic outbreaks. Clearly, early detection and action are a key to avoiding significant damage. However, it is often difficult to predict where a species will spread (e.g., *Thaumetopoea pityocampa*, section 2.3.3) or when more virulent strains will emerge (e.g., Dutch elm disease, section 2.3.7). This complicates decision-making for managers. Furthermore, most management decisions have associated socio-economical or environmental costs, and should thus be weighed carefully before implementation.

General recommendations for managing forests under changing climate and biotic pressures involve:
The impact of a biotic agent depends both on its aggressivity, and on the health of its hosts. Risk analysis of biotic agents should account for the risks of spread, of increasing virulence, and of weakening of the potential hosts. However, evaluation of biotic impact remains difficult, and often perceptions differ between forest managers and the wide public. This should account for the adaptive potential of host tree and biotic agents populations, which may be one of the most important predictors of the level of climate change effects.

Forest health monitoring is a crucial element of strategies for managing forests under climate change. It seems the NFIs, if integrated into wider multi-functional monitoring systems, already offer a strong basis on which to build an efficient EU-wide monitoring. Harmonisation procedures are in place or undergoing between MS, which should facilitate the establishment of shared information systems.

While it is often difficult to predict which alien biotic agents will become pests, legislation to prevent introduction, establishment and spread of alien pests and diseases in the EU is an important instrument to preserve forest health.

Management decisions should integrate the forest management aims, along with the characteristics of the host-biotic agent interaction. No common guidelines can be provided concerning biotic disturbances for the entire EU, but simple frameworks for assessment, helping to prioritise actions can be developed (see an example in Section 3.3.2).

In the future, forest management may include both increased ‘close-to-nature’ forest management methods to satisfy social and ecological functions, and more intensive methods more similar to agriculture than forestry based on naturally occurring tree species. This will lead to challenges in term of forest governance for an integrated landscape approach to spatial planning across forest land ownerships.

The diversity of Europe’s natural forest and cultural woodland landscapes is a tool that can help to stimulate mutual learning and knowledge production towards ecosystem-based forest management as pronounced by SFM policy, both locally and at the Pan-European level. Systematic comparative studies can support both successful SFM governance processes and tangible outcomes on the ground in forest and woodland landscapes. A prerequisite for a successful sustainable development process in line with SFM is that related policies and reference conditions are reflected in relevant education programmes. In actual operational forest management and governance there is a need for forest landscape managers who are capable to collaboratively lead sustainable development processes, including but not limited to forest management to produce a wide range of products, ecosystem services and values. These professionals would be key actors in as different areas as rural development, public health,

conservation, cultural heritage management, bio-energy production and timber production. This requires a comprehensive understanding of how to implement an integrated landscape approach, i.e. SFM as a collaborative learning process and sustainability as measurable objectives.

To conclude, there is a need for both (1) trans-disciplinary knowledge production that use landscapes as sampling units (Roux et al. 2010); and (2) encouraging local collaborative learning processes involving researchers, practitioners and other stakeholders on the ground, and in education. Ideally, networks of case study landscapes should be developed, representing different forest use phases and governance arrangements, and different collaborative learning approaches to develop locally adapted SFM solutions.
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Chapter 7: References


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Promoth, Final Report: WP1 “Core and expansion range”


References


References


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Disturbances of EU forests caused by biotic agents


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Studhalter RA (1914) Insects as carriers of the chestnut blight fungus. Phytopathology 4 : 52.


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Wallmo, U. 1897 Rationell skogsafverkning. Praktiska råd till såväl större som mindre enskilde skogsägare, samt svar på en fråga för dagen [Rational forest fellings. Practical advice to large as well as small private forest owners and an answer to a question of this day]. Länstidningens tryckeri, Örebro. (In Swedish.)


