USDA Interagency Research Forum on Gypsy Moth and Other Invasive Species
January 16-19, 2001
Loews Annapolis Hotel, Annapolis, Maryland

AGENDA

Tuesday Afternoon, January 16

REGISTRATION
POSTER DISPLAY SESSION I

Wednesday Morning, January 17

PLENARY SESSION .... Moderator: J. Robert Bridges, USDA-FS
Welcome
Michael McManus, USDA-FS

The Siege of Invasive Species in Midwestern Ecosystems
Robert N. Wiedenmann, Illinois Natural History Survey

The Brown Spruce Longhorn Beetle in Halifax: Pest Status and Preliminary Results of Research
Jon Sweeney, Natural Resources Canada

PLENARY SESSION .... Moderator: Robert Mangold, USDA-FS

The National Council on Invasive Species
Lori Williams, Department of the Interior

A Multi-year Project to Detect, Monitor, and Predict Forest Defoliator Outbreaks in Central Siberia
Max McFadden, The Heron Group, LLC

Wednesday Afternoon, January 17

GENERAL SESSION .... Moderator: Cynthia D. Huebner, USDA-FS
Invasive Plants: Organismal Traits, Population Dynamics, and Ecosystem Impacts
Presenters: E. Nilsen, Virginia Polytechnic Institute & State University; D. Gorchov, Miami University of Ohio; F. Wei, State University of New York at Stonybrook; K. Britton, USDA-FS; C. D'Antonio, University of California at Berkeley

GENERAL SESSION .... Moderator: Kathleen Shields, USDA-FS
Research Reports
Presenters: J. Colbert, USDA-FS; J. Elkinton, University of Massachusetts; J. Cavey, USDA-APHIS

POSTER DISPLAY SESSION II

Thursday Morning, January 18

GENERAL SESSION .... Moderator: Victor Mastro, USDA-APHIS
Asian Longhorned Beetle
Presenters: M. Stefan, USDA-APHIS; D. Nowak, USDA-FS; S. Teale, SUNY College of Environmental Science and Forestry; B. Wang, USDA-APHIS; R. Mack, USDA-APHIS

GENERAL SESSION .... Moderator: Kevin Thorpe, USDA-ARS
Research Reports
Presenters: S. Frankel, USDA-FS; B. Geils, USDA-FS; D. Gray, Natural Resources Canada

Thursday Afternoon, January 18

GENERAL SESSION .... Moderator: Vincent D'Amico, USDA-FS
Gypsy Moth in the Midwest
Presenters: D. McCullough, Michigan State University; A. Liebhold, USDA-FS; W. Kauffman, USDA-APHIS; A. Diss, Wisconsin Department of Natural Resources; L. Solter, Illinois Natural History Survey; K. Raffa, University of Wisconsin

GENERAL SESSION .... Moderator: Vincent D'Amico, USDA-FS
Research Reports
Presenters: B. Hrašovec, University of Zagreb, Croatia; E. Burgess, Hort-Research, Auckland, New Zealand; C. Maier, Connecticut Agricultural Experiment Station

Friday Morning, January 19

GENERAL SESSION .... Moderator: Sheila Andrus, USDA-FS
Asian Longhorned Beetle: Detection and Monitoring Panel Discussion
Panel Participants: J. Aldrich and A. Zhang, USDA-ARS; R. Haack, USDA-FS; D. Lance and B. Wang, USDA-APHIS; D. Williams, USDA-FS; S. Teale, SUNY College of Environmental Science and Forestry; M.T. Smith, USDA-ARS; K. Hoover, The Pennsylvania State University

GENERAL SESSION .... Moderator: David Lance, USDA-APHIS
Asian Longhorned Beetle: Control Options Panel Discussion
Panel Participants: V. D'Amico, USDA-FS; T. Poland and R. Haack, USDA-FS; A. Hajek, Cornell University; L. Hanks, University of Illinois at Champaign-Urbana; M. Keena, USDA-FS; B. Wang and W. McLane, USDA-APHIS; Z. Yang, Chinese Academy of Forestry; M.T. Smith, USDA-ARS

Closing Remarks

CONTENTS

ECOSYSTEM IMPACTS OF INVASIVE EXOTIC PLANTS IN WILDLANDS – CHARACTERIZING THE CULPRITS

Carla M. D'Antonio

Department of Integrative Biology, University of California, Berkeley, CA

ABSTRACT

Until the past two decades, the focus of invasive plant research was largely on agricultural and rangeland weeds because of the direct economic costs associated with crop and grazing land losses. Over the past two decades, however, the continued decline of parks and wild lands in the face of invasion by non-native species has contributed to the rising awareness and concern over species that threaten native biodiversity and ecosystem functioning including many introduced plants. While non-native plants can cause the decline of native species by directly competing with them for limiting resources, they may also alter the fundamental processes that support native species assemblages. Species that alter ecosystem processes change the rules of the game for native species and potentially shift ecosystems down new trajectories from which it may be hard to return. They therefore represent serious management and restoration challenges.

Introduced plants affect ecosystem processes if they change rates of nutrient accumulation, retention, and turnover; alter patterns of energy flow and water movement; alter geomorphology and physical structure of the site; and/or alter the disturbance regime. Many plant invaders do more than one of these things at once (e.g., grasses that promote fire cause changes in N cycling and loss). Currently, we lack a framework for reliably predicting which species will have ecosystem-scale impacts. The purpose of this talk is to present an overview of the factors that will determine which species might have ecosystem impacts and highlight the interplay of invading species with the recipient community in determining the outcome of species invasions.

The impact of a plant species might be thought of as being a function of (1) its demography (birth and death rate, dispersal characteristics, etc.); (2) its morphology and physiology (this can be thought of as the effect at the individual level: how fast does it take up nutrients, what does it do with them, how does it affect heat flux, physical habitat structure, etc.); and (3) the recipient environment. Regarding the latter, a species is unlikely to have the same effect in each environment it enters, both because the environment affects demography and physiology of a species and because of varying degrees of resistance to change by the resident biota or abiotic features of the environment.

Efforts to characterize plant invaders in terms of their demographic traits date back to the 1960s when ecologist Herbert Baker wrote a now classic article, "On the modes and origins of weeds." But Baker did not distinguish high impact from low impact invaders and for the most part he focused on agricultural weeds, many of which do not have impacts in wildlands.

Since Baker's time, efforts to find general traits to explain "invasiveness" have not been fruitful. Perhaps this is because the term "invasiveness" itself is ambiguous and is generally defined as the ability to increase when rare; almost any species can do this under at least some circumstances. More fruitful efforts to characterize traits of invaders have focused on prediction within particular groups, such as *trees* (Reichard and Hamilton 1997) or specific genera such as *pines* (*Pinus*) (Rejmanek and Richardson 1996). These authors distinguish species either as ones that tend to spread or do not tend to spread from planted sites. A consistent finding is that successful invaders have (1) fairly fast growth through the juvenile period, (2) fewer years between large seed crops, and (3) generally smaller seeds than non-invaders within their category.

While these studies give us some idea of what it takes to be an invader for trees, neither of the studies distinguish high- from low-impact invaders. It may be that there are no unique demographic traits associated with high-impact species, but this has not been specifically addressed. In California, there are an estimated 1,050 introduced plant species with self-replacing populations, yet only a small fraction of them actually cause measurable change in some feature of wildland ecosystems. The California Exotic Pest Plant Council (CalEPPC) concluded that 78 of these species merit listing as invasive and damaging in wildlands and an additional 50+ merit further study. We can consider these to be "high-impact" invaders. An interesting feature of this group that relates to their demography is that 78% of them were purposefully (as opposed to accidentally) introduced to California or the western USA. This is also true in Hawaii where 91% of the 107 introduced plant species considered a threat to natural areas were also purposefully introduced species. This stands in contrast to damaging introduced insects which largely come in as accidental introductions (an OTA report estimates 81% are accidentals). It also contrasts with the proportion of purposefully versus accidentally introduced plant species in the general flora (for example, in Hawaii only 58% of all of the established plant invaders were purposefully introduced.). Hence it seems that there is a disproportionate representation of purposefully introduced species in the "high-impact" group. This suggests that these high-impact species are not necessarily species with good long-distance dispersal (which is a good thing in terms of control) and because most of them are perennial, they are not likely to be ruderal species. In other words, local *persistence* is part of what these species are all about. In addition to being perennial, most of these species maintain large seed banks, have a wide tolerance for environmental conditions once established, and show rapid recovery after disturbance. There are several high-impact invaders on the CalEPPC list that are annual species and were accidental imports. Notable examples include yellow star thistle (*Centaurea solsticialis*) and cheatgrass (*Bromus tectorum*). Their persistence in part comes from large seed banks and wide environmental tolerance but it may also come in part from positive feedbacks that they generate (e.g., cheatgrass/fire cycles). Mechanisms promoting their persistence merit further study.

