

**Two-year Bark and Ambrosia Beetle Diversity Study at the Talladega National Forest in
the Southeastern United States**

by

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Abstract

Over two year period (March 2008 to February 2010), bark and ambrosia beetles were captured in three different trap types (intercept panel, flight intercept and pitfall) at the Talladega National Forest in Chilton, Bill and Perry Counties in Alabama. Of 85,876 total insects captured, 23,030 were *Hylastes salebrosus* Eichhoff, 21,283 were *Gnathotrichus materiarius* and 8,004 were *Ips grandicollis*. The number of root dwelling insects captured emphasized their importance in the loblolly pine ecosystem. Several beetle species were more common at predicted loblolly pine decline plots compared to predicted non-decline plots, including several ambrosia beetle species. Elevated numbers of beetle species that feed on weakened trees indicated that loblolly pine decline may be occurring in the study plots. Trees in decline are less capable of resisting insect attacks. Previous thinning that occurred at 8 of the 24 study plots during the 1990s appeared to reduce the insect numbers in declining plots. Loblolly pine decline may be managed through careful thinning.

Intercept panel traps captured many species associated with the mid-bole and higher of trees in addition to insect species associated with tree roots. Intercept panel traps captured the broadest representation of insects and appeared to be a useful trap to supplement pitfall traps in the capture of root-feeding bark beetles. Flight intercept traps captured many of the same species as intercept panel traps but captured fewer individuals of all species.

Invasive non-native plants were detected at study plots during 5 surveys conducted in 2008 and 2009, but the non-native plants observed were generally common across the region. Most non-native plant observations occurred along roads and adjoining private property did not appear to facilitate invasion. Non-native plants were more common at plots that had been thinned during the 1990s than at plots that had not been thinned at all during the rotation, affirming previous studies that found non-native plants respond to disturbance.

Biomass removal and conventional thinning was expected to take place at all the research plots predicted to have loblolly pine decline. Conventional precommercial thinning is expected to increase short-term bark and ambrosia beetle populations because of the release of attractant chemicals and increase of logging debris at the plots that would follow treatment. Insect populations following biomass removal plots would not be expected to increase to the same extent as in conventionally thinned plots because of the habitat removal and the subsequent release of remaining trees. Both biomass removal and conventional thinning treatments are expected to increase invasive plants because of the disturbances involved. Expected follow-up research will compare insect collections and non-native plant totals from before and after the treatments.

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Chapter 1

Introduction and Literature Review

1.1 General overview of diversity

Insect attacks on pines have become widespread and severe in recent years (Klenner and Arsensault, 2009). The increased insect damage has occurred as forests have become increasingly biologically stressed and homogenous in tree species composition (Nowak et al., 2008). A wide range of bark beetles are involved in this phenomenon but most research seeking the association between forest health and insect attacks have studied only bark beetles associated with widespread tree mortality and financial loss. Other bark and ambrosia beetle species are only recorded in projects surveying insect species over an area or tracing the spread of select exotic invasive species.

Ecosystem and whole plant characteristics, such as surface area and plant volume, size and frequency of habitat gaps, quality and refuge from predators and the size and shape of habitat all play a role in insect diversity (Warfe et al., 2008). Species diversity is often reduced when a habitat type is less connected (Thomas et al., 1992) and greater primary productivity has been suggested to reduce overall diversity. Local diversity within a site is been hypothesized to be proportional to predation intensity and interspecies competition may eventually result in an increased number of realized habitat niches (Paine, 1966).

Many of these broad considerations also affect bark and ambrosia beetle populations. Insect diversity is generally proportional to plant diversity and the number of habitat niches available. Schaffers et al. (2008) found that insect biodiversity on a site was most influenced by plant species present, interspecies competition and abiotic factors. Werner and Raffa (2000) observed that the populations of different insect species were potentially affected by a range of factors including the presence of a single tree species, the presence of appropriate habitat, canopy structure or land management practices. Opportunistic insects are associated with open, disturbed conditions such as gaps and thinned forests (Warriner et al., 2002). The presence of several different forest types was found to maximize the diversity of ground-dwelling beetle species (Werner and Raffa, 2000).

1.2 Bark beetle and weevil species collected in study

Bark beetles are capable of causing widespread changes within forest ecosystems leading to long term changes in forest composition and potential widespread economic damage. Tree damage occurs as galleries that insects create for feeding and reproduction disrupt the cambium and introduce pathogenic fungi. Approximately 6,000 species of sub-family Scolytinae are currently believed to exist worldwide, the majority of which primarily attack either declining or stressed trees (Dajoz, 2000). However, the chemical ecology and life histories of a relatively small group of species that cause economic damage are thoroughly studied. The vast majority of bark beetle species are poorly understood.

1.2.1 Genus *Dendroctonus*

Members of the genus *Dendroctonus* include the most economically destructive species in the subfamily Scolytinae. The southern pine beetle, *D. frontalis* Zimmermann, kills more trees than any other bark beetle species in the southeastern United States. The western pine beetle (*D. brevicomis* LaConte), mountain pine beetle (*D. ponderosae* Hopkins) and Douglas-fir pine beetle (*D. pseudotsugae* Hopkins) are additional bark beetle species associated with catastrophic economic damage and stand replacing attacks (Furniss and Carolin, 1977).

1.2.1.1 *Dendroctonus frontalis* (Zimmermann)

In the southeastern United States, southern pine beetles (*D. frontalis*) are associated with far more damage than any other species of pine beetle and, as a result, have been studied more than any other species. In their range in the United States, which includes the entire southeastern United States and extends northward to Pennsylvania and west to Arizona, they are associated with widescale economic damage (Thatcher and Berry, 2002). Their capacity to kill healthy trees over large geographic areas has important ecological consequences. The life history of southern pine beetle has been extensively documented and is understood to the extent that outbreaks can be predicted with some accuracy (Coulson, 1979). The insect averages 2-4 mm in length and is differentiated from other regional species in the genus *Dendroctonus* by the presence of a deep median groove on the frons (USDA, 1989). Additionally, unlike many bark beetles, the head of *Dendroctonus* species are clearly visible from above. Frontalin is the primary attractant pheromone for *D. frontalis* (Moser et al., 2005). Other important chemicals in the lifecycle of the species include trans-verbenol, endo-brevicommin and verbinone (Vite et al. 1985).

Temperature influences the ecology of southern pine beetles in several ways. The number of generations per year increases by one with an average increase of 1.6 to 6.3° C (Waring et al., 2009). Exposure to temperatures of -12 to -18° C induces winter mortality in forest populations of *D. frontalis* (Tran et al., 2007). At the other extreme, developmental rates of *D. frontalis* decline at temperatures higher than 32° C and larvae are killed outright beyond 43° C (Gagne et al., 1982).

Generations of land managers have attempted to control southern pine beetle. Chemical control was originally the most used control method but was often ineffective and largely discontinued after 1970. Forest management techniques are now widely employed to limit the occurrence and extent of *D. frontalis* outbreaks. It is now known that aging stands dominated by single pine species are particularly susceptible to insect attacks, therefore many land managers try to avoid creating these conditions. Removal of dead and dying material can reduce the severity of infestations (Price et al., 1998), as can maintaining vigor by managing density (Warriner et al., 2002).

1.2.1.2 *Dendroctonus terebrans* (Olivier)

Black turpentine beetles (*D. terebrans*) are relatively large (6.5-8.4 mm.) beetles that attack roots, stumps and the lower bole of trees. They range from New Hampshire, south to Florida and west to Texas (USDA, 1989). Their numbers increase following fires, windstorms and other areas where there are widespread dead and dying trees (Bennett et al, 1958).

Dendroctonus terebrans are captured year throughout the year except for winter in the southeastern United States. The species is the largest bark beetle in the southeastern United

States and is identified by the absence of a deep groove between the eyes, a longer and less dense elytral vestiture and consistent black coloration. Ethanol and turpentine are known attractants of *D. terebrans* (Fatzinger, 1985) with frontalin, exo-brevicommin, trans-pinocarveol, cis-verbenol, trans-verbenol, 1R5S-myrtanal, verbenone, myrtenol are natural pheromones (Phillips et al., 1984).

1.2.2 Genus *Hylastes*

Members of the genus *Hylastes* typically feed on, and breed in, conifer roots, lower stems and in felled trees or stumps (Blackman, 1941). They do not kill healthy trees within their native ranges, but are often associated with trees that may be either declining or dying due to other causes. This genus, in the subtribe Hylastina (Arnett et al., 2002), is morphologically similar to *Hylurgops* LeConte, which also has root- and stump-feeding members. In the southeastern United States, the most frequently encountered members of the *Hylastes* genus are *H. salebrosus* Eichhoff and *H. tenuis* Eichhoff (Wood, 1982). Another species collected in the southeast is *H. porculus* Erichson (Blackman, 1941; Wood, 1982, Miller and Rabaglia, 2009). The range of this species extends through much of the eastern United States. In abundant numbers, the species can damage roots either directly through feeding or indirectly through their role in insect-fungal associations (Erbilgin et al., 2001).

1.2.2.1 *Hylastes salebrosus* Eichhoff

At approximately 3.3-5.0 mm long, *H. salebrosus* is about 2.5 times as long as wide in both sexes (Wood, 1982). The insect has been captured on loblolly, longleaf and shortleaf pine

from Texas to Florida, and north to New Jersey (Wood, 1982). The compounds 1S alpha-pinene and ethyl alcohol have been known to work as attractants (Miller and Rebaglia, 2009).

1.2.2.2 *Hylastes tenuis* Eichhoff

Hylastes tenuis is the smaller of the two most commonly captured southern species, approximately 2.1-2.7 mm long in both sexes, and about 3 times as long as wide (Wood, 1982). The insects' range extends from Hidalgo, Mexico, north and east to New York State. With rare exceptions, it is found exclusively on pines (Wood, 1982). The species is attracted to 1S alpha-pinene and ethyl alcohol (Miller and Rebaglia, 2009).

1.2.2.3 *Hylastes porculus* Erichson

In red pine decline, *H. porculus* has been shown to vector *Leptographium* spp. (Klepzig et al., 1995). The species is observed across the eastern United States and southeast Canada (Wood, 1982). They are associated with declining pines and are known to feed in the lower roots. Attacks are initiated by females but both sexes are attracted to pine resin from wounded roots. *Hylastes porculus* is also a known prey species of the bark beetle genus *Ips* (Erbilgen and Raffa, 2001). Adults are black, approximately 5-6 mm in length and nearly hairless (Lee et al., 2007). *Hylastes porculus* is attracted to 1S alpha-pinene, ethyl alcohol (Miller and Rebaglia, 2009), ipsdienol and lanierone (Erbilgen et al., 2003).

1.2.2.4 *Hylastes opacus* Erichson

Hylastes opacus, recently introduced into North America from the palearctic region (Bridges, 1995), has spread into the eastern United States (Haack, 2006) and breeds in stumps and roots of either dead or dying pines. Adults are 2.5-3.0 mm in length, with a black body and dull elytra. The frons and vertex are closely and coarsely punctured. The pronotum is quadrate, as long as wide and constricted anteriorly. Interstriae are flat, wider than the striae and the elytral declivity have erect seta (Hoebeke, 1994). The species is attracted to volatiles emitted from wounded pines and the attraction can be enhanced through ethanol and nonanol (de Groot and Poland, 2003).

1.2.3 Genus *Ips*

While *Ips* typically attack either weakened or recently dead trees, they are associated with intermittent economic and ecological damage. *Ips* can kill healthy trees in the aftermath of either a southern pine beetle attack or when trees are weakened from drought, fire or storm damage. *Ips* species are closely associated with specific portions of the main bole, although their habitats can overlap. Up to four or five generations per year can be produced in the southeastern United States (Bennett et al., 1958). While they attack loblolly pines in the southeastern United States, *Ips* attack different tree species in other regions. Red pine (*Pinus resinosa*) with root-insect infestation were more likely to be attacked by *Ips* in the northern United States (Aukema et al., 2010). The insects are cylindrical to hemispheric in shape and range from 0.5 to 8.0 mm. *Ips* have broad elytral declivities with elevated margins. Mid and hind-tibiae are rather slender, abruptly truncate apically and armed by a few widely spaced coarse teeth. Length and the number of spines on the declivity are typically used to identify species (Bennett et al., 1958).

Ips are primarily attracted to terpenes, with pheromones having a secondary affect. Males excavate breeding tunnels, eject frass and protect the tunnel entrances. Each tunnel typically contains 2-5 mating females and galleries of *Ips* have a characteristic Y-shape. Depending on species and location, adults emerge between February and June. Activity resumes once average air temperature increases to 7 to 10° C. *Ips* fly to new habitat as individuals or in small groups in either daylight or twilight when temperatures are between 20 and 45° C (EPPO, 1996).

1.2.3.1 *Ips grandicollis* Eichhoff

Ips grandicollis typically attack either the midbole or large branches of pines. Their range extends across the eastern United States (Eickwort et al., 2006). Individuals are approximately 2.8 to 4.7 mm in length and up to 6 generations per year are produced in the southeastern United States. *Ips grandicollis* have five spines on their elytral declivity. Males attract females with ipsdienol, and with other chemical attractants including cis-verbenol and trans-verbenol (Vite et al., 1978).

1.2.3.2 *Ips calligraphus* Germar

Individuals of *I. calligraphus* are approximately 3.5 to 6.5 mm long with 6 spines on the elytral declivity. *Ips calligraphus* ranges across the eastern United States and southern Ontario (Eickwort et al., 2006). They usually attack the lower bole of the tree and are often the first *Ips* species to attack stressed trees (Conner and Wilkenson, 1983). Males attract mates through the release of cis-verbenol, trans-verbenol and ipsdienol (Renwick and Vite, 1972).

1.2.3.3 *Ips avulsus* (Eichhoff)

Ips avulsus has 4 spines on the elytral declivity and individuals range from 2.3 to 2.8 mm in length. *I. avulsus* is generally restricted to the southeastern United States, occurring no farther north than southern Pennsylvania and no more westward than Texas (USDA, 1985). The species most often attacks the upper bole of the tree and will feed on areas of trees previously attacked by other insects. This insect is often closely associated with fungal species in the genus *Leptographium* (Klepzig et al, 2001) and also can be associated with annosum root rot. Males attract females with ipsdienol (Vite et al., 1978). R-ipsdienol and S-ipsdienol are also known to also act as attractant chemicals (Miller and Asaro, 2005).

1.2.4 Weevil species

Several weevil species in the subfamily Molytinae can cause significant economic damage in pine plantations and Christmas tree plantations (Dixon and Foltz, 1990). Insects lay eggs within boles of living trees and larvae overwinter in downed trees. Once mature they feed on the stumps of dead or dying trees. Weevil feeding can kill pine seedlings through the stripping away of bark (Price, 2008). Weevils can become especially destructive when seedlings are planted immediately following a harvest. *Hylobius pales* Herbst, *Pachylobius picivorus* Germar, and to a lesser extent, *Pissodes nemorensis* Germar are associated with plantation damage. Weevil populations are known to increase during large disturbances in forests but under most conditions they are not problematic.

1.2.4.1 *Pachylobius picivorus* Germar

Pachylobius picivorus are brownish-black beetles with patches of short, flattened white or reddish hair on elytra. The species has a North American range extending as far north as

Canadian Labrador and as far west as Oklahoma. *Pachylobius picivorus* feed on numerous pine species and are attracted to stands that have been either disturbed or recently cut. Individuals are typically 7.9-11.1 mm in length. Beaks of *P. picivorus* are robust, cylindrical and feebly curved. Tibiae are short and thick with dilated outer apical angles (Dixon and Foltz, 1990). Adults feed nocturnally on the inner bark of trees and stumps but remain in leaf litter during daylight hours. Signs of *P. picivorus* feeding include irregularly shaped patches of missing bark, white, crystallized resin and the presence of dead saplings and branches (Dixon and Foltz, 1990). In Christmas tree plantations, the insect can cause up to 60% mortality (Thatcher, 1960). Ethyl alcohol and turpentine have been used to successfully lure *P. picivorus* (Rieske and Raffa 1991).

1.2.4.2 *Hylobius pales* Herbst

Adults are reddish-brown with scattered tufts of long yellow-white or grey hairs on elytra and thorax. Males are approximately 5.8 to 11.3 mm in length while females are 7.4 to 10.3 mm. Beaks are robust, cylindrical and feebly curved. *Hylobius pales* are distributed throughout the eastern United States. Host trees consist of species in the genus *Pinus* and *H. pales* are attracted to cutting and disturbance of pine stands. *Hylobius pales* have a shorter, more depressed thorax than *P. picivorus* (Dixon and Foltz, 1990).

The insect is a common pest in Christmas tree plantations and can cause up to 90% mortality (Thatcher, 1960). In the southeastern United States, *H. pales* feed all year and produce 2 generations annually. When Christmas tree plantations were infected with the fungus *Leptographium procerum*, *H. pales* was as common as *Pissodes nemorensis*, although *H. pales* was more common than *P. nemorensis* when the fungus was not detected, succeeding *P.*

nemorensis may be a better vector. Many of the same chemicals used to attract *P. picivorus* are also effective attractants of *H. pales* (Miller, 2006).

1.2.4.3 *Pissodes nemorensis* Germar

Pissodes nemorensis are typically 6 mm long, rusty-red to gray-brown weevils with characteristic whitish spots on the wing covers (USDA, 1989). They are a native insect species found throughout the eastern United States. *Pissodes nemorensis* feeds on inner bark and wood of twigs and leading terminals. Males attract females with the pheromone grandisol and grandisal, in the presence of cut wood (Landolt and Phillips, 1997). The species is attracted to a mixture of ethyl alcohol and turpentine (Rieske and Raffa, 1991) and grandisol (Phillips et al., 1984). The insect attacks pines during the winter season, particularly young trees that are growing poorly. Economic damage associated with *P. nemorensis* is primarily a function of attacks on commercially valuable ornamental plants (Dees et al., 1999) despite the species ability to vector *Fusarium circinatum* Nirenberg & O'Donnell, the causal agent of pitch canker (USDA, 1989).

Although generally associated with less damage than *H. pales* and *P. picivorus*, *P. nemorensis* can injure trees if present in sufficiently high numbers. Signs of infestation include seedling mortality, patches of peeling bark or the presence of dying branch terminals. *Pissodes nemorensis* is also a known

1.3 Ambrosia beetles

Associated with both softwoods and hardwoods, ambrosia beetles are generally restricted to dead and dying trees and often increase in numbers after disturbances. Ambrosia beetles are more likely to be polyphagous than bark beetles (Hulcr et al, 2007) and their damage is primarily associated with the staining of timber and wood products due to the fungus they transport into the tree. Many commonly observed ambrosia beetle species in the southeastern United States are exotic species (Coyle et al., 2005).

1.3.1 *Gnathotrichus materiarius* Fitch

Gnathotrichus materiarius typically ranges from 2.7 to 3.1 mm in length and has a sleek, smooth black body surface with small and shallow punctures. They are found across the eastern United States and as far west as Texas and South Dakota (Wood, 1982). The metepisternum is covered by elytra and visible only in the anterior portion (Rebaglia et al., 2006). The appearance of the insect is similar to *Monarthrum mali* Fitch, but individuals can be distinguished by the absence of a large antennal club and darker coloration (Rebaglia et al., 2006).

Males and females are equally abundant in the woods. Males initiate attacks on host trees and attract females with the pheromone sulcatol (Fleichtmann and Berisford, 2003).

Gnathotrichus materiarius is reported to be found in either the presence of dying trees or in areas that have recently been cut. It is generally not associated with significant damage. The species is most commonly associated with economic damage in Europe, where it is non-native and invasive (Flehtmann and Berisford, 2003). Attractant chemicals include alpha-pinene, ipsdienol, lanierone (Erbilgen et al. 2003) and ethyl alcohol (Miller and Rebaglia, 2009).

1.3.2 *Monarthrum mali* Fitch

Monarthrum mali an ambrosia beetle native to North America, is a generalist feeder on hardwoods through eastern North America. Individuals range from 1.8 to 2.4 mm in length and has a relatively long pronotum and an antennal club three times the length of the funicle (Rebaglia et al., 2006). This insect attacks dying or recently cut trees and the males, which colonize new host trees, are believed to attract females through long distance pheromones. Under experimental conditions colonizing males can sense the presence of ethanol (Kirkendall et al., 2008) which has been used effectively as an attractant in forests (Miller and Rebaglia, 2009).

1.3.3 *Monarthrum fasciatum* Say

Monarthrum fasciatum is shorter than *M. mali* and is distinguished from it by a pale yellow patch on the elytral declivity (Rebaglia et al., 2006). They are found from Wisconsin to Massachusetts and Texas to Florida (Wood, 1982). Range, host trees and chemical attractants are believed to be similar to *M. mali* (Kirkendall et al., 2008). The species is attracted to ethyl alcohol (Roling and Kearby, 1975).

1.3.4 *Xyleborinus saxesenii* Ratzburg

Individuals are typically 2.0-2.4 mm in length and have conical granules at ventrolateral margin that are not hooked. The elytral declivity of the species can be identified by the presence of small spines. The scutellum is conical with a depression between elytral bases.

Xyleborinus saxesenii is found on six continents (Rebaglia et al., 2006). Distinctly eastern and western populations of *X. saxesenii* are found within the United States (Atkinson et al., 1990). Collections of *X. saxesenii* have been reported to be greater in the presence of dead

trees (Campbell et al., 2008). Ethyl alcohol can be used as bait (Miller and Rebaglia, 2009) and *X. saxesenii* has been attracted to alpha-pinene and beta-pinene (Petrice et al., 2004).

1.3.5 Genus *Xyleborus*

Xyleborus is a large genus with worldwide distribution that feeds on trees in dozens of genera. Species in the genus range from 1.6 mm to 5.9 mm in length and yellowish-brown to black with extreme sexual dimorphism (Wood and Bright, 1982). *Xyleborus* species are also associated with hardwoods and usually feed on small twigs and branches (Schiefer and Bright, 2004).