References


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Annex 1: Correspondence between the European forest type of the EEA and the more general classification used throughout the report

<table>
<thead>
<tr>
<th>CLASSIFICATION TYPE OF THE FOREST USED IN THE REPORT</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Boreal</strong></td>
</tr>
<tr>
<td>Mainly hemiboreal forest (2), boreal forest (1) and non-riverine alder, birch or aspen forest. In less proportion: mire and swamp forest</td>
</tr>
</tbody>
</table>
Annex 2: Questionnaire sent for the data collection on information, monitoring and early warning systems

SURVEY ON DISTURBANCES OF EU FORESTS CAUSED BY BIOTIC AGENTS

The aim of this survey is to increase the European Commission awareness about the existing information, monitoring and early warning systems currently used to preserve the European forests from biotic disturbances. These disturbances include pest attacks, invasive species expansion or grazing due to some animal species proliferation, but exclude game. Thanks in advance for your time and for returning the completed questionnaire to ulrike.jana@biosis.com by Monday 2nd May 2011

I- GENERAL INFORMATION ABOUT THE RESPONDENT

Identity of the respondent
Family name: ___________________________
First name: ___________________________
Email address: ___________________________
Name of your organisation: ___________________________

II- INFORMATION, MONITORING, EARLY-WARNING SYSTEMS

Please list below the information systems (ie databases), Monitoring systems and Early warning systems of which you are aware and that cover biotic agents, forest dieback, but also more generally any relevant system that covers e.g. forests, insect species or invasive species. You can also mention any related systems from which good practices can be learnt (e.g., agriculture or plant/animal health systems).

<table>
<thead>
<tr>
<th>Information systems</th>
<th>Monitoring systems</th>
<th>Early warning systems</th>
</tr>
</thead>
<tbody>
<tr>
<td>1-</td>
<td>1-</td>
<td>1-</td>
</tr>
<tr>
<td>2-</td>
<td>2-</td>
<td>2-</td>
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<tr>
<td>3-</td>
<td>3-</td>
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<td>5-</td>
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<td>6-</td>
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</tr>
<tr>
<td>7-</td>
<td>7-</td>
<td>7-</td>
</tr>
</tbody>
</table>

General Information Form 1/1
### SURVEY ON DISTURBANCES OF EU FORESTS CAUSED BY BIOTIC AGENTS

#### DETAILED INFORMATION ON THE SYSTEM

Please provide below detailed information on any of the above-listed systems you are familiar with. Even incomplete information is useful, so do not hesitate to fill in a detailed information form even if you only have information for a limited number of questions. Seven empty forms are available for you to provide information in this questionnaire. You can simply duplicate the form if you need more.

#### Detailed information Form 1

1. Name of the system described:

2. What is your position/role in this system?

3. How did you become aware of this system?

4. Can you provide us the address of the main contact in charge of this system (email address and phone number)?

5. Tick the corresponding type of system (X)

   - [ ] Information system
   - [ ] Monitoring system
   - [ ] Early warning system

   Comments:

6. What is the scale of the data provided by this system (you can tick several boxes)? (X)

   - [ ] International
   - [ ] National
   - [ ] Local
   - [ ] European (e.g. DAISIE)
   - [ ] Regional
   - [ ] I don't know

   Comments:

7. Can you tick which biotic agent(s) is (are) concerned/likely to be concerned by this system? (X)

   - [ ] Pathogens (bacteria, virus)
   - [ ] Fungi
   - [ ] Nematodes
   - [ ] Mammals
   - [ ] Insects
   - [ ] Plants
   - [ ] Other

   Comments:
SURVEY ON DISTURBANCES OF EU FORESTS CAUSED BY BIOTIC AGENTS

8. What is the size of the sampling area? No answer needed for the information systems

9. What is the frequency of the surveillance and reporting of this system? No answer needed for the information systems

10. Please specify the start and end date of the system?
    Start date:  
    End date (if relevant):

11. Do you know why this system has been implemented?

12. What kind of data are provided by this system?

13. How much does the implementation of this system cost (please give overall total or total per year, month as data availability permits)?