My field research has focused on two groups of high-impact invaders that I believe have some of the most long-term ecosystem effects: (1) those that alter N accumulation and cycling, and (2) those that affect disturbance regime. Among ecosystem ecologists, there is a great deal of work looking at how species traits (physiology/morphology) such as leaf litter chemistry, rates of N uptake, timing of uptake, and allocation affect rates of N turnover and retention. A question of great interest is how different does an invader have to be from resident species to have an impact on rates of cycling, retention, or accumulation? With my former PhD student Michelle Mack, we investigated the effect of introduced perennial bunchgrasses on ecosystem N cycling in seasonally dry Hawaii woodlands which previously lacked a perennial grass component. These grasses are different from natives in their nutrient use efficiency and their leaf tissue chemistry. Some of these characteristics, such as lignin content and C:N ratio, are known to have a strong control over rates of litter decomposition. Hence we predicted that these grasses should cause a change in ecosystem N cycling. Yet we found that despite being in these sites for 25+ years, they are not having a measurable effect on soil nutrient cycling. We believe this is because their major interaction with native species is a competitive one—they compete for the same pool of limiting resources and are not different enough in traits that influence N cycling to create an effect despite their strong superficial differences. It may also be that it will take another decade for effects to develop because the soils in this area contain a large amount of organic carbon and it may take a long time for an invader to dilute this pool if the natives are still in the system. We suggest that species that are not discretely different from natives in their traits and interact with them largely through competition may take a long time to show ecosystem process changes.

By contrast, species such as the N fixing shrub, *Myrica faya*, which is invading these same sites, are very different from most of the species present in terms of litter chemistry and can much more rapidly alter soil N accumlation and cycling in these N-limited communities. Peter Vitousek found that *Myrica faya*, first introduced to Hawaii as a forestry tree and now widely invasive in mesic and seasonally dry forests and shrublands, can add up to 4 times as much N per year as all native sources combined. It also substantially elevates rates of N mineralization. *Myrica* does something qualitatively different from the other species present and this characteristic—fixing atmospheric N—makes its litter contributions substantially different in their chemistry from native species. In other words, *Myrica* has no functional analog relative to N cycling in the invaded system and most likely because of that it has a rapid and dramatic impact on ecosystem processes. In California, five out of seven shrubs on the CalEPPC "A" list are nitrogen fixers. Karen Haubensak, a graduate student in my lab, has documented that two of these species have a strong effect on soil N, but this effect seems dependent on the abundance of the invader and perhaps the duration of presence on the site. These shrubs are invading sites with native N fixers but these tend to be species with small stature. So these N fixing shrub invaders again appear to have no functional analog in the invaded sites. Elevated soil N has a long-term effect on community development. N fixing invaders are frequently the target of removal efforts or insect attack. A variety of investigators, including people in my lab, have demonstrated that N left over from these N fixers can have a long-term effect on community development by favoring fast-growing exotic species.

A second difficult-to-reverse impact appears to occur with species that alter disturbance regime. In the western U.S. and Hawaii, there are numerous introduced plant species that alter the frequency and intensity of fire (list presented). In general, alterations to fire regime are due to changes in the continuity of the fuel bed due to the way in which grasses fill in the space between woody species. In ecosystems where native species are not adapted to intense or frequent fire, the effects of this alteration on native species composition and ecosystem structure have been very dramatic such as in Hawaii where I have been working. However, the impacts of grass-fueled fires are dependent upon the recipient environment. We found that in the coastal lowlands of Hawaii Volcanoes National Park, alien grass-fueled fire does not have a negative impact on cover of native species. We concluded that this was because the native species in these sites are fire tolerant whereas in higher elevation (submontane) sites, different native species are present and they are not fire tolerant. Grass/fire cycles can be very difficult to reverse due to the inevitability of fire in human-dominated landscapes, feedbacks from grasses to microclimate, and the ability of these grasses to regenerate quickly and suppress the recruitment of woody species either directly or because of high fire frequency.

Often traits that enhance the spread of fire may not be initially obvious. For example, *Tamarix*, or saltcedar, is a fast-growing, salt-tolerant tree that has invaded naturally saline river courses throughout the American West. Its salty litter and foliage would not appear to be fire promoting. Yet in a study of fire in riparian habitats in parts of the lower Colorado River and nearby drainages, D. Busch found a high, and increasing, frequency of fire in areas dominated by *Tamarix*. He accounts part of the apparent increasing fire frequency to the accumulation of dead litter of native species because *Tamarix*, a species highly consumptive of groundwater, gradually reduces available water in the system and slowly outcompetes or adds stress to life for native species. In this case, the interplay of *Tamarix* with the native community adds more fuel to the impending fire. These examples highlight that in order to really predict the impact a species will have, you need to know about the potential recipient environments and how native species in these sites will interact with the invader and respond to the altered ecosystem processes that will occur once the invader is established.

So do we simply throw up our hands and say it is close to impossible to predict which species will have ecosystem impacts because we need to know too much about both the species and the potential environments? I believe that we need to rely on empirical evidence to guide decision making. We know, for example, that N fixers have tremendous potential to impact sites. We should carefully review introduced N fixers in the horticultural trade and look for those species with traits that will promote spread and persistence using known high-impact invaders as a point of reference. Since we already know who many of these invaders are, we should reduce their abundance in the horticultural trade in states where they are a risk and look carefully at species that share many traits with them but might not be perceived as a threat yet. Likewise, we already know that many introduced grasses can cause difficult-to-reverse, ecosystem-level change. We need careful controls over further introductions of non-native grasses and better coordination among agencies so that agencies that promote outplanting of grasses are not in direct conflict with conservation groups.

I conclude by saying that our search for a general framework for predicting species impacts is still in an early stage; it may be that no such general framework will emerge that works for all plant species. Nonetheless, data on wildland weeds and *the conditions under which they create impact* is essential to helping managers identify areas most at risk from invader impacts and species that should be prioritized for control or removal. While it is difficult to remain optimistic in the face of dramatic changes in the composition and function of our forests and wildlands, it is important for research biologists to interact with managers, policy makers, and stake holders to help identify areas at risk, identify species that pose the greatest threats, and work in teams at a regional scale to find creative solutions to control, removal, and/or restoration.

LITERATURE CITED

Reichard, S.; Hamilton, C.W. 1997. **Predicting invasions of woody plants introduced into North America**. Conservation Biology. 11: 193-203.

Rejmanek, M.; Richardson, D. 1996. **What attributes make some plant species more invasive?** Ecology. 77: 1655-1661.

ASSISTED AND NATURAL SPREAD OF PARASITOIDS AND DISEASES IN NEWLY ESTABLISHED POPULATIONS OF GYPSY MOTH: THE WISCONSIN EXPERIENCE

Andrea Diss

Wisconsin Department of Natural Resources, 101 S. Webster St., Madison, WI 53703

ABSTRACT

Gypsy moth was isolated from the nearest long-established population in Michigan for years following establishment in Wisconsin. Given this isolation, we were concerned that natural spread of specialist enemies of this foreign pest might be slowed. We decided to conduct an initial survey of natural enemies of gypsy moth and implement a biological control introduction program.

In the summers of 1996 and 1997, workers from the Department of Natural Resources placed burlap bands around preferred host trees at sites with relatively higher catches of male moths. Sites were checked twice weekly starting the first week of June through the third week of July. This corresponded to the period of third instar larvae through pupation. Larvae found under the bands were sent to a rearing facility where the cause of death of any that died prematurely was determined. When gypsy moth egg masses were found during population monitoring, we checked them for the presence of parasitoids.

