

Chapter 17

IPM: The Forest Context



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Integrated pest management (IPM) is perhaps best described as “...the maintenance of destructive agents, including insects, at tolerable levels by the planned use of a variety of preventative, suppressive or regulatory tactics that are ecologically and economically efficient and socially acceptable. It is implicit that the actions taken are fully integrated into the total resource management process in both planning and operation” (Waters 1974). Another useful definition of IPM is “an ecosystem-based strategy that focuses on long-term prevention of pests or their damage through a combination of techniques such as biological control, habitat manipulation, modification of cultural practices, and use of resistant varieties. Pesticides are used only after monitoring indicates they are needed according to established guidelines, and treatments are made with the goal of removing only the target organism. Pest control materials are selected and applied in a manner that minimizes risks to human health, beneficial and non-target organisms, and the environment” (University of California, Davis 2015). The spatial and temporal scale of forests demands landscape-level and long-term planning with an emphasis on preventive measures, e.g. silviculture. IPM programs in forests have historically been concerned mainly with pests that have large

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impacts on fibre and wood supply and the livelihood of resource-dependent communities, e.g. the spruce budworm, *Choristoneura fumiferana* (Clemens), and mountain pine beetle, *Dendroctonus ponderosae* Hopkins, in North America. However, in Europe and more recently in North America, there has been a shift away from a focus on individual pests and towards IPM as part of ecosystem management (Häusler and Scherer-Lorenzen 2001; Alfaro and Langor 2016). Ideally, IPM should be an integral part of sustainable forest management, which in addition to sustained forest productivity, includes principles such as maintenance of biodiversity and ecological processes, carbon sequestration, and protection of soil and water quality (Holvoet and Muys 2004).

17.1 Components of IPM

17.1.1 *Biology and Ecology of the Pest-Tree-Forest System*

A central component of IPM in forests is knowledge and understanding of the biology and ecology of the pests, their host trees, and the forest system in which they interact. Effective strategies for reducing the negative impact of an insect pest requires sufficient knowledge of the pest's life history and the factors that affect its population dynamics, such as host susceptibility and natural enemies. A key aim is to reduce pest impacts while minimizing negative effects on ecosystem services and function.

17.1.1.1 Systematics and Taxonomy

The first step in IPM is accurate identification of the causative pest(s) and that requires some knowledge of taxonomy, and not infrequently, the assistance of taxonomic specialists. The next step would be to determine what is known about the pest species' biology, and suitable methods for its survey and control. If the pest can be identified to genus only (e.g. it may be a non-native species accidentally introduced to a region) it may still be possible to determine some of its biology based on what is known of other species in the same genus (Huber and Langor 2004). Accurate identification can be difficult for species with only subtle morphological differences from other species. Misidentifications can be costly, as illustrated in Box 17.1.

Box 17.1 Importance of taxonomy and accurate species identification

Specimens of the brown spruce longhorn beetle, *Tetropium fuscum* (Fabr.), a European native, were collected in Point Pleasant Park, Halifax, Nova Scotia, Canada in 1990 during a trapping survey for spruce beetle, *Dendroctonus rufipennis* (Kirby), but were misidentified as the native species, *Tetropium*

cinnamopterum Kirby. It was not until 1999 that the causal organism was correctly identified as *T. fuscum* (Smith and Hurley 2000). A quarantine and eradication program was initiated in 2000 at an estimated cost of CAN\$4–6 million per year (Huber and Langor 2004). The goal of eradicating *T. fuscum* was abandoned in 2007 when it was clear that the beetle had established itself over a large area. Although it is quite possible that *T. fuscum* was already established in Nova Scotia several years before specimens were first collected in 1990, the delay of almost a decade in the accurate identification of *T. fuscum* likely made effective containment and eradication more difficult (Huber and Langor 2004) (see Chapter 19).

17.1.1.2 Pest Life History and Factors Affecting Pest Populations

Knowing a pest's life history is fundamental to developing effective survey and control methods. Furthermore, understanding the key factors that affect pest population biology makes it possible to develop tools and tactics that have less interference with natural mortality factors. For example, the discovery that nucleopolyhedrosis viruses cause the collapse of outbreaks of defoliators such as Douglas-fir tussock moth, *Orgyia pseudotsugata* (McDunnough), and balsam fir sawfly, *Neodiprion abietis* (Harris), has led to the mass production and application of species-specific viruses to suppress defoliator populations (Shepherd et al. 1984; Otvos et al. 1987; Lucarotti et al. 2007). Below, we briefly highlight some of the natural factors affecting pest distribution and abundance that must be considered when developing IPM programs. The myriad of interacting abiotic and biotic factors and their effects on insect populations is beyond the scope of this chapter. For more information, see Price et al. (2011) and Schowalter (2016).

Climate

Climate, especially temperature and precipitation patterns, has a substantial influence on the distribution of plants and the animals that feed on them (Merriam 1894). All insect pests have upper and lower temperature limits beyond which they do not survive, and these limits are useful for predicting their potential geographic distribution, e.g. an exotic species introduced to a new continent or a native pest expanding its range in a changing climate. Knowing a pest's distribution in the landscape and how it may vary in response to climate is a prerequisite for efficient targeting of IPM tactics. Warming temperatures in the last couple of decades have enabled range expansions of some species like the pine processionary moth, *Thaumetopoea pityocampa* Schiff. (Battisti et al. 2005), mountain pine beetle (Logan and Powell 2001; Carroll et al.

2003; Weed et al. 2013), and southern pine beetle, *Dendroctonus frontalis* Zimmermann (Dodds et al. 2018), and generated the need for temperature-based models to predict where range expansions may occur (e.g. Buffo et al. 2007; Lesk et al. 2017). Exposure to unseasonal cold temperature is often the largest single source of mortality in mountain pine beetle populations (Safranyik 1978). In Alberta, Canada, ground surveys are conducted every spring to estimate mountain pine beetle overwintering survival and forecast population trends, which are in turn used to focus management activities where they are most effective in slowing the beetle's spread (Anon. 2007b). In addition to overwintering survival, temperature affects the rate of development, voltinism (number of generations per year), dispersal, reproduction, and degree of phenological synchrony with their hosts (Hansen and Bentz 2003). Favourable temperatures during larval development can shift spruce beetle populations from a 2-year life cycle to a 1-year life cycle and contribute to large-scale outbreaks, whereas cold temperatures that occur before spruce beetles have acclimatized can contribute to outbreak collapse (Aukema et al. 2016). Knowledge of temperature-phenology relationships is useful in models for predicting the impact of pests (Powell and Bentz 2014) and the need for management actions.

Natural Enemies

Most insect herbivores serve as food or brood hosts for a large assortment of natural enemies (i.e. predators, parasitoids, and pathogens) which have been implicated as major mortality factors of about half of the pest species for which long-term population studies have been conducted (Price et al. 2011). For example, pupal mortality from parasitism (Fitzgerald 1995) and bird predation (Parry et al. 1997) contribute to population regulation of the forest tent caterpillar, *Malacosoma disstria* Hübner, and density-dependent pupal predation regulates low-density populations of the winter moth, *Operophtera brumata* L. (Varley and Gradwell 1968; Roland 1994). Baculoviruses infect many species of forest Lepidoptera and sawflies (Cory et al. 1997) and have been used to control defoliators like the balsam fir sawfly, *Neodiprion abietis* (Harris) (Moreau and Lucarotti 2007). Knowledge of a pest's natural enemies and their impacts on pest populations is beneficial when developing an IPM program, e.g. to reduce negative impacts when using insecticides (Williams et al. 2003) or for classical biological control of exotic, invasive forest pests (Bauer et al. 2015).

Host Tree-Insect Interactions and Food Quality

In addition to the top-down effects exerted on herbivore populations by natural enemies, the quality and availability of food (e.g. host trees) exert considerable bottom-up effects, and this is the basis of IPM tactics that affect tree vigour (e.g. thinning) and breeding for genetic resistance. Host resistance is one of the main factors regulating endemic populations of bark beetles (Aukema et al. 2016).

Factors that stress trees and reduce their vigour, such as root rots, overstocked growing conditions, drought, defoliation, or root damage from wind events, reduce host defenses and make trees more susceptible to colonization and mortality by bark beetles (Fettig et al. 2007; Kolb et al. 2016). When this occurs on a large scale it often leads to greater reproductive success and higher bark beetle populations (Werner et al. 2006; Aukema et al. 2016). When bark beetle populations reach the epidemic phase, they can overcome the defenses of healthy trees thanks to mass attack facilitated by aggregation pheromones (Wallin and Raffa 2004; Boone et al. 2011). Lodgepole pines, *Pinus contorta* Dougl. ex Loud. var. *latifolia* Englem., with thicker phloem offer more food and space for larval development and produce more mountain pine beetle brood than trees with thinner phloem (Amman 1972). Suppressed trees with smaller diameters are habitat for *Sirex noctilio* F. and removing these through silvicultural treatments can reduce tree mortality attributed to *S. noctilio* in a stand (Neumann et al. 1987; Dodds et al. 2014).

In contrast to bark beetles and *S. noctilio* that perform better in stressed or weakened hosts, some species, such as the white pine weevil, *Pissodes strobi* (Peck), prefer vigorous hosts (Alfaro et al. 1995). In addition, many defoliators prefer more vigorous hosts with leaves of high protein and water content (Dury et al. 1998). Furthermore, the species, size, and age of trees and foliage may also affect the development rate and survival of insect herbivores. For example, the spruce beetle has larger broods in white spruce, *Picea glauca* Moench (Voss), than in Sitka spruce, *Picea sitchensis* (Bong.) Carr., and Lutz spruce, *Picea* × *lutzii* Little (i.e. a hybrid of white and Sitka spruce) (Holsten and Werner 1990). However, although vigorous growth (foliage) may be more susceptible to herbivores, the tree as a whole may have other defence mechanisms to tolerate such damage (Stone 2001). For example, the foliage of fast-growing species of *Eucalyptus* is highly susceptible to herbivorous insects, but the trees can tolerate the defoliation because of their fast growth, i.e. they use foliage replacement as their defence mechanism. Compare this to slow-growing *Eucalyptus* species that utilise phytochemical and physical properties within leaves as the main defence against herbivorous insects.

Dispersal

Emigration and immigration are key processes in the life history of many forest insects, allowing some species to move out of unsuitable habitats, and to expand, or contract their range. Factors affecting dispersal include pest density, host density, body size, lipid content, and weather conditions (e.g. temperature, wind) (Smith et al. 2001; Evenden et al. 2014; Jones et al. 2019). Although dispersal is often short range within forest stands, wind-assisted long-range dispersal flights (30–100 km) of the mountain pine beetle are considered to be partly responsible for the beetle's range expansion across the Rocky Mountains into northern Alberta (Jackson et al. 2008; Safranyik et al. 2010).

Pests may also be spread by people via the inadvertent movement of infested material such as firewood (Jacobi et al. 2012) and solid wood packaging of goods in

shipping containers arriving at international ports (Haack 2006; Haack et al. 2014). This pathway has led to the establishment of invasive bark and wood boring insects outside their native ranges, such as the polyphagous shot hole borer, *Euwallacea formicatus* (Eichhoff) (Eskalen et al. 2012; Smith et al. 2019), and emerald ash borer, *Agrilus planipennis* Fairmaire (Herms and McCullough 2014), in North America, and Asian longhorned beetle, *Anoplophora glabripennis* Motschulsky, in North America and Europe (Haack et al. 2010). Importation of unprocessed logs from North America is thought to be the likely pathway that led to establishment of the red turpentine beetle, *Dendroctonus valens* LeConte, in China (Yan et al. 2005). Regulatory controls and phytosanitary treatment of wood packaging has reduced the risk of anthropogenic dispersal of pests (Haack et al. 2014). However, exotic, invasive species like spongy moth, *Lymantria dispar* (L.) (formerly gypsy moth), and emerald ash borer, spread by a process referred to as ‘stratified dispersal’, involving long distance movement of the insect by people to locations far beyond the area where the pest is established, combined with natural dispersal from outlier populations that establish at these new locations (Sharov and Liebhold 1998a, b; Herms and McCullough 2014). The better we understand factors that influence both natural and human-assisted dispersal of pests, the better we can predict rates and direction of spread and develop effective IPM strategies.

Forest/Stand Structure and Susceptibility to Pests

Tree species composition, age class distribution, stand density and host tree condition affect the susceptibility of forests to insect pests. By knowing stand conditions that favour a particular pest, or vice-versa, it is possible to develop *risk and hazard rating* models (see Sect. 17.1.4) that predict the potential impact of a pest in different stands. For high impact pests such as the mountain pine beetle and spruce budworm, this information can be used in decision-support systems to direct where and when management is implemented.

Monocultures often tend to be more susceptible to insect herbivory than mixed species forests (Jactel and Brockerhoff 2007; Guyot et al. 2015, 2016). One mechanism thought to be responsible for greater herbivory in less diverse plant communities is greater host availability and increased foraging efficiency. For example, vast areas of mature, even aged lodgepole pine, along with warmer than average winter temperatures that increase overwintering survival of mountain pine beetle, are considered important factors inciting mountain pine beetle outbreaks (Bentz et al. 2010; Safranyik et al. 2010). In contrast, host plants are less plentiful and also more patchy and more difficult for herbivores to locate in diverse stands. Diverse forests often have more complementary resources (pollen, nectar) and alternative hosts than monocultures, and this supports a more robust assemblage of natural enemies that can exert greater top-down regulation of herbivores (Lawton and Strong 1981).

