

**ACADIAN FLYCATCHER RESPONSE TO DETERIORATING EASTERN HEMLOCK
CONDITIONS IN TWO HEMLOCK DOMINATED HABITATS IN THE DELAWARE WATER
GAP NATIONAL RECREATION AREA AND DELAWARE STATE FOREST**

By

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A Thesis Submitted in Partial Fulfillment of
The Requirements for the Degree of
Master of Science in Biology
To the Graduate College of
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APPROVAL PAGE

This thesis by Megan Napoli submitted to the Graduate College in partial fulfillment of the degree of Master of Science in Biology on May 8, 2015 has been examined by the following faculty and it meets or exceeds the standards required for graduation as testified by our signatures below.

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ABSTRACT

A Thesis Submitted in Partial Fulfillment of the Requirement for the Degree of Master of Science in Biology to the Graduate School of East Stroudsburg University of Pennsylvania

Student's Name: Megan Napoli

Title: ACADIAN FLYCATCHER RESPONSE TO DETERIORATING EASTERN HEMLOCK CONDITIONS IN TWO HEMLOCK DOMINATED HABITATS IN THE DELAWARE WATER GAP NATIONAL RECREATION AREA AND DELAWARE STATE FOREST

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Abstract

Eastern hemlock (EAHE) is an essential tree found normally along riparian corridors whose stands create a unique habitat that governs ecological functions and distinct communities. The biggest threat to these EAHE forests are an invasive insect from Japan, the hemlock woolly adelgid (HWA) which are spreading rapidly and desiccating whole stands of this tree. The Acadian flycatchers (ACFL) show preference for nesting in EAHE in Delaware State Forest (DSF) and the Delaware Water Gap National Recreation Area (DEWA) and therefore, can act as a bioindicator to deteriorating EAHE conditions. To understand the response of the ACFL to changes in hemlock stands, comparisons were made to a previous study conducted in the same locations (Sheehan 2003) and between two habitat types, ravines and benches that exhibit varying levels of hemlock condition. Results indicated a significant decrease overall EAHE health between study years as well as in ACFL pair density. An increase in nesting in various deciduous tree species was also found, as was a significant increase in nest height in EAHE. Ravines were found to have significantly lower pair/fledgling densities, greater understory mean density and greater abundance of EAHE in poor health. These results support that EAHE health has and continues to deteriorate in DSF and DEWA, and that bench sites are generally more productive and contain healthier stands of EAHE than ravines making them a higher priority for HWA management in DSF and DEWA.

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Chapter I.

INTRODUCTION

Eastern Hemlock: Importance in the Riparian Ecosystem

The eastern hemlock (EAHE; *Tsuga canadensis*) is a coniferous tree that is native to the northeastern United States. This tree's range extends from Nova Scotia southward through New England, the Mid-Atlantic States and to northern Georgia/Alabama and westward through the southern provinces of Canada, northern Michigan, Wisconsin, and some portions of Minnesota (USDA 1965a). The hemlock is the longest living and most shade tolerant tree in the northeastern U.S. and is normally the most dominant tree in areas where it occurs (Quimby 1996, Evans 2002). Eastern hemlock has been described as a foundation species, a common or abundant species that defines microclimate and processes of an ecological community (Ellison et al. 2005). Eastern hemlock functions as a foundation species by suppressing plant diversity, resource availability and regulating nutrient and energy cycling (Martin and Goebel 2012). The structure of this tree is unique; its branches extend nearly to the forest floor and it has two-ranked needles that spread sideways into a flattened plane (Ward et al. 2004). Because of this unique structure, mature hemlock stands form a dense canopy and create a microclimate that is cool, moist and allows little penetration of sunlight.

The combination of shade and acidic soils (generated by shed needles) suppresses herbaceous and understory growth, leaving much of the ground under hemlock stands bare (Snyder et al. 2005).

This distinctive microclimate creates a biotic community different from surrounding deciduous woodlands with unique species and high biodiversity (Howe and Mossman 1995, Snyder et al. 2002, Ross et al. 2003, 2004). Mammals including, black bear (*Ursus americanus*), bobcat (*Lynx rufus*), snowshoe hare (*Lepus americanus*), red squirrel (*Tamiasciurus hudsonicus*), southern red-backed vole (*Myodes gapperi*), and northern flying squirrel (*Glaucomys sabrinus*) are to varying extent dependent on the hemlock forest (Yamasaki et al. 2000). Plethodontid salamanders, such as the eastern red-backed (*Plethodon cinereus*), northern two-lined (*Eurycea bislineata*), and northern dusky (*Desmognathus fuscus*) salamanders, are abundant within the hemlock forests and rely on the more moist and cool environment for their ecophysiological requirements (Brooks 2001, Kizlinski et al. 2002). Streams shaded under hemlocks contain a higher abundance of brook (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) than deciduous forest streams, as well as numerous representatives of the aquatic invertebrate orders Ephemeroptera, Plecoptera, and Trichoptera (Evans et al. 1996; Ross et al. 2003). There are approximately 97 bird species commonly found in hemlock forests, with several species being obligate or near obligate inhabitants. These species include the black-throated green warbler (*Setophaga virens*), Blackburnian warbler (*Setophaga fusca*), blue-headed vireo (*Vireo solitarius*), and the Acadian flycatcher (*Empidonax virescens*), among others (Benzinger 1994b; Ross et al. 2004).

Normally found along the edges of moist riparian corridors of headwater streams, the eastern hemlock occurs in dense stands within mixed deciduous forests (Quimby 1996). These headwater streams are classified into two main stream types; (1) ravines (Fig. 1), having one or two steep slopes descending to the stream with water flowing down an inclined gradient and (2) benches (Fig. 2), with slow moving, braided streams meandering across a flat floodplain (Ross et al. 2004). Previous studies

conducted in the Delaware River Basin by Ross et al. (2004) reported higher bird species richness and territorial densities on benches compared to ravines, but they did not focus on a particular species or potential causes of this disparity. Ernst (2012) and Barnes (2014) reported that Louisiana waterthrushes exhibit significantly higher pair densities, fledglings produced per km of stream reach and incidence of double brooding on hemlock benches suggesting that benches support higher ecological productivity than ravines. This study uses the Acadian flycatcher as a bioindicator of both habitat productivity and the ongoing effects of deteriorating hemlock condition. Together, the two species complement each other with regard to the portions of the riparian habitat that they are regularly exposed to thus providing a broader picture/reflection of impacts to their overall headwater stream environment.



Figure 1. Ravine stream type; Tumbling Waters, DEWA, Pike County, PA. (Photo by author Megan Napoli)

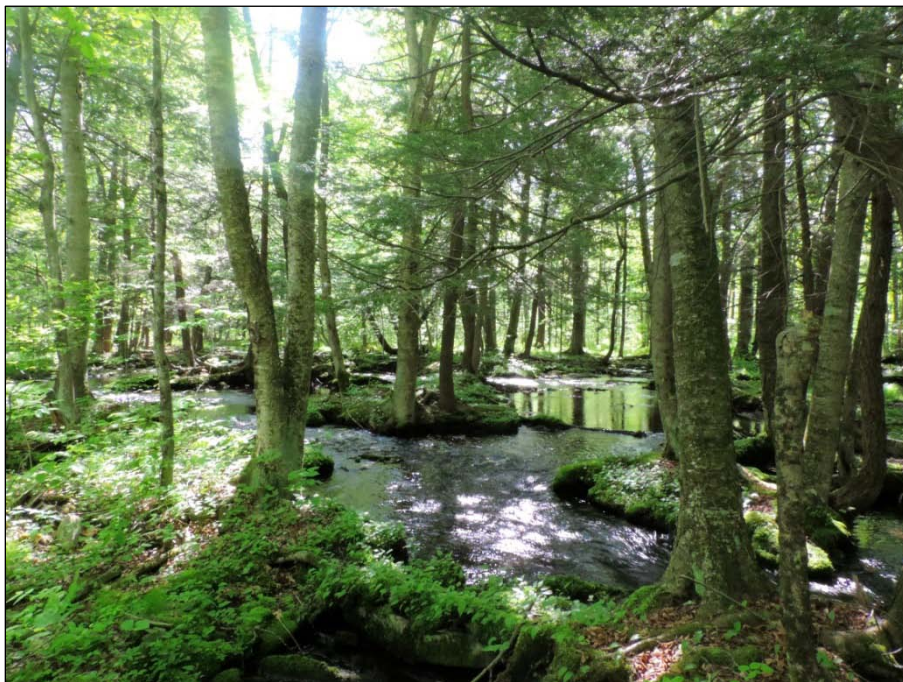


Figure 2. Bench stream type; Spruce Run, DSF, Pike County, PA. (Photo by author Megan Napoli).

Hemlock Woolly Adelgid

The hemlock woolly adelgid (HWA; *Adelges tsugae* [Insecta: Hemiptera: Adelgidae]) is a very small (1-2 mm in length) aphid-like insect that is native to East Asia. In its native habitat, this insect is normally harmless to the Asian hemlock species it inhabits, as it is to the North American Western Hemlock (*Tsuga heterophylla*). After traveling east from the western U.S., it quickly became apparent that it caused extensive damage to eastern hemlock and the closely related Carolina hemlock (*Tsuga caroliniana*) (Cheah et al. 2004). HWA was first observed in North America in the 1920s on western hemlock and mountain hemlock (*T. mertensiana*) in Oregon (Cheah et al. 2004). By 1951, HWA had made its first appearance in the eastern U.S. in Richmond, VA and by 1989 it was found within the boundaries of the Delaware Water Gap National Recreation Area (DEWA) (Cheah et al. 2004; Evans 2004). This invasive insect can now be found in 19 eastern states, from northeastern Georgia to southwestern Maine, with recent observations in southeastern Ohio, northern Virginia, midwest Tennessee, western New York, and mid-northern New Hampshire (Fig. 3).

In North America, adelgids are parthenogenetic (only producing females) and produce two generations a year, the sistens (winter generation) and the progrediens (spring generation) (Cheah et al. 2004) (Fig. 4). The sistens eggs hatch in late spring and these first instar nymphs are called “crawlers”. Crawlers are the progrediens generation that relocate to the bases of needles on the tree to feed on sap (Fig. 5). While feeding, the nymphs are covered in a white, woolly wax to protect

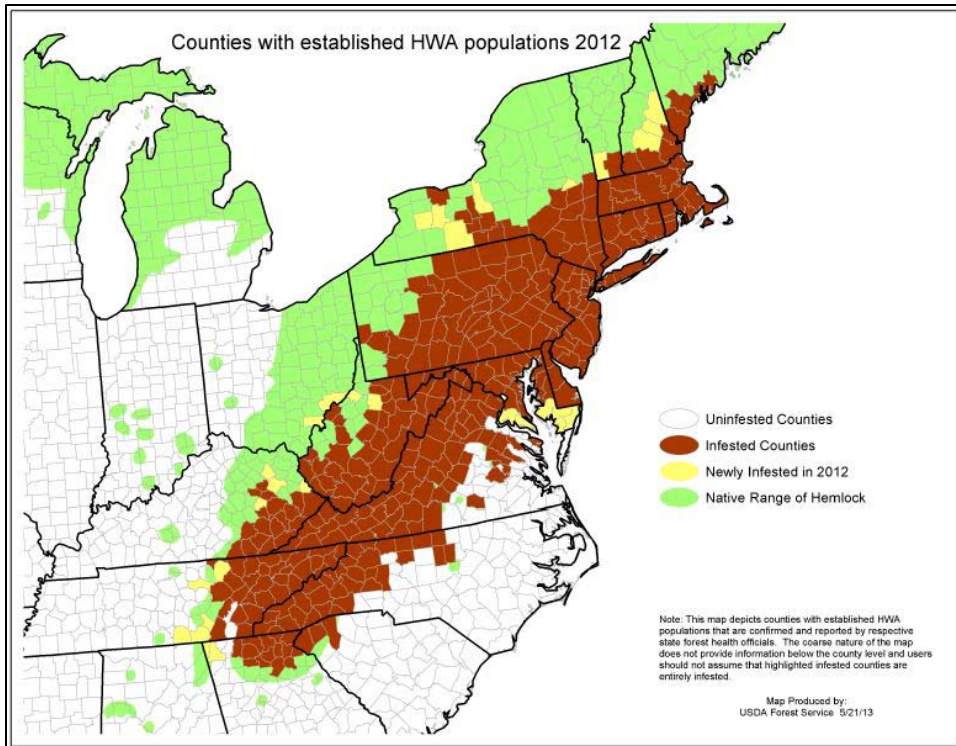


Figure 3. Distribution map of HWA in the eastern United States. Produced by USDA Forest Service: (<http://na.fs.fed.us/fhp/hwa/maps/2012.pdf>)

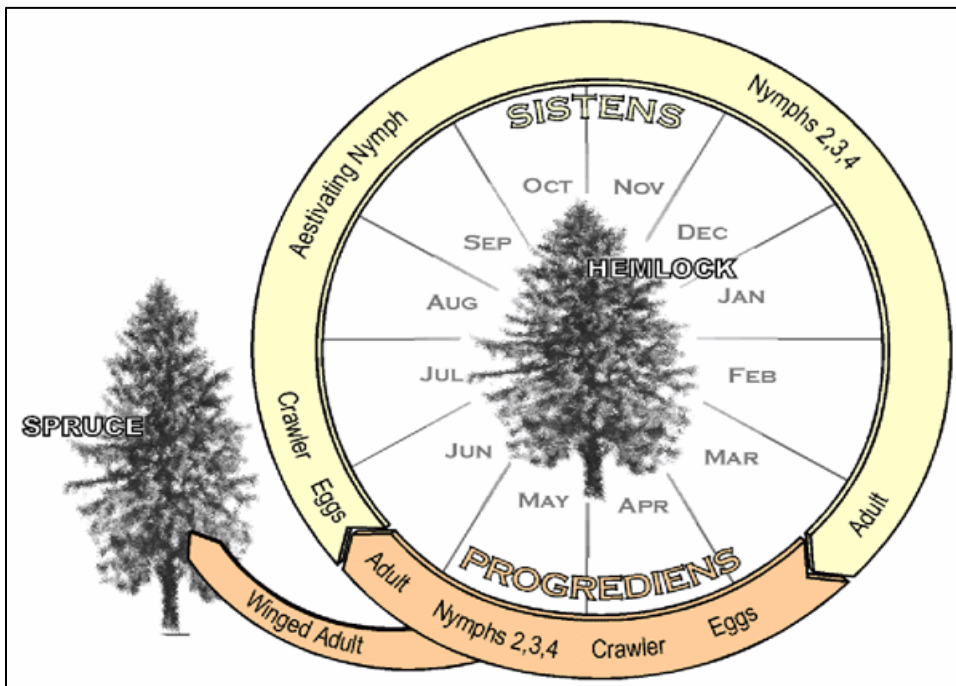


Figure 4. HWA annual life cycle on hemlock in North America (From Maine Forest Service) (<http://www.forestryimages.org>).

themselves from outside elements. Through spring and early summer the nymphs develop into adults, with some developing wings and others remaining wingless. The winged adults, called “sexpara”, fly from the hemlock in search of a suitable species of spruce on which to deposit eggs. Because there are no suitable spruce species in North America, this portion of the population dies before sexual reproduction (Cheah et al. 2004). The wingless adults lay up to 125 eggs in mid-June, which hatch by July (adults and eggs can be found under the white, woolly wax for protection; Fig. 5). These crawlers are the sistens generation and relocate to the base of needles and become dormant until autumn. In the cool days of autumn, these nymphs become active and start to feed on the sap of the tree. Through the winter they develop into adults and lay up to 300 eggs in March, thus completing the HWA life cycle (McClure et al. 2001, Cheah et al. 2004).

The adelgid feeds on hemlock sap by inserting its sucking/piercing mouthpart, the stylet, into the parenchyma cells of the plant’s xylem tissue at the base of the needles (Wallace and Hain 2000; Fig. 5c). Most of feeding occurs in the nymphal stages in early spring and late autumn when temperatures are cool and potential predators are less active. Also during this time, hemlocks produce an abundance of sugar and amino acids that are beneficial to the HWA and deplete the hemlock of their nutrient reserves (Cheah et al. 2004). As adelgids feed, the needles on the hemlock begin to discolor, desiccate and eventually drop off reducing the amount of new bud growth on hemlock branches the following season (McClure 1987). The slow deterioration of a hemlock can be observed as a series of steps including; (1) branch die-off from the tip to the bole; (2) branches becoming brittle and breaking off from the ground upward; (3) nearly complete defoliation with only the crown remaining alive; and finally (4) the tree, typically dies within 4-10 years, and becomes a dead snag (McClure 1991, Evans 2003; Fig. 6).

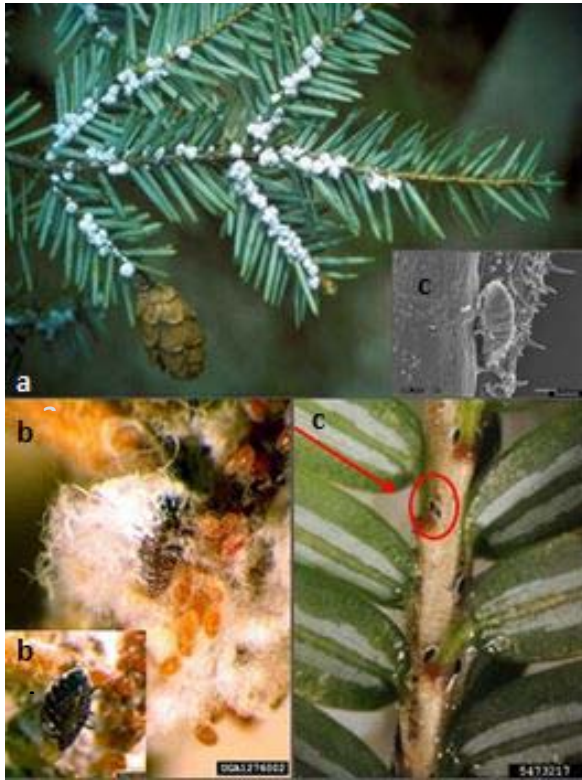


Figure 5. HWA. a) Ovisacs on branch, photo from Connecticut Agricultural Experiment Station Archive, Connecticut Agricultural Experiment Station; b) adult and eggs inside ovisac, photo by Michael Montgomery, USDA Forest Service; c) HWA feeding, photo by Kelly Oten, North Carolina Forest Service (top), Ashley Lamb, Virginia Polytechnic Institute and State University (bottom). All photos obtained from: (<http://www.invasive.org/browse/subthumb.cfm?sub=289>).



Figure 6. Eastern hemlock. a) Unhealthy branch and tree, photo by Chris Evans, Illinois Wildlife Action Plan (top) and Megan Napoli (bottom); b) Healthy branch and tree, photo obtained from <http://savethehemlock.blogspot.com> (top) and <http://www.scituatescience.com/station-1.php> (bottom).

Decline of the Eastern Hemlock

During the egg and active nymph stages, HWA is easily spread by wind, wildlife and people (Cheah et al. 2004). Currently, half of the native range of eastern hemlock is occupied by HWA infestation and it is expanding at a mean rate of 12.5 km per year westward (USFS 2012; Evans and Gregoire 2007). The spread of HWA is limited in the northeast hemlock range by cold temperatures. HWA populations normally stop expanding into areas experiencing mean winter temperatures of -5°C (Paradis et al. 2008). Projected scenarios resulting from climate change predict that much of the northeast will gradually experience milder winters with mean winter temperatures above -5°C (Paradis et al. 2008). Without the protection of this -5°C isotherm, HWA populations are expected to spread throughout the entire northeast and upper midwest (Paradis et al. 2008).

Eastern hemlock forests have been devastated by this invasive insect, causing wide-spread mortality and near-eradication of hemlock in its southern range (Young et al. 2002). As hemlock trees perish and fall, the resulting large canopy gaps allow more sunlight to penetrate the forest floor causing an increase in sapling and shrub growth (Orwig 2002, Small et al. 2002). Deciduous species that are frequently observed growing in areas of eastern hemlock loss include black birch (*Betula lenta*), yellow birch (*Betula alleghaniensis*) red maple (*Acer rubrum*), yellow poplar (*Liriodendron tulipifera*) and rhododendron (*Rhododendron* sp.) (Orwig and Kizlinski 2002, Brantley et al. 2013). Replacement of eastern hemlock with deciduous trees is likely to also affect stream flow as deciduous trees have higher mean annual water use so flow of smaller streams would likely be reduced (Daley et al. 2007). During summer drought, this would potentially result in smaller streams drying up. Water flow would also be reduced due to an increase in downed hemlock trees and debris in streams. With increased light penetration soil temperature increases and soil moisture decreases which alters the carbon/nitrogen ratio in soils and affects decomposition rates (Cobb et al. 2006,

Brantley et al. 2013). As a consequence of these environmental alterations, the unique biocommunity that depends on eastern hemlock forests is likely to be impacted drastically (Benzinger 1994a, Orwig and Foster 1998). Bird communities were impacted by loss of American chestnut (*Castanea dentata*) and American elm (*Ulmus americana*) but at least the former was replaced ecologically by various species of mast-producing oaks (*Quercus* spp.) and hickories (*Carya* spp.). More drastic effects will be seen with the loss of eastern hemlock because there are no comparable ecological surrogates that will thrive in its absence (Benzinger 1994b). The black-throated green warbler, blue-headed vireo, ovenbird (*Seiurus aurocapilla*), hermit thrush (*Catharus guttatus*), and Acadian flycatcher have all shown apparent population declines as a result of hemlock mortality (Tingley et al. 2002). Pervasive effects on stream vegetation structure/composition and nutrient disruptions will likely trigger a trophic cascade of changes, e.g. aquatic invertebrate assemblages are anticipated to be altered which in turn could impact density and productivity of fish, herpetofauna, birds and other wildlife (Snyder et al. 2002). Thus, with the disappearance of hemlock stands, we are likely to witness a chain of dramatic changes within our forest ecosystem which requires documentation and hopefully mitigation in the future. It is imperative to continue to study and monitor HWA effects on our eastern hemlock forests and the wildlife that depend on and reside within this unique ecosystem.

The Acadian Flycatcher

The Acadian flycatcher (ACFL) is a tyrant flycatcher (Family Tyrannidae) and neotropical migrant which breeds throughout the eastern U.S. and winters in Central America and northern South America (Figs. 7-8). Throughout most of its range, this flycatcher typically breeds in wetlands or mature deciduous riparian forests (Whitehead and Taylor 2002). However, it nests almost exclusively in hemlock stands in the northeastern portion of its range that was reoccupied by this species following extensive

deforestation in the early 1900s (Lyons & Livingston 1997, Shriner 2001, Tingley et al. 2002, Ross et al. 2004, Becker et al. 2008). Males arrive on breeding grounds about a week before females in late April/early May to establish territories (Mumford 1964). Acadian flycatchers are mainly monogamous within a breeding season, are known to have double broods (more often reported in the southern portion of breeding range) and will almost always renest after a first failed nest (Mumford 1964). An adept flier, the Acadian will often catch its prey on the wing by aerial hawking but sally glean and hovering are also employed to catch prey items include insects, insect larvae, spiders, and harvestmen (Mumford 1964).

Inter-habitat selection. – Acadian flycatchers are found in a variety of broadly defined forest types throughout their breeding range. In the south, they breed in both hardwood and cypress dominated bottomland forests (Wilson and Cooper 1998b, Whitehead and Taylor 2002) and at higher elevations in riparian hardwood dominated forests types including oak-hickory, oak-pine, and American beech (Smith 1977, Annand and Thompson 1997). In the central portion of their breeding range ACFLs breed in hardwood and mixed riparian forests types; west of the Appalachians beech-maple, oak-hickory, and western mesophytic forests are used while east of the Appalachians (Atlantic coastal plain) hardwood bottomland swamps are utilized (Sallabanks et al. 2000, Whitehead and Taylor 2002).

It is mostly in the northern portion of the breeding range where ACFLs show preference for nesting in hemlock-dominated habitats (Benzinger 1994b, Sheehan 2003, Ross et al. 2004, Allen 2008), and to a lesser extent, other forest types such as oak-hickory kettle holes, tamarack swamps, and beech-maple swamps (Mumford 1964, Bisson et al. 2000, Whitehead and Taylor 2002). Hemlock stands are also an important breeding habitat in the southern Appalachian Mountains including southern Pennsylvania (Allen 2008, Becker et al. 2008), the Blue Ridge Mountains of Virginia



Figure 7. ACFL nesting in eastern hemlock. Photo by author Megan Napoli.

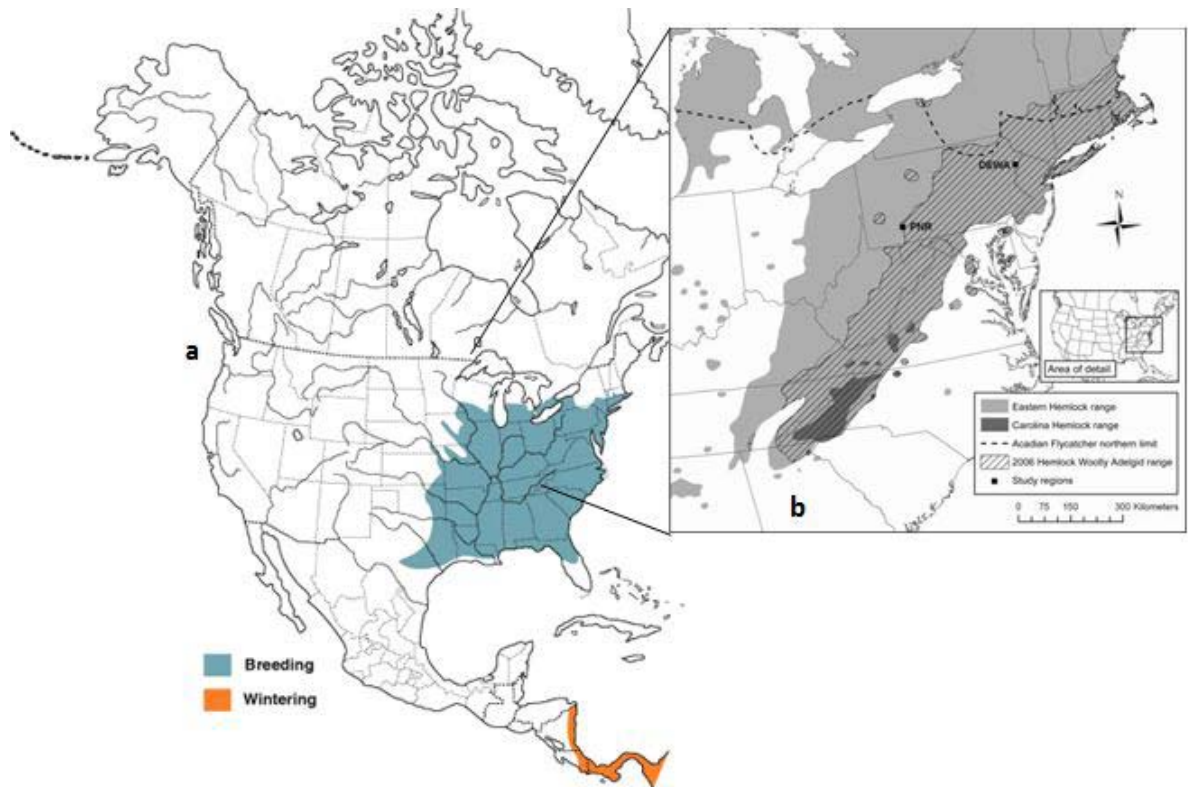


Figure 8. a) Distribution map, obtained from (<http://bna.birds.cornell.edu/>); b) northern limit of ACFL in relation to EAHE range and HWA infestation, obtained from Allen et al. (2009).

(Murray and Stauffer 1995, Lewis 1999; Fauth and Cabe 2005), West Virginia (Shearer 1990), southeastern Kentucky (Brown and Weinkam 2014), and Great Smokey Mountains National Park (Wilcove 1988, Shriner 2001). In Westmoreland County, southwestern Pennsylvania, at Powdermill Nature Reserve, Allen (2008) reported a slightly higher pair density in hemlock stands compared to hardwood forest and in Lebanon County, at Fort Indiantown Gap, Becker et al. (2008) found that ACFLs are present only in forest stands with greater than 10% hemlock. Preference for hemlock vs. deciduous habitat has been demonstrated in the Great Smokey Mountains National Park by Wilcove (1988) who found higher breeding densities of ACFL in hemlock vs. deciduous stands as did Brown and Weinkam (2014) who reported ACFL having a stronger positive association with hemlock vs. deciduous forests in southeastern Kentucky.

Areas in the Northeast that have reported eastern hemlock forest as the preferred habitat for ACFLs include: northern Pennsylvania (Brauning 1992, Haney 1999, Sheehan 2003, Ross et al. 2004), western New York (Eaton 1988), northern New Jersey (Leck 1984, Benzinger 1994b), northeastern Massachusetts and Connecticut (Lyons and Livingston 1997, Tingley et al. 2002), and southern New England (Zelanski and Baptist 1990, Lyons and Livingston 1997, Tingley et al. 2002).

Acadian flycatchers have been found breeding in all counties in Pennsylvania with hemlock habitat preference more frequently reported in northern Pennsylvania (Brauning 1992). In northwestern Pennsylvania, Haney (1999) found ACFL to be very common in old growth hemlock habitat, and in the northeast, in Monroe and Pike Counties, ACFL show preference for nesting in both eastern hemlock dominated ravines and benches compared to riparian hardwood forest stands (Sheehan 2003, Ross et al. 2004). These numerous studies clearly show that hemlock-dominated habitats are of great importance to breeding Acadian flycatchers, especially in the northeastern U.S. and northern Pennsylvania.

Intra-habitat selection. – The typical Acadian flycatcher nest is placed hammock-like in a nest fork towards the end of a tree branch, appears flimsy, and often has streamers composed of plant material hanging from bottom of the nest (Whitehead and Taylor 2002, Allen 2008, Master and Allen 2012). Nests are typically built overhanging or relatively near water or over areas with low vegetative cover, including trails, slopes, and areas with relatively bare ground (Wilson and Cooper 1998a, Whitehead and Taylor 2002, Sheehan 2003, Allen 2008). Nests are therefore conspicuous from below and often laterally, but frequently have higher amounts of cover overhead (Whitehead and Taylor 2002). Nest height ranges from 3-6.5 m, which corresponds with the top of the shrub layer and the bottom of the subcanopy layer (Whitehead and Taylor 2002, Sheehan 2003).

Depending on the geographic area that ACFL are breeding in, a variety of different tree/shrub species are selected for nesting. Acadian flycatchers also tend to choose tree species out of proportion to their availability in the habitat, thus showing a preference for nesting in particular species. In Arkansas, Wilson and Cooper (1998a) reported preferences for nesting in Nuttall oak (*Quercus nuttallii*) and possumhaw (*Ilex decidua*). In Pennsylvania, preference for nesting in hemlock was reported by Sheehan (2003) with 92% of nests placed in eastern hemlock even though relative density of hemlock trees was only 42%. Ross et al. (2004) reported ACFL as an obligate hemlock species. In Massachusetts, Lyons and Livingston (1997) reported 100% of nests placed in eastern hemlock. In New Jersey, Benzinger (1994b) classified ACFL as a primary facultative hemlock facultative species, with the majority of the population nesting in hemlock.

Hardwood tree species that are frequently documented as being used for nesting by ACFL include American beech (*Fagus grandifolia*), witch-hazel (*Hamamelis virginiana*), and flowering dogwood (*Cornus florida*) (Christy 1942, Mumford 1964, Walkinshaw 1966, Bisson et al. 2000). Allen (2008) studied hardwood and hemlock dominated forests with ACFL populations and found that in the hemlock dominated

forests, eastern hemlock was significantly preferred for nesting compared to all other tree species. In the hardwood dominated forests, witch-hazel and American beech were the preferred nesting species.

