

## 7 Forest Health in a Changing Environment

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**Abstract:** Ongoing changes in Earth's atmosphere and global climate change will have profound effects on future forest distribution and composition, as well as on the organisms that live and depend on forests. It is anticipated that due to increased stress, trees will become more susceptible to insects and diseases. At the same time, changes in temperature and precipitation regimes will cause alterations in the lifecycles of insects and diseases, increasing their host ranges and virulence, and becoming important drivers of change in forest ecosystems. In addition to these effects, forest health is threatened by invasive species. Accelerated global trade will increase the likelihood of cross-continental introduction of insects and pathogens, as well as plants and animals alien to native ecosystems, causing transformation and economic loss in forests already stressed by climate change. Forest resistance to climate-induced forest health decline can be increased by a number of measures, which include increasing forest complexity, conservation of genetic variability by protecting key populations, and assisting forests to migrate and adapt to the change. Complex decision-making systems may need to be developed to assess climate risk and forest health impacts. Forest pest management can play a role in mitigating the effects of climate change on forests and society.

**Keywords:** forest health, climate change, insects, diseases, invasive species



### 7.1 Introduction

Forests cover approximately 30% of the world's total land mass (FAO 2005). They form an integral part of life on Earth, and provide a range of benefits at local, national, and global levels. In a report to the eighth Session of the United Nations Forum on Forests (UNFF8 2009), the Secretary-General tabled a report (E/CN.18/2009/4) in which it was recognised that climate change poses a serious threat to forests throughout the world, and to the millions of people who depend on forests for their livelihoods (UN 2009). The report addressed the important role that carbon sequestration by forests can play in mitigating and adapting to climate change. The multi-year program proposed calls for actions to address the theme of forests in a changing environment, preventing deforestation and forest degradation, and combating de-

sertification. The United Nations Forum recognised that, while carbon sequestration is foremost in the public's attention at this moment, forests produce not only timber and important non-timber forest products, they also provide a large number of important ecosystem services. These include maintenance of biodiversity, water and temperature regulation, flood and landslide controls, and others. A healthy, sustainably managed forest is essential for the production of vital ecosystem services.

Forests are comprised of distinct ecosystems, each consisting of coherent communities of life forms and the physical environments with which they interact (Slocumbe 1993). As a consequence, forests are normally highly dynamic and subject to a variety of changes. Separating normal forest changes from changes caused by declining forest health requires that we understand the dynamics of change.



**Photo 7.1** Stands in the Chilcotin Plateau of British Columbia (BC), Canada normally sustain recurrent mountain pine beetle and fire disturbances. **Top:** Photograph taken in 2008 of an ongoing mountain pine beetle infestation (red trees) and evidence of past infestations that killed the trees on the ground (downed coarse woody debris). **Bottom:** Photograph taken in 2009 of the same stand after a wildfire.

Brad Hawkes

Changes in forests come primarily in the form of “disturbances,” which can be defined as any relatively discrete events in time that disrupt the successional development of a forest stand, ecosystem, or landscape, by affecting its population structure and changing resources, substrate availability, or the physical environment (Runkle 1985, Rykiel 1985, Attiwill 1994). Disturbances are often thought of as destructive events (Pickett et al. 1987) that set back, or even change the direction of the successional processes of regeneration and recovery. Disturbances occur in many forms (e.g., wind, fire, disease, insect disturbance, and logging) and over a wide range of scales (Dale et al. 2001). Disturbances leave ecological legacies (Dupouey et al. 2002) that determine future species composition, age structure, and spatial heterogeneity of the area (Radeloff et al. 2000, James et al. 2007) and, consequently, facilitate or impede the occurrence of future disturbances (Fleming et al. 2002, Kulakowski et al. 2003, Holzmüller et al. 2008).

The concept of disturbance regime (Sousa 1984, Johnson and Gutsell 1994) is typically used to classify disturbances of a particular type (e.g., insect outbreak) in terms of their areal extent, severity, duration, and frequency (Martell 2002, Bergeron et al. 2004). Since many areas are prone to multiple kinds of disturbances (e.g., wind, fire, disease, and insect outbreak), focusing on a single type of disturbance may not lead to a comprehensive understanding of a site’s disturbance history (Fleming 2000). Hence, it is also important to consider how different disturbances interact (Photo 7.1) (Fall et al. 2004, Parker et al. 2006, Bouchard and Pothier 2008). It is the disturbance regime complex that ultimately results in modifications of forest environments. This is exemplified by the large-scale landscape alterations that occur after the interaction of intense pest activity and wildfires.

### 7.1.1 Influence of Disturbances on Ecosystem Succession

It is widely recognised that different kinds of ecosystems are associated with different disturbance regime complexes (Runkle 1985), which points to the fundamental role that disturbance plays in the development of ecosystem structure and function (Aber and Melillo 1991, Attiwill 1994, Johnson and Miyanishi 2007). In sustaining itself at a site, an ecosystem goes through repeated cycles of maturation and renewal. Disturbances are the principal agents in these “renewal” cycles for releasing the tightly bound accumulations of biomass, energy, and nutrients that characterise an ecosystem at maturity (Gunderson and Holling 2002). This sudden release produces a pulse of available resources, which opportunistic species exploit as they effectively colonise the site and launch another successional sequence. This is a critical time in ecosystem recovery. The next successional sequence could retrace previous successional pathways at the site and thus culminate in an ecosystem with much the same properties (in terms of species composition, spatial distribution, soil qualities, etc. [Bever et al. 1997]) as the one that existed on the site immediately before the disturbance. In this sense, the original ecosystem has effectively retained the site. However, there are natural situations that result in the original ecosystem not retaining the site and another, sometimes radically different, ecosystem may begin to assert itself (Holling 2001, Scheffer et al. 2001, Folke et al. 2005). Depending on the mix of available resources and micro-environments available at the site, and the variety of species with an opportunity to exploit them, the new successional sequence may veer away from the successional pathway leading back to the ecosystem originally situated at the site, and move in a pathway culminating in a totally new ecosystem (Oliver 1980, Bazzaz 1996, Scheffer and Carpenter 2003, Jasinski and Payette 2005, Schröder et al. 2005) with its own unique disturbance regime complex (Mack and D’Antonio 1998, Raffa et al. 2008). Heath and Alfaro (1990) reported the replacement of lodgepole pine ecosystems by Douglas-fir in response to beetle epidemics.

### 7.1.2 Ecosystem Disturbances and Forest Health

The capacity of a forest ecosystem to absorb disturbance and reorganise while undergoing change and retain essentially its original function, structure, identity, and feedbacks has been termed “ecological resilience” (Gunderson and Holling 2002, Walker

et al. 2004). Within this context, a “healthy forest” can be considered one in which the underlying ecological processes of its ecosystems operate so that, on any temporal or spatial scale, they are resilient to the historical disturbance regime with which it evolved. However, when a forest ecosystem is subjected to disturbances outside its historical range, such as those expected from rapid climate change, large-scale land use-change, or logging, its resilience may be inadequate and collapse becomes possible. The challenge ahead is to ensure that forests remain healthy *for as long as possible* in face of the impending changes to Earth’s climate so that they remain productive and sustainable. This includes continuing to provide a share of world production and retain jobs, and having the capability to provide the economic, recreational and other environmental benefits and ecosystem services of the original ecosystem, and maintain the natural biodiversity of plant and animal life associated with that ecosystem.

This chapter reviews the concept of forest health in a changing environment, and discusses how to maintain healthy forests under climate change given anticipated increases in the extent and severity of disturbance by agents such as insects, diseases, and pollution. We note that the threat to forest health arises primarily from the independent and interacting effects of climatic warming, invasive species, inadequate forest management, and atmospheric pollutants.