17.1.2 Survey and Monitoring

Effective techniques to survey and monitor forest insect populations are critical components of IPM, and when available, provide natural resource managers with important information on how to prioritize management actions (Edmonds et al. 2000; Carnegie et al. 2005b). For example, decisions to harvest an area prior to an anticipated outbreak; to perform sanitation of infested trees to reduce pest populations; or to increase survey efforts in surrounding forests, are best made from predictions of tree damage or tree mortality estimated from survey data. However, sampling insects in forested environments presents unique challenges that may not be encountered in agricultural systems or urban forests where damage is more easily observed and quantified (Fettig et al. 2001, 2005). In addition, the cryptic nature of many insects and their presence in portions of trees that are difficult to sample (e.g. upper tree crowns) can make surveys of forest insects challenging (Ric et al. 2007).

17.1.2.1 Pest Density-Damage Relationships

Estimates of pest density based on regular surveillance provide managers with the opportunity to pre-emptively plan and implement IPM. Pest density-damage estimates are much more common for defoliators than other feeding guilds. For example, systems have been developed for spruce budworm using light traps to predict population trends (Simmons and Elliott 1985) and pheromone traps to predict larval densities (that can then be related to tree damage) (Sanders 1988; Rhainds et al. 2016). Pheromone-baited trapping systems have also been developed for the western spruce budworm, *Choristoneura occidentalis* Freeman (Niwa and Overhulser 2015), European sawfly, *Neodiprion sertifer* (Geoffroy) (Lyytikäinen-Saarenmaa et al. 2006) and spongy moth (Gage et al. 1990), among other species. Pheromone-baited traps are a preferred sampling tool in many cases as they are effective at detecting low-density populations, are often species-specific, especially with moth pests, and are relatively easy to use.

There are very few examples of using bark beetle trap catches to successfully predict tree mortality. However, trap catches of spruce beetle can be used to estimate its population phase (i.e. endemic vs. epidemic), which is linked to tree mortality (Hansen et al. 2006). Damage thresholds predicted from pheromone-baited traps have also been developed for European spruce beetle, *Ips typographus* (L.), in Italy (Faccoli and Stergulc 2004) and Sweden (Weslien 1992b). In the southeastern U.S., Billings (1988) developed a practical method of forecasting population trends and infestation levels of southern pine beetle based on captures of southern pine beetle and the ratio of southern pine beetle to one of its major predators. Attempts to predict tree mortality in western North America from trap catches of western pine beetle, *Dendroctonus brevicomis* LeConte, have been unsuccessful (Hayes et al. 2009).

17.1.2.2 Trap Trees

Trap trees are tools used to survey or monitor bole-infesting insects. Trees selected as trap trees are either artificially stressed through chemical or mechanical means, or pheromones are used to initiate insect colonization. Depending on the life history traits of the target insect, trap trees are either left standing (Neumann et al. 1982) or felled (Hodgkinson 1985). Trap trees provide multiple opportunities to detect target insects, including the capture of insects in traps attached to trap trees, collecting adult insects that emerge from sections of trap trees removed from the field and placed in rearing containers, or through signs such as galleries, resinosis or emergence holes that are reliably diagnostic (McCullough et al. 2009; Zylstra et al. 2010).

Trap trees have been used operationally to detect exotic species. Probably the best example of this is the use of chemically girdled pine trees as detection tools for *S. noctilio* in the Southern Hemisphere. Positive trap trees (those colonized by *S. noctilio*) are then integrated into the biological control program for managing *S. noctilio* using a parasitic nematode (Neumann et al. 1982; see Sect. 17.3.3). Trap trees have also been used operationally to detect emerald ash borer in North America (McCullough et al. 2009). In this context, ash trap trees are girdled using a chainsaw and a section of bark is removed at about breast height. Later in the summer, these trap trees are felled, and the bark is peeled to determine if larvae are present (Fig. 17.1).

A benefit of trap trees over pheromone-baited detection/monitoring traps, especially for an insect that does not utilize long-range sex pheromones, is that they provide a more complete suite of chemical and visual cues to attract the target insect, which often results in a more sensitive survey tool (Mercader et al. 2013). However, using trap trees is logistically more difficult than semiochemical-baited traps and creating dead trees in many areas can often create safety hazards if precautions are not undertaken. Felling, handling, and transporting tree sections into rearing facilities can also be challenging, expensive and time consuming. Where colonizing insects do not make signs of infestation that result in species determination, there can be an extensive lag time between when trap trees are colonized and when adults emerge from the wood allowing for species identification. However, using molecular techniques to identify pest species from larvae or frass can reduce the lag time for some species (Kethidi et al. 2003; Wilson and Schiff 2010; Ide et al. 2016).

17.1.2.3 Semiochemical-Baited Traps

Semiochemicals, including pheromones and kairomones, are used by insects to find mates or to locate suitable habitats and hosts (Roelofs and Cardé 1977; Wood 1982). These chemicals can be strong sources of attraction for insects and provide excellent survey and monitoring tools. Some semiochemicals are attractive to only one or very few species (e.g. *L. dispar* moth sex pheromone) while others may attract a broad range of species (e.g. alpha-pinene, a host plant volatile that is emitted from many tree species). Pheromones, used alone or with host volatiles, have been used extensively to detect and monitor Lepidoptera (Elkinton and Cardé 1981; Grant 1991; Jactel



Fig. 17.1 Ash trap trees, girdled in spring to increase their stress levels and attraction to the emerald ash borer, *Agrilus planipennis*, have been used in surveys to detect and delimit infestations of emerald ash borer (Photo credit: Pennsylvania Department of Conservation and Natural Resources)

et al. 2006; Jones et al. 2009), Coleoptera (Weslien 1992b; Brockerhoff et al. 2006; Sweeney et al. 2006; Billings and Upton 2010), and Hymenoptera (Lyytikäinen-Saarenmaa et al. 1999; Dodds and de Groot 2012) (Fig. 17.2). Combining more than one host volatile (e.g. alpha-pinene and ethanol) or combining pheromones with host volatiles can synergize attraction and increase trap captures of some Coleoptera (Chenier and Philogene 1989; Silk et al. 2007; Allison et al. 2013).

A wide array of traps are available for use with semiochemicals (Fig. 17.3). Multiple-funnel traps and panel intercept traps were designed specifically for bark beetles (Lindgren 1983) or bark beetles and woodborers (Czokajlo et al. 2001), respectively (Fig. 17.3). Modifications of these traps, including enlarging funnel holes (Miller et al. 2013), applying lubricants (de Groot and Nott 2003; Graham et al. 2010; Allison et al. 2011), and extending a collar above the bottom funnel (Allison et al. 2014) can improve trap captures. Canopy malaise traps have also shown promise for sampling bark beetles and woodborers (Vance et al. 2003; Dodds et al. 2015). Traps commonly used to monitor lepidopteran pests include pheromone-baited delta sticky traps or non-sticky traps that use a dry collecting cup and pesticide strip to kill captured insects (e.g. the Unitrap [Fig. 17.3d]). Sticky traps are efficient at catching moths but the sticky surface becomes saturated with moths (and moth

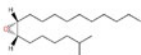






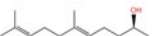




Lepidoptera Sex Pheromones		
<i>Lymantria dispar</i>	<i>Choristoneura fumiferana</i>	<i>Operophtera brumata</i>
		
disparlure	(<i>E</i>)-11-tetradecenal	(<i>Z,Z,Z</i>)-1,3,6,9-Nonadecatetraene
Coleoptera: Curculionidae: Scolytinae Aggregation Pheromones		
<i>Ips typographus</i>	<i>Dendroctonus ponderosae</i>	<i>Dendroctonus frontalis</i>
		
2-methyl-3-buten-2-ol	trans-verbenol	frontalin
Coleoptera: Cerambycidae Sex-aggregation Pheromones		
		
3-hydroxyhexan-2-one	<i>S,E</i> -fuscumol	monochamol
Host Plant Volatiles (General Attractants)		
		
ethanol	alpha-pinene	1- <i>S</i> -beta-pinene

Fig. 17.2 Examples of chemical structures (El-Sayed 2022) of some insect pheromones and plant volatiles used in forest insect pest surveys. Lepidoptera and Scolytinae pheromones are usually more specific than Cerambycidae pheromones and all of these pheromones are more specific than host volatiles

scales) at relatively low population densities, so they are not as suitable as non-sticky traps for monitoring large changes in population densities (Sanders 1986).

Various factors influence trap captures and can broadly be categorized as intrinsic and extrinsic. Intrinsic factors include the type of trap (Flechtmann et al. 2000; Sweeney et al. 2006; Dodds et al. 2015), trap color (Campbell and Borden 2009; Francese et al. 2010; Rassati et al. 2019), trap surface treatments (de Groot and Nott



Fig. 17.3 Examples of traps used to survey for forest insects: **a** multiple funnel and **b** intercept panel traps are commonly employed to survey for bark beetles and woodborers; **c** canopy malaise traps are used for bark beetles and woodborers as well as other taxa (Photo credits: K. J. Dodds); and **d** Unitraps are used to collect Lepidoptera (Photo credit: M. MacDonnell, University of New Brunswick, Fredericton, NB)

2003; Graham et al. 2010; Allison et al. 2011), type of collection cup or adhesive (Miller and Duerr 2008), placement of traps along environmental gradients (Dodds 2014; Schmeelk et al. 2016; Allison et al. 2019; Sweeney et al. 2020), and other trap modifications (Allison et al. 2014). Extrinsic factors include variables such as local forest disturbance history (wildfire and silvicultural treatments) (Sullivan et al. 2003; Dodds 2011), forest stand composition and structure (Ohsawa 2004), volume and decay class of downed wood (Lee et al. 2014), and local insect population levels where traps are deployed. Meteorological variables including temperature, relative humidity, precipitation, and wind may also influence trapping results (Salom and Mclean 1991; Peng et al. 1992; Jönsson and Anderbrant 1993).

The potential economic and ecological impacts of exotic, invasive forest insects have been the impetus for nationwide detection surveys using semiochemical-baited traps. Many countries, including Canada (Canadian Food Inspection Agency 2016), New Zealand (Brockerhoff et al. 2006), Australia (Wylie et al. 2008; Carnegie et al. 2018), and the U.S. (Rabaglia et al. 2008; Jackson et al. 2014), among others, have well-developed annual surveys that target bark beetles, woodborers, moths, and other damaging insects. These surveys may focus on individual target species (Wylie et al. 2008; Jackson et al. 2014) or on broader target taxa, e.g. species of bark and wood boring insects in the families Cerambycidae, Buprestidae, Curculionidae (Scolytinae), and Siricidae, at risk of transcontinental movement in wood packaging. In both cases, the goal is to detect introduced and established pests as early as possible, when populations and infested areas are small and the chances of eradication are good (Tobin et al. 2014). If a newly detected exotic species is considered to pose a threat, surveys with semiochemical-baited traps are implemented to delimit the population in the invaded region (Liebhold et al. 2016). Examples of large-scale delimitation efforts include *S. noctilio* (Dodds and de Groot 2012) and emerald ash borer (USDA APHIS PPQ 2017) in North America and *Uraba lugens* Walker in New Zealand (Suckling et al. 2005).

Semiochemical-based detection and monitoring traps are easy to deploy and consequently frequently used in IPM programs. Lures and traps are relatively inexpensive and most traps can be used for many years. However, understanding and interpreting what trap captures mean, and do not mean, is critical. Presence of an insect in a trap may or may not indicate the existence of a local population, as many insects can disperse long distances. Conversely, for several reasons a trap that is negative for a target species cannot be interpreted as evidence that the area is free of that species. Traps have a defined active sampling space, capture efficiencies are often low, and for some species, retention of individuals captured in traps is low (Elkinton and Childs 1983; Byers 2008; Allison and Redak 2017). For example, mark-release-recapture studies have shown that only 1–29% of bark beetles that are marked and released are recaptured in traps (Birch et al. 1982; Weslien and Lindelöw 1989; Zolubas and Byers 1995).

17.1.2.4 Ground-Based Surveys

When signs and/or symptoms of infestation are obvious, ground surveys can be very effective, especially in small stands or other areas where trees are easily accessible. Signs and symptoms of infestation, such as resin or staining (Coleman and Seybold 2008; Ryan et al. 2013), oviposition sites on tree boles (Ric et al. 2007), defoliation in crowns (MacLean and Lidstone 1982), tree crown fade (Billings and Pase 1979), bark flaking by woodpeckers (de Groot et al. 2006) or egg masses (Shepherd and Brown 1971; Liebhold et al. 1994) can all provide evidence of insect presence and in some cases, population levels. (Fig. 17.4). An advantage of ground surveys over pheromone-baited trap surveys is that signs or symptoms that are strongly correlated with a specific insect provide direct evidence that the species is established in the area. Further sampling of trees detected during ground surveys often yields more information, such as pest population estimates and identification of mortality factors. Although they are laborious and time consuming, ground surveys of infested trees are conducted on a systematic basis throughout the year in Europe during European spruce beetle outbreaks (Fettig and Hilszczański 2015). Once identified, infested trees are marked, numbered, and mapped for sanitation (see Sect. 17.1.4.2.4).

17.1.2.5 Remote Sensing and Aerial Detection Surveys

Remote detection methods are also useful tools for assessing the effects of insects on forest resources (Ciesla 2000). Surveys may be conducted using manned or unmanned aircraft (McConnell et al. 2000; Lehmann et al. 2015) or may integrate detailed information related to plant growth and stress from satellite systems or aircraft-based sensors. Remote sensing provides opportunities to rapidly gather information on changes in forest condition over large spatial scales (Stone and Mohammed 2017).

Aerial detection surveys using aircraft have been used in parts of the U.S. since the 1940s (Wear and Buckhorn 1955) and have occurred annually over much of the forested lands in the U.S. since the 1970s. Aerial surveys are also an important component of insect and disease monitoring in Canada (British Columbia Ministry of Forests 2000), New Zealand (Kershaw 1989), and Australia (Carnegie et al. 2008). Originally used to track tree mortality and damage, these surveys can also detect new pests or damage in new areas, estimate levels of damage, and provide guidance for further survey or management (McConnell et al. 2000; Johnson and Wittwer 2008). Aerial detection surveys are occasionally followed with *ground-based surveys* to more precisely delimit damage observed from the air.