ACFL and Hemlock Decline. - Given the demonstrated preference for hemlock as a nesting substrate in the northeastern United States in general, and northern Pennsylvania in particular, it has been predicted that breeding areas containing weakened/dead hemlocks would be abandoned by ACFL and densities would decline (Tingley et al. 2002, Sheehan 2003, Ross et al. 2004, Becker et al. 2008, Allen 2009, Brown and Weinkam 2014). Tingley et al. (2002) and Sheehan (2003) both demonstrated that ACFL densities were higher in healthier hemlock stands compared to those heavily infested with HWA. Tingley et al. (2002) found ACFL breeding only in the least infested stands. Sheehan (2003) found ACFL pair densities to be the lowest in the most infested sites. Abandonment of weakened hemlock stands is likely due to changes in vegetation structure and composition including increased light penetrance, less canopy cover, increased deciduous sapling/shrub densities, and a decrease in suitable nesting branches. A breeding range retraction has already been observed in ACFL in the early 1900's in New England and the Northeast at the height of deforestation in the east. Subsequently, recolonization has occurred (Whitehead and Taylor 2002). With the increasing spread of HWA, it is possible that another range retraction of ACFL from the northeast is in the offing. This study focuses on the same area, and in fact the same streams, as Sheehan (2003) in order to follow up on these predictions and determine how ACFL have responded to increasing hemlock woolly adelgid infestation and declining eastern hemlock health in Delaware State Forest (DSF) and the Delaware Water Gap National Recreation Area (DEWA).

ACFL as a bioindicator species. – A bioindicator species is described as any species, or group of species that can be monitored to assess health of an environment

or ecosystem by virtue of measureable changes in their behavior resulting from anthropogenic impacts to the environment (Caro 2010). Avian species are particularly useful as bioindicators because of; (1) a well understood ecology; (2) bird community links to habitat use; (3) easily determined presence/absence/abundance/foraging tactics, and; (4) ability to capture public interest and serve as keystone environmental health messengers (Padoa-Schioppa et al. 2006, Caro 2010). In Pennsylvania, the Louisiana waterthrush has been studied as a bioindicator for headwater stream ecological integrity and for effects of declining eastern hemlock condition in the same habitats used in this study (O'Connell et al. 2003, Mulvihill et al. 2008, Ernst 2012, Barnes 2014).

The Acadian flycatcher, as a near obligate hemlock species in northeastern PA, meets these bioindicator criteria and thus should serve as an effective species to monitor with regard to declining hemlock condition in northeastern Pennsylvania. Logistically, compared to other obligate/near-obligate hemlock species such as blue-headed vireo and Blackburnian warbler, Acadians are relatively more abundant in the selected study areas in DSF and DEWA, and therefore can provide sufficient sample sizes for statistical analyses. ACFL nest lower in the canopy than other obligate/near obligate hemlock bird species making nests easier to locate and monitor. The song/call of the ACFL is frequently performed close to the nest making it easier to locate. Ecologically, because studies have shown that ACFL prefer eastern hemlock for nesting in northeastern Pennsylvania, they should provide measureable responses (changes in pair densities, measures of reproductive success, nest tree preferences) to hemlock decline. ACFL metrics can be compared across time as well as between hemlock ravines and benches to interpret overall ecological integrity and prioritize hemlock habitats for management actions.

Study Justification

This study has two goals including; (1) compare Acadian Flycatcher and hemlock woolly adelgid metrics with those of Sheehan (2003) in order to gauge what changes, if any, have occurred in over the 12 years between studies and; (2) compare hemlock-dominated habitat types, ravines and benches, with regard to HWA effects and ACFL metrics. Accomplishing these goals will provide a historical perspective with regard to ongoing effects of HWA infestation on hemlock stands and a comparison of the relative productivity of each hemlock-dominated habitat. If one or the other habitat type is demonstrated to be more productive as a result of better hemlock condition and ACFL reproductive metrics, that habitat type would logically receive priority should an economically effective treatment be developed for combating HWA infestations on a large scale. Thus, the combination of these goals will help DEWA and DSF managers develop effective and efficient habitat management plans.

Null Hypotheses

- 1) **H₀:** Eastern hemlock health is the same across site types and study sites.
- 2) **H₀:** Eastern hemlock health has not changed since the Sheehan (2003) study.
- 3) **H₀:** Acadian flycatcher reproductive success and pair densities are the same across site types and study sites.
- 4) **H₀:** Acadian flycatcher reproductive success and pair densities have not changed since the Sheehan (2003) study.
- 5) **H₀:** Eastern hemlock health does not affect Acadian flycatcher reproductive success, pair density or nest preference (between studies, site types, and sites).

Chapter II.

METHODS AND MATERIALS

Study Site Descriptions

Research was conducted between May and September 2013 and 2014 at eight study sites. Six sites were located in Monroe and Pike Counties, in northeastern Pennsylvania and two were located in Warren County, northwestern New Jersey (Fig. 9). Sites were selected to provide data complimentary to that previously acquired during a study by Sheehan (2003) using six of the same study sites. Selection of sites in the current study was based on, (1) logistical considerations (travel time and ability to adequately survey study stream reaches); (2) Acadian flycatcher presence/recent absence; (3) forest structure (mature eastern hemlock dominated forests); and (4) stream size (1st-2nd headwater streams). Eastern hemlock was the most dominant and dense tree species at all study sites. Each site description below includes tree species with the highest ecological importance value (EIV), a combination of relative frequency, density, and dominance that characterizes the site. Sites were categorized by stream type, each having different ecomorphologies. Hemlock benches are characterized by slow moving, braided streams meandering across a flat floodplain, while ravines have

one or two steep slopes descending to a stream with water flowing down an inclined gradient (Ross et al. 2004). Three bench sites were studied in this project including, Saw Creek (SAW), Spruce Run (SPRU), and Marshall's Creek (MARS). They were located within/near DSF, an 80,267 acre state forest in Pike (SAW/SPRU) and Monroe Counties (MARS). Ravine sites were located in the DEWA, a 70,000 acre national park extending 40 miles along the Delaware River in both Pennsylvania and New Jersey. Three ravine sites were located on the Pennsylvania side of DEWA including Adam's Creek (ADAM), Hornbeck's Creek (HORN), and Tom's Creek (TOM), in Pike County. On the New Jersey side, two ravine sites were studied, VanCampen's Brook (VAN) and Dunnfield Creek (DUNN). Each study site was composed of stream reaches ranging from 1.45-2.85 km in length, with the width of each site extending 75-100 m into the surrounding forest. Study site characteristics are shown in Table 1 below.

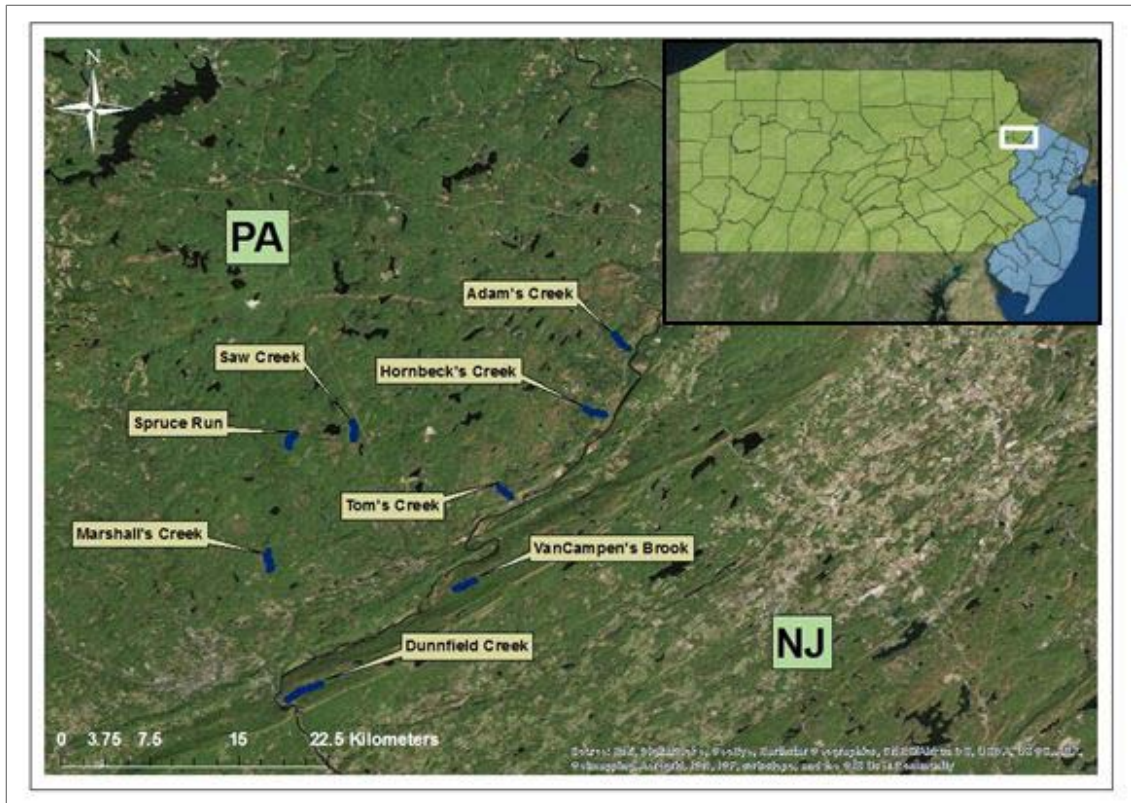


Figure 9. Map of all study sites in northeastern PA and northwestern NJ. PA data obtained from (<http://www.pasda.psu.edu/>) and NJ data obtained from (<https://njgin.state.nj.us/>)

Table 1. Study sites used in the current study and by Sheehan (2003) and their basic characteristics.

Site	Site abbr.	Location	Years used	Site Length (km)	Area studied (ha)	Elev. Range (ft.)	Upstream Endpoint	Downstream Endpoint
Ravine								
Adam's Creek	ADAM	PA, DEWA	2013-2014	1.66	0.56	398-603	41.250514N	41.240233N
Hornbeck's Creek	HORN	PA, DEWA	2001-2002, 2013-2014	1.67	0.84	379-512	41.194444N	41.188858N
Tom's Creek	TOM	PA, DEWA	2001-2002, 2013*-2014	1.46	0.48	375-484	41.126778N	41.134822N
VanCampen's Brook	VAN	NJ, DEWA	2002, 2013-2014	1.68	0.68	360-524	41.058761N	41.057925N
Dunnfield Creek	DUNN	NJ, DEWA	2001-2002, 2013-2014	2.85	1.88	319-819	40.985892N	40.972669N
Bench								
Spruce Run	SPRU	PA, DSF	2001-2002, 2013-2014	1.45	1.72	1025-1101	41.175439N	41.1646N
Saw creek	SAW	PA, DSF	2013-2014	1.62	2	985-1051	41.184153N	41.170758N
Marshall's Creek	MARS	PA	2001-2002, 2014	1.91	0.56	654-738	41.086808N	41.072078N

*used in 2013 only to confirm absence of ACFL.

Spruce Run (SPRU, Figure 10a). – Spruce Run was located in Pike County within DSF. This bench site was included in research conducted by Sheehan (2003) in his 2001/2002 study, and was incorporated in both 2013 and 2014 as well. Eastern hemlock had the highest EIV of 66.52, followed by yellow birch (*Betula alleghaniensis*) with 51.42 and sugar maple (*Acer saccharum*) with 33.78. Basswood (*Tilia americana*) and red maple (*Acer rubrum*) were also commonly observed. The understory layer contained a high density of yellow birch saplings, with American hop hornbeam (*Ostrya virginiana*), basswood and American beech (*Fagus grandifolia*) also relatively common. The shrub layer consisted mainly of witch-hazel (*Hamamelis virginiana*), spice bush (*Lindera benzoin*), and blueberry (*Vaccinium* sp). Eastern hemlock health was essentially good at this site with low infestation and reasonably good overall condition. This site had human activity confined to a rarely used hiking trail paralleling a portion of the study reach with a small foot bridge located on the end of the study reach. The forest surrounding the stream was mature, unfragmented and comprised largely of hardwood trees with the eastern hemlock stands primarily confined to a width of 75 m for most of the stream reach.

Marshall's Creek (MARS, Figure 10b). – Marshall's Creek was located in Monroe County. This bench site was also previously researched in Sheehan's study, however, it was only researched in 2014 during the current study. Eastern hemlock had the highest EIV at 92.05 with white oak (*Quercus alba*) and sugar maple having similar values of 39.89 and 39.05, respectively. The understory layer had a high density of eastern hemlock saplings with American beech and sugar maple saplings also frequently encountered. The eastern hemlock saplings were more dense here than at the other two bench sites combined. At this site, shrubs were infrequently observed, while witch-hazel was the only shrub/small tree found in research plots. Eastern hemlock health was fair at this site. A road with moderate to heavy traffic and dotted with residential housing ran along the length of the study reach. In the middle of the site a house with a large yard effectively separated the study reach into upper and lower segments. There

were a handful of trails along the stream, evidence of some clear-cutting in small patches, and three recently developed man-made ponds. The surrounding forest was largely dominated by hardwood trees, with some eastern hemlock stands extending laterally considerably farther (> 100m) than at SPRU.

Saw Creek (SAW, Figure 10c). – Saw Creek, a bench site, was located in Pike County within DSF. This bench site was not used in Sheehan’s study. Eastern hemlock had the highest EIV at 84.46, with yellow birch and sugar maple having similar values of 30.56 and 30.39, respectively. The understory layer consisted largely of American beech and yellow birch saplings, with witch-hazel, spicebush and serviceberry (*Amelanchier canadensis*) commonly found in the shrub layer. Eastern hemlock health at this site was relatively good. A narrow gravel road bridge crosses over the middle of the study reach near three small hunting cabins. The downstream portion of the study reach was frequented by trout fisherman in the spring. Otherwise, there was little disturbance to the overall forest at this site. This site was surrounded mainly by hardwood forest, although some eastern hemlock stands extend laterally into wetlands adjacent to the stream.

Adam’s Creek (ADAM, Figure 10d). – Adam’s Creek was the most northerly site, located in Pike County on the Pennsylvania side of DEWA. This ravine site was not used in Sheehan’s study, but was in both 2013 and 2014. This site was included in this study because it was previously treated by the National Park Service (NPS) in April 2000 for HWA infestation by releasing Pt beetles (*Pseudoscymnus tsugae*). Eastern hemlock had the highest EIV at 109.02, with yellow birch and sugar maple having values of 44.46 and 31.21, respectively. The understory layer consisted of yellow birch, American beech, and sugar maple saplings, with no eastern hemlock saplings occurring along the study reach. The invasive Japanese barberry (*Berberis thunbergii*) was the most frequently observed shrub while serviceberry was also common. Eastern hemlock health at this site was poor in most areas. This site hosted human activity almost daily in summer due to the popular waterfall and swimming hole at the upstream end of the reach. A trail paralleled

the entire length of this site, but the overall forest structure was intact. The surrounding forest was a mix of eastern hemlock and hardwood trees.

Hornbeck's Creek (HORN, Figure 10e). – Hornbeck's Creek was located south of Adam's Creek, in Pike County on the Pennsylvania side of DEWA. This ravine site was previously used in Sheehan's study and was investigated in 2013 and 2014. Eastern hemlock had the highest EIV at 90.16, followed by yellow birch at 42.49 and black birch (*Betula lenta*) at 20.87. American beech and eastern hemlock saplings were the most common sapling species observed. Serviceberry was the most common understory tree while spicebush and Japanese barberry were the most abundant shrubs. Eastern hemlock health at this site was poor to fair. This site had a maintained trail following the stream for the entire length of the study reach. Hikers and swimmers were frequently seen walking to the waterfall and swimming hole at the upstream end of the study reach. The surrounding forest was intact and consisted of a mix of eastern hemlock and hardwood trees. Hemlock stands extended at least 100 m upslope along several tributaries entering Hornbeck's Creek.

Tom's Creek (TOM, Figure 10f). Tom's Creek was located south of Hornbeck's Creek, also in Pike County, on the Pennsylvania side of DEWA. This ravine site was previously used in Sheehan's study, and was utilized primarily in 2014. In 2013, this site was only monitored briefly to confirm a total lack of Acadian flycatcher activity along the study reach. Eastern hemlock had the highest EIV at 97.71, followed by American beech at 29.17 and yellow birch at 28.3. American beech, eastern hemlock and yellow birch saplings were common. Witch-hazel was commonly seen in the understory layer with Japanese barberry, spicebush and rhododendron (*Rhododendron sp.*) most common in the shrub layer. Eastern hemlock health at this site was generally poor. A picnic area at the downstream portion of the study reach was used regularly by visitors using the trail that ran along the length of the study reach. In 2014, there was extensive reconstruction of a small dam adjacent to the picnic area at the downstream terminus of the study reach. The surrounding forest was intact consisting of a mix of eastern hemlock and

hardwood trees. Hemlock stands extended up to 100 m from the stream along tributaries leading into Tom's Creek.

VanCampen's Brook (VAN, Figure 10g). – VanCampen's Brook was located in Warren County on the New Jersey side of DEWA. This ravine site was previously used in Sheehan's study during 2002 and in the current study in 2013 and 2014. Eastern hemlock had the highest EIV followed by black birch, yellow birch, and sugar maple (38.48, 20.36, and 19.63, respectively). The most commonly observed saplings were eastern hemlock followed by striped maple (*Acer pensylvanicum*) and American beech. The shrub layer consisted primarily of Japanese barberry with some invasive winged burning bush (*Euonymus alatus*) and rhododendron. Serviceberry was a common understory tree. The health of hemlock trees at this site was mostly poor with a few trees in fair condition. This site, like several others, had popular swimming holes and a maintained trail paralleling the entire study reach. A moderately busy road also ran along the top of the eastern slope of this site, but was at least 100 m distant for most of the study reach.

Dunnfield Creek (DUNN, Figure 10h). – Dunnfield Creek was the most southerly site used in the current study and it is located in Warren County on the New Jersey side of DEWA. This ravine site was previously used in Sheehan's study and used during 2013 and 2014 in the current study year. Eastern hemlock had the highest EIV at 56.46, with yellow birch at 53, and tulip poplar (*Liriodendron tulipifera*) having the third highest value at 39.23. The eastern hemlocks at this site were in poor condition and interspersed with hemlock stands consisting entirely of snags. American beech, sugar maple, striped maple saplings, and witch-hazel were commonly seen throughout the understory. Japanese barberry was very dense throughout this study reach. This site is a very popular hiking area with a well-used recreational trail that once again, extended the length of the study reach, crossing the stream in several areas. The hemlock stands were more sparse and narrow (extending only 50 m inland from either side of the stream) than on any other stream study reach.

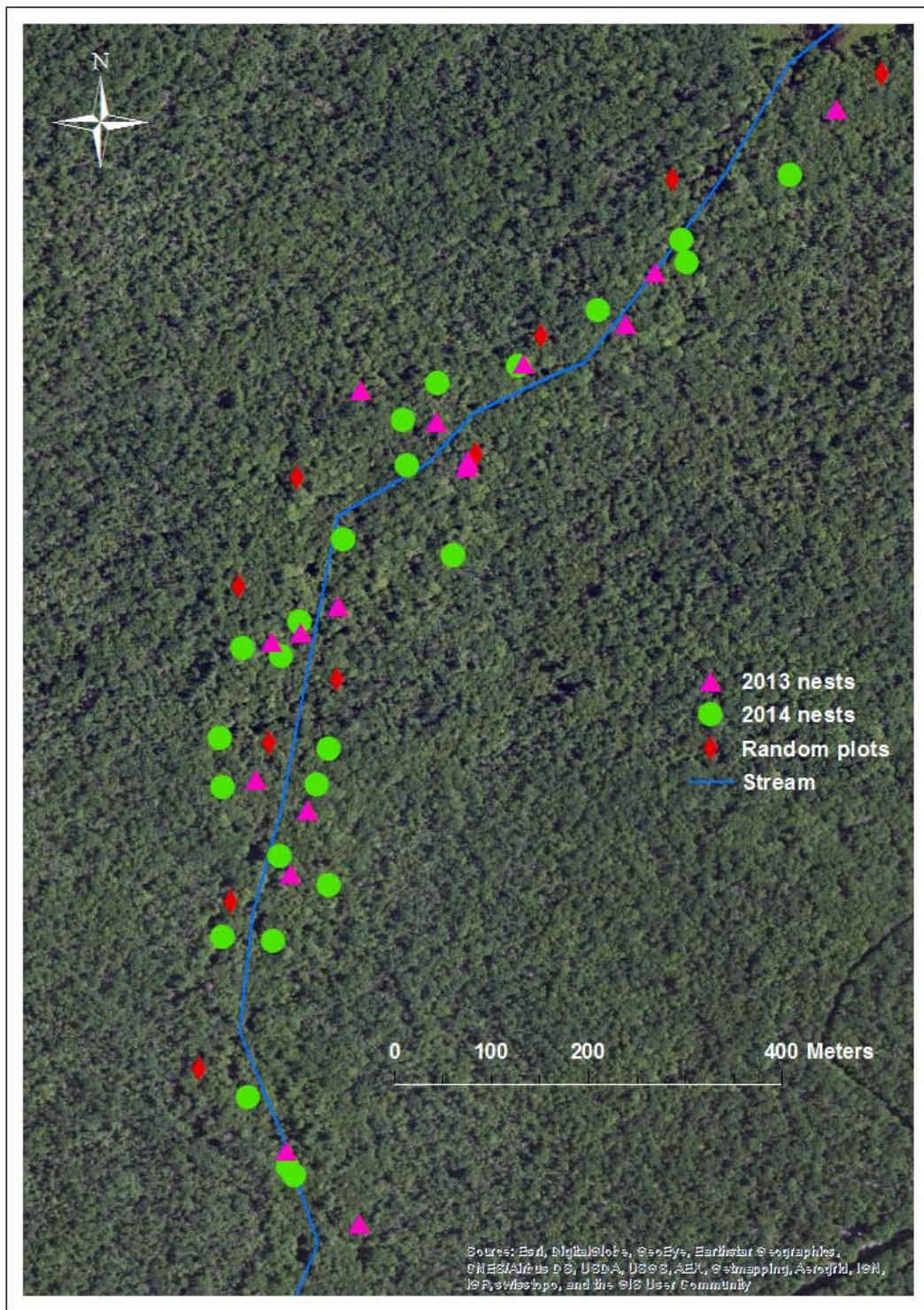


Figure 10a. Spruce Run

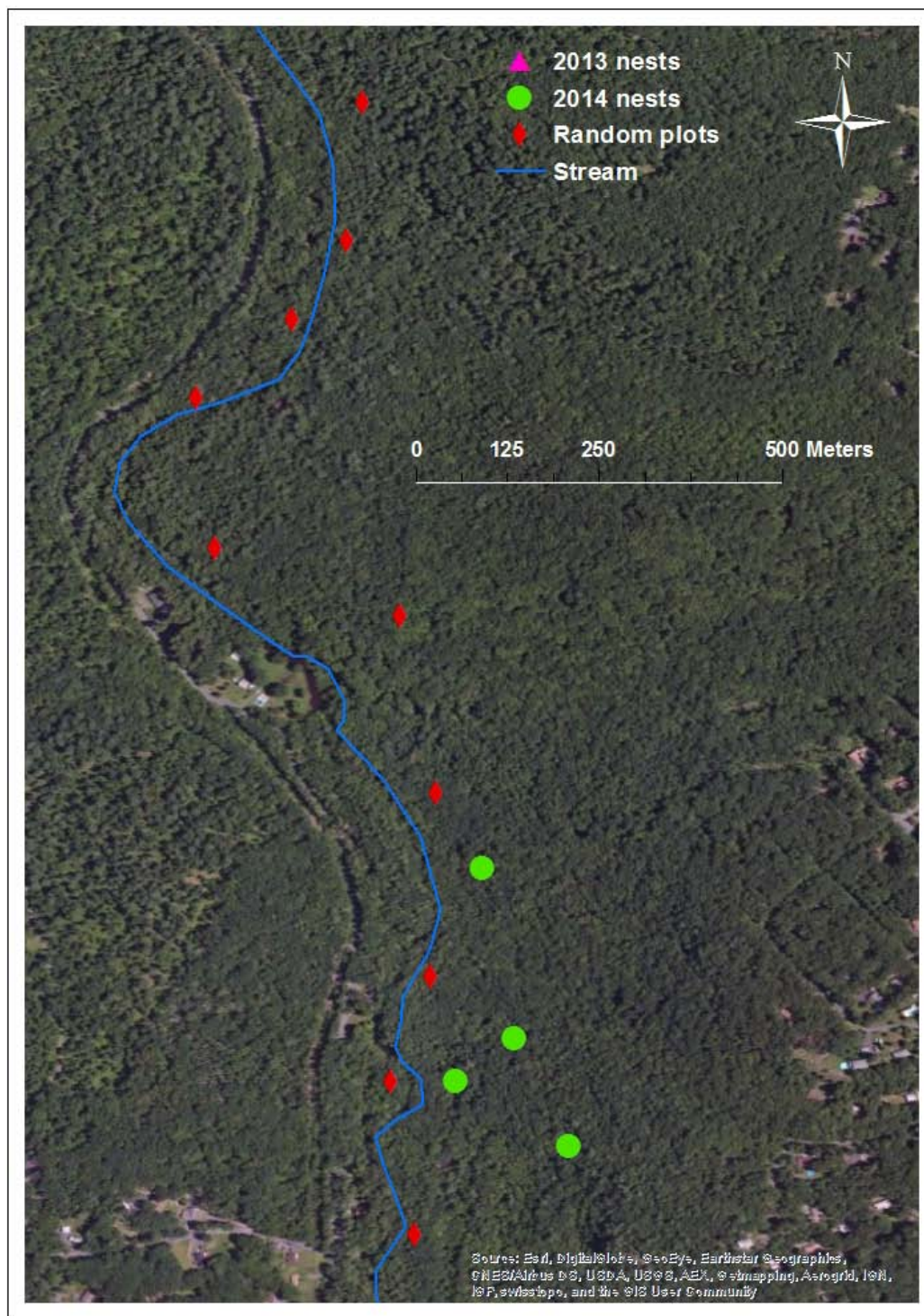


Figure 10b. Marshall's Creek.

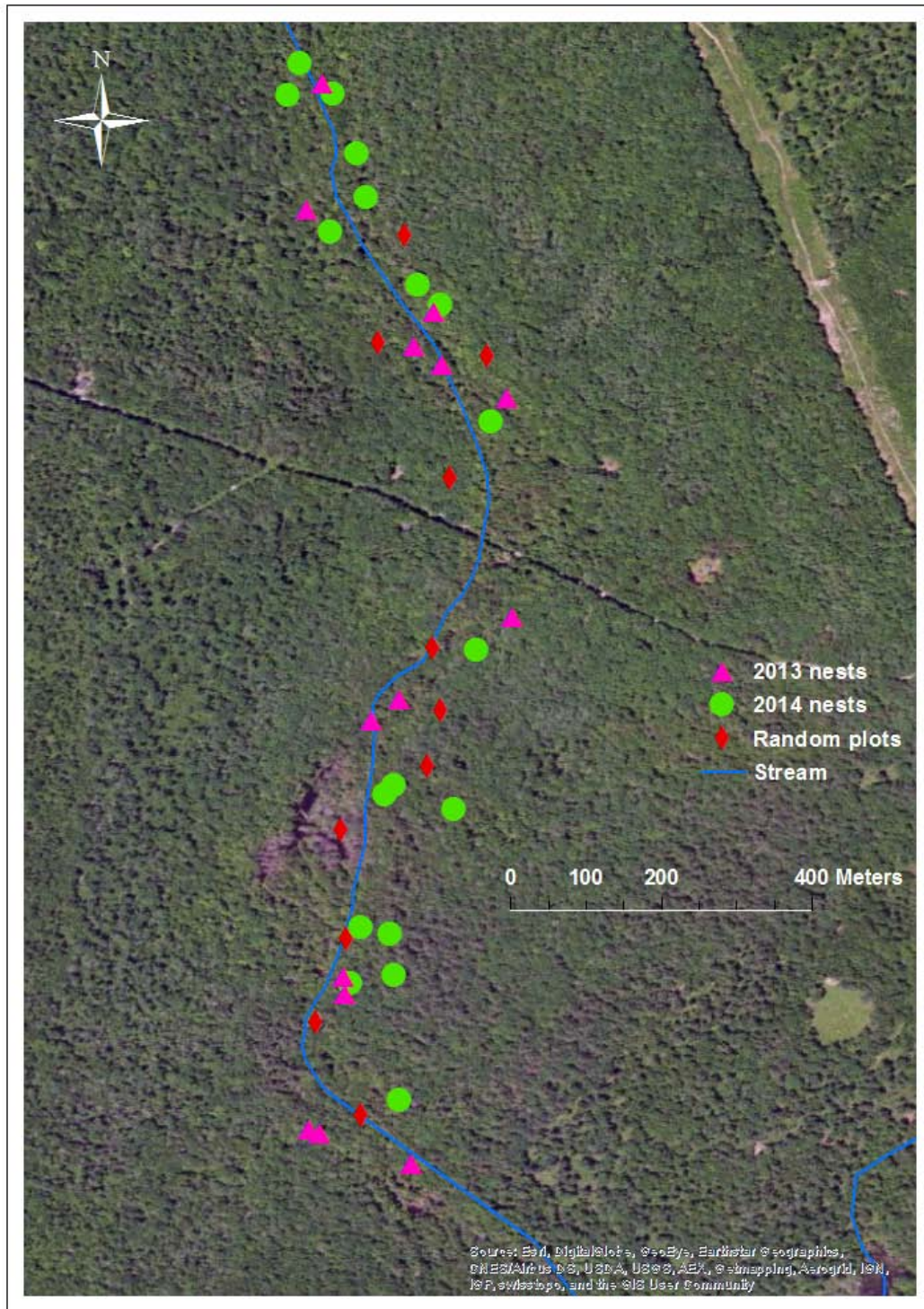


Figure 10c. Saw Creek



Figure 10d. Adam's Creek.