## 7.2 Drivers of Ecosystem Health Change

### 7.2.1 Changes in Disturbance Intensity

Initially, ecosystem instability and loss of resilience in response to climate change will likely be a gradual process, occurring over a time scale of multiple decades. Outbreaks of insects and diseases, and increased fire frequency, will gradually alter the forest cover, giving rise to succession by species more adapted to the new environmental conditions. However, as disturbances reach catastrophic levels, such as the pine beetle situation in British Columbia, Canada, which began in 2001, disturbances could become drivers of change in their own right as dead trees return their stored carbon dioxide (CO<sub>2</sub>) to the atmosphere, thereby accelerating the process of change (Kurz et al. 2008). It is important to anticipate the threshold or tipping point (Fischlin et al. 2009) in which massive ecosystem transformation and reorganisation may occur.

Climate change and increased extreme weather events predicted by the Intergovernmental Panel on Climate Change (IPCC 2007) could alter many of



the basic processes that determine pest outbreaks (Ayres and Lombardero 2000, Parmesan et al. 2000, Volney and Fleming 2000, Walther et al. 2002). With climate change, disturbance regimes are expected to change, and the process of ecosystem replacement will accelerate. Changes in temperature will alter tree/pest synchrony, tree species distributions, and tree resistance. Simultaneously, changes in moisture and precipitation regimes – becoming drier in certain areas and wetter in others (IPCC 2007) – will cause geographic and host range expansion of insects and diseases, allowing organisms to occupy areas or hosts that were previously out of reach because of climatic constraints.

Insects can respond rapidly to environmental change due to their short lifecycles, mobility, great reproductive potential, and physiological sensitivity to temperature and moisture (Andrewartha and Birch 1954, Ayres and Lombardero 2000). Thus, climate change is expected to have strong direct and indirect effects on the survival, development, reproduction, dispersal, and geographic distribution of forest pests and their hosts (Ayres and Lombardero 2000, Volney and Fleming 2000, Bale et al. 2002, Walther et al. 2002). Warmer spring and summer temperatures will accelerate development rates and increase species' reproductive potential (e.g., multivoltine species may increase the number of broods produced per year) (Bale et al. 2002, Battisti et al. 2006, Raworth 2007, Finch et al. 2008, Tobin et al. 2008); warmer winter temperatures will increase overwintering survival (Battisti et al. 2005 and 2006, Veteli et al. 2005, Buffo et al. 2007). Low winter survival is normally responsible for maintaining many pest species below outbreak levels. Other climatic factors, such as humidity, also play a role in controlling insect survival and reproduction. In the Southern Hemisphere, where winters are not as cold and therefore overwintering survival not as important (because it rarely gets cold enough to kill overwintering insects), even milder winter temperatures may result in an extra generation of insects and result in increased damage.

As insects and pathogens modify their ranges in response to changes in host tree distribution and changes in climatic suitability, pest organisms will vacate areas of their “normal” distribution that will become no longer suitable in which to complete their lifecycles and move into new, more suitable areas and utilise existing hosts or shift to new hosts (Wilf et al. 2001). These changes are already causing some pests to expand their geographic ranges, either poleward latitudinally and/or upward altitudinally (e.g., Hodar et al. 2003, Parmesan and Yohe 2003, Root et al. 2003, Karban and Strauss 2004, Battisti et al. 2005 and 2006, Hickling et al. 2006, Buffo et al. 2007, Musolin 2007, Vanhanen et al. 2007). Thus, high latitude and high altitude pest species that are at the extremes of their distribution and have little



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**Photo 7.2** Egg mass of the pine processionary moth, *Thaumetopoea pityocampa* Denis & Schiff., on *Pinus sylvestris* Linnaeus, in the expanded outbreak area, at 1450 m, Mont Avic (Aosta, Italy), an unusual elevation for this insect species. The insect shows high performance and almost no mortality due to the absence of major natural enemies. A genetic analysis has shown that the population originated from different sources, suggesting that female moths are flying over larger distances than previously believed (Simonato et al. 2007).

ability to shift their range, may sustain loss of habitat and decline or become extinct, while low latitude or altitude species are expected to benefit, and expand their ranges northward and upward (Hill et al. 2002, Walther et al. 2002). However, many low latitude or altitude species (particularly specialists) may decline due to a lack of suitable habitat into which they can expand their range (Warren et al. 2001, Hill et al. 2002, Merrill et al. 2008).

An example of a forest pest shifting its range in response to climate change is the pine processionary moth (*Thaumetopoea pityocampa* Denis & Schiff.), an important pest of pine forests in southern Europe. This pest has expanded its range substantially (both northward latitudinally and upward altitudinally) during the last ten years, with high attack rates in previously unaffected areas (Hodar et al. 2003, Battisti et al. 2005 and 2006, Staley et al. 2006). This has been attributed to warmer winter temperatures that allow greater survival of overwintering larvae, and increased warm summer nights above a threshold

temperature required for flight take-off and nocturnal dispersal of females (Battisti 2006). If range shift and increased survival events become a global trend, there may be serious consequences to silviculture worldwide.

Effects of climate change that are more difficult to predict include altered synchrony between host-pest phenology, e.g., pest feeding stages must coincide with host food availability, and reproduction must coincide with suitable seasonal timing (Bale et al. 2002, Walther et al. 2002, Nealis and Regniere 2004, Harrington et al. 2007, Musolin 2007). Thus, changing climate regimes have the potential to disrupt the host/pest equilibrium, altering pest density and outbreak severity. Additionally, species that interact with insect pests (including parasitoids, predators, mutualists, or competitors) and help maintain pest populations under control, may also be affected by climate change (Ayres and Lombardero 2000).

Increased CO<sub>2</sub> levels in the atmosphere will not only increase temperatures, but will change precipitation patterns. Many shoot and needle diseases will respond to increasing precipitation by spreading to new ranges and hosts, and become more virulent. This is because tree pathogens (as any other fungi) need humidity for reproduction. Increased humidity makes forests more susceptible to diseases. Woods et al. (2005) linked increases in *Dothistroma* needle blight to increased precipitation.

However, anticipating which pathogens will increase in activity requires more precise forecasts not only of overall precipitation increase and seasonality, but also an accurate determination of the pathogen's lifecycle. For example, diseases may not intensify if precipitation increases only in wintertime, or if total precipitation increases are due to rain events that occur at larger intervals. In such cases, the outcome may actually be a negative influence on the development of forest diseases. Therefore, in order to correctly assess increased disease risk caused by climate change, it is of high importance to accurately predict future precipitation pattern changes, especially changes in humidity, during the sporulation period of fungal pathogens. Unfortunately, current climate models do not give reliable estimates of the level of humidity at a local scale, and, therefore, it is unclear if diseases like *Scleroderris* canker or pine needle casts, for example, will benefit from or be restricted by future climate conditions.

Range shifts and increased survival in pests, along with reduced host resistance due to drought stress, could have serious worldwide consequences for the health, productivity, and ecosystem services obtained from forests.

### 7.2.2 Invasive Species in a Changing Environment

Climate change may also create environmental conditions conducive to the establishment of invasive species (Ayres and Lombardero 2000, Walther et al. 2002, Buffo et al. 2007, Regniere et al. 2009). Some invasive species (including plants, fungi, animals, and microbes) have profound effects on the structure and function of forest ecosystems around the world. They can negatively affect tree growth, vigour, and survival; the yield and quality of plant products; and forest habitat, recreation, aesthetics, and cultural values (FAO 2009). They can also limit reforestation programs, cause the commercial abandonment of certain tree species, and may make it necessary to cut large areas of infested trees (FAO 2009).

Invasive species are often characterised by high population growth rates, high reproductive capacities, short lifecycles, high dispersal rates, and efficient utilisation of environmental resources (Orians 1986). It is these traits – combined with a lack of native predators, pathogens, or competitors in the new habitat – that allow invasives to prosper in new habitats, often out-competing native species for resources and reducing the richness, evenness, and diversity of invaded communities (Harrod 2001, Collier et al. 2002, Hejda et al. 2009).