Satellite technologies, including multispectral and hyperspectral sensors (airborne and spaceborne) and LiDAR (Light Detection and Ranging—a laser-based method of mapping landscape features) have been useful for survey and monitoring of forest insects. These approaches have also been used to identify tree species and to map tree distributions (Somers and Asner 2012), determine areas where tree stress is occurring (Hanavan et al. 2015), as well as mapping locations where insects have caused damage (White et al. 2006; Fassnacht et al. 2014; Stone and Mohammed 2017).

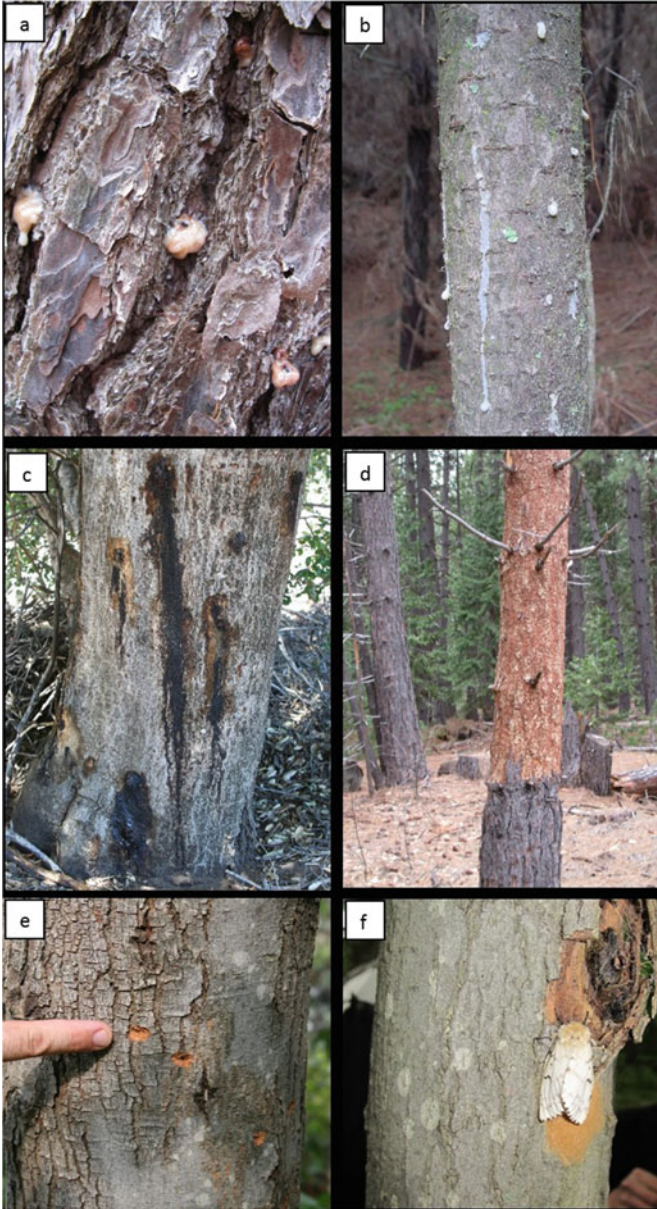


Fig. 17.4 Examples of signs used to survey for insects: **a** resin at the sites of southern pine beetle, *Dendroctonus frontalis*, attack (Photo credit: K. J. Dodds); **b** resin beading associated with *Sirex noctilio* oviposition sites (Photo credit: A. J. Carnegie); **c** bark staining resulting from goldspotted oak borer, *Agrilus auroguttatus*, attacks (Photo credit: T. W. Coleman, Forest Health Protection, USDA Forest Service); **d** woodpecker flaking on a ponderosa pine, *Pinus ponderosa*, colonized by western pine beetle, *Dendroctonus brevicornis* (Photo credit: C. J. Fettig); **e** Asian longhorned beetle, *Anoplophora glabripennis*, oviposition sites (Photo credit: K. J. Dodds); and **f** spongy moth, *Lymantria dispar*, egg mass (Photo credit: K. J. Dodds)

A drawback of remote sensing is that information collected at such large scales is often incomplete, e.g. it may be possible to detect tree stress or tree mortality but not necessarily determine a specific causal agent of tree mortality. Another challenge is the time lag between when the information is collected and when it is processed into a format that is useful to managers, although this is rapidly improving. Integration of a number of survey and monitoring tools provides the best chances to provide reliable information within the context of an IPM program.

17.1.3 Pest Impact Assessment and Cost–Benefit Analysis

Pest impact assessment considers the ecological and economic impacts of a pest. Defining the ecological impacts or changes to an ecosystem associated with particular pests is difficult (Swank et al. 1981; Reynolds et al. 2000; Lovett et al. 2002, 2006; Lewis and Liken 2007). Assessment of cumulative impacts must consider effects on individuals (e.g. reproductive success, growth), populations (e.g. genetics, population dynamics), communities (e.g. species diversity, species composition), ecosystems (e.g. nutrient cycling), and regions (Parker et al. 1999; Ricciardi et al. 2013) and is often context dependent. For example, bark beetle outbreaks are often detrimental to many ecological goods and services, while at the same time benefiting other ecological goods and services (Morris et al. 2018). By opening forest canopies and creating large gaps, grazing habitat may be enhanced. In rural areas, real-estate values may also increase due to better scenic views and transition to tree species more appealing to landowners.

Depending on how widespread tree damage or tree mortality is, different approaches to quantify ecological impacts are used. Among stands of similar type and conditions, comparisons can be made between infested and uninfested sites. The first step is often focused on plot-level vegetation assessment and establishment of permanent plots. Through this type of data collection, information can be gained on changes in forest condition, often with an emphasis on tree structure and composition. These types of assessments have been conducted for both native (Donato et al. 2013; Zeppenfeld et al. 2015) and invasive insect species (Dodds et al. 2010; Dodds and Orwig 2011; Simmons et al. 2014; Haavik et al. 2015). While standard vegetation plots provide strong information on impacts occurring at the stand level, knowledge of factors acting at larger spatial scales can also be investigated by dispersing vegetation plots or increasing their size throughout an impacted landscape (Orwig et al. 2008). Coarser landscape-scale assessments may be made through aerial surveys or remote sensing data.

Quantifying the costs associated with ecological impacts of forest pests is difficult due to challenges valuating and monetizing ecological goods and services (Boyd and Banzhaf 2007; Holmes et al. 2009; Stenger et al. 2009). However, the market value of forest trees for timber and fiber can easily be estimated and compared to management costs to determine the net cost or benefit of actions to reduce tree losses. Considering treatment options for insects that do not kill trees but cause growth

reduction is more challenging, as economic projections over the life of trees must be considered. Models may be used to predict the value of a stand that can then be used to guide pest management decision-making; these can range from simple stand growth models to more elaborate models that incorporate non-traditional forest products (Fox et al. 1997). Comparing growth over the rotation of treated and untreated stands can provide cost-benefits of control programs (Cameron et al. 2018b; Wardlaw et al. 2018), but such analyses are rare.

Although costs and benefits are underlying principles of IPM, full cost-benefit analysis rarely occurs in forest pest management (MacQuarrie et al. 2016; Niquidet et al. 2016; Cameron et al. 2018b). More frequently, cost-benefit analysis is done to determine the lowest cost option to achieve a specific management objective, for example, protecting foliage from defoliation by spruce budworm by using the lowest effective dose of *Bacillus thuringiensis kurstaki* (*Btk*, a soil-dwelling bacterium commonly used as a biological pesticide) (Morris 1984). Alternatively, cost-benefit analysis may be used to detect pest threshold levels that justify treatment (Niquidet et al. 2016).

Estimating the benefits of management interventions is often difficult in forestry due to long delays (25–60 years) between management actions and harvests combined with volatility in forest product prices and economic parameters (Niquidet et al. 2016). However, there have been some thorough cost-benefit analyses of pest management programs against exotic, invasive forest pests (e.g. Sharov and Liebhold 1998a; Cameron et al. 2018b). For example, the costs of protecting urban ash trees from the emerald ash borer have been demonstrated to be substantially lower than costs of tree removal and replacement following mortality due to emerald ash borer infestation (McCullough and Mercader 2012). Tobin (2008) showed that slowing the spread of spongy moth in North America is a cost-effective strategy in spite of the large infested area because it delays the costs associated with maintaining expanded quarantine zones and managing spongy moth outbreaks.

17.1.4 Management Strategies

There are two basic strategies to reduce the negative impacts of insects on forests. *Prevention* is designed to reduce the probability and severity of future infestations by manipulating stand, forest and/or landscape conditions. *Suppression* is designed to reduce current infestations by manipulating pest populations using *remedial tactics*. In some cases, *risk and hazard rating systems* or *decision support systems* are available to identify stands that should be prioritized for management (see Sect. 17.1.4.2). For example, the Spruce Budworm Decision Support System (SBWDSS) is used to quantify returns in marginal timber supply from protecting stands against spruce budworm infestations in Canada (MacLean et al. 2001).

When implementing prevention or suppression, managers should be cognisant of opportunities to address additional objectives with little or no additional cost. For example, in pine-dominated forests of the southern and western U.S., fuel reduction

treatments, such as mechanical thinning and prescribed fire, are frequently used to reduce forest fuels (Stephens et al. 2012). While prescriptions differ between thinning treatments implemented for fuels reduction and those for managing pest infestations, there are opportunities to alter fuel reduction treatments without reducing their efficacy while increasing the effectiveness of these treatments to reduce the susceptibility of forests to certain pests. In the latter case, *crown* or *selection thinning* (removal of larger trees in the dominant and codominant crown classes) may be necessary to achieve suitable reductions in the abundance of preferred hosts of certain pests. In other situations, some resource objectives may be negatively impacted by preventive and suppressive tactics, and it is prudent to identify as many of these impacts as possible and to adjust management strategies accordingly (e.g. changing the timing, scale, frequency and/or intensity of treatments) (e.g. Fettig et al. 2008, 2014).

As indicated earlier, in many cases, management strategies may not be justified due to ecological or social constraints. In other cases, the benefits of intervention may not justify the costs. Furthermore, metrics used to assess impacts caused by forest insects have traditionally been based on timber values yet, increasingly, emphasis is placed on the full range of ecological goods and services derived from forests (Morris et al. 2018; Fettig 2019). Unfortunately, empirical estimation of potential market and nonmarket values for most ecological goods and services is in its infancy (Stenger et al. 2009; McCollum and Lundquist 2019), and as such is a major obstacle in establishing credible linkages between management interventions and changes in economic valuations in forests.

17.1.4.1 Prevention

Regulatory Controls

Regulatory controls are designed to prevent the introduction of exotic, invasive species and/or to reduce their spread once established. These are usually informed by *pest risk assessments*, which quantify risks associated with the introduction and/or spread of exotic, invasive species based on assessments of relevant factors, such as invasion pathways, host distributions, and impacts. Data on the ecology and life history of many insect species are limited even in their native environments. Consequently, many pest risk assessments rely heavily on expert judgment and assessment. Although pest risk assessments can be useful in explaining the general causes and consequences of an invasion, more formalized and quantitative estimates of risk based on spatially explicit, multi-scale decision support systems are becoming more common (e.g. due to uncertainty associated with the impacts of climate change). Risk assessments for exotic species are now standard procedure prescribed by the World Trade Organization Agreement on Sanitary and Phytosanitary Procedures (Yemshanov et al. 2009).

Quarantines are used to reduce the spread of exotic, invasive species once established in a new environment, and include information on the regulated species and articles (e.g. host materials), the geographic scope of the quarantine, and penalties

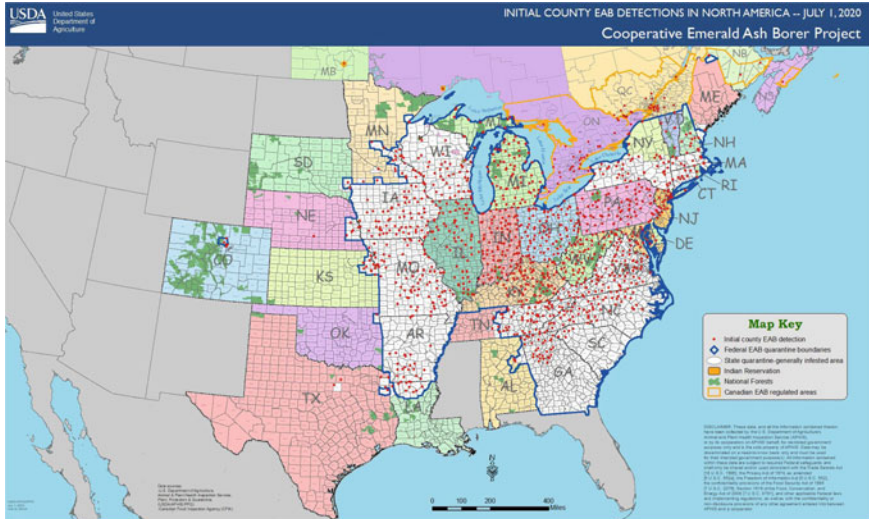


Fig. 17.5 Quarantines are regulatory measures designed to reduce the spread of exotic, invasive species once established in a new environment. The emerald ash borer, *Agrilus planipennis*, was first detected in Michigan in 2002, and by January 2021 had spread to 36 U.S. states and 5 Canadian provinces despite federal, provincial and state quarantines (Stone 2021)

for noncompliance. As an example, the state of Minnesota established a quarantine of pine wood with bark (Minnesota Statute 18G.06, subd. 4, 2013), exclusive of pine mulch or chips, pine Christmas trees and pine nursery stock, from areas of the U.S. determined to have established mountain pine beetle populations. Any person violating the quarantine is subject to civil and criminal penalties. The U.S. government, through the Animal and Plant Health Inspection Service (APHIS), imposed a quarantine on emerald ash borer in the eastern U.S. from 2003 to 2020 (Fig. 17.5). Internationally, standards to prevent the introduction and spread of exotic species are established by the International Plant Protection Convention (IPPC, www.ippc.int/en/). The International Standards for Phytosanitary Measures No. 15 (ISPM 15), first established in 2002, provides standards for wood packing materials (e.g. dunnage, crates, and pallets) used in international trade, and requires that they be heat treated or fumigated and branded with a seal of compliance (Haack et al. 2014).