We found *Entomophaga maimaiga* at one site in the Door Peninsula. Nucleopolyhedrosis virus (NPV) and *Cotesia melanoscela* were present at a site near the southern end of Green Bay. Dipteran parasitoids were present throughout the range of gypsy moth and some of these were identified as *Compsilura concinnata*. *Ooencyrtus kuvanae* was present at five sites in the base of the Door Peninsula.

Following the survey of parasitoids and diseases already present, we started an introduction program to supplement natural spread. We used three criteria in selecting candidate species: a high degree of specificity to the gypsy moth, the ability to establish in low-density gypsy moth populations and to survive in our climate, and availability. Using these criteria, we selected *E. maimaiga, O. kuvanae*, and *C. melanoscela* for release. Introductions were made throughout the area where gypsy moth was established in Wisconsin for four years.

We introduced *E. maimaiga* using ground cadavers. Releases were done in September, October, and November. By using cadavers, we expected to provide a more consistent distribution of spores and a higher concentration. We also avoided the possibility of introducing soil-borne pathogens. We introduced *O. kuvanae* as adults collected off of infested gypsy moth egg masses supplied from wild populations in Michigan. Release of the

early fall generation of this species was emphasized to maximize the proportion of females. Captive bred *C. melanoscela* were released in late fall as overwintering cocoons as previous studies had shown this reduced mortality from parasitism. Despite this, emergence was an average of only 62%.

Post-release surveys were done using the same procedure as before. Sites were surveyed each year following an introduction through the summer of 2000. *E. maimaiga* was recovered at approximately 50% of its sites. Recovery appeared to be influenced by rainfall; natural spread also occurred into central Wisconsin in 2000. *O. kuvanae* was recovered at about 35% of the sites where we released it. Unassisted spread of this species has been rapid, however. We have not yet recovered *C. melanoscela* from sites where it was released or at other sites.

The spread of *E. maimaiga* appears to have been assisted by our introductions. Natural spread, however, will play a larger role during wet years now that gypsy moth is more generally established in the eastern areas of the state. This species is a good choice for introductions into newly established and isolated populations of gypsy moth. The risk of effects on non-targets is small, it can persist for many years, and while it can be difficult to make collections, it is relatively cheap. We would recommend to pest managers that the establishment rate might be improved by distributing spores soon after collection in mid summer to avoid desiccation. Variation in the likelihood of recovery emphasizes the need to conduct post-release surveys for several years.

The spread of *Ooencyrtus* was not greatly aided by our introductions. Most populations found during surveys were not associated with introductions. In terms of public relations though, introductions of this species were a success. *Ooencyrtus* is an easy species for the public to find when gypsy moth is at moderate levels, so it makes a good ambassador species for the concept of biological control. It is also easy to obtain, cheap, and fairly specific to gypsy moth when collected from wild populations.

We can't say how successful our introductions of *Cotesia* were due to limited efficacy of the burlap band survey technique at recovering this parasitoid. Simple searches of young larvae at release sites also failed to find evidence of this species, however. Pest managers might consider the ability to recover a species when selecting them because a lack of results could be a problem in maintaining support for a biological control program.

In order to improve biological control introduction programs, pest managers need more information on which natural enemies benefit from deliberate introductions, particularly when a species is expensive or difficult to obtain. More information on how to optimize conditions for establishment of each species would also be helpful.

CONTROL OF *ANOPLOPHORA GLABRIPENNIS* WITH ENTOMOPATHOGENIC FUNGI

Thomas L. M. Dubois[1], Hu Jiafu[2], Zengzhi Li[2], Meizhen Fan[2], and Ann E. Hajek[1]

[1]Department of Entomology, Cornell University, Ithaca, NY 14853-0901

[2]Anhui Agricultural University, Hefei, Anhui, P.R. China

ABSTRACT

Two species of entomopathogenic fungi applied in two different ways for control of *Anoplophora glabripennis* adults were compared during field trials near Wuhe, Anhui Province, China, in July 2000. A strain of *Beauveria bassiana* originally isolated from beetles and presently marketed by Mycotech (Butte, MT) and a strain of *Beauveria brongniartii* isolated from a cerambycid in Japan and marketed by Nitto Denko (Osaka, Japan) were used. For application, use of non-woven fiber bands impregnated with fungal cultures and attached around tree trunks was compared with spraying comparable doses of fungal spores onto tree trunks. Thirty poplar trees were used for each treatment with an additional 30 untreated trees as controls. For each study tree, five field-collected *A. glabripennis* adults were placed within a cage made of window screening, 70 cm in length and wider than the diameter of the tree so that beetles could move freely, placed 1.5 to 2 m up the tree trunk (= 750 total adult ALB were used for this study). Beetles were counted and provided with food and water daily for 10 days, after which adults were removed and monitored further in individual cups. Fungal treatments always differed from controls. Oviposition rate per female and the number of oviposition sites per tree were significantly lower for cages where *B. brongniartii* was applied compared with *B. bassiana*, although results for *B. brongniartii* did not differ by application method. Time to 50% mortality (LT_{50}) was significantly lower for cages where *B. brongniartii* was applied compared with *B. bassiana*. Once again, the method that *B. brongniartii* was applied did not yield statistically significantly differences although we saw a trend of lower LT_{50}s for bands (7.8 days) than for sprays (9.2 days).

Laboratory bioassays to compare additional fungal strains were conducted both with adult ALB from the colony in the USDA quarantine in Ithaca, NY, and with field-collected adult ALB in China. Ithaca bioassays comparing two isolates of *B. bassiana* and two isolates of *B. brongniartii* demonstrated faster LT_{50}s for *B. brongniartii*; these results are consistent with field bioassay results. Bioassays in China compared five strains of *B. bassiana*, one strain of *B. brongniartii*, and one strain of *Metarhizium anisopliae*. LT_{50}s for adults exposed to one *B. bassiana* strain, the *B. brongniartii* (Nitto Denko), and the *M. anisopliae* strain were all < 6 days.

ASSESSING THE IMPACT OF NORWAY MAPLE ON NATURAL FOREST COMMUNITIES

Wei Fang

Department of Ecology and Evolution, State University of New York at Stony Brook, Stony Brook, NY 11794-5245

INTRODUCTION

Unprecedented high rates of biological invasion are increasingly homogenizing the earth's biota, with serious consequences both for humans and for natural communities and ecosystems (Lodge 1993, Vitousek et.al. 1997). Invasions vary in the extent to which they alter the structure, function, and dynamics of natural ecosystems (Williamson and Fitter 1996). There are very few accepted generalizations about which kinds of invasive species cause the greatest ecological effects. One of them is the "superior competitor hypothesis." It predicts that the invader that has a superior competitive ability can cause a significant effect by drawing down resources to very low levels (e.g., Huenneke and Thomson 1995, Meekins and McCarthy 1999) and may lead to competitive displacement and numerical reductions in native species. This hypothesis is applicable to many invasive species because they tend to excel in a new area where their congeners have existed (Williamson and Fitter 1996). The superior competitor hypothesis can also be readily extended to any species (native or exotic) to address a more fundamental question: how does a single species interact with its neighbors at the individual level and produce consequences at the community and ecosystem levels? Current mass invasion and the rapid community/ecosystem changes as consequences of invasion have offered a great opportunity to answer this question.

Trees are the dominant life form and comprise the majority of the biomass and productivity in forests. Canopy trees drive succession and ecosystem functions in forests. Subsequently, invasive tree species may easily have a large effect on a forest at several organizational levels. However, it is often difficult to distinguish between the effects of tree invasion and effects of prior land use. Among the few studies on invasive tree species, only Vitousek and Walker (1989) pin down the mechanism of the invasion's effect by linking physiological processes and population dynamics to ecosystem properties. None of these studies has directly tested the superior competitor hypothesis in a field experimental setting.

In this study, I test the superior competitor hypothesis by assessing the impact of *Acer platanoides* (Norway maple, NM), an invasive tree species, on natural forest communities. I hypothesize that the superior competitive ability of *A. platanoides* at the individual level causes their negative effects at the levels of community and ecosystem. I further hypothesize such superior competitive ability of *A. platanoides* adults is achieved by three mechanisms: (1) aboveground competition for light, (2) belowground competition for water and nutrients, and (3) litter effect, including possible nutrient release and/or physical effects.