1.3.5.1 *Xyleborus atratus* Eichhoff

Xyleborus atratus are approximately 3 mm in length with interstrial spines that are smaller than strial punctures on the elytral declivity. Individuals are about 2.6 times longer than wide (Atkinson et al., 1990), are sparsely setose and the pronotum has several small serrations (Rebaglia et al., 2006). Individuals have been collected across the eastern United States (Atkinson et al., 1990).

1.3.6 Genus *Xylosandrus*

Species in the genus *Xylosandrus* are characterized by a stout body shape and widely separated procoxae. Additionally, they have less elongated bodies than insect members of *Xyleborinus*. The genus contains 54 species and has a worldwide distribution (Dole et al., 2010).

1.3.6.1 *Xylosandrus crassiusculus* (Motschulsky)

This non-native species was first detected in the United States in Charleston, South Carolina in 1974 (Atkinson et al., 1988a) and now extends throughout the eastern United States to Oklahoma. The global range of *X. crassiusculus* includes southeast Asia, East Africa, India, Sri Lanka, China and Japan (Atkinson et al., 2010).

Adult females are typically 2.3 to 3.0 mm in length and have a reddish head region and dark brown to black elytra. Elytral declivities lack punctures and the surface of the declivity is granulate and dull. Females are believed to be approximately ten times more numerous than the males, which are flightless and approximately 1.5 mm long (Rebaglia et al., 2006).

Adults overwinter in galleries formed in the host tree. Females bore into twigs, branches and trunks of small hardwoods and mate within the galleries (Atkinson et al., 1988b). Flight typically occurs after air temperatures reach 21° C. Depending on air temperatures the adults emerge in early spring (Mizell and Riddle, 2004). In the southeastern United States there are typically two generations per year (Dees et al., 1999). The species has been observed to be attracted to ethyl alcohol and alpha-pinene (Pennacchio et al., 2003). *Ambrosiella* and *Fusarium* fungi have been isolated from *X. crassiusculus* (Oliver and Mannion, 2001). Infested trees sometimes display wilted leaves and show protrusions of collected wood dust.

1.3.6.2 *Xylosandrus compactus* Eichhoff

This species, native to the continent of Asia, is found throughout the southeastern United States and Hawaii (Ellis and Hodges, 2006). Weakened trees in dozens of pine and hardwood species are more susceptible to *X. compactus* and twigs are typically colonized by 1 to 20

individuals (Cognato, 2005). Males are reddish-brown and typically 0.8-0.9 mm in length while females are black and approximately 1.6-1.8 mm long. Strial setae is absent on the elytral declivity. The life cycle is approximately one month and there are several generations per year (Ellis and Hodges, 2006). Potential new host trees are occupied by females while flightless male *X. compactus* are rarely observed. The insects are attracted to ethanol (Miller and Rebaglia, 2009).

1.3.6.3 *Xylosandrus germanus* Blandford

Xylosandrus germanus attack twigs of trees and shrubs (Rebaglia et al., 2006). This non-native species originated from Asia and has a current range through the eastern United States extending westward to Texas. *Xylosandrus germanus* is attracted to over 200 host trees worldwide but the species prefers trees with a diameter of less than 6 cm.

Individuals of *X. germanus* typically have a body length of 2.0-2.3 mm. Adults are stout and cylindrical and brown to black. Strial setae on the elytral declivity are present and have clearly impressed punctures on the elytral declivity. Females overwinter in leaf litter and excavate galleries on hardwoods. Galleries are composed of entrance tunnels and brood chambers with 1-3 branch tunnels. Like *X. crassiusculus*, *X. germanus* produces “toothpick-like” expulsions of frass from trees on which they feed (Mizell and Riddle, 2004). Several hundred larvae can live in a gallery. Adult flight occurs from late March to early September and females typically fly at a height ranging from 1.2 to 1.6 meters (Weber and McPherson, 1983).

Progeny emerge from late May to late July to attack healthy trees. The insect is a vectors *Fusarium* spp. and is dispersed long-distance through the transport of packing crates,

firewood, wood products and nursery stock (Gill et al., 1998). Ethyl alcohol can attract *X. germanus* (Miller and Rebaglia, 2009). Males are flightless (Ellis and Hodges 2006).

1.3.6.4 *Xylosandrus mutilatus* (Blandford)

Xylosandrus mutilatus is the longest insect in the genus in the United States. Mature adults are over 3.0 mm long, black and have an amber-colored head capsule which is heavily concealed. Females heavily outnumber the flightless males which die after mating. Females overwinter within trees before colonizing new trees in the spring. Feeding in trees, they create galleries which have 1 to 4 cm long central galleries with short brood chambers (Coyle, 2006). This insect species is native to Asia and currently ranges throughout the eastern United States. Symptoms of infestations include pinhole sized holes in bark with light colored boring dust at the base of the trees. The species is spread long distances through nursery stock, international shipping and fire wood. *Xylosandrus mutilatus* Blandford produce one generation per year and prefers host material less than 5 cm DBH (Schiefer and Bright, 2004). Females can fly up to 2-3 km if aided by air movement. Adult individuals make dispersal flights between June and August (Schiefer and Bright, 2004). *Xylosandrus mutilatus* is associated with a wide range of hardwood species in Japan and is known to feed on *Acer rubrum* L., *A. saccharum* Marshall, *A. palmatum* Thunb., *Ostrya virginiana* (Mill.) K.Koch, *Cornus florida* L., *Fagus grandifolia* Ehrh, *Luquidambar styraciflua* L., *Liriodendron tulipifera* (L.), *Melia azedarach* L., *Pinus taeda* L. and *P. serotina* Michaux. in North America (Stone et al., 2007).

1.3.7 *Orthotomicus caelatus* Eichhoff

Orthotomicus caelatus have recurved antennal clubs. *O. caelatus* is distributed across North America as far north as Alaska (Wood, 1982). The elytral declivity is narrowly excavate with 3 pairs of spines in which the third spine is displaced mesally, not on the summit of declivital margin. Mid-and-hind-tibiae are slender and abruptly truncate apically (Rebaglia et al., 2006). The species responds to ethyl alcohol bait in traps (Miller and Rebaglia, 2009).

1.3.8 *Platypus flavicornis* Fabricius

Platypus flavivornis, a member of the subfamily Platypodinae, is frequently observed after bark beetle mass attacks and is found throughout the eastern United States (Wood and Bright, 1992). Attacks of *P. flavivornis* peak ten days after initial southern pine beetle activity and the species is thought to be attracted to volatiles released from trees more than interspecies pheromones. Sulcatol is believed to be an attractant pheromone for this species (Renwick and Vite, 1972).

1.4 Predator species

A variety of insects, mites and birds feed on bark and ambrosia beetles (Moore, 1972). Predators may specialize on a particular insect species or feed as generalists (Reeve et al., 2009). Predator insect species generally detect and are drawn to the attractant chemicals issued by their prey and may also be attracted to chemicals released by trees after bark and ambrosia beetle attacks (Mizell et al., 1984). Predators can affect bark beetle populations, although many studies have emphasized that bark beetle population numbers are more influenced by habitat availability than predation (Reeve, 1997).

1.4.1 *Thanasimus dubius* Fabricius

The best understood predator beetle species in the southeastern United States is *T. dubius*, which feeds on southern pine beetle (Berisford, 1980). *T. dubius* has a United States range west to Wisconsin and Texas (Reeve, 2000). Known as the checkered beetle, this insect feeds on southern pine beetles but are unable to regulate populations during southern pine beetle outbreaks. *Thanasimus dubius* eggs take 7 days to hatch and have a larval stage of 42 days in laboratory rearings. Male adult longevity is about 50 days (Lawson and Morgan, 1992). Individuals are attracted to frontalin and alpha-pinene associated with the aggregation pheromones of their prey species (Aukema and Raffa, 2005).

1.4.2 *Temnochila virescens* Fabricius

Along with *T. dubius*, this metallic green predator insect has been observed to reduce prey populations (Aukema and Raffa, 2004). *T. virescens* has been captured across the eastern United States (Reeve et al., 2009). Eggs of *T. virescens* take 9 days to hatch and adult life span is approximately 230 days (Lawson and Morgan, 1992). *Temnochila virescens* is more strongly attracted to *Ips* pheromones than chemical attractants of other species (Billings and Cameron, 1984). Exo-brevacomin is also known to be an attractant chemical (Bedard et al., 1969).

1.5 Interactions between beetle species and host trees

The overall health of trees significantly affects their resilience against insect attacks. The effectiveness of pine defenses against bark beetles is a function of the biological characteristics of the tree species, on-site competition and abiotic factors that in turn influence both the numbers and diversity of insects. Aside from physical defenses such as bark, trees rely on chemical

defenses for protection. Chemical repellents and, oleoresins, keep insects away from trees and flush insects out of trees respectively (Hanover, 1975).

Widespread planting of commercially important trees such as loblolly pine (*Pinus taeda* L.) has increased the range of insects that feed on loblolly pine, notably southern pine beetle (Fettig et al., 2007). Bark beetles usually become more widespread as the proportion of pines in a forest increases (Schowalter and Turchin, 1993). Because of this, *D. frontalis* is generally less destructive in landscapes where host trees are interspersed with non-host trees (Price, 1997). Bark beetle populations are also influenced by the health of the host tree species. Bark beetles are more successful following disturbances such as severe drought, wind damage or lightning strikes (Christiansen et al., 1987). Presence or absence of attractant chemicals, wind speed, turbulence, temperature, topography, tree species composition and time scale influence how effectively bark beetles can detect and infect pines (Byers, 2008). Populations of bark beetles can become more influenced by concentrations of insect attractant pheromones than site conditions once outbreak conditions are achieved (Wallin and Raffa, 2004).

While many bark beetle species can live on an individual tree, most typically occupy specific parts of tree (Payne et al., 1990). The middle bole is usually the most species rich area while upper and lower boles have less insect diversity (Beresford, 1980). Bark beetles can detect the tunnels of other species while feeding and avoid the tunnels of competitors (Byres, 1989). Most bark beetles are unable to overcome healthy trees and may increase their populations when attacks by species such as *D. frontalis* or *Ips* species increase the amount of dead and dying material in the forest, which they can use as habitat.

Many species of bark and ambrosia beetles are associated with downed material and arrive after trees have died. *Pissodes nemorensis*, *H. pales* and *P. picivorus* are associated with coarse woody debris and *Ips* populations often build up in this decomposing material. Moisture content, size and wood temperature of the debris all affect its potential utility to insects (Hanula, 1993). Because of the more rapid decomposition of phloem, bark beetles which feed on phloem, must feed more quickly on downed material than ambrosia beetles. When fresh loblolly pine logs were exposed to forest insects, bark beetles appeared during the first three weeks, *X. saxesenii*, in the fourth through sixth weeks, and ambrosia beetles were the only species found after six weeks. After eight weeks, neither bark nor ambrosia beetles were present in the logs (Flectman et al., 1999).

Many non-native beetle species have been introduced to the United States and have become widely established. Between 1985 and 2000, beetles in the subfamily Scolytinae were intercepted at United States ports 6,825 times. Non-native species were most frequently transported in shipments of tile, marble, machinery, steel parts, iron ware, granite, aluminum and slate (Lee et al., 2007). The 44 non-native species of the subfamily Scolytinae found in North America in 2001 included 16 ambrosia beetles, 11 pithfeeders of hardwood twigs, 7 seed feeders of hardwoods and palms, 6 true bark beetles of conifers and 3 true bark beetles of hardwoods (Haack, 2001). *Xylosandrus germanus* was first discovered in the United States in 1932, *X. compactus* was first found in 1952 and *X. crassiusculus* was detected in 1974 (Oliver et al., 2001). Species of ambrosia beetles discovered in the southeastern United States in recent years include *X.s glabratus* in 2002, *X. mutilates* in 2002 and *X. atratus* in 1987 (Haack, 2006). Since

2000, APHIS and Forest Health Protection have coordinated efforts to track and detect non-native invasive insect species (Rebaglia et al., 2008).

1.6 Potential impacts of land management on bark beetle populations

Forest management can have widespread effects on bark beetle populations. Trees growing within thinned, less densely stock stands produced a greater quantity of defensive resin (Negron, 1998). Bark beetle populations often increase in the immediate aftermath of a thinning because of the temporary increase of debris but then decline to baseline levels. In an Alaskan study, bark beetle and wood borer populations increased the first year after harvest and burning, but populations declined within ten years (Werner, 2002). After thinning, attractant plumes are spread over a broader range, but are less concentrated and more difficult to detect than in an unthinned stand (Thistle et al., 2005). Longer term changes in microclimate associated with thinning, such as increased wind and warmer temperatures, may act to increase species diversity after openings (Hindmarch and Reid, 2001). Several types of thinning have been used to control insects. Both “cut and remove” and “cut and leave” thinning regimes have been more successful in regulating southern pine beetle (*D. frontalis*) populations than in regulating populations of other primary bark beetles of the *Dendroctonus* genus. Shelterwood management has also been observed to reduce bark beetle attacks, particularly when enhanced with the use of insecticides (Petersson, 2004).

While trees may grow faster and more effectively resist insect attack during thinning, diversity of bark beetles may increase following stand management. Thinning increases

diversity of bark beetle species through increasing breeding material for some species and diversifying potential habitat (Peltonen et al., 1998). Bark and ambrosia beetle species associated with coarse woody debris were more successful in recently cut areas compared to stands which had not been thinned in the previous 15 years (Sippola et al., 2002). Despite this, trees in thinned stands are less likely to be killed by bark beetles even when they are equally likely to be attacked (Safranyik et al., 2004b).

The effect of fire on insects is inconsistent and is often a function of habitat. Insects that live in bark, species attracted to fungi and soil-dwelling insects are attracted to recently burned areas. Scolytid beetles have been found to benefit from fire (Muona and Rutanen, 1994). Some root dwelling species such as *H. salebrosus* and *H. tenuis* along with some ambrosia beetle and weevil species (e.g. *Xyleborus pubescens* Zimmermann, *P. picivorus*) increased following fires (Sullivan et al., 2003). Spring burns were correlated with a doubling of *Ips pini* (Say) populations and a 30-90 percent increase in *T. dubius* but no increase in *I. grandicollis* numbers (Santoro et al., 2001). Fires generally resulted in increases in bark beetle populations and diversity that did not persist long term (Werner, 2002).

Trees growing in environmentally unsuitable conditions are more likely to have reduced growth. Excessively dense loblolly pine stands (basal area $\geq 37\text{m}^2/\text{ac.}$) can stress trees and have been associated with greater insect attacks. Loblolly pine is also frequently planted in less than ideal conditions because of its widespread commercial use. Considered to be primarily a bottom land or old field species, it has invaded and planted in former agricultural areas and many areas formerly dominated by longleaf (*Pinus palustris* Mill.) and shortleaf pine (*Pinus echinata* Mill.) (Bragg, 2002). On sites that have previously been used for agriculture, pine beetles can cause

substantial losses to loblolly pine. Loblolly pine that grows at a slower rate have been observed to be attacked more heavily by bark beetles. Reduced mortality of longleaf pine compared to loblolly pine during bark beetle outbreaks has traditionally been attributed to greater resin production but it has also been theorized that longleaf pine has evolved along with bark beetles and developed defenses against them (Martinson et al., 2007).

1.7 Possible role of bark beetles in forest decline and decline history at the Oakmulgee Ranger District

Forest declines are species specific, occur over a broad area and result from multiple biotic and abiotic causes. Combinations of insects, disease, stand succession and weather acting in concert have been associated with declines (Keene and Arno, 1993). Decline symptoms include sparse crowns, reduced radial growth and deterioration of fine roots and premature mortality. Trees in declining stands typically show reduced diameter growth over the previous ten years. Previously thinned stands were less likely to undergo decline (Oak and Tainter, 1988). Widespread loblolly pine decline, which has been a concern since the 1960s, increases the danger of insect outbreaks because declining trees are less resistant to insect attacks.

The Oakmulgee Ranger District of the Talladega National Forest in Alabama was established during the 1930s. During this time, loblolly pines were planted on former cropland to reduce soil erosion (Pasquill, 2008). Pine decline was originally reported within the district in 1959 and then again in 1968 and 1978 and pine decline has been monitored at the Oakmulgee Ranger District at the Talladega National Forest since 1990 (Hess et al., 2004). Analysis of site soil, weather, tree growth and crown condition at declining sites were compared with non-

declining sites (Hess et al., 1999). Soil conditions, root fungi and insects in the District are thought to be associated with loblolly pine decline (Eckhardt et al., 2007). Because of soil exhaustion resulting from years of cotton cultivation, some forests regenerated in former agricultural areas may not be restorable to their pre-settlement vigor (Balbach et al., 2008).

Premature loblolly pine mortality has been recorded since the 1950s (Roth and Peacher, 1971). Initial studies at the Oakmulgee Ranger District dismissed *Heterobasidion irregulare* Garbelotto & Otrrosina (Formerly known as *Heterobasidion annosum* (Fr.) Bref.) as a possible cause of decline, although that species as well as *Phytophthora cinnamoni* Rands were found at many of the decline sites (Roth and Peacher, 1971). Recently, stronger connections have been detected between pine decline and the fungal genus *Leptographium*, potential vector insects and their root habitat (Eckhardt et al., 2007). Several studies have investigated the roles of root feeding bark beetles, particularly in the genus *Hylastes*, in spreading pathogenic fungi that weaken southern pines and contribute to decline symptoms. Seedlings attacked by *Hylastes ater* (Paykull) were more likely to be infected with sapstain fungi (Reay et al., 2002). Outside of outbreak conditions it is believed that mortality caused by bark beetle damage occurs through damage resulting from destruction of fine roots (Farrell and Parmeter, 1989). Roots infested with *Leptographium terebrantis* S.J. Barras & T.J. Perry and *Leptographium procerum* (W.B. Kendr.) M.J. Wingf. have been associated with pine decline (Eckhardt et al., 2007) and have been isolated in the majority of roots in decline trees in a longleaf pine study (Otrrosina et al., 1999).

The widespread occurrence of premature loblolly pine mortality has inspired studies of how topography affects pine decline. Slopes greater than 20 percent and southern and

southeastern exposures have been associated with pine decline, while loblolly pine growing on level terrain and northerly aspects were associated with greater vigor (Eckhardt and Menard, 2008). Similarly, increased mortality of lodgepole pines (*Pinus contorta* Dougl) is observed on xeric south facing slopes (Biondi and Fessenden, 1999) and sugar maple stands (*A. saccharum*) growing at higher elevation were more likely to undergo decline than those at lower elevations (Drohan et al., 2002).

1.8 Invasive plant species

Numerous invasive plant species have spread throughout the southeastern United States and 20 percent of forested areas in Alabama are believed to be affected by invasive plants (Alabama Forestry Commission, 2009). According to USDA Forest Inventory Analysis data, the most widespread invasive plant species within Alabama are Japanese honeysuckle (*Lonicera japonica* Thunb.), Chinese privet (*Ligustrum sinense* Lour.), kudzu (*Pueraria Montana* (Lour.) Merr.), cogongrass (*Imperata cylindrica* (L.) P. Beauv.), Japanese climbing fern (*Lygodium japonicum* (Thunb.) SW.) and mimosa (*Albizia julibrissin* Durz.) (Loewenstein et al., 2008). Many forest management operations create disturbances and disperse invasive plant propagules, providing opportunities for invasive species to become established.

1.8.1 *Albizia julibrissin* Durz.

Known as mimosa or silk tree, which was introduced in 1745, occurs as far north as New York State, throughout the entire southeast United States and west to California (USDA, 2010). It spreads via seed through wind, water and animals (Remaley, 2005). *Albizia julibrissin* prefers full sun and is common along roadsides and riparian areas. The species is nitrogen fixing and

can affect the nutrient availability at sites. Seeds are spread via seed through wind, water and animals and often remain viable for over five years (Miller, 2002). Seeds resprout quickly when cut. Trees can be controlled by cutting at ground level when followed by herbicide treatments of the cut stump. Girdling can work as a control technique under circumstances when herbicide application is impractical. Herbicides can be applied to either the foliage or through basal bark treatments (Remaley, 2005).

1.8.2 *Imperata cylindrica* (L.) P. Beauv.

Cogongrass is considered to be one of the most potentially stand-changing invasive species in the southeastern United States. This grass forms aggressive, dense colonies which can change the plant composition once established. *Imperata cylindrica* can prevent the growth of other plants through aggressive rhizome growth and changes to the fire regime (Lippencott, 2000). The plant was accidentally introduced to the United States in 1912 and then again, when briefly, though unsuccessfully, tested as a forage crop. The current range in the United States extends from South Carolina through Texas (Rawlins et al., 2011).

1.8.3 *Lespedeza bicolor* Turcz.

Shrubby lespedeza (*L. bicolor*) is a branched, deciduous shrub which usually grows to a height of 0.9 to 3.0 meters. This plant rapidly colonizes openings (USDA, 2002) and alters soil through nitrogen fixation. Dense stands can prevent regeneration of native plants. Leaves are alternate with three elliptical leaflets, purple flowers bloom from June through September and single seeded legumes are present from August through March (Miller, 2002). This species can be controlled through mowing followed up within several months by herbicide treatments.

1.8.4 *Lespedeza cuneata* (Dumont) G. Don

Chinese lespedeza (*Lespedeza cuneata*) is native to Asia and was imported to the United States to provide soil stabilization wildlife forage and hay (Remaley, 2006). The current range of the species is primarily in the southeastern United States extending west to Texas and along the Coastal Plain to lower New England (USDA, 2010). *Lespedeza cuneata* spreads quickly and can prevent reestablishment of native grass species. It maintains its dominance through rapid establishment of an extensive seed bank. The high tannin content of the shoots renders the plant unpalatable to many foragers. Underground rhizomes make hand pulling impractical, but the weed can be control by either repeated mowing or by herbicides (Miller, 2002).

1.8.5 *Ligustrum sinense* Lour.

Chinese privet (*L. sinense*) is an invasive shrub with a current range of the entire southeastern United States north to Kentucky to New England and west to Texas (USDA, 2010). The species was initially brought to Florida as a hedge shrub in 1852. *Ligustrum sinense* has a high tolerance for air pollution and adverse soil conditions and is capable of forming dense thickets that prevent the establishment of native plants (Urbatsch, 2000). Plants can grow as tall as 10 m and spread either by seed or from root sprouts. Seeds are dispersed over long distances by birds and other wildlife. Individual plants can produce hundreds of seeds annually. Thickets of Chinese privet can be treated by combinations of cutting, mowing, burning and herbicides (Miller, 2002).