Tree Breeding

Forest geneticists and tree breeders have traditionally focused on polygenic traits, because very few important traits in trees are controlled by single genes. An important exception is that of some disease resistance genes. As an example, populations of sugar pine, *Pinus lambertiana* Dougl., have been heavily impacted for decades by *Cronartium ribicola* J.C. Fisch., the exotic pathogen that causes white pine blister

rust. Although white pine blister rust can be fatal to all species of white pine, a gene occurs at low frequency in sugar pine that confers resistance to *C. ribicola* (Kinloch et al. 1970). Restoring populations of sugar pine involves, among other factors, identifying white pine blister rust-resistant trees in the field followed by selective breeding of these individuals, and eventual outplanting of white pine blister rust-resistant seedlings.

In general, tree breeders have largely ignored opportunities to increase insect resistance. For example, for decades it has been recognized that oleoresin flow characteristics in some pines are predictable and heritable (Smith 1975; Hodges et al. 1979) and potentially could be selected for in tree breeding programs to increase resistance to bark beetles (Strom et al. 2002). However, little progress has been made. Perhaps the most successful example of harnessing natural genetic variation for forest insect resistance involves the white pine weevil in North America. Several seed orchards have been established to grow white pine weevil-resistant spruce seedlings for use in reforestation (Alfaro et al. 2013).

There have been rapid advances in the application of plant biotechnology in the last two decades (Harfouche et al. 2011). While applications in forestry are experimental, there have been achievements in poplars, pines, and eucalypts, for example, involving insertion of *Bt* genes to increase resistance to insect defoliators. For genetically modified trees, significant study is required to evaluate the stable expression of genes after insertion into the tree, as biosafety concerns involving potential drift of genes into the environment must be considered (Vettori et al. 2016). As such, future uses of genetically modified trees in IPM programs will likely be limited to short-rotation woody cropping systems (Fig. 17.6).

Silvicultural Tactics

Silviculture is the backbone of IPM in forests (see Chapter 20), and in some cases begins with the proper selection of planting stock that is pest-free and appropriate for site conditions. When selecting planting stock managers should not only consider the climate of today, but that likely to be experienced in the future. Some experts suggest that *assisted migration*, the practice of planting tree species outside of their current distribution due to anticipated changes in the climatic niche, is important and should be applied more widely than has occurred (Gray et al. 2011). Planting may also provide an opportunity to increase tree species diversity, which as indicated earlier, is often associated with reductions in insect herbivory (Jactel and Brockerhoff 2007).

Managing stand density through thinning is an important silvicultural tactic in several systems (Figs. 17.7 and 17.8). Thinning operations vary in their prescription (e.g. some remove many trees of a particular species or size class whereas others may remove few trees) resulting in different stand structures and tree species compositions that influence susceptibility in different ways to different forest pests. While it is widely accepted that thinning is effective for reducing future levels of tree mortality attributed to some bark beetles (Fettig et al. 2007), there is no clear evidence that



Fig. 17.6 A five-year old poplar, *Populus* spp., research plantation at the Savannah River Site, South Carolina, U.S. This site received irrigation and fertilization throughout the growing season. Future applications of genetically modified trees to increase insect resistance will likely be limited to short-rotation woody cropping systems (Photo credit: D. R. Coyle, Clemson University)

thinning reduces losses from forest defoliators (Muzika and Liebhold 2000). In addition, there are examples where forest insects have greater impacts in thinned than unthinned stands, including the balsam fir sawfly in eastern Canada (Ostaff et al. 2006) and white pine weevil in western Canada (Alfaro and Omule 1990), but these tend to be the exception. In some cases, proper management of logging residues is important to reduce risks of future infestations by species that may breed in this material (Fettig et al. 2006). One unique variation to thinning is pre-emptive removal of certain host species in an attempt to limit the spread of a particular pest (Vannatta et al. 2012).

17.1.4.2 Suppression

Pesticides

Pesticides are an integral part of IPM, but social concerns and environmental considerations restrict their use in many forests, particularly in Europe. They may be applied manually, with ground-based equipment (e.g. soil and tree injection systems,



Fig. 17.7 Thinning ponderosa pine, *Pinus ponderosa*, in California to increase resistance to bark beetles, primarily western pine beetle *Dendroctonus brevicomis*, and mountain pine beetle, *D. ponderosae*. Among other factors, thinning reduces host availability; reduces competition among trees for nutrients, water, and other resources thereby increasing vigor; and affects microclimate decreasing the effectiveness of chemical cues used in host finding, selection and colonization by bark beetles (Photo credit: C. J. Fettig)

sprayers, blowers, and related equipment) or aerially with fixed-wing aircraft or helicopters. In forests, pesticides most commonly used for management of forest insects include contact and systemic insecticides, microbials (bacteria, viruses, pathogens and nematodes), insect growth regulators, soaps and horticultural oils. Most applications are confined to intensively managed areas, such as nurseries, seed orchards, short-rotation woody cropping systems and recreation sites.

Insecticides used to protect individual trees from colonization by bark beetles, and to a lesser extent woodborers, usually consist of ground-based sprays applied to the tree bole (Fig. 17.9). Residual activity varies by active ingredient, bark beetle species, tree species and associated climatic conditions (Fettig et al. 2013). In the western U.S., ten of thousands of trees may be treated annually to protect them from bark beetles during large-scale outbreaks, such as observed with mountain pine beetle in the mid-2000s (Fettig et al. 2021).

In recent years, researchers attempting to find safer, more portable, and longer lasting alternatives to bole sprays have evaluated the effectiveness of injecting small quantities of systemic insecticides directly into the tree bole with pressurized systems. These systems push low volumes of product, generally less than several hundred



Fig. 17.8 A loblolly pine, *Pinus taeda*, plantation thinned in Virginia as part of the Southern Pine Beetle Prevention Program. Since 2003, the Southern Pine Beetle Prevention Program, a joint effort of the USDA Forest Service and Southern Group of State Foresters, has encouraged and provided cost-share assistance for silvicultural treatments to reduce stand and forest susceptibility to southern pine beetle, *Dendroctonus frontalis* (Photo credit: J. T. Nowak, Forest Health Protection, USDA Forest Service)



Fig. 17.9 Protection of individual trees from mortality attributed to bark beetles may involve applications of liquid formulations of contact insecticides to the tree bole (Photo credit: C. J. Fettig)

milliliters for even large trees, into the small vesicles of the sapwood. Following injection, the product is transported throughout the tree to the target tissue (i.e. the phloem where bark beetle feeding occurs) (Fettig et al. 2013). In North America, bole injections have been demonstrated effective for mountain pine beetle, spruce beetle, and western pine beetle (Fettig et al. 2020) but are used most commonly in urban forests for control of exotic, invasive species such as emerald ash borer (Hermes and McCullough 2014).

Synthetic formulations of entomopathogenic microorganisms may also be useful for managing bark beetles and wood borers. Research efforts have focused on the fungus *Beauveria bassiana* (Bals.-Criv.) Vuill. Tactics under development include contaminating beetles collected in traps and then releasing these individuals back into field populations to contaminate the pest population (Kreutz et al. 2000; Lyons et al. 2012) and applying various suspensions of spores to the surfaces of felled and standing trees (Davis et al. 2018).

Most large-scale insecticide applications for defoliators involve the use of fixed-wing aircraft (Fig. 17.10). Among the first were applications in eastern Canada in the late 1920s when >85,000 kg of calcium arsenate dust was applied in attempts to control outbreaks of spruce budworm and hemlock looper, *Lambdina fiscellaria* (Guen.) (Holmes and MacQuarrie 2016). The development of synthetic organic insecticides in the early 1940s led to use of dichlorodiphenyltrichloroethane (DDT) for control of spruce budworm and other forest defoliators in North America. DDT remained the preferred control option for spruce budworm throughout the 1950s and 1960s (Nigam 1975) but by the mid-1950s the negative impacts of DDT were recognized (Turusov et al. 2002) and by the mid-1970s several countries banned most uses of DDT (Fig. 17.11). Today, microbial agents such as *B. thuringiensis* and insect growth regulators, such as diflubenzuron, have replaced the use of most synthetic insecticides for management of forest defoliators.

The Slow the Spread Program for management of spongy moth in the U.S. is a great example of the incorporation of insecticides into an IPM program. This combined federal and state effort involves detecting isolated populations of spongy moth with pheromone-baited traps (see Sect. 17.1.4.2.2) placed along the expanding population front from North Carolina to Wisconsin. In most cases, detected colonies are treated with *Btk*, diflubenzuron or mating-disruption pheromone. It has been estimated that this project has reduced the spread of *L. dispar* from infested areas to adjacent uninfested areas by >50% (Sharov et al. 2002).

Semiochemical Tactics

Semiochemicals are used to disrupt mating behaviors, mass trap pest insects, attract and kill insects, and to inhibit colonization of individual trees and forest stands. Semiochemicals have the benefit of being environmentally benign compared to insecticides, and many are species- or genera-specific. It is common practice to combine several semiochemical treatments, such as aggregation and anti-aggregation pheromones, into one IPM program targeting an insect species.



Fig. 17.10 Applications of insecticides for management of forest defoliators usually involve the use of fixed-wing aircraft. This photo shows an application of DDT to control the Douglas-fir tussock moth, *Orgyia pseudotsugata*, in Idaho in 1947. Today, microbial agents such as *Bacillus thuringiensis* and insect growth regulators such as diflubenzuron have replaced DDT and other broad-spectrum insecticides (Photo credit: Furniss [2007])

Mating disruption has been used successfully to manage some lepidopteran pests, primarily in agricultural and orchard settings (Cardé and Minks 1995). Fundamental to mating disruption success is release of a highly attractive sex pheromone from multiple points throughout a treatment area that makes mate location difficult (Cardé and Minks 1995; Miller et al. 2006). The Slow the Spread Program for spongy moth management represents the most extensive example of mating disruption for a forest pest. Synthetic spongy moth pheromone, disparlure, is spread over targeted landscapes through flakes applied by airplane (Tobin and Blackburn 2007). Since 1993, >5.5 million ha have been treated with a spongy moth mating disruption product (USDA 2016). Aerial applications of sex pheromone in large-scale field trials have suppressed spruce budworm mating (Rhainds et al. 2012) but not budworm egg or larval densities, likely due to immigration of mated female moths (Régnière et al. 2019). Other large-scale mating disruption programs for forest Lepidoptera are rare (Rhainds et al. 2012; Svensson et al. 2018). Examples of mating disruption are less common in other insect orders, however promising results have been demonstrated for pinhole borers (Coleoptera: Platypodinae) (Funes et al. 2011), cerambycids (Maki et al. 2011; Sweeney et al. 2017), and sawflies (Anderbrant et al. 1995; Martini et al. 2002).

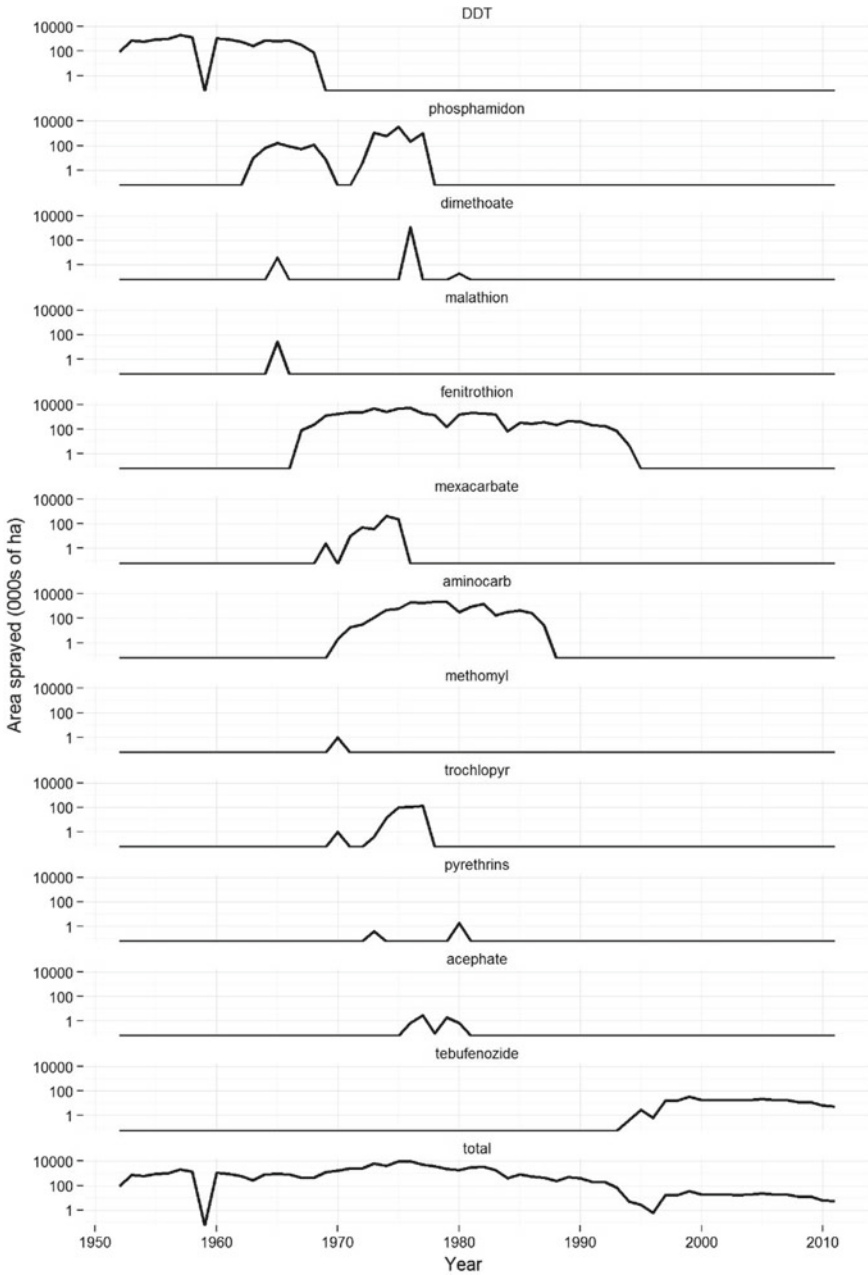


Fig. 17.11 Insecticides used for management of spruce budworm, *Choristoneura fumiferana*, in Canada (Figure credit: C.J.K. MacQuarrie, Natural Resources Canada; reproduced from Holmes and MacQuarrie [2016])