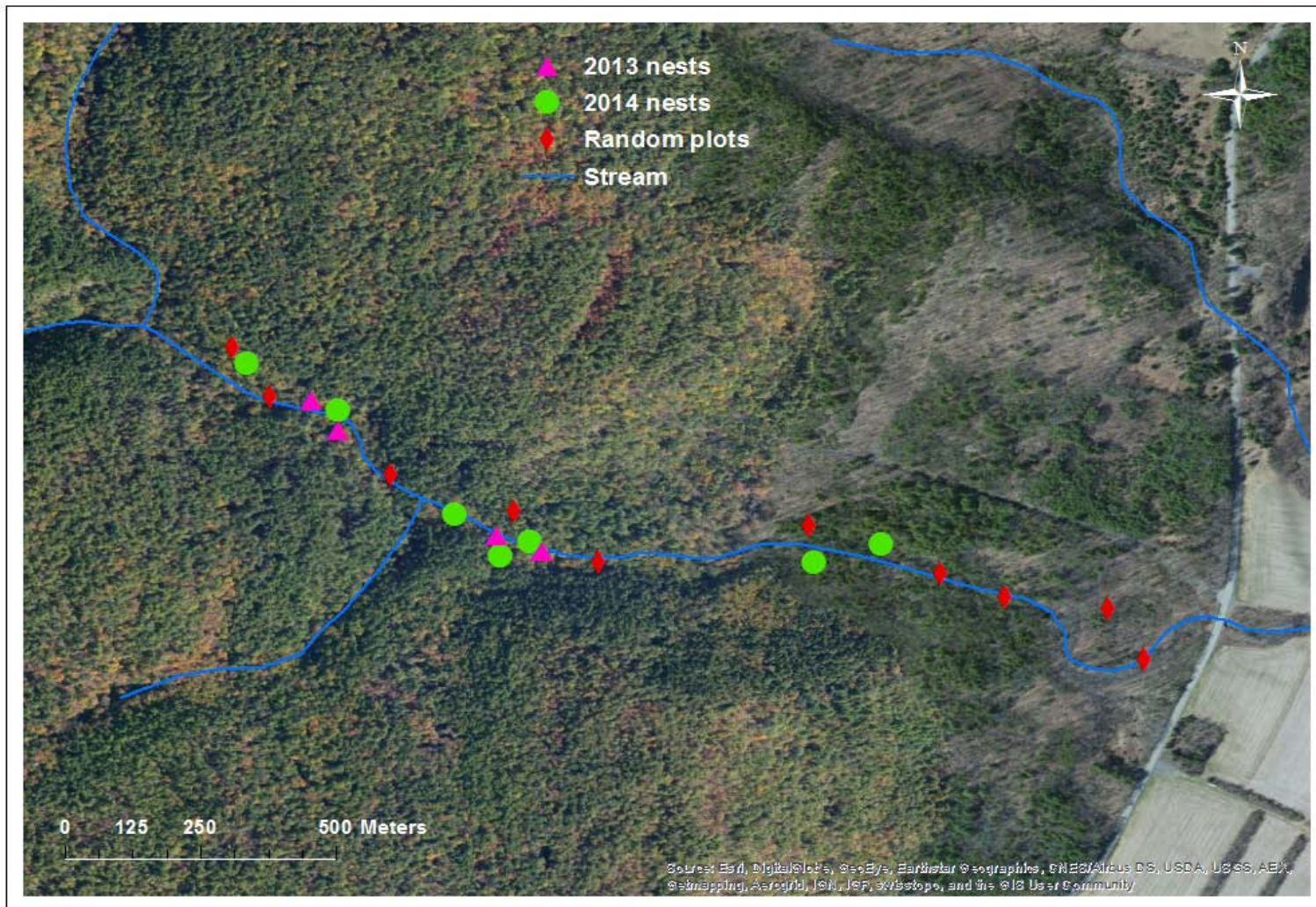


Figure 10e. Hornbeck's Creek

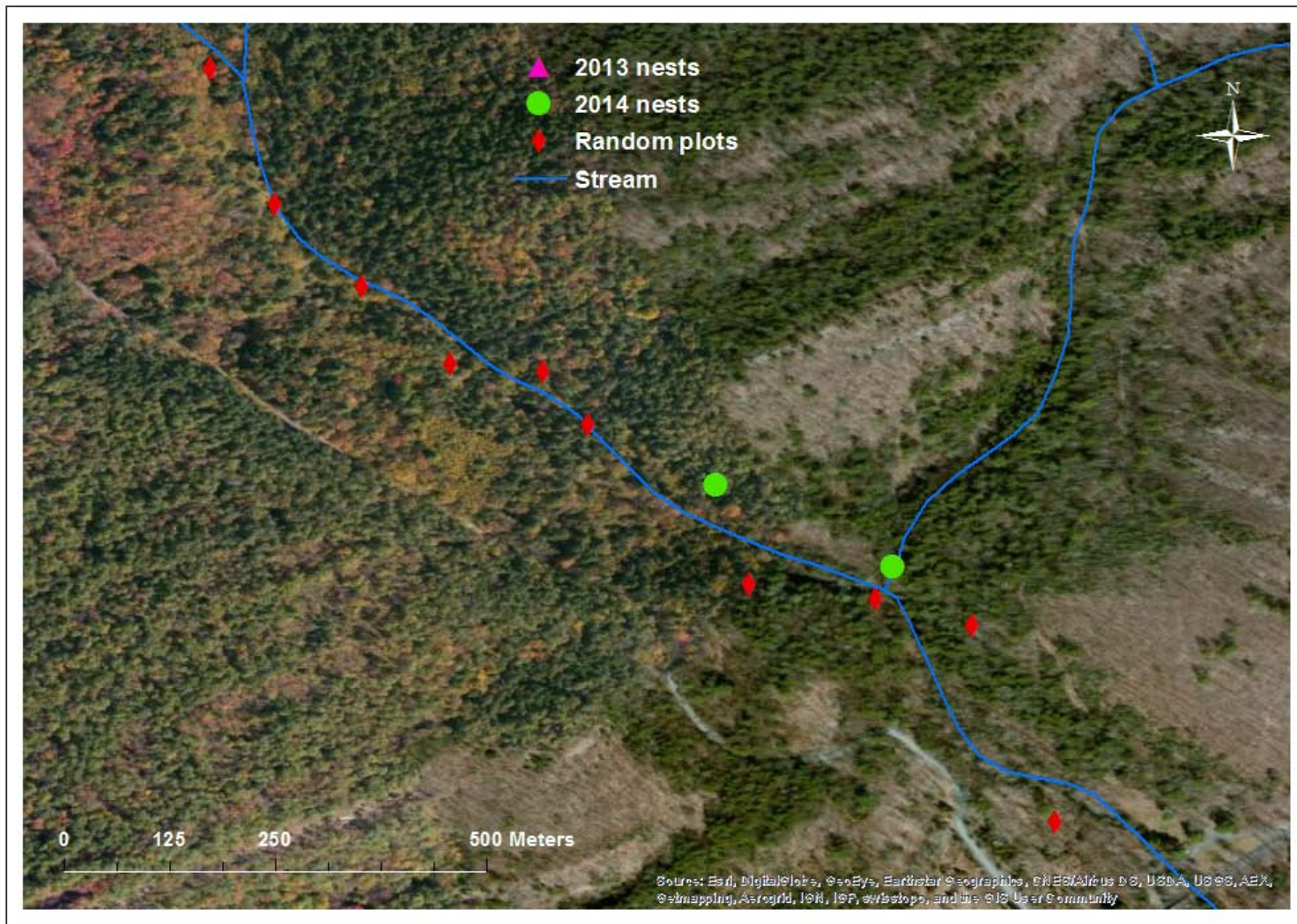


Figure 10f. Tom's Creek

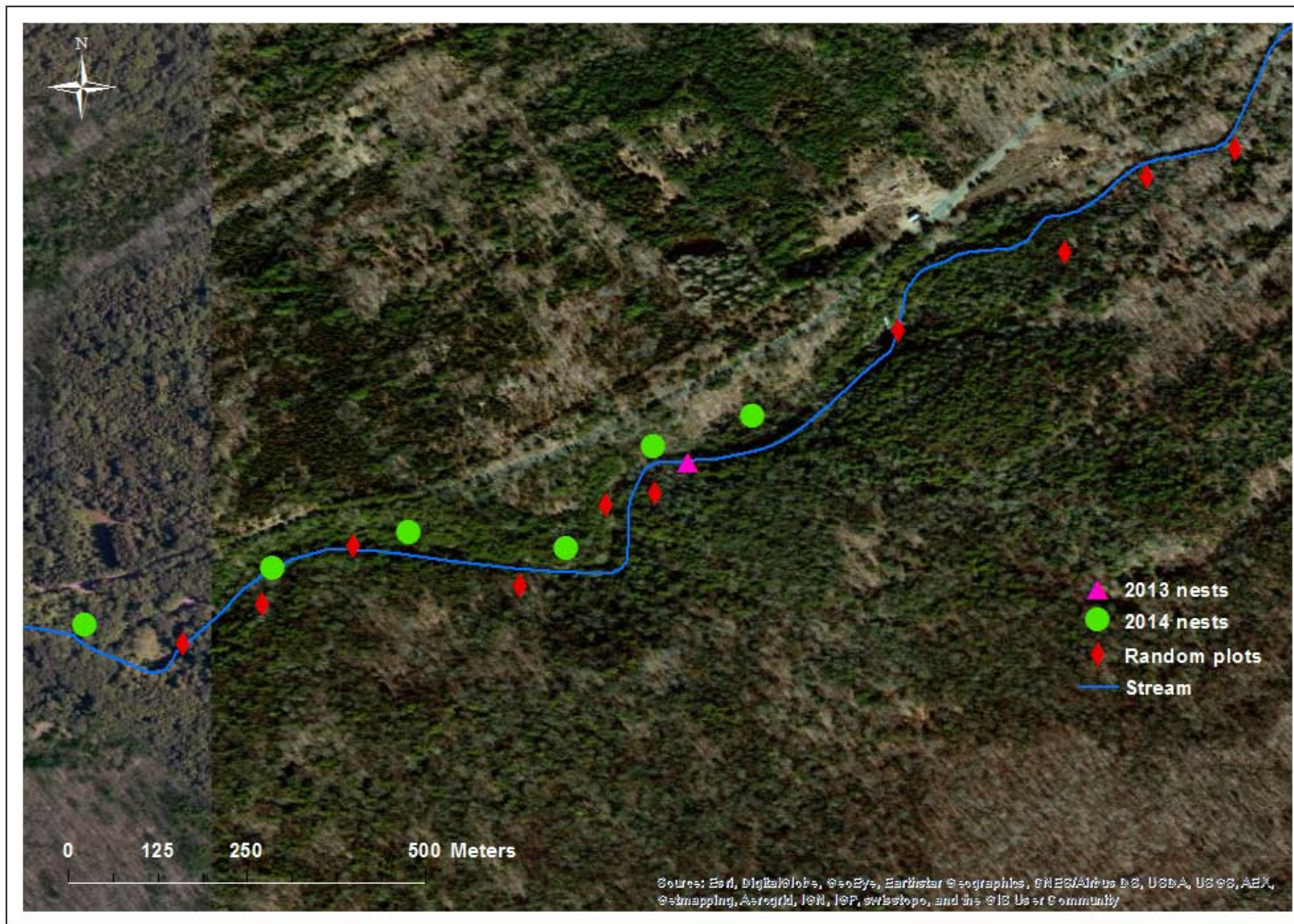


Figure 10g. VanCampen's Brook.

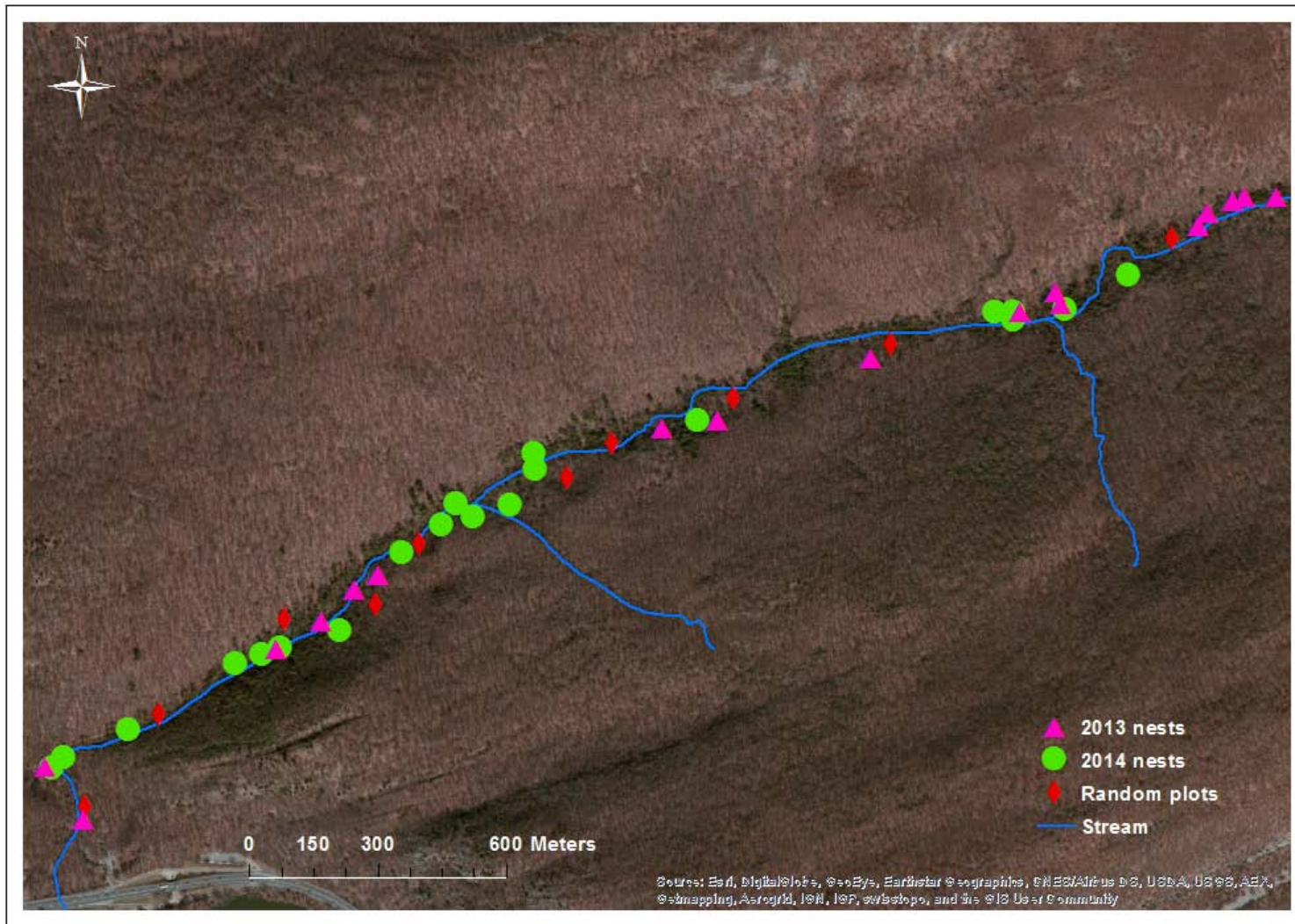


Figure 10h. Dunnfield Creek.

Eastern Hemlock Conditions

Eastern Hemlock Health and HWA Survey. – Following a protocol developed by the Delaware Water Gap National Recreation Area (Evans 1996) and previous methods used by Sheehan (2003), an HWA survey was performed to evaluate eastern hemlock condition and HWA infestation level at all study sites. All surveys were performed in June 2014 when both the adelgid and new hemlock growth are visible. An additional site, Tumbling Waters, located in the DEWA between TOM and HORN, was included in the survey in early July because no ACFL were observed there in 2013-14 and we wanted to see if this lack of ACFL activity would correlate with infestation levels along a continuum with the other study sites.

The survey began at a randomly chosen distance (using a random number generator) between 0-50 m at the downstream end of the study reach at each site. After location of this first plot, each subsequent plot was spaced 50 m apart down the length of the study reach. The 50 m distance between each plot was measured with the Garmin handheld GPS unit. Plots were randomly selected to occur on the left or right stream banks, or in some cases if hemlock did not occur on one bank, the alternate bank was chosen. If hemlock did not occur on either side of the bank, then the plot was marked as having no hemlocks available. The center point of each plot was arrived at after 50 m was traversed. The four closest hemlock trees to the plot center point were chosen to be surveyed. Plots were continued until the end of the site was reached.

After selecting the four hemlock trees in each plot, each tree's dbh was estimated in 5 cm intervals and the tree height was also estimated. The tree's lowest dead and live branch heights were measured to the nearest half meter. Lowest dead branch condition was rated as; (1) stubs, branch was broken off with no twigs remaining; (2) coarse-twigs, twigs that remained on these branches were relatively thick and; (3) fine-twigs, twigs that remained on these branches were fine and only recently defoliated. Lowest live branch condition was rated as; (1) poor, these branches were

brittle with many yellowing /missing needles; (2) fair, branches having some flexibility with a moderate amount of yellowing/missing needles and; (3) good, branches were pliable with many healthy needles and very few yellowing/missing needles. A 30 cm long tip of the lowest live branch from each tree was then examined for HWA infestation and terminal analysis. The number of non-new (dark green), new (light green), bare and HWA infested terminals were counted on this 30 cm branch tip. Each of the four trees was then given an overall health condition rating from 1 to 5; (1) extremely poor, only the very top of tree crown still alive, nearly all branches dead/gone; (2) poor, some live branches extending down the bole from crown, but most branches dead/gone; (3) fair, live branches extending halfway down the bole, but other half of branches dead/gone; (4) good, nearly all branches still alive with few branches dead/gone, sparse/no HWA present and; (5) pristine, all branches healthy with no HWA present.

Calculation and Statistical Methods. – All statistical analyses in this thesis were performed using IBM's Statistical Package for the Social Sciences (SPSS 22.0). All analyses were performed using a confidence interval of 95% ($p = 0.05$). For all Chi-square tests, when significant differences were found, a non-parametric post-hoc test was done to see where significant differences occurred. To do this, in a contingency table each individual cell's adjusted standardized residuals (z scores) were compared to an adjusted p value that reflected the number of cells that were included in the contingency table (Beasley 1995, García-pérez and Núñez-antón 2003). Data was tested for normality with the Shapiro-Wilk test; all data analyzed in this thesis was found to be not normally distributed ($p < 0.05$). For Kruskal-Wallis analyses of variance, when an overall significant difference was found, a pair-wise post-hoc test was used to see where significant differences occurred.

Data collected from the HWA survey was compared within the current study and also to data collected by Sheehan (2003). Overall tree rating and lowest live branch rating were compared using Chi-square tests between studies, site types, and sites.

Terminal data and height of lowest live branches were analyzed using Wilcoxon matched pair signed-rank tests between studies, Mann-Whitney U tests between site types, and Kruskal-Wallis variance tests between sites.

Acadian Flycatcher Demography

Pair Density. – Starting in early May, each stream reach was visited at 3-4 day intervals, with the field season ending in mid-September for both study years. Each site was investigated by slowly walking down one side of the stream reach, meandering up to 50 m away from the stream depending upon sightings and song cues from ACFL, and then repeating this protocol for the opposite side of the stream reach. Key ACFL cues that were searched for included; (1) ACFL vocalizations including adult songs and several types of calls and; (2) individual movement, pair interactions, and indications of nest building. When singing males were encountered, detailed notes were taken on his location. Observations would continue for roughly 15 minutes to see if there were any indications of a female nearby. Nest locations were commonly found by following an ACFL calling (typically a female) during construction, incubation, or rearing periods. All nest locations were recorded using a Garmin® 60CSX handheld GPS unit ([Appendix...](#)), and were occasionally marked with flagging attached to trees approximately 20 m from nest location in areas of dense vegetation. Each stream reach was searched approximately 25 times each study year initially to locate pairs/nests and subsequently to monitor the fate of nests, eggs, and fledglings; exceptions included DUNN in 2013 (started in late June), TOM in 2013 (no ACFL activity), and MARS (not included until 2014). ACFL are extremely vocal, in particular females which emit a characteristic call when near the nest, thus, there is a high confidence that nearly every nest within each study reach was eventually located.

Total number of successful and failed nests was recorded for each site. If a nest fails, Acadian flycatchers often renest in a continued attempt to produce offspring. More rarely, with a first successful nest, ACFL can attempt to double brood, thus roughly doubling their reproductive success for the breeding year (Mumford 1964, Whitehead and Taylor 2002). When determining the number of breeding ACFL pairs on a study reach, the number of renests was added to the number of double brooded nests and then that total was subtracted from the total number of nests. Because ACFL were not banded there was never 100% certainty that any two nests were from the same pair. Mumford (1964) stated that there is usually a lag time between nests of 4-10 days with spacing between nests of 18-94 meters. Allen (2008) found lag times up to 20 days (allowing fledglings to become independent) and distances between nests of 90 meters. Using these criteria it was reasonably easy to infer the number of renesting or double brooding nesting attempts, if any, for each pair.

Nest Survival. – When an active nest was located, it was visited every 3-4 days to monitor its status. Nests that were low enough (accessible) were checked using a mirror-pole, which was essentially a small handheld mirror attached to the end of a telescoping painter's pole. The pole extended to about 8 m, and by holding the mirror above the nest the contents of the nest could be observed with minimal disturbance. This method was only used when no apparent predators were visible or heard especially blue jays (*Cyanocitta cristata*) and eastern gray squirrels (*Sciurus carolinensis*). As nestlings reached the age where their eyes were open and they were capable of moving around, use of the mirror-pole was stopped to eliminate the risk of causing nestlings to fledge early. In rare cases, nests were low enough to be checked without the mirror-pole. If a nest was too high (inaccessible) to be checked with the mirror-pole, nest contents could often be seen through the bottom of the flimsily built nest. Nestlings could also be seen at high nests when taking food from adults or protruding from the nest cup. A nest was considered successful when empty on or after the expected fledging date (assuming that nestlings were seen regularly prior to that date), fecal sacs

were seen in the area around and below the nest, and in some cases adults and fledglings were seen within the nest territory. If a nest was empty before the expected fledging date it was classified as failed. Inaccessible nests were visited several more times to conform that they failed. In some cases, nests were partially torn apart, missing completely, and egg/nestling remains were found nearby. The most likely cause of nest failure was predation, but a few nests also appeared to fail because of high winds and/or heavy rain. A nest was classified as failed by abandonment if all eggs failed to hatch and no ACFL activity was observed at the nest.

Two methods were used to calculate nest survival. Apparent nest success was calculated by taking the number of successful nests at a site and dividing it by the total number of nests at that site. This method produces a positive bias because it does not account for potential nests that were short-lived (lost quickly and not found). To reduce the potential positive bias that occurs when using apparent nest success, the Mayfield (1975) method for calculating nest survival was also performed. This method used exposure days, i.e. the number of days the nest was known to be exposed to the elements such as predation and harsh weather (Mayfield, 1961, 1975). Subtracting the number of failed nests at a site by the total number of exposure days (days for both incubation and nestling periods) for all nests and then dividing by the total number of exposure days resulted in daily survival probability. This was the probability that a nest would survive from one day to the next. By taking the daily survival probability and raising it to the 30th power, the nest survival probability was determined. This number reflects the probability of a nest surviving the typical ACFL nesting cycle of 30 days.

Productivity. – Measures that quantified productivity at each site included pair density (pairs/km), nest density (nests/km), fledgling density (fledglings/km), and pair productivity (fledglings/pair, nests/pair). To determine how many fledglings a site produced, the number of fledglings from successful nests was added together. When a successful nest was inaccessible and the number of nestlings fledged was difficult to

determine (n= 15), the average brood size of 2.8 was assumed (Whitehead and Taylor 2002, Allen 2008).

Nest Placement. – Multiple nest-specific measurements were recorded for each nest. The nest tree species was identified and if it was an eastern hemlock it was given a tree rating from 1-5; (see **Eastern Hemlock Conditions**) and checked for HWA infestation (yes/no). Nest tree diameter at breast height (dbh) was measured to the nearest tenth (0.1) of a cm using a dbh tape from Forestry Suppliers. The height of nests that were ≤ 8 m was measured to the nearest tenth of a meter using a telescoping rod from Forestry Suppliers. For nests that were > 8 m a clinometer was used to measure height. Nest tree height was similarly measured. Using a 100 m measuring tape, also from Forestry Suppliers, the distance from the nest to the tip of the nest branch and the nest to the bole of the nest tree was measured. Standing at the bole of the tree, using a compass, the bearing from the bole to the nest was recorded. The terrain below the nest was classified into five categories including stream, tributary, trail, clearing, and slope ($>10^\circ$). Nest placement in the branch (nest-fork) was classified into five categories based on how many main twigs the nest was attached to (Allen 2008; Fig. 11). ACFL will often incorporate nest “tails” on the bottom of their nests made from plant fibers such as grass, catkins, fine twigs, rootlets, etc. (Harrison 1975, Allen 2008). The number and length of such tails that were > 5 cm were recorded for each nest.

Calculations and Statistical Methods. – Demographic and nesting data that was collected in this study was compared between each site type (ravines vs. benches) with Mann-Whitney U tests and between each study site with Kruskal-Wallis tests; any categorical data was compared using Chi-square tests (Fisher’s exact test was used in cases with low sample size). A goodness-of-fit Chi-square test was used to compare nest tree species preference in relation to habitat availability. Demographic and nesting data was also compared to Sheehan (2003) to see if there were any statistical differences in the twelve years between the studies (except fledgling data which was not reported in Sheehan 2003). Differences between the two studies were compared within each site

type (ravines vs. benches) and study site using Wilcoxon matched pair signed-rank tests or Chi-square tests for categorical data (Fisher's exact test was used in cases with low sample size).

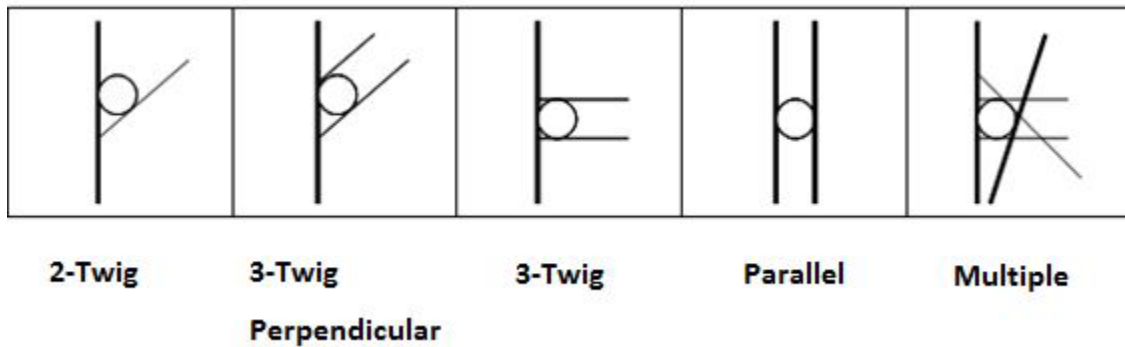


Figure 11. Illustration of nest-fork categories from a bird's eye view. Twigs and nest are represented by lines and circles, respectively (Allen 2008).

Habitat Selection and Nest Measures

Nest and Random Vegetation Plots. – Vegetation analysis for nests was done by establishing a 22.6 m circular plot with the nest at the center of the circle. This was based on BBIRD protocol (Martin et al. 1997), with a few additional measurements drawn from research done by Allen (2008). Thus, the area of each plot was 0.04 ha. Extending outward from the center point (directly under nest), four 11.3 m transect lines were laid out along each cardinal direction. Each line was marked with colored tape every 5 m from the center. This distance marked 5 m radius (10 m diameter) nested plot within the larger circle. (Fig. 12). Within the nested plot more detailed measurements for trees, saplings, shrubs and ground cover were recorded. For trees, the species, dbh, distance to nest (m), and bearing to nest were recorded. Trees were defined as having a dbh of 10 cm or greater. Any tree species with a dbh of < 10cm was deemed a sapling. For saplings and shrubs, the species, number of stems, percent cover and average height were recorded. The nested circular plot also created quadrants or “wedges”. For each quadrant, five categories of ground cover were estimated including percent cover of shrubs, herbaceous plants, water, soil/rock, and leaf. These categories could then be consolidated into categories of water, bare ground and covered ground categories. In the larger circular plot only tree species and dbh were recorded. The canopy cover for each tree species found in a plot was also estimated. At the center of each plot a densiometer was used to estimate the canopy cover over the nest. From the center of each plot, a 100 m measuring tape was used to determine the distance to the closest water source (main stream or occasionally a tributary). Each plot was categorized into one of three terrain zones based on topography; (1) stream; (2) floodplain and; (3) slope (Fig 13). Additional data recorded for each plot included clinometer measurements for slope, aspect and canopy height (m).

At each site 10 random vegetation plots were also established (SAW had 11 due to accidental miscount) to determine if there were any statistical differences in vegetative structure/composition between areas that ACFL chose or did not choose to

nest. To determine the location of random vegetation plots, random numbers were assigned to segments of stream determined by the length of the study reach. For example, SPRU was 1,450 meters long so segments of stream 30-145 m in length were assigned randomly thus ensuring there would be 10 random plots along the study reach ($10 \times 145 = 1,450$ m). The 30 m lower limit of segment length ensured that every random plot was at least 30 m away from an adjacent random plot linearly along the stream. Next, random compass bearings were chosen to determine in what direction the random plot would be situated relative to the stream bank. Finally, random numbers were also assigned to distances from 0-75 m to locate the plot within the maximum distance that ACFL nested away from a particular stream. After taking into account direction, if a random plot happened to fall within 30 m of a nest plot, a new set of random numbers would be used to establish a new random plot. The vegetation measurements done at random plots were the same as for nest plots except for nest-specific variables.

Foliage Height Diversity. – An additional variable measured in vegetation plots was vertical foliage distribution (MacArthur et al. 1966). For the 11.3 m transect line laid down in each cardinal direction, colored tape was used to mark 2.26 m distances along the line resulting in 5 equidistant markings along each line (See Fig. 12). Foliage height diversity (FHD) was measured at each of these marked distances along each line for a total of 21 measurements (including the center point where the lines met). At each station, the presence/absence of live vegetation was noted within six height intervals including; 0.5-3 m, 3-6 m, 6-12 m, 12-18 m, 18-24 m, and > 24 m (MacArthur and MacArthur 1961). The end result was a measure of foliage distribution above the ground.

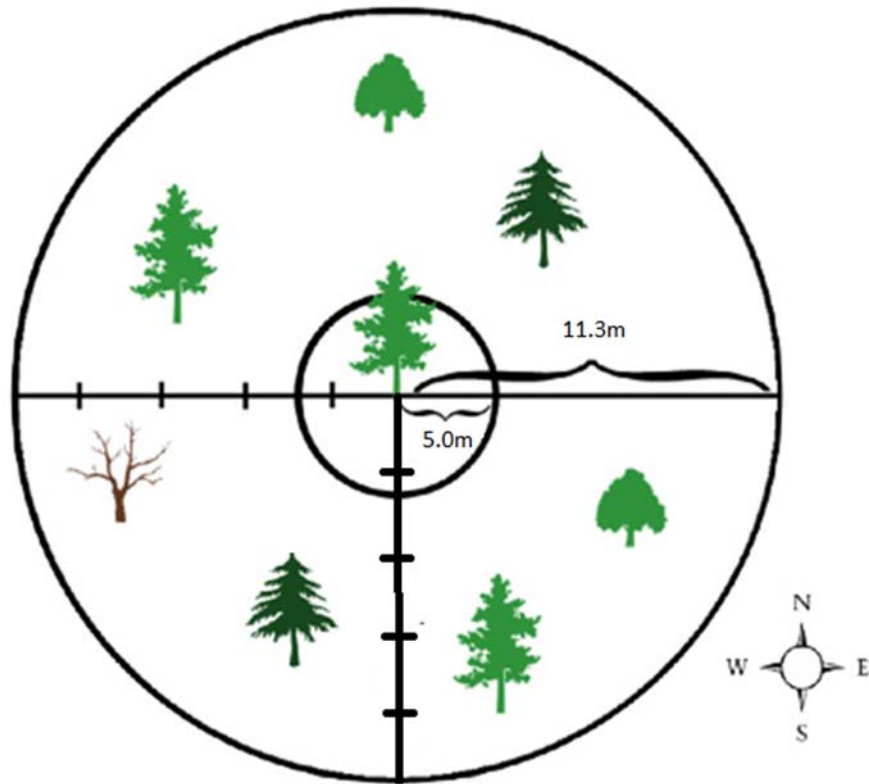


Figure 12. Illustration of a vegetation plot where the outer circle represents the area where the trees were analyzed, the inner circle represents the area where the shrubs and saplings were analyzed, and the hash marks represent the intervals where foliage distribution was observed (East Stroudsburg University's Avian Ecology and Behavior Laboratory).

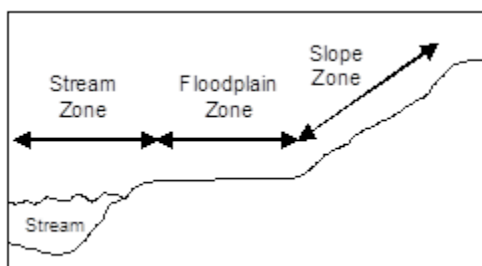


Fig 13. Illustration of terrain zones. The stream zone extends 10 m from each bank (Allen 2008).