1.8.6 *Lonicera japonica* Thunb.

Japanese honeysuckle (*L. japonica*), was introduced to the United States for forage and erosion control in the mid 19th century. It is currently found through much of the contiguous United States (Bravo, 2009). A vine which climbs trees, dense infestations of *L. japonica* can kill host trees through either strangulation or by overtopping. Dense thickets of trailing vines and rhizomes can displace native competitors (Bravo, 2009). The species is spread by vines that root at the nodes, rhizomes and transport of seeds by birds and other animals. Individuals can be evergreen in the lower southeastern United States.

Lonicera japonica can be distinguished by white, tubular flowers, opposite leaves and fine hairs on the stems and leaves. Control can be via mechanical removal occupied areas in July and September followed by chemical treatment. Hand pulling is only effective with small populations (Miller, 2002).

1.8.7 *Lygodium japonicum* (Thunb.) Sw.

Also known as Japanese climbing fern, this perennial viney fern can invade open forests and right-of-ways, forming thick mats which smother small trees and can act as ladders that increase damage during prescribed fires (Munger, 2005). Late successional, forested habitats may be particularly vulnerable to climbing fern invasion (Munger, 2005). Fronds are opposite, compound and once-or-twice divided (Miller et al., 2010). Spores can be spread long distance by wind and are also often transported in pine straw (Loewenstein et al, 2008).

1.8.8 *Paspalum dilatatum* Poir.

This perennial grass species, commonly known as dallisgrass, was initially planted to provide pasture. Native to South America, the species was introduced to the United States

before 1840. It initiates growth earlier in the spring and persists longer into the fall compared to most warm season native perennial grasses (Burson et al., 2009). Rapid growth and profuse seed production enhances the ability of the species to invade. It grows well in open disturbed areas and commonly invades agricultural lands, margins of water ways and urban sites but is poor forage for both mammals and birds (Miller and Miller, 1999). The current range of this species extends from New Jersey west to Oregon (USDA, 2010).

1.8.9 *Paspalum notatum* Flueggé

Commonly known as bahiagrass, this deep-rooted, warm season perennial grass spreads through rhizomes. Initially introduced to the United States in Gainesville, Florida in 1913, the species invaded pastures and disturbed rights-of-way, forming dense mats that displaced native vegetation (Violi, 1999). Shoots are 20 to 75 cm tall and linear to elongate in shape (Miller, 2002). Bahiagrass can alter fire regimes and regrows quickly following late summer and autumn burns. Direct seeding of native vegetation by humans can enable native plants to out-compete *P. notatum* (Gordon et al., 1999).

1.8.10 *Pueraria montana* (Lour.) Merr.

Kudzu is widely distributed through the southeastern United States. The species is a trifoliolate-leaved, semi-woody, trailing or climbing perennial vine in the Fabaceae (legume) family. Vines thrive near rights-of-way, stream banks and can grow up to 30 cm per day (Miller et al, 2010). Kudzu ranges north to New York State and west to Texas. Beginning in the late nineteenth century, it was widely planted as both erosion control and as an ornamental (Everest

et al., 1991). The rapidly growing species alters soil through nitrogen fixation and through shading, can photosynthesis on trees on which they climb (Forseth and Innis, 2004).

1.9 Overall summary

Premature loblolly pine mortality has become widespread in the southeastern United States. Bark beetles and weevils are vectors of pathogenic fungi believed to contribute to decline and declining trees are less able to resist subsequent insect attacks. Given the possibility that little studied root and lower bole dwelling bark beetles may be an important component of the loblolly pine complex, better reckoning of the population of these often overlooked species is potentially valuable. This study captured the diversity of these insects over two years using three different insect trap types. Possible effects of both predicted loblolly pine decline and previous thinning operations on bark and ambrosia beetle species were tested. More bark and ambrosia beetles were expected to be captured in predicted decline areas, fewer were expected in thinned areas and most insect species were expected to have population peaks during spring. Additionally, while thinning and biomass removal are widely believed to mitigate bark beetle attacks little work has been conducted on the subsequent effects of these treatments on the spread of invasive non-native plant species. A pretreatment invasive was conducted on the basis that many regional non-natives thrive on disturbance and can capitalize on many land management treatments with the hypothesis that non-native plant species would be more common if and when follow-up surveys occurred at the research plots.

Chapter Two

Bark and Ambrosia Beetle Diversity at the Oakmulgee Ranger District

2.1 Abstract

Over a two-year period, 85,876 insects were captured with three different trap types located in the Oakmulgee Ranger District in the Talladega National Forest in Alabama. *Hylastes salebrosus* was the most commonly collected bark beetle species, while *Dendroctonus frontalis* were captured infrequently. The ambrosia beetle *Gnathotrichus materiarius* was the second most commonly collected insect species. Differences in insect collection totals were observed between predicted loblolly decline plots versus predicted non-decline plots. No differences in insect collection were found between adjacent plot pairs that differed in slope and aspect. A strong positive relationship between insect collection numbers and higher foliar transparency was observed. Bark beetle species tended to be positively correlated with one another, suggesting that pine bark beetle outbreaks may be compounded by additional bark beetle species which often go unobserved. Many of these less studied species, particularly weevils and insects in the genus *Hylastes*, have been linked with pine decline that may in turn weaken tree resistance to subsequent insect attacks. In addition to providing a picture of bark and ambrosia beetle

diversity at the Talladega National Forest, data collected over the two-year period could serve as pre-treatment data for any subsequent work performed in the plot locations.

2.2 Introduction

Insect attacks on pines have become widespread and severe in recent years (Western Forestry Leadership Coalition, 2009). In western North America alone, mountain pine beetle epidemics have killed over a million acres of forest trees (Robbins, 2008). Many bark and ambrosia beetles are poorly understood and thorough diversity studies may uncover previously obscure interactions between beetles and their host trees. It is important to fully understand bark beetles so their attacks can be either minimized or mitigated.

Most research exploring the relationships between forest health and insect attacks have concentrated on the few bark beetle species associated with widespread mortality and financial damage (Berisford, 1980). In the southeastern United States, *Dendroctonus frontalis* Zimmerman is associated with far more damage than any other species and have received more attention than other regional bark beetles. Their ability to kill healthy trees over large geographic areas has shaped forest ecology and forest management (Boyle et al., 2004). Insects within the genus *Ips* are also associated with economic damage and tree mortality. *Ips* species usually attack declining trees but can kill healthy trees during outbreaks that most often follow disturbances such as major storms or *D. frontalis* outbreaks (Erbilgen and Raffa, 2002).

Healthy trees are usually able to resist beetle attacks. In addition to physical defenses such as bark, trees employ chemical defenses such as chemical repellents and oleoresin (Hanover, 1975). When trees are stressed the effectiveness of these defenses is reduced. In particular, stressed pines produce lower resin and are less capable of flushing out bark beetles (Perkins and Matlack, 2002). Bark beetle attacks are also more successful following disturbances such as wind throw or lightning strike.

Many of the other bark and ambrosia beetle species are not associated with mortality and are rarely the direct focus of insect research. Recent studies, however, suggest they exert an influence on forest health. These beetles are capable of introducing pathogenic fungal associates into trees which may reduce tree resistance to subsequent insect attacks (Paine et al., 1997). *Hylastes tenuis* Eichhoff and *Hylastes salebrosus* Eichhoff have been observed to be more common in declining loblolly pine stands than non-declining stands (Eckhardt et al., 2007). Most studies devoted to *Hylastes* and other root and lower bole feeders have investigated their potential contribution to the pine decline complex.

Forest management practices can have powerful and widespread affect on tree resistance to insects and subsequent bark beetle populations. Excessively dense planting has been associated with greater insect attacks. Trees in thinned stands were found to be as likely to be attacked, but less likely to be killed by a mass attack (Safranyik et al., 2004b). Trees growing within thinned, less densely stocked stands, produced a greater quantity of defensive resin (Negrón, 1998), which enhances resistance to insects. Bark beetle populations increase in the immediate aftermath of thinning but decline over the long-term (Werner, 2002). This short-term increase in beetle species diversity and bark beetle populations is due to the increase of coarse

woody debris associated with thinning (Peltonen et al., 1998). Thinning may also cause changes in microclimate, such as increased wind and warmer temperatures that act to increase beetle diversity (Hindmarch and Reid, 2001). Attractant chemicals are spread over a broader range following thinnings but are less concentrated and are therefore more difficult for insects to detect (Thistle et al., 2005).

Loblolly pine is the most common commercially planted tree species in the southeastern United States and is the dominant tree species in many forests. Widespread premature mortality of these species has important ecological and economic repercussions. Trees begin to suffer decline symptoms at age 25 to 35 and die well before the expected lifespan of 120-275 years. The prospect that little studied bark and ambrosia beetles may be vectors in a disease complex make a broad insect survey in a declining area valuable. While testing possible associations between beetle capture and tree health indicators it will also be possible to test models estimating loblolly pine decline probability. Steeper slope (greater than 10%) and south and southeast facing aspect has been associated with pine decline in prior reports (Eckhardt and Menard, 2008), which would be expected to reduce insect resistance. Greater insect capture was therefore expected in predicted decline plots.

2.3 Methods and Materials

2.3.1 Plot Measurements

Bark and ambrosia beetles were trapped at twenty-four research plots located in the Oakmulgee Ranger District of the Talladega National Forest in Perry, Chilton and Bibb Counties, Alabama. Study sites, selected on the basis of stand history, slope and aspect, were

former agricultural lands converted to forest between 1973 and 1984. Each study plot included three subplots located 36.57 meters from plot center at bearings of 120, 240 and 360 degrees. Subplots were an array of circles with a radius of 7.62 meters. Plot design followed methods used in the USDA Forest Inventory and Analysis Program (Dunn, 1999).

Research plots were selected on the basis of whether or not they were in compartments in which loblolly pine decline occurred. Nine of twelve plot pairs were within compartments considered to be experiencing loblolly pine decline on the basis of topography, stand age and management. Three pairs of plots were located in stands not expected to exhibit decline symptoms. Insect collection data from both predicted decline and predicted non-decline compartments could subsequently be compared. The 24 study plots were grouped into 12 pairs and plots within a pair were separated by topographical features with one plot located in a predicted high risk area (slope >10%) and an adjacent plot in a predicted lower risk area (slope <10%). Plots were randomly selected within predicted decline or predicted non-decline plots and slopes were confirmed with a clinometer at each research plot.

One hundred latitude and longitude plot center coordinates were measured and averaged per plot within the GPS unit (Garmin GPSMAP 76Cx, Garmin International Inc., Olathe, KS) before data was recorded. GPS measurements were accurate within 10 meters. Pitfall traps, panel traps and modified flight intercept traps were placed within each of the three subplots for a total of nine traps per plot.

2.3.2 Insect trapping

Three different trap types were utilized to capture the widest range of species. Intercept panel traps (Fig. 1a) and flight intercept traps (Fig. 1b) were installed to capture flying insects and pitfall traps were placed in the ground to capture root dwelling insects. Pitfall traps were placed at the center of each subplot, panel traps were located approximately 3 meters west of the pitfall trap and flight intercept traps were placed approximately 3 meters east of the pitfall traps. Intercept panel traps (APTIV Company, Portland, Oregon) were hung approximately two meters by wire from a metal pole kept in the ground by a metal sleeve. The bucket of each panel trap was filled with approximately 45 ml of a mixture of two parts distilled water to one part antifreeze to prevent the escape of captured insects.

Flight intercept traps consisted of a plastic gallon milk jug cut open on three sides with the fourth side attached to a pole approximately 0.60 meters above the ground. A 120 ml plastic cup attached to the lip of the milk jug served as the receptacle for insects. Two 5 cm long by 2 cm diameter pine stems were placed in the cup to attract and protect insects.



Figure 2.1. (A) Intercept panel trap and (B) flight intercept trap deployed on plots at the Oakmulgee Ranger District, Talladega National Forest

Pitfall traps consisted of 10 cm by 20 cm diameter of PVC plastic drain pipe with eight entrance holes equally spaced around the circumference. The traps were buried with the entrance holes within 5 cm of ground level at each subplot. A plastic skirt was fitted around the trap to reduce the risk of flooding (Menard, 2007). Two loblolly pine stems 5 cm by 2 cm were placed in the trap base. The cap was kept loose to facilitate access. Escape of captured insects was prevented though coating with a thin layer of liquid TeflonTM (Northern Products Woonsocket, RI) each collection period. Trapped insects remained in the cup until the following collection period.

All traps were baited with two 8 ml glass vials, filled either with 90% ethanol or steam-distilled southern pine turpentine. Bait vials were replenished every two weeks during insect collections. Intercept panel traps were also refilled with the diluted antifreeze during collections. The antifreeze deterred predaceous insects and prevented insect loss from intercept panel traps. Pine stems in the flight intercept and pitfall traps were replaced every two weeks. Insect collections began March 2008 and continued biweekly through February 2010. Captured insects were taken to the Forest Health Dynamics Laboratory at Auburn University (Auburn, AL, USA), sorted and identified to species.

2.3.3 Tree measurements

2.3.3.1 Crown rating

Live crown ratio, crown density, crown transparency, crown dieback, crown position and number of sides of crown sunlight exposure were measured on each of the three subplots and at plot center in accordance with Forest Health Monitoring (FHM) procedures (Dunn, 1999). Pine

trees with DBH greater than four inches (10 cm) were evaluated. Three trees were randomly selected for resin and growth increment collection at center plot and the three subplots. Selected trees were spray painted with one pink band both above and below the tag in May 2009 to indicate leave trees in future thinning or biomass removal work so post-treatment data could be collected. Pine and hardwood basal areas were measured with a prism at the center plot of all research plots. Stem count within 7.32 meters of center plot and all subplots was determined in July 2009.

2.3.3.2 Additional tree measurements

Resin was collected from 12 trees per plot (three trees at center plot and all three subplots) in July 2009. Trees were wounded with a ½ inch arch punch (Osbourne No. 149) at a height of 1.37 m and 15 mL pre-weighed vials were hung below the wound. Vials containing resin were collected 24 hours later and weighed upon return to the Forest Health Dynamics Laboratory at Auburn University. Increment cores were collected from the selected trees during July and August 2009 and returned to the Forest Health Dynamics Laboratory where 5-and-10-year growth measurements were conducted to the nearest 0.01 mm with a Mitutoyo Digimatic (Mitutoyo Corporation, Maplewood, NJ) electronic ruler. Pine and hardwood basal area was measured with a prism.

2.4 Statistical Analysis

Statistical analysis was performed using SAS 9.1 (SAS Institute Inc., Cary, N.C.). Correlations between collected insect species and crown rating variables were obtained using Pearson correlations. Multivariate linear regression was used to determine significant

relationships between insect numbers and crown rating, stem count, growth increment and resin production. Analysis of variance was used to determine differences between groups of plots stratified by forest health characteristics. A repeated measures General Linear Model (GLM) test using Tukey-Kramer analysis to obtain least-squared means was used to identify differences in totals between plots within and outside areas of predicted loblolly pine decline.

Insect collection totals within plots with slopes greater than 10 percent were compared with collections from plots with slopes less than 10 percent with a two-sided t-test. On the basis of National Forest data (stand age, slope, aspect and prior management), nine pairs of study plots were expected to undergo loblolly pine decline and three were not. Differences in insect collection totals between the eighteen study plots located within National Forest compartments considered to be experiencing loblolly pine decline were compared to insect collections in six plots from compartments considered to be outside the area of decline. Because of the unequal number of plots, a repeated measures General Linear Model (GLM) was used to test for differences in SAS 9.1. Least-squared means were obtained using Tukey-Kramer analysis. Alpha-levels of significance for these and all subsequently mentioned testing was set at $P \geq 0.05$.

2.4.1 Stratification methods

For each of the forest health characteristics measured resin production, 5-and-10-year growth increments, stem count, pine basal area, hardwood basal area, mean diameter at breast height (DBH), mean crown ratio, mean crown density, mean crown transparency, study plots were stratified into groups of the eight lowest, eight median and eight highest. Stratification was

performed to uncover relationships in data that may be obscured (Randolph, 2006). Analysis of variation was used to test differences in insect populations between stratified groups.

2.5 Results

2.5.1 Total number of insects

Between March 2008 and February 2010, 85,876 insects were collected and identified to species. Twenty-three different bark and ambrosia beetle species and two predator beetle species (*Thanasimus dubius* Fabricius (family Cleridae) and *Temnoscheila virescens* Fabricius (family Trogossidae) were captured (Table 2.1). *Hylastes salebrosus* was the most frequently collected insect during the study. *Ips grandicollis* Eichhoff was the second most common bark beetle. Few *D. frontalis* (n=23) were collected throughout this same period. Three weevil species were identified, *Pachylobius picivorus* Germar, *Hylobius pales* Germar and *Pissodes nemorensis* Germar. *Gnathotrichus materiarius* Fitch was the most numerous ambrosia beetle recovered. *Xyleborinus saxesenii* Ratzburg, *Xylosandrus crassiusculus* (Motschulsky) and *Xylosandrus mutilatus* (Blandford) were the next three most common ambrosia beetle species collected in the traps.

Table 2.1. Total bark, ambrosia and predator beetles trapped over two years (March 2008-February 2010).

Species	Habitat role	Total capture
<i>Hylastes salebrosus</i>	Bark beetle	23,030
<i>Gnathotricus materiarius</i>	Ambrosia beetle	21,283
<i>Ips grandicollis</i>	Bark beetle	8,004
<i>Hylastes porculus</i>	Bark beetle	4,612
<i>Hylastes tenuis</i>	Bark beetle	3,927
<i>Xyleborinus saxesenii</i>	Ambrosia beetle	3,748
<i>Temnoscheila virescens</i>	Predator beetle	3,720
<i>Thanasimus dubius</i>	Predator beetle	2,848
<i>Xylosandrus crassiusculus</i>	Ambrosia beetle	2,242
<i>Hylobius pales</i>	Weevil	2,044
<i>Pachylobius picivorus</i>	Weevil	2,027
<i>Dendroctonus terebrans</i>	Bark beetle	1,994
<i>Xylosandrus mutilatus</i>	Ambrosia beetle	1,989
<i>Playpus flavicornis</i>	Ambrosia beetle	1,616
<i>Xyleborus atratus</i>	Ambrosia beetle	896
<i>Orthotomicus caelatus</i>	Bark beetle	676
<i>Pissodes nemorensis</i>	Weevil	248
<i>Monarthrum faciatum</i>	Ambrosia beetle	135
<i>Hylastes opacus</i>	Bark beetle	125
<i>Xylosandrus compactus</i>	Ambrosia beetle	111
<i>Ips avulsus</i>	Bark beetle	99
<i>Xylosandrus germanus</i>	Ambrosia beetle	51
<i>Ips calligraphus</i>	Bark beetle	25
<i>Dendroctonus frontalis</i>	Bark beetle	23

2.5.2 Correlations between insect species

Bark beetle species were closely associated with one another. *Hylastes salebrosus*, the most frequently observed species, was significantly correlated with more species than any other bark beetles captured. *Hylastes salebrosus* was positively correlated with *H. tenuis* ($r^2= 0.4222$, $p\text{-value}=0.0399$), *P. picivorus* ($r^2=0.4592$, $p\text{-value}=0.0240$), *D. terebrans* ($r^2=0.6464$, $p\text{-value}=0.0006$) and *I. grandicollis* ($r^2=0.6542$, $p\text{-value}=0.0005$) and *X. crassiusculus* ($r^2=0.4200$, $p\text{-value}=0.0006$) and *I. grandicollis* ($r^2=0.6542$, $p\text{-value}=0.0005$) and *X. crassiusculus* ($r^2=0.4200$, $p\text{-value}=0.0006$).

value=0.0140). *Ips grandicollis* was positively associated with *D. terebrans* ($r^2=0.4806$, p-value=0.0174) (Table 2.2). In addition to their significant correlation to *H. salebrosus*, *H. tenuis* was positively correlated with the ambrosia beetle *X. crassiusculus* ($r^2=0.4200$, p-value=0.0410). *Pachylobius picivorus* had a positive correlation with *D. terebrans* ($r^2=0.5882$, p-value=0.0025).

Ambrosia beetles were less related to one another although there was a positive correlation between *X. saxesenii* and *G. materiarius* ($r^2=0.4168$, p-value=0.0427). The predator beetle *T. dubius* was negatively correlated with *H. salebrosus* ($r^2=0.5770$, p-value=0.0032), *I. grandicollis* ($r^2=-0.3971$, p-value=0.0547), *X. crassiusculus* ($r^2=0.5113$, p-value=0.0107) and *X. saxesenii* ($r^2=0.4167$, p-value=0.0428). Relationships between *T. dubius* and other insect species were generally negative, an unexpected result given that *T. dubius* is a known predator of some of the other species, such as *Ips* beetles.