Mass trapping of insects is a population reduction technique in which pest insects are lured into pheromone-baited traps or stressed trees for purposes of collection and removal. Most attempts to mass trap forest pests have focused on bark beetles (Seybold et al. 2018). Evaluating the effectiveness of these programs is difficult, and often hindered by the inability to accurately estimate insect populations and determine the impact of mass-trapping on local pest populations or their damage. An extensive mass trapping program was used to reduce populations of European spruce beetle, but it was difficult to determine whether declining populations were due to mass trapping or to coincidental natural abiotic and biotic factors (Bakke 1991; Weslien 1992a). Despite this, Schlyter et al. (2001) provided evidence that northern spruce bark beetle, *Ips duplicatus* (Sahlberg), populations were reduced by mass trapping using pheromone-baited traps in Mongolia. Similarly, traps and trap logs were used to reduce tree mortality from European spruce beetle in spruce forests of Italy (Faccoli and Stergulc 2008). Another case where success was documented, in terms of total beetle captures and reduction in log degrade, was with ambrosia beetles in western North America. Ambrosia beetles create small holes in the sapwood of logs where they also inoculate fungi that stain and degrade the wood. Pheromone-baited traps were deployed in log yards to reduce populations of ambrosia beetles and to reduce log damage (McLean and Borden 1979; Lindgren and Borden 1983), and later developed into a larger annual mass trapping effort (Lindgren and Fraser 1994; Borden and Stokkink 2021). In other systems, successful reduction in damage has been less clear (Bakke 1991; Weslien 1992a; Ross and Daterman 1997). In an attempt to develop a tool that may aid in the reduction of pine wood nematode, *Bursaphelenchus xylophilus* (Steiner & Buhner) Nickle, in Europe, mass trapping of the cerambycid, *Monochamus galloprovincialis* (Olivier), was tested and showed promise for reducing *M. galloprovincialis* populations at local scales as part of an IPM program (Sanchez-Husillos et al. 2015).

The use of pheromone-baited traps for mass trapping of bark beetles has disadvantages. First, many of these semiochemicals elicit strong responses from insects and can result in spillover attacks on adjacent healthy trees. Consequently, managers should carefully consider placement of pheromone-baited traps and select areas where tree loss is acceptable and where attacked trees can be removed to further reduce local bark beetle populations (Ross and Daterman 1997). Another issue related to mass trapping is the large numbers of beneficial insects, especially natural enemies, which are also captured, but this can be reduced by trap modifications (Ross and Daterman 1998).

The use of trap trees can serve a similar function to mass trapping. Trap trees are baited with semiochemicals or induced (e.g. herbicide treatment) to release attractive semiochemicals in order to stimulate attack by target taxa. When implemented as part of a suppression plan, trap trees can be chemically treated with contact insecticides or injected with systemic insecticides to kill arriving insects (i.e. attract and kill) (Lister et al. 1976; Lanier and Jones 1985; Gray et al. 1990; Drumont et al. 1992; Hansen et al. 2016; McCullough et al. 2016) or left untreated to allow successful colonization by insects. Untreated trap trees are then removed and destroyed before

the next generation of insects emerges (Bakke 1989). Trap trees have been implemented in several countries for management of European spruce beetle (Bakke 1989) and have been effective (Raty et al. 1995), but they were less effective than pheromone-baited traps for suppressing Douglas-fir beetle, *Dendroctonus pseudotsugae* Hopkins, populations in western North America (Dodds et al. 2000; Laidlaw et al. 2003).

Push–pull systems use a combination of tools in an attempt to manipulate insect populations to colonize one area and ignore another. Common components of push–pull systems include attractants, such as aggregation pheromones, with simultaneous use of repellants, such as antiaggregation pheromones (Cook et al. 2007). For this method to be practical, the insect must have a strong response to semiochemicals used as attractants and repellants. Because of this, successful push–pull systems are rare but promising results have been demonstrated for a few bark beetle species (Lindgren and Borden 1993; Ross and Daterman 1994; Borden et al. 2006; Gillette et al. 2012; Seybold et al. 2018). However, because of concerns about spillover attacks induced by the use of synthetic pheromone, push only (repellent) treatments have received more attention. Of note, a push–pull strategy was ineffective in protecting trap trees (used for application of the biocontrol nematode for *S. noctilio*) from colonization by the eastern fivespined ips, *Ips grandicollis* (Eichoff), in Australia (Carnegie and Loch 2010).

Push–pull systems have also been tested on tree pests in settings outside of forests. In a nursery setting, a push–pull system using ethanol-baited traps and verbenone was unsuccessful at protecting trees from attack by non-native ambrosia beetles (Werle et al. 2019). The use of attractive UV lights to pull burnt pine longicorn, *Arhopalus ferus* (Mulsant), away from log storage and processing facilities has been tested in New Zealand as a means of reducing infestation of logs (Pawson and Watt 2009).

Biological Controls

Biological control is the reduction of pests through the activity of one or more biological control agents. We use the terminology of Eilenberg et al. (2001) who describe four different strategies: classical, inoculation, inundation, and conservation. Classical biological control involves “the introduction of an exotic, usually co-evolved, biological control agent for permanent establishment and long-term pest management” (Eilenberg et al. 2001). The main goal of classical biological control is the permanent establishment of biological control agent(s) to provide long-term pest control. It usually involves the importation and release of insect parasitoids (or occasionally predators) to control non-native insect pests. Classical biological control has by far been the most common method of biocontrol used in forest pest management, and has had reasonable success, providing long-term control of tree pests in 34% of cases (Kenis et al. 2017). In Canada, more than 150 species of biocontrol agents have been released against 41 different forest insects, resulting in long-term control of nine target species, all defoliators, such as the winter moth, *Opherophthera*

brumata (L.), the larch casebearer, *Coleophora laricella* (Hübner), and the European spruce sawfly, *Gilpinia hercyniae* (Hartig) (MacQuarrie et al. 2016). Parasitoid wasps released and established in the Southern Hemisphere for control of *S. noctilio*, primarily *Ibalia leucospoides* Hochenw. and *Megarhyssa nortoni* (Cresson) (Taylor 1976; Cameron 2012), can provide up to 50% parasitism (Carnegie et al. 2005a; Collett and Elms 2009). Some other examples include *Diaeretus essigellae* Stary and Zuparko for control of the Monterey pine aphid, *Essigella californica* (Essig.), in Australia (Kimber et al. 2010) (see Box 17.2), the predatory beetle *Rhisophagus grandis* Gyll. for control of the great spruce bark beetle, *Dendroctonus micans* Kug., in Great Britain (Evans and Fielding 1994), and the egg parasitoid, *Avetianella longoi* Siscaro, for control of the woodborer, *Phoracantha semipunctata* Fabr., in California, U.S. (Hanks et al. 1996).

In contrast to these successes, the introduction of more than 700,000 individuals of about 33 different predator species provided no measurable control of the balsam woolly adelgid, *Adelges piceae* (Ratzeburg), in North America (Kenis et al. 2017). A large-scale classical biological control program underway to control the highly destructive and invasive emerald ash borer in North America has successfully established egg and larval parasitoids and measured some impact on populations (Duan et al. 2014), but its long-term success remains uncertain (Bauer et al. 2015; Jennings et al. 2016). Non-target effects can also be significant, as the case of *Compsilura concinnata* (Meigen) illustrates. A highly generalist tachinid parasitoid introduced into North America in 1906 to control *L. dispar* and browntail moth, *Euproctis chrysorrhoea* (L.), *C. concinnata* had little impact on *L. dispar*, effectively controlled browntail moth, but likely caused the decline of several species of silk moths (Elkinton and Boettner 2012). Analyses of cost: benefit ratios of classical biological control of forest pests are rare but have been estimated at 1:15 for the winter moth and 1:19 for the European spruce sawfly, compared to about 1:2.5 for most chemical control programmes (Tisdell 1990).

Inoculation biological control is “*the intentional release of a living organism as a biological control agent with the expectation that it will multiply and control the pest for an extended period, but not permanently*” (Eilenberg et al. 2001). The distinguishing feature of this strategy is that control is not permanent, and additional releases of the biological control agent are necessary. An example of this strategy is the annual release of the nematode *Deladenus siricidicola* Bedding for *S. noctilio* in the Southern Hemisphere (Bedding and Akhurst 1974). The nematode is mass cultured and inoculated into trap trees weakened by herbicide treatment or girdling to increase attraction and susceptibility to colonization by the woodwasp. The nematodes infect the woodwasp larvae and render adult females sterile, effectively filling the woodwasp eggs with juvenile nematodes. Infected females then spread the infection when they lay their sterile nematode-filled eggs into other host trees (Bedding and Akhurst 1974) (see Sect. 17.3.3).

Inundation biological control is “*the use of living organisms to control pests when the control is achieved exclusively by the released organisms themselves*” (Eilenberg et al. 2001). In this strategy, the released biological control agents must control a sufficiently high proportion of the pest population, or reduce damage significantly,

before dispersing or dying. Control relies solely on the released biological control agent(s), not on their progeny. Examples of the use of this strategy in forestry are rare, but field trials in Canada showed that inundative releases of the native egg parasitoid, *Trichogramma minutum* Riley, suppressed spruce budworm populations and reduced defoliation (Smith et al. 1990a, b). Unfortunately, populations collapsed before commercial production could be made viable and the method was never used operationally (MacQuarrie et al. 2016). In the southeastern U.S., inundative releases of encapsulated *Trichogramma exiguum* Pinto and Platner increased rates of egg parasitism in the Nantucket pine tip moth, *Rhyacionia frustrana* (Comstock), but was considered impractical as a control strategy due to high predation of encapsulated *T. exiguum* by ants (Asaro et al. 2003).

Conservation biological control is the “*modification of the environment or existing practices to protect and enhance specific natural enemies or other organisms to reduce the effect of pests*” (Eilenberg et al. 2001). This strategy includes activities that protect or enhance populations of biological control agents, such as reduced or more targeted use of pesticides (e.g. Cadogan et al. 1995) or providing alternate hosts and food sources for natural enemies. For example, supplemental feedings of southern pine beetle parasitoids in the laboratory and field with Eliminate™, an artificial diet consisting largely of sucrose, was shown to increase their longevity and fecundity (Stephen and Browne 2000).

Box 17.2 Case history of classical biological control: Monterey pine aphid, *Essigella californica*

The Monterey pine aphid is native to western North America where it feeds on pines (Sorensen 1994) but is not considered of economic importance (Ohmart 1981). It was detected in Australia in 1998 (Carver and Kent 2000) and once established, spread quickly throughout the major pine growing regions (Anon. 2000; Carver and Kent 2000). The Monterey pine aphid has been associated with severe chlorosis and defoliation across much of the Monterey pine, *Pinus radiata* D. Don, plantation estate in mainland Australia and is considered a significant pest, especially following years of below-average rainfall (Eyles et al. 2011; Stone et al. 2013a, b) (Fig. 17.12a, b, c). Defoliation tends to be more severe in mid-rotation (16–20-year-old) to mature (30–35-year-old) stands, more often in the upper crown, and can cause up to 95% crown loss. In some years, 30–45% of the plantation estate is impacted. Defoliation by the aphid has been estimated to cause losses of AU\$21 million in annual wood production (May 2004). Investigations into management options in Australia determined that biological control would be the most cost-effective option, with an estimated net present value of around AU\$15 million over 30 years and providing a benefit in perpetuity (May 2004). The aphid’s only known parasitoid, *D. essigellae*, was described from museum specimens but live specimens had not been observed in the field (Kimber et al. 2010) (Fig. 17.12d, e). *Diaeretus essigellae* were subsequently located in California and imported to