Calculations and Statistical Methods. – Vegetative structure and composition, foliage height diversity, surface water and terrain data was only analyzed for this study and not compared to Sheehan (2003). For each tree species found in vegetation plots, relative density, relative dominance, relative frequency and ecological importance value were calculated. These values assist with comprehending the total tree composition in detail for each site. Mann-Whitney U tests were performed to determine significant differences in vegetation structure between plot types (nest vs. random), sites, and site types. Vegetation composition (tree, sapling, shrub species) was compared only magnitudinally between site types, and between nest and random plots for each site type and study site. Nest tree species preference/avoidance was compared between nest and random plots within each site type and each study site. The foliage height diversity proportion of “hits” across intervals was compared with Mann-Whitney U tests between site type, plot type, and each study site. Chi-square tests were used when determining if there were significant differences in terrain zones between site types, plot types, and each study site. Plot center distances to water were compared with Mann-Whitney U tests between site types, plot types, and each study site.

Chapter III

RESULTS

Eastern Hemlock Conditions

The HWA survey completed in 2014 was performed on nine stream reaches in DSF (SPRU, MARS, SAW) and DEWA (ADAM, HORN, TUWA, TOM, VAN, DUNN). Comparisons were evaluated between site type (ravines vs. benches) and among stream sites. The variables determined to be most indicative of defining overall hemlock health were; (1) hemlock health rating; (2) mean height of the 1st live branch; (3) health rating of the 1st live branch; and (4) mean number of new, bare and HWA infested terminals summed.

The HWA survey completed in 2002 was performed on seven stream reaches in DSF (SPRU, MARS) and DEWA (ADAM, HORN, TOM, VAN, DUNN). Comparisons between the 2002 and 2014 surveys were performed using only these sites (eliminating TUWA and SAW which were not surveyed in 2002), and were evaluated between survey years,

site type, and among stream sites. The variables determined to be most indicative of defining overall hemlock health were the same as above except for (4) where only mean total terminals per branch were compared between surveys. Number of bare terminals per branch was relatively lower during the 2002 survey compared to the 2014 survey and was therefore not recorded. Total terminals were determined by summing HWA infested, non-new and new terminals per branch for both surveys.

Eastern Hemlock Health Rating. – During the 2014 survey 653 trees were surveyed in ravine sites while 284 were sampled on benches. There was a significant difference between ravines and benches in the proportion of EAHE trees rated in health categories 1-4 ($\chi^2 = 269.11$, $df = 4$, $p < 0.001$; Fig. 14). There were significantly more trees rated in categories 1 and 2 (very poor, poor) in the ravines compared to benches. Hemlocks rated 3 and 4 (fair, good) were significantly more common on the bench sites (adj. $p < 0.005$). There was no significant difference in trees rated in health category 5 (pristine), largely because there weren't any (adj. $p > 0.005$). Overall, the benches had significantly more highly rated EAHE trees than the ravines while the ravines had significantly more poorly rated EAHE trees than the benches. Significant differences in number of EAHE trees in various health categories were also found between sites ($\chi^2 = 549.03$, $df = 32$, $p < 0.001$; Fig. 15). TUWA had the highest percentage of poor EAHE trees (rated at 1 and 2) at 79.3%, followed by ADAM with 72.3% (Table 2a). These sites also had the lowest percentages of healthy trees (rated 4 or 5) with ADAM at 0% and TUWA at 1.7%. SPRU and SAW had the lowest percentage of poorly rated trees at 4.4% and 6%, respectively. SPRU and SAW also had the highest percentage of healthy trees at 43.5% and 33.3%, respectively.

In 2002, 548 EAHE trees were surveyed in ravines compared to 537 during the 2014 survey. On benches, 224 trees were surveyed in 2002 and 200 in 2014. Overall, there was a significant difference between survey years in the proportion of EAHE trees in health categories ($\chi^2 = 585.06$, $df = 4$, $p < 0.001$), with each health category (1-5) being significantly different between surveys (adj. $p < 0.005$, Fig. 16). The 2002 survey had a

significantly higher proportion of healthy trees and fewer trees in poor condition than the 2014 survey.

Within ravines, there was a significant difference between survey years in health categories ($\chi^2 = 461.41$, $df = 3$, $p < 0.001$), with each health category (1-4) being significantly different between surveys (adj. $p < 0.006$; Fig. 17). No trees were categorized as pristine in either survey year in ravines. Within benches, there was a significant difference between survey years in health categories 2-5 ($\chi^2 = 177.42$, $df = 4$, $p < 0.001$; adj. $p < 0.005$). No significance difference was observed in the proportion of very poor (1) trees in benches between survey years (adj. $p > 0.005$).

For each study site, there was an overall significant difference in health categories between survey years (ADAM: $\chi^2 = 88.08$, $df = 3$, $p < 0.001$; HORN: $\chi^2 = 61.94$, $df = 3$, $p < 0.001$; TOM: $\chi^2 = 138.81$, $df = 3$, $p < 0.001$; VAN: $\chi^2 = 101.91$, $df = 3$, $p < 0.001$; DUNN: $\chi^2 = 150.48$, $df = 3$, $p < 0.001$; SPRU: $\chi^2 = 73.97$, $df = 3$, $p < 0.001$; MARS: $\chi^2 = 109.16$, $df = 3$, $p < 0.001$). In summary, there were significantly more healthy trees in the 2002 survey and more unhealthy trees in the 2014 survey overall, between site types and among stream sites (Table2b).

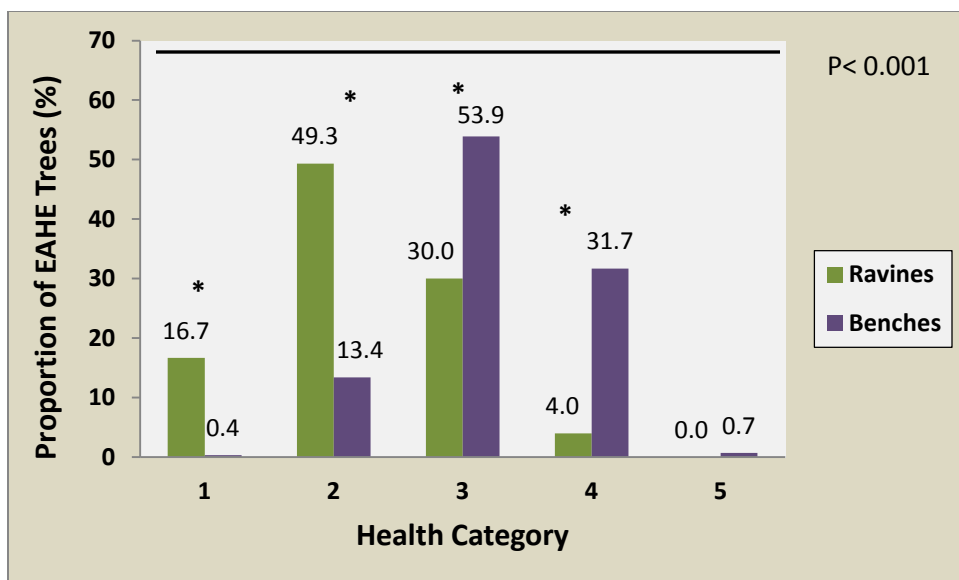


Figure 14. Proportion of EAHE trees evaluated in each health category between site types for 2014 survey. Line at top of graph denotes overall significant difference between site types in health categories ($p < 0.001$). Asterisks denote where significant differences occurred between site types in each health category (adj. $p < 0.005$).

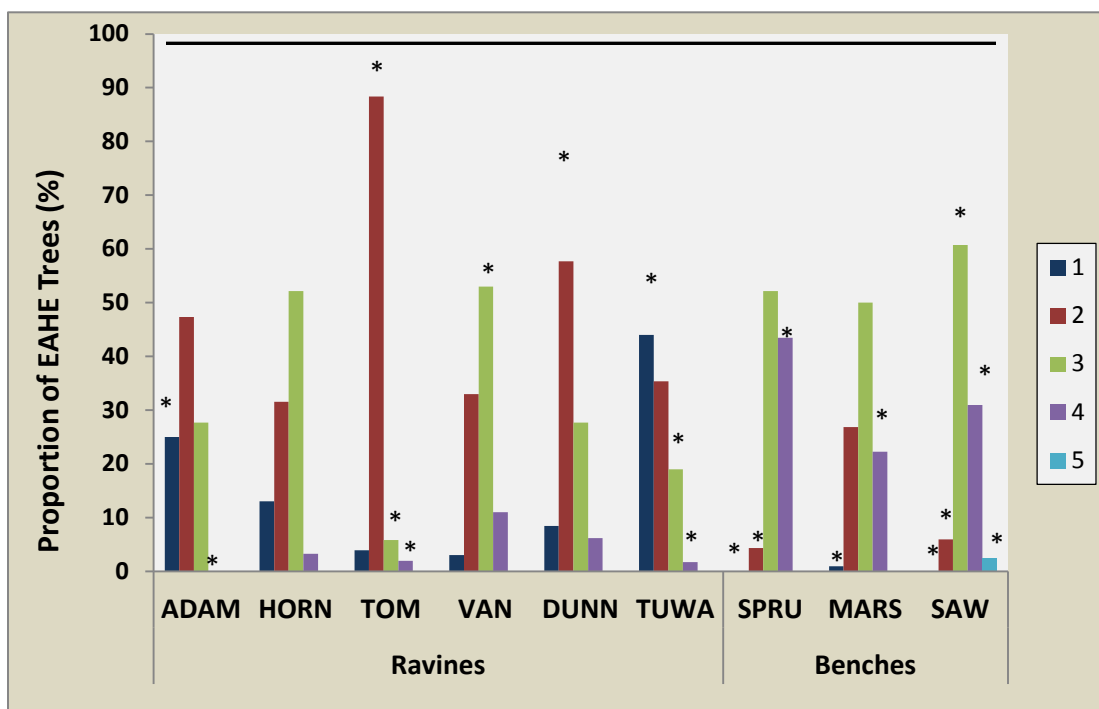


Figure 15. Total EAHE trees evaluated in each health category among sites for 2014 survey. Line at top of graph denotes overall significant difference between sites in health categories ($p < 0.001$). Asterisks denote where significant differences occurred among sites in each health category (adj. $p < 0.0011$).

Table 2a. Total number and proportion of EAHE trees in health categories from 2014 survey.

Site	Very Poor (1)		Poor (2)		Fair (3)		Good (4)		Pristine (5)		N
	n	%	n	%	n	%	n	%	n	%	
Ravines											
ADAM	28	25.00*	53	47.32*	31	27.68	0	0.00*	0	0.00	112
HORN	12	13.04	29	31.52*	48	52.17*	3	3.26*	0	0.00	92
TOM	4	3.88	91	88.35*	6	5.83*	2	1.94*	0	0.00	103
VAN	3	3.00	33	33.00*	53	53.00*	11	11.00*	0	0.00	100
DUNN	11	8.46	75	57.69*	36	27.69	8	6.15*	0	0.00	130
TUWA	51	43.97	41	35.34	22	18.97	2	1.72	0	0.00	116
Benches											0
SPRU	0	0.00	4	4.35	48	52.17*	40	43.48*	0	0.00*	92
MARS	1	0.93	29	26.85*	54	50.00*	24	22.22*	0	0.00	108
SAW	0	0.00	5	5.95	51	60.71	26	30.95	2	2.38	84
<i>Ravines total/mean</i>	<i>109</i>	<i>16.69*</i>	<i>322</i>	<i>49.31*</i>	<i>196</i>	<i>30.89*</i>	<i>26</i>	<i>3.98*</i>	<i>0</i>	<i>0.00</i>	<i>653</i>
<i>Benches total/mean</i>	<i>1</i>	<i>0.35</i>	<i>38</i>	<i>13.38*</i>	<i>153</i>	<i>54.30*</i>	<i>90</i>	<i>31.69*</i>	<i>2</i>	<i>0.70*</i>	<i>284</i>
<i>Total/mean</i>	<i>110</i>	<i>11.74*</i>	<i>360</i>	<i>38.42*</i>	<i>349</i>	<i>38.69*</i>	<i>116</i>	<i>12.38*</i>	<i>2</i>	<i>0.21*</i>	<i>937</i>

*Denote where significant differences occurred in health categories within a study site between survey years (adj. $p < 0.0063$; Table 2b). No comparisons were performed between surveys for TUWA and SAW.

Table 2b. Total number and proportion of EAHE trees in health categories from 2002 survey.

	Very Poor (1)		Poor (2)		Fair (3)		Good (4)		Pristine (5)		
	n	%	n	%	n	%	n	%	n	%	N
Ravines											
ADAM	0	0.00*	16	19.05*	26	30.95	42	50.00*	0	0.00	84
HORN	12	10.00	16	13.33*	28	23.33*	64	53.33*	0	0.00	120
TOM	1	0.76	19	14.39*	40	30.30*	72	54.55*	0	0.00	132
VAN	0	0.00	0	0.00*	9	12.50*	63	87.50*	0	0.00	72
DUNN	1	0.71	7	5.00*	26	18.57	106	75.71*	0	0.00	140
Benches											
SPRU	0	0.00	0	0.00	5	4.63*	81	75.00*	22	20.37*	108
MARS	0	0.00	0	0.00*	11	9.48*	105	90.52*	0	0.00	116
Ravines total/mean	14	2.55*	58	10.58*	129	23.13*	347	63.32*	0	0.00	548
Benches total/mean	0	0.00	0	0.00*	16	7.06*	186	83.04*	22	9.82*	224
Total/mean	14	1.81*	58	7.51*	145	18.54*	533	69.04*	22	2.85*	772

*Denote where significant differences occurred in health categories within a study site between survey years (adj. p< 0.0063).

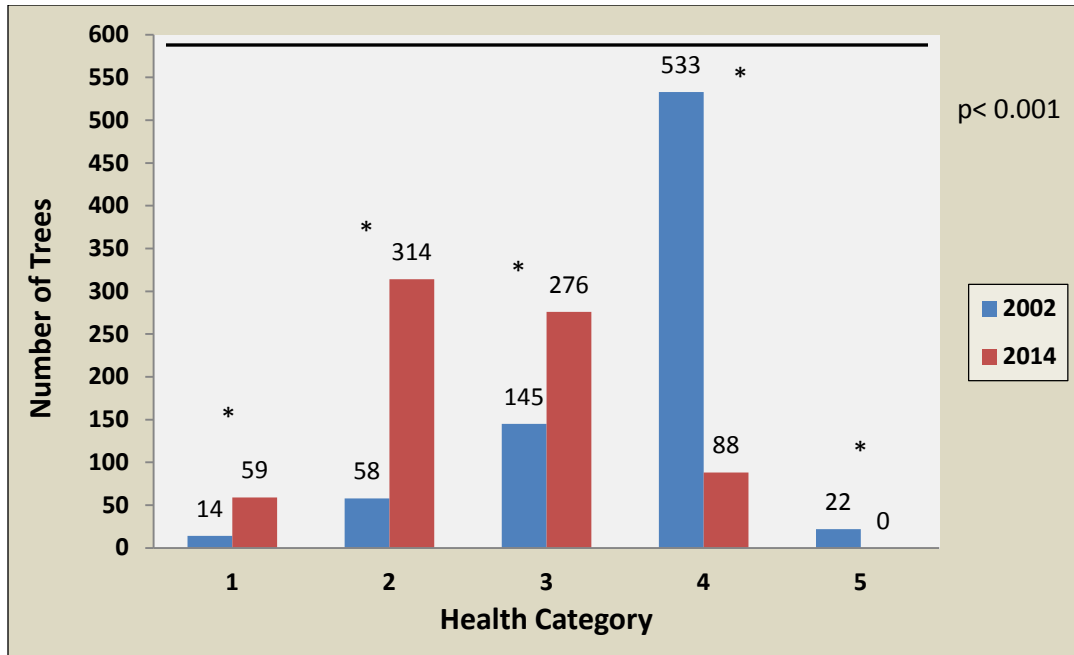


Figure 16. Total EAHE trees in each health category between survey years. Line at top of graph denotes overall significant difference between surveys for tree counts in each health category ($p < 0.001$). Asterisks denote where significant differences occurred in each health category between survey years (adj. $p < 0.005$).

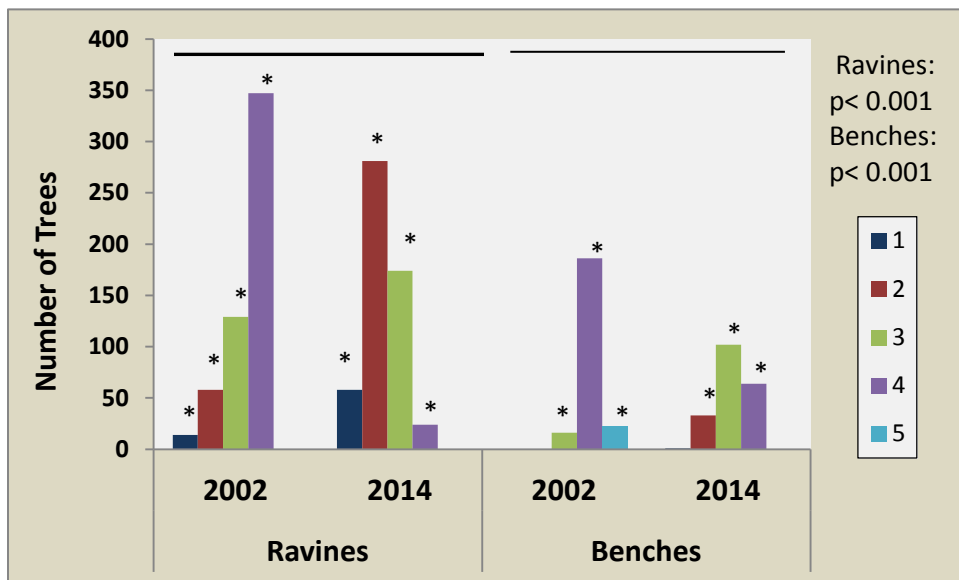


Figure 17. Total trees counted in each health category for both site types between survey years. Lines at top of graph denote significant differences within the ravines ($p < 0.001$) and benches ($p < 0.001$). Asterisks denote where significant differences occurred in health categories within site type between surveys (adj. $p < 0.001$).

Height of 1st Live Branch. – In the 2014 survey, 649 branches were measured for height of the 1st live branch on EAHE trees in ravines while benches had 284 branches measured. The mean height of the 1st live branch in the ravines was 9.84 m while the benches had a significantly lower mean of 3.90 m ($U= 44955$, $z= -12.506$, $p< 0.001$; Fig. 18). There was an overall significant difference among sites for mean height of the 1st live branch ($H= 230.834$, $df= 8$, $p< 0.001$). TUWA had the highest mean height at 12.91 m followed by ADAM at 11.26 m, and then TOM at 11.08 m. MARS had the lowest mean height at 3.46 m followed by the other benches, SPRU at 4.03 m and SAW at 4.33 m. Of the ravine sites, HORN had the lowest mean height of 6.2 m, followed closely by VAN at 6.75 m.

In 2002, 548 EAHE lowest branch heights (m) were measured in the ravines compared to 649 in 2014. On the benches, 224 branches were measured in 2002 compared to 284 in 2014. Overall, there was a significant difference between survey with a mean height of 5.02 m in 2002 and 7.69 m in 2014 ($Z= -8.524$, $p< 0.001$; Fig. 19). In the ravines, there was a significant difference between survey years with ravines in 2002 having a mean height of 5.52 m and in 2014 having a mean height of 9.17 m ($Z= -9.003$, $p< 0.001$). On the benches, there was no significant difference between survey years; 2002 had a mean height of 3.79 m while in 2014 the mean height was 3.73 m ($Z= -0.668$, $p= 0.504$).

Within the ravines, study sites that had a significant difference in mean height of 1st live branch between 2002 and 2014 included ADAM ($\bar{X}=5.98$ vs. 11.26; $Z= -4.568$, $p< 0.001$), TOM (5.16 vs. 11.08; $Z= -7.265$, $p< 0.001$), VAN (4.91 vs. 6.75; $Z= -2.520$, $p= 0.012$), and DUNN (5.95 vs. 9.87; $Z= -4.536$, $p< 0.001$; Fig. 20). HORN was not significantly different between survey years (5.45 vs. 6.2; $Z= -0.689$, $p= 0.491$). On the benches, SPRU had a significant difference in mean height of 1st live branch between 2002 and 2014, with 3.45 vs. 4.03 m ($Z= -2.026$, $p= 0.043$). MARS was not significantly different with 4.11 vs. 3.46 m ($Z= -1.061$, $p= 0.289$). MARS was the only site that had a lower mean height in 2014 compared to 2002.

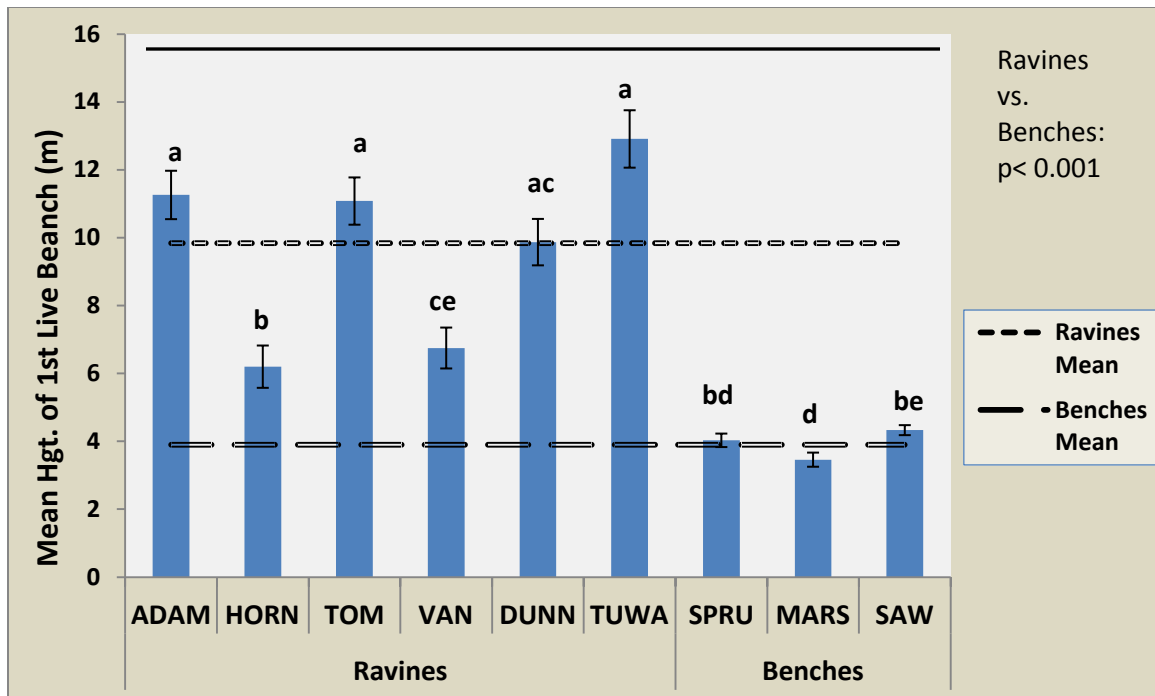


Figure 18. Comparison of mean height of the 1st live branch on EAHE for study sites in 2014. Line at top of graph denotes a significant difference between ravines and benches ($p < 0.001$). Bars that do not share a letter have significantly different mean heights between sites ($p < 0.05$).

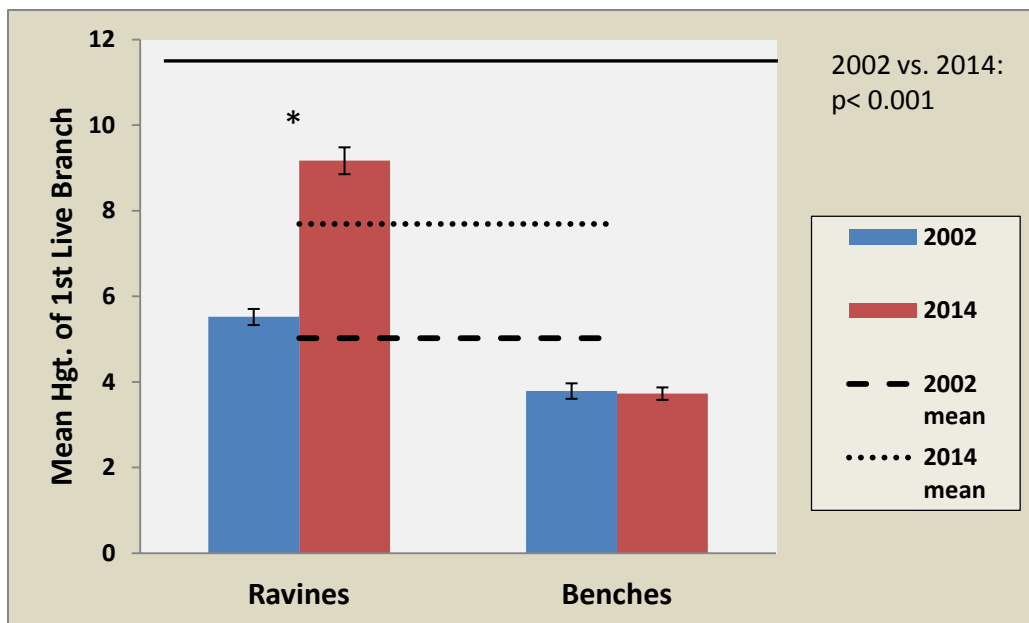


Figure 19. Mean height of 1st live branch between survey years for both site types. Line at top of graph denotes a significant difference in overall mean height of 1st live branch between survey years ($p < 0.001$). Asterisk denotes significant difference between surveys within the ravine sites ($p < 0.001$).

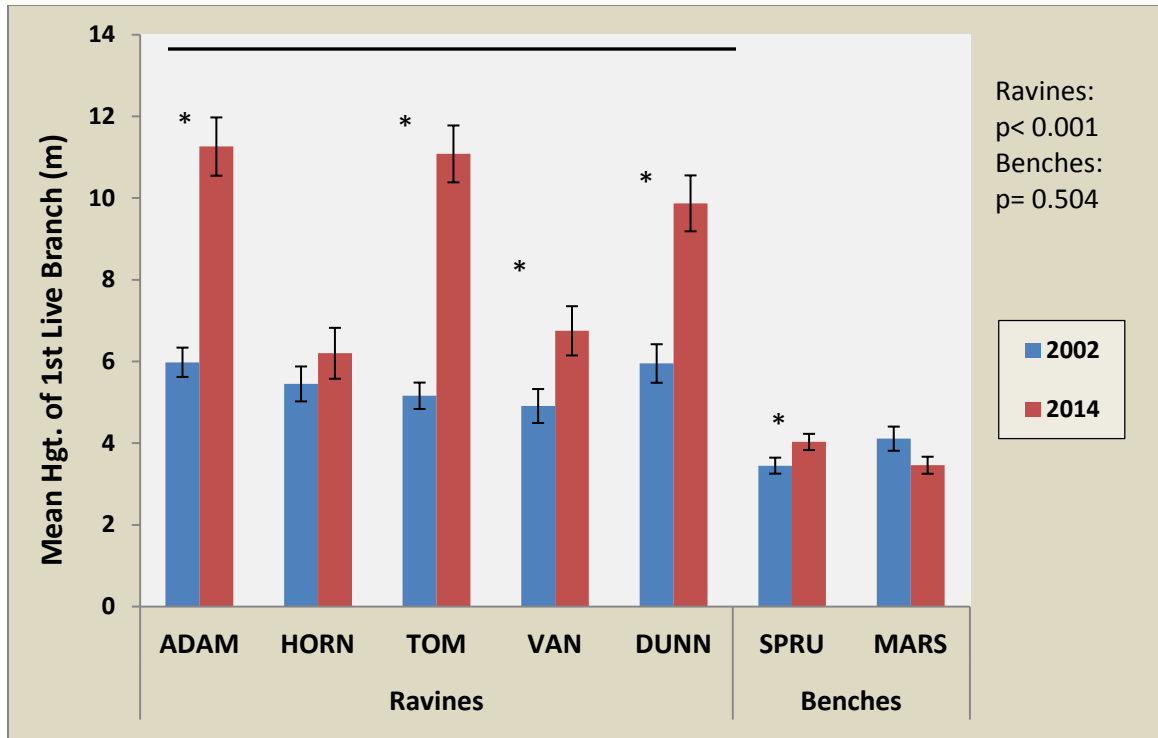


Figure 20. Mean height (m) of 1st live branch on EAHE trees between survey years among study sites. Line at top of graph denotes a significant difference within the ravines between surveys ($p < 0.001$). Asterisks denote a significant difference in mean height within a study site between survey years ($p < 0.05$).

1st Live Branch Health Rating. – In the 2014 survey, 648 lowest live branches in the ravines were assigned to health categories (1, 2, or 3) reflecting their overall health, while benches had 284 branches surveyed. Between site types, there was an overall significant difference in the proportion of branches in the three health categories ($\chi^2=103.25$, $df=2$, $p<0.001$; Fig. 21). There were significantly more branches categorized as poor (1) in the ravines and significantly more branches categorized as good (3) on the benches (adj. $p<0.0083$). There was no significant difference observed in health category 2 (fair) between site types (adj. $p>0.0083$). Significant differences in the proportion of branches in health categories were also found between sites ($\chi^2=276.06$, $df=16$, $p<0.001$; Fig. 22). DUNN had the highest percentage of unhealthy branches (1) at 70.63%, followed by ADAM with 58.04%, and HORN with 48.91% (Table 3a). SPRU and SAW had the highest percentages of healthy branches with 38.04% and 35.71%, respectively. For both ravines and benches, approximately half (ravines= 51.84%, benches= 51.28%) of branches were categorized as fair (2).

In 2002, 548 branches were surveyed in ravines, compared to 532 in 2014. On the benches, 200 branches were surveyed in 2002, while 248 branches were surveyed in 2014. Overall, there was a significant difference between survey years in the proportion of branches in health categories ($\chi^2=389.06$, $df=2$, $p<0.001$; Fig. 23). In 2014, there were significantly more branches categorized as poor (1) and fair (2) than in 2002, and significantly more branches categorized as good (3) in 2002 compared to 2014 (adj. $p<0.0083$). Within the ravines, there was a significant difference in the proportion of hemlock branches in all health categories (1-3) between survey years ($\chi^2=265.33$, $df=2$, $p<0.001$; adj. $p<0.0083$; Fig. 24). Similarly, there was also a significant difference within benches between survey years in all health categories ($\chi^2=156.08$, $df=2$, $p<0.001$; adj. $p<0.0083$). For each study site, there was an overall significant difference in health categories between survey years (ADAM: $\chi^2=44.06$, $df=2$, $p<0.001$; HORN: $\chi^2=46.64$, $df=2$, $p<0.001$; TOM: $\chi^2=11.92$, $df=2$, $p<0.001$; VAN: $\chi^2=83.83$, $df=2$, $p<0.001$; DUNN: $\chi^2=172.05$, $df=2$, $p<0.001$; SPRU: $\chi^2=82.92$, $df=2$, $p<0.001$; MARS: $\chi^2=81.56$,

df= 2, $p < 0.001$). In summary, for both site types and each study site, there were significantly more healthy branches during the 2002 survey and more unhealthy branches in 2014 (Table3b).