However, not all introductions turn into bio-catastrophes. Once into a novel environment, a species may be unsuccessful and disappear, or it may survive and continue to persist at a low, endemic level, achieving a “natural balance” maintained by local predators and parasitoids that adapt to the newcomer. The availability of an introduced pest in the new land of the hosts assists in the establishment of the insects and diseases that associate with them. This is the case of the successful establishment of the bark beetle *Tomicus piniperda* Linnaeus into North America onto populations of its natural host, the Scotch pine (*Pinus sylvestris* Linnaeus) (Niemela and Mattson 1996), which is itself an introduced species.

The spread of invasives appears to be increasing globally. This is most likely due to increased travel and globalisation of trade, which facilitates the movement of alien species to new habitats – whether intentionally (e.g., ornamental plants, domestic pets) or unintentionally (e.g., insect pests in wood packaging). The increasing extent of disturbances caused by changing land-use patterns (e.g., the exploiting of natural forests, creation of roads, and conversion of forests to agricultural land) also plays a role in the vulnerability of ecosystems to invasion. These disturbances can create a sudden abundance of resources (e.g., light, soil nutrients, and space), remove native species, and facilitate the movement of exotics (Parendes and Jones 2000, Lugo and Helmer 2004,

### Box 7.1 Lessons from a *Dothistroma* needle blight epidemic in northwestern British Columbia, Canada

Alex Woods

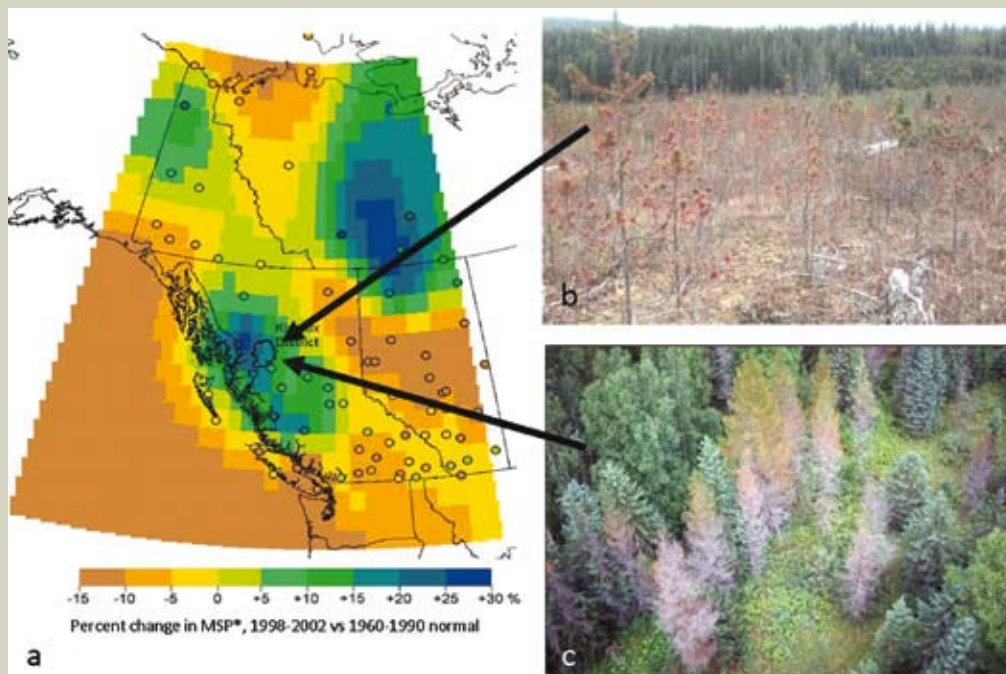
*Dothistroma* needle blight caused by the fungus *Dothistroma septosporum* (Dorog) Morelet is a major pest of pine plantations in the southern hemisphere, where both the host and the pathogen have been introduced (Gibson 1974). In northern temperate forests, where the pest and host trees are native, damage levels have historically been low. However, *Dothistroma* needle blight is currently causing extensive defoliation and mortality in plantations of lodgepole pine in northwestern British Columbia (BC), Canada. The severity of this epidemic is such that mature lodgepole pine trees in the area are dying, which is an unprecedented occurrence (Woods et al. 2005).

The epidemiology of *Dothistroma* needle blight has been well studied (Gadgil 1967, Peterson 1967), and the environmental conditions that favour the disease are well known. Elevated summer precipitation combined with increased overnight minimum temperatures and warm rain events have been associated with past outbreaks of *Dothistroma* needle blight (Peterson 1973, Gibson 1974), and have been clearly associated with the current epidemic in BC (Woods et al. 2005).

Climate change studies that have considered pathogens tend to look at how the impacts of

drought and associated stress on host organisms (plants) have affected fungal pathogens (Brasier 1996, Lonsdale and Gibbs 1996). Drought stress has not been a factor in northwestern BC, and temperature and moisture conditions for lodgepole pine populations in the northwest should have become more favourable, resulting in possibly improved growth performance (Rehfeldt et al. 1999, Nigh et al. 2004). However, the increase in summer precipitation has instead favoured the development of a foliar pathogen that has far outweighed any benefits the improved environmental conditions may have had on tree growth.

The current *Dothistroma* needle blight epidemic in northwestern British Columbia is not an example of an ecosystem out of control. The natural ecosystem in this region is diverse, resilient, and adaptable, and is possibly as well-equipped as any forest ecosystem to cope with the effects of climate change. For the past three decades, however, forest managers in northwestern BC have attempted to grow a managed forest dominated by two species: lodgepole pine, *Pinus contorta* Dougl. and spruce, *Picea glauca* (Moench) Voss (Woods 2003). They did so because they sought to improve upon both the timber productivity and the lack of management predictability associated with the naturally diverse forests of the region (Woods 2003). The *Dothis-*



**Figure 7.1** Map of British Columbia, Canada showing (a) percent increase in mean summer precipitation during the period of the current epidemic initiation (1998–2002) over the climate normal of 1961–1990, and (b) photographic examples of severe *Dothistroma* needle blight infestations showing failed plantations in the Kispiox District, and (c) dying mature lodgepole pine trees. \*MSP = mean summer precipitation. (Woods et al. 2005).



troma needle blight epidemic has demonstrated how such forest management goals can be upset by even minor changes in climate and the associated effects on forest health. The increase in summer precipitation that has occurred is not extreme, but it has, perhaps, been enough to pass a biological threshold that had previously kept *Dothistroma* needle blight in check. This disease epidemic illustrates our inability to control what was believed to be a predictable forest management model. It is exactly this lack of control in light of climate change that we must learn from. Such overly simplistic management intentions have shaped landscapes and ecosystems for human values around the world, often leading to similar unforeseen consequences (Kareiva et al. 2007).

We must now recognise that through global warming, our human footprint has been firmly set around the world (Hansen et al. 2005). The *Dothistroma* needle blight epidemic in northwestern BC illustrates how even a small change in the weather of a resilient natural territory can disrupt forest management plans, particularly if those plans have failed to integrate the inherent natural wealth of the area. Forest management must embrace the new reality of uncertainty associated with climate change and its direct and indirect effects. When faced with such uncertainty, the best option is to build on the natural diversity that already exists within forests and facilitate the migration of species that may be better adapted to cope with the future (Millar et al. 2007).

McDonald et al. 2008).

Climate change may play an important role in facilitating species invasions by creating opportunities for invasive species to become established in new areas as they become climatically suitable, and by stressing some native plant species, thus making them more susceptible to pathogen infection, insect attack, and vegetative competition. Thus, given current global trends, invasive species should be considered an important component of global change scenarios, guiding forest and pest management plans – as their potential to impact forest ecosystems is significant.