A. platanoides was introduced as an ornamental shade tree from Eurasia in 1762 and is now the most planted street tree in the U.S. (Webb 1997). It has been reported that its dense canopy reduces wildflower diversity (Wyckoff and Webb 1996, Webb et al. 2000) and that it exhibits higher water, nitrogen, and phosphorus use efficiencies than its congeneric species *Acer saccharum* (Kloeppel and Abrams 1995), but no quantitative measurements were taken to assess these hypothesized effects. *A. platanoides* appears to be aggressively invading forests in the northeast U.S. and has even captured the attention of the popular media. *Acer rubrum* (red maple, RM), a congener of *A. platanoides* and a dominant canopy species in the mesic urban forest on Long Island, NY, will be compared as the representative native species in terms of effect.

The study is conducted in the nature preserves. The neighboring natural community is a typical mesic temperate deciduous forest. The dominant tree species are *Quercus rubra, Q. velutina, A. rubrum, Betula lenta, Cornus florida, Sassafras albidum*, and *Carya* sp. The shrub layer is dominated by *Viburnum acerifolium, Viburnum dentatum, Vaccinium vacillans*, and *Gaylussacia baccata*. The herb layer is carpeted by *Maianthemum canadense* and *Parthenocissus quinquefolia*, and scattered *Polygonatum biflorum, Smilacina racemosa, Smilax rotundifolia*, and *Aster divaricatus*. Under an *A. platanoides* dominated forest, however, these species are largely absent and the understory is dominated by *A. platanoides* juveniles, *Lonicera japonica*, and *Alliaria petiolata*.

METHODS AND RESULTS

Vegetation Survey and Mapping. To detect whether *A. platanoides* has significant spatial associations with particular species, either natives or exotics, I mapped all individual adult trees (DBH > 2.0cm) in a 100x50 m^2 area (100m W-E) in Muttontown, NY. The mapping area covers the highly invaded area (west) and its neighboring native forest (east). I recorded species, DBH, and the X,Y coordinates of each tree. I then divided this area into 200 5X5 m^2 plots. I recorded the density of tree seedlings (height < 0.3m) of each species in each plot and recorded the coverage of each shrub, herb, and vine species in each plot as well.

An initial inspection indicated that (1) *A. platanoides* was negatively associated with native species diversity. (2) Very few species grew under *A. platanoides* monoculture except *A. platanoides* and two other invasive exotic species. They are much less abundant towards the interior of the native forest (east of the mapping area). (3) *A. platanoides* is invading the interior of the native forest (*A. platanoides* adult density drops to nearly 0 beyond 65 m eastward, while seedlings established beyond the full 100m).

Belowground Competition Experiments. To explore the competitive effects of *A. platanoides* roots on seedling performance of *A. platanoides* (NM) and *A. rubrum* (RM), I conducted two experiments with a manipulation of trenching the treated plot to 25cm deep to remove shallow root effects. Log-transformed growth data were analyzed with ANOVA, and the survival data were analyzed with Chi-square test.

<u>Experiment 1.</u> I transplanted a total of 240 seedlings under three NM and three RM adult trees in the summer of 1999. Two of four 1x1 m^2 plots under each tree were trenched. Five seedlings of NM and RM were transplanted in each plot. The results showed that (1) RM

seedlings grew better under RM trees, while NM seedlings grew better under NM trees; (2) all seedlings survived better under NM than under RM ($P = 0.036$, n=6), while NM seedlings survived better than RM seedlings ($P<0.001$, n=6); and (3) seedlings grew better in trenched plots than control plots, more so under RM than NM, but not significant due to the high mortality of seedlings.

Experiment 2. I repeated the experiment with a total of 640 seedlings in the fall of 1999 with two additional components in the design: (1) I used two sites to see if the pattern from Muttontown was applicable in other mesic forest on LI; (2) in addition to four NM in monoculture and two RM in mixed area, I also used four NM adults in the same mixed area for each site. The results showed that (1) seedling growth did not differ significantly between the two sites ($P=0.478$, initial n=320); it would not be the priority of my future experiment to use more than one site; (2) seedling survival under two NM canopy types was very similar, but significantly lower than under RM, which strongly suggested that NM adults, not the prior land use of NM monoculture, caused the lower seedling survival; (3) NM seedlings survived better than RM ($P<0.001$), but RM seedlings grew faster if they survived ($P=0.008$); this implied certain life history trade-offs in the seedling performance; and (4) trenching (removing shallow root effects) significantly improved the growth of RM seedlings under NM, but not of NM seedlings. This suggests that the root of NM adults suppressed the native RM seedlings' growth, while did no harm to their own seedlings.

Above + Belowground + Litter Experiment. Based on what I had learned up to the spring of 2000, I set up a formal experiment to simultaneously test all three proposed mechanisms with which *A. platanoides* adults compete with understory seedlings, compared to its native congener *A. rubrum*. The performances of transplanted responding seedlings (e.g. phytometers) will measure the competitive hierarchy of the canopy species. The phytometer species are chosen according to the mapping study: four native species dominated in the mixed forest (*A. rubrum* (tree), *B. lenta* (tree), *P. quinquefolia* (vine), and *M. canadense* (herb)); three exotic species persisted under the *A. platanoides* monoculture (*A. platanoides* (tree), *L. japonica* (vine), and *Alliaria petiolata* (herb)); and one more exotic vine (*Celastrus orbiculata*, which is most invasive and destructive at many forests on Long Island and likely to be the next invader to Muttontown Preserve). All the phytometer seeds were collected from LI mesic forests (mostly from Muttontown) in 1999 and seedlings were germinated in the Stony Brook greenhouse in the spring of 2000.

The experiment tests the presence and the relative importance of the competitive effects of *A. platanoides* adults on seedling performance due to its canopy, root, and litter, in comparison to those of native *A. rubrum*. It also tests the competitive response hierarchies of eight phytometer species in various experimental conditions, which may predict the direction of community transition under the effect of *A. platanoides* invasion. Eight *A. platanoides* adults were randomly chosen in an *A. platanoides* monoculture, and another eight *A. platanoides* and eight *A. rubrum* were chosen in an adjacent mixed forest. Half of the trees had the south half of the canopy removed in May 2000. Under each tree, three 1.5mx1.5m plots were randomly located at 2m north of the trunk, one trenched, one with litter removed, and one intact. In each plot, five seedlings of each of the eight phytometer species were transplanted in May, 2000, with a total of 2,880. Their performances were recorded in late July and September of 2000.

The preliminary analysis of the seedling growth showed that (1) light is the most limiting factor under NM canopy; (2) root competition for resources is also limiting the understory; (3) litter, in general, protects understory seedlings from herbivory; and (4) different seedling species respond to experimental treatment differentially, suggesting that NM may potentially alter species composition.

CONCLUSIONS

This proposed study is one of the first field experimental studies on the mechanism of invasive tree species' effects at both community and ecosystem levels. The challenge of distinguishing the effects of invasive tree species and land use history has prevented all except one existing case to demonstrate the mechanism of the effects by tree invaders. This study will use a combination of comparative and experimental approaches to address this issue, which will greatly enhance the power of hypothesis testing and interpretation of confounded empirical patterns. Results of the study will not only offer the much needed knowledge on the effects of invasive tree species and have broad implications on predicting the effects of invasive species in general, but also will shed new light on how species interaction at the individual level has consequences at higher organizational levels, e.g., community and ecosystem.

LITERATURE CITED

Huenneke, L.F.; Thomson, J.K. 1995. **Potential interference between a threatened endemic thistle and an invasive nonnative plant**. Conservation Biology. 9(2): 416-425.

Kloeppel, B.D.; Abrams, M.D. 1995. **Ecophysiological attributes of the native *Acer saccharum* and the exotic *Acer platanoides* in urban oak forests in Pennsylvania, USA**. Tree Physiology. 15(11): 739-46.

Lodge, D.M. 1993. **Biological invasions: lessons for ecology**. Trends in Ecology and Evolution. 8:133-137.