14. What do you think about the transferability of this system?

15. How could the EU help in improving this system?

16. Additional comments
Annex 3: Identification of the experts contacted during the survey

<table>
<thead>
<tr>
<th>Name</th>
<th>Institution</th>
<th>Country</th>
<th>Contact details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Milos Knizek</td>
<td>Forestry and Game Management Research Institute</td>
<td>Czech Republic</td>
<td><a href="mailto:knizek@vulhm.cz">knizek@vulhm.cz</a></td>
</tr>
<tr>
<td>Pasi Rautio</td>
<td>Finnish Forest Research Institute</td>
<td>Finland</td>
<td><a href="mailto:pasi.rautio@metla.fi">pasi.rautio@metla.fi</a></td>
</tr>
<tr>
<td>Jean-Luc Flot</td>
<td>Forest Health Department</td>
<td>France</td>
<td><a href="mailto:jean-luc.flot@agriculture.gouv.fr">jean-luc.flot@agriculture.gouv.fr</a></td>
</tr>
<tr>
<td>Louis-Michel Nageleisen</td>
<td>Forest Health Department</td>
<td>France</td>
<td><a href="mailto:nageleisen.dsf@wanadoo.fr">nageleisen.dsf@wanadoo.fr</a></td>
</tr>
<tr>
<td>Ricardas Beniusis</td>
<td>State Forest Survey Service</td>
<td>Lithuania</td>
<td><a href="mailto:ricardasben@yahoo.com">ricardasben@yahoo.com</a></td>
</tr>
<tr>
<td>Leen Moral</td>
<td>Alterra, Wageningen UR / Centre Ecosystems</td>
<td>The Netherlands</td>
<td><a href="mailto:Leen.Moraal@wur.nl">Leen.Moraal@wur.nl</a></td>
</tr>
<tr>
<td>Paloma Garcia</td>
<td>Servicio de Sanidad Forestal y Equilibrios Biológicos (SSF)</td>
<td>Spain</td>
<td><a href="mailto:at_pgarciaf@mma.es">at_pgarciaf@mma.es</a></td>
</tr>
<tr>
<td>Soren Wulff</td>
<td>Swedish University of Agricultural Sciences</td>
<td>Sweden</td>
<td><a href="mailto:soren.wulff@slu.se">soren.wulff@slu.se</a></td>
</tr>
</tbody>
</table>
### Annex 4: Main characteristics and differences between the NFIs of 10 EU countries

<table>
<thead>
<tr>
<th>Organism</th>
<th>Name of the institute</th>
<th>Date of creation</th>
<th>Status</th>
<th>Type of inventory</th>
<th>Cover</th>
<th>Sampling</th>
<th>Stratification</th>
<th>Remote sensing</th>
<th>Other</th>
<th>Form</th>
<th>Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Austria</td>
<td>Federal Research and Training Centre for Forests, Natural Hazards and Landscape</td>
<td>1961</td>
<td>Research</td>
<td>Forests only (the category “other woody areas” do not exist)</td>
<td>Systematic grid: 2.75x2.75km</td>
<td>No</td>
<td>No</td>
<td>Study on the type of properties and forestry roads</td>
<td>Circle and transects</td>
<td>Fixed and variable</td>
<td></td>
</tr>
<tr>
<td>Finland</td>
<td>Finnish Forest Research Institute (Metla)</td>
<td>1922</td>
<td>Research</td>
<td>All the categories</td>