### 7.2.3 Pollutants and Forest Health

Atmospheric and soil pollution are recognised as hazards to the health of trees in both urban and forest environments. These human-caused disturbances affect forests at a scale that is far from natural (Bormann 1985, Mattson, 1996). Because they often occur long after an ecosystem has become established, the system lacks the resiliency needed to adapt to these disturbances. Increased pollutants in the environment may cause severe shifts in the established balance of pest-host relationships; the ecosystem may not be able to adjust to the rapidity or severity of these shifts. Pollutants affect CO<sub>2</sub> concentrations in the atmosphere, nitrogen cycles in the soils, stratospheric ozone, and ambient temperatures, all of which reduce tree vigour and increases their susceptibility to insect and diseases. Haack (1996) reported evidence of increased activity of lepidopteran larvae, gypsy moths, borers, and phloem-feeding beetles in oak-hickory forests that corresponded to increases in

soil acidity purportedly due to pollutants. The topic of pollutants and forest health is treated in a separate chapter of this book.

## 7.3 Monitoring Forest Health in a Changing Environment

Parties to international agreements, such as the Montréal Process (Montréal Process 2009), have agreed upon a set of criteria and indicators for the conservation and sustainable management of forests. Criterion 3 of the Montréal Process recognises that maintenance of forest health and vitality is the foundation of sustainable forest management, and establishes indicators for the measurement of this criterion. The indicators are the Area and Percent of forest affected by biotic and abiotic processes and agents beyond reference conditions. Signatory countries have agreed on timely reporting of the state of their forests using these criteria. Some of the challenges in applying these criteria are discussed below.

Long-term records are required to determine if the agents are operating beyond the reference conditions, i.e., beyond the range of historic variation. Past history serves as a useful baseline to help in measuring alterations in tree and pest distributions that expose forests to new or enhanced pressures from pests.

Direct methods of forest health assessment are based on quantifiable measures, such as levels of tree mortality, crown vigour (defoliation, discoloration, leaf necrosis, crown density, foliage transparency), and tree growth (tree rings, size of basal area, shoot growth or dieback) (Hall 1995, Campbell and Liegel 1996, Wermann 1997). These direct methods will

**Box 7.2 Climate change effects on the decline of yellow-cedar***Paul Hennon*

Yellow-cedar (*Chamaecyparis nootkatensis* (D. Don) Spach) is a culturally and economically valuable tree that has been dying in over 200 000 ha of pristine forests for the past 100 years in southeast Alaska, USA, and nearby British Columbia, Canada (Hennon et al. 2008). Over 20 years of research has revealed a condition termed “yellow-cedar decline” as a leading example of widespread tree mortality caused by climate change. Yellow cedar decline is characterised by trees experiencing slow decline in vigour over several years, ultimately resulting in the death of the tree.

Paleoecology and dendrochronology indicate that yellow-cedar probably survived the Pleistocene and early Holocene in small Alaskan refugia and began active migration about 4000 years ago when the climate shifted to a cool and wet regime, and increased its abundance at lower elevations during the Little Ice Age (Beier et al. 2008, Hennon et al. 2008). Reconstruction of yellow-cedar population changes is possible because dead trees remain standing for about 100 years. From these reconstructions, it has been established that the onset of yellow-cedar decline began in about 1880 (Hennon et al. 1990) – a date that roughly coincides with the end of the Little Ice Age.

After evaluating a number of abiotic and biotic risk factors associated with yellow-cedar decline, a working hypothesis was developed around a cascading complex of landscape- and site-specific factors, and to a physiological vulnerability of yellow-cedar: the susceptibility of its fine roots to spring freezing injury (Schaberg et al. 2008).

Long-term climate or short-term weather events influence each of the landscape and site factors that predispose yellow-cedar to this freezing injury. The cool, moist climate that developed in the late Holocene created bog and forested wetland conditions that favoured the abundance of yellow-cedar, but also forced shallow rooting (D’Amore and Hennon 2006, D’Amore et al. 2009). The open canopy condition on boggy soils makes these open forests more susceptible to more extreme microclimates, in particular to rapid warming, which triggers cedar dehardening in late winter (Schaberg et al. 2005) and to cold temperature penetration into soils during cold weather in spring (D’Amore and Hennon 2006). Research on cold tolerance of yellow-cedar roots in early spring demonstrated that soil temperatures below  $-5^{\circ}\text{C}$  are lethal (Schaberg et al. 2008).

The presence of snow buffers soil temperatures, disrupting the progression of events leading to tree



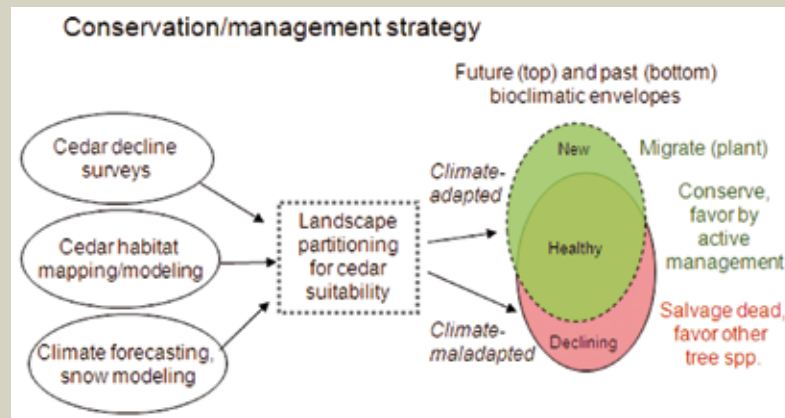
**Photo 7.3 Intensive yellow-cedar decline in Alaska.**

injury. Analysis of weather station data confirms a trend towards warmer winters and reduced snow, but persistent early spring freezing events throughout the 1900s; these are all conditions necessary for yellow-cedar decline (Beier et al. 2008). Comparing snow models to the distribution of yellow-cedar decline further illustrates the controlling influence of snow in the health of yellow-cedar forests (Hennon et al. 2008). Yellow-cedar is healthy where snow persists past the last cold period in spring, or where yellow-cedar is deep-rooted on better drained soils.

Recommendations for the conservation or active management of yellow-cedar are based on current and future habitat suitability of this tree in the context of its vulnerability to climate conditions. Partitioning of the landscape into suitable and unsuitable areas is based on information from yellow-cedar decline surveys, yellow-cedar habitat modelling from inventory plots, and climate and snow forecasting (Hennon et al. 2008). This helps to expose key habitat features in which yellow-cedar is dying, stable, and regenerating – an approach consistent with the shifting bioclimatic envelope concept (Hamann and Wang 2006).

Planting or thinning is often needed to ensure the initial regeneration and competitive status of





**Figure 7.2** The conservation and management strategy for yellow-cedar partitions in the landscape of coastal Alaska into areas that are unsuitable and are, or will be, suitable as the climate warms (modified from Hennon et al. 2008).

yellow-cedar; thus, there is a role for active management (Hennon et al. 2009). These activities are directed at higher elevation or on well-drained soils where snow or deeper rooting, respectively, protects yellow-cedar roots from lethal cold temperatures. Recent studies have also demonstrated that dead cedar forests represent a surprisingly valuable wood resource for salvage recovery (Hennon et al. 2007) because the tree's unique heartwood chemistry retains wood properties for up to a century after death (Kelsey et al. 2005). Salvage logging of dead cedar where it is now maladapted to climate can relieve pressure from other areas more suitable for long term conservation of yellow-cedar.

Yellow-cedar decline highlights the paradoxes

and complexities that might be expected in future climate change-forest scenarios. The paradox of a freezing injury triggered by climate warming probably could not have been predicted. The complex interaction of climate, landscape, and site factors, as well as cedar's unique physiological vulnerability to freezing, required time for research to unravel. New information on the autecology of yellow-cedar helped uncover its unique physiological vulnerability. Once the cause was finally elucidated, however, this new knowledge was coupled with recent advances in Geographic Information System (GIS) analysis and climate projections to provide managers with guidance on how to adapt cedar management to a changing climate.

continue to be useful for monitoring forest health as climate pressure mounts.