Table 2.2. Significant Pearson correlation results between bark and ambrosia beetles species ($\alpha=0.05$)

Species	Correlated species	R-sq.	P-value
<i>H. salebrosus</i>	<i>P. picivorus</i>	0.4592	0.0240
<i>H. salebrosus</i>	<i>D. terebrans</i>	0.6464	0.0006
<i>H. salebrosus</i>	<i>I. grandicollis</i>	0.6542	0.0005
<i>H. salebrosus</i>	<i>X. crassiusculus</i>	0.4949	0.0140
<i>H. salebrosus</i>	<i>T. dubius</i>	-0.5770	0.0032
<i>P. picivorus</i>	<i>D. terebrans</i>	0.5882	0.0025
<i>H. tenuis</i>	<i>X. crassiusculus</i>	0.4200	0.0410
<i>D. terebrans</i>	<i>I. grandicollis</i>	0.4806	0.0174
<i>X. saxesenii</i>	<i>G. materiarius</i>	0.4168	0.0427
<i>X. saxesenii</i>	<i>T. dubius</i>	0.4167	0.0428
<i>X. crassiusculus</i>	<i>T. dubius</i>	-0.5113	0.0107

2.5.3 Multivariate analysis of insect collection numbers and tree measurements

Mean crown transparency was highly correlated with insect collection totals. Higher transparency was associated with greater collections of *H. salebrosus* ($r^2=0.3465$ p-value=0.0025), *H. tenuis* ($r^2=0.2398$, p-value=0.0152), *P. picivorus* ($r^2=0.2820$, p-value=0.0076), *D. terebrans* ($r^2=0.2471$, p-value= 0.0135), *I. grandicollis* ($r^2=0.1808$, p-value= 0.0384), *G.*

materiarius ($r^2=0.3023$, $p\text{-value}=0.0054$) and *X. saxesenii* ($r^2=0.1205$, $p\text{-value}=0.0161$). Mean crown density was significant for *H. salebrosus* ($r^2=0.0668$, $p\text{-value}=0.0336$) and *X. crassiusculus* ($r^2=0.2459$, $p\text{-value}=0.0137$). Other forest health variables were significant for only single species. Ten-year growth was significant for *X. saxesenii* ($r^2=0.0690$, $p\text{-value}=0.0330$). *Pachylobius picivorus* was negatively correlated with hardwood basal area ($r^2=0.2585$, $p\text{-value}=0.0025$) and *G. materiarius* was negatively associated with mean diameter at breast height ($r^2=0.1031$, $p\text{-value}=0.0198$). There was also a strong positive association between *X. saxesenii* and mean diameter at breast height per plot ($r^2=0.5102$, $p\text{-value}<0.0001$). A full list of significant regression results is shown of Table 2.3.

Table 2.3. Multivariate analysis regression results for comparisons between bark and ambrosia beetles and forest health measurements. ($\alpha=0.05$).

Insect species	Forest health variable	Partial R-square	P-value
<i>Hylastes salebrosus</i>	Mean crown transparency	0.3465	0.0025
<i>Hylastes salebrosus</i>	Resin	0.0668	0.0336
<i>Hylastes tenuis</i>	Mean crown density	0.2398	0.0152
<i>Pachylobius picivorus</i>	Mean crown transparency	0.2820	0.0076
<i>Pachylobius picivorus</i>	Mean crown transparency	0.2585	0.0025
<i>Dendroctonus terebrans</i>	Mean crown transparency	0.2471	0.0135
<i>Ips grandicollis</i>	Mean crown transparency	0.1808	0.0384
<i>Gnathotrichus materiarius</i>	Mean crown transparency	0.3023	0.0054
<i>Gnathotrichus materiarius</i>	Mean crown density	0.2731	0.0014
<i>Gnathotrichus materiarius</i>	DBH	0.1031	0.0198
<i>Xylosandrus crassiusculus</i>	Mean crown density	0.2459	0.0137
<i>Xyleborinus saxesenii</i>	DBH	0.5102	0.001
<i>Xyleborinus saxesenii</i>	Mean crown transparency	0.1205	0.0161
<i>Xyleborinus saxesenii</i>	10-year growth	0.0690	0.0330
<i>Temnoschila virescens</i>	Mean crown density	0.1528	0.0341

2.5.4 Effects of prior thinning at research plots

Three pairs of the research plots were thinned in 1996 and two additional pairs were thinned in 1999. None of the remaining seven pairs underwent thinning. General linear model analysis was used to compare insect counts at the seven plot pairs that had never been thinned with the five plot pairs that had. Higher mean captures of *P. picivorus* (p-value= 0.0008) were observed at unthinned plots and insect counts were higher in thinned stands for *X. mutilatus* (p-value=0.0194).

The presence of pine decline may have confounded effects of prior thinning on insect presence. To account for this possibility, data from unthinned predicted decline stands were separated from that of unthinned, predicted non-decline stands. Insect totals from both plot groups were compared to those from those from thinned stands (n=10) with Tukey-Kramer comparisons using a General Linear Model Procedure. Significant differences were observed between thinned and unthinned predicted decline plots for *H. tenuis* (p-value= 0.0494), *P. picivorus* (p-value= 0.0793), *X. saxesenii* (p-value=0.0429), *X. crassiusculus* (p-value=0.0198) and *P. flavicornus* (p-value= 0.0286). Differences were obscured between previously thinned predicted decline plots and previously unthinned predicted decline plots after plots in non-declining areas were separated as a potential confounding factor. No predicted decline plots had been thinned. After presence of possible pine decline was accounted for, several insect species were more common in predicted decline plots including *P. picivorus* (p-value=0.0003), *H. porculus* (p-value=0.0143), *X. crassiusculus* (p-value=0.0449) and *P. flavicornus* (p-value=0.0293). Only *X. mutilatus* (p-value= 0.0378) was more common in the thinned plots.

When compared to the plots located in predicted non-decline compartments, the unthinned predicted decline plots generally had greater totals of several insect species (Tables 2.4 and 2.5)

Table 2.4. Comparison of mean bark beetle collections per plot at thinned predicted decline plots, unthinned predicted decline plots and unthinned predicted non-decline plots. Different letters within a row represent significant differences ($\alpha= 0.05$).

Species	Thinned decline (n=8)	Unthinned decline (n=10)	Unthinned non-decline (n=6)
<i>Hylastes salebrosus</i>	737	1181	815
<i>Ips grandicollis</i>	307	339	339
<i>Hylastes tenuis</i>	146 ab	186 a	131 b
<i>Hylastes porculus</i>	127	175	131
<i>Dendroctonus terebrans</i>	59	109	68
<i>Pachylobius picivorus</i>	40 ab	122 a	79 b
<i>Hylobius pales</i>	70	91	88
<i>Pissodes nemorensis</i>	10	12	10

Table 2.5. Comparison of mean ambrosia and predator beetle collections per plot at thinned predicted decline plots, unthinned predicted decline plots and unthinned predicted non-decline plots. Different letters within a row represent significant differences ($\alpha= 0.05$).

Species	Thinned decline (n=8)	Unthinned decline (n=10)	Unthinned nondecline (n=6)
<i>Gnathotrichus materiarius</i>	877	836	630
<i>Xylosandrus crassiusculus</i>	70 b	135 a	54 b
<i>Xyleborinus saxesenii</i>	172 a	172 ab	104 b
<i>Xylosandrus mutilatus</i>	111 a	62 b	80 b
<i>Monarthrum mali</i>	14	22	13
<i>Monarthrum fasciatum</i>	4	6	8
<i>Platypus flavicornis</i>	49 b	95 a	45 b
<i>Thanasimus dubius</i>	130	100	128
<i>Temnochila virescens</i>	180	155	121

2.5.5 Stratification method and results

The stratified groups consisted of the eight plots with the highest, median and lowest values for each forest health variable. The plots with the lowest resin production had the lowest number of *P. picivorus* and *X. crassiusculus* (Table 2.6). *Dendroctonus terebrans* was captured more frequently in plots with median resin production. *Gnathotrichus materiarius* and *X. saxesenii* were collected in greater numbers in plots that had greater five and ten-year growth. *T. virescens* was most common in the group of plots with the lowest five-year growth (Table 2.7).

Stands with the highest stem count were associated with the greatest numbers of *P. picivorus*, *D. terebrans* and *X. crassiusculus* while the stands with the lowest stem count had greater totals of *I. grandicollis* than the median stem count plots (Table 2.9). *Hylastes tenuis* was captured least frequently in the the low basal area plots (Table 2.10). The lowest hardwood basal area plots had the largest number of *H. salebrosus* and *D. terebrans*. More *G. materiarius* insects were collected at the lowest hardwood basal area plots than at the highest (Table 2.11).

Ips grandicollis was most frequently collected at plots with the lowest DBH. *Temnosheila virescens* was least frequently trapped at the plots with the highest DBH. *Xyleborinus saxesenii* was collected most in the plots with the greatest DBH, while *G. materiarius* was captured more at the highest DBH plots than at the lowest DBH (Table 2.12). Plots with the lowest mean live crown ratios (LCR) were associated with higher collections of *P. picivorus* and *X. crassiusculus*. Median LCR plots had the greatest numbers of *I. grandicollis* collected. *Hylastes salebrosus* was more common at low mean LCR plots than at median mean

LCR plots while *T. virescens* was most often collected in the median than low LCR plots (Table 2.13).

Capture of *H. tenuis* and *X. crassiusculus* were lowest at plots with high mean crown densities. *Pachylobius picivorus* were captured more in stands with the lowest mean crown density plots compared to plots with highest mean crown density (Table 2.14). Plots with the highest mean crown transparency had the greatest number of *P. picivorus*, *D. terebrans* and *G. materiarius* collected. Plots with the greatest mean foliar transparency had significantly greater numbers of *I. grandicollis* and *X. crassiusculus* than plots with the lowest mean crown transparency. The plots with the lowest mean crown transparency had the fewest of *H. salebrosus* (Table 2.15).

Table 2.6. Mean insect collection totals per plot within eight highest, median and lowest resin collection plots. Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (2.64-4.79g) (std error=0.76)	Median (2.01-2.58g) (std error=0.73)	Lowest (1.19– 1.99g) (std error=0.58)
<i>H. salebrosus</i>	22 a	20 a	16 a
<i>H. tenuis</i>	3 a	4 a	3 a
<i>H. porculus</i>	4 a	4 a	4 a
<i>P. picivorus</i>	2 a	2 a	1 a
<i>H. pales</i>	1 b	2 b	2 a
<i>D. terebrans</i>	1 ab	2 a	1 b
<i>I. grandicollis</i>	7 a	7 a	6 a
<i>G. materiarius</i>	13 b	20 a	19 a
<i>X. crassiusculus</i>	2 a	2 a	1 b
<i>X. saxesenii</i>	3 a	3 a	3 a
<i>T. dubius</i>	2 a	3 a	2 a
<i>T. virescens</i>	3 a	4 a	2 b

Table 2.7. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to five-year radial growth increment (mm). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (37-51) (std error=0.52)	Median (33-37) (std error=0.15)	Lowest (27-33) (std error=0.28)
<i>H. salebrosus</i>	19 a	20 a	18 a
<i>H. tenuis</i>	3 a	4 a	3 a
<i>H. porculus</i>	4 a	4 a	4 a
<i>P. picivorus</i>	2 a	2 a	2 a
<i>H. pales</i>	2 a	2 a	2 a
<i>D. terebrans</i>	1 a	2 a	1 a
<i>I. grandicollis</i>	6 b	7 ab	8 a
<i>G. materiarius</i>	21 a	16 b	15 b
<i>X. crassiusculus</i>	1 b	2 ab	3 a
<i>X. saxesenii</i>	4 a	3 b	3 b
<i>T. dubius</i>	2 a	3 a	2 a
<i>T. virescens</i>	2 b	3 a	4 a

Table 2.8. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to ten-year growth increment (mm). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (65-82) (std error=0.18)	Median (60-64) (std error=0.47)	Lowest (48-59) (std error=0.52)
<i>H. salebrosus</i>	21 a	18 a	18 a
<i>H. tenuis</i>	3 a	3 a	3 a
<i>H. porculus</i>	4 a	4 a	4 a
<i>P. picivorus</i>	2 a	2 a	2 a
<i>H. pales</i>	2 a	2 a	2 a
<i>D. terebrans</i>	2 a	2 a	1 a
<i>I. grandicollis</i>	7 a	6 a	7 a
<i>G. materiarius</i>	21 a	18 ab	14 b
<i>X. crassiusculus</i>	2 ab	2 b	2 a
<i>X. saxesenii</i>	4 a	3 b	3 b
<i>T. dubius</i>	2 a	3 a	2 a
<i>T. virescens</i>	3 a	3 a	3 a

Table 2.9. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to stem count (stems/acre). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (48-59) (std error=1.94)	Median (36-46) (std error=0.93)	Lowest (12-34) (std error=4.79)
<i>H. salebrosus</i>	21 a	17 a	20 a
<i>H. tenuis</i>	3 a	3 a	3 a
<i>H. porculus</i>	4 a	4 a	4 a
<i>P. picivorus</i>	2 a	1 b	2 ab
<i>H. pales</i>	2 a	2 a	2 a
<i>D. terebrans</i>	2 a	1 b	1 ab
<i>I. grandicollis</i>	7 ab	5 b	8 a
<i>G. materiarius</i>	18 a	18 a	17 a
<i>X. crassiusculus</i>	2 a	2 ab	2 b
<i>X. saxesenii</i>	3 a	4 a	3 a
<i>T. dubius</i>	2 a	2 a	2 a
<i>T. virescens</i>	3 a	3 a	3 a

Table 2.10. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to pine basal area (sq. ft./acre). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (120-170) (std error=2.00)	Median (90-110) (std error=0.95)	Lowest (30-90) (std error=2.18)
<i>H. salebrosus</i>	15 b	22 a	20 ab
<i>H. tenuis</i>	4 a	4 a	3 b
<i>H. porculus</i>	4 a	4 a	4 a
<i>P. picivorus</i>	1 b	2 a	2 ab
<i>H. pales</i>	2 a	2 a	2 a
<i>D. terebrans</i>	1 b	2 a	2 ab
<i>I. grandicollis</i>	6 a	6 a	8 a
<i>G. materiarius</i>	15 a	18 ab	19 a
<i>X. crassiusculus</i>	2 b	3 a	1 b
<i>X. saxesenii</i>	3 a	4 a	3 a
<i>T. dubius</i>	3 a	2 a	2 a
<i>T. virescens</i>	3 a	3 a	3 a

Table 2.11. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to hardwood basal area (sq. ft./acre). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (30-60) (std error=1.28)	Median (0-20) (std error=0.73)	Lowest (0-0) (std error=0.0)
<i>H. salebrosus</i>	16 b	16 b	25 a
<i>H. tenuis</i>	3 ab	3 b	4 a
<i>H. porculus</i>	3 a	5 a	4 a
<i>P. picivorus</i>	1 b	2 b	3 a
<i>H. pales</i>	1 a	2 a	2 a
<i>D. terebrans</i>	1 b	1 b	3 a
<i>I. grandicollis</i>	7 ab	6 b	8 a
<i>G. materiarius</i>	15 b	17 ab	20 a
<i>X. crassiusculus</i>	2 ab	1 b	2 a
<i>X. saxesenii</i>	3 a	4 a	3 a
<i>T. dubius</i>	2 a	3 a	2 a
<i>T. virescens</i>	4 a	2 b	3 ab

Table 2.12. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to mean diameter at breast height (in). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (9-11) (std error=0.28)	Median (8-9) (std error=0.15)	Lowest (7-8) (std error=0.09)
<i>H. salebrosus</i>	19 a	20 a	19 a
<i>H. tenuis</i>	3 a	3 a	3 a
<i>H. porculus</i>	4 a	4 a	3 a
<i>P. picivorus</i>	2 a	2 a	2 a
<i>H. pales</i>	2 a	2 a	2 a
<i>D. terebrans</i>	1 a	2 a	2 ab
<i>I. grandicollis</i>	6 b	6 b	8 a
<i>G. materiarius</i>	20 a	17 ab	15 b
<i>X. crassiusculus</i>	2 a	2 a	2 a
<i>X. saxesenii</i>	4 a	2 b	3 b
<i>T. dubius</i>	2 a	3 a	2 a
<i>T. virescens</i>	2 b	4 a	3 a

Table 2.13. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to mean live crown ratio (percent). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (35-47) (std error=0.58)	Median (30-34) (std error=0.58)	Lowest (25-29) (std error=0.58)
<i>H. salebrosus</i>	19 ab	14 b	24 a
<i>H. tenuis</i>	3 b	3 a	4 a
<i>H. porculus</i>	4 a	3 a	4 a
<i>P. picivorus</i>	1 b	1 b	2 a
<i>H. pales</i>	1 a	2 a	2 a
<i>D. terebrans</i>	2 a	1 a	2 a
<i>I. grandicollis</i>	8 a	5 b	7 a
<i>G. materiarius</i>	16 a	18 a	18 a
<i>X. crassiusculus</i>	1 b	2 b	3 a
<i>X. saxesenii</i>	3 a	3 a	3 a
<i>T. dubius</i>	2 a	3 a	2 a
<i>T. virescens</i>	4 a	3 b	3 ab

Table 2.14. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to mean live crown density (percent). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (37-43) (std error=0.67)	Median (33-37) (std error=0.44)	Lowest (29-33) (std error=0.49)
<i>H. salebrosus</i>	18 a	21 a	18 a
<i>H. tenuis</i>	3 b	4 a	3 a
<i>H. porculus</i>	4 a	4 a	3 a
<i>P. picivorus</i>	1 b	2 ab	2 a
<i>H. pales</i>	2 a	2 a	2 a
<i>D. terebrans</i>	1 a	2 a	2 a
<i>I. grandicollis</i>	7 a	7 a	6 a
<i>G. materiarius</i>	17 b	21 a	15 b
<i>X. crassiusculus</i>	1 b	2 a	2 a
<i>X. saxesenii</i>	3 b	4 a	3 b
<i>T. dubius</i>	3 a	2 a	3 a
<i>T. virescens</i>	3 a	3 a	3 a

Table 2.15. Mean insect collection totals per plot within eight highest, median and lowest plots with regard to mean live crown transparency (percent). Different letters within a row represent significant differences ($\alpha= 0.05$) within a species.

Species	High (35-38) (std error=0.29)	Median (33-35) (std error=0.27)	Lowest (30-33) (std error=0.37)
<i>H. salebrosus</i>	23 a	20 a	14 b
<i>H. tenuis</i>	4 a	3 a	3 a
<i>H. porculus</i>	4 a	4 a	4 a
<i>P. picivorus</i>	2 a	1 b	1 b
<i>H. pales</i>	2 a	1 a	2 a
<i>D. terebrans</i>	3 a	1 b	1 b
<i>I. grandicollis</i>	8 a	7 ab	5 b
<i>G. materiarius</i>	23 a	15 b	14 b
<i>X. crassiusculus</i>	2 a	2 ab	1 b
<i>X. saxesenii</i>	3 a	3 a	3 a
<i>T. dubius</i>	2 a	2 a	3 a
<i>T. virescens</i>	4 a	3 a	3 a

2.5.6 Insect totals in predicted decline stands versus those in predicted non-declining stands

Hylastes tenuis, *G. materiarius*, *X. crassiusculus*, *X. saxesenii* and *T. virescens* were more frequently captured in the predicted decline plots than predicted non-decline plots. No differences occurred between plots for *T. dubius*, *H. salebrosus*, *H. porculus*, *P. picivorus*, *H. pales*, *D. terebrans* or *I. grandicollis* (Table 2.16).

Table 2.16. Differences in pooled insect collection totals between plots in stands suffering from loblolly pine decline (n=18) and plots outside of decline area (n=6). “Predicted loblolly pine decline (LPD)” and “predicted non-loblolly pine decline (non-LPD)” refer to mean insect collection per plot per collection. ($\alpha= 0.05$).

Species	Predicted LPD	Predicted non-LPD	F-value	P-value
<i>Hylastes salebrosus</i>	17	20	2.15	0.1432
<i>Hylastes tenuis</i>	3	3	7.61	0.0059
<i>Hylastes porculus</i>	4	4	0.01	0.9386
<i>Pachylobius picivorus</i>	2	2	0.33	0.5659
<i>Hylobius pales</i>	2	2	0.41	0.5198
<i>Dendroctonus terebrans</i>	1	2	1.08	0.2998
<i>Ips grandicollis</i>	7	7	0.05	0.8260
<i>Gnathotrichus materiarius</i>	15	18	4.82	0.0283
<i>Xylosandrus crassiusculus</i>	1	2	11.73	0.0006
<i>Xyleborinus saxesenii</i>	2	3	10.29	0.0014
<i>Thanasimus dubius</i>	3	2	0.26	0.6090
<i>Temnoscheila virescens</i>	2	3	4.41	0.0359

2.5.7 Timing of insect collection peaks

The majority of species showed a spring population peak, reduced numbers from June to August and few if any individuals from December through February. Some species, however, were captured during times of year in which traps are rarely utilized. *Gnathotrichus materiarius* was captured primarily during relatively warm periods during December through February when contents of the intercept panel trap collection bucket were not frozen. The two most common weevils, *P. picivorus* and *H. pales* were generally captured during different seasons. *Hylobius pales* were most frequently captured during March and September while *P. picivorus* was captured more frequently from May through August.

2.6 Discussion

Predicted decline plots often had more insects recovered than predicted non-decline plots. Increased numbers of fungi-vectoring root feeders may be contributing to loblolly pine death within the stands believed to be in decline. One reported insect vector (*H. tenuis*) of *Leptographium* fungi associated with pine decline and several ambrosia beetle species were more common in predicted decline plots than predicted non-decline plots. Many trees in the predicted decline stands exhibited signs of stress and the elevated presence of ambrosia beetles provides additional evidence of poor tree health as ambrosia beetles have been reported to be more numerous in the increased presence of dead and dying trees (Gorsuch, 2003). Several ambrosia beetle species in this study, notably *G. materiarius* are known to benefit from the presence of dead and dying pines (Wood and Bright, 1992) and this appeared to be occurring here.

Crown characteristics in general were often more accurate predictors of insect populations than non-crown measurements and mean foliar transparency were particularly associated with higher insect captures. Compared to trees within predicted non-declining plots, those within predicted declining plots were expected to show signs of distress including reduced resin production, slower growth, lower live crown ratios, lower live crown density and higher live crown transparency. Greater bark and ambrosia beetle capture had been expected at plots where trees showed those outward characters than where there were healthier looking trees. Higher mean crown density and live crown ratios were associated with fewer insect totals for observed bark and ambrosia beetle species when differences were detected. This finding was not surprising as distressed crowns are often apparent before loblolly pine death. Given that most

ambrosia beetles and many bark beetle species avoid the defenses of healthy trees, greater collection totals of beetles was anticipated in declining plots.

Crown transparency was not strongly correlated with non-foliar tree measurements such as resin production, radial growth or basal area. Non-foliar tree measurements had weaker and less consistent relationships with insect collection counts. Resin production and growth rate were not strongly associated with insect collection totals despite a documented association between increased pine decline and slower growth (Eckhardt et al., 2007). This may be because those are earlier signs of distress than damage to needles and are not as directly associated with mortality. Pine and hardwood basal area did not appear to affect insect collections despite the documented preference of some ambrosia beetle species (such as *Monarthrum* sp.) for hardwoods. Measuring basal area with a hand prism may have artificially increased the basal area of loblolly pine at the expense of hardwoods given the greater diameter of pines at the plots.