Meekins, J.F.; McCarthy, B.C. 1999. **Competitive ability of *Alliaria petiolata* (garlic mustard, Brassicaceae), an invasive, nonindigenous forest herb**. International Journal of Plant Sciences. 160(4): 743-52.

Vitousek, P.M.; D'Antonio, C.M.; Loope, L.L.; Rejmanek, M.; Westbrooks, R. 1997. **Introduced species: a significant component of human-caused global change**. New Zealand Journal of Ecology. 21(1): 1-16.

Vitousek, P.M.; Walker, L.R. 1989. **Biological invasion by *Myrica faya* in Hawaii – plant demography, nitrogen-fixation, ecosystem effects**. Ecological Monographs. 59(3): 247-65.

Webb, S.L. 1997. ***Acer platanoides***. In: Randall, J.M.; Marinelli, J., eds. Invasive plants: weeds of the global garden. Brooklyn Botanic Garden, Inc., New York.

Webb, S.L.; Dwyer, M.; Kaunzinger, C.K.; Wyckoff, P.H. 2000. **The myth of the resilient forest: case study of the invasive Norway maple (*Acer platanoides*)**. Rhodora. 102(911): 332-354.

Williamson, M.; Fitter, A. 1996. **The varying success of invaders**. Ecology. 77(6): 1661-66.

Wyckoff, P.H.; Webb, S.L. 1996. **Understory influence of the invasive Norway maple (*Acer platanoides*)**. Bulletin of the Torrey Botanical Club. 123(3): 197-205.

THE PHYTOSANITARY ALERT SYSTEM OF THE NORTH AMERICAN PLANT PROTECTION ORGANIZATION: WWW.PESTALERT.ORG

Daniel A. Fieselmann[1], Woody D. Bailey[1], Robert Favrin[2], Raul Santibanez[3], Ian McDonell[4], and Leah C. Millar[1]

[1]USDA Animal and Plant Health Inspection Service, Plant Protection and Quarantine, Center for Plant Health Science and Technology, 1017 Main Campus Dr., Suite 2500, Raleigh, NC 27606-5202

[2] Canadian Food Inspection Agency, 3851 Fallowfield Rd., Nepean, ON K2H 8P9 Canada

[3] Comision Nacional De Sanidad Agropecuaria, Secretario Tecnico/ Particular, Guillermo Perez Valenzuela No. 127, Col. El Carmen, Coyaocan, C.P. 04100 Mexico, D.F.

[4] NAPPO, Observatory Cres. Bldg. #3, Central Experimental Farm, Ottawa, ON Canada K1AOC6

ABSTRACT

The introduction of exotic plant pests into North America has resulted in both ecological and economic costs of immense proportion. These pests include arthropods, mollusks, pathogens, and weeds. Quarantine efforts are under increasingly intense pressure from factors including globalization and the accompanying movement of people, increases in international tourism and trade, and smuggling. As a result, there is a need for more effective exclusion and detection measures to defend against increased pressure from pest invasions. A new tool for those on the front line of safeguarding North American plant resources is an information network that provides alerts about emerging pest threats. The North American Plant Protection Organization (NAPPO), which is comprised of members from Canada, Mexico, and the United States, has established the Phytosanitary Alert System. This web-based resource consolidates international pest information of significance to North America. It provides concise pest alerts enabling focused inspections on high-risk cargo, passenger baggage, and other pathways of entry.

NAPPO panel members have constructed the Phytosanitary Alert System located at www.pestalert.org. Sources of information for the System may include: the world-wide web, port of entry interception records from the NAPPO countries, journals, newsletters, databases, conferences, and personal networks, as well as international organizations and plant protection services. The website also solicits input from the scientific community and others who have information to contribute about emerging pest threats. Quality control filters will ensure that only accurate, reliable information is posted. The System will strive to focus on that which is truly important to Mexico, Canada, and the United States. In addition to posting concise pest alerts, the site displays or offers links to relevant information sources, data sheets, and photographs. Eventually, the System may offer an email capability similar to that used by ProMed to actively notify personnel. Access by hand-held device is currently available. Finally, not only will the System serve as a practical tool, it can become a valuable archive of the historical record of pest threats to North America.

TECHNOLOGIES FOR PREVENTING INTRODUCTIONS OF EXOTIC WOOD-BORING INSECTS INTO THE UNITED STATES

Mary Fleming[1], Kelli Hoover[2], Yi Fang[1], Dinesh Agrawal[1], Vic Mastro[3], Jeffrey Shield[4], Yuejin Wang[5], and Rustum Roy[1]

[1] Materials Research Lab, The Pennsylvania State University, University Park, PA 16802

[2] Department of Entomology, The Pennsylvania State University, University Park, PA 16802

[3] USDA Animal and Plant Health Inspection Service, PPQ, Methods Development Lab, Otis ANGB, MA 02542

[4] Materials Science & Engineering, University of Utah, Salt Lake City, UT 84112

[5] State Administration for Entry-Exit Inspection and Quarantine of P.R. China

ABSTRACT

The use of microwaves to destroy Asian longhorned beetles in wood used for making pallets and crates was investigated in China. Our preliminary research found that green wood is an excellent absorber of microwave energy regardless of wood type studied, whereas dry wood is a much poorer absorber, but a better transmitter, of microwave energy. Consequently, because larvae and pupae contain a high volume of water, insects are killed much more quickly in dry wood than wet wood using microwaves. Initial experiments conducted in China on 4"x4"x1" and 4"x4"x4" blocks of poplar showed that irradiation at 100% power using a 900 W microwave oven kills ALB larvae and pupae in 5 to 30 seconds in dry poplar and 3 minutes or less in wet poplar. Our preliminary data suggest that microwaves are a feasible, practical alternative for eradication of exotic wood-boring insects in wood used to construct solid wood packing materials. We are currently working on optimizing the parameters of this technology and the inclusion of chemical indicators that can be applied to wood to verify that effective treatment of the wood has occurred.

RELATIONSHIPS BETWEEN ENDOGENOUS JUVABIONE IN FRASER FIR AND BALSAM WOOLLY ADELGIDS

Glenn A. Fowler, Jie Zhang, and Fred P. Hain

Department of Entomology, North Carolina State University

ABSTRACT

Balsam woolly adelgid (*Adelges piceae* (Ratzeburg)) is an introduced pest of little concern in its native range, but devastating to Fraser fir. Balsam woolly adelgids (BWA) typically attack the trunk of Fraser fir. Loss of apical dominance usually occurs early in a BWA infestation. Old growth Fraser fir stands are virtually eliminated, but individual trees still survive. In many cases vigorous Fraser fir reproduction has replaced the old growth as seen at the top of Mt. Mitchell, NC. In central Europe, fir trees tolerate a BWA infestation by producing thick outer bark, which is an unsuitable substrate for further development. There is some indication that Fraser fir may have a similar mechanism. Juvabione, which is found in the wood and bark of Fraser fir, may also play a role in resistance to BWA. Juvabione concentrations increase through the growing season. Juvabione concentrations increase with tree age. Juvabione concentrations may be induced to increase with BWA infestation levels. Juvabione has been correlated with reduced BWA egg production. When topically applied, juvabione can eliminate BWA egg production. Additional questions to be explored are: Does juvabione within the tree influence BWA reproduction and metabolism? Can we select for trees with high juvabione levels that will be resistant to BWA?

DEVELOPMENT OF ODOR-BASED MONITORING SYSTEMS FOR EXOTIC PESTS

Joseph A. Francese, David R. Lance, and Victor C. Mastro

USDA Animal and Plant Health Inspection Service, PPQ,
Otis Plant Protection Laboratory, Bldg 1398, Otis ANGB, MA 02542-5008

ABSTRACT

The ability to rapidly detect and delimit incipient populations is critical to efforts to protect managed and natural ecosystems from invasive species. We are developing semiochemical-based monitoring technology for exotic pests that are recently established in, or pose a significant threat to, U.S. ecosystems. Current projects, with one exception, involve forest pests. Our general approach is: (1) collect and identify potential attractants (pheromones and kairomones), (2) assay them to determine their biological activity, and (3) develop lure/trap combinations that can be used in action programs.