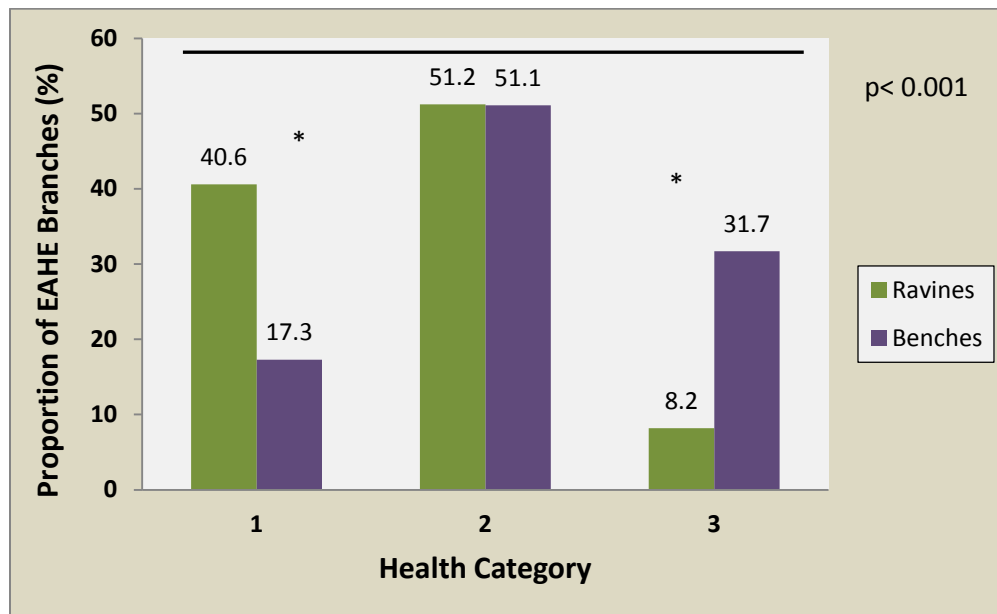


Figure 21. Proportion of EAHE branches observed in each health category between site types from 2014 survey. Line at top of graph denotes overall significant difference between site types ($p < 0.001$). Asterisks denote where significant differences occurred in the three health categories (adj. $p < 0.0083$).

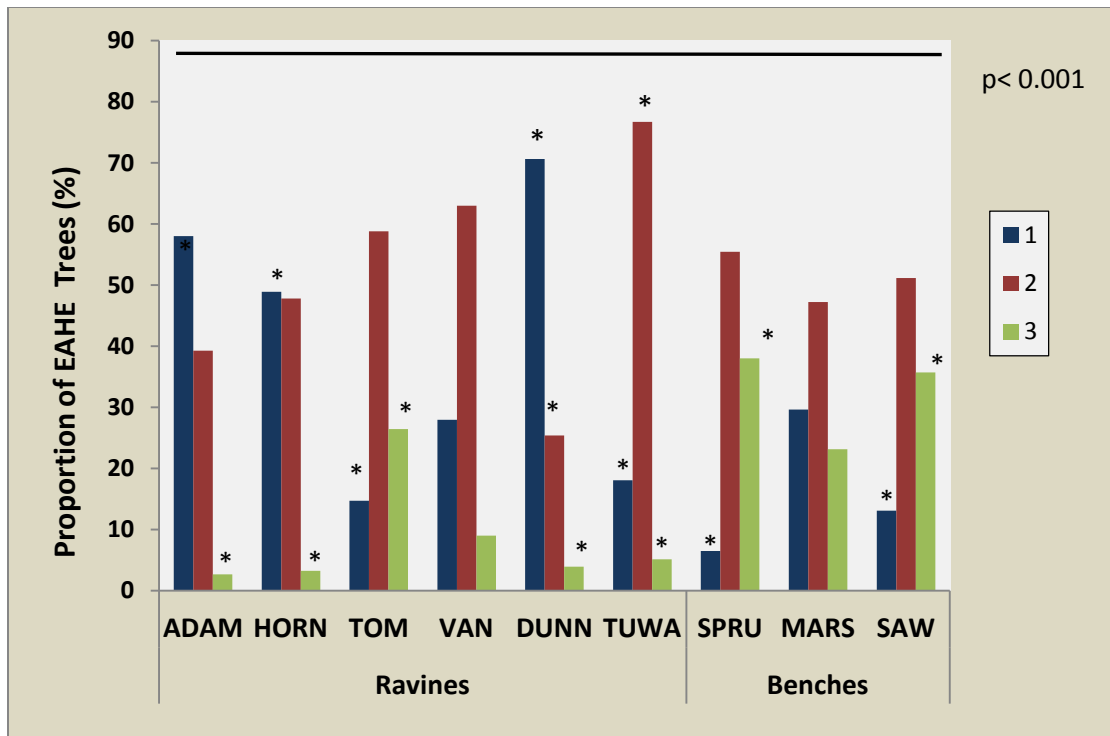


Figure 22. Total EAHE branches evaluated in each health category between sites from 2014 survey. Line at top of graph denotes overall significant difference between sites in health categories ($p < 0.001$). Asterisks denote where significant differences occurred between sites in each health category (adj. $p < 0.0019$)

Table 3a. Total number and proportion of EAHE 1st lowest branches in the three health categories from 2014 survey.

	Poor (1)		Fair (2)		Good (3)		
	n	%	n	%	n	%	N
Ravines							
ADAM	65	58.04*	44	39.29	3	2.68*	112
HORN	45	48.91*	44	47.83	3	3.26*	92
TOM	15	14.71	60	58.82*	27	26.47	102
VAN	28	28.00*	63	63.00*	9	9.00*	100
DUNN	89	70.63*	32	25.40	5	3.97*	126
TUWA	21	18.10	89	76.72	6	5.17	116
Benches							
SPRU	6	6.52*	51	55.43*	35	38.04*	92
MARS	32	29.63*	51	47.22*	25	23.15*	108
SAW	11	13.10	43	51.19	30	35.71	84
<i>Ravines total/mean</i>	<i>263</i>	<i>39.73*</i>	<i>332</i>	<i>51.84*</i>	<i>53</i>	<i>8.43*</i>	<i>648</i>
<i>Benches total/mean</i>	<i>49</i>	<i>16.42*</i>	<i>145</i>	<i>51.28*</i>	<i>90</i>	<i>32.30*</i>	<i>284</i>
<i>Total/mean</i>	<i>312</i>	<i>31.96*</i>	<i>477</i>	<i>51.67*</i>	<i>143</i>	<i>15.59*</i>	<i>932</i>

* Denote where significant differences occurred in the three health categories within a study site between survey years (adj. p< 0.0083). No comparisons between surveys for TUWA and SAW were performed.

Table 3b. Total number and proportion of EAHE 1st lowest branches in health categories from 2002 survey.

	Poor (1)		Fair (2)		Good (3)		
	n	%	n	%	n	%	N
Ravines							
ADAM	23	27.38*	29	34.52	32	38.10*	84
HORN	19	15.83*	55	45.83	46	38.33*	120
TOM	34	25.76	48	36.36*	50	37.88	132
VAN	1	1.39*	16	22.22*	55	76.39*	72
DUNN	3	2.14*	31	22.14	106	75.71*	140
Benches							
SPRU	0	0.00*	3	2.78*	105	97.22*	108
MARS	0	0.00*	23	19.83*	93	80.17*	116
<i>Ravines total/mean</i>	<i>80</i>	<i>14.50*</i>	<i>179</i>	<i>32.21*</i>	<i>289</i>	<i>53.28*</i>	<i>548</i>
<i>Benches total/mean</i>	<i>0</i>	<i>0.00*</i>	<i>26</i>	<i>11.31*</i>	<i>198</i>	<i>88.70*</i>	<i>224</i>
<i>Total/mean</i>	<i>80</i>	<i>10.36*</i>	<i>205</i>	<i>26.24*</i>	<i>487</i>	<i>63.40*</i>	<i>772</i>

*Denote where significant differences occurred in health categories within a study site between survey years (adj. p< 0.0083).

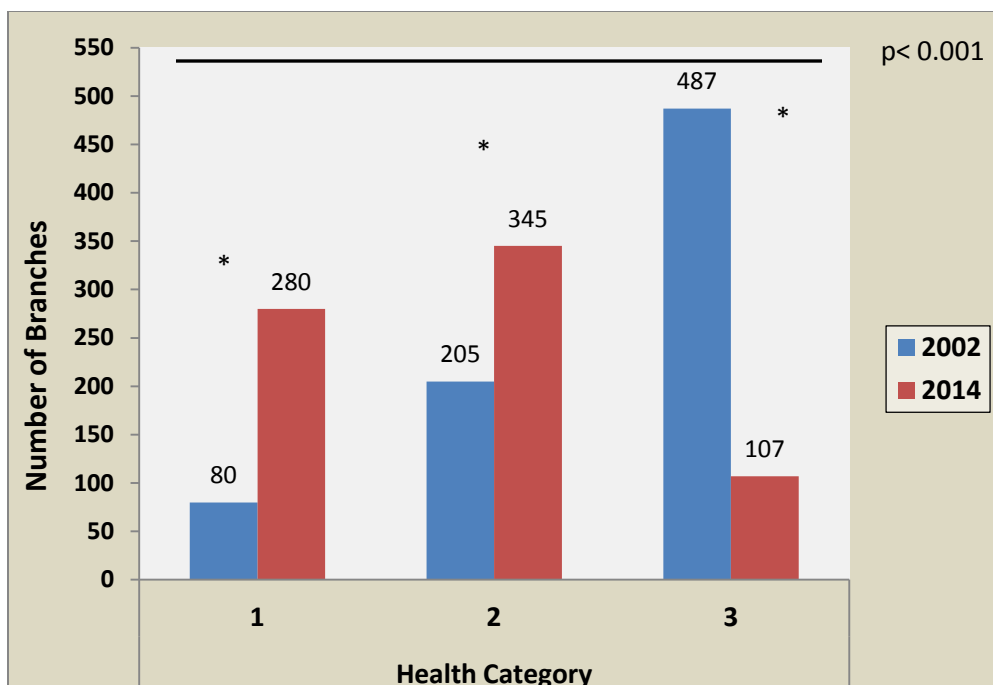


Figure 23. Total EAHE branches in each health category between survey years. Line at top of graph denotes overall significant difference between survey years for number of branches in the three health categories ($p < 0.001$). Asterisks denote where significant differences occurred in each health category between survey years (adj. $p < 0.0083$).

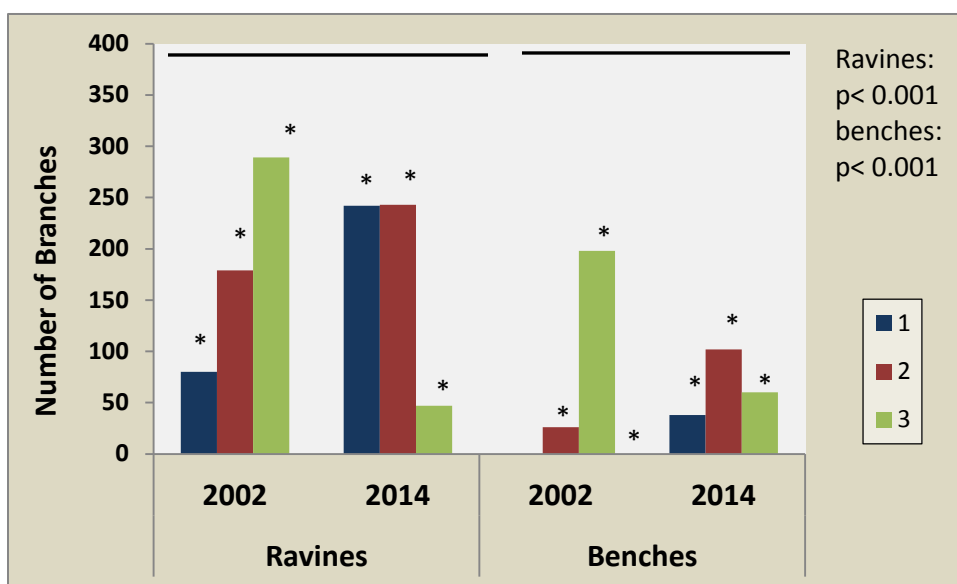


Figure 24. Number of branches in each health category for both site types between survey years. Lines at top of graph denote significant differences within the ravines ($p < 0.001$) and benches ($p < 0.001$). Asterisks denote where significant differences occurred in each health category within site types between survey years (adj. $p < 0.0083$).

Terminal Measurements. – In 2014, terminal type (non-new, new, bare, and HWA infested) was quantified for the lowest live branch. In ravines, non-new and new terminals were counted on 354 branches, bare terminals were counted on 351 branches, and HWA infested terminals were counted on 243 branches. On benches, non-new, new and bare terminals were counted on 282 branches, and HWA infested terminals were counted on 270 branches (Table 4). Total branches at ravines and benches for each terminal type differ due to accessibility of the lowest branch. Between the ravines and benches, there was a significant difference in mean number of bare ($U=43173$, $z=-2.797$, $p=0.005$) and HWA infested terminals ($U=22556.5$, $z=-7.074$, $p<0.001$; Fig. 25). Ravines had a significantly higher mean number of bare and HWA infested terminals with 16.9 and 2.8, respectively, compared to benches which had a mean of 13.6 bare and 1.3 HWA infested terminals per branch. There was no significant difference in mean number of non-new ($U=47618$, $z=-1.016$, $p=0.309$) and new terminals ($U=49344$, $z=-0.249$, $p=0.803$) between the ravines and benches. Between ravines and benches, the mean number of non-new terminals per branch was 27.5 and 30, respectively, and the mean number of new terminals per branch was 11.7 and 10.6, respectively. For all four types of terminals, there were overall significant differences between study sites (Non-new: $H=29.44$, $df=6$, $p<0.001$; new: $H=64.04$, $df=6$, $p<0.001$; bare: $H=119.02$, $df=6$, $p<0.001$; HWA infested: $H=85.63$, $df=6$, $p<0.001$; Fig. 26-29)

For survey comparison, only mean total terminals per branch were compared. In 2002, 629 branches were sampled, compared to 431 branches in 2014. There were significantly more total terminals per branch in 2002 with a mean of 60.2 compared to 42.3 in 2014 ($Z=-1.115$, $p<0.001$; Fig. 30). In the ravines, there were significantly less total terminals per branch in 2014 with a mean of 43.5 when compared to 58.4 in 2002 ($Z=-8.591$, $p<0.001$). Similarly, on the benches, there were significantly less total terminals per branch in 2014 with a mean of 40.7 compared to 63.5 in 2002 ($Z=-9.950$, $p<0.001$).

Within the ravines, study sites that had a significant difference in mean number of total terminals per branch between 2002 and 2014 included HORN ($\bar{X}$ =60.46 vs. 41.33; $Z = -6.466$, $p < 0.001$), VAN (58.64 vs. 40.58; $Z = -4.677$, $p < 0.001$), and DUNN (62.82 vs. 51.07; $Z = -3.299$, $p = 0.001$; **Fig. 31**). ADAM was not significantly different between survey years with a mean of 47.64 vs. 41.75, respectively ($Z = -1.637$, $p = 0.102$). Both bench sites had a significant difference in number of total terminals between 2002 and 2014 with SPRU having 63.35 vs. 42.3 ($Z = -6.256$, $p < 0.001$) and MARS having 63.53 vs. 39.3 ($Z = -7.710$, $p < 0.001$).

Table 4. Total branches surveyed and mean number of each terminal type per branch for each survey year.

Site	<u>Non-new (n^a, $\bar{X}$)</u>		<u>New (n, $\bar{X}$)</u>		<u>HWA (n, $\bar{X}$)</u>		<u>Bare (n, $\bar{X}$)</u>	
	2002	2014	2002	2014	2002	2014	2002	2014
Ravines								
ADAM	80, 28.31	83, 30.84	80, 9.59	83, 8.71	80, 9.74 *	48, 2.65 *	NA	80, 11.02
HORN	120, 37.51 *	84, 31.25 *	120, 9.65 *	84, 6.65 *	119, 13.38 *	73, 3.52 *	NA	84, 14.67
TOM	129, 36.27 *	NA	129, 9.68 *	NA	129, 11.12	NA	NA	NA
VAN	72, 35.19 *	94, 25.34 *	72, 14.18	94, 14.14	72, 9.26 *	65, 1.09 *	NA	94, 12.29
DUNN	140, 34.57 *	93, 29.84 *	140, 21.84 *	93, 13.71 *	136, 6.53 *	57, 4.07 *	NA	93, 28.6
Benches								
SPRU	106, 43.27 *	92, 29.67 *	106, 19.00 *	92, 11.87 *	106, 1.08	88, 0.97	NA	92, 12.92
MARS	116, 40.21 *	106, 27.80 *	116, 15.87 *	106, 10.73 *	116, 7.46 *	100, 0.81 *	NA	106, 13.11
SAW	NA	84, 33.21	NA	84, 8.99	NA	82, 2.15	NA	84, 14.77
<i>Ravines total/mean</i>	<i>541, 34.79 *</i>	<i>354, 27.47 *</i>	<i>541, 13.41</i>	<i>354, 11.66</i>	<i>536, 10.00 *</i>	<i>243, 2.83 *</i>	<i>NA</i>	<i>351, 16.89</i>
<i>Benches total/mean</i>	<i>222, 41.67 *</i>	<i>282, 30.02 *</i>	<i>222, 17.36 *</i>	<i>282, 10.58 *</i>	<i>222, 4.41 *</i>	<i>270, 1.27 *</i>	<i>NA</i>	<i>282, 13.55</i>
<i>Total/mean</i>	<i>763, 36.79 *</i>	<i>636, 29.71 *</i>	<i>763, 14.56 *</i>	<i>636, 10.69 *</i>	<i>758, 8.36 *</i>	<i>513, 2.01 *</i>	<i>NA</i>	<i>633, 15.34</i>

^a Sample sizes (n) for terminal type within a site for 2014 survey differ due to some branches being inaccessible.

*Denote significant difference in mean terminals in each category between survey years. Results must be interpreted carefully due to significant difference in total terminals between surveys (seen in Fig. 18-20). No comparisons made within Bare category and site SAW because these were not surveyed in 2002. TUWA and TOM were not included in the terminal survey.

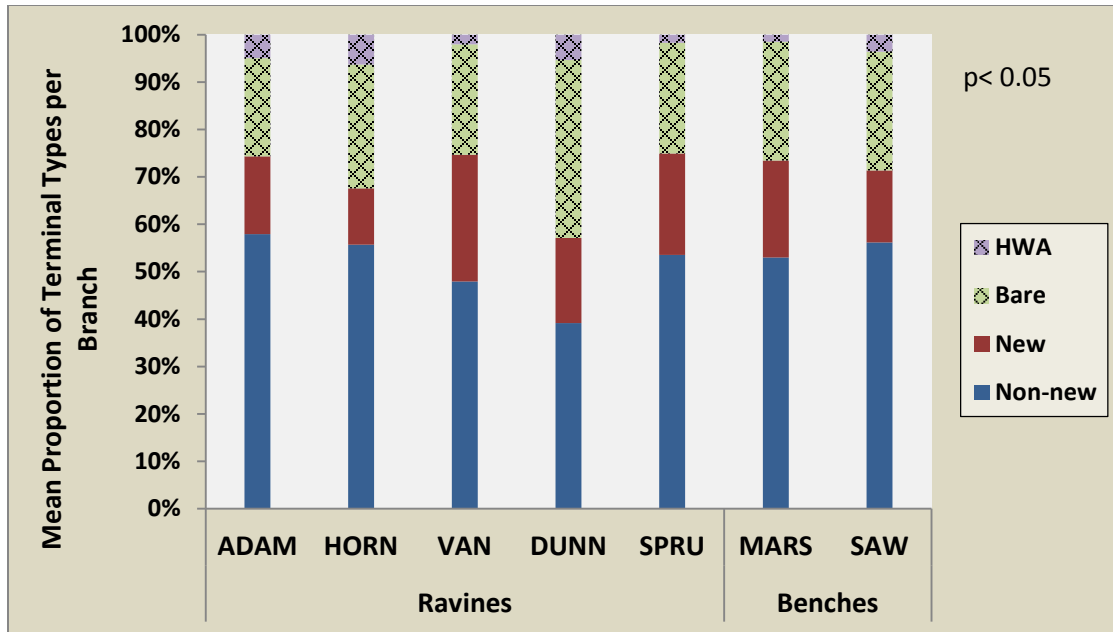


Figure 25. Mean proportion of terminal types per branch at each study site in 2014. Bars with crosshatches denote a significant difference between ravines and benches ($p < 0.05$).

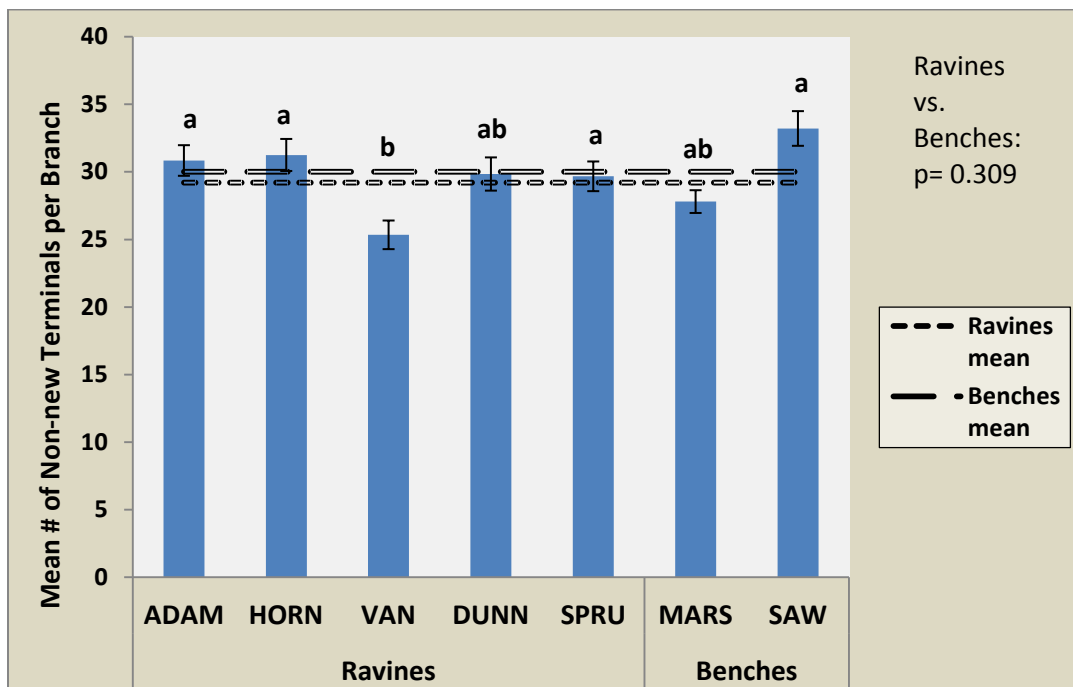


Figure 26. Comparison of mean number of non-new terminals per branch for study sites in 2014. Bars that do not share a letter have significantly different mean heights between sites.

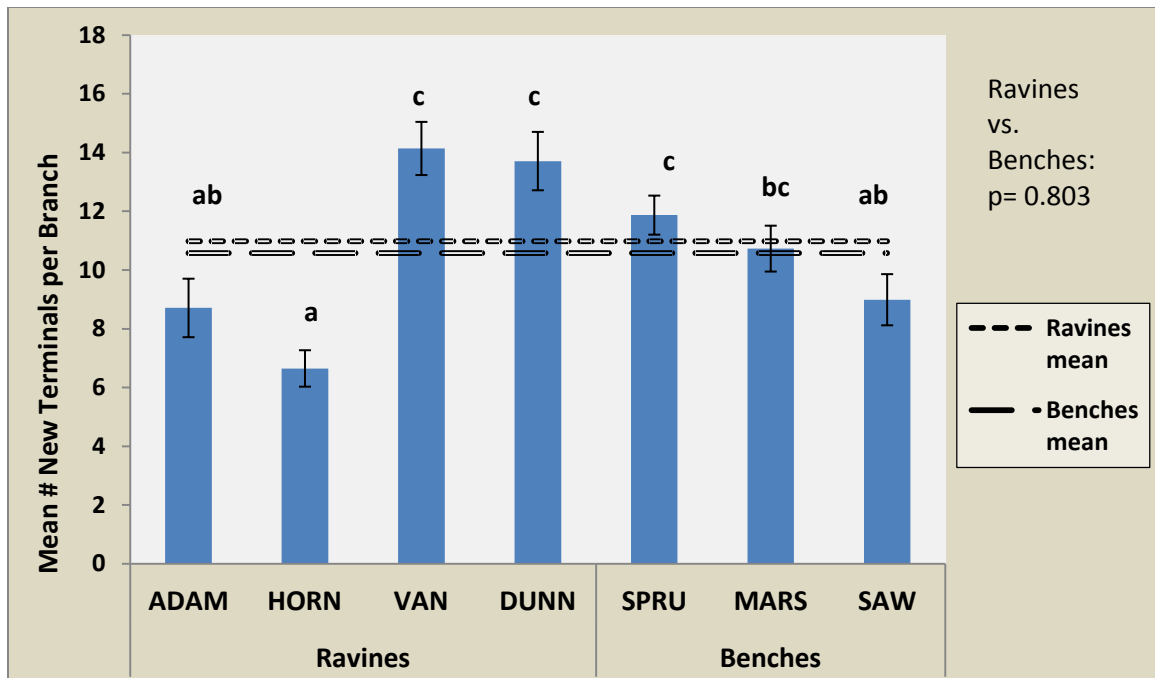


Figure 27. Comparison of mean number of new terminals per branch for study sites in 2014. Bars that do not share a letter have significantly different mean heights between sites.

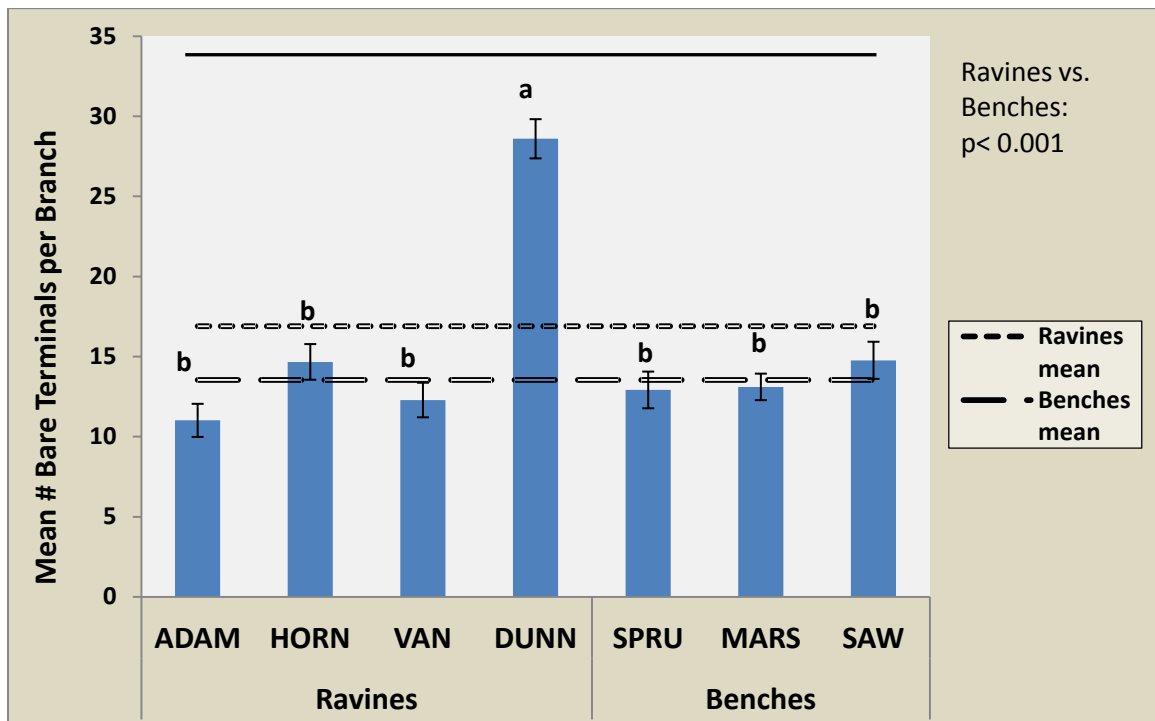


Figure 28. Comparison of mean number of bare terminals per branch for study sites in 2014. Line at top of graph denotes a significant difference between ravines and benches. Bars that do not share a letter have significantly different mean heights between sites.

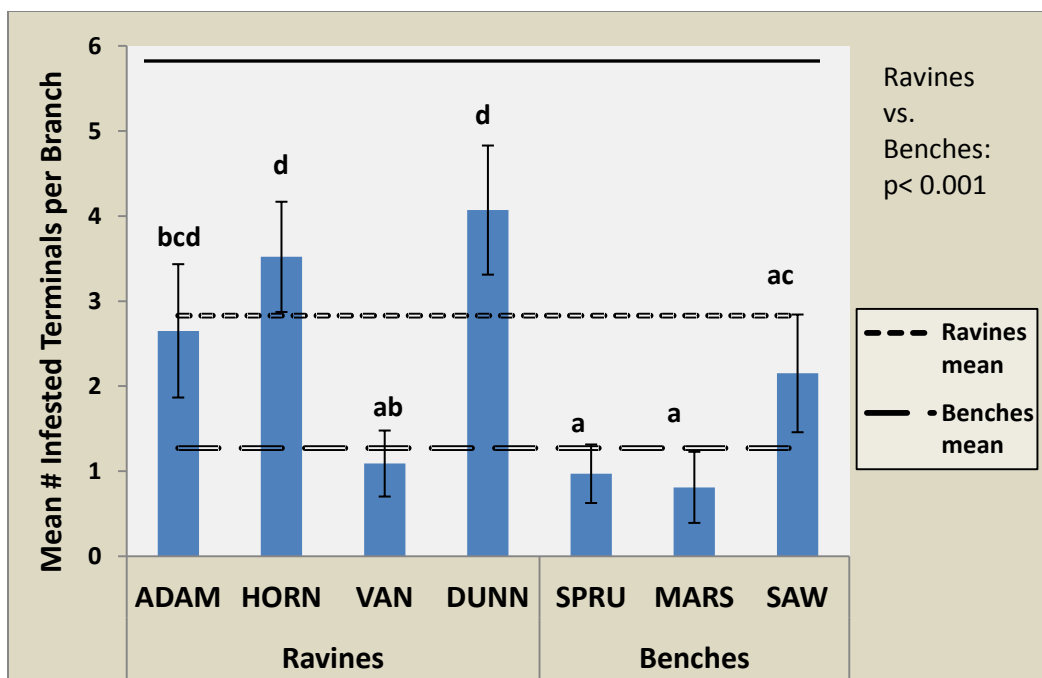


Figure 29. Comparison of mean number of HWA infested terminals per branch for study sites in 2014. Line at top of graph denotes a significant difference between ravines and benches. Bars that do not share a letter have significantly different mean heights between sites.

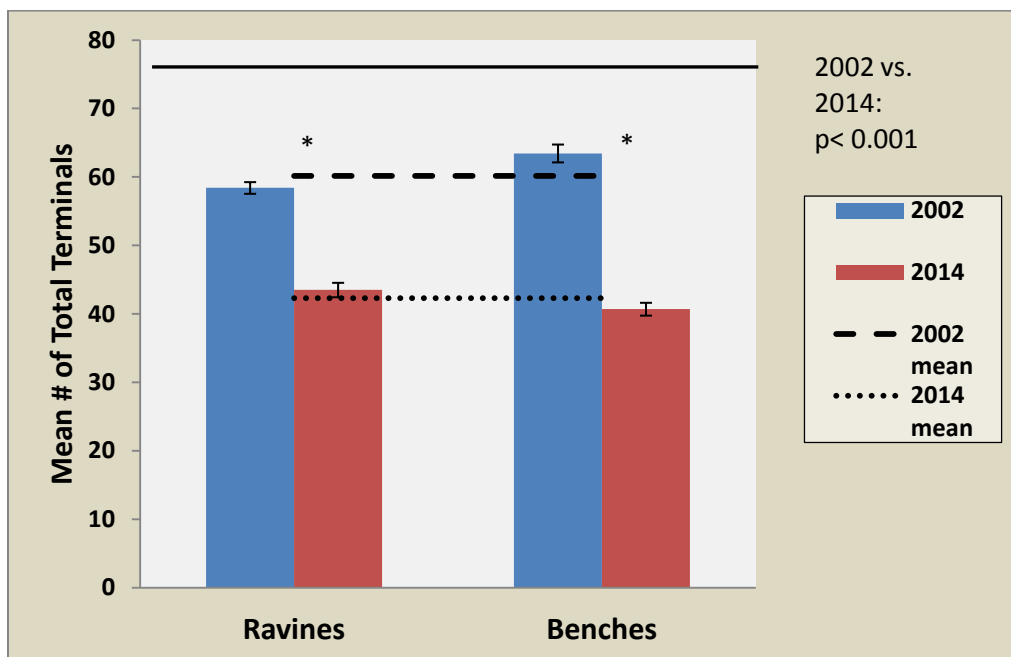


Figure 30. Comparison of mean number of total terminals between survey years at each site type. Line at top of graph denotes a significant difference in overall mean total terminals between survey years ($p < 0.001$). Asterisks denote a significant difference within a site type between survey years ($p < 0.001$).