Thinning generally benefits forest health and reduces susceptibility to insects (Werner, 2002). Thinning, for example, can be used to offset effects of loblolly pine decline by reducing competition between remaining trees. Following a thin, standing trees often have increased access to light and nutrition. Insect counts in thinned stands in this study were lower than insect counts in unthinned stands, suggesting that thinning of stressed trees may have improved tree insect resistance even within predicted decline areas. On private land, this may serve to keep loblolly pine trees alive long enough to be more profitably harvested. Many land managers practice thinning for their own economic benefit in order to increase growth during rotations. Private land owners do not grow loblolly pine to its full age during commercial rotation and thinning may enable loblolly pine plantations to enhance their revenue even within areas of

loblolly pine decline. In non-commercial forests, such as the study sites, growing alternate species that may be more site-appropriate, such as longleaf pine may be a better long-term strategy.

Because many bark beetle species coexist in the same habitat, some insect species were expected to be associated with each other and their associated predators. Bark beetle species were often found together but associations between ambrosia beetles were less strong. Correlations between ambrosia beetle species were expected to be weaker because ambrosia beetles often live in dead or dying material and exploit a broader range of tree species. Some ambrosia beetle genera, such as *Xylosandrus* and *Xyleborinus* include many species that are capable of feeding on dozens of tree species, both softwood and hardwood (Wood and Bright, 1992). Surprisingly, the predator species *T. dubius* was negatively correlated with all other insect species with which it had a significant relationship. This was inconsistent with the hypothesis that *T. dubius* feeds on its prey in a density dependant manner (Turchin et al., 1999). However, if *T. dubius* is more specifically attracted to frontalin, the primary attractant chemical of southern pine beetle (*D. frontalis*) the low numbers of southern pine beetle captured during the survey may have influenced *T. dubius* numbers. Additionally, many of the more common bark beetle species (in genus *Hylastes*) live much of their life cycle underground, whereas the predator beetles captured in this study feed primarily on in beetles associated with other areas of the tree.

Overall bark and ambrosia beetle capture was heavily weighted towards members of genus *Hylastes* and the ambrosia beetle *G. materiarius*. These species represented over half of the total insect captures. The seemingly high number of correlations between *Hylastes*

salebrosus and other bark beetles may have simply been a function of how many individuals were captured. The *Hylastes* species and weevils captured are known vectors of fungi associated with loblolly pine decline and their high capture may be an indicator of the widespread problems with loblolly pine decline in the District (Eckhardt et al., 2007). The fact that only *Hylastes tenuis* among these species appeared to be greater in predicted decline plots was perhaps because bark beetles tend to live in living trees rather than dead material. The greater collection of ambrosia beetles at the predicted decline plots supported the accuracy of the decline model and the low insect resistance of many trees expected to be in decline (Table 2.15). This conclusion was supported by the distressed foliage at predicted decline plots. Total capture of *D. frontalis* was low and insect diversity during a southern pine beetle outbreak would be expected to be considerably different.

The southern pine beetle (*D. frontalis*) was rarely collected in the study. These observations were in line with regional observations of low southern pine beetle (SPB) activity in 2008, with SPB captured in only 1,433 spots recorded in the southeastern United States, compared to 60,000 spots in 2000 (Billings, 2008). Southern pine beetle levels in 2009 were expected to remain at 2008-levels in most states and to decline in Alabama (Billings, 2008). Southern pine beetle population patterns are erratic and a sudden outbreak of SPB would have provided an opportunity to document the extent of *T. dubius* population growth following an increase in prey population. Another possibility in the aftermath of a *D. frontalis* outbreak would be an upsurge of activity by bark beetles such as *Ips* and ambrosia beetles with the capability to exploit the increased potential habitat of dead and dying trees following a mass attack.

Insects captured in this study were similar to insects captured in Lindgren funnel traps during other trapping surveys in the southeastern United States. In prior studies, native species predominated among bark beetle collections whereas many of the ambrosia beetles encountered were non-native (Oliver and Mannon, 2001). This was true in this study as well (Table 2.1). While the most commonly observed bark beetles were native, several invasive ambrosia beetle species were prominent. In other studies conducted in different areas of the southeastern United States, *X. saxesenii* and *X. crassiusculus* were the most common ambrosia beetles rather than the native *G. materiarius* (Miller and Rebaglia, 2009). At three hardwood stands in Tennessee invasive *X. saxesenii* was the most abundant species (nearly half of all insects observed) with *X. crassiusculus*, *M. fasciatum*, *M. mali*, *X. atratus*, *Xyleborus pelliculosus* Eichoff and *Ambrosiodus tachygraphus* Zimmerman the other species with greater than 100 total individuals collected (Oliver and Mannion, 2001). The biggest difference between the findings of this study and previous studies of ambrosia beetles was that this project captured a disproportionately high number of *G. materiarius*.

Although insect collection totals were generally lower from November through February, insects were captured throughout the year. Many *G. materiarius* were captured in winter trapping that which detected population peaks during months not surveyed by earlier work. Extending the length of the trapping season appears to show that several species, including some that carry fungi associated with loblolly pine decline (Eckhardt et al., 2007) have been previously undercounted. This finding affirms results from earlier studies by Zanzot et al. (2010) who observed September peaks of several species, most frequently *H. tenuis* and *H. pales*, that supplemented more commonly detected spring peaks.

Survey accuracy is also influenced by how effectively traps capture target insect species. As in earlier studies, aerial traps captured the broadest diversity of insects. Window, panel intercept and funnel traps are typically used in general population surveys and studies aimed at invasive species. Pitfall traps are considered a supplemental tool in general insect surveys (Hyvarinen et al., 2006) but have been the primary trap used in surveys of root feeding beetles. A relatively high proportion of *P. picivorus* and *H. pales* in this study were collected in pitfall traps, appearing to confirm the attraction of these weevil species to roots given the greater effort required for these relatively large species to enter the smaller entrances of the pitfall traps. *Pachylobius picivorus* was four times more frequently collected in pitfall traps than at flight intercept traps in Wisconsin but in Louisiana *P. picivorus* was collected twice as often in flight intercept traps than pitfall traps (Erbilgin et al., 2001). This variation appears to validate the use of multiple traps. Collections from Wisconsin pine plantations and the Kisatchie National Forest in Louisiana reported many species captured in this study. While different traps were more effective in different regions, many of the same insect species were captured as at the Oakmulgee Ranger District (Erbilgin et al., 2001). Previous studies that have trapped bark beetles through pitfall trapping have also captured large proportions of *Hylastes* and weevil species (Zanzot, et al., 2010). The appropriateness of different traps types appears to be a function of the range and habitat of target species.

Biomass removal of woody debris may become more common given environmental and social pressures for the development of alternative fuels. This treatment, essentially a thin from below that removes smaller diameter vegetation, may also be used for some wildlife management. Under a biomass removal regime, trees selected for fuel harvest would be

completely removed from the plot. The effects of this treatment have not been tested on bark and ambrosia beetle populations and it is uncertain whether biomass removal would either increase or decrease insect numbers. While a temporary increase in release of attractant chemicals could draw bark beetles to the experimental plots the removal of potential habitat could reduce long term bark and ambrosia beetle populations. Also, following the treatments, trees would be released from competition with each other and expected to be more insect resistant. Effects of biomass removal may vary by insect type. Ambrosia beetles largely rely on declining and dead trees for habitat and their numbers would be expected to decline. Root dwelling beetles and insects would still have some habitat when roots are still around but many of those species are usually found on living hosts (Wood and Bright, 1992). Finally, *T. dubius* and *T. virescens* feed largely on these bark and ambrosia beetles and their population numbers could be expected to mirror those of their prey.

2.7 Conclusion

The large numbers of *Hylastes* species collected at the plots indicate that they may play a more important role in forest health than previously believed. Slope and aspect differences between adjacent plots impacted insect population numbers. Crown condition was the most positive indicator of insect collection totals, as stands with more dense and less transparent tree crowns were associated with fewer insect numbers. Many bark beetle species were positively correlated with both poor crown condition and each other. If these correlations consistently hold, the populations of many bark and ambrosia beetles are likely to increase during southern pine beetle outbreaks.

A wider spectrum of insects was captured by collecting all year with a variety of traps. Panel intercept traps appeared to successfully capture a wide range of species, including those such as in the genus *Hylastes*, that are generally associated with roots. Deployment of the traps year round resulted in rarely observed peaks of some species (such as *G. materiarius*). Wider employment of year round trapping could provide for a more accurate idea of insect population and species richness.

The findings from the studies provided confirmation of the decline model at the Oakmulgee Ranger district. That a greater total of root and lower bole feeding beetles were captured at the stands considered to be in decline provides additional evidence that these species are involved in the loblolly pine decline complex. Previous thinning appeared to reduce the presence of some insect species, perhaps indicating that long-term resistance of trees within thinned stands has been enhanced. Subsequent biomass removal performed at the plots will test how bark and ambrosia beetle populations respond to these experimental treatments.

Chapter 3

Efficacy of Three Insect Trap Types at the Oakmulgee Ranger District

3.1 Abstract

Three different trap types (intercept panel, flight intercept and pitfall traps) were directly compared at three subplots of 24 research plots in the Oakmulgee Ranger District of the Talladega National Forest in west-central Alabama. Panel intercept traps captured the majority of insects for nearly all bark and ambrosia beetles, including species that are primarily associated with root habitats. While pitfall traps captured fewer insects than intercept panel traps, they preferentially captured root and lower bole dwelling species. Flight intercept traps captured lower numbers of the same species as panel intercept traps. Panel intercept traps appear to be effective for use in general insect surveys and studies that currently use Lindgren funnel traps. Pitfall fulfilled a specialized function and acted as a useful supplement.

3.2 Introduction

Insect trapping allows consistent, long-term surveys to be conducted in remote areas where insect populations would not otherwise be known. The ideal trap and lure used depends on both the insect species and the variety of insects being targeted. Effective trapping methods mimic the habitat and chemical attractants of the target insects. The efficiency of

different bark and ambrosia beetle traps varies between different species and at different settings. Trapping can target individual species or be used to conduct a general survey. Insect traps are often used to monitor the presence and spread of exotic species, for observation of population peaks and better document life histories of insect species (Weinzieri et al., 2005).

Traps attract insects with visual attractants, scent attractants or both. Size, shape, elevation and color are visual components of traps which can mimic habitat and attract insects. For example, many bark beetles are attracted to traps that match the profile of trees (deGroot and Nott, 2001). Dark colored traps have been found to capture more bark beetles and weevils than lighter colored traps. Reflective traps appear to reduce capture of insects sensitive to different visual wavelengths (Mizell and Tedders, 1999). White and yellow traps captured fewer insects than black, blue, brown, green, grey and red traps. However, Lindgren (1983) concluded that ambrosia beetles are not affected by trap color which was supported by Strom and Goyer (2001). Pheromones and chemicals issued by trees associated with both habitat and prey species have also been effectively used in trapping (Weinzieri et al., 2005). Baiting traps with synthesized chemical attractants to mimic these attractants is considered to be one of the most cost-effective and efficient ways to survey insect populations (Dodd et al., 2010).

Baited traps capture insects more effectively than unbaited traps (Bouget et al., 2009) but effectiveness of chemical lures vary by insect species. Ethanol, turpentine and ethanol and turpentine used in combination are known bark and ambrosia beetle attractants. The combination of ethanol and turpenes is a strong synergistic attractant for bark beetles (Gandhi et al., 2010). Ethanol enhanced the attraction of alpha-pinene for *Xyleborus pubescens* Zimmermann, *Hylastes porculus* Erichson, *H. salebrosus* Eichhoff and *H. tenuis* Eichhoff (Miller and Rebaglia, 2009)

and also has a synergistic effect when mixed with turpentine to trap bark beetles such as *Dendroctonus terebrans* (Oliver), pales weevils and pitch eating weevils (Fatzinger, 1985). *Pachylobius picivorus* Germar were strongly attracted to ethanol to turpentine ratios greater than 1:1. *Hylobius pales* Herbst were slightly more attracted to ethanol alone than *P. picivorus*. The turpentine component may be needed for host recognition in baits (Rieske and Raffa, 1991). Ethanol alone has been successfully employed as bait for *Xyleborus* species in 22 states (Rebaglia et al., 2008). Scolytid species have also been more attracted to traps baited with ethanol compared to unbaited traps in hardwood forests (Montgomery and Wargo, 1983). The placement of bait within a trap also affects attraction of insects to traps. Lures placed inside traps were generally found to be more effective than lures above traps (Dodd et al., 2010).

Both flying and root dwelling insects have ramifications on forest health. Aerial traps, such as panel intercept traps, are used to survey southern pine beetle, *Ips* species and other beetles associated with feeding on trees. Some root feeders which are typically captured in pitfall traps have been implicated in premature loblolly pine decline which occurs across the southeastern United States. Because of the great importance of loblolly pine in both unmanaged and commercial forestry it is important to better know the populations of insects such as *Hylastes* species that are known to vector fungi associated with loblolly pine decline. A more comprehensive understanding of populations of southeastern bark and ambrosia beetles is needed and the use of multiple trap types was intended to capture more of this diversity. This study compared the relative efficacy of panel intercept traps, flight intercept traps and pitfall traps to capture bark and ambrosia beetles. Because of their black coloration, large entrance, profile that resembles trees and broad dispersion of chemicals, panel intercept traps were expected to capture

the largest number of insects. Pitfall traps were expected to capture predominantly root feeding insects. It was expected that some traps would be particularly effective at capturing these different species and that this study would be a useful way to observe those details.

3.3 Methods and Materials

Bark and ambrosia beetles were trapped at twenty-four plots located in the Oakmulgee Ranger District of the Talladega National Forest in Perry, Chilton and Bibb County Alabama. Study sites, selected on the basis of stand history, slope and aspect, were former agricultural lands converted into forest between 1973 and 1984. Each plot included three subplots located 36.57 meters from the center of the plot at bearings of 120, 240 and 360 degrees. Subplots were an array of circles with a radius of 7.62 meters and an overall area of 0.0182 hectares. Plot design followed methods used in the USDA Forest Inventory and Analysis Program (Dunn, 1999).

The three traps used were panel intercept traps, flight intercept traps and pitfall traps. Intercept panel traps (APTIV Company, Portland, Oregon) were hung approximately two meters by wire from a metal pole kept in the ground by a metal sleeve. The bucket of each panel trap was filled with approximately 45 ml of a mixture of two parts distilled water to one part antifreeze, added to prevent the escape of captured insects. Flight intercept traps consisted of a clear plastic 1-gallon milk jug cut open on three sides with the fourth side attached to a pole approximately 0.60 m above the ground. A 120 ml plastic cup attached to the lip of the milk jug served as the receptacle for insects. Two 5 cm long by 2 cm dia pine stems were placed in the cup to attract insects. Pitfall traps consisted of 10 cm by 20 cm diameter PVC plastic pipe with eight entrance holes equally spaced around the circumference. The traps were buried with the

entrance holes within 5 cm of ground level at each subplot. A plastic skirt was fitted around the trap to reduce the risk of flooding (Menard, 2007). Two loblolly pine cuttings 5 cm by 2 cm length and diameter were placed in the trap base. The cap was kept loose to facilitate access. Escape of captured insects was prevented though coating with a thin layer of liquid Teflon™ (Northern Products Woonsocket, RI) each collection period. Trapped insects remained in the cup until the following collection period.

Traps were collected on a biweekly basis between March 2008 and February 2010. Each trap was baited with a mixture of 95% ethanol and turpentine. Because each trap type was present at each subplot, it was possible to directly compare the relative effectiveness of each trap at capturing insects present at the Oakmulgee Ranger District. Through direct comparisons the project should allow a determination of whether alternatives to pitfall traps could adequately capture bark and ambrosia beetles associated with roots and the lower bole.

3.4 Statistics

SAS 9.1 ((SAS Institute Inc., Cary, N.C.), analysis of variation (ANOVA) was used to compare capture data from intercept panel traps, flight intercept traps and pitfall traps. The data included was the total collection totals per species at each plot during each collection. Analysis of variation was determined using a Tukey test and the threshold of significant difference was set at 0.05.

3.5 Results

3.5.1 Differences between trap types

Panel intercept traps captured a significantly greater total of nearly every insect species observed in the study (Table 3.1). The only exception was for the root feeding bark beetle *H. tenuis*, in which pitfall traps captured significantly more individuals than panel traps. The vast majority of individuals of bark and ambrosia beetles, not associated with roots, were captured in panel traps. Overall, panel intercept traps captured the most insects and the broadest range of species. Insect species primarily associated with flight, with no root feeding associated with their lifecycle (*Gnathotrichus materiarius* Fitch, *Thanasimus dubius* Fabricius and *Temnochila virescens* Fabricius) were captured so predominantly in intercept panel traps that differences between other trap types were not statistically significant.

Differences between flight intercept trap and pitfall trap collection totals were also observed for many species (Table 3.2). Flight intercept traps collection totals were significantly greater than in pitfall trap capture for *H. salebrosus*, *Ips grandicollis* Eichhoff, *G. materiarius*, *Xylosandrus crassiusculus* (Motschulsky), *Xyleborinus saxesenii* Ratzburg, *T. dubius* and *T. virescens*. Pitfall traps captured more *H. tenuis*, *P. picivorus*, *H. pales* than flight intercept traps. Collections of *Hylastes salebrosus*, *H. porculus*, *D. terebrans*, *G. materiarius*, *T. dubius* and *T. virescens* were not significantly different between flight intercept and pitfall traps.

Table 3.1. Mean collection totals of insect species per trap per collection period. Significant differences have different letters. ($\alpha= 0.05$) indicate significant difference in insect collection between trap type.

Species	Panel trap	FIT trap	Pitfall trap	F-value	P-value
<i>H. salebrosus</i>	16 a	2 b	1 b	254.92	<0.0001
<i>H. tenuis</i>	1 b	0 c	2 a	155.72	<0.0001
<i>H. porculus</i>	3 a	1 b	0 b	131.29	<0.0001
<i>P. picivorus</i>	1 a	0 c	1 b	55.62	<0.0001
<i>H. pales</i>	1 a	0 c	1 b	56.54	<0.0001
<i>D. terebrans</i>	1 a	0 b	0 b	87.11	<0.0001
<i>I. grandicollis</i>	6 a	1 b	0 c	368.15	<0.0001
<i>G. materiarius</i>	17 a	1 b	0 b	502.72	<0.0001
<i>X. crassiusculus</i>	1 a	1 b	0 c	101.20	<0.0001
<i>X. saxesenii</i>	3 a	0 b	0 c	202.63	<0.0001
<i>T. dubius</i>	2 a	0 b	0 b	88.61	<0.0001
<i>T. virescens</i>	3 a	0 b	0 b	263.27	<0.0001

3.5.2 Differences between timing of species abundance

The majority of insect species were captured in the greatest abundance during April and May of each year but no two species displays identical population patterns. *Hylastes salebrosus* was most frequent during March and April in both 2008 and 2009 and had a smaller peak that occurred in the fall, which was larger in 2009 than in 2008 (Fig. 3.1). The peak collection period for *H. tenuis* was in April and May with a second, smaller peak in September (Fig. 3.2). The number of *H. porculus* trapped during October and November of 2009 was larger than at any other time although smaller peaks were observed in the fall of 2008 and April of both years (Fig. 3.3).

Hylobius pales and *P. picivorus*, although about equally common, were largely captured at different times of year. *Pachylobius picivorus* were recovered in April and May of both years,

with more individuals collected in 2009 than in 2008 (Fig. 3.4). *Hylobius pales* were most common in March and April with a smaller fall peak, during the 2009 season (Fig. 3.5). *Dendroctonus terebrans* collections showed a small peak in March and early April 2009 followed by a larger peak in July and August 2009. The summer 2009 peak constituted a significant part of the total *D. terebrans* collected (Fig. 3.6). *Ips grandicollis* was collected most frequently from April to June (Fig. 3.7).

Gnathotrichus materiarius was the only species captured in the largest numbers during the winter months (Fig. 3.8). Capture was reduced, however, when temperatures dropped below freezing for several days. *Xylosandrus crassiusculus* was most numerous in March and April during both years (Fig. 3.9). *Xyleborinus saxesenii* was more common in early spring than many other species, peaking in March during both years (Fig. 3.10). Of the predator species, *Thanasimus dubius* was captured most frequently during May and June, particularly in 2009 (Fig. 3.11). *Temnosheila virescens* peaked earlier in the year and was most numerous in late March and early April (Fig. 3.12).

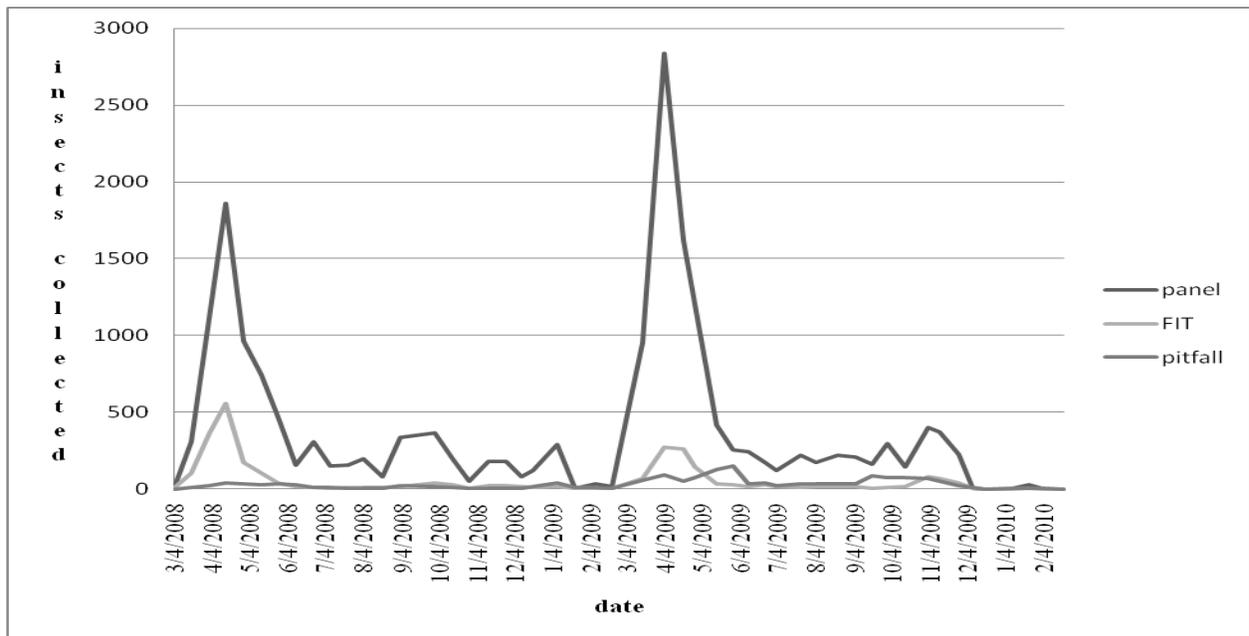


Figure 3.1. *Hylastes salebrosus* totals at panel, flight intercept and pitfall traps over two-year period.