<td>Systematic grid variable (different densities between the regions)</td>
<td>Yes but only in the north of the country</td>
<td>Yes</td>
<td>/</td>
<td>Circular</td>
<td>variable</td>
<td></td>
</tr>
<tr>
<td>France</td>
<td>National Forestry Inventory</td>
<td>1958</td>
<td>Administration</td>
<td>Forests and hedges</td>
<td>Systematic grid: 1.41x1.41</td>
<td>Yes</td>
<td>Yes</td>
<td>/</td>
<td>Concentric circles + transects</td>
<td>Fixed</td>
<td></td>
</tr>
<tr>
<td>Country</td>
<td>Name of the institute</td>
<td>Date of creation</td>
<td>Type of inventory</td>
<td>Status</td>
<td>Cover</td>
<td>Sampling</td>
<td>Stratification</td>
<td>Remote sensing</td>
<td>Other</td>
<td>Form</td>
<td>Area</td>
</tr>
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</tr>
<tr>
<td>Germany</td>
<td>Federal Research Centre for Forestry and Forest Products</td>
<td>1986</td>
<td>Research</td>
<td>Research</td>
<td>All the category</td>
<td>Systematic grid: 4x4km</td>
<td>No</td>
<td>Yes</td>
<td>Inventory along the forestry roads</td>
<td>Concentric circle</td>
<td>Fixed and variable</td>
</tr>
<tr>
<td>Italy</td>
<td>Agricultural Research Council – Forest Monitoring and Planning Research Unit</td>
<td>1983</td>
<td>Research</td>
<td>Research</td>
<td>Forests, other woodlands and isolated trees</td>
<td>Systematic grid: 1x1km</td>
<td>Yes</td>
<td>No</td>
<td>/</td>
<td>Circular</td>
<td>Fixed</td>
</tr>
<tr>
<td>Poland</td>
<td>Forest Research Institute</td>
<td>1995</td>
<td>Research</td>
<td>Research</td>
<td>All the forest areas</td>
<td>Systematic grid: 4x4 km</td>
<td>/</td>
<td>/</td>
<td>/</td>
<td>Concentric circles</td>
<td>fixed</td>
</tr>
<tr>
<td>Romania</td>
<td>Forest Research and Management Institute</td>
<td>1990</td>
<td>Research</td>
<td>Research</td>
<td>Only for forest with management plan</td>
<td>Systematic grid: 2x2km</td>
<td>No</td>
<td>No</td>
<td>Forestry road inventory, use of the information from the management plans</td>
<td>Concentric circles</td>
<td>fixed</td>
</tr>
<tr>
<td>Spain</td>
<td>National Forestry Inventory</td>
<td>1965</td>
<td>Administration</td>
<td>All the forest lands</td>
<td>Systematic grid: 1x1 km</td>
<td>Yes</td>
<td>Yes</td>
<td>/</td>
<td>Concentric circles</td>
<td>Fixed</td>
<td></td>
</tr>
<tr>
<td>Sweden</td>
<td>Swedish University of Agricultural Sciences</td>
<td>1923</td>
<td>Research</td>
<td>Research</td>
<td>All types of territory use</td>
<td>Systematic and random</td>
<td>Yes</td>
<td>Yes</td>
<td>Teledetection as a second informative source</td>
<td>Circular</td>
<td>fixed</td>
</tr>
</tbody>
</table>
### Annex 5: Presentation of the 17 national monitoring systems (15 from EU and 1 from US)