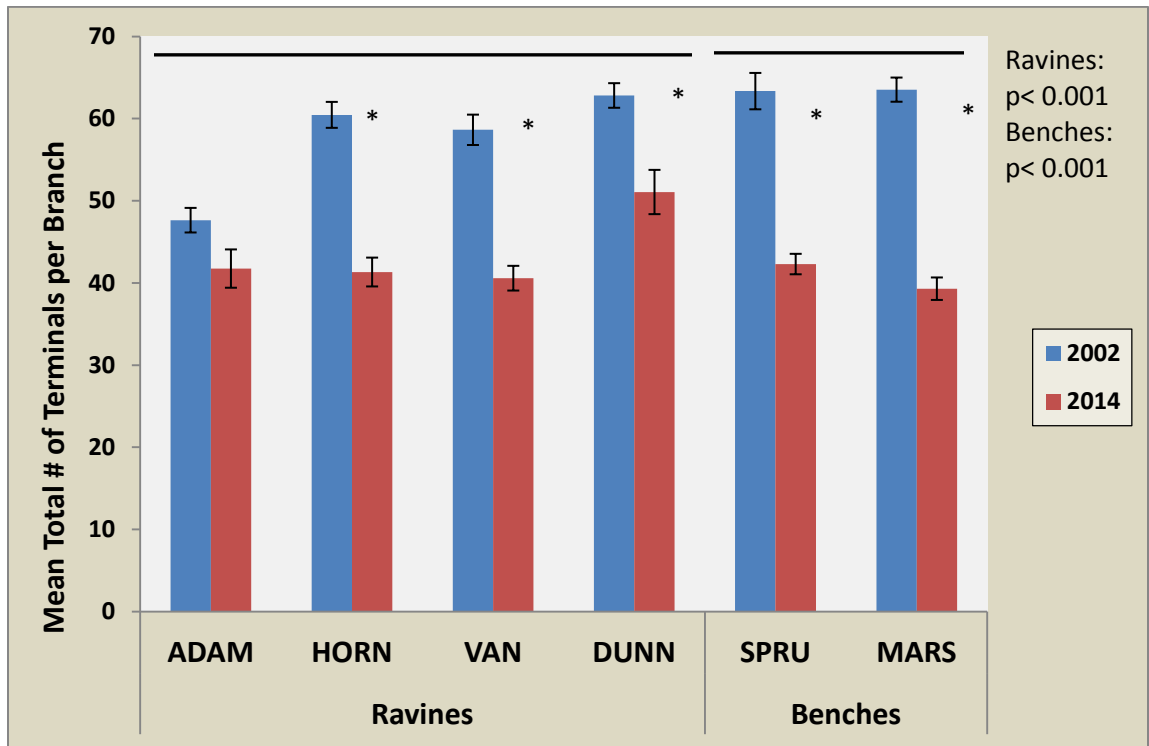


Figure 31. Comparison of mean number of total terminals between survey years at each study site. Lines at top of graph denote a significant difference within site types between survey years (Ravines: $p < 0.001$; Benches: $p < 0.001$). Asterisks denote a significant difference within a study site between survey years ($p < 0.05$).

Reproductive Success

Pair Density. - In the current study, a total of 142 nests were found among all study sites over two field seasons (Fig. 32). Sixty-four nests were found in the ravines while 78 were found on benches. For ravine sites, DUNN had the highest number of total nests with 39, followed by HORN with a considerably lower number of only 11 nests. With only two nests found, TOM had the lowest number of nests. For bench sites, SPRU had the highest number of total nests with 41, and MARS had the lowest number of nests with only four.

Mean pair density across all sites was 3.4 pairs/km (range: 0.0-9.0 pairs/km; Table 5a). The benches had a significantly higher mean pair density of 6.4 pairs/km (range: 1.6-9 pairs/km) than the ravines with a mean pair density of 1.9 pairs/km (range: 0-4.6 pairs/km; $U = 6.000$, $z = -2.333$, $p = 0.019$; Fig. 33). Within the ravine sites, DUNN had the highest mean pair density with 4.6 pairs/km, followed by HORN with 2.4 pairs/km. TOM had the lowest mean pair density with 0.68 pairs/km. Within the bench sites, SPRU had the highest mean pair density with 9.0 pairs/km, while MARS had 1.6 pairs/km. There was no significant difference in pair density between study sites ($H = 12.643$, $df = 7$, $p = 0.081$). Pair densities were similar for site types and each site between 2013 and 2014.

Study sites comparable with 2001/02 pair density data include HORN, TOM, VAN, DUNN, SPRU and MARS. Between studies, differences were seen in number of nests located at each site (Table 5a-b). At HORN, DUNN, and SPRU more nests were found in the current study and at TOM, VAN, and MARS fewer nests were found. This pattern is the same for pair densities at each site between studies, except for DUNN, which is likely due to a high number of renests in 2014. In comparison with the 2001/02 study, 2013/14 had a significantly lower overall mean pair density with 4.8 vs. 3.1 pairs/km, respectively ($Z = -1.962$, $p = 0.050$; Fig. 34). Within the ravines, 2013/14 also had a significantly lower mean pair density than 2001/02 with 2.1 vs. 4.0 pairs/km,

respectively ($Z = -2.24$, $p = 0.025$; Fig. 35). Similarly, on the benches, there was a lower mean pair density in 2013/14 than 2001/02 with 5.3 vs. 6.4 pairs/km respectively, however, it was not significant ($Z = -0.736$, $p = 0.461$). There were no significant differences within sites between study years for mean pair density. HORN and SPRU were the only sites where the mean pair density was higher in 2013/14 than in 2001/02.

Table 5a. Acadian flycatcher nesting success at study sites in 2013 and 2014.

Site	<u>Site Length</u>	<u>No. Nests</u>		<u>No. Pairs</u>		<u>Pairs / km</u>		<u>Nests / pair</u>		<u>Nests / km</u>	
	(km)	2013	2014	2013	2014	2013	2014	2013	2014	2013	2014
Ravines											
ADAM	1.66	1	4	1	3	0.60	1.81	1.00	1.33	0.60	2.41
HORN	1.67	4	7	3	4	1.80	2.40	1.33	1.75	2.40	4.19
TOM	1.46	0	2	0	1	0.00	0.68	0.00	2.00	0.00	1.37
VAN	1.68	1	6	1	3	0.60	1.79	1.00	2.00	0.60	3.57
DUNN	2.85	17	22	13	13	4.56	4.56	1.31	1.69	5.96	7.72
Benches											
SPRU	1.45	17	24	13	13	8.97	8.97	1.31	1.85	11.72	16.55
MARS	1.89	NA	4	NA	3	NA	1.57	NA	1.33	NA	2.09
SAW	1.62	15	18	10	10	6.17	6.17	1.50	1.80	9.26	11.11
Ravine totals/mean	-	23	41	18	24	1.51	2.25	0.93	1.75	1.91	3.85
Bench totals/mean	-	32	46	23	26	7.57	5.57	1.41	1.66	10.49	9.92
Pooled totals/mean	-	55	87	41	50	3.24	3.49	1.06	1.72	4.36	6.13

Table 5b. Acadian flycatcher nesting success at study sites in 2001 and 2002.

Site	<u>Site Length</u>	<u>No. Nests</u>		<u>No. Pairs</u>		<u>Pairs / km</u>		<u>Nests / pair</u>		<u>Nests / km</u>	
	(km)	2001	2002	2001	2002	2001	2002	2001	2002	2001	2002
Ravines											
HORN	1.63	2	1	3	2	1.84	1.23	0.67	0.50	1.23	0.61
TOM	1.65	4	6	3	4	1.82	2.42	1.33	1.50	2.42	3.64
VAN	1.15	NA	9	NA	7	NA	6.09	NA	1.29	NA	7.83
DUNN	1.56	10	12	9	11	5.77	7.05	1.11	1.09	6.41	7.69
Benches											
SPRU	1.3	10	15	8	10	6.15	7.69	1.25	1.50	7.69	11.54
MARS	1.89	14	14	11	11	5.82	5.82	1.27	1.27	7.41	7.41
Ravine totals/mean	-	16	28	15	24	3.14	4.20	1.04	1.10	3.35	4.94
Bench totals/mean	-	24	29	19	21	5.99	6.76	1.26	1.39	7.55	9.48
Pooled totals/mean	-	40	57	34	45	4.28	5.05	1.13	1.19	5.03	6.45

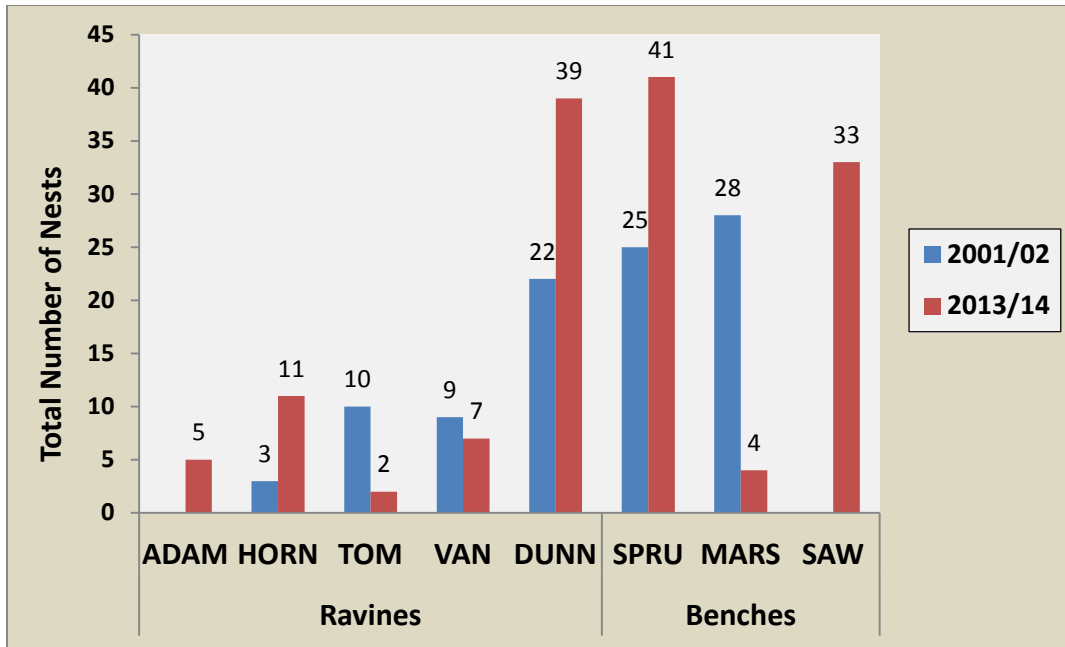


Figure 32. Total number of nests found between studies at each study site.

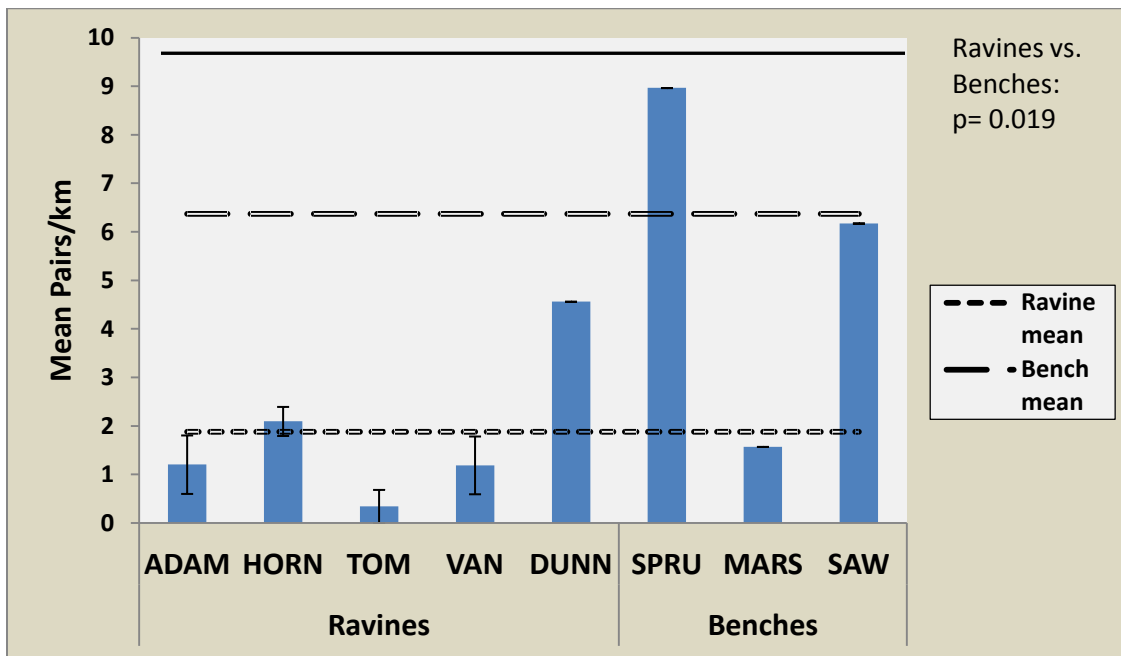


Figure 33. Pair density at each study site in 2013/14. Line at top of graph denotes a significant difference in mean pair density between ravines and benches.

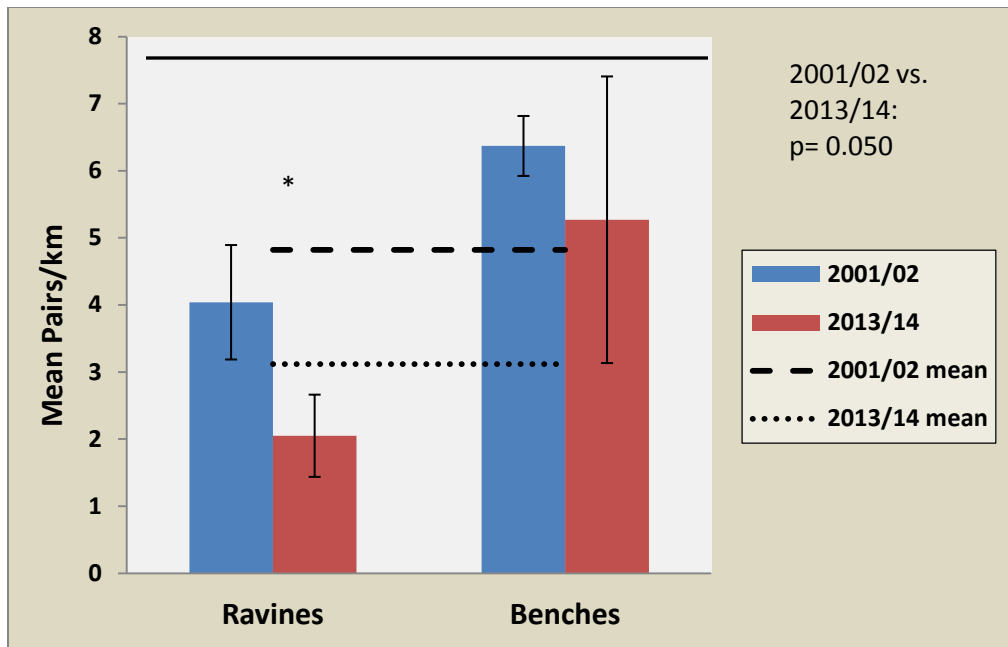


Figure 34. Comparison of pair densities between studies at both site types. Line at top of graph denotes significant difference in mean pair/density between studies ($p = 0.050$). Asterisk denotes a significant difference between studies within the ravines ($p = 0.025$).

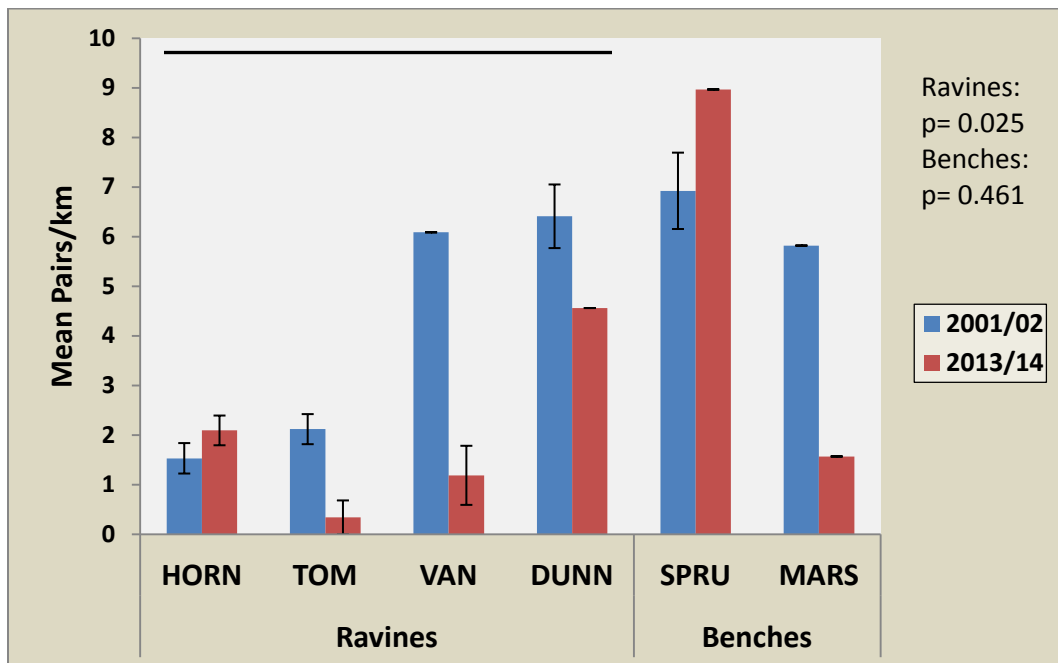


Figure 35. Comparison of pair densities between studies at each study site. Line at top of graph denotes a significant difference with the ravines ($p = 0.025$).

Nest Survival. – The overall apparent success rate was 57.7% and the Mayfield survival rate was 51.1% (Table 6a). The benches had a non-significant higher Mayfield survival rate than the ravines with 55.3% vs. 48.7%, respectively ($U = 18$, $z = -0.860$, $p = 0.440$; Fig. 36). However, if the atypical low mean survival rate of 3.9% at TOM is excluded, the mean survival rate for ravines increases to 59.8%, which is higher than the benches, though not significantly. There were no significant differences in Mayfield survival rates between study sites ($H = 5.892$, $df = 7$, $p = 0.552$).

Of comparable sites (HORN, TOM, VAN, DUNN, SPRU, MARS), apparent nest success in 2001/02 was similar to the current study with a nest survival rate of 56.9% compared to the 2013/14 survival rate of 55.3%. The Mayfield survival rate in 2001/02 of 57.8% was non-significantly higher than the Mayfield survival rate of 50.4% in 2013/14 ($Z = -1.098$, $p = 0.272$; Table 6b). Ravines had a non-significant higher Mayfield survival rate of 62.8% in 2001/02 compared to 43.5% in 2013/14, and benches had a lower survival rate of 51.7% compared to 56.1%, respectively (Ravines: $Z = -1.400$, $p = 0.161$; benches: $Z = -0.365$, $p = 0.715$; Fig. 37). No significant differences in Mayfield survival rates occurred at any sites between studies (Fig. 38).

Overall in the current study, nestling-stage Mayfield survival was non-significantly greater than egg-stage Mayfield survival with 74.2% vs. 66.9%, respectively (Table 7). There were no significant differences between the ravines and benches in the number of nests that failed during egg-stage, nestling-stage, or that were abandoned ($\chi^2 = 2.262$, $df = 3$, $p = 0.52$; Fig. 39). Nestling-stage survival was greater than egg-stage survival for both the ravines and benches; 69.5% vs. 61.1% and 86% vs. 81.6%, respectively. There were no significant differences for type of nest failure between sites ($\chi^2 = 22.341$, $df = 21$, $p = 0.38$). Egg-stage was slightly greater than nestling-stage survival at sites HORN and SAW; all other sites had greater nestling-stage survival rates.

In the current study, for nests placed in EAHE, reproductive success was not influenced by health of the hemlock tree. There was no significant difference between

successful and failed nests in each hemlock health category ($\chi^2 = 2.147$, $df = 3$, $p = 0.543$; Fig. 40). Ravines had a non-significant higher proportion of nests placed in very poor and poor EAHE trees compared to the benches; 46.8% vs. 33.3%, respectively (Table 8). Benches had a non-significant higher proportion of nests placed in EAHE trees rated as good than in ravines with 19.4% vs. 4.9%. There were no significant differences in successful and failed nests between site types for each EAHE health category (very poor: $p = 0.667$, FET; poor: $p = 0.181$, FET; fair: $p = 0.058$, FET; good: $p = 0.625$, FET). Similarly, there were no significant differences in successful and failed nests among sites for each EAHE health category (very poor: $\chi^2 = 3.000$, $df = 2$, $p = 0.223$; poor: $\chi^2 = 6.355$, $df = 6$, $p = 0.385$; fair: $\chi^2 = 7.789$, $df = 6$, $p = 0.254$; good: $\chi^2 = 3.139$, $df = 4$, $p = 0.535$). No nests were placed in EAHE trees rated as pristine (highest rating) because there were virtually no pristine trees remaining among these sites.

In the current study, nests in eastern hemlock had a non-significant slightly lower apparent success rate (56.6%, $n = 113$) than nests placed in hardwood trees (62.1%, $n = 29$; $p = 0.676$, FET; Table 9a). Rhododendron and black gum (*Nyssa sylvatica*), had 100% apparent success ($n = 1$ each spp.), with witch-hazel having the lowest apparent success at 40%. No significant differences in apparent nest survival and mean number of fledglings per nest were observed between each nest tree species ($\chi^2 = 4.081$, $df = 7$, $p = 0.770$; $H = 6.356$, $df = 7$, $p = 0.499$).

The current study's apparent nest survival in eastern hemlock (56.6%) was nearly identical with the apparent success rate of 56% ($n = 91$) in 2001/02 (Table 9a/9b). Nest survival in hardwood trees was lower in 2001/02 than 2013/14 with an apparent success rate of 50% vs. 62.1%, respectively. No significant differences were observed between studies in apparent survival for eastern hemlock nests and hardwood nests (EAHE: $\chi^2 = 2.517$, $df = 2$, $p = 0.284$; HARD: $p = 0.690$, FET). Apparent nest survival rates did not differ significantly with respect to other variables including surrounding vegetation structure, nest placement, and nest concealment (Table 10).

Table 6a. Acadian flycatcher nest survival at sites during 2013 and 2014.

Year	Site	<u>No. Nests</u>	<u>No. Successful Nests</u>	<u>No. Successful Nests / km</u>	<u>Apparent Success (%)</u>	<u>Mayfield Success (%)</u>
2013	Ravine					
	ADAM	1	1	0.60	100	100
	HORN	4	4	2.40	100	100
	TOM	0	0	0.00	0.00	0.00
	VAN	1	1	0.60	100	100
	DUNN	17	13	4.56	76.47	57.28
	Bench					
	SPRU	17	13	8.97	76.47	67.56
	SAW	15	9	5.56	60.00	43.71
	MARS	NA	NA	NA	NA	NA
	<i>Ravine totals/mean</i>	23	19	1.63	75.29	71.46
	<i>Bench totals/mean</i>	32	22	7.27	68.24	55.64
	<i>Pooled totals/mean</i>	55	41	3.78	85.49	78.09
2014	Ravine					
	ADAM	4	2	1.20	50.00	38.58
	HORN	7	2	1.20	28.57	30.69
	TOM	2	0	0.00	0.00	7.78
	VAN	6	3	1.79	50.00	24.77
	DUNN	22	8	2.81	36.36	27.35
	Bench					
	SPRU	24	10	6.90	41.67	31.19
	SAW	18	13	8.02	72.22	63.65
	MARS	4	3	1.57	75.00	62.81
	<i>Ravine totals/mean</i>	41	15	1.40	32.99	25.83
	<i>Bench totals/mean</i>	46	26	5.50	62.96	52.55
	<i>Pooled totals/mean</i>	87	41	2.94	44.23	35.85
Both yrs.	<i>Ravine totals/mean</i>	64	34	1.52	54.14	48.65
	<i>Bench totals/mean</i>	78	48	6.20	65.02	55.29
	<i>Pooled totals/mean</i>	142	82	3.08	57.78	51.14

Table 6b. Acadian flycatcher nest survival at sites during 2001 and 2002.

Year	Site	<u>No. Nests</u>	<u>No. Successful Nests</u>	<u>No. Successful Nests / km</u>	<u>Apparent Success (%)</u>	<u>Mayfield Success (%)</u>
2001	Ravines					
	HORN	2	2	1.23	100	55.60*
	TOM	4	2	1.21	50.00	55.60*
	VAN	NA	NA	NA	NA	NA
	DUNN	10	6	3.85	60.00	52.20
	Benches					
	SPRU	10	7	5.38	70.00	68.50
	MARS	14	8	4.23	57.14	50.90
	<i>Ravine totals/mean</i>	<i>16</i>	<i>10</i>	<i>2.10</i>	<i>70.00</i>	<i>52.20</i>
	<i>Bench totals/mean</i>	<i>24</i>	<i>15</i>	<i>4.81</i>	<i>63.57</i>	<i>59.70</i>
	<i>Pooled totals/mean</i>	<i>40</i>	<i>25</i>	<i>3.18</i>	<i>67.43</i>	<i>56.56</i>
2002	Ravines					
	HORN	1	0	0.00	0.00	67.20*
	TOM	6	5	3.03	83.33	67.20*
	VAN	9	6	5.22	66.67	76.20
	DUNN	12	6	3.85	50.00	62.50
	Benches					
	SPRU	15	8	6.15	53.33	53.10
	MARS	14	5	2.65	35.71	34.30
	<i>Ravine totals/mean</i>	<i>28</i>	<i>17</i>	<i>3.03</i>	<i>50.00</i>	<i>69.35</i>
	<i>Bench totals/mean</i>	<i>29</i>	<i>13</i>	<i>4.40</i>	<i>44.52</i>	<i>43.70</i>
	<i>Pooled totals/mean</i>	<i>57</i>	<i>30</i>	<i>3.48</i>	<i>48.17</i>	<i>60.08</i>
Both yrs.	<i>Ravine totals/mean</i>	<i>44</i>	<i>27</i>	<i>2.63</i>	<i>58.57</i>	<i>62.74*</i>
	<i>Bench totals/mean</i>	<i>53</i>	<i>28</i>	<i>4.60</i>	<i>54.05</i>	<i>51.70</i>
	<i>Pooled totals/mean</i>	<i>97</i>	<i>55</i>	<i>3.35</i>	<i>56.93</i>	<i>57.83*</i>

*TOM/HORN data was combined in 2001/02 by Sheehan (2003).

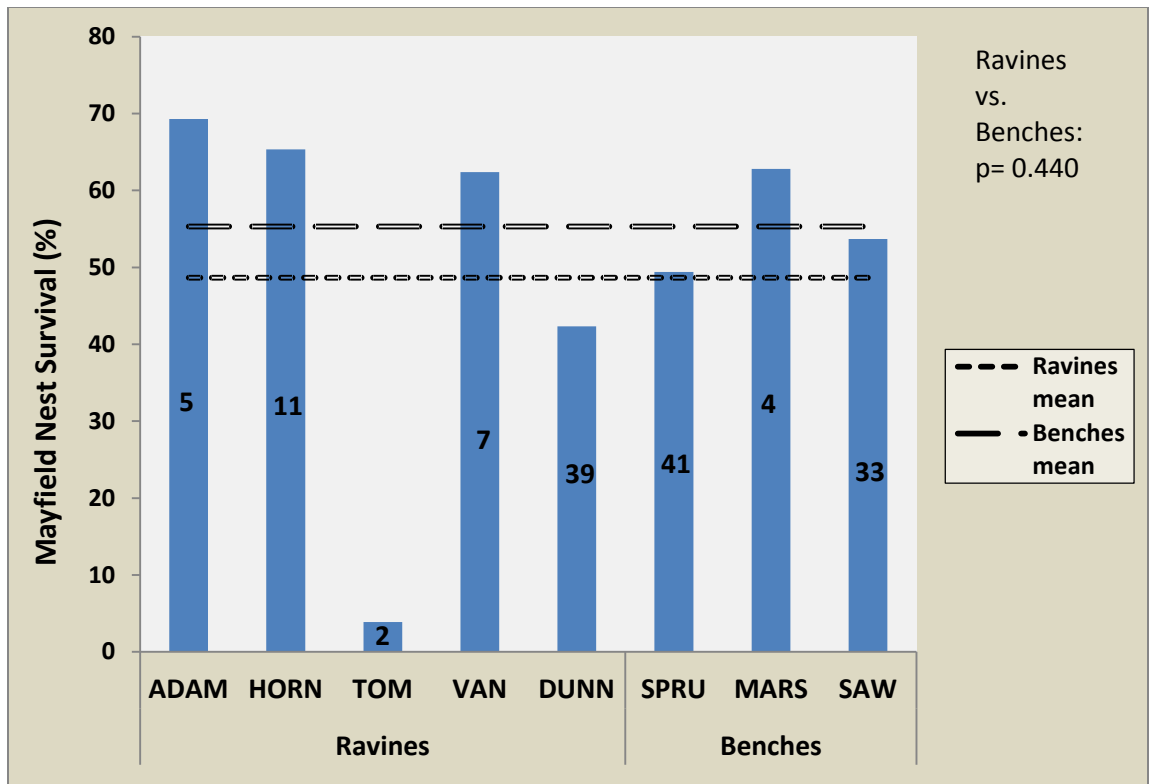


Figure 36. Mayfield survival rates (%) at each study site in 2013/14. Numbers in each bar represent the number of nests found at each site.