Indirect methods of forest health assessment have been used in the past. Abundance of canopy invertebrates is higher in mature forests that have not been disturbed by humans (Schowalter 1995), which makes them useful bioindicators of forest health. Hilszczanski (1997) used the diversity of beetles from different trophic groups as indicators of long-term influences of insecticide treatments on forest ecosystems. Populations of many songbirds are strongly influenced by local forest structure, which affects micro-habitat and availability of food and nesting sites. Therefore, the presence or absence of certain bird species can reflect forest conditions (Campbell and Liegel 1996). These indicators are specific to the successional stage of the forest as determined by their particular disturbance regime. In a changing environment, the value of these indirect indicators of forest health will need to be demon-

strated, as the populations of the biological indicators will respond in unpredictable ways to changes in any of the components of their respective food webs.

Assessment of the complex symptoms of forest decline caused by air pollution requires both direct and indirect methods, which are often sophisticated and intensive (Wermann 1997). For example, Butorina et al. (1997) monitored the state of Russian and Ukrainian forests that have been subjected to radiation emissions by studying the cytogenetics of trees and assessing chromosome and mitosis abnormalities. Often, a particular stress causing no or little apparent damage will predispose the forest to be more susceptible to attack by insects and pathogens, or other kinds of environmental damage. Poor forest health in Alaska, including the decline of the Alaska yellow-cedar (*Chamaecyparis nootkatensis*), has been attributed to a combination of climate change and natural susceptibility to root-freezing.

### Box 7.3 Forest management, climate change, and feedbacks: the mountain pine beetle in western North America

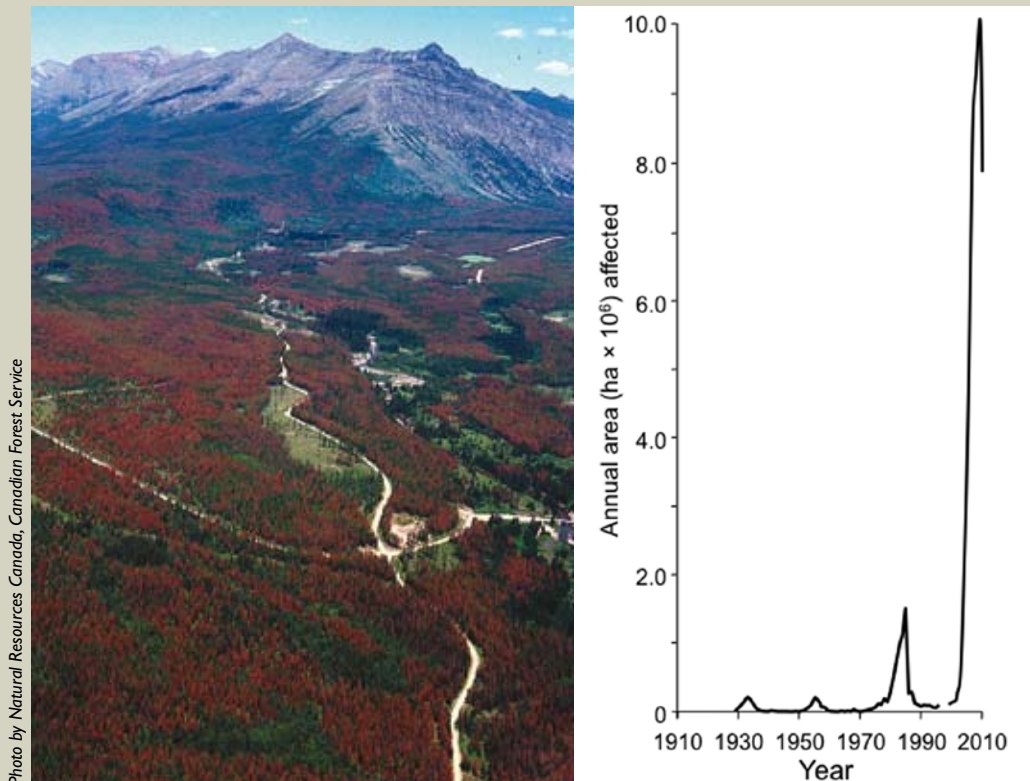
Allan Carroll

The mountain pine beetle is native to the pine forests of western North America, where it normally exists at very low densities, infesting only weakened or damaged trees. Under conditions conducive to survival, populations may erupt and spread over extensive landscapes, causing the mortality of large numbers of healthy trees (Safranyik and Carroll 2006). While there have been four significant eruptions by mountain pine beetle during the past century in western North America (Taylor et al. 2006), the ongoing epidemic is unprecedented in its size and severity – causing the mortality of mature pines over a cumulative area of 14 million hectares in western Canada (Photo 7.4), and an additional ~4 million ha in the USA (Cain 2009).

For a mountain pine beetle outbreak to occur, there must be an abundance of large, mature pine trees (the beetle's preferred resource) combined with several years of favourable weather for beetle survival (Safranyik and Carroll 2006). Evidence indicates that, as a result of anthropogenic modifications to the environment, these conditions have coincided over an exceptionally large proportion of

the pine forests of western North America in recent decades. Taylor and Carroll (2004) showed that due to fire suppression and selective harvesting (for species other than pine) during the latter half of the previous century, there was more than three times the amount of mature pine in western Canada at the start of the current outbreak, when compared with 100 years earlier. Furthermore, Carroll et al. (2004) demonstrated that as a result of climate change, conditions relevant to mountain pine beetle survival have improved over much of western Canada during recent decades, allowing populations to invade formerly climatically unsuitable pine forests. Thus, both conditions for an outbreak have coincided with sufficient magnitude to cause the largest outbreak in recorded history.

The current mountain pine beetle epidemic has resulted in extensive direct ecological and economic impacts, affecting wildlife habitat (Bunnell et al. 2004), water quantity/quality (Walker and Sydneysmith 2008), recreational values (Sheppard and Picard 2006), and timber quantity (Pedersen 2004) and quality (Byrne et al. 2006, Watson 2006). Recent evidence also indicates that the widespread tree mortality inflicted by the mountain pine bee-



**Photo 7.4** Annual area affected by the mountain pine beetle (a), and tree mortality associated with the current outbreak in British Columbia, Canada (b). Data courtesy of the Canadian Forest Service, Forest Insect and Disease Survey and the British Columbia Ministry of Forests and Range.

tle may also have significant longer-term impacts to regional carbon dynamics. Indeed, the loss of carbon uptake and the increased emissions from decaying trees as a result of the mountain pine beetle epidemic has converted the forests of western Canada from a small net carbon sink into a large net carbon source to the atmosphere (Kurz et al. 2008). In fact, during 2009, it is anticipated that carbon emissions to the atmosphere will be similar in magnitude to the average annual direct emissions from forest fires for all of Canada. Moreover, through reductions to net primary productivity and increases in heterotrophic respiration, beetle im-

pacts have negated any potential gains in carbon sequestration associated with climate warming, elevated atmospheric CO<sub>2</sub> concentrations, and nitrogen deposition (Kurz et al. 2008). Exacerbation of insect outbreaks through anthropogenic intrusions, such as has occurred with the mountain pine beetle, could undermine the ability of northern forests to take up and store atmospheric carbon. Resultant increases in atmospheric carbon could further increase warming and accompanying insect outbreaks thereby leading to positive feedbacks to the global climate system.

## 7.4 Management of Insects and Diseases in a Changing Environment

Forest disturbances are characterised by a number of parameters, which include disturbance size, intensity, and recurrence rate (Kohm and Franklin 1997). When assessing the significance of a particular forest insect or disease outbreak, it is of critical importance to compare the magnitude of the disturbance in all its parameters with the historical pattern, particularly in terms of extent, intensity, and frequency. If the disturbance pattern is outside the historical range of variation, there is a probability that forest health may be declining and corrective management action may be needed. For example, Canadian boreal ecosystems are healthy when fire and spruce budworm, the two most common disturbances in boreal forests, recur at their natural average intervals, magnitude, and size. This, of course, does not mean that economic loss caused by these disturbances is not important.