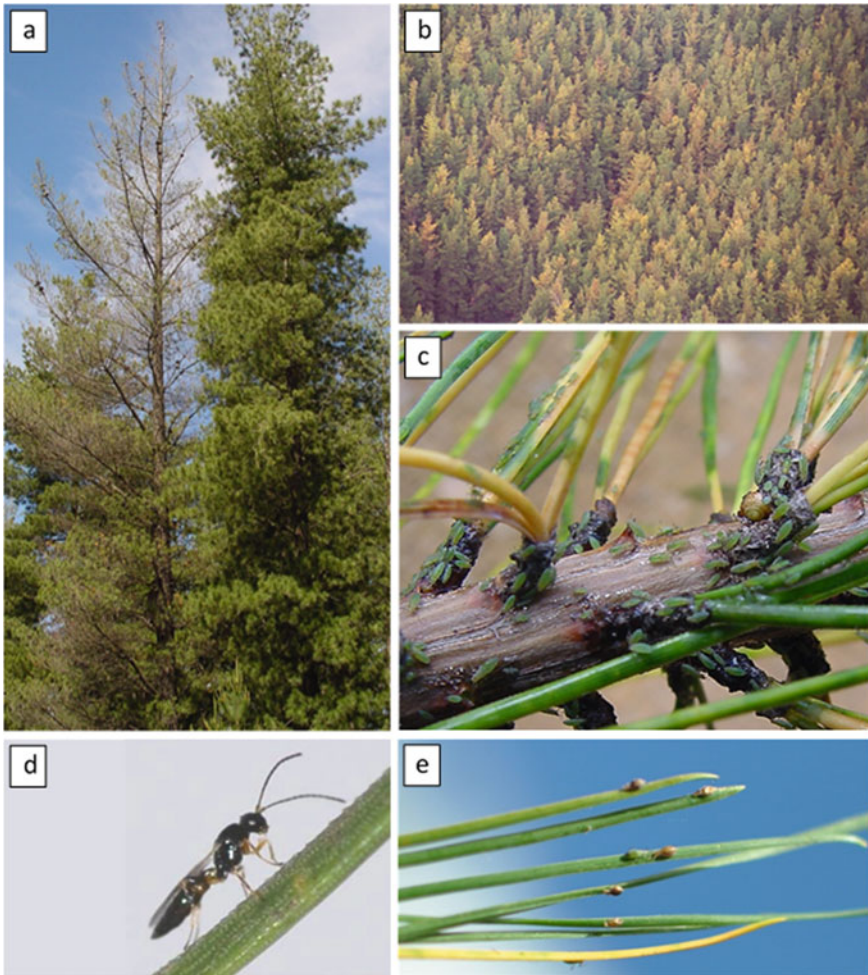


Fig. 17.12 The Monterey pine aphid, *Essigella californica*, on *Pinus radiata* in Australia—(a) defoliation of upper crown, compared to unaffected tree, (b) chlorosis in mature stand, (c) aphids on branch and needles (note needle chlorosis); (d) female *Diaeretus essigellae*; and (e) *D. essigellae*-parasitized *E. californica* (mummies) on pine needles (Photo credits: A. Carnegie [a, b, c, e], Forests and Wood Products Australia [d])

an approved quarantine facility where host-specificity testing was performed (Kimber et al. 2010). The first releases occurred in late 2009, with subsequent releases during the next three years in all Australian States where Monterey pine aphid was present. Annual monitoring for *D. essigellae* has occurred since 2010. There was initial concern when no established populations were detected by 2012 and no further releases were planned after 2012. However, in 2013,

a few established populations were detected, some up to 50 km from release points. By 2017, surveys found *D. essigellae* established in most pine growing regions in New South Wales and Victoria, and more recently in South Australia, but not in Tasmania, Queensland, or Western Australia. Established populations have been found over 100 km from release sites. In terms of impact, the area of New South Wales affected by the aphid has decreased following release of *D. essigellae*. However, in some years (e.g. 2017) there were localized areas with high populations of both the Monterey pine aphid and *D. essigellae*.

Cultural Tactics

Sanitation is a cultural tactic that involves the identification of currently infested trees, and subsequent felling and removal or treatment to destroy pests within the tree in order to reduce pest populations. At the smallest scale, this may include *pruning* of affected portions of the tree (e.g. twigs or branches). Sanitation is commonly employed for management of bark beetles in Europe and North America (Fettig and Hilszczański 2015). Where it is economically feasible, trees may be harvested and transported to mills where broods are killed during processing and milling of lumber and some economic return may be realized (Fig. 17.13). Otherwise, felled trees are burned, chipped, debarked, or treated by solarization (placement of infested material in the direct sun, which is often sufficient to kill brood beneath the bark in warmer climates). In some cases, an emphasis is placed on sanitation of newly infested trees during the early stages of tree colonization in order to reduce the quantity of attractive semiochemicals (e.g. aggregation pheromones) released into the stand. Synthetic attractants may be used to concentrate existing infestations within small groups of trees prior to sanitation (see Sect. 17.1.4.2.2).

Sanitation is used to disrupt the unique attack behavior of southern pine beetle in the southeastern U.S., which relies on the release of aggregation pheromones by pioneering beetles for initial (*spot*) infestations to expand. By harvesting and processing southern pine beetle-infested trees, plus a buffer strip of uninfested trees, spot growth can be halted and some economic return realized. However, timely sanitation is often not possible due to limitations in labor, processing, milling and other factors. In these cases, the best alternative consists of felling all freshly attacked and brood-bearing trees toward the center of the spot. In addition, a horseshoe-shaped buffer of uninfested trees at the spot's expanding front is felled to help disrupt pheromone plumes and recruitment of other southern pine beetle (Fettig et al. 2007). Typically, the width of the buffer is equivalent to the height of the average tree in the stand, although actual buffer width (3–90 m) varies depending on spot size and the rate of spot growth (i.e. numbers of recently attacked trees).

Salvage involves harvesting and processing dead trees usually to recover some economic value that would otherwise be lost or for safety concerns as dead trees



Fig. 17.13 Sanitation is considered the most effective tactic for reducing levels of tree mortality attributed to European spruce beetle, *Ips typographus*, in Germany and many other countries. During outbreaks, it is common for large numbers of currently infested trees to be harvested, decked, and transported to local mills for processing (Photo credit: C. J. Fettig)

pose hazards to forest visitors and workers. Technically, salvage is not a suppressive tactic as its implementation has no immediate effect on insect populations in most cases. However, the term commonly appears in the literature, and in certain cases salvage of damaged, broken or windthrown trees may limit future increases in insect populations. Salvage is commonly used for management of European spruce beetle (Wermelinger 2004; Fettig and Hilszczański 2015) as outbreaks are often incited by windstorms, which provide an abundance of downed and weakened host material that fosters rapid increases in European spruce beetle populations (Marini et al. 2017). In some situations, timing of salvage operations is critical to reduce economic losses, e.g. pine plantations in Australia weakened by fire or windstorms are susceptible to infestation by eastern fivespined ips and its associated blue stain fungus—the sooner timber is salvaged the lower the loss in value due to blue stain (Wylie et al. 1999).

17.1.5 Integrating IPM Within Overall Forest Ecosystem Management

Ideally, pest managers should work directly with foresters and other natural resource managers to develop forest management plans. This ensures that tactics to prevent or reduce pest impacts are considered from the outset (see Sect. 17.1.4.1). Clearly, IPM cannot focus on solving individual pest problems to the exclusion of other natural processes and management actions. For example, use of fertilizers to increase tree growth rates must be balanced against potentially greater feeding damage by pests such as the cottonwood leaf beetle, *Chrysomela scripta* F., and the cottonwood leafcurl mite, *Tetra lobulifera* (Keifer) (Coyle et al. 2005). Society's demand for multiple and sometimes conflicting values from forests, such as wildlife habitat, recreation, biodiversity, and forest products, requires that IPM be practiced within a framework of ecosystem management (Alfaro and Langor 2016). Integration of pest management within overall forest management planning is a start. Tools that can facilitate this process are models, risk analysis frameworks and decision support systems.

17.1.5.1 Modeling as an IPM Tool

Models are useful tools to synthesize what is known about processes affecting pest populations and pest density-damage relationships so that predictions can be made under different management scenarios. They are also useful for revealing gaps in understanding and directing future research. Many models have been developed to describe pest population dynamics and improve IPM of eruptive defoliators like the spruce budworm (Régnière and You 1991; Sturtevant et al. 2015), spongy moth (Liebhold et al. 1998; Sharov and Liebhold 1998a, b; Tobin et al. 2004), and forest tent caterpillar (Cooke et al. 2012), as well as bark beetles (Logan et al. 1998; Perez and Dragicevic 2010; Duncan et al. 2015). Statistical regression models quantify relationships between pest density and damage to determine action thresholds (Johns et al. 2006; Fry et al. 2008). Temperature-driven phenology models like BioSIM (Régnière 1996; Régnière et al. 2014) and the spongy moth life stage model (Gray 2010) forecast the timing of events in a pest's life cycle. Models have been used to improve the efficacy of sampling and control methods (Tobin et al. 2004), to predict changes in pest distributions under climate change scenarios (Carroll et al. 2006; Régnière et al. 2009), and to assess the risk of invasive species establishment (Pitt et al. 2007; Yemshanov et al. 2009; Gray 2010, 2016, 2017). Models have also explored factors affecting dispersal patterns and rates of spread of native and exotic, invasive forest pests (Sharov and Liebhold 1998a, b; Prasad et al. 2010; Křivan et al. 2016). Finally, outputs from models of pest population dynamics and impacts can be integrated with models of stand dynamics and forest inventory to develop decision support systems that can forecast damage and guide management actions.

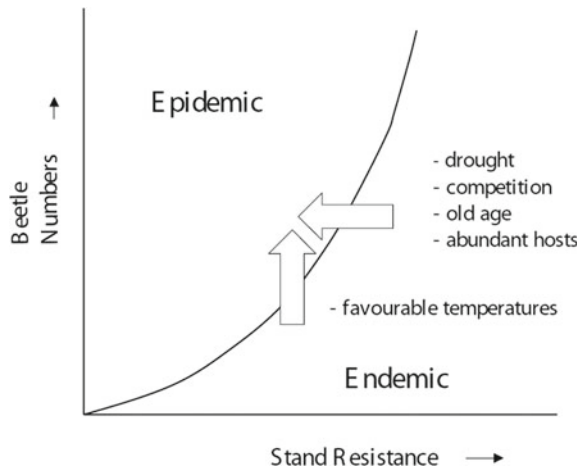
17.1.5.2 Risk Analysis Frameworks and Decision Support Systems

Risk analysis frameworks can help determine the risks of economic losses from major insect pests and identify optimal management responses (Fuentelba et al. 2013; Nealis 2015). They are particularly useful for facilitating cooperation among multiple landowners and jurisdictions with a common pest problem but with different priorities and policies (Nealis 2015) (see Sect. 17.2.1). The basic elements of risk analysis frameworks are risk assessment, risk response, and risk communication (Nealis 2015). Risk assessment and response address the questions of “what do we know?”, “what does it mean?” and “what should we do?” and works in an iterative fashion. Risk communication establishes a back-and-forth dialogue between managers and multiple stakeholders to establish community-based estimates of risk tolerance and to make the process as transparent as possible.

Using a risk assessment framework, Alfaro and Fuentelba (2016) outlined four steps when planning forest stand regeneration. First, the probability of pest damage is evaluated based on site variables such as climate, exposure, soil type, proximity to sources of potential pests, e.g. an early risk rating model for white pine weevil indicated low risk of damage to Sitka spruce in areas with low temperatures and high humidity (McMullen 1976). Secondly, models are used to forecast economic impacts at different infestation levels and integrated with cost and efficacy of control measures to determine the infestation threshold(s) where costs of pest damage exceed those of control actions. Third, the vulnerability of planting stock is determined and the tree species and provenance with least risk of pest damage are selected. For many years, Sitka spruce was not considered for regenerating many areas of British Columbia due to the impact of white pine weevil, but this has changed since the development of weevil-resistant Sitka spruce (Alfaro et al. 2013). The final step is to determine the appropriate management response, based on the previous three steps, and to communicate the level of risk to managers (Alfaro and Fuentelba 2016; Alfaro and Langor 2016).

Decision support systems incorporate models of stand-level and forest-level dynamics with models of pest population dynamics and pest impact to forecast damage and prioritize areas for management. They are particularly useful in regions that experience large-scale cyclical outbreaks of pests with significant impacts on tree mortality or tree growth and yield, e.g. the spruce beetle in Alaska (Reynolds et al. 1994), mountain pine beetle in western North America (Shore and Safranyik 2003) (see Sect. 17.3.2), spruce budworm in eastern North America (MacLean et al. 2001; Hennigar et al. 2007) (see Sect. 17.3.1), and spruce weevil in British Columbia (Alfaro et al. 1997). The SBWDSS forecasts the impact of budworm outbreaks on tree volume growth and tree mortality and integrates stand-level impacts across the forest to set priorities for foliage protection and/or harvest. Management of mountain pine beetle aims to keep populations in the endemic phase using direct controls (e.g. removal and burning of currently infested trees) to suppress beetle numbers, and preventive management to reduce the susceptibility of the forest to the beetle (Berryman 1978; Shore and Safranyik 2003) (Fig. 17.14). Parts of the mountain pine

Fig. 17.14 Factors contributing to a shift from endemic to epidemic populations in mountain pine beetle, *Dendroctonus ponderosae*. Management of mountain pine beetle is aimed at keeping populations in the endemic phase using direct controls to suppress beetle populations on a localized basis and preventive controls to increase resistance of trees and stands [after Shore and Safranyik 2003]



beetle decision support system have been adopted as part of the overall management strategy to contain the spread of mountain pine beetle in Alberta and provide a good example of efforts to integrate IPM within overall forest management planning (Anon. 2007b).

17.2 Constraints to Implementing IPM

17.2.1 Multiple Jurisdictions and Conflicting Priorities

Implementation of IPM within a large and diverse landscape is limited by available knowledge, tools, and budgets, and complicated by multiple landowners with different priorities and responsibilities (Nealis 2015). There are also legal and organizational constraints as well as differences in perspective among stakeholders that affects adoption of IPM in forests (Stark et al. 1985). Pest outbreaks do not recognize jurisdictional boundaries. In some cases, the actions of multiple agencies and jurisdictions are coordinated through pest-specific strategic planning committees. For example, a “Strategic Direction Council” was formed in response to the potential (and eventual) spread of mountain pine beetle into central Alberta, with members representing the Canadian Forest Service, Alberta Sustainable Resource Development, Alberta Community Development, Parks Canada, and forest industry. The council provides broad policy direction and priorities and ensures effective communication among the various agencies, while technical sub-committees develop and recommend specific management actions (Dalman 2003).