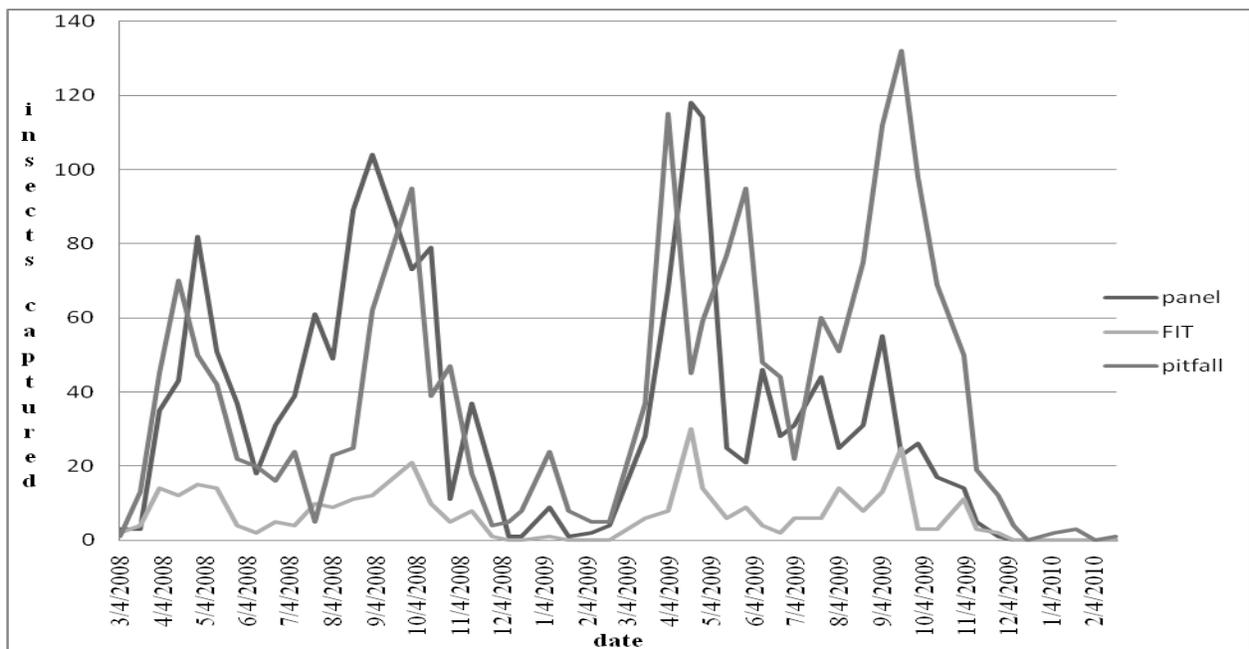


Figure 3.2. *Hylastes tenuis* totals at panel, flight intercept and pitfall traps over two-year period

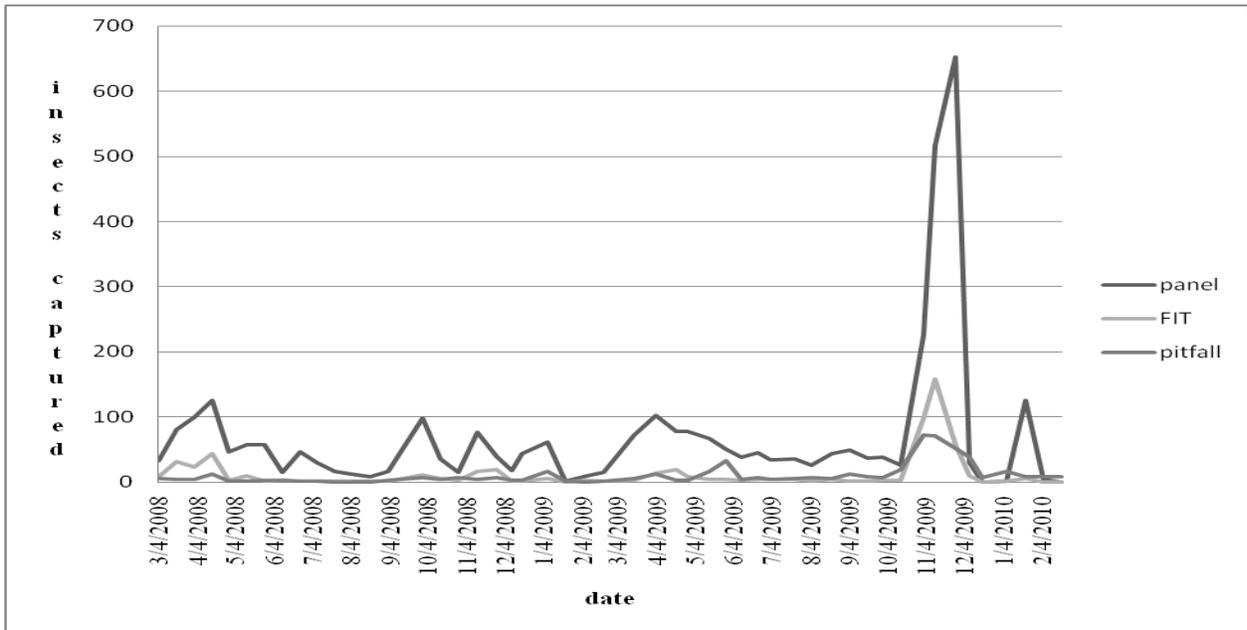


Figure 3.3. *Hylastes porculus* totals at panel, flight intercept and pitfall traps over two-year period.

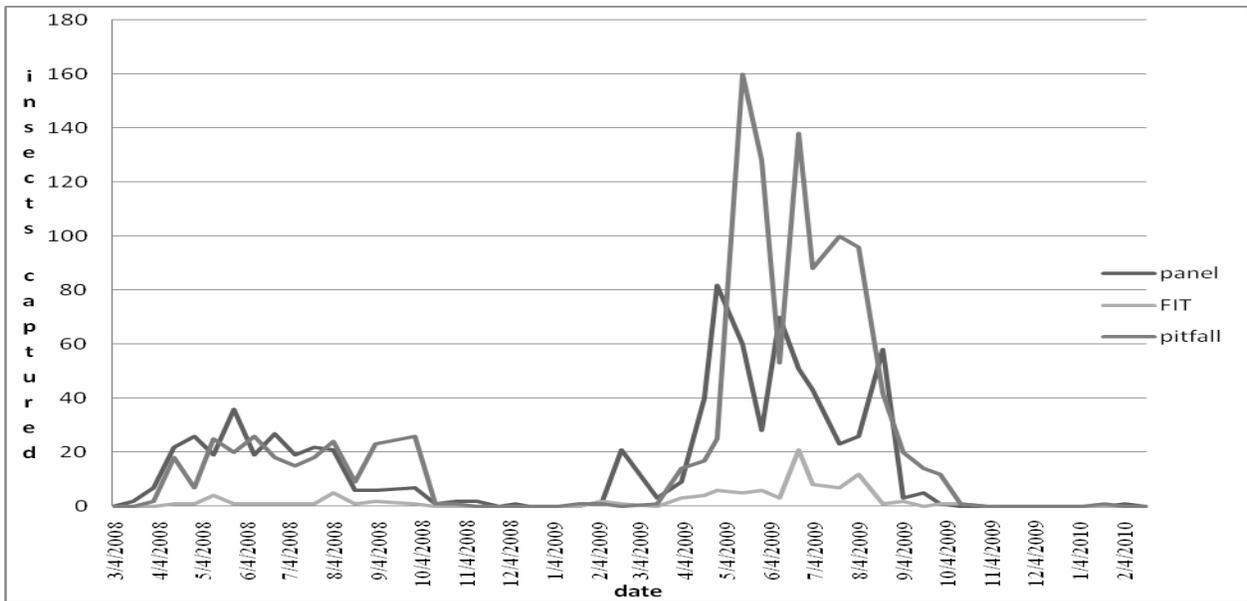


Figure 3.4. *Pachylobius picivorus* totals at panel, flight intercept and pitfall traps over two-year period.

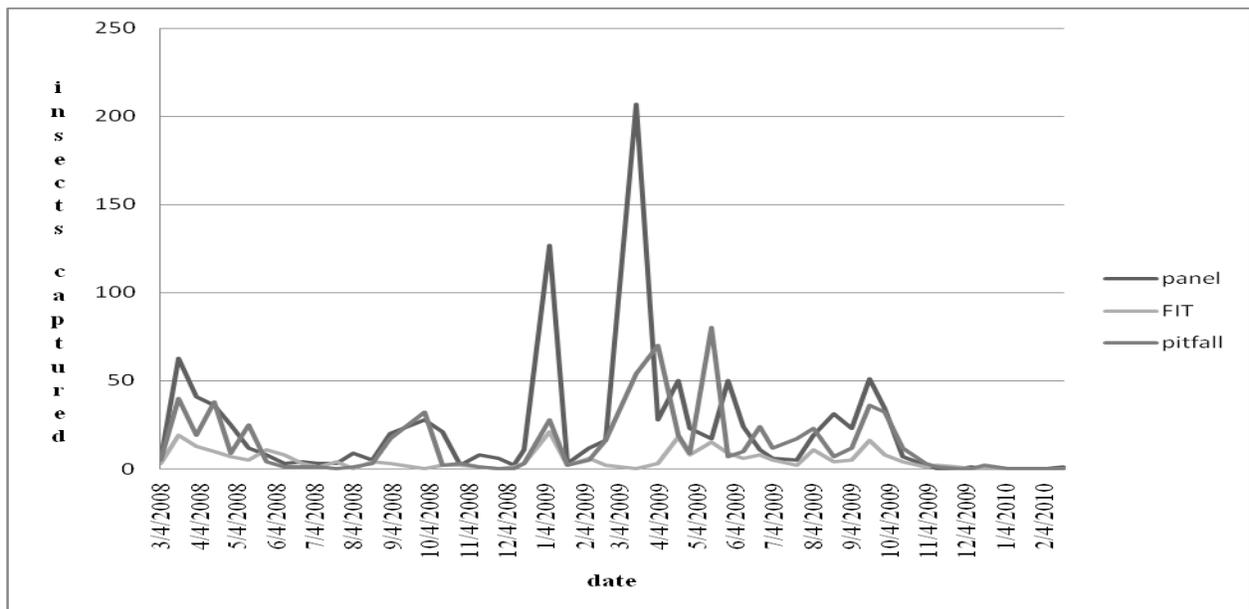


Figure 3.5. *Hylobius pales* totals at panel, flight intercept and pitfall traps over two-year period.

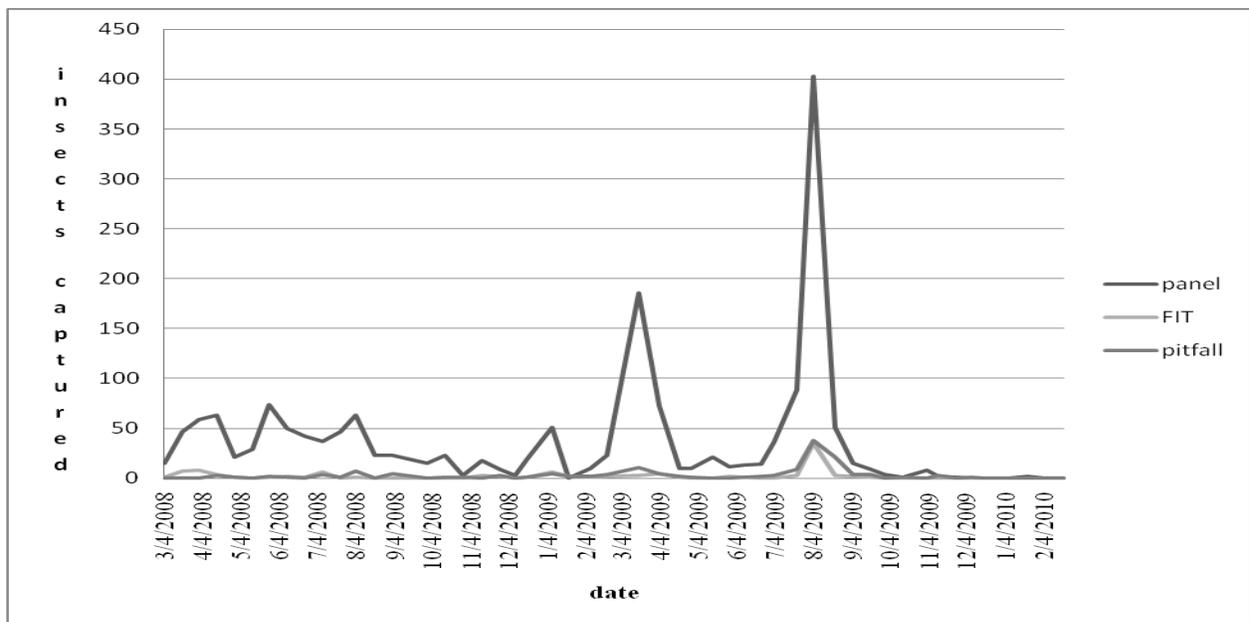


Figure 3.6. *Dendroctonus terebrans* totals at panel, flight intercept and pitfall traps over two-year period.

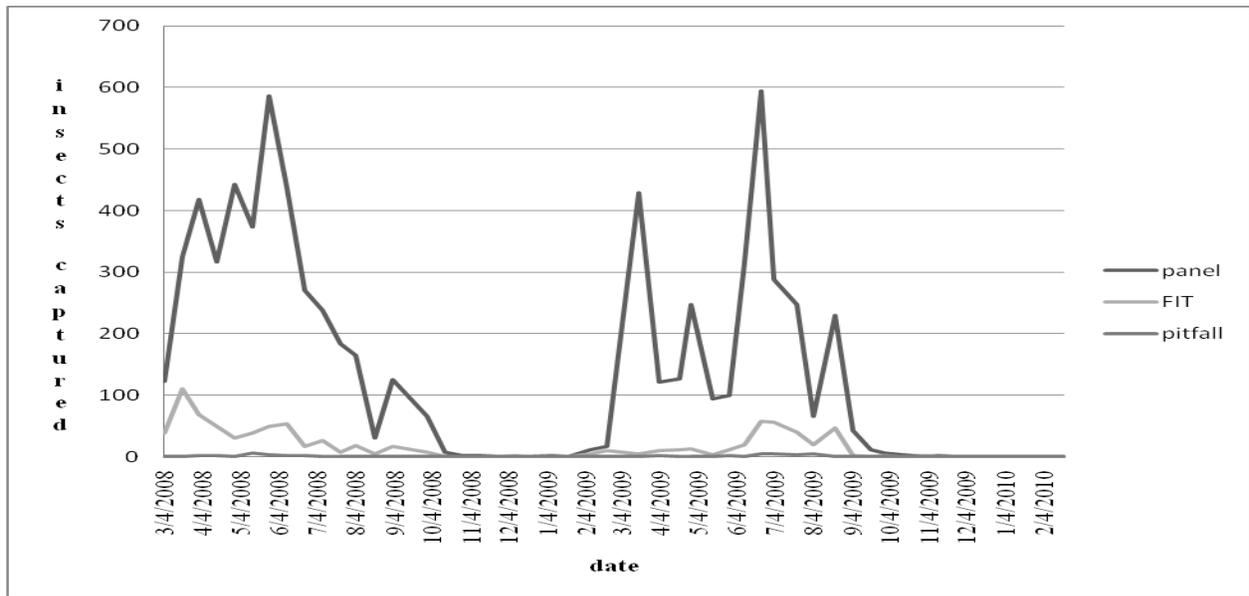


Figure 3.7. *Ips grandicollis* totals at panel, flight intercept and pitfall traps over two-year period.

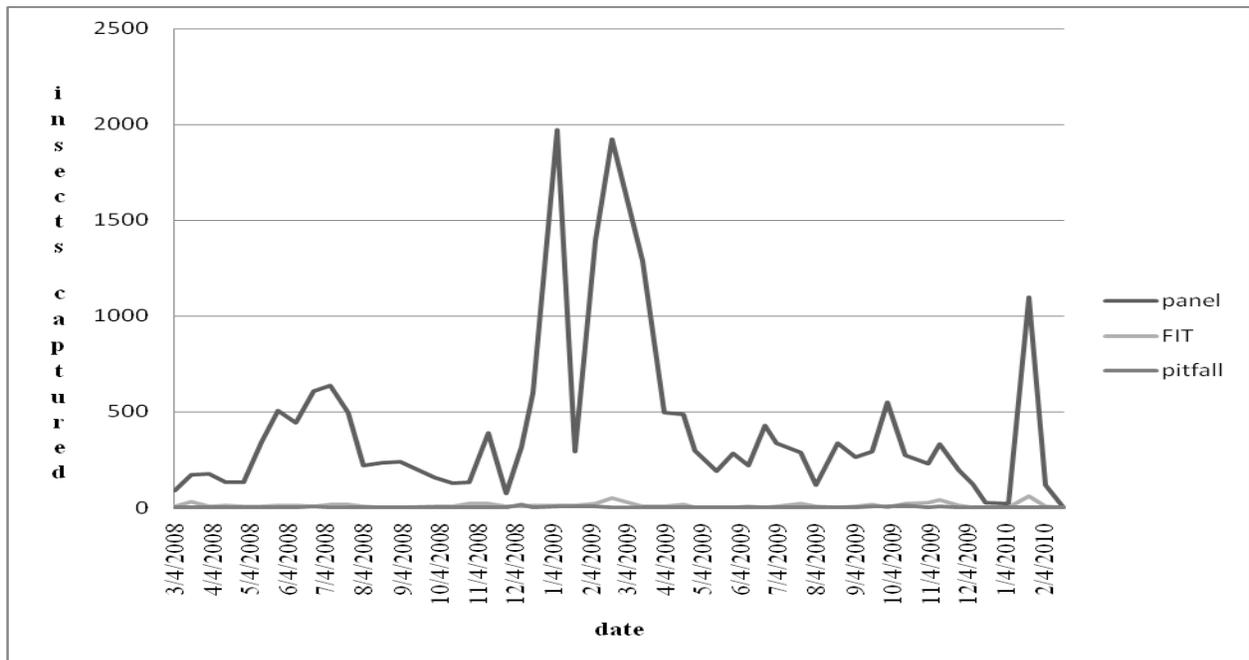


Figure 3.8. *Gnathotrichus materiarius* totals at panel, flight intercept and pitfall traps over two-year period.

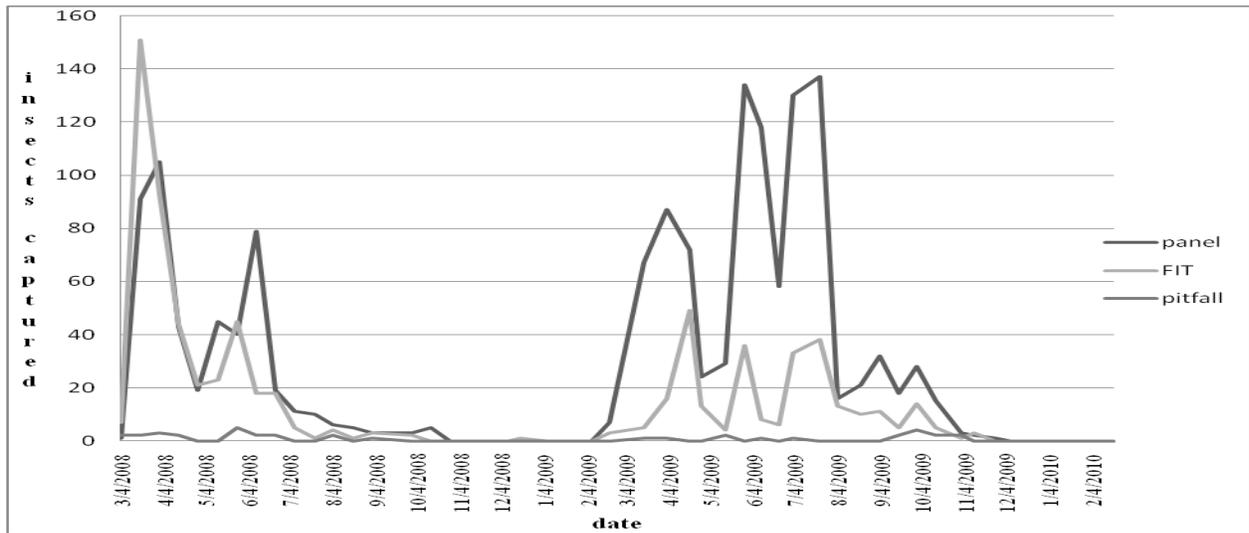


Figure 3.9. *Xylosandrus crassiusculus* totals at panel, flight intercept and pitfall traps over two-year period.