Many forest insects and disease outbreaks play a natural role in forest succession by selectively killing or retarding the growth of certain tree species while leaving others untouched (Castello et al. 1995). For example, in some ecosystems of the North American Northwest, the mountain pine beetle, *Dendroctonus ponderosae* Hopkins, by selectively killing mature and overmature lodgepole pine, hastens the succession from fire-origin lodgepole pine to more shade tolerant species, such as Douglas-fir (Heath and Alfaro 1990) or, in combination with fire disturbance, it perpetuates lodgepole pine in a seral stage (Axelson et al. 2009). These outbreaks, while creating substantial economic loss to industry and society, when operating within the natural range of variation for this disturbance, do not necessarily reduce forest health. In this case, the beetle is an agent of transformation and, by killing off the mature, stagnant overstorey,

allows more light and nutrients to reach the understorey species, which then exhibit accelerated growth and occupy the free space.

Management of native insects and diseases should concentrate on restoring ecosystem balance and resiliency. Understanding how the pest organism affects the forest can be used to develop management systems that prevent losses by anticipating which trees are more susceptible to the pest organism. In the case of the mountain pine beetle, economic losses can be reduced and forest susceptibility lowered by selective removal of the susceptible hosts following a hazard rating plan in which susceptible areas are delineated (Shore and Safranyik 1992). Enhancement of tree vigour through thinning and fertilisation is known to increase tree resistance to beetle attack.

Managing native pests in a changing environment requires anticipating their interaction with their hosts in new environments. Many pests depend on a close synchrony between the phenology of the host and the lifecycle of the pest. The spruce budworms, *Choristoneura* spp., for example, require a precise synchrony between bud development and the end of the first instar diapause. If, due to favourable climatic conditions for the insect, there is an early ending of the first instar diapause, the young larvae will perish while attempting to penetrate undeveloped buds. Gray (2008) modelled the future distribution of the spruce budworm in Canada under different future climate scenarios.

Because of the several tactics that are available, management of native defoliators requires an integrated approach. Forests should be intervened to reduce the susceptible host (to mimic natural processes, prescribed burning may be recommended). Pheromone monitoring and hazard rating should be practised to warn of impending outbreaks and to allow orderly salvage of timber at risk. Direct control through application of biocontrol agents (the use of parasitoids, bacteria, viruses, or fungi) are now rou-



**Box 7.4 New forest epidemics in northern Europe**

*Jarkko Hantula, Michael Müller and Arja Lilja*

Several forest diseases have been newly discovered during the 20th century as they have rapidly reached epidemic levels in northern Europe. Here we discuss the case of *Dothistroma septosporum* (Dorog) Morelet on pines, and of *Chalara fraxinea* Kowalski on ash trees.

The fungus *Dothistroma septosporum* causes a needle disease in many species of pine trees worldwide. During the 20th century in northern Europe, this fungus caused sometimes considerable damage to many pines, especially to *Pinus nigra* J.F. Arnold. Another European species, *Pinus sylvestris* Linnaeus, was usually free of symptoms, even when growing next to heavily infested *P. nigra*.

Historically, the disease has been absent in Finland and Estonia, but in 2006, the long time balance between pine trees and *D. septosporum* in Europe was suddenly broken, and this fungus was first observed in Estonia (Hanso and Drenkhan 2008) on both *P. nigra* and *P. sylvestris*. Initially, the disease had a sporadic distribution area, as it was observed in only a few locations. However, by summer 2007, the disease had spread rapidly over the whole of Estonia, becoming an epidemic. By 2008, *D. septosporum* was found for the first time in Finland (Müller et al. 2009), where *P. sylvestris* is the dominating pine species, becoming common, especially in the central region of the country. By summer 2009, the situation was similar or the level of the epidemic was slightly decreasing.

*D. septosporum* has spread rapidly through northern Europe during the last few years, geographically and among hosts. Previously almost-resistant *P. sylvestris* trees are now frequently carrying the disease not only in Estonia and Finland, but also in some central European countries, including the Czech Republic.

A new disease also began to affect ash trees (*Fraxinus excelsior* Linnaeus) in the 1990s in Poland and Lithuania (Juodvalkis and Vasiliauskas 2002, Przybył 2002). After the first observations, this disease has rapidly moved towards the rest of Europe (Cech 2006, Schumacher et al. 2007, Thomsen et al. 2007, Halmschlager and Kirisits 2008), reaching Finland and Norway in 2007. The symptoms of this disease are highly variable, but in the end most ash trees die.

This new disease has recently been shown to be caused by the fungal pathogen, *Chalara fraxinea* (Kowalski 2006, Kowalski and Holdenrieder 2008), which was not known before the epidemic began. The spreading nature of the disease from Eastern Europe in all directions where ash trees grow, suggested that the disease would have been caused by an alien fungus introduced recently to Europe. However, this conclusion was contradictory to the observation that the teleomorph of *C. fraxinea* actually is *Hymenoscyphus albidus* (Rob. ex. Desm.) W. Phillips, a fungus known to have already occurred in Europe centuries ago (Kowalski and Holdenrieder 2009).

Could there be a common explanation for the observations described above? We hypothesise that the increased virulence of these diseases is caused by changes in plant-pathogen interactions, which could be due to two possibilities: (i) an environmental change, or (ii) some kind of genetic change in the fungus.

Causes of environmental change could include climate warming or changes in forest management. Changes in forest management can be ruled out since none have occurred throughout northern Europe in the last decades. While the weather in Europe has rapidly warmed during the last few decades, attributing the spread of these diseases to climate change seems premature. Global warming would nicely explain the spread of both pathogens toward the north. However, it seems unlikely that warming would also explain the host range changes of *D. septosporum*, or the spread of ash decline towards the south.

Possible genetic changes could include: pathogenicity-affecting mutations, introduction of new or more pathogenic races to Europe, or even fungal hybridisations; but there is no hard evidence supporting any of these possibilities.

As a conclusion, the increased pathogenicity of *D. septosporum* and *C. fraxinea* in northernmost Europe highlights the difficulties in understanding the causes of pathogen spread or altered behaviour. Both diseases were initially associated with climate change, but since new data has appeared, this view has turned out to be too simplistic. It may turn out to be that the changes result from several unrelated causes, of which warmer weather is only one.

tinely recommended when available (e.g., Shamoun et al. 2003). Similarly, in Europe, the incidence root and butt pathogen, *Heterobasidion annosum*, which increased with the adoption of summer harvesting in northern conifer forests, was addressed by developing a biocontrol based on a competing fungus, *Phlebiopsis gigantea* (Fr.) J (Risbeth 1963, Korhonen et al. 1994, Pratt et al. 2000).

An Integrated Pest Management System is recommended for the management of the white pine weevil (*Pissodes strobi* Peck), a severe pest of reforestation (Alfaro et al. 1995). However, the tactics depend on the weevil hazard rating of the area, which is based on a determination of the potential degree-day accumulation for the site. Other strategies included in this approach aim at restoring ecosystem balance by reducing the conditions that lead to the development of an outbreak. The system recommends treatments that diminish heat accumulation in the stand by encouraging growth of suitable non-host conifers as well as an overstorey of deciduous species, such as aspen or alder. These conditions may also enhance natural enemy populations (Altieri 1994).

In anticipation of future climate change, forests will have a higher chance of remaining healthy if they have high genetic variability so that adaptation to a changing environment is more likely. Thus, managing the ecosystems of the future requires proactive conservation of genetic resources to secure a sound genetic base for pest resistance. Host genetic resistance is useful in the management of native insects and diseases because it can be easily integrated with other control methods, particularly those based on silviculture (Hunt 1983a, b; Alfaro et al. 2008, King and Alfaro 2009). Kemp and Palmberg-Lerche (1994) recommended conserving the widest possible range of provenances, giving particular emphasis to sampling the extremes of the species distribution.