<table>
<thead>
<tr>
<th>Name of the monitoring system</th>
<th>Country</th>
<th>Harmful biotic agents targeted</th>
<th>Start and end date</th>
<th>Frequency of the surveillance/reporting</th>
<th>Sampling</th>
<th>Institute in charge of the system</th>
</tr>
</thead>
<tbody>
<tr>
<td>Walloon Permanent Forest Resources Inventory</td>
<td>Belgium</td>
<td>* Insects * Fungi</td>
<td>1994 - 2006</td>
<td>Annually</td>
<td>grid of 8.7 x 8.7 km which covers 534 plots</td>
<td>Division de la Nature et des Forêts</td>
</tr>
<tr>
<td>Monitoring programme in forest reserves and unmanaged forests - Fungi</td>
<td>Belgium</td>
<td>* Fungi</td>
<td>2000 – to date</td>
<td>Every 10 years</td>
<td>15 sampling sites randomly chosen</td>
<td>Research Institute for Nature and Forest (INBO)</td>
</tr>
<tr>
<td>Official surveillance (imports controls, plant health checks, detection surveys) based on phytosanitary legislation of the Czech Republic</td>
<td>Czech republic</td>
<td>* Pathogens * Fungi * Nematodes * Insects * Plants</td>
<td>Depend of the phytosanitary situation</td>
<td>Once a year, or more according to legislative requirements</td>
<td>Plant health checks are focused on plants for planting (forest nurseries, import of trees); in forest stands, detection surveys are carried out according to valid legislation and with respect to the level of risk of organisms.</td>
<td>State Phytosanitary Administration</td>
</tr>
<tr>
<td>Occurrence of Forest Damaging Agents</td>
<td>Czech republic</td>
<td>* Pathogens * Fungi * Nematodes * Mammals * Insects</td>
<td>1964 – to date</td>
<td>Main annually, other continuously</td>
<td>Whole Czech Republic</td>
<td>Forest Protection Service</td>
</tr>
<tr>
<td>Name of the monitoring system</td>
<td>Country</td>
<td>Harmful biotic agents targeted</td>
<td>Start and end date</td>
<td>Frequency of the surveillance/reporting</td>
<td>Sampling</td>
<td>Institute in charge of the system</td>
</tr>
<tr>
<td>-------------------------------</td>
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<td>----------------------------------</td>
</tr>
<tr>
<td>Forest Health Department Monitoring French Network</td>
<td>France</td>
<td>* Plants</td>
<td>1989 – to date</td>
<td>Each observer devotes 30 to 50 days per year to forest observations and sanitary diagnosis</td>
<td>The network uses the ICP Forest 16 X 16 km grid for its observations.</td>
<td>Forest Health Department</td>
</tr>
<tr>
<td>Hungarian Biodiversity Monitoring System: Plant community monitoring - Forests</td>
<td>Hungary</td>
<td>* Plants (5 invasive species are monitored)</td>
<td>2000 – to date</td>
<td>Annually</td>
<td>Total area monitored: 93 000 km² covers by 124 plots</td>
<td>Ministry of Environment and Water, Authority for Nature Conservation</td>
</tr>
<tr>
<td>Monitoring of Hymenoptera: Forest trap-nesting Hymenoptera</td>
<td>Lithuania and Poland</td>
<td>* Insects (all the hymenoptera)</td>
<td>2003- to date</td>
<td>Twice a year</td>
<td>30 sampling sites are chosen according to expert knowledge and cover 65200 km². 5 different samples are collected by plot</td>
<td>Institute of Ecology of Vilnius University</td>
</tr>
<tr>
<td>Monitoring of insect pests on trees</td>
<td>Netherlands</td>
<td>* Insects</td>
<td>1946 to date</td>
<td>Annually</td>
<td>Nation wide</td>
<td>Alterra, Wageningen UR</td>
</tr>
<tr>
<td>Polish Phytopathological Monitoring Programme to Assess the threat posed by fungi to forest</td>
<td>Poland</td>
<td>* Fungi</td>
<td>1996 – to date</td>
<td>Unknown</td>
<td>Unknown</td>
<td>Unknown</td>
</tr>
<tr>
<td>Wood-Eating Insects in the Pieniny National Park in Poland: monitoring of the</td>
<td>Poland</td>
<td>* Insects</td>
<td>1989 – to date</td>
<td>20 sampling campaigns per year</td>