Risk analysis frameworks have been used to address multi-jurisdictional impediments to proactive IPM of major forest insect pests (Nealis 2015). A risk analysis framework was used to address the question of whether range expansion of mountain

pine beetle to central Alberta represented a significant threat to the boreal forest. It was successful in providing a timely science-based response and led to increased communication and cooperation among jurisdictions, e.g. Saskatchewan partially funds mountain pine beetle monitoring and control actions in Alberta (Nealis 2015). One of the most important benefits of a risk analysis framework is its emphasis on open communication and engagement with the public on the potential risks and uncertainties of pest impacts and management responses. When a risk analysis framework was applied to the spongy moth eradication program in British Columbia, public opposition to the program was reduced and some communities initiated programs to survey and monitor spongy moth populations (Nealis 2009).

17.2.2 Legal, Policy, and Economic Constraints

Most countries have laws and policies designed to protect the environment and to reduce the risk of negative impacts on environmental quality from human activities. For example, the National Environmental Policy Act (NEPA) in the U.S. and the Environmental Assessment Act in Canada require the completion of environmental impact assessments before undertaking any major management actions. Other levels of government have similar laws designed to protect people and the environment from potential harm from pest management activities, e.g. pesticide applications. While these laws are necessary to protect the environment and human health, the requirements can be complex and may constrain the implementation of IPM programs. For example, 60–120 m buffer zones are required around streams and other water bodies to protect aquatic ecosystems from aerially-applied pesticides, which can present a significant operational challenge. Fortunately, the efficacy and efficiency of aerial applications of pest control products has increased significantly in the last couple of decades with ultra-low volume sprays, use of process-oriented models (Cooke and Régnière 1996; Régnière and Cooke 1998), and on-board electronic guidance systems that optimize aircraft flight lines to compensate for changes in altitude and wind direction, and buffer zones (McLeod et al. 2012). These advances have greatly minimized drift of pesticides onto non-target areas and buffer zones (Thompson et al. 2010).

The Environmental Protection Agency (EPA) in the U.S. and the Pest Management Regulatory Agency (PMRA) in Canada have strict requirements for the registration and use of pest control products. Before approval of a new control product, much research and development are necessary to determine its efficacy against the target pest(s) and its impact on human health and the environment, and this requires substantial investment. For example, it costs an estimated \$286 million and 11 years to develop and register a new pesticide; not surprisingly, <0.001% of newly discovered active ingredients become registered pest control products (Anon. 2017). For private companies, this largely makes it cost prohibitive to develop anything other than broad-spectrum pesticides with large potential markets to ensure a return on

investment. Development and registration of more environmentally benign, species-specific products like baculoviruses or semiochemicals have usually been done with government assistance due to limited markets and return on investment. This is particularly evident in the forestry sector (Gillette and Fettig 2021). Fortunately, since around 2001, regulatory agencies have reduced the requirements for registration of pest control products like pheromones and microbial insecticides that, due to their less toxic nature and greater species-specificity, are considered lower risk (Anon. 2001, 2002). However, even for environmentally benign products, strict data requirements must be met, and costs can be substantial (Lucarotti et al. 2007).

17.2.3 Attitudinal Constraints and Social License

Public opposition to the application of pesticides has severely limited their use in most forests. To that end, it is difficult to imagine present day public acceptance of DDT applications in Stanley Park, Vancouver, Canada, but such applications were made in the 1950s for control of hemlock looper and the greenstriped forest looper, *Melanolophia imitata* (Walker) (Holmes and MacQuarrie 2016). Between 1985 and 1990, *Btk* gradually replaced synthetic insecticides in spruce budworm control programmes in Canada (Van Frankenhuyzen et al. 2000). However, aerial applications of *Btk* in urban settings for control of exotic, invasive insects like the Asian spongy moth, *Lymantria dispar asiatica* Vnukovskij, remains contentious (Ginsburg 2006) in spite of studies that indicate no effects on public health (Green et al. 1990). Interestingly, a survey of public attitudes toward control of forest insects in Ontario, Canada found that acceptance of insecticide use by the public was higher in people that had experienced outbreaks of pest insects (MacDonald et al. 1998).

The Forest Stewardship Council (FSC) and similar organizations were established to promote sustainable forest management and environmental integrity and they influence many pest management practices in forestry. Many consumers and some companies will not purchase forest products from industries that do not follow FSC standards. In 2015, an estimated 181 million ha of forests in 80 countries were FSC certified (Zanuncio et al. 2016). Pesticides designated as highly hazardous are severely restricted and cannot be used in FSC-certified estates "...without specific derogation, regardless of prevailing national approvals system" (Anon. 2007a). FSC prohibition of certain insecticides has made it difficult to manage some forest insect pests, e.g. chrysomelid beetles in eucalyptus plantations (Carnegie et al. 2005b) and leaf-cutting ants, *Atta* and *Acromyrmex* spp., in Brazil (Zanuncio et al. 2016).

17.3 IPM Programs for Major Forest Insect Pests

17.3.1 *Spruce Budworm in Eastern North America*

The spruce budworm is a major defoliating pest of balsam fir, *Abies balsamea* (L.) Mill., and spruce forests in North America. Populations erupt every 30–40 years and outbreaks last 1–20 years (Gray et al. 2000), causing extensive tree growth loss and tree mortality (MacLean 2016). Between 1975 and 2000, budworm caused moderate to severe defoliation on >450,000,000 ha of forests in Canada, reducing radial growth rates by as much as 75% and killing an average of 85% of trees in mature balsam fir stands and 36% of trees in mature spruce stands (MacLean 2016; Miller 1977). In the past few decades, spruce budworm management has consisted mainly of aerial applications of insecticides to protect foliage and reduce volume loss and tree mortality. Surveys are done by aerial sketch mapping to locate areas of moderate to severe defoliation, and by branch sampling of the overwintering second instar larvae to estimate budworm abundance and to predict subsequent levels of defoliation. Phenology models like BioSIM aid managers in the most effective timing of *Btk* applications (Régnière and Sharov 1998; Régnière et al. 2014).

The SBWDSS (Erdle 1989; Hennigar et al. 2007; MacLean et al. 2001), which projects the effects of budworm outbreaks on stand volume growth and tree mortality, is built on four relationships that quantify budworm impacts: (1) forecasts of budworm population levels over time, based on population dynamics models and previous outbreak cycles; (2) the relationship between pest population level and defoliation level; (3) the relationship between damage level (defoliation) and tree/stand growth loss and mortality; and (4) effects on the forest landscape as a function of accumulated stand-level impacts (Erdle and MacLean 1999). For defoliators such as spruce budworm, the main factor influencing the impact on tree growth loss and mortality is cumulative defoliation (MacLean 2016). The model allows managers to compare the effects of various scenarios of foliage protection, harvest scheduling, and salvage on future wood supply. Additional software, called the *Accuair Forest Protection Optimization System* (ForPRO), integrates the SBWDSS with the Woodstock timber supply model and allows users to simulate the impacts of spruce budworm on stand- and forest-level growth and yield under different protection scenarios (Hennigar et al. 2007) (Fig. 17.15). The SBWDSS and ForPRO have been tested in Maine, U.S., and Quebec, Alberta, Saskatchewan, New Brunswick, and Newfoundland, Canada, and were used to plan insecticide spray operations for several years during the latest budworm outbreak in Saskatchewan.

Faced with the likelihood of another spruce budworm outbreak in New Brunswick, the ‘Healthy Forest Partnership’ (a consortium of federal and provincial governments, forest industry, and universities) was formed in 2014 with the goal of testing an ‘early intervention strategy’ against spruce budworm (Anon. 2022). The strategy involves intensive monitoring and early detection of populations in supposed “hotspots” and suppression of these populations with applications of *Btk* or tebufenozide (Mimic®) to prevent the outbreak from spreading via moth dispersal (MacLean 2016). The

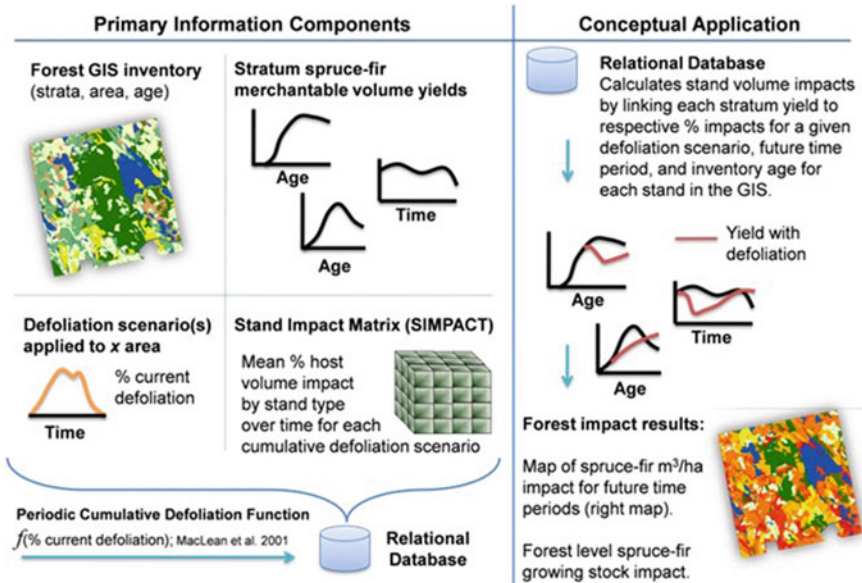


Fig. 17.15 Information sources required and conceptual application of the SBWDSS to calculate spruce-fir stand volume impacts for alternative spruce budworm, *Choristoneura fumiferana*, defoliation scenarios (From: C. H. Hennigar 2009)

most significant change in management strategy is the switch from foliage protection to that of population suppression. This has been controversial and is based on the notion that spruce budworm outbreaks spread from localized foci or hotspots, as suggested by early models of spruce budworm population dynamics (Morris 1963; Clark et al. 1979). That early notion had been rejected in subsequent analyses of budworm population dynamics (Royama 1984, 2012), which instead hypothesized that the oscillation in budworm populations was driven by density-dependent mortality from a complex of natural enemies (Eveleigh et al. 2007), and that dispersal of egg-carrying moths contributed only to secondary fluctuations or “noise” about the basic cycle. Subsequent studies supported Royama’s hypothesis that mortality from natural enemies is the main driver of budworm population cycles, but also suggested that food availability and moth dispersal may play larger roles in population dynamics, e.g. an influx of moths might counteract mate-finding Allee effects observed in low-density populations (Régnière et al. 2013; Pureswaran et al. 2016).

As part of the early intervention strategy, SBWDSS and ForPRO have been modified from the original objective of foliage protection to that of population suppression based on overwintering budworm larval populations and were used to plan budworm suppression treatments in New Brunswick and Newfoundland from 2015–2020 (Johns et al. 2019). It is too early to determine the success of the budworm early intervention strategy, but results from the first five years are promising (MacLean et al. 2019). The program has generated data on dynamics of low-density budworm

populations, fostered communication and collaboration among government, industry, and academia, and is stimulating public involvement through citizen science initiatives to help monitor budworm dispersal events through an extensive network of pheromone-baited traps and smart phone applications (Carleton et al. 2020).

17.3.2 Mountain Pine Beetle in Western North America

In western North America, about 15 species of bark beetles are capable of causing large amounts of tree mortality (Bentz et al. 2020). Most notable is the mountain pine beetle, which colonizes several tree species, including lodgepole pine, ponderosa pine, *Pinus ponderosa* Dougl. ex Laws., sugar pine, limber pine, *Pinus flexilis* James, western white pine, *Pinus monticola* Dougl., and whitebark pine, *Pinus albicaulis* Engelm., among others (Negrón and Fettig 2014). Recent outbreaks of mountain pine beetle have been severe, long lasting, and well documented (Audley et al. 2020; Fettig et al. 2021). While a formal IPM program is not universally recognized for mountain pine beetle, many of the associated components have been developed and are being implemented at different scales.

Information on the intensity and extent of mountain pine beetle infestations is most often accomplished by aerial detection surveys using fixed-wing aircraft and/or helicopters (Wulder et al. 2006) followed by ground-based surveys of areas with noticeable levels of tree mortality. During surveys, a common method of estimating when trees were colonized and killed by mountain pine beetle uses needle color and retention with three stages: *green stage* (within one year of attack; green foliage or foliage just beginning to fade); *red stage* (1–3 years since death; red foliage); and *grey stage* (>3 years since death; grey, limited or no foliage). However, it is important to emphasize that these are crude estimates that may vary by several years from the actual time since tree death.

Several risk and hazard rating systems have also been developed to describe the susceptibility of a stand to infestation by mountain pine beetle. The most frequently used was developed by Shore and Safranyik (1992) for lodgepole pine. Stand susceptibility is calculated based on four factors: (1) percentage of susceptible basal area (trees ≥ 15 cm dbh), (2) average stand age of dominant and co-dominant trees, (3) stand density of all trees ≥ 7.5 cm dbh, and (4) the geographic location of the stand in terms of latitude, longitude, and elevation. Insect population data, referred to as a “beetle pressure index”, incorporates the proximity and size of the mountain pine beetle population. The stand susceptibility index and the beetle pressure index are then used to compute an overall stand risk index (Shore and Safranyik 1992).