#### 7.4.1 Management of Introduced Pests

The introduction of non-indigenous species can result in catastrophic changes to forests because local trees lack naturally co-evolved defences to the invading organism, and because ecosystems do not have the biocontrollers (parasitoids and predators) that keep them in check in their native lands. This applies to species that are introduced from other continents as well as species within a country that expand their range due to changing environmental conditions or human activity.

Because of mounting criticism of the use of chemicals in pest control, resulting in increased public concern and the banning of pesticides, there is renewed interest in classical biological control, i.e., the importation of the parasites and predators that

regulate the pest in its place of origin. The intent is to increase ecosystem resiliency rather than eradication of the pest (Clausen 1978, Caltagirone 1981). The literature has many examples of biological control programs against invasive pests. Selected examples include the case of the pine shoot moth, *Rhyacionia buoliana* Schiff. in Chile, (Lanfranco 1994), the biocontrol attempts on Reunion Island against invasion by exotic weeds, and the attempts by Asian countries to control the introduced *Leucaena psyllid* (Banpot 1995).

The best option for controlling potential damage by invasive species is prevention. In 2009, the Food and Agricultural Organization of the United Nations (FAO) -based Interim Commission on Phytosanitary Measures revised a global standard for the regulation of wood packaging material in international trade. Under these measures, the international community has agreed on a variety of practices and regulations aimed at preventing the accidental transport of foreign species into new habitats (FAO 2009). Chile has proposed a novel approach (Lanfranco et al. 2002) in which purchasers of forest goods from Chile are made aware of the potentially invasive pests that could hitch a ride in their exported commodities. This “coming out clean” approach makes the prevention of insect and disease introduction *a two-way street* in which the exporter and importer take responsibility to prevent the spread of invasive species.

Once established, the eradication of an alien species is often difficult. Agencies in charge of eradication often enter into jurisdiction and public opinion battles that make effective eradication difficult. Nealis (2009) analysed the case of the introduced gypsy moth *Lymantria dispar* Linnaeus in Canada, and concluded that for effective eradication there is a need for improved dialogue between scientists and the public, including effective communication about the risk posed by the invading organism if it is not eradicated.

However, in a changing environment, existing bio-control programs, which have been successful in keeping introduced pests in check, may need to be reconsidered under the conditions created by future climate change. New climatic conditions and ecosystem migration will alter the balance of all trophic levels associated with pest-host systems.

Increased international trade will inevitably result in increased risk of introductions of foreign organisms. Some of these may not be considered pests in their place of origin, but may attain pest status in a different environment. Introductions can be regulated at the level of the organism, or restrictions can be imposed upon the vector or the host, or the means by which the introduction might occur. Since trade goods cannot always remain in quarantine while risk assessments are made, admission decisions are often made with little biological knowledge of the potential

### Box 7.5 Biological control of the invasive pine shoot moth, *Rhyaciona buoliana*, in changing Chilean environments

Dolly Lanfranco and Mar Ramos

*Rhyacionia buoliana* (Schiff.) was the first serious exotic forest pest to affect Chile's extensive *Pinus radiata* D. Don. plantations. Right from the introduction, it presented densities, dispersions, and damage never seen before in other parts of the world (Cerdeña et al. 1985, Ide and Lanfranco 1996a, b). Within a few years, the pest was established in the entire distribution area of pine in the central-south regions of Chile, affecting approximately 2 million ha of plantations, and triggering pest management actions that included within-country quarantine areas, and chemical and biological control. Given the ineffectiveness and harmful effects of chemical control, the biological control option was favoured, which focused on the introduction of natural insect enemies. Basic and applied research was conducted to mass-rear and release *Orgilus obscurator* (Nees) (Hym.: Braconidae), a parasitoid of *R. buoliana* native to Europe. The objective was to use the biological control agent along with silviculture measures to minimise losses in height, diameter, and volume by decreasing pest population densities and levels of infestation. However, after 20 years, it is clear that the biological control is currently not functioning well in all the regions where it was implemented (Table 7.1).

In the areas with major concentration of pine plantations (Bío Bío Region) and to the north, levels of control are significantly lower than in areas to the south, particularly on poor sites. In spite of the large amount of mated females released and parasitised larvae inoculated in northern regions, the control percentages are not always acceptable (Table 7.1).

This is not the case south of the Bio Bio Region, in La Araucanía and Los Ríos, where control was so effective that presently it is difficult to find the pest. Because of a marked north-south gradient of climatic conditions in the areas planted with pine, with the northern areas hot and arid, and temperate and humid climatic conditions in the south, we hypothesise that *O. obscurator* response may not be the same under changing environmental conditions. This could be due to climate-related variation in the multi-trophic interaction between tree, pest, the biocontrol agent, and the natural enemies of the biocontroller. One factor that could influence the success of *O. obscurator* as a biocontroller is the fact that, since introduction, several native parasitoids are also attacking *R. buoliana*, and hyperparasitoids are parasitising *O. obscurator*. These include *Perilampus tristis* (Mayr) (Hym.: Perilampidae) and *Temelucha* sp. (Foerster) (Hym.: Ichneumonidae). *P. tristis* is an exotic parasitoid, whose date of introduction to Chile is unknown and its function in the parasitoid complex is unclear. *Temelucha* sp. an endoparasitoid koinobiont of *Rhyacionia buoliana*, is the main competitive species of *O. obscurator*. Additionally, records of superparasitism and multiparasitism have been detected in both species. *Temelucha* was detected from the start of the biological control effort in Chile, but only in 2000 was the interference with the biocontroller noted. (Lanfranco et al. 1994, Lanfranco et al. 1998, Ide et al. 2007).

Impending climate change will have a profound effect in Chile, causing the migration of Chilean ecosystems in manners that have not yet been anticipated. Likely, northern areas will become unsuitable for pine plantations due to increases in aridity, while

**Table 7.1 Effectiveness of biological control of *Rhyacionia buoliana* by *Orgilus obscurator* in a north-south gradient of ecoclimatic regions of Chile. % indicates the percent of *R. buoliana* parasitised by *O. obscurator* in the first (1998) and last (2006) year of survey.**

Region	Latitude range	Mean temp. °C	Annual precipitation (mm)	1998 %	2006 %
South					
O'Higgins	33°52'–34°58'	14.7	562.8	0	56
Maule	34°58'–36°32'	14.8	942.8	1.5	68
Bío Bío	36°32'–38°29'	12.4	1008.2	3.38	42
Araucanía	38°29'–39°17'	12.0	1624.8	38.89	90
Los Ríos	39°17'–40°40'	11.8	2348.7	50.75	Host no longer detectable
Difference	6 °48'	2.9	1785.9		

Sources: Lanfranco and Ide 1998, Lanfranco and Ide 2000, CONAF 2006.



the best pine-growing areas will move further to the south. Insects respond quickly to changes in climatic conditions. Primary pests, such as *R. buoliana*, will most likely follow the future distribution of their hosts (Hance et al. 2007). However, it is harder to predict the response of members of other trophic levels, such as competitive parasitoids and hyper-

parasitoids, which will have more unpredictable variations in response to changes in climate. Thus, climate change may create conditions in which the present biocontrol system based on *O. obscurator* may lose its efficacy in areas where presently it is performing well.

dangers. Hence the approach of “guilty until proven innocent” has been adopted. Ruesink et al. (1995) recommend that evaluators should include ecologists, evolutionary biologists, and conservationists to better assess community consequences, possible genetic change (host shifts), and threatened species.