<td>The total area monitored covers 23 km² and 38 sampling sites are implemented.</td>
<td>Pieniny National Park</td>
</tr>
<tr>
<td>Name of the monitoring system</td>
<td>Country</td>
<td>Harmful biotic agents targeted</td>
<td>Start and end date</td>
<td>Frequency of the surveillance/reporting</td>
<td>Sampling</td>
<td>Institute in charge of the system</td>
</tr>
<tr>
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</tr>
<tr>
<td>Number of <em>Ips typographus</em>, <em>Pityogenes chalcographus</em> and <em>Trypodendron lineatum</em></td>
<td></td>
<td><em>Insects (Pine Wood Nematode and butterflies)</em></td>
<td>1974 – to date</td>
<td>Annually</td>
<td>1 sample is collected per plot.</td>
<td></td>
</tr>
<tr>
<td>Monitoring of Wood Insects and the Pine Wood Nematode</td>
<td>Poland</td>
<td><em>Insects (Pine Wood Nematode and butterflies)</em></td>
<td>1974 – to date</td>
<td>Annually</td>
<td>The total area monitored covers 84 km² and 147 plots are implemented.</td>
<td>Roztocze National Park</td>
</tr>
<tr>
<td>Monitoring scheme: Monitoring of Ground Beetles</td>
<td>Poland</td>
<td><em>Insects (beetles)</em></td>
<td>2004 – to date</td>
<td>6 sampling campaigns per year</td>
<td>The total area monitored covers 151 km² and 2 sampling sites are implemented. 15 samples are collected per plot.</td>
<td>Wigry National Park</td>
</tr>
<tr>
<td>Monitoring of Fauna Abundance in the Pieniny National Park in Poland: Estimation of Number of the Red Deer, the Roe Deer and the Feral Pig</td>
<td>Poland</td>
<td><em>Mammals (Red Deer, the Roe Deer and the Feral Pig)</em></td>
<td>1994 - to date</td>
<td>Annually</td>
<td>The total area monitored covers 50 km² and a single sampling plot is monitored.</td>
<td>Pieniny National Park</td>
</tr>
<tr>
<td>Swedish Target tailored inventory of forest damages</td>
<td>Sweden</td>
<td><em>Pathogens</em> <em>Fungi</em> <em>Nematodes</em> <em>Mammals</em> <em>Insects</em> <em>Plants</em></td>
<td>2005 – to date</td>
<td>Annual</td>
<td>Varying depending of the actual damage outbreak</td>
<td>Swedish University of Agricultural Sciences</td>
</tr>
<tr>
<td>Name of the monitoring system</td>
<td>Country</td>
<td>Harmful biotic agents targeted</td>
<td>Start and end date</td>
<td>Frequency of the surveillance/reporting</td>
<td>Sampling</td>
<td>Institute in charge of the system</td>
</tr>
<tr>
<td>-------------------------------</td>
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<td>----------------------------------</td>
</tr>
<tr>
<td>Monitoring spruce bark beetle (<em>Ips typographus</em>) flight activity</td>
<td>Sweden</td>
<td>* Insect (<em>Ips typographus</em>)</td>
<td>1995 – to date</td>
<td>During the flight period of the beetle</td>
<td>The monitoring of the spruce bark beetle is ongoing in eight regions</td>
<td>Swedish University of Agricultural Sciences</td>
</tr>
<tr>
<td>Monitoring Pine weevil (<em>Hylobius abietis</em>)</td>
<td>Sweden</td>
<td>* Insect (<em>Hylobius abietis</em>)</td>
<td>2007 – to date</td>
<td>Annual</td>
<td>Sampled regeneration forests all over Sweden</td>
<td>Swedish University of Agricultural Sciences</td>
</tr>
<tr>
<td><strong>Example outside EU</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Forest Health Monitoring Program</td>
<td>US</td>
<td>* Insects * Disease agents * Alien species</td>
<td>1990 – to date</td>
<td>Annually</td>
<td>A permanent plot system is located on a based sampling grid, with fixed-area plots approximately 27 km apart. There are about 12,000 plots nationwide; about one-third of them occur in forests.</td>
<td>Partnership among the USDA Forest Service, State Foresters, universities, and the USDI Bureau of Land Management</td>
</tr>
</tbody>
</table>
Annex 6: Correlation between scientific and common names of the biotic agents mentioned in the study