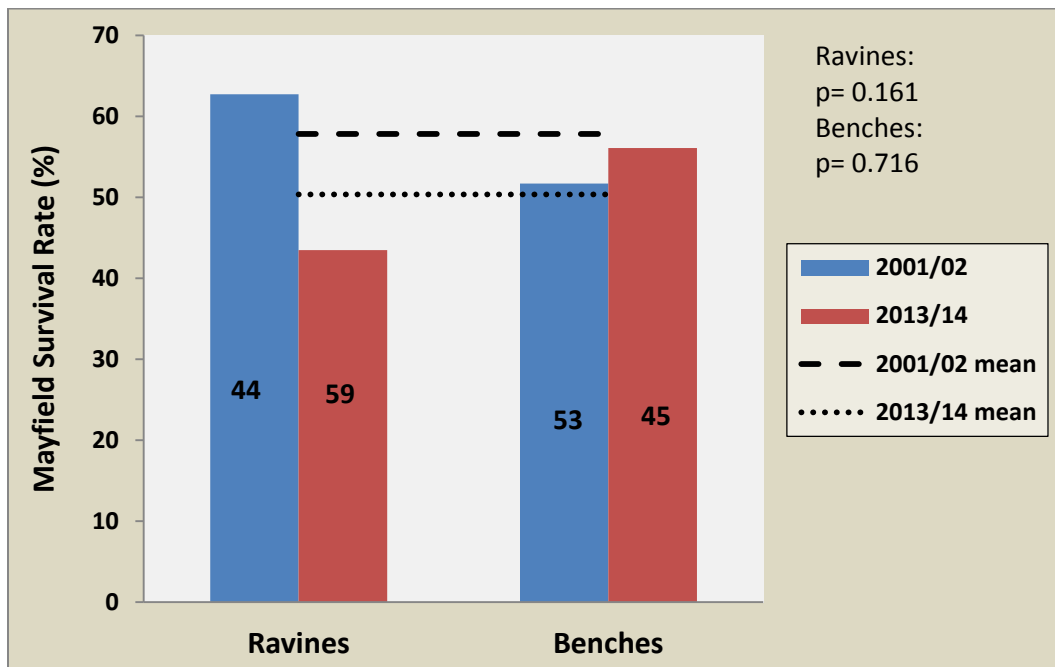


Figure 37. Comparison of Mayfield survival rates (%) in both site types between studies. Numbers in each bar represent the total nests compared at each site type.

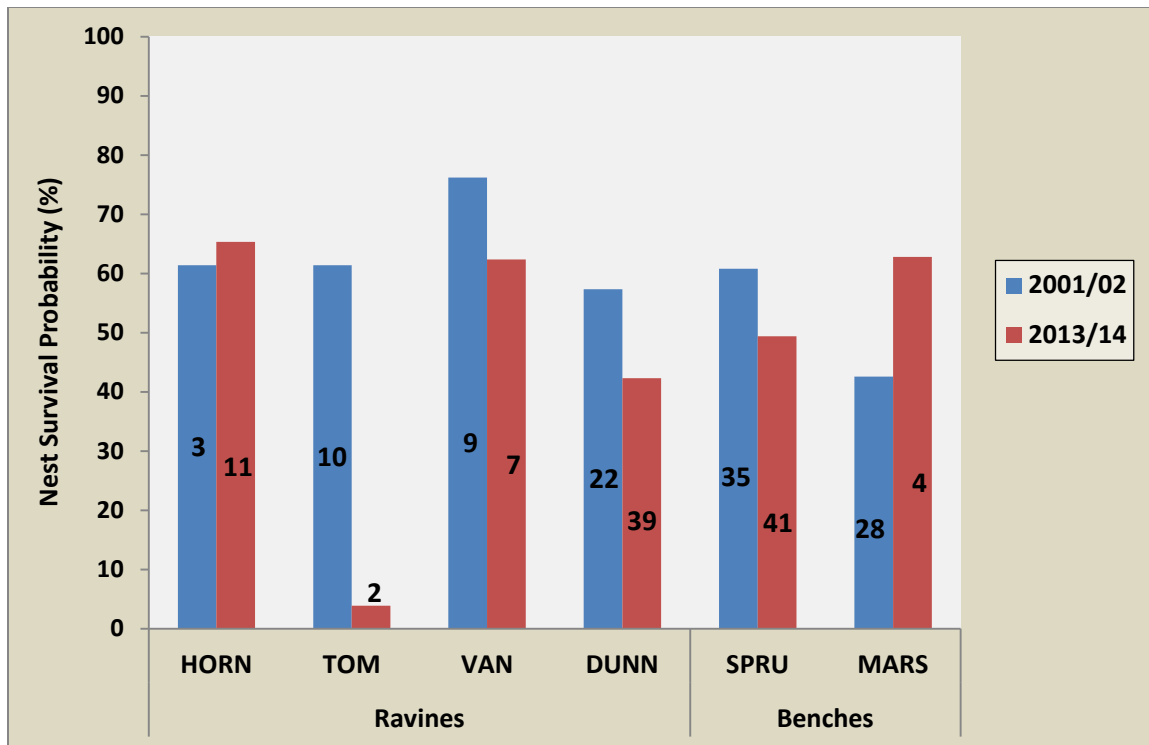


Figure 38. Comparison of Mayfield survival rates (%) at each site between studies. Numbers in each bar represent the total nests compared at each site.

Table 7. Acadian Flycatcher apparent and Mayfield nest survival at sites pooled across 2013 and 2014.

Site	^a Egg Stage			Nestling Stage			All Stages	
	# Failed Nests	Exp. Days	Mayfield Succ. (%)	# Failed Nests	Exp. Days	Mayfield Succ. (%)	Mayfield Succ. (%)	Apparent Succ. (%)
Ravine								
ADAM	1	52.5	73.51	1	61.5	79.49	43.46	60.00
HORN	2	123.5	77.01	3	159	76.59	42.95	54.50
TOM	1	11.5	23.33	1	13	32.61	7.77	0.00
VAN	2	66	61.12	1	48	74.47	26.09	57.10
DUNN	9	414.5	70.38	5	410	84.22	34.97*	53.80
Bench								
SPRU	10	585	75.89	6	600	86.87	44.45*	56.10
MARS	0	NA	NA	0	NA	NA	62.81*	75.00
SAW	4	474.5	87.33	6	522.5	85.07	54.85*	66.70
<i>Ravine totals/mean</i>	15	668	61.07	11	691.5	69.48	31.05	45.08
<i>Bench totals/mean</i>	14	1059.5	81.61	12	1122.5	85.97	54.04	65.93
<i>Pooled totals/mean</i>	29	1727.5	66.94	23	1814	74.19	39.67	52.90

^aEgg Stage: $100 \times (\text{daily survival rate})^{16}$; Nestling Stage: $100 \times (\text{daily survival rate})^{14}$; All stages: $100 \times (\text{daily survival rate})^{30}$.

*MARS and SAW had 1 failed nest due to abandonment and DUNN had 3. SPRU had 2 unknown type nest failures (Fig. 7).

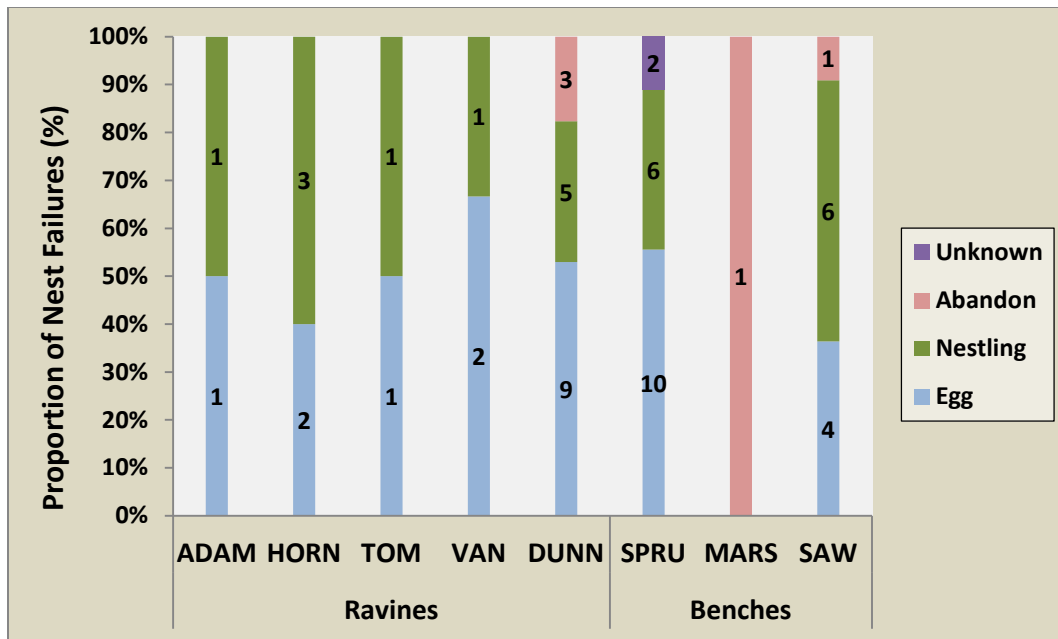


Figure 39. Comparison of nest failure type at each site in 2013/14.

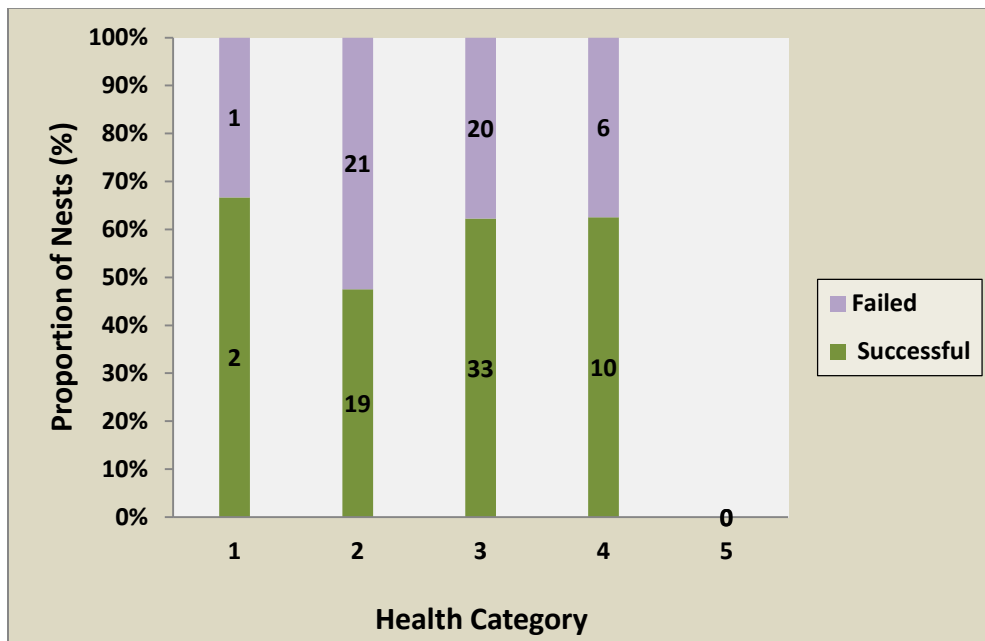


Figure 40. Proportion of EAHE placed nests in each EAHE health category. Numbers in bars represent total EAHE nests in each category.

Table 8. Proportion of Acadian flycatcher EAHE nests in each health category

	<u>Very Poor</u>		<u>Poor</u>		<u>Fair</u>		<u>Good</u>		<u>N</u>	
	Successful (%)	Failed (%)	Successful (%)	Failed (%)	Successful (%)	Failed (%)	Successful (%)	Failed (%)	Successful	Failed
Ravine										
ADAM	0.00	0.00	20.00	40.00	40.00	0.00	0.00	0.00	3	2
HORN	10.00	0.00	20.00	10.00	30.00	30.00	0.00	0.00	6	4
TOM	0.00	0.00	0.00	50.00	0.00	0.00	0.00	50.00	0	2
VAN	0.00	0.00	33.33	0.00	16.67	50.00	0.00	0.00	3	3
DUNN	0.00	5.56	27.78	16.67	16.67	27.78	5.56	0.00	9	9
Bench										
SPRU	2.70	0.00	8.11	21.62	32.43	16.22	10.81	8.11	20	17
MARS	0.00	0.00	0.00	0.00	50.00	0.00	25.00	25.00	3	1
SAW	0.00	0.00	19.35	19.35	32.26	12.90	12.90	3.23	20	11
<i>Ravine totals</i>	2.44	2.44	24.39	17.07	21.95	26.83	2.44	2.44	21	20
<i>Bench totals</i>	1.39	0.00	12.50	19.44	33.33	13.89	12.50	6.94	43	29
<i>Pooled totals</i>	1.77	0.88	16.81	18.58	29.20	18.58	8.85	5.31	64	49

Table 9a. Apparent nest survival (%) by nest tree species in 2013/14.

Nest Tree Species	No. Nests	Apparent Succ. (%)	Mean No. Fledglings
Eastern Hemlock	113	56.64	1.6
"Hardwood"	29	62.07	1.4
Birches	9	55.56	1.3
American Beech	8	75.00	1.9
Sugar Maple	5	60.00	1.2
Witch-hazel	5	40.00	1
Rhododendron	1	100.00	3
Black Gum	1	100.00	1

Table 9b. Apparent nest survival (%) by nest tree species in 2001/02.

Nest Tree Species	No. Nests	Apparent Succ. (%)
Eastern Hemlock	91	56.04
"Hardwood"	8	50.00
American Beech	3	33.33
Red Maple	1	0.00
Witch-hazel	1	0.00
Rhododendron	3	100.00

Table 10. Characteristics of successful and failed nests (Numerical data: mean $\pm$ S.D.; Categorical: % frequency)

Measurement Type	Measurement	Successful (n=82)	Failed (n=59)	Statistical Results
Vegetation Structure	Trees/plot	25.5 $\pm$ 10.2	24.5 $\pm$ 10.6	U= 2239.5 , z= -0.751, p=0.453
	Closest tree (m)	3.1 $\pm$ 1	2.8 $\pm$ 1.1	U= 1968, z= -1.838, p=0.066
	Saplings/plot	7.8 $\pm$ 10.5	7 $\pm$ 9.5	U= 2324 , z= -0.399, p=0.690
	Sapling cover (%)	22.2 $\pm$ 22.9	19.6 $\pm$ 22.6	U= 2195, z= -0.940, p=0.347
	Shrubs/plot	4.8 $\pm$ 9.7	6.7 $\pm$ 12.2	U= 2388 , z= -0.136, p=0.892
	Shrub cover (%)	19 $\pm$ 12.1	19.3 $\pm$ 12.8	U= 2390.5, z= -0.119, p=0.905
	Water cover (%)	10.2 $\pm$ 14.4	14.5 $\pm$ 15.8	U= 2095, z= -1.413, p=0.158
	Herb cover (%)	28 $\pm$ 17.1	25.3 $\pm$ 16.9	U= 2204.5, z= -0.897, p=0.370
	Bare cover (%)	42.8 $\pm$ 22.8	40.8 $\pm$ 19.4	U= 2392, z= -0.113, p=0.910
Nest Placement	FHD Vegetation "Hits" (%)	56.3 $\pm$ 9.5	53.9 $\pm$ 9.8	U= 2059, z= -1.505, p= 0.132
	Tip distance (cm)	70.9 $\pm$ 37.3	74.9 $\pm$ 43	U= 2297.5, z= -0.341, p=0.733
	Trunk distance (m)	3.8 $\pm$ 1.4	3.8 $\pm$ 1.6	U= 2068.5, z= -1.617, p=0.106
	Nest height (m)	7.7 $\pm$ 5.1	6.9 $\pm$ 4	U= 2336.5, z= -0.510, p=0.610
	Nest tree dbh (cm)	29.7 $\pm$ 16.3	29.9 $\pm$ 16.1	U= 2359, z= -0.251, p=0.802
	Nest tree height (m)	17.2 $\pm$ 8.4	17.6 $\pm$ 8.4	U= 2349.5, z= -0.457, p=0.648
	Nest distance to water (m)	16.5 $\pm$ 26.7	12.4 $\pm$ 14.7	U= 2217, z= -1.009, p=0.313
	Slope (median)	8	10	$\bar{X}$ = 0.260, df=1, p=0.610
	Aspect (median)	230	223	$\bar{X}$ =0.029, df=1, p=0.865
	Nest fork (most freq.)	2-twig, (38.3%)	Multiple, (45.3%)	$\bar{X}$ =1.851, df=4, p=0.763
Nest Concealment	Nest overhang (most freq.)	Clearing, (43.9%)	Clearing, (40.7%)	$\bar{X}$ =8.157, df=4, p=0.086
	Nest zone (most freq.)	Slope, (54.9%)	Slope, (50.8%)	$\bar{X}$ =1.419, df=2, p=0.492
	Canopy coverage above nest (%)	95.9 $\pm$ 4.4	96.2 $\pm$ 3.4	U= 2417.5, z= -0.006, p=0.995
	Canopy coverage/plot (%)	96 $\pm$ 5	94 $\pm$ 7.6	U= 2070.5 , z= -1.563, p=0.118
	Canopy height (m)	24.6 $\pm$ 11	23.1 $\pm$ 5.1	U= 2131.5, z= -1.206, p=0.228

*"Tip distance": n= 82, 57; "Nest fork": n= 81, 53; "Closest tree": n= 86, 56 (more than one closest tree in some plots)

Pair Productivity. – Mean fledgling density across both years at all sites was 8.0 fledglings/km (range: 0.0-22.6; Table 11). There was a significantly higher mean fledgling density on benches with 16.2 fledglings/km compared to 3.5 fledglings/km in the ravines ($U= 3.000$, $z= -2.697$, $p= 0.005$; Fig. 41). Even when TOM, which had 0 fledglings, was removed from analysis the mean fledgling density was still significantly different ($p= 0.011$). Within benches, SPRU had the highest fledgling density with 19.5 fledglings/km with SAW having a similar density of 18.8. The other bench site, MARS, was much lower with a mean fledgling density of 4.5. Within ravines, DUNN had the highest fledgling density with 9.3 fledglings/km. With no successful nests, TOM had the lowest fledgling density of 0.0. There was no significant difference in fledgling density between sites ($H= 13.224$, $df= 7$, $p= 0.067$).

Mean pair productivity across all sites for 2013/14 was 2.1 fledglings/pair (range: 0.0-3.6; Table 11). Pairs on benches produced a non-significantly higher mean number of fledglings with 2.7 fledglings/pair compared to 1.2 fledglings/pair in ravines ($U= 13.000$, $z= -1.471$, $p= 0.165$; Fig. 42). Within benches, SAW had the highest pair productivity with a mean of 3.1 fledglings/pair, and SPRU had the lowest with 2.2. Within ravines, VAN had the highest pair productivity with a mean of 2.7 fledglings/pair, while TOM had the lowest with 0.0. There was no significant difference in pair productivity between sites ($H= 7.313$, $df= 7$, $p= 0.397$).

Table 11. Pair productivity in 2013/14.

Site	<u>No. Fledglings</u>		<u>No. Fledglings/km</u>		<u>No. Fledglings/pair</u>	
	2013	2014	2013	2014	2013	2014
Ravines						
ADAM	2.8	5	1.69	3.01	2.80	1.67
HORN	10	6	5.99	3.59	3.33	1.50
TOM	0	0	0.00	0.00	0.00	0.00
VAN	3	1.79	7.00	4.17	3.00	2.33
DUNN	31.2	21.8	10.95	7.65	2.40	1.68
Benches						
SPRU	32.8	23.8	22.62	16.41	2.52	1.83
MARS	NA	8.6	NA	4.50	NA	2.87
SAW	25	36	15.43	22.22	2.50	3.60
<i>Ravine totals/mean</i>	<i>47</i>	<i>34.59</i>	<i>5.13</i>	<i>3.68</i>	<i>2.31</i>	<i>1.44</i>
<i>Bench totals/mean</i>	<i>57.8</i>	<i>68.4</i>	<i>19.03</i>	<i>14.38</i>	<i>2.51</i>	<i>2.77</i>
<i>Pooled totals/mean</i>	<i>104.8</i>	<i>102.99</i>	<i>9.10</i>	<i>7.69</i>	<i>2.36</i>	<i>1.94</i>

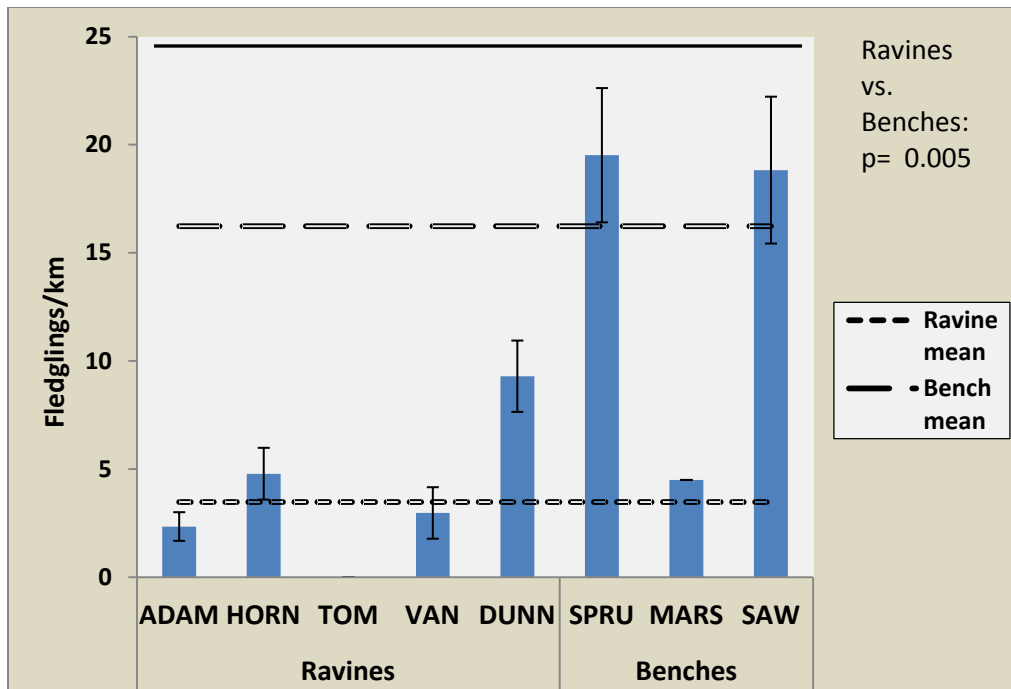


Figure 41. Fledgling density at each study site. Line at top of graph denotes a significant difference between site types ($p = 0.005$).

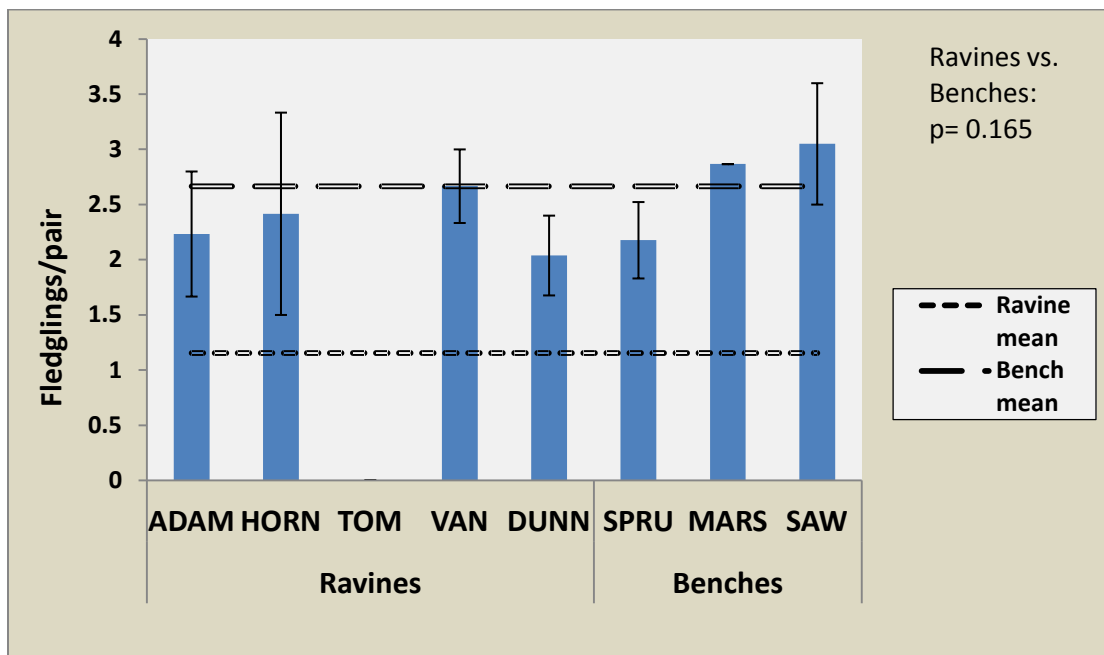


Figure 42. Pair productivity at each study site.

Nest Productivity. – In the current study, pooled across all sites, there was an average of 1.6 nests/pair (range: 1-3 nests/pair) across study sites over both years including renesting and double brooding attempts (Table 12a). Nests/pair were similar between sites types with ravines having a mean of 1.5 nests/pair and benches having 1.6 nests/pair. Across all sites, renesting attempts occurred in 26 out of 41 first failed nests (63.4%). Of these 26 renests, 11 failed again (42.3%). A third nesting attempt occurred three times, with one nest failing a third time. Fourth renesting attempts were not observed. Ravine sites had a slightly higher incidence of first nest failure (n= 22) than the benches (n= 19). ACFL at ravine sites also had a slightly higher but non-significant occurrence of attempting to renest after a first failed nest (n= 14) compared to the benches (n= 12; FET, $p=0.614$).

Across all sites, double-brooding attempts occurred 21 times (42%; Table 12b). Of these 21 nests, 15 were successful (71.4%). Triple-brooding was not observed in this study. Benches had a non-significant higher occurrence of double-brooding with 14 attempts (46.7% of 30 first successful nests), compared to ravines with 7 attempts (35% of 20 first successful nests, FET, $p=0.300$).

Nest Placement

Nest Placement Variables by Site Type. – Mean nest placement variables, including nest distance to tip of branch, nest height, nest distance to water, and canopy coverage of plot, were significantly different between site types (Table 13). In the ravines, the mean distance from the nest to the tip of the branch was significantly shorter (66.6 cm) than on the benches (76.9 cm) ($U= 1908$, $z= -2.198$, $p= 0.028$). Nests on the benches were significantly higher than at the ravines with a mean height of 7.8 m vs. 6.8 m , respectively ($U=1769.5$, $z= -2.979$, $p= 0.003$). Nests on the benches were found to be significantly closer to water than nests in the ravines with a mean distance of 14 m vs.

15.8 m, respectively ($U = 1732$, $z = -3.148$, $p = 0.002$). The mean canopy cover on benches was significantly greater than at ravines with 96.2% vs. 93.7%, respectively ($U = 1742$, $z = -3.313$, $p = 0.001$). Nest placement variables including nest distance to trunk, nest tree dbh, nest tree height, canopy coverage above nest, and canopy height did not differ significantly between site types. Nests on benches were found most often overhanging a clearing (52.6%), while nests in the ravines were found most often overhanging a slope (46.9%). There were significantly more nests on the benches that were placed over the main stream channel compared to the ravines, with 35.9% vs. 7.8%, respectively (adj. $p < 0.005$), and significantly more nests placed over a slope in the ravines than on the benches with 46.9% vs. 3.9%, respectively (adj. $p < 0.005$). Similarly, nests found on benches were significantly more often placed in the stream zone (39.7%) than nests in the ravines (7.8%) (adj. $p < 0.008$), and nests in the ravines were significantly more often found in the slope zone (70.3%) than nests on the benches (38.5%) (adj. $p < 0.008$). In the ravines, nests were most often found in a 2-twig nest fork (43.5%), while on the benches, nests were most often found in a multiple nest fork. No significant difference in nest fork type was found between site types.

Among sites, significant differences in nest distance to water and canopy coverage of plot were seen between SPRU and DUNN. Nests at SPRU were found significantly closer to water than at DUNN with 5.4 m vs. 13.6 m, respectively ($H = 11.257$, $p = 0.022$). There was significantly higher mean canopy coverage/plot at SPRU than DUNN with 97.2% vs. 93.1%, respectively ($H = 29.194$, $p = 0.019$). Nests at DUNN were found significantly less than other sites in the stream zone (7.7%) (adj. $p < 0.002$), and nests at SPRU were found significantly more in the stream zone (48.8%) and significantly less in the slope zone (24.4%) (adj. $p < 0.002$) than other sites.

Table 12a. Renesting productivity for 2013/14 data across study sites.

Site	No. Pairs	No. Nests	No. Nests/pair	1st nest fail	Renested after 1st fail	2nd nest fail	Renested after 2nd fail	3rd nest fail
Ravine								
ADAM	4	5	1.25	2	1	0	0	NA
HORN	7	11	1.57	3	3	2	0	NA
TOM	1	2	2.00	1	1	1	0	NA
VAN	4	7	1.75	2	2	0	0	NA
DUNN	26	39	1.50	14	7	3	1	0
Bench								
SPRU	26	41	1.58	11	7	4	1	1
SAW	20	33	1.65	7	4	1	1	0
MARS	3	4	1.33	1	1	0	0	NA
<i>Ravine totals/mean</i>	<i>42</i>	<i>64</i>	<i>1.52</i>	<i>22</i>	<i>14</i>	<i>6</i>	<i>1</i>	<i>0</i>
<i>Bench totals/mean</i>	<i>49</i>	<i>78</i>	<i>1.59</i>	<i>19</i>	<i>12</i>	<i>5</i>	<i>2</i>	<i>1</i>
<i>Pooled totals/mean</i>	<i>91</i>	<i>142</i>	<i>1.56</i>	<i>41</i>	<i>26</i>	<i>11</i>	<i>3</i>	<i>1</i>

Table 12b. Double-brooding productivity for 2013/14 data across study sites.

Site	No. Pairs	No. Nests	No. Nests/pair	1st nest succeeded	Double- bred	2nd nest succeeded
Ravine						
ADAM	4	5	1.25	2	0	NA
HORN	7	11	1.57	4	1	1
TOM	1	2	2.00	0	0	NA
VAN	4	7	1.75	2	1	0
DUNN	26	39	1.50	12	5	4
Bench						
SPRU	26	41	1.58	15	6	5
SAW	20	33	1.65	13	8	5
MARS	3	4	1.33	2	0	NA
<i>Ravine totals/mean</i>	<i>42</i>	<i>64</i>	<i>1.52</i>	<i>20</i>	<i>7</i>	<i>5</i>
<i>Bench totals/mean</i>	<i>49</i>	<i>78</i>	<i>1.59</i>	<i>30</i>	<i>14</i>	<i>10</i>
<i>Pooled totals/mean</i>	<i>91</i>	<i>142</i>	<i>1.56</i>	<i>50</i>	<i>21</i>	<i>15</i>

Nest Placement Variables by Nest Tree. – In this study, there were 113 nests placed in eastern hemlock and 29 nests placed in seven different hardwood tree species (Table 14a-b). With low sample sizes for each individual hardwood nest tree species, all hardwood tree species were lumped together to form one category of tree type for analysis. Mean nest placement variables that were found to be significantly different between tree types included nest tree height, nest tree dbh, and canopy cover/plot. Eastern hemlock nest trees were significantly taller than hardwood nest trees with a mean height of 18.3 m vs 13.9 m, respectively ($U = 1173$, $z = -2.358$, $p = 0.018$). Eastern hemlock nest trees had a significantly larger dbh than hardwood nest trees with a mean dbh of 33.3 cm vs. 16.5 cm, respectively ($U = 536.5$, $z = -5.47$, $p < 0.001$). Plots that had an eastern hemlock tree nest had significantly more canopy coverage with a mean cover of 95.4% vs. 93.4%, respectively ($U = 1257$, $z = -2.069$, $p = 0.039$). No significant differences in terrain zone and nest fork type were observed between eastern hemlock and hardwood nest trees ($\chi^2 = 5.649$, $df = 2$, $p = 0.059$; $\chi^2 = 7.25$, $df = 4$, $p = 0.123$). There was a significant difference between tree types in nest overhang ($\chi^2 = 10.956$, $df = 4$, $p = 0.027$) with hardwood nests found overhanging trails and slopes more often than eastern hemlock nests (13.8% vs. 3.5%; 34.5% vs 20.4%, respectively) and eastern hemlock nests were found overhanging streams more often than hardwood nests (26.5% vs. 10.3%, respectively).