The first documented use of suppressive tactics for mountain pine beetle occurred in the early 1900s in the Black Hills of South Dakota and Wyoming, U.S. (Hopkins 1905). Today, strategies often aim to reduce localized populations, slow the rate of spread of infestations, and to provide protection of individual trees or stands. These focus on the use of insecticides, semiochemicals and sanitation harvests. Coggins et al. (2011) found that mitigation rates of >50% (sanitation harvests) coupled with

ongoing detection and monitoring of infested trees within treated sites in British Columbia was sufficient to control mountain pine beetle infestations, especially with persistent implementation. Alternatively, other researchers have stressed that many large-scale, well-funded and well-coordinated sanitation efforts were largely ineffective, and that resources would be better allocated to prevention (e.g. Wickman 1987). Sanitation is likely to be most effective if the following IPM principles are followed: (1) early detection, (2) rapid response, (3) continued monitoring, and (4) persistent application of suppressive treatments until populations return to endemic levels.

Age-class structure and tree species composition are dominant factors influencing the extent and severity of mountain pine beetle infestations (Taylor and Carroll 2003). Preventive tactics, such as thinning, that address these factors will influence the susceptibility of forests to mountain pine beetle infestations. Among other factors, thinning reduces host availability; reduces competition among trees for nutrients, water, and other resources thereby increasing vigor; and affects microclimate, decreasing the effectiveness of chemical stimuli used by mountain pine beetle in host finding, selection and colonization (Progar et al. 2014). Thinning implemented for mountain pine beetle in lodgepole pine, where the species has its greatest impacts, include thinning from above or diameter-limit thinning, and thinning from below (i.e. focusing on removal of trees in the suppressed and intermediate crown classes) applied to reduce basal area, remove trees with thick phloem, and/or increase residual tree spacing (Fettig et al. 2014). Thinning from below may optimize the effects of microclimate, inter-tree spacing, and tree vigor even though residual trees are of diameter classes considered more susceptible to mountain pine beetle (Mitchell et al. 1983). Bollenbacher and Gibson (1986) provide a list of attributes useful for assessing the potential effectiveness of thinning for reducing the probability of mountain pine beetle infestation and extent of tree mortality in lodgepole pine forests (Table 17.1).

Cottrell et al. (2020) examined the current state of knowledge regarding institutional, social, and environmental factors that influence the ability to manage

Table 17.1 Favourable conditions for reducing the probability of mountain pine beetle, *Dendroctonus ponderosae*, infestation and extent of tree mortality by thinning of lodgepole pine, *Pinus contorta*, forests in the Intermountain West, U.S. (adapted from Bollenbacher and Gibson 1986)

Parameter	Value
Stand composition	>80% <i>Pinus contorta</i>
Stand age	60–110 years
Basal area	>29.8 m ² /ha
Stand density	750–1500 trees/ha (>7.5 cm dbh ¹)
Average diameter	>20 cm dbh
Elevation	<1800 m
Percentage of trees currently infested	<10%

¹Diameter at breast height

mountain pine beetle (i.e. “adaptive capacity”). Three main categories were identified: (1) environment including stressor (i.e. mountain pine beetle), exposure (i.e. system connectivity) and sensitivity (i.e. forest health) factors; (2) society including impacts (i.e. metrics), public opinion (i.e. communication, perceptions, and attitudes), and management (i.e. proactive and reactive); and (3) ecosystem services including aesthetics, air quality, carbon sink/source, timber resources and water quality/quantity. Their research provides a framework for managers and policy-makers that is useful in identifying limitations in adaptive capacity in hopes of addressing them more effectively in the future. Public opinion, and the availability of human and financial capital, were identified as significant constraints.

17.3.3 *Sirex noctilio* in Australia

Sirex noctilio is native to Eurasia and northern Africa and is now a significant pest in exotic pine plantations in the Southern Hemisphere (Slippers et al. 2011) (Fig. 17.16a, b). Females lay eggs into trees along with a white rot fungus, *Amylostereum areolatum* (Chaillat ex Fries) Boidin, and a phytotoxic mucus (or toxin); the combination of which eventually kills the tree (Ryan and Hurley 2012). First reported in New Zealand in 1900, it was not until severe droughts occurred in unthinned, over-stocked stands that *S. noctilio* became a serious pest, with 33% of trees killed on 120,000 ha between 1946 and 1951 (Morgan and Stewart 1966; Bain et al. 2012). *Sirex noctilio* was first detected in Australia in Tasmania in 1952 (Gilbert and Miller 1952), on the mainland in 1961 (Irvine 1962), and is now established in major pine growing regions in Victoria, South Australia, New South Wales, and Queensland (Neumann et al. 1987; Carnegie and Bashford 2012). Between 1987 and 1989 >5 million trees were killed in a single area in southern Australia (Haugen 1990), and a contemporary analysis of this outbreak calculated the value of lost wood production at AU\$22.3 million (Cameron et al. 2018a). *Sirex noctilio* is also a significant pest in major pine growing regions in South America (Iede et al. 2012; Klasmer and Botto 2012) and South Africa (Hurley et al. 2012).

In response to severe losses in Australia, a *Sirex Management Strategy* was developed (Haugen et al. 1990), which includes biological control (Bedding et al. 1993), forest surveillance, quarantine and silvicultural methods (National Sirex Control Committee 2022). The first attempts at biological control occurred in New Zealand in the late 1920s and 1930s, with releases of *Rhyssa persuasoria* (L.) (Cameron 2012). *Ibalia leucospoides* was later introduced and established in the 1950s, followed by *M. nortoni* in the 1960s (Fig. 17.16c). In response to the establishment of *S. noctilio* on the mainland of Australia, the Sirex Biological Control Unit was established in the United Kingdom by the Commonwealth Scientific and Industrial Research Institute (CSIRO) with the aim to collect and identify parasitoids of potential use in Australia (Carnegie and Bashford 2012). Over 20 parasitoids were collected and sent to Australia and New Zealand, including *Schlettererius cinctipes* (Cresson), *I. leucospoides*, *M. nortoni*, and *R. persuasoria* (Taylor 1976; Nuttall 1989; Hurley



Fig. 17.16 *Sirex noctilio*: **a** tree mortality from an outbreak in a 13-year-old Monterey pine, *Pinus radiata*, plantation in Australia; **b** *S. noctilio* female; **c** *Megarhyssa nortoni* female; **d** *Deladenus siricidicola* being injected into trap tree (Photo credits: A. Carnegie [a, b, d], New South Wales Department of Primary Industries [c])

et al. 2007). Releases of parasitoids in South America and South Africa mostly originated from Australia and New Zealand (Hurley et al. 2007; Cameron 2012). The most successful parasitoids in Australia and elsewhere have been *I. leucospoides* and *M. nortoni* (Carnegie et al. 2005a; Hurley et al. 2007; Collett and Elms 2009). *Ibalia leucospoides* is the most abundant and effective parasitoid in Australia, with parasitism generally from 30–50% (Carnegie et al. 2005a; Collett and Elms 2009). It is possible that *M. nortoni* abundance is quite high in some areas in Australia, but current sampling techniques are not optimal for monitoring this parasitoid, due to asynchronous emergence times of the pest and parasitoid (Carnegie et al. 2005a; Collett and Elms 2009).

The nematode *D. siricidicola* was first found parasitizing *S. noctilio* in New Zealand (Zondag 1969), likely introduced with *S. noctilio* (Hurley et al. 2007). Subsequent surveys revealed up to 90% parasitism of *S. noctilio* in some forests (Zondag 1979). Concurrent with the CSIRO survey for *S. noctilio* parasitoids was a search for parasitic nematodes, with several hundred strains from seven species screened and tested for parasitism (Bedding and Akhurst 1978; Bedding and Iede 2005). A single strain, Sopron, grew well on *A. areolatum* cultures, did not parasitize the siricid-attacking parasitoids, and emerging wasps were larger than those parasitized by other strains (Bedding and Akhurst 1978; Bedding 2009). This strain resulted in almost 100% parasitism of *S. noctilio* in Australia (Bedding and Akhurst 1974) and was released operationally into areas where *S. noctilio* spread (Fig. 17.16d). In addition to the use of biological controls, silvicultural treatments such as thinning from below, to remove suppressed trees and to increase vigor of the remaining trees, are important IPM components, and are commonly used to reduce the impact of *S. noctilio* in commercial forests throughout the Southern Hemisphere (Dodds et al. 2014).

17.3.4 *Eucalyptus* Leaf Beetles in Australia

The leaf beetle, *Paropsisterna bimaculata* (Olivier), is endemic to Tasmania, and has long been a significant defoliator of eucalypts in native forests (Greaves 1966; de Little 1979) (Fig. 17.17a, b). Early attempts to establish eucalypt plantations in Tasmania resulted in severe damage to trees from *P. bimaculata* (de Little 1989; Candy et al. 1992), and it was realized that an effective management strategy was needed (Elliott et al. 1992). A large research program ensued from the 1970s to the 1990s that included study of the biology of *P. bimaculata* and its natural enemies (Elliott and de Little 1980; de Little 1983; de Little et al. 1990) and the impacts of defoliation on tree growth (Candy et al. 1992; Elliott et al. 1992; Elek 1997; Candy 2000). From this research, an IPM program for *P. bimaculata* in eucalypt plantations was developed (Elliott et al. 1992) and later refined (Candy 2000, 2003). This program includes: (1) monitoring egg and larval populations in young plantations; (2) predicting damage based on population estimates; and (3) determining economic injury thresholds to guide control decisions, (i.e. aerial application of insecticides) (Fig. 17.17c, d).

Forestry Tasmania (now Sustainable Timber Tasmania) implemented the IPM program operationally in 1998 coinciding with a rapid expansion of the eucalypt plantings in Tasmania (Wardlaw et al. 2018). Research into non-insecticidal control options was initiated during the early stages of the IPM program to avoid disrupting the high levels of natural predation (Elliott et al. 1992) and threats to non-target aquatic animals (de Little 1989), but more recently the focus has turned toward gaining FSC certification (Wardlaw et al. 2018). The use of *Bacillus thuringiensis* var. *tenebrionis* (*Btt*) was investigated (Beveridge and Elek 1999; Elek and Beveridge 1999), but its efficacy was limited to first instar larvae, and it was insufficiently

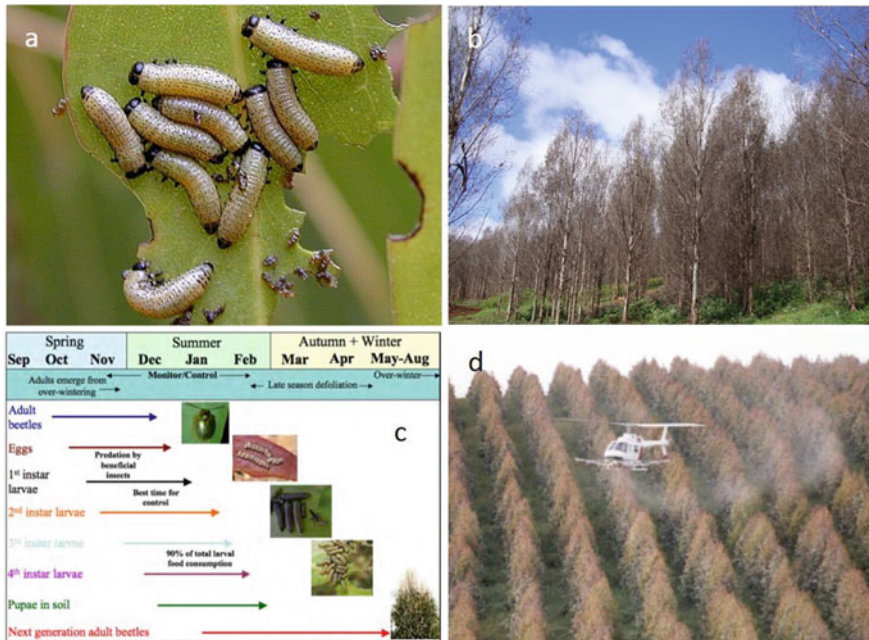


Fig. 17.17 The IPM program for leaf beetle, *Paropsisterna bimaculata*, in Tasmania, Australia: **a** *P. bimaculata* larvae feeding on a young eucalypt leaf; **b** severe defoliation of an 11-year old *Eucalyptus nitens* plantation; **c** life cycle of *P. bimaculata*, used to monitor population levels to optimize control program; and **d** aerial application of insecticides (Photo credits: T. Wardlaw and J. Elek, Forestry Tasmania)

effective when applied operationally (Wardlaw et al. 2018). Success™, one of the spinosyn group of biological insecticides, was shown to be effective against young *P. bimaculata* with no impact on non-target insects (Elek et al. 2004). Although used operationally from 2003–2011, its use progressively declined due to the higher cost and operational complexity compared to α -cypermethrin (Wardlaw et al. 2018). The IPM program for leaf beetles in Tasmania has proven to be cost beneficial over the long term (Cameron et al. 2018b; Wardlaw et al. 2018).

17.4 Summary

Insects have important roles in forest ecosystems as disturbance agents, decomposers, nutrient cyclers, and natural enemies, but a small fraction are considered pests because they compete directly with people for ecological goods and services. The goal of IPM in forests is to keep pest populations and their damage at acceptable levels using a variety of tactics that are ecologically based, cost-effective, and socially and environmentally acceptable. This requires effective tools and methods for survey and

monitoring, and an understanding of the relationships between pest populations and their impact on forest resources to determine when and where management actions are necessary. Management tactics may be preventive, e.g. regulatory controls to reduce the risk that wood used to pack shipping containers contains live wood boring beetles, or remedial, e.g. the aerial application of *Btk* or sex pheromone to slow the spread of spongy moth in North America. Ideally, IPM programs should be part of overall ecosystem management and integrated within forest management plans from the outset. Though many IPM strategies and decision support tools have been developed, their operational implementation in forests is often limited by legal, attitudinal, organizational, and financial constraints, and complicated by multiple landowners with different needs, priorities, and responsibilities. However, IPM is continuously evolving through more research, application, and adaptive management.

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