Callidiellum rufipenne (Coleoptera: Cerambycidae) (Motschulsky) has been found in North America to infest Eastern red cedar (*Juniperus virginiensis*) and Northern white cedar (*Thuja occidentalis*). Several host-produced compounds elicit antennal responses, and this spring we plan to perform field assays to test the behavioral activity of these compounds.

Dendrolimus superans sibiricus (Lepidoptera: Lasiocampidae) Butler is an important pest of fir, spruce, and larch in its native range of northern Asia. Although it is not established in the U.S., it poses a significant threat to our coniferous forests. An attractant has been found for this pest[1], but current work involves determining the exact structure of the sex pheromone.

Copitarsia spp. (Lepidoptera: Noctuidae) (Walker) are polyphagous pests of ~40 crop plants in their native range of Mexico, Central America, and South America. We have obtained permits to import these pests and are making arrangements to collect them.

Anoplophora glabripennis (Coleoptera: Cerambycidae) (Motschulsky) is a serious pest of poplar, maple, and willow in its native range of China. In the past few years, infestations have been discovered in the New York and Chicago areas. Current research involves, among other things, the development of lures and traps (in collaboration with groups from SUNY-ESF, Syracuse, NY; USDA ARS, Beltsville, MD; and Beijing Forestry University, China) to be used in the survey and detection effort.

---

[1] Klun, J.A.; Baranchikov, Y.N.; Mastro, V.C.; Hijji, Y.; Nicholson, J.; Ragenovich, I.; Vshivkova, T.A. **A sex attractant for the Siberian moth *Dendrolimus superans sibiricus* (Lepidoptera: Lasiocampidae).** J. Entomol. Sci. 35: 158-166.

SUDDEN OAK DEATH IN COASTAL CALIFORNIA

Susan Frankel

USDA Forest Service, S&PF, Pacific Southwest Region Office,
1323 Club Dr., Vallejo, CA 94592

ABSTRACT

Thousands of tanoak (*Lithocarpus densiflorus* Hook. & Arn. Rehd.), coast live oak (*Quercus agrifolia Nee*), and California black oak (*Quercus kelloggii* Newb.) have died in coastal areas of California. This unprecedented oak die-off is believed to be caused by a previously unknown *Phytophthora* species discovered by David Rizzo, UC-Davis, in June 2000. Western oak bark beetle, Ambrosia beetles, and *Hypoxylon thourasianum* are commonly found on infected trees. Considered secondary, these organisms accelerate deterioration of the weakened trees.

The disease was named by Pavel Svhira, UC-Cooperative Extension Horticultural Advisor in Marin County in 1995 when he first reported dying tanoak in Mill Valley, CA. Affected trees show a rapid foliar color change with the entire crown changing from green to yellow to brown in a few weeks.

Sudden oak death has been found in the following six coastal counties: Marin, Sonoma, Napa, San Mateo, Santa Cruz, and Monterey. Infested areas are close to the Pacific coast with the farthest inland extent being 35 miles in Napa County. We do not know if this distribution pattern is due to temperature and humidity constraints or because it is a recent introduction.

In January 2001, it was confirmed that the unknown *Phytophthora* was genetically identical to an unknown *Phytophthora* discovered in 1993 that is causing a dieback in rhododendron in Germany and the Netherlands. The *Phytophthora* causing Sudden oak death was also isolated from rhododendron in a commercial rhododendron nursery in Santa Cruz County.

In August 2000, the California Oak Mortality Task Force was created to address Sudden oak death for the state of California. The task force goals are to provide a safe environment for the citizens in infested areas and to limit the spread of Sudden oak death. Further information on Sudden oak death may be found at the task force web site www.suddenoakdeath.org.

MALE-BIASED SEX RATIOS IN *GLYPTAPANTELES FLAVICOXI* (MARSH) (HYMENOPTERA: BRACONIDAE) PARASITIZING GYPSY MOTH (LEPIDOPTERA: LYMANTRIIDAE)

Roger W. Fuester, Kenneth S. Swan, Kyle Dunning, and Philip B. Taylor

USDA Agricultural Research Service, Beneficial Insects Introduction Research,
501 South Chapel St., Newark, DE 19713-3814

ABSTRACT

Efforts to obtain biological control of this pest have involved the importation, study, and release of parasitoids, predators, and pathogens. One of the parasitoids imported for trial against the gypsy moth was *Glyptapanteles flavicoxis* (Marsh), a gregarious larval endoparasite of the Indian gypsy moth (*Lymantria obfuscata* (Walker)). Laboratory studies in quarantine showed that *G. flavicoxis* readily attacked the gypsy moth, and releases of this species were made in the Middle Atlantic States, but did not result in its establishment. Because of its gregarious development, large numbers of this parasitoid can be reared with relatively few hosts, and the species was considered to have potential for inundative releases directed at specific gypsy moth populations.

Unfortunately, sex ratios in laboratory rearings of *G. flavicoxis* are usually male-biased. Male-biased sex ratios in populations of parasitic wasps used in biological control are undesirable, because they can prevent the establishment of introduced species or hinder commercial production of species used for augmentative control. Studies were conducted on potential factors contributing to male-biased sex ratios in laboratory rearings of the braconid endoparasitoid *Glyptapanteles flavicoxis* (Marsh) with the gypsy moth (*Lymantria dispar* (L.)) as a host. Sex determination in this wasp is arrhenotokous, a system in which fertilized (diploid) eggs give rise to female progeny, and unfertilized (haploid) eggs, male progeny. In the first experiment, we found that sex ratios did not differ among progenies of parents stored at 13 or 16°C and allowed to mate at 20 or 25°C, but that many females produced all male progeny, indicating that they had not mated. In the second experiment, females were exposed to hosts soon (0-60 min) after mating or 23-25 h later. Sex ratios were higher in progenies of females provided with a rest period than in those which were not. In a third experiment, females were allowed to mate from one to four times with a given male. Although differences between these groupings were not statistically significant, the data suggested that more than two matings might depress sex ratios of progeny. An alternative analysis with only two groupings (1-2 matings and 3-4 matings) suggested that more than two matings might be detrimental. Therefore, it is concluded that matings of this species be controlled (limited to 1 or 2) and that females should be provided with a period of repose after mating before they are offered hosts for parasitization.

REVIEW AND PROSPECT OF RESEARCH ON *ANOPLOPHORA GLABRIPENNIS* (COLEOPTERA: CERAMBYCIDAE) IN CHINA

Ruitong Gao and Guohong Li

Institute of Forest Ecology, Environment and Protection, Chinese Academy of Forestry,
Beijing 100091, China

ABSTRACT

Geographic Distribution of *Anoplophora glabripennis* in China

Anoplophora glabripennis (Motsch.) is one of the most important wood borers damaging a great number of broad-leaved tree species in China. It has broadly expanded in 24 Provinces of China, including Hebei, Shaanxi, Gansu, Ningxia, and Inner Mongolia, etc. The main tree species damaged by *A. glabripennis* are poplar (*Populus*), willow (*Salix*), elm (*Ulmus*), and maple (*Acer*).

Research History of *A. glabripennis* in China

The species was reported briefly by Liu Hechang in 1937 and named as *Melanauster glabripennis* Mots. In the 1950s, Qin Xixiang started to study and describe the life cycle in detail, biology, and control methods of *A. glabripennis* (Motsch.). In the 1960s and 1970s, only less than 10 papers were published in study of *A. glabripennis*. After the 1980s, along with the increase of poplar plantation area in China, the damage caused by *A. glabripennis* (Motsch.) became more and more serious. Therefore, the Chinese government began to pay more attention to the problem and more entomologists started to participate in studies in finding effective methods for controlling the beetle. More than 200 papers have been published during this period.

Research Status

The research on *A. glabripennis* by Ruitong Gao and Ping He showed that the adult's life span and fecundity depend on different tree species fed for replenishing nutrition. And a laboratory study on mating behavior showed that sex pheromone and vision acted together on the male beetle to find its mate. Results of research on reproductive behavior of *A. nobilis* in field conducted by Kebin Zhang showed that adult activity in one day could be divided into five periods. Moisture, sugar, and protein contents as well as sclereid contents in tree bark are important factors that influence host selection as reported by different researchers.