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Agrilus planipennis</em></td>
<td>Alien emerald ash borer</td>
</tr>
<tr>
<td><em>Agrilus planipennis or marcopoli</em></td>
<td>Emerald ash borer</td>
</tr>
<tr>
<td><em>Alces alces</em></td>
<td>Moose</td>
</tr>
<tr>
<td><em>Anoplophora chinensis</em></td>
<td>Citrus longhorned beetle</td>
</tr>
<tr>
<td><em>Anoplophora glabripennis</em></td>
<td>Asian longhorned beetle</td>
</tr>
<tr>
<td><em>Armillaria spp.</em></td>
<td>Armillaria root disease</td>
</tr>
<tr>
<td><em>Biscogniauxia mediterranea</em></td>
<td>Charcoal disease</td>
</tr>
<tr>
<td><em>Bursaphelenchus xylophilus</em></td>
<td>Pine wood nematode</td>
</tr>
<tr>
<td><em>Capreolus capreolus</em></td>
<td>Roe deer</td>
</tr>
<tr>
<td><em>Cephalcia abietis</em></td>
<td>Webspinning sawfly</td>
</tr>
<tr>
<td><em>Ceratocystis fagacearum</em></td>
<td>Oak wilt</td>
</tr>
<tr>
<td><em>Cervus elaphus</em></td>
<td>Red deer</td>
</tr>
<tr>
<td><em>Chalara fraxinea</em></td>
<td>Ash dieback</td>
</tr>
<tr>
<td><em>Cryphonectria parasitica</em></td>
<td>Chestnut blight</td>
</tr>
<tr>
<td><em>Dendroctonus micans</em></td>
<td>Great spruce bark beetle</td>
</tr>
<tr>
<td><em>Dendroctonus ponderosae</em></td>
<td>Mountain pine beetle</td>
</tr>
<tr>
<td><em>Diplodia pinea</em></td>
<td>Diplodia blight</td>
</tr>
<tr>
<td><em>Dothistroma septosporum</em></td>
<td>Red band needle blight</td>
</tr>
<tr>
<td><em>Dryocosmus kuriphilus</em></td>
<td>Oriental chesnut gall wasp</td>
</tr>
<tr>
<td><em>Elatobium abietinum</em></td>
<td>Green spruce aphid</td>
</tr>
<tr>
<td><em>Epirrita autumnata</em></td>
<td>Autumnal Moth</td>
</tr>
<tr>
<td><em>Erwinia nimipressuralis and amylovora</em></td>
<td>Fire blight</td>
</tr>
<tr>
<td><em>Fusarium circinatum</em></td>
<td>Pitch canker</td>
</tr>
<tr>
<td><em>Gremmeniella abietina</em></td>
<td>Brunchorstia disease</td>
</tr>
<tr>
<td>Scientific name</td>
<td>Common name</td>
</tr>
<tr>
<td>--------------------------------------------------------------------------------</td>
<td>----------------------------------------------------------------------------</td>
</tr>
<tr>
<td><em>Heterobasidion</em> spp.</td>
<td>Root rot</td>
</tr>
<tr>
<td><em>Hylobius abietis</em></td>
<td>Pine weevil</td>
</tr>
<tr>
<td><em>Ips acuminatus</em></td>
<td>Engraver beetle</td>
</tr>
<tr>
<td><em>Ips confusus</em></td>
<td>Pinyon ips beetle</td>
</tr>
<tr>
<td><em>Ips sexdentatus</em></td>
<td>Six-toothed bark beetle</td>
</tr>
<tr>
<td><em>Ips typographus</em></td>
<td>European spruce bark beetle</td>
</tr>
<tr>
<td><em>Lymnaea dispar</em></td>
<td>Gypsy moth</td>
</tr>
<tr>
<td><em>Lymnaea monacha</em></td>
<td>Nun moth</td>
</tr>
<tr>
<td><em>Matsucoccus feytaudi</em></td>
<td>Maritime pine scale</td>
</tr>
<tr>
<td><em>Melampsora farlowii, medusa, pinitorqua, larici-populina and allii-populina</em></td>
<td>European rust</td>
</tr>
<tr>
<td><em>Mycosphaerella laricis leptolepidis</em></td>
<td>Needle cast of Japanese larch</td>
</tr>
<tr>
<td><em>Neodiprion sertifer</em></td>
<td>European pine sawfly</td>
</tr>
<tr>
<td><em>Odonta dorsalis</em></td>
<td>Locust leafminer</td>
</tr>
<tr>
<td><em>Operophtera brumata</em></td>
<td>Winter moth</td>
</tr>
<tr>
<td><em>Ophiostoma novo-ulmi</em></td>
<td>Dutch Elm Disease</td>
</tr>
<tr>
<td><em>Phaenops cyanea</em></td>
<td>Wood boring beetle</td>
</tr>
<tr>
<td><em>Phoracantha semipunctata</em></td>
<td>Lonhorned borer</td>
</tr>
<tr>
<td><em>Phytophthora ramorum</em></td>
<td>Sudden Oak Death</td>
</tr>
<tr>
<td><em>Pityogenes chalcographus</em></td>
<td>Bark beetle</td>
</tr>
<tr>
<td><em>Rhyacionia buoliana</em></td>
<td>Pine Shoot Moth</td>
</tr>
<tr>
<td><em>Rhyacionia frustrana and rigidana</em></td>
<td>Pine tip moth</td>
</tr>
<tr>
<td><em>Rhyynchophorus ferrugineum</em></td>
<td>Red palm weevil</td>
</tr>
<tr>
<td><em>Sciurus carolinensis</em></td>
<td>Grey squirrels</td>
</tr>
<tr>
<td><em>Scolytus multistriatus and scolytus</em></td>
<td>European elm bark beetle</td>
</tr>
<tr>
<td><em>Thaumetopoea pityocampa</em></td>
<td>Pine processionary moth</td>
</tr>
<tr>
<td><em>Tomica piniperda and minor</em></td>
<td>Common pine shoot beetle</td>
</tr>
<tr>
<td><em>Tortrix viridana</em></td>
<td>Green oak leaf roller</td>
</tr>
<tr>
<td><em>Zeiraphera improbana and diniana</em></td>
<td>Larch bud moth</td>
</tr>
</tbody>
</table>
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