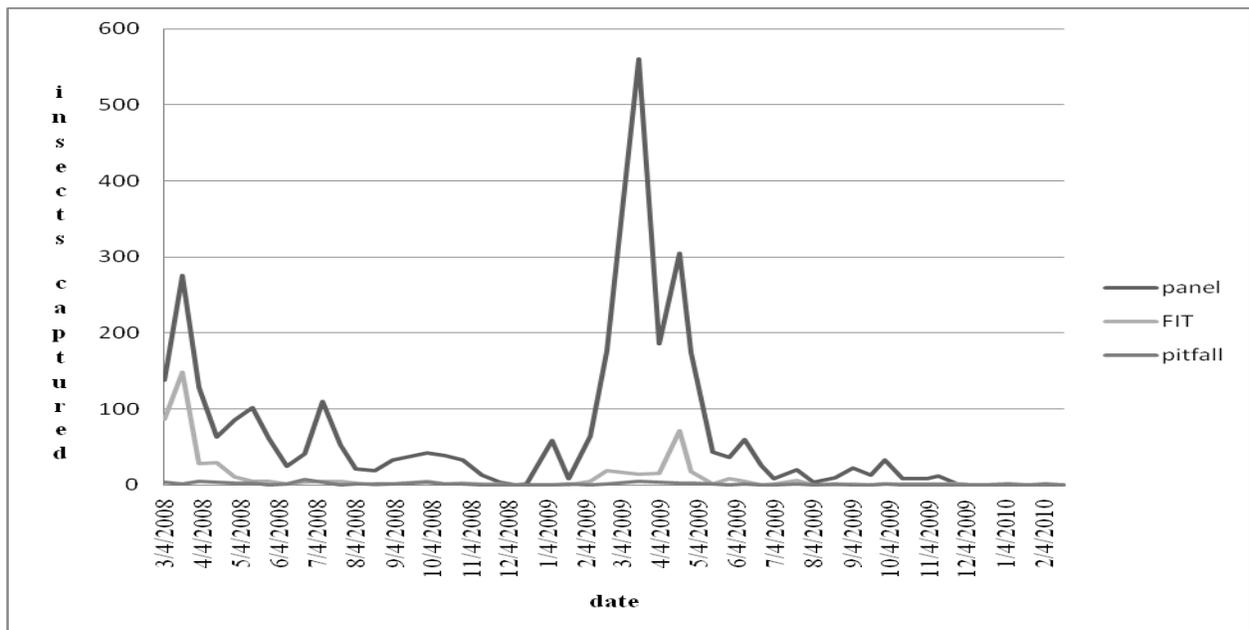


Figure 3.10. *Xyleborinus saxesenii* totals at panel, flight intercept and pitfall traps over two-year period.

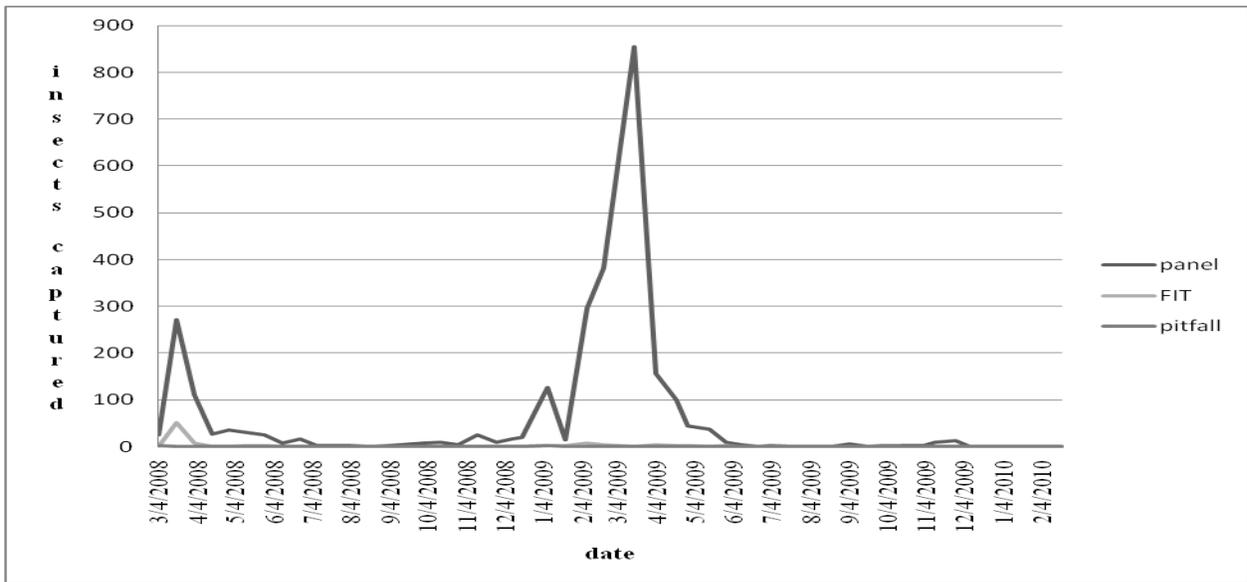


Figure 3.11. *Thanasimus dubius* totals at panel, flight intercept and pitfall traps over two-year period.

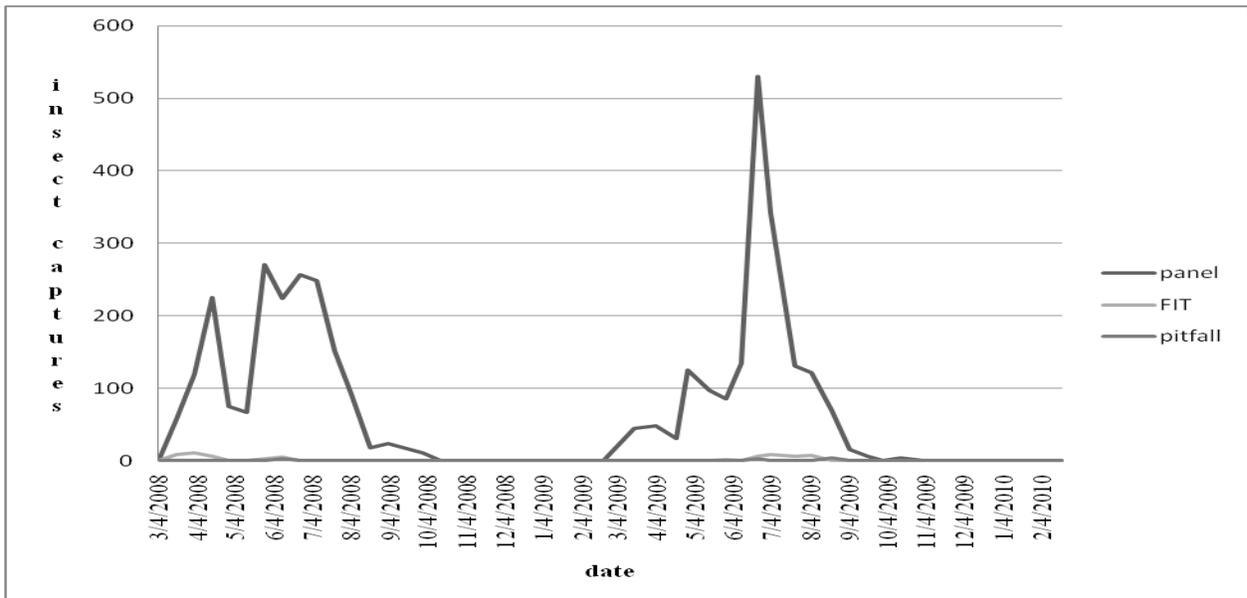


Figure 3.12. *Temnoscheila virescens* totals at panel, flight intercept and pitfall traps over two-year period.

3.6 Discussion

Pitfall traps are generally used to capture bark beetle species but the supplemental use of other trap types appeared to increase the accuracy of the survey. Because capturing overall diversity of bark and ambrosia beetle populations was an important goal, aerial traps such as intercept panel traps were employed. The use of intercept panel traps in particular greatly increased the total capture of bark and ambrosia beetles. The most important diversity trends appeared to be the large proportion of overall insect capture that occurred in intercept panel traps, the presence of root dwelling bark beetles and weevils in the panel traps and the specialized function of pitfall traps. Intercept panel traps may, therefore, have potential to be used to survey general bark beetle populations the way that funnel traps are used in seasonal surveys of species such as southern pine beetle.

Almost every species collected during the study was captured more frequently in intercept panel traps. Intercept panel traps possessed several characteristics which likely increased their insect capture. Entrances of these traps were larger than those of flight intercept and pitfall traps and correspondingly easier for insects to enter. A comparatively higher elevation allowed panel intercept traps to better capture insects flying through the forest. Since the late 1970s, scientists have known that the similarity of the silhouette of the panel traps to host pine species helps attract insects (Lindgren, 1983), so visual resemblance to trees probably enhanced effectiveness of intercept panel traps. Black traps are unreflective and therefore attract bark beetles. Profile and color likely heightened the attractiveness of the intercept panel traps. Flight intercept traps did not have the silhouette of pine trees, were more difficult to enter and easier to escape from due to the absence of teflon or antifreeze. Flight intercept traps generally captured

the same species as intercept panel traps whereas pitfall traps captured proportionately more root-dwelling insects. In this study, with its emphasis on capturing bark beetle diversity, the use of panel intercept traps probably enhanced the accuracy of the count through greatly increased capture of non-root feeders.

Overall, species that are associated with the mid-bole of trees and higher were captured almost exclusively in aerial traps and root and lower bole feeders were captured in both aerial and pitfall traps. The proportion of relatively large *P. picivorus* and *Hylobius pales* in the pitfall traps and the greater amount of effort required for the weevils to enter these traps leads credence to the idea that these species are particularly attracted to the roots and will expend a greater amount of energy to get to them. *Hylastes* species and weevils were also significantly more abundant in pitfall traps than flight intercept traps. This was expected considering that these species are associated with pine roots (Eckhardt, 2003). Many of the root-feeding species are associated almost exclusively with roots and the lower bole, so while the presence of *Hylastes* spp., *Pachylobius picivorus* and *Hylobius pales* in pitfall traps were unsurprising, it was less expected that collection totals of all other root feeders except for *H. tenuis* were greatest in panel traps. Panel traps, even hung nearly two meters in the air typically captured more individuals of root-dwelling species than pitfall traps. The significantly greater capture of these insects in panel traps demonstrates the often underestimated importance of flight in the lifecycle of these insects.

The presence of nine traps in each of the collection plots may have increased the overall capture totals within each plot. The release of chemical attractants from different trap types and different subplots within the plot may have interacted with one another. This could have potentially been a confounding factor if insects were frequently captured by a different trap than

the one which initially attracted them. *Ips* collections have been affected by both the number of traps in a block and the total concentration of attractants (McMahon et al., 2010) in Wisconsin. However, attractants are less widely dispersed in pitfall traps because of their trap design, so all traps may not have made the same contribution to the plume. Possible interference between different traps and whether bait from some traps may attract insects to different, adjacent traps may be a potential topic for future research.

Many studies only keep traps out for a few months and few trap for insects later in the year than September. Detection of unexpected population patterns is a possible justification for winter trapping. Many of the insect species in the study showed population peaks in March, after leafout and were seldom or never trapped after October but there were several exceptions, including several species associated with loblolly pine decline. September and October collection peaks of *H. tenuis* and December and January peaks of *G. materiarius* appeared to be the most consistent off-season increases of beetles in the study. The bigger fall populations of *H. porculus* compared to *H. salebrosus* in the second year of the study was not entirely surprising, given that the range of *H. porculus* is more northerly (Edmonds et al., 2000). Both *Hylastes tenuis* and *Hylobius pales* were had peaks during fall in prior studies (Zanzot et al., 2010) and considering the population patterns that emerged in this study, it appears autumn peaks typical of the life history of this species. *Dendroctonus terebrans* had a particularly large population peak during July and August following resin sampling while many species were relatively rare. A large increase in *D. terebrans* collection numbers in only a few plots accounted for this peak. Like the addition of panel intercept traps, trapping insects during the fall appeared to increase the accuracy of surveys.

Although this study used different traps than other studies, the same bark and ambrosia beetle species were observed as were reported in other work conducted in the southeastern United States. Many studies trap insects such as *D. frontalis* and *Ips* species with Lindgren funnel traps. Lindgren funnel traps were not used in this study but have captured similar insects as intercept panel traps did at the Oakmulgee Ranger District. Dodd et al. (2010) compared insect capture of intercept panel traps to Lindgren funnel traps and canopy malaise traps in a mature *Pinus strobus* L. stand in New Hampshire and found that total bark beetle capture and species richness did not vary between traps. Ambrosia beetles were captured in greater numbers in the intercept panel traps, but differences in species richness were not found between the three traps. In a Christmas tree plantation in upstate New York, panel intercept traps were used along with Theyson slot traps and Lindgren funnel traps. Funnel traps captured greater totals of *Hylastes opacus* Erichson, Theyson traps captured more *Orthotomicus caelatus* Eichhoff and *I. grandicollis* while *X. saxesenii* did not differ between traps (Petrice et al., 2004). The most common species observed at the Oakmulgee Ranger District reflect data from similar studies elsewhere (Eckhardt et al., 2003, Zanzot et al., 2010) and the utility of pitfall traps to capture insects more specifically attracted to roots has also been observed in previous studies (Hyvarinen et al., 2006). Comparing the findings of these traps to the trapping results from these previous studies, it seems likely that both trapping results appeared to be credible and that the addition of panel intercept traps increased the accuracy of the survey.

3.7 Conclusion

Intercept panel traps proved to be an efficient means to capture a broad range of insect species, including those associated with roots, the lower bole or mid-bole and higher and the best

use of these traps would be in general surveys. Bark and ambrosia beetle species captured by intercept panel traps were similar to captures in Lindgren funnel traps in previous studies in the southeastern United States. When trapping species associated with roots, the use of pitfall traps to supplement panel intercept traps appears to be advisable. Pitfall traps would also be a useful supplement if the capture of live insects is an objective. Flight intercept traps captured the same species as panel intercept traps but in fewer numbers. Panel intercept traps proved to be durable and capable of use over long-term surveys with only minimal maintenance. Year-long intercept panel trapping appear to be applicable in instances when the traps must be sturdy, and a broad range of species are to be collected. Compared to the flight intercept and pitfall traps, intercept panel traps had an obvious superiority at capturing flying insects. Pitfall traps, due to their specialized role in capturing root feeding insects provided a valuable complement. This study was well suited to traps which captured a wide assortment of insect species but more specific traps, such as pitfall traps, would better suit researchers seeking insects living in highly specialized habitats.

Chapter 4

Pretreatment Invasive Plant Survey at the Oakmulgee Ranger District

4.1 Abstract

Non-native invasive plant species are considered to be one of the greatest threats to native biodiversity. Despite an increased focus on invasive plants, most research efforts have been directed toward wild areas. However, many of the forests of the eastern United States are commercially managed and are exposed to some of the known vectors of invasive plants (man-made paths, transport of equipment). An experimental biomass removal that would be more intensive than conventional thinning was scheduled to take place at the Oakmulgee Ranger District in the Talladega National Forest in west-central Alabama. Pre-treatment non-native plant surveys were conducted during summer and fall 2008 and spring, summer and fall 2009. Surveys revealed few non-native plants with *Lespedeza cuneata*, *Lonicera japonica*, *Lespedeza bicolor* and *Albizia julibrissin* present; all of which are widely distributed through the region. Many non-native plants were observed where roads passed through research plots and a higher number were present in compartments predicted to be in loblolly pine decline. Adjoining private property did not have an effect on the presence of non-native invasive plants.

4.2 Introduction

4.2.1 Overview

Invasive plants are exotic, non-native species that colonize and persist in a new environment (Mooney and Drake, 1989). Increases in travel and trade, habitat fragmentation and disturbance escalate opportunities for invaders to expand their ranges and invade forests (Chornesky et al., 2005). Invasive species directly compete with native species for resources, alter ecosystem processes, hybridize with natives and affect gene pools (Richberg and Patterson, 2003). Early successional invaders often opportunistically occupy openings following large disturbances. Non-native exotic plants often grow faster and are subject to less predation than native plants upon arrival in a new habitat (Flory and Clay, 2009). Invaders which eventually become dominant in their new habitat may be capable of making permanent changes in their ecosystems through such means as nitrogen fixation, physical exclusion of native plants or changes in fire regimes (Brown et al., 2006). Techniques used to manage forests facilitate the spread and establishment of invasive species through introduction via contaminated machinery, equipment or materials and canopy and soil disturbance. Right of ways, stream banks, roads and trails are important pathways in the movement of invasive plant species (Miller, 2002). Areas of urban/forest interface, sometimes near commercial forestry operations, are also known to be especially vulnerable to invasive species. Many invasive species, especially those at the urban/forest interface, are escaped ornamental species, many of which are still legally sold by the landscaping industry (Miller, 2002).

4.2.2 Causes of invasiveness and resistance of native ecosystems

Susceptibility of ecosystems to invasion by non-native organisms is affected by a variety of factors including climate, disturbance regime and competitiveness of native species (Lonsdale, 1999). Various theories suggest that invaders are superior competitors for limited resources, that new environments enable invasive plants to escape natural enemies or even that invasiveness is an inherited characteristic and survival strategy (Inderjit et al., 2005). In some instances co-introduced enemies of the invading plant species may also attack native plants as well.

Humans transport plants to new environments regularly but of thousands of introduced species, relatively few survive outside of cultivation in their new habitat. Of those that survive, few become established and only a small proportion of those become invasive (Williamson and Fitter, 1996). The numerous theories explaining success or failure of invasions are not mutually exclusive. So many site specific and sometimes confounding factors are involved with each individual invasion that broad generalizations are difficult to make (Ewers and Didham, 2006) and global hypotheses are difficult to test in isolation (Inderjit et al., 2005). In addition, the lag between initial introduction and when a species becomes invasive can be long, adding to the challenge of identifying problem species (Mack et al., 2008).

4.2.3 Land management and its effects

Difficulties inherent in long-term monitoring and translation of results of studies conducted in different regions have resulted in little analysis of long-term effects of forest management on invasive plants. The affects of commercial forestry activities on invasive species are poorly understood (Ohio Invasive Plant Conference, 2005). In lodgepole pine (*Pinus*

contorta Dougl) stands in British Columbia, Canada overall species richness and diversity were higher in thinned stands than in unthinned stands. While few of the plants in the study were invasive, invasives were more prevalent after the treatment (Lindgren et al., 2006). A study using a wider variety of treatments in Oregon found that the only management activities to facilitate invasion were those that created gap environments (Boggs and Puettmann, 2005).

4.2.4 Invasive plant observations and mapping

Before invasive species can be treated they must be found and mapped. Many surveying techniques exist, but ideal methods of invasive plant monitoring should combine efficiency and accuracy. Of seven tested survey methods, the most consistent invasive plant population estimates were obtained by using a grid and random points following targeted transects (Rew et al., 2006). Surveying areas with no known populations of non-native invasive plants is believed to better detect outlying populations (Maxwell et al., 2009). Finding satellite populations is important as these areas have the greatest potential to increase the overall range of the species (Radosevich et al. 2003). Monitoring of outlying areas may be useful for this reason, even if nothing is found.

Biomass removal treatments are to occur on the Oakmulgee study plots, reducing plot basal area from approximately 120 to 70 sq.²/ac. (37 to 21 m²/ha). Effects of forest thinning on invasive plant populations have been studied infrequently (Ohio Invasive Plant Research Conference, 2005) and those studies have produced inconsistent results (Beggs and Puettmann, 2005). Even less is known about the effects of biomass removal on invasive plant presence. The objective of this study is to determine levels of invasive non-native plant populations. Aspects of

thinnings that can increase invasive plants may be amplified in a biomass removal. Because of the creation of roads and the presence of machinery that may transport plants and propagules, the presence of invasive plants is expected to be greater following these treatments.

4.3 Methods and Materials

The study was conducted at the Oakmulgee Ranger District of the Talladega National Forest in Bibb, Perry and Chilton counties in west-central Alabama on 24 plots located within the district. Eighteen plots were located within compartments considered to be in pine decline while 6 plots were in predicted non-declining compartments. Invasive non-native plant surveys were conducted in research plots, where insect collection traps were already established (Chapter 2).

Invasive plant surveys were conducted during June and November 2008 and May, August and November 2009. Four transects extending 122 m north, east, south and west from plot center were marked and non-native plants within 0.3 m on either side of the transect were noted. Occurrence of invasive plants was monitored along transects that extended 37 m from center plots to the insect traps at 120, 240 and 360 degrees supplemented by a circular transect between the traps themselves (Figure 4.1). Transect lines were abbreviated when steep terrain, large creeks or private property boundaries were encountered. Intersections with roads (timber or public) were recorded. Private property within 122 m was also noted. Representative plant specimens of known invasive and unidentified species were placed in a paper bag, pressed and taken to the Auburn University Herbarium (Auburn, AL) for confirmation of field identifications.

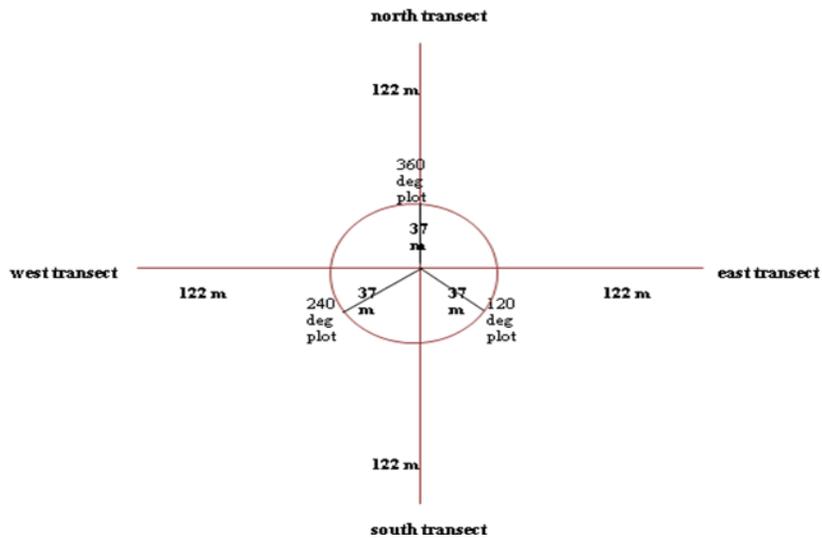


Figure 4.1. Invasive plant transect layout at Oakmulgee Ranger District Research Plots 2008-2009.