According to the widely distributed newspaper in China, the Guangming Daily, about 40% of the 2,330,000 ha poplar plantation in China had been damaged by the beetle. About 50,000,000 trees had been cut down from 1991 to 1993 in Ningxia province alone, and the direct economic lose was 300 millions Yuan RMB. It also caused severe damage to the ecosystems of Northwest China was destroyed severely.

Natural enemies of the beetle include *Dastarcus longulus*, *Iphiaulax imposter*, *Tetramorium caespiyum*, *Lasioseius ometes*, *L.* sp., *Proctolaelaps cossi*, *Dendrolaelaps* sp., *Paecilomyces farinosus*, *Pseudomonas alcaligenes*, *P. putida*, *Enterbacter agglomerans*, *Beauveria*

bassiana, *B. brongniartii*, and *Metarhizium anisopliae*. The woodpecker *Dendrocopos major* feeds on many larvae of insects in the Cerambycidae, Scolytidae, Buprestidae, Sesiidae, Limacodidae, and Notodontidae families.

Peaks of adult emergence and larval hatch were monitored by using development stage method, phenology method, and bait tree method, etc. Gao Ruitong (1997) determined appearance period of *Anoplophora glabripennis* adults by using *Acer negundo* as the bait tree in Henan Province. The results showed that adult elementary period in the middle 10 days of June, peak time in the last 10 days of July, and telophase period in the last 10 days of September.

The most important beetles in the *Anoplophora* genus are *A. glabripennis* and *A. nobilis*. *A. chinensis* is widespread in China and infests many tree species. *A. chinensis* mainly infests *Casuarina* spp. seriously in China. But we have never seen other reports on economic losses of damages caused by *A. chinensis*. Just a few reports about other species can be found in China.

Control Methods

Controls of both *A. glabripennis* and *A. nobilis* are based on cultural practices, with emphasis on IPM to build a healthy forest ecosystem. Other methods, such as plant inspection and quarantine, biological control, chemical control, and physical control may also be integrated into the management strategy, which aims to regulate the interaction among host plants, the beetle, and its natural enemies to minimize damage by the beetle.

Remaining Problems

A. glabripennis bionomics and its hosts records accord with those of *Apriona germari* (Hope) and *Anoplophora chinensis*, other species of longhorned beetles. Taxonomy problem of *A. glabripennis* and *A. nobilis* is as a focus along with their research work. As the description of taxonomy, adults whose scutellum and elytra have white patches of hairs are *A. glabripennis*; adults with yellowish or yellow patches are *A. nobilis*. But according to our long-term observation and crossbreeding experiment from 1998 to 2000, we have made the conclusion that the two species are at most two different subspecies within one species.

Research Prospect

Emphasizing studies on adjusts and control action of natural factors in forest ecosystem, using the interacting relations of tree-pest-natural enemy in forest ecosystem, adjusting defensive function, repair and reduced resistance of tree to high-point by resistant selection, cultivation and water as well as fertilizer management.

Selecting resistant tree species to plant, make it unsuitable for longhorned beetle to eat and develop. We think it is most important to select the tree species inhibiting eggs and young larvae to develop. Emphasizing studies on high and new technology, taking full advantage of genetic engineering technique and leading insect-resistant genes in the tree makes it resistant to the pest; protecting and utilizing natural enemy. That is to say, we need to create the advantage of subsistence and propagation of natural enemy, and should be selective to use insecticide or bio-insecticide. Researching the behavior and biology of longhorned beetle adults in more detail, the relations between longhorned beetle adult and host, and special information relations between male and female adults. This research will be the base for developing artificially synthesized bio-insecticide.

IMPACTS OF WHITE PINE BLISTER RUST

Brian W. Geils

USDA Forest Service, Southwest Forest Science Complex,
2500 South Pine Knoll Dr., Flagstaff, AZ 86001

ABSTRACT

The introduction of *Cronartium ribicola* Fisher to North America early in the 1900s began the first major forest disease outbreak American forestry seriously confronted. White pine blister rust has caused significant impacts in eastern North America, the Lake States, Pacific, and Rocky Mountain forests. Blister rust is still a damaging pathogen and important ecological factor in forests of those regions. It is now spreading in the Great Basin, Southern Rocky Mountains, and Southwestern forests, and may soon reach northern Mexico (if it has not already). Because white pines are so highly valued, their loss to blister rust and our efforts to mitigate those losses have generated incalculable impacts—socio-economic, political-administrative, ecological, scientific, and aesthetic. The history of blister rust in North America has already gone through two management cycles (eradication and integration), and I propose it is entering a third (adaptation).

Before blister rust was discovered in 1906 in Geneva, New York, forest pathologists knew of its destructive potential to American white pines (Spaulding 1922, 1929). American white pines had been planted widely across cool, moist Europe where nearly every rural family cultivated black currant. When *Cronartium ribicola* was introduced from Asia, it found both a suitable environment and the two hosts it required to complete its life cycle. The disease eventually caused European foresters to abandon the highly susceptible white pines. In America, however, forestry was white pine forestry. When the white pine of a region was exhausted, logging moved west. The timber industry had just set up in the Pacific and Inland Empire forests of western white pine and sugar pine. In the East and Lake States, replanting eastern white pine was underway using cheap and plentiful nursery stock from Europe. By the time the rust was discovered, it was already widely established. There had not been much which could be done to prevent the other catastrophic outbreak of the times, chestnut blight. But, white pines were worth saving (Pack 1933) and this pathogen had a weakness (Spaulding 1922). The first attack on blister rust consisted of quarantines (especially to keep the rust out of western North America) and eradication (first, cultivated black currant and then wild *Ribes*). Some of the first impacts of blister rust were political-administrative—passage of our first quarantine and weed laws and regulations and the first organization of agencies and cooperative federal-state programs dealing with forest pests (Maloy 1997). The laborers employed in *Ribes* eradication no doubt saw the opportunity as an economic boon (especially during the Depression when the eradication program reached its peak in employment and expenditure).

In the Eastern states, conditions were especially suitable for *Ribes* eradication to succeed and it continues to be effective for protecting young plantations (see Ostrofsky et al. 1988, Lombard and Bofinger 1999). *Ribes* was plentiful in the Lake States and could be removed with reasonable effort. Regional and landscape features, however, made the environment for the rust more variable (Van Arsdel 1961, Charlton 1963). Where conditions were not suitable for infection, *Ribes* eradication was not economically efficient (Anderson 1973). Although the Office of Blister Rust Control (BRC) was dedicated to *Ribes* eradication, the strategy was severely tested in the West (Benedict 1981). In eastern Washington, northern Idaho, and western Montana, wild *Ribes* were not only important sources of rust inoculum, they were large and abundant (Mielke 1943). Considering transportation and labor, eradication became a Herculean task, met by a determined BRC with chemicals, heavy machinery, and explosives.

Although some of the first research in biological control was undertaken, managers were mostly concerned with improving the efficiency of eradication. Accomplishments were reported as acres treated. In some areas only a few white pines survived exposure to the rust; many were salvaged. Control probably did reduce the velocity and severity of the outbreak. Research was begun on hazard rating (Van Arsdel 1961), genetic breeding (Bingham 1983), and silvicultural control. With improvements in fire suppression, reserve crews of rust busters were no longer needed. Eradication moved to areas of marginal benefit (Toko et al. 1967). The crisis for the BRC was probably the deployment of an antibiotic strategy without an adequate understanding of canker development and without controlled, replicated testing (Leaphart and Wicker 1968). Not only was the blister rust program reorganized under the Forest Service, but the entire strategy of forest disease control shifted to integrated management as a part of forest operations.