## 7.5 Decision Support Systems for Forest Health in a Changing Environment

Decision makers need Decision Support Systems (DSS) to forecast the risk of ecosystem change under varying scenarios of future climate conditions. By necessity, such systems will be complex because they need to operate at different spatial scales (from stand to landscape) and at temporal scales of many decades. The ultimate goal would be to forecast the social and economic cost to society of increased pest activity in terms of impacts on timber supply, as well as in terms of the value of the environmental services expected from the forests of the future. In other words, a decision maker would like to know what will be the social and economic costs of maintaining a healthy, well-adapted forest under diverse climate change scenarios. A well-adapted forest is one that continues to produce goods and services in a changing climate. The information needs for DSS in a changing environment are quite large. There is a need to increase our understanding of what will be the expected changes in forest vulnerability to insects and diseases, as well as how the expected changes will affect the biology of pest organisms in terms of disturbance duration and intensity. Pest disturbance is one of the main drivers of ecosystem change. There are several components or stages that such DSS systems should have in order to be useful to policy makers:

- 1) *Forecasting the risk of ecosystem change under future climate scenarios.* This requires analytical systems capable of anticipating the risk of ecosystem change in terms of potential for ecosystem migration, changes in species composition, and potential forest health problems in the new environment. Each risk category is prioritised and the likelihood of each risk occurring evaluated. Hamann and Wang (2006) developed an ecosystem-based climate envelope modelling approach to assess potential climate change impacts on forest ecosystems and tree species. The system predicts current and future niches for various tree species of British Columbia, Canada. Using this model, Hamann and Wang (2006) predict that the most important species of British Columbia will lose significant proportions of their suitable habitat. Murdock and Flower (2009) projected the future geographic range of some of the conifer species of British Columbia under the various climate scenarios and overlapped those ranges with climatic suitability maps for the most significant insect pests affecting these species. In this manner, Murdock and Flower (2009) could identify areas of the future ranges for the tree species that were at high or low risk of pest infestation.
- 2) *Establishing the desired condition of future landscapes.* After stage 1, there is a need to decide what composition the future landscape should have or is capable of having in a changing environment. This is a goal-setting stage where the timber and non-timber products and ecosystem services desired from the changing landscape are set.
- 3) *Simulating the effects of alternative forest and pest management options* and their effects on future landscape characteristics, and comparison with the goals set for the particular landscape. Forest health management options for controlling new insect pests and diseases need to be analysed for their social, economic, and environmental costs and benefits. This part of the analysis will be recurrent because decision makers will need to balance the social and economic cost-benefit of each action.

## 7.6 Conclusions: New Paradigms Needed to Maintain Healthy Forests in a Changing Environment

Maintenance of a healthy forest is intimately connected with the concept of sustainable forest management (SFM). SFM must be viewed as an ecological approach to achieve multiple uses of forests by blending the needs of the people that derive their livelihood from the forest with the all-encompassing needs of world societies for environmental services, such as biodiversity conservation, water regulation, and carbon sequestration. Thus, SFM is meant to preserve biodiversity and to mimic natural processes so that ecosystem integrity is maintained at the landscape level. At the same time, it is meant to be responsive to the broad range of social values in a way that is equitable to both urban and rural resource-dependent communities.

SFM recognises that an ecosystem is healthy when all the natural disturbances that act on a forest ecosystem are operating within their historical range of variation. The challenge for SFM in a changing environment is to determine how future climatic conditions will affect the disturbance regimes of the landscape.

There are a number of initiatives that could help to maintain forest health under future changes of the world's climatic regimes:

### Managing for complexity

Faced with climate uncertainty, an extremely important role for forest decision makers will be to favour management approaches that encourage and sustain ecosystem complexity. Ecosystems that are variable in age and species structure, as well as genetically diverse, have the best chance of adapting to a changing climate.

Genetic resistance to pests is one of the favoured methods of pest control because it offers a durable form of pest management that does not rely on chemical pesticides. The large tree genetic improvement work completed in British Columbia, Canada, to find, propagate, and utilise tree resistance in pest management against the white pine weevil (summarised by King and Alfaro 2009) disclosed that resistant genotypes were found most commonly at the edge of the species distribution. Unfortunately, these areas are the most threatened by climate change. Therefore, in order to save potentially valuable genotypes for use in reforestation efforts to create forests adapted to future conditions, gene conservation programs should prioritise these areas.

### Role of governments and universities

Research efforts should be maintained and directed to mitigate future forest health problems in the new environments. A changing environment may require that new solutions be found for forest health problems that may presently be considered solved. The biological control of the pine shoot moth, *Rhyacionia buoliana* Schiff. in Chile is working effectively in some ecosystems. However, migration of these ecosystems may create imbalances in the parasitoid complex, which may destabilize the system.

### Risk analysis and decision support systems

The procedures of risk management, already implemented in industry (e.g., financial institutions) since the early 1970s (Crouhy et al. 2001), provide a useful framework for building Decision Support Systems to help decision makers maintain forests that are more adaptable to a changing environment. Such systems will need to identify the risks of change in various ecosystem components and build future scenarios assuming different likelihoods of risk occurrence. The consequences for society of each risk occurring will need to be estimated and evaluated considering economic and environmental values.

### Climate mitigation through forest health management

Large scale tree mortality, accompanied by severe forest degradation caused by pests (i.e., following an outbreak, land often becomes hard to regenerate, or regeneration may have poor health), becomes a driver of climate change because dead trees eventually release stored carbon into the atmosphere through decomposition, further aggravating the problem. Pest management, by reducing tree mortality and tree growth loss, can contribute to the mitigation of the effects of climate change. Maintaining a healthy forest helps ensure that CO<sub>2</sub> bound up in forests remains sequestered, a most important environmental service from forests in an era of global climate change.

### Helping forests to adapt

Forest adaptation to future conditions will occur naturally as major forest species are driven from their current geographic distribution and gradually settle into their new habitats as the habitats become suitable (e.g., they meet the degree-day requirements for survival). This process of colonisation of new habitats has occurred in the past history of the planet and will result in other components of the ecosystems, including the pest organisms, moving along with their host trees.

The concept of assisted migration is being considered as a way of helping ecosystems to adapt (McLachlan et al. 2006). This implies determining the locations where future ecosystems will reside following gradual climate change and moving north

(in the northern hemisphere) plant species or varieties adapted to more southern conditions. It will be important to look ahead and determine what forest pest problems could move from old bio-ecological zones to the new areas. For example, as the boreal ecosystems become sub-boreal, will sub-boreal pests adapt to the boreal species?

### International efforts to control invasive species

Invasive species will continue to be a threat to forest health worldwide and an important driver of forest change. Prevention of forest destruction by invasive pests also has a role in climate change mitigation. Keeping forest pests with potential to cause forest degradation out of a country requires international agreements, regulations, and point-of-entry inspections.

Given the increased movement of goods in international trade and increased global development, the control of many forest pests and pathogens will prove to be a challenging balance between controlling and preventing the spread of these invasives without unnecessarily restricting trade or development. In order to minimise the risks and impacts of invasive species on forest health, forests should be managed and existing regulations enforced to prevent their introduction, establishment, and spread.

Effective management of invasives requires comprehensive knowledge about a wide range of organisms, their biology, ecology, distribution, their impacts on forest ecosystems, and possible methods of control (FAO 2009). Identifying the pathways by which these organisms spread is especially important since once established, eradication can be difficult due to the extent of the problem, associated expenses, and conflicts with public agencies over eradication methods (such as the use of pesticides) (e.g., Nealis 2009). There are many research needs aimed at the prevention of introduction and pest detection, as well as control options for organisms that slip through the cracks.

Forest invasives are an international problem, so a solution requires cooperation, knowledge-sharing, and coordination of pest management activities among countries (FAO 2009). This is especially relevant to developing countries, where there is currently less knowledge about invasive species and fewer resources available to address this problem. Independent bodies, such as the Technical Panel on Forest Quarantine and the International Forest Quarantine Research Group, are helping to address these gaps in the knowledge base by focusing research where it is most needed.

Similar efforts in international regulations will be required for effective reduction of pollution impacts across international boundaries, involving public and private sector organisations to secure enforceable agreements on emission reductions.

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