4.4 Statistics

Binomial logistic regression on SAS 9.1 (SAS Institute, Cary, NC) was used due to the extremely frequent occurrences of zeroes in the data set (no invasive species). This enabled the data to be analyzed on a presence/absence basis and also enabled species which grow as individual shoots (e.g. *Albizia julibrissin* Durazz.) to be considered on the same scale as clonal species which spread via underground rhizomes (e.g. *Lespedeza cuneata* (Dum. Cours.) G. Don. var. *cuneata*). When binomial regression was performed, maximum likelihood and odds ratio findings were often absent, making statistical analysis impractical.

4.5 Results

4.5.1 Overall observations

Total observations of non-native plants varied by season in which the survey occurred. The total number of non-native plant observations were greatest during the August 2009 survey (56 total observations), followed by the May 2009 survey (37 total observations), October 2009 survey (31 total observations), June 2008 survey (28 observations) and November 2008 survey (10 observations) (Table 4.1). Non-native plants were most common during the summer months.

Table 4.1. Total number of observations of non-native species by survey season at research plots at the Oakmulgee Ranger District, Talladega National Forest.

Species	Summer 2008	Fall 2008	Spring 2009	Summer 2009	Fall 2009
<i>Lonicera japonica</i>	4	4	15	12	5
<i>Ligustrum sinense</i>	0	0	0	0	1
<i>Lespedeza bicolor</i>	8	0	10	17	10
<i>Lespedeza cuneata</i>	12	6	10	17	12
<i>Paspalum dilatatum</i>	1	0	0	0	0
<i>Paspalum notatum</i>	3	0	0	0	0
<i>Albizia julibrissin</i>	0	0	2	10	3

4.5.2 Invasive plants and loblolly pine decline

There were a greater number of non-native plants in the predicted decline plots compared to the predicted non-decline plots. Every non-native species was more common in the predicted decline plots and that was not a function of a greater number of predicted decline plots. For example, during the summer of 2009, predicted decline plots contained all observations of *Lonicera japonica* Thunb., all 17 observations of *Lespedeza bicolor* Turcz., 13 of 17

observations of *L. cuneata*, and 9 of 10 observations of *A. julibrissin*. These disproportionate findings were representative of the overall findings (Table 4.2).

Table 4.2. Comparison of non-native plant observations in plots with or without loblolly pine decline by survey season at research plots at the Oakmulgee Ranger District, Talladega National Forest. D (n=18) indicates declining plots, N (n=6) indicates non-declining plots.

Season		<i>L. japonica</i>	<i>L. sinense</i>	<i>L. bicolor</i>	<i>L. cuneata</i>	<i>P. dilatatum</i>	<i>P. notatum</i>	<i>A. julibrissin</i>
Sum 08	D	3	0	7	12	1	2	0
	N	1	0	1	0	0	1	0
Fall 08	D	4	0	0	6	0	0	0
	N	0	0	0	0	0	0	0
Spr 09	D	10	0	7	9	0	0	2
	N	5	0	3	1	0	0	0
Sum 09	D	12	0	17	13	0	0	9
	N	0	0	0	4	0	0	1
Fall 09	D	5	1	8	12	0	0	3
	N	0	0	2	0	0	0	0

4.5.3 Effects of road and private property interceptions

Non-native plants were more common in plots where roads passed within 37 m of plot center (n=14) than plots in which roads were further away (n=10) (Table 4.3). While the majority of plots had a road within 37 m, a disproportional number of non-native plant observations came from these plots. During the summer 2009 survey, all 12 *L. japonica* observations, all 17 *L. bicolor* observations, 15 of 17 *L. cuneata* observations and all 10 *A. julibrissin* observations were at plots transected by roads. A large majority of non-native plants were observed by roadsides in other study periods.

Table 4.3. Comparison of non-native plot observations between plots with or without roads within 37 m of plot center at the Oakmulgee Ranger District, Talladega National Forest. ‘Yes’ (n=14) indicates roads within 37 m of plot center, ‘No’ (n=10) indicates no road. Roads consisted of logging trails, unpaved public roads and paved public roads.

Season	Road	<i>L. japonica</i>	<i>L. sinense</i>	<i>L. bicolor</i>	<i>L. cuneata</i>	<i>P. dilatatum</i>	<i>P. notatum</i>	<i>A. julibrissin</i>
Sum 08	Yes	3	0	8	12	1	3	0
	No	1	0	1	0	0	0	0
Fall 08	Yes	4	0	0	6	0	0	0
	No	0	0	0	0	0	0	0
Spr 09	Yes	10	0	8	9	0	0	2
	No	5	0	2	1	0	0	0
Sum 09	Yes	12	0	17	15	0	0	10
	No	0	0	0	2	0	0	0
Fall 09	Yes	4	0	8	12	0	0	3
	No	0	1	2	0	0	0	0

4.5.4 Effects of prior thinning

In addition to roads, some of the plots were exposed to additional disturbance and possible introduction to non-native plants when stands were thinned. While only a third (8 of 24) plots had been thinned, the majority of non-native plant observations were made in stands that had been thinned (Table 4.4). Aside from a greater number of *L. japonica* in unthinned plots and an equal number of *L. bicolor* in thinned and unthinned plots during the summer 2009 surveys, greater totals of non-native plants were detected in the thinned plots.

Table 4.4. Comparison of non-native plot observations between thinned in and unthinned plots at the Oakmulgee Ranger District, Talladega National Forest. ‘T’ (n=8) indicates thinning had occurred at the plots, ‘U’ (n=16) indicates plots had never been thinned.

Season		<i>L.</i>	<i>L.</i>	<i>L.</i>	<i>L.</i>	<i>P.</i>	<i>P.</i>	<i>A.</i>
		<i>japonica</i>	<i>sinense</i>	<i>bicolor</i>	<i>cuneata</i>	<i>dilatatum</i>	<i>notatum</i>	<i>julibrissin</i>
Sum 08	T	3	0	5	7	1	2	0
	U	1	0	3	5	0	1	0
Fall 08	T	4	0	0	5	0	0	0
	U	0	0	0	1	0	0	0
Spr 09	T	6	0	5	8	0	0	1
	U	9	0	5	2	0	0	1
Sum 09	T	10	0	15	11	0	0	6
	U	2	0	2	6	0	0	4
Fall 09	T	3	1	6	8	0	0	3
	U	2	0	4	4	0	0	0

4.6 Discussion

Many prominent invasive species in Alabama were not observed on the study sites. While the possibly exists that non-native plant observations may increase following thinning or biomass removal, only a few species of non-native plants were regularly observed along the research transects. Many of the observations were unsurprising given the widespread distribution of species such as *L. japonica* (Schierenbeck, 2004). As expected, these species were most common along roadsides and in previously disturbed areas but surprisingly, non-native presence was all but non-existent in the few plots bordering private lands. It is hypothesized that follow up surveys expected to follow thinning and biomass removal will detect more invasive plants and that biomass removal will increase non-native plant presence more than conventional thinning. The species currently at the plots are capable of quickly responding to disturbance and may rapidly fill openings that this work leaves behind. Follow up studies may

subsequently test if non-native plants respond to thinning as expected and increase from their current low populations.

Thinning operations and biomass removal are expected to occur at the Oakmulgee Ranger District but little is known of the effects these treatments have on invasive plants. Most observations of invasive plants were associated with either roads or past stand disturbance, indicating that additional management may introduce non-native plants into new areas. Invasive species were rarely observed on transects that were not intercepted by a road within 132 m of plot center. Several invasive species observed in this study are shade intolerant and benefit from open areas along roadsides. Even plant species capable of thriving under shade, such as Japanese honeysuckle, were more abundant by roadsides exposed to light and disturbance. This indicates openings created in the forest by a biomass removal project could facilitate the spread and establishment of both shade tolerant and shade intolerant species.

Previous forest management activities in the National Forest may have contributed to the establishment of non-native plants. Plots that were thinned in 1996 and 1999 tended to have greater numbers of invasive plants compared to those which had not been thinned since regeneration. Thinnings may have created opportunities for invasive plants through either increased light or through movement of propagules on machinery. Species such as *L. cuneata* may have spread along logging roads used during the thinning operations. The previous thinning conducted at the plots appears to have had long term consequences, a pattern not always observed following thinning (Dodson, 2004), but worth considering as further work is conducted at the plots.

More invasive plants were observed in plots located in predicted loblolly pine decline compartments, even when considering the larger sample size of predicted decline plots compared to predicted non-decline plots. Potential associations between pine decline and increased presence of invasive plants have not been tested but under decline conditions loblolly pine crowns are less dense and more sunlight passes through. Increased light penetration to the forest floor may benefit the invasive plants (Snitzer et al., 2005). However, none of the three predicted non-decline pairs were within 122 m of a public road. That decline is a factor in invasive plant presence cannot be stated with confidence until tested without confounding factors such as the affects of roads. Loblolly pine has been so widely distributed across both the area and region that any hypothetical widespread dieoff could be expected to create openings in many areas for opportunistic species, including many invasives.

Adjacent private property was expected to facilitate the spread of non-native plants. Adjoining private areas included recently thinned private forests near several research plots and a residential lawn bordered another plot. While many invasive species have been introduced to new environments as escaped ornamentals, ornamental species did not appear to be invading National Forest land from private property. Plots that bounded private property did not have higher totals of invasive species. There were no non-native plant observations at any plots with private property within 122 m of plot center. This was probably because all three of these plots were located more than 122 m from roads. Given that in some instances the plot bordered either private yards or stands that had been recently thinned, road effect appeared to be stronger. Only two of the plots, however, directly bordered residential yards, so it may not be possible to make a strong comparison. The eastern half of the District in particular, however, has many houses

surrounded by public land so a stronger comparison could be made if different study areas were selected.

Increases in survey counts of perennial non-native plants may have been as much a function of human error as actual population increase given the difficulty of walking identical transects between different survey periods. Multiple pretreatment surveys occurred because thinning treatments were expected to take place between the first and second years of the study. If transects can be followed with more accuracy in different surveys, the method used to survey invasive plants in this study is still expected to capture non-native plant diversity in both thinned areas and unthinned areas nearby. Potential changes outside the treatment areas could indicate the extent to which changes in non-native plant populations in disturbed areas can affect adjoining non-disturbed areas.

A biomass removal study consisting of a complete removal of all woody material greater than 7.5 inches in diameter at breast height may be conducted in the near future. The level of disturbance associated with roads and other access, the use of off-site machinery and the opening of the overstory all could increase the presence of invasive plants. In light of the shortage of research regarding the impact of forest management on invasive plant levels (Ohio Invasive Plant Conference, 2005) a follow up survey investigating the effect of biomass removal on invasive plant species would be of great interest. Few studies have considered the response of invasive plant species to management, despite the widespread belief that disturbance facilitates invasion. Dodson (2004) found that invasive plants spread after thinning treatments, particularly when burning accompanied thinning. However, general studies show inconsistent response of non-native plants to thinning. In an Oregon study conducted five years after several different

types of thinning, the overall presence of herbaceous plants increased but few long-term changes persisted in the understory (Beggs and Puettmann, 2005). Because of the inconsistent response of non-native invasive plants to forest management, potential changes in the presence of invasive plants following biomass removal treatment are difficult to predict and unlikely to become apparent until years after the treatment. Because non-native plants were more prevalent in previously thinned plots, one could reasonably predict that a more intensive treatment such as biomass removal would result in an increase in invasive plants.

4.7 Conclusions

Many of the non-native species observed in this study are widely distributed species associated with disturbance but none are considered to be among the most ecologically harmful. Overall, the plots appeared to be relatively free of invasive plants but non-native species were generally more common where thinning had been conducted in close proximity to roads. Private property boundaries were not associated with elevated numbers of invasive plants. Areas that had been thinned tended to have greater numbers of invasive plants, primarily along old logging roads. National Forest lands often have fewer human disturbances and roads than surrounding areas but even on the National Forest plots, roads and thinning appeared to influence the distribution of invasive plants. Additional disturbances or road construction following biomass removal treatments are likely to increase numbers of invasive species. The probability that invasive plants will exploit disturbances relating from future management will be affected by the degree of disturbance, the response of native plants to the treatments and the dispersal of invasive plant propagules onto the site. The response of invasive plants to biomass removal is

important to understand given the increasingly widespread public discussion of harvesting biomass for fuel.

Chapter 5

Summary and Conclusions

5.1 Overview

This project measured bark and ambrosia beetle populations and insect diversity in advance of biomass removal and conventional thinning treatments and associated insect levels with tree health measurements and predicted loblolly pine decline. Crown transparency was the characteristic most associated with elevated insect populations. This broad survey of bark and ambrosia beetles showed a high proportion of species in plots that could vector pathogenic fungi. Previous thinning regimes appeared to reduce the numbers of some bark and ambrosia beetles and could be a valuable tool in managing stressed areas. Intercept panel traps effectively captured a broad range of flying insects. A non-native invasive plant survey was completed in anticipation of a biomass removal. Few non-native invasive plants were observed, but if biomass removal is practiced, it will now be possible to see if stand management affects invasive plant establishment.

5.2 Insect diversity

More individuals were captured within the genus *Hylastes* than any other insect genus, with *H. salebrosus* Eichhoff being the most numerous insect species. *Hylastes*, as well as other insect genera that are not associated with primary damage in commercial tree species have been

more sparsely documented than southern pine beetle (*Dendroctonus frontalis* Zimmermann) and engraver beetle species (*Ips* spp.). Their numbers alone may give them an importance often not considered in prior studies, especially given their role as a vector of pathogenic fungi. Because they do not kill host trees, *Hylastes* have been a practical species to study in seeking the relationship between insect vectors their associated fungi and host trees (Eckhardt et al., 2004). Transport of fungi into roots is believed to weaken the defenses of trees through long term nutrient depletion and lack of water conduction due to deterioration of fine roots (Eckhardt et al., 2010). It may be worth considering the possibility that *Hylastes* may damage the trees through simple physical damage to root systems if present in extremely high numbers collected in through much of the collection period.

The most interesting insect observation was that predator beetles (*Thanasimus dubius* Fabricius and *Temnochila virescens* Fabricius) were not associated with other potential prey beetles. This finding reinforces the hypothesis that *T. dubius* responds primarily to frontalin, the attractant chemical for southern pine beetle (Raffa, 2001) and generally ignores other bark beetles. *Dendroctonus frontalis* was almost entirely absent from the research plots during the two-year collection period and it would be interesting to compare proportions of bark and ambrosia beetle species found in this study to those during long-term periods when southern pine beetle outbreaks occurred. Previous studies have examined population fluctuations of predator beetles and other primary bark beetles, such as *Ips* during southern pine beetle outbreaks, but work has seldom ventured beyond species not associated with widespread damage.

Insect capture was greater for several insects at predicted loblolly pine decline plots than predicted non-decline plots. More insects were expected within declining plots as a result of

anticipated lower tree resistance. Several of the species that were more common in declining stands were ambrosia beetles. Because ambrosia beetles are known to primarily exploit weakened trees their greater presence may be expected in declining areas.

Foliar characteristics such as crown transparency were correlated with insect collection totals. Greater crown transparency was associated with higher insect counts. Distressed appearance of needles is typically a sign of physical distress in pine species (Ustin, 2008). Highly stressed trees are unable to effectively resist insect attacks, which in turn additionally weakens the stressed trees.

Traps used in the survey were placed near one another and the number of traps at each plot may have affected insect collections. Nine traps were located within 37 m from plot center and the three traps at subplots were separated by only approximately 3 m. Some studies have concluded that placing traps within 20 m of one another may lead to interference which complicates detecting the effects of individual traps (Sappington, 2002). Bark beetle attractants have been observed to have an attractive radius of over 100 m (Chen et al., 2010). Interference between traps could have potentially affected both total concentration of attractants and perhaps the trap types in which they were captured. Follow up research could test whether the insects that were attracted to one of the traps at a subplot were ultimately captured by a different trap at the subplots.

Most of the captured bark beetles were native species while some of the commonly captured ambrosia beetles were non-native, such as *Xyleboriunus saxesenii* Ratzburg and *Xylosandrus crassiusculus* (Motschulsky). The large capture of non-native ambrosia beetles

reflected findings at prior studies in the southeastern United States. A higher than usual proportion of *Gnathotrichus materiarius* Fitch was captured during the winter. Had these peaks not been observed, the most common species in these studies would have been non-native as well.

5.3 Trapping conclusions

Intercept panel traps appeared to capture similar bark beetle species as Lindgren traps did in other studies. In most of those prior studies bark beetle and weevil species that have been primarily associated with roots have been trapped in pitfall traps (Johnson et al., 2010). That the majority of *Hylastes salebrosus* Eichhoff were caught in panel intercept traps during this study implies a greater capability of flight intercept traps to capture root dwelling species and supplement the use of pitfall traps than generally considered. The use of pitfall traps remains advisable as the smaller plumes of attractant pheromones associated with pitfall traps and their much smaller entrances, particularly for larger species, lead to pitfall trap captures being much more specialized in what they captured. Fewer studies use flight intercept traps to trap bark beetles. The milk jug design used by this study proved to be fragile over the duration of the study. Traps exposed to direct sunlight needed to be replaced several times, as a result of physical breakdown at several points (the insect cup and at the attachment to the pole).

The ideal trap depends primarily on the objectives of the study. For example, the use of pitfall traps would be advised over panel intercept traps in studies which require live insect trapping. Live insects are needed when studying insect behavior, life history or when collecting fungi and other associated organisms from insects (Eckhardt et al., 2004, 2007). In studies

which seek to capture insect diversity, estimate current or future populations of problematic pest species or trap potential invasive species, aerial traps such as a Lindgren funnel or a panel intercept are more appropriate given the broader diversity of species captured by them (Weinzieri et al., 2005).

Several population peaks occurred during this study outside of the usual spring and summer trapping window. Use of long-term trapping beyond the Talladega National Forest could result in a more accurate understanding of underreported insect species. While many species more accurately observed from the addition of winter trapping may have negligible commercial importance, early detection of invasive insects could potentially be improved as non-native species expand their range (Brockerhoff et al., 2006).

5.4 Potential biomass work

Biomass is widely expected to be an increasingly important source of energy in the future (Perschel et al., 2007). Given that wood, if affordably converted, could be more efficiently converted to energy than food crops it is important to understand the possible ecological effects in areas such as the southeastern United States (Lattimore et al., 2009). If similar work is carried out elsewhere, comparison of pre-treatment and post-treatment data from the Oakmulgee Ranger District could provide an example of changes in insect population following disturbances such as biomass removal (Mitchell, 2007).

Biomass removal was expected to take place throughout the District in 2008 at 4 of the 12 paired plots. All understory material with DBH lower than 7.5 inches would be removed and a stand density of 40 to 60 would be obtained. Material removed from the biomass treatment

plots would be chipped, completely removed from the plot and co-fired with coal at a power plant in Gadsden, Alabama (Mitchell, 2007). Changes in forest health in the research plots are expected to be reflected by crown density and direct comparison of pre-treatment and post-treatment insect collections.

A short-term increase of insects would be expected at the plots given a conventional thinning treatment due to the widespread release of attractant chemicals when trees were cut. Multiple studies have detected a short term increase in bark and ambrosia beetle species after thinning treatments (Fettig et al., 2006), followed by an eventual return to baseline levels and an increase in the growth rate of the remaining trees (Fettig et al., 2007). Plots that receive a conventional thinning are expected to have an increase in bark beetles. Insect collections following the biomass removal plots would be less predictable. As at conventional thinning sites, chemical attractants would be released during the disturbance. While insects would be attracted, complete removal of material from the site would remove much of the insect habitat. Trees left on the research sites following biomass removal would theoretically be released from competition (as they are at conventional thinning) and are expected to undergo more rapid growth and more effectively flush out even a temporarily increased number of bark and ambrosia beetles.

5.5 Non-native Invasive plant observations

Few studies have been conducted on the potential effect of biomass removal on the spread of non-native invasive plants. In addition to the drastic changes in light availability and stand density after biomass removal, heavy equipment would be moved to work areas and at

some study plots temporary roads would be constructed to provide access. These activities have been associated with the spread of non-native invasive plants.

Some opportunistic invasive plant species appear to increase in the short-term following biomass removal, but those changes recede as natural succession proceeds (Dodson, 2004) and forest thinning operations are not typically associated with an increase in the presence of invasive plant species (Beggs and Puettmann, 2003). The ecological hazards of invasive species are now recognized enough that protocols are in place to reduce the chances of invasion. Potential connections between silvicultural forest treatments and in the spread of invasive plants have not been studied.

Many exotic plants considered to be the most ecologically hazardous by the Alabama Invasive Plant council are believed to be either absent or rare at the Oakmulgee Ranger District at the time of this writing. The invasive plant species observed in the study are widely distributed across the state of Alabama (Miller and Chambliss, 2008). Cogongrass (*Imperata cylindrica* (L.) P. Beauv.) is widespread in south Alabama, primarily by roadsides and open areas but only a few isolated patches of cogongrass have been detected in the Oakmulgee Ranger District and none were observed during the survey.

Few invasive plant species were observed in the plots. In many ways the plots at the Oakmulgee Ranger District do not fit the profile of an area which would be heavily occupied by invasive species. Most of the area within the National Forest was cotton acreage until forests were planted to prevent soil erosion during the New Deal era (Shands, 1992). Since that time, the Oakmulgee Ranger District has been under less land development pressure and human

release of invasives than many surrounding areas. Paved roads nevertheless run through the park and patches of private property remain within the National Forest. These patches are more common in the eastern half of the district, where work was conducted and some of the research plots were surrounded by private property, but this had no effect on the presence or absence of invasives.

While there was not enough data to perform statistical analysis, it appeared as though previous thinning operations in plots may have led to an increase in non-native invasive plants. Most observations of non-native invasive plants appeared to be found by human-made paths. If true this would seem to confirm that disturbances and conduits during previous work may have favored the presence of non-native plants. Disturbances benefit non-native invasive plants and human activities tend to create disturbances.

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