The rust outbreak and salvage created new conditions. Other forest tree species came into economic and ecological importance. Successional changes and new disturbance regimes required foresters to deal with new pests—defoliators, bark beetles, and root diseases. The rust spread into the Sierra Nevada and high-elevation pines of the Northern Rockies. Supported by new research (McDonald et al. 1991), new approaches were developed for integrating pest management into forest practice (Hagle et al. 1989). Tactics included: silviculture to reduce regeneration of *Ribes* (Moss and Wellner 1953), genetically improved planting stock, thinning and pruning (Hunt 1998), and decision-support tools (McDonald et al. 1981). The paradigm of this era was "if we can quantify the economic costs and ecological impacts of forest pests, we can implement appropriate, intensive-management policies." Unfortunately, this approach did not adequately consider "surprises" (Holling and Meffe 1996) and change in ecosystems and institutions (Gunderson et al. 1995). Although the rust in North America began with a genetic bottleneck (Hamelin et al. 2000), it seemed to be evolving (McDonald 1996). Society's demands from public forests also raised the importance of biological diversity and ecological sustainability. Forest insects and pathogens were no longer viewed as "pests," they were "natural disturbance agents."

Cronartium ribicola is an exotic, invasive species which has proven capable of instigating catastrophic ecological change. Our challenge is to design research for and adapt management for a dynamic, unpredictable environment. The issues are likely to include impacts in sensitive ecosystems and on dependent species (Hoff and Hagle 1990), bridging genetic barriers of the rust (e.g., by introduction of new Asian races), accommodating the cultivation of *Ribes* (Hummer and Sniezko 2000), and expansion of the rust into new regions (such as Mexico). Our most important tool will no longer be the hodag (developed for eradication) but communication and cooperation.

LITERATURE CITED

Anderson, R.L. 1973. **A summary of white pine blister rust research in the Lake States.** Gen. Tech. Rep. NC-6. U.S. Department of Agriculture, Forest Service, North Central Experiment Station. 12 p.

Benedict, W.V. 1981. **History of white pine blister rust control–a personal account**. FS-355. Washington, DC: U.S. Department of Agriculture, Forest Service. 47 p.

Bingham, R.T. 1983. **Blister rust resistant western white pine for the Inland Empire: the story of the first 25 years of the research and development program**. Gen. Tech. Rep. INT-146. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 45 p.

Charlton, J.W. 1963. **Relating climate to eastern white pine blister rust infection hazard.** Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Eastern Region. 38 p.

Gunderson, L.H.; Holling, C.S.; Light, S.S., eds. 1995. **Barriers and bridges to the renewal of ecosystems and institutions**. New York, NY: Columbia University Press. 593 p.

Hagle, S.K.; McDonald, G.I.; Norby, E.A. 1989. **White pine blister rust in northern Idaho and western Montana: Alternatives for integrated management**. Gen. Tech. Rep. INT-261. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 35 p.

Hamelin, R.C.; Hunt, R.S.; Geils, B.W.; Jensin, G.D.; Jacobi, V.; Lecours, N. 2000. **Barrier to gene flow between eastern and western populations of *Cronartium ribicola* in North America**. Phytopathology. 90: 1073–1078.

Hoff, R.J.; Hagle, S.K. 1990. **Diseases of whitebark pine with special emphasis on white pine blister rust**. In: Symposium on Whitebark Pine Ecosystems: Ecology and Management of a High-Mountain Resource; 1989 March 29–31; Bozeman, MT. Gen. Tech. Rep. INT-270. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 179–190.

Holling, C.S.; Meffe, G.K. 1996. **Command and control and the pathology of natural resource management**. Conservation Biology.10(2): 328–337.

Hummer, K.E.; Sniezko, R. 2000. **Introduction**. In: Ribes, Pines, and White Pine Blister Rust. Corvallis, OR. 8–10 September 1999. HortTechnology. 10(3): 514.

Hunt, R.S. 1998. **Pruning western white pine in British Columbia to reduce white pine blister rust losses: 10-year results**. Western Journal of Applied Forestry. 13(2): 60–63.

Leaphart, C.D.; Wicker, E.F. 1968. **The ineffectiveness of cycloheximide and phytoactin as chemical controls of the blister rust disease**. Plant Disease Reporter. 52: 6–10.

Lombard, K.; Bofinger, J. 1999. **White pine blister rust, *Cronartium ribicola*, infestation incidence for selected areas of New Hampshire**. Durham, NH: Department of Resources and Economic Development, NH Division of Forests and Lands.

Maloy, O.C. 1997. **White pine blister rust control in North America: a case history.** Annual Review of Phytopathology. 35: 87–109.

McDonald, G.I. 1996. **Ecotypes of blister rust and management of sugar pine in California.** In: Kinloch, B.B., Jr,; Morosy, M.; Huddleston, M.E., eds. Sugar Pine: Status, values, and roles in ecosystems: Proceedings of a symposium presented by the California Sugar Pine Management Committee. Publication 3362. Davis CA: University of California, Division of Agriculture and Natural Resources: 137–147.

McDonald, G.I.; Hoff, R.J.; Bingham, R.T. 1991. **History and accomplishments of white pine blister rust research in the USDA Forest Service.** In: Proceedings IUFRO Rusts of Pine Working Party Conference. 1989 September 18–22; Banff, Alberta. Info. Rep. NOR-X-317. Forestry Canada. Northern Forestry Centre: 45–53.

McDonald, G.I.; Hoff, R.J.; Wykoff, W. 1981. **Computer simulation of white pine blister rust epidemics.** I. Model formulation. Res. Pap. INT-258. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 136 p.

Mielke, J.L. 1943. **White pine blister rust in western North America.** School of Forestry Bulletin. 52. New Haven, CT: Yale Univ. 155 p.

Moss, V.D.; Wellner, C.A. 1953. **Aiding blister rust control by silvicultural measures in the western white pine type.** Circular 919. Washington, DC: U.S. Department of Agriculture. 32 p.

Ostrofsky, W.D.; Rumpf, T.; Struble, D.; Bradbury, R. 1988. **Incidence of white pine blister rust in Maine after 70 years of a *Ribes* eradication program.** Plant Disease. 72: 967–970.

Pack, C.L. 1933. **White pine blister rust, a half-billion dollar menace.** Washington, DC: Charles Lathrop Pack Forestry Foundation. 14 p.

Spaulding, P. 1922. **Investigations of the white-pine blister rust.** Bulletin 957. Washington, DC: U.S. Department of Agriculture, Forest Service. 100 p.

Spaulding, P. 1929. **White-pine blister rust: a comparison of European with North American conditions.** Tech. Bull. 87. Washington, DC: U.S. Department of Agriculture. 58 p.

Toko, E.V.; Graham, D.A.; Carlson, C.E.; Ketcham, D.E. 1967. **Effects of past *Ribes* eradication on controlling white pine blister rust in northern Idaho.** Phytopathology. 57: 1010.

Van Arsdel, E.P. 1961. **Growing white pine in the Lake States to avoid blister rust.** Stn. Pap. 92. St. Paul. MN: U.S. Department of Agriculture, Forest Service Lake States Experiment Station.

ENTOMOPHAGA MAIMAIGA AT THE SOUTHERN EDGE OF THE GYPSY MOTH RANGE

Hilary Gillock and Fred P. Hain

Department of Entomology, North Carolina State University

ABSTRACT

The fungus *Entomophaga maimaiga* Humber, Shimazu & Soper was originally introduced in 1910 for gypsy moth control. However, it was first recovered from North American populations in 1989. The fungus is a very effective pathogen and very host specific. Therefore, when gypsy moth is driven to low levels by *E. maimaiga*, there may be too few hosts to maintain high levels of *E. maimaiga* in the soil. Low levels of the fungus can create a delay in the negative feedback loop. A time lag may exist between gypsy moth recovery and *E. maimaiga* recovery. Our objective was to introduce sterile gypsy moth egg masses to sites where *E. maimaiga* is present and observe what effect host augmentation has on *E. maimaiga* levels in the soil. Three sites were selected in Camden County and two in Currituck County, NC. On 13 March, 2000, soil was collected from around bases of three trees at each site and processed to extract preseason resting spore density. On 20 April, 2000, sterile gypsy moth egg masses were distributed at three different levels. On 5 July 2000 soil samples were collected again and processed to determine postseason resting spore density. We concluded that (1) host augmentation can be used successfully to increase resting spore densities in the soil; (2) high levels of egg masses are more effective than low levels; (3) host augmentation is ecologically safer and less labor intensive than other resting spore enhancement techniques; and (4) host augmentation could be incorporated into long-term management plans for selected sites.