Among comparable sites, there were 77 eastern hemlock placed nests in the current study and 91 EAHE nests in 2001/02. In the current study, the mean nest height for EAHE nests was 6.9 m which was significantly higher than the mean EAHE nest height of 5.8 m from 2001/02 ($Z = -3.907$, $p < 0.001$; Fig 43). There was a significant higher EAHE nest height of 7 m in the ravines in 2013/14 compared to 5.7 m in 2001/02 ($Z = -4.190$, $p < 0.001$; Fig. 43). On benches, EAHE nests in 2013/14 were not significantly higher than in 2001/02 with mean heights of 6.8 m vs. 5.9 m, respectively ($Z = -0.433$, $p = 0.665$). Between studies, DUNN was the only site to have a significantly different higher EAHE mean nest height of 8.3 m in 2013/14 vs. 6.3 m in 2001/02 ($Z = -3.724$, $p < 0.001$; Fig. 44). All other sites, with the exception of MARS, had non-significant higher mean EAHE nest

heights in 2013/14 compared to 2001/02. This pattern of significantly higher mean EAHE nest heights in 2013/14 reflects the significantly higher mean heights of 1st live branches of EAHE from the HWA survey (Fig. 45).

Table 13. Mean nest placement variables by site type.

Measurement Type	Measurement	<u>Ravine</u> (n=64)	<u>Bench</u> (n=78)	Statistical Results
Nest Placement	Tip distance (cm)	66.6 ± 43.6	76.9 ± 35.6	U= 1908, z= -2.198, p=0.028
	Trunk distance (m)	3.9 ± 1.7	3.7 ± 1.2	U= 2399, z= -0.398, p=0.691
	Nest height (m)	6.8 ± 5.4	7.8 ± 3.9	U= 1769.5, z= -2.979, p= 0.003
	Nest tree dbh (cm)	27.7 ± 16.2	31.7 ± 16	U= 2063, z= -1.661, p= 0.097
	Nest tree height (m)	17.8 ± 9.9	17.1 ± 6.9	U= 2453, z= -0.174, p= 0.862
	Nest distance to water (m)	15.8 ± 13.9	14 ± 27.7	U= 1732, z= -3.148, p= 0.002
	Nest Fork (most freq.)	2-twig, 43.5%	Multiple, 43.8%	$\chi^2 = 4.798$, df=4, p= 0.309
	Nest Overhang (most freq.)	Slope, 46.9%	Clearing, 52.6%	$\chi^2 = 50.296$, df=4, p<0.001
	Terrain Zone (most freq.)	Slope, 70.3%	Stream, 39.7%	$\chi^2 = 20.891$, df=2, p<0.001
Nest Concealment	Canopy coverage above nest (%)	95 ± 4.9	96.8 ± 3	U= 2105.5, z= -1.604, p= 0.109
	Canopy coverage/plot (%)	93.7 ± 5.8	96.2 ± 6.9	U= 1742, z= -3.313, p=0.001
	Canopy height (m)	26 ± 12.4	22.4 ± 4.1	U= 1888, z= -2.502, p= 0.12

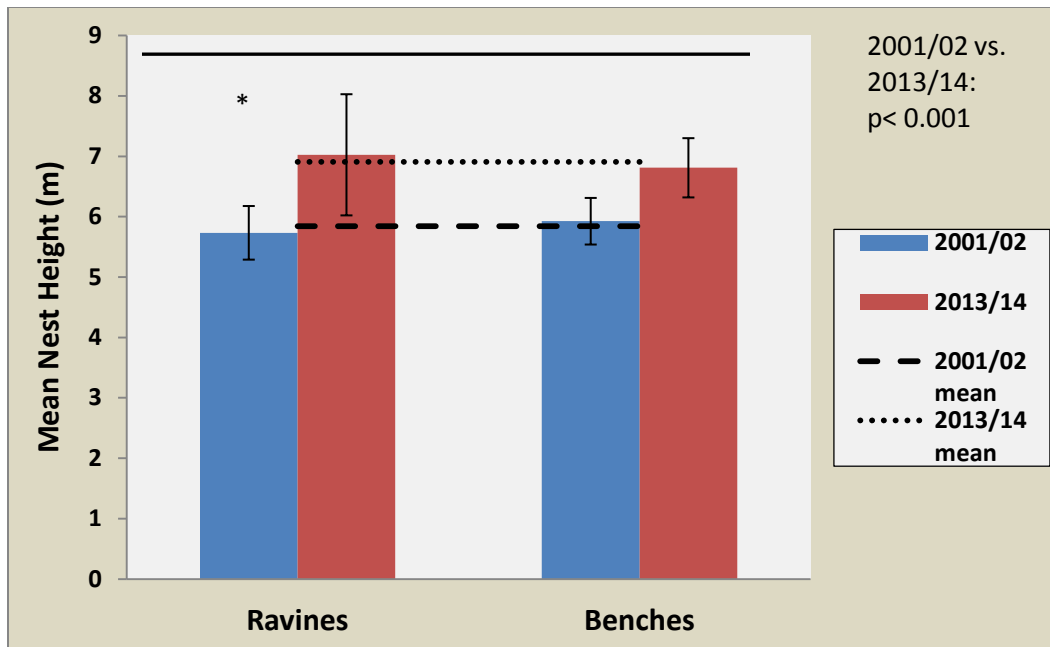


Figure 43. Comparison of mean EAHE nest height (m) between studies at each site type. Line at top of graph denotes a significant difference in overall mean nest height between studies ($p < 0.001$). Asterisk denotes a significant difference in mean nest height within the ravines between studies ($p < 0.001$).

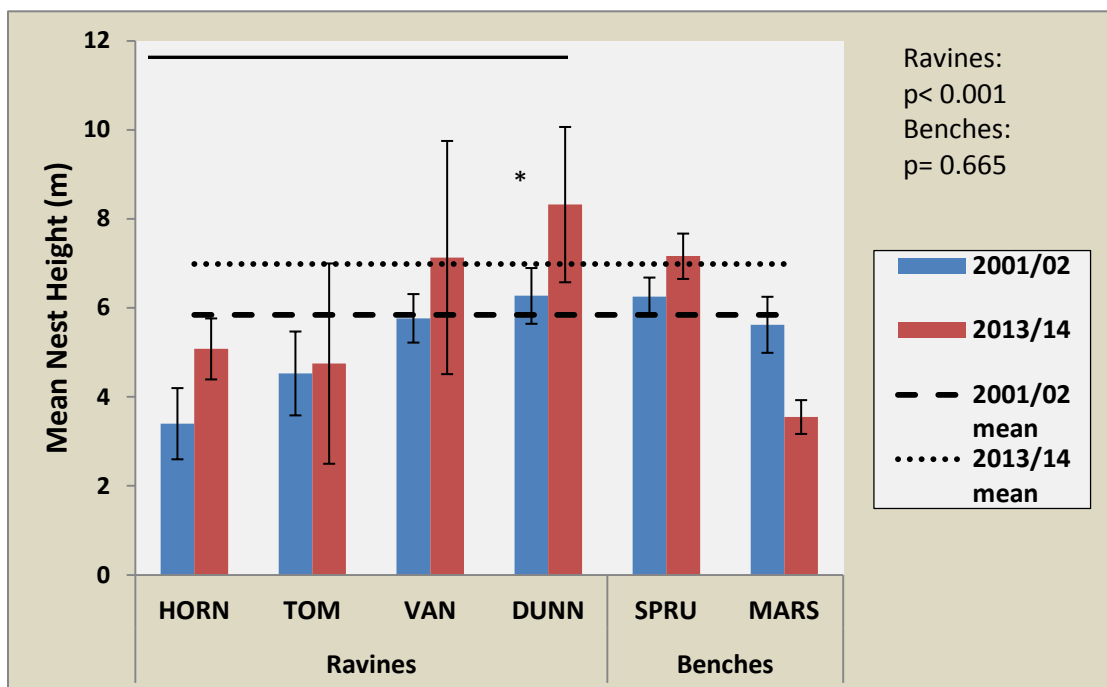


Figure 44. Comparison of mean EAHE nest height between studies at each site. Line at top of graph denotes a significant difference within ravines between studies ($p < 0.001$). Asterisk denotes a significant difference at DUNN between studies ($p < 0.001$).

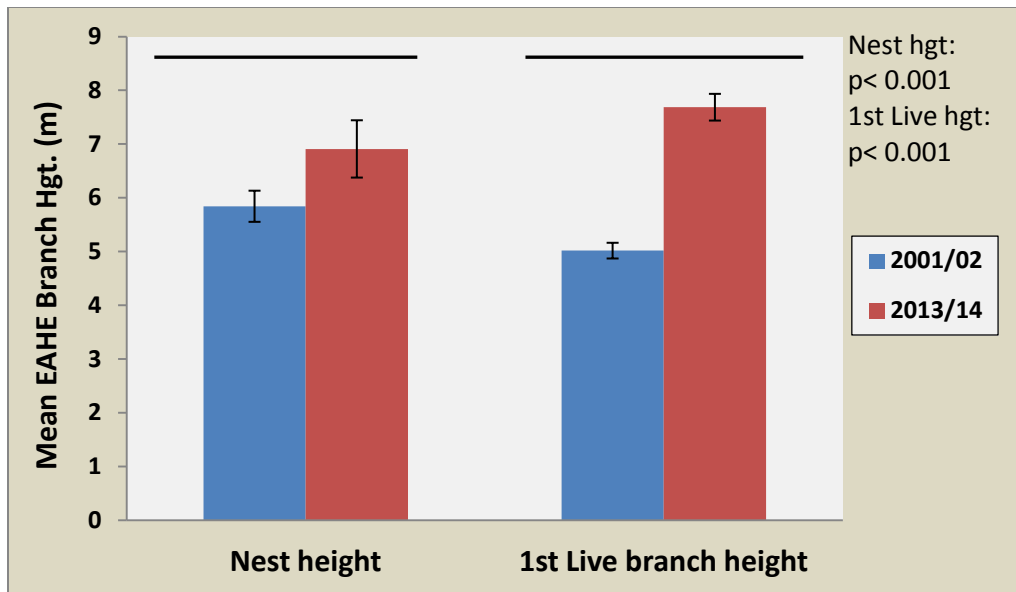


Figure 45. Comparison of mean EAHE nest height (m) and mean height of 1st live branch (m) between studies. Lines at top of graph denote significant differences in mean height of EAHE nests and 1st live branches between studies ($p < 0.001$; $p < 0.001$).

Table 14a-b. Nest placement variables in each nest tree species.

A

N	Nesting Tree	Nest Ht. (m)	Tree ht. (m)	Tree dbh (cm)	Nest distance to tip (cm)	Nest distance to bole (m)	Nest distance to water (m)	Canopy cover above nest (%)	Canopy cover of plot (%)	Canopy ht. (m)
113	Eastern Hemlock	7.4 ± 4.6	18.3 ± 8.2	33.3 ± 15.7	68.7 ± 35.2	3.8 ± 1.2	15 ± 24.4	96 ± 3.8	95.4 ± 6.6	24.2 ± 10
9	Birches*	11.2 ± 5	20.1 ± 5.4	22.6 ± 9.4	78.5 ± 43.3	4.5 ± 2.7	13.3 ± 16.7	92.7 ± 6.7	90 ± 6.1	22.8 ± 4.2
8	American Beech	3.5 ± 1.2	9.4 ± 5.6	10.9 ± 4.9	135 ± 58.6	2.7 ± 1.5	8.7 ± 3.6	98.1 ± 2.8	94.4 ± 5	23.8 ± 3.5
5	Sugar Maple	10.3 ± 3.4	20.6 ± 5.5	25.9 ± 7.2	74 ± 15.2	4.5 ± 3.3	19.7 ± 16.5	98.6 ± 1	98 ± 2.7	22.8 ± 3.7
5	Witch-hazel	4.3 ± 1.3	5.5 ± 0.6	6.1 ± 2.7	62 ± 25.9	3.2 ± 1.4	16.3 ± 12.3	96.8 ± 4.5	92 ± 7.6	22.8 ± 2.8
1	Rhododendron	2.8 ± 0	4 ± 0	10 ± 0	10 ± 0	3.5 ± 0	13.7 ± 0	96.4 ± 0	95 ± 0	20 ± 0
1	Black Gum	10.5 ± 0	12.8 ± 0	17.4 ± 0	20 ± 0	4.7 ± 0	28 ± 0	94.3 ± 0	100 ± 0	27 ± 0

* "YEBI" n=8, "BLBI" n=1

B

N	Nesting Tree	Most freq. tree zone	Most freq. nest fork	Most freq. nest overhang
113	Eastern Hemlock	Slope (47.8%)	Multiple (41.6 %)	Clearing (48%)
9	Birches	Slope (66.7%)	2-twig (66.7%)	Slope (44.4%)
8	American Beech	Slope (87.5%)	2-twig (62.5%)	Clearing (50%)
5	Sugar Maple	Floodplain/Slope (each 40%)	2-twig/multiple (each 40%)	Clearing (80%)
5	Witch-hazel	Slope (80%)	Multiple (40 %)	Clearing/slope (40%)
1	Rhododendron	Slope (100%)	2-twig (100%)	Slope (100%)
1	Black Gum	Slope (100%)	3-twig slant (100%)	Slope (100%)

Nest Tree Preference. – In the current study, 113 nests of out 142 (79.6%) were placed in eastern hemlock trees and 29 (20.4%) were placed in seven species of hardwood trees (Fig. 46a-n). Hardwood nest tree species included nine yellow birch (YEBl; 6.3%), eight American beech (AMBE; 5.6%), five sugar maple and witch-hazel (SUMA and WIHA; 3.5% each), and one black gum, black birch, and rhododendron (BLGU, BLBl, and RHODO; 0.7% each). Ravines had 23 hardwood nests out of the 29 (79.3%), with DUNN having the majority of these nests (21 out of the 29; 72.4%). VAN had one nest in AMBE and HORN had one in YEBl. ADAM and TOM had 100% of their nests in EAHE. Benches had six of the 29 nests (20.7%) in hardwood trees, with SPRU having two in SUMA, one in WIHA, and one in YEBl. SAW had one nest in YEBl and one in WIHA. MARS had 100% of nests in EAHE.

When considering nest tree species usage in relation to habitat availability, each site showed a preference for nesting in eastern hemlock trees compared to nesting in hardwoods; observed nests in EAHE trees exceeded the expected number for each site based on availability (Table 15a). Goodness-of-fit Chi-square indicates a significant preference for nesting in EAHE and avoidance of hardwoods in relation to habitat availability. ADAM, HORN, DUNN, SPRU, and SAW showed a significant trend for nesting in EAHE out of proportion to availability. TOM, VAN, and MARS all had more observed nests than expected in EAHE, however, the trend was not significant. HORN, SPRU, and SAW had significantly fewer nests than expected in hardwood trees in relation to habitat availability. ADAM, TOM, VAN, DUNN and MARS all had fewer observed nests in hardwood trees than expected, however the trend was not significant. At DUNN, because there was a larger hardwood nest sample size, individual hardwood species preference can be considered (Table 15b). Eastern hemlock was selected out of proportion to availability, thus, ACFL showed a significant preference for nesting in EAHE. American beech was the only other tree species that was significantly selected out of proportion to availability. Birches, witch-hazel, and rhododendron were all selected less than expected for nesting in relation to availability, though the trend was

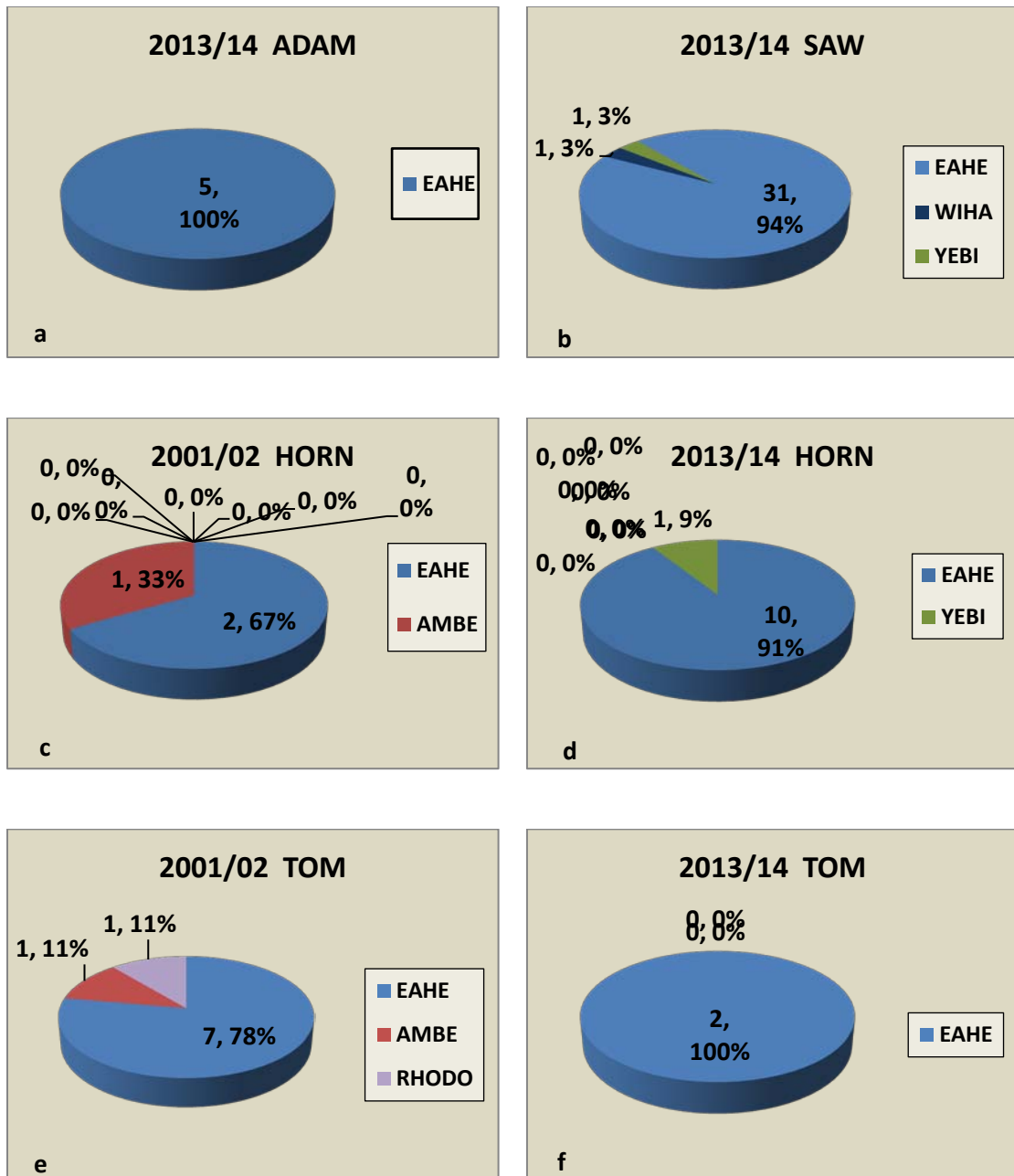


Figure 46. a-n. Comparison of nest tree species composition between studies at each site. Asterisk denotes a significant difference in nest tree use at DUNN between studies ($p < 0.001$).

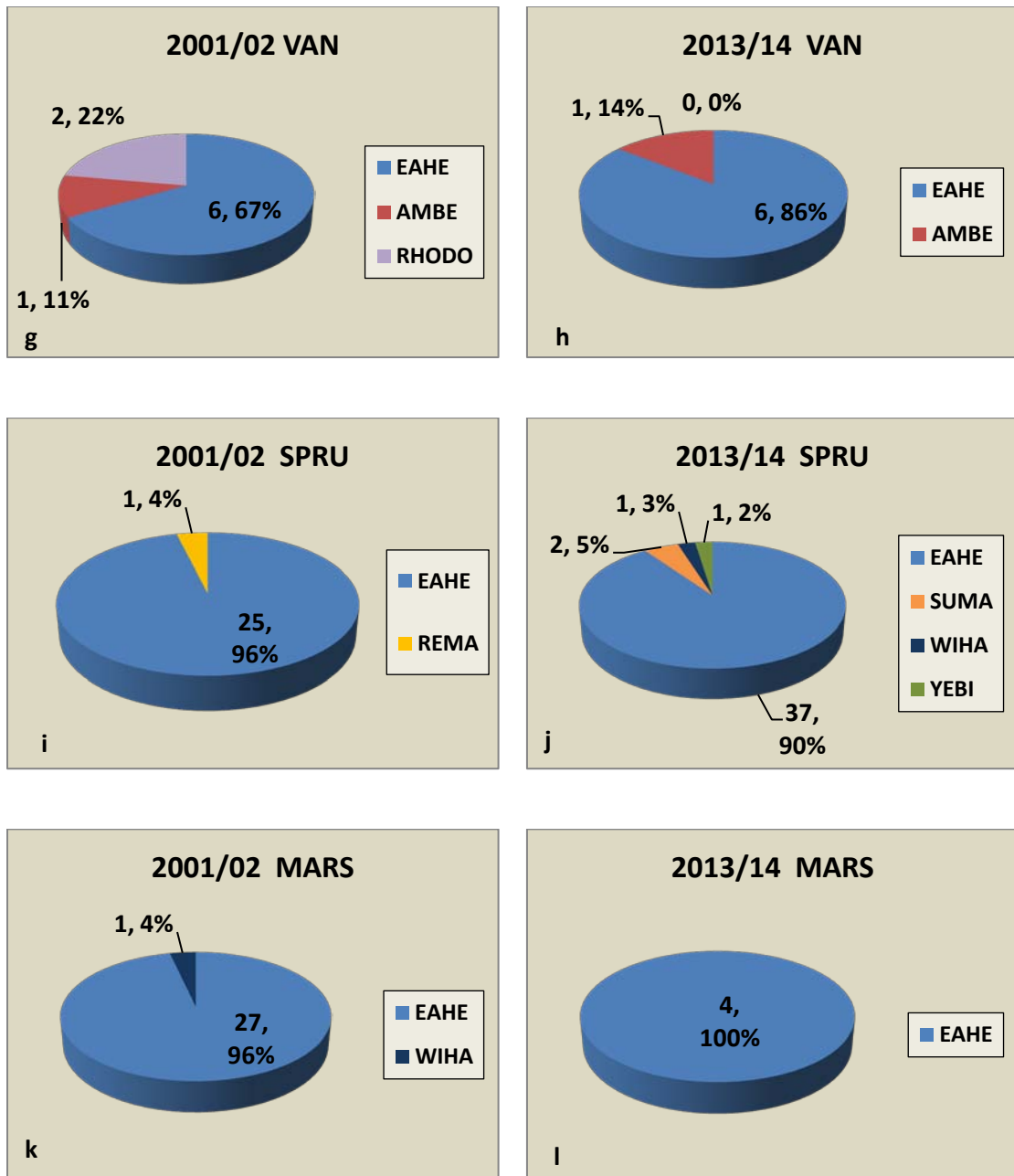


Figure 46. a-n cont'd. Comparison of nest tree species composition between studies at each site. Asterisk denotes a significant difference in nest tree use at DUNN between studies ($p < 0.001$).

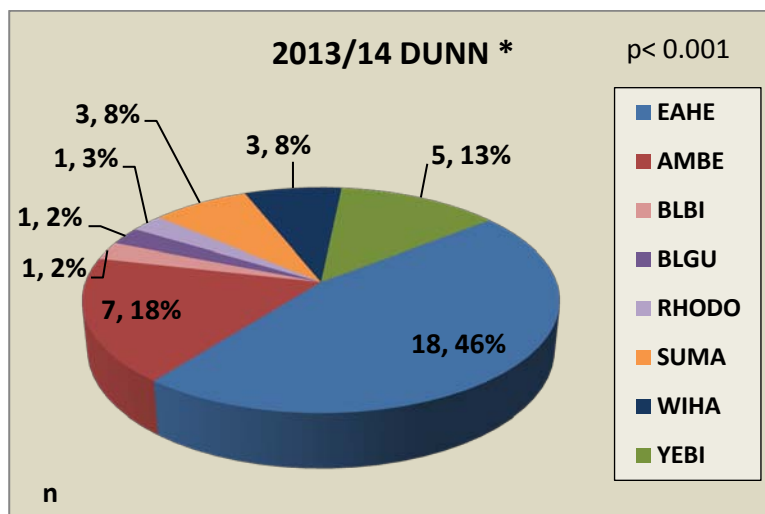
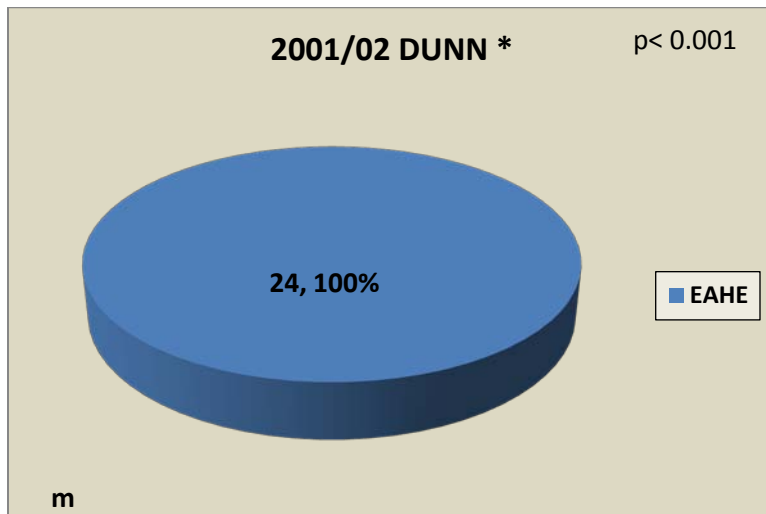


Figure 46. a-n cont'd. Comparison of nest tree species composition between studies at each site. Asterisk denotes a significant difference in nest tree use at DUNN between studies ($p < 0.001$).

not significant. Sugar maple and black gum were selected more than expected for nesting in relation to availability, though the trend was not significant. The tree species selected for nesting at DUNN account for ~73% of all tree species observed at this site. Two other tree species, tulip poplar and red maple, had higher habitat availability (6.9% and 3.9%, respectively) than some tree species selected for nesting, yet ACFL did not nest in these species.

Nest tree preference between studies (no SAW or ADAM) was significantly different (FET, $p = 0.011$). In the current study, there was a significantly higher rate of nesting in hardwood tree species than in 2001/02 (26.0% vs. 8.2%, respectively) and a significantly lower rate of nesting in eastern hemlock trees (74% vs. 91.8%, respectively). Within ravines, there was a significantly higher rate of nesting in hardwood trees than in 2001/02 (22.1% vs. 13.3%, respectively) and a significantly lower rate of nesting in EAHE trees (34.6% vs. 86.7%, respectively) (FET, $p = 0.007$). On benches, there was no significant difference in nest tree preference (FET, $p = 0.471$). DUNN was the only site to have significant differences in nest tree species use with a significantly higher rate of nesting in hardwood trees than in 2001/02 (53.8% vs. 0.0%, respectively) and a significantly lower rate of nesting in EAHE trees (46.2% vs. 100%, respectively) (FET, $p < 0.001$).

Nest Reuse. -Same season nest reuse was observed once in 2013 at SAW when a female, whose first nest was successful, successfully double brooded using the same nest. Same season nest tree reuse (EAHE) was observed once in 2013 at SPRU by one female, whose first nest being successful, double-brooded in the same tree on a slightly higher branch directly above her first nest (this second nest failed). In 2014, one female was observed twice using a previously built nest at DUNN, with the first nest failing and the second succeeding. Interestingly, these two nests were used by one female in the 2013 field season as well, though it is not known if the same female was involved between seasons. Between seasons, nest tree reuse (EAHE) was observed twice, once at DUNN and once at ADAM, with the new nests built on slightly higher branches. On many

occasions, nests in the second field season were observed being built quite close to old successful nests from first field season. Same season use of different tree species by pairs renesting or double brooding was observed 16 times; once at SAW, once at HORN, once at VAN, three instances at SPRU, and ten times at DUNN. Nests that switched from hardwood to EAHE were observed six times (5/6 had successful hardwood nests and attempted to double brood in EAHE). Switching from EAHE to hardwoods occurred six times (5/6 times EAHE nests failed and renest was attempted in a hardwood tree). Switching between hardwood species occurred four times (3/4 nests failed initially).

Habitat Selection

Vegetative Structure. – A total of 114 plots in the ravines and 109 plots on benches were surveyed to characterize vegetation composition and structure. When comparing vegetation structural differences between these site types, data from both nest and random plots was used to provide a more thorough and detailed assessment of vegetation structure (Table 16). Ravines and benches differed significantly in all structural vegetation categories except herbaceous cover (%), bare ground cover (%), and foliage height diversity “hits” (%). Benches had significantly higher tree densities and shorter closest tree distances per plot than the ravines. There was a significantly higher proportion of water cover in plots on the benches than in ravines. Benches had significantly more canopy cover over each plot than ravines. Ravines had significantly higher sapling densities, sapling cover, shrub densities, and shrub cover per plot than the benches. Mean canopy height was significantly higher at the ravines than the benches.

Among plots surveyed, 142 were nest (use) plots and 81 were random (non-use) plots. Vegetation structure variables in nest and random plots were compared overall, between site types and at each site (Table 17). Overall, when data was pooled across

A Table 15a-b. Nest tree species usage in relation to habitat availability.

Site	Availability	Eastern Hemlock			Hardwood Species			(O-E) ² /E	χ^2
		Observed	Expected	(O-E) ² /E	Availability	Observed	Expected		
Ravines									
ADAM	0.41	5	2.05	4.268	0.59	0	2.95	2.955	7.222
HORN	0.38	10	4.18	8.094	0.62	1	6.82	4.965	13.059
TOM	0.32	2	0.64	2.870	0.68	0	1.36	1.358	4.227
VAN	0.45	6	3.13	2.621	0.55	1	3.87	2.125	4.745
DUNN	0.18	18	6.94	17.616	0.82	21	32.06	3.814	21.430
Benches									
SPRU	0.33	37	13.67	39.837	0.67	4	27.33	19.919	59.756
MARS	0.44	4	1.75	2.909	0.56	0	2.25	2.254	5.163
SAW	0.45	31	14.72	18.022	0.55	2	18.28	14.504	32.525
Ravine totals	0.33	41	21.31	18.180	0.67	23	42.69	9.078	27.258
Bench totals	0.39	72	30.69	55.617	0.61	6	47.31	36.073	91.690
Pooled totals	0.37	113	52.42	70.011	0.63	29	89.58	40.968	110.980

"Habitat Availability"= proportion of tree species to the total species stem density from all plots at site; "O" = Observed nests at site;

"E"= Expected nests (Total nests observed at a site * Habitat Availability). Critical Chi-square Value= 3.841, df=1, p= 0.05. Bold values: p≤ 0.05.

B. Dunnfield Creek nest tree species use

Nest Tree Spp.	Habitat Availability	Observed	Expected	$(O-E)^2/E$
Eastern Hemlock	0.18	18	6.94	17.616
Birches	0.27	6	10.39	1.856
American Beech	0.05	7	2.10	11.397
Sugar Maple	0.03	3	1.35	2.031
Witch-hazel	0.14	3	5.64	1.234
Rhododendron	0.04	1	1.51	0.175
Black Gum	0.02	1	0.76	0.078
$\sum \chi^2$				34.387

sites there were significant differences in vegetation structure between nest and random plots in canopy height, proportion of foliage height diversity “hits”, and proportion of sapling, shrub, water and bare cover per plot. Nest plots had a non-significant lower tree density than random plots. Mean canopy cover and cover over the plot center was greater in nest vs. random plots, though the result was not significant. Nest plots had a significantly higher mean canopy height than random plots ($U = 3816.5$, $z = -4.194$, $p < 0.001$). Nest plots had a significantly higher proportion of total FHD “hits” than random plots ($U = 2312.5$, $z = -7.423$, $p < 0.001$). Nest plots had significantly lower proportions of sapling and shrub cover ($U = 4592, 3646$; $z = -2.506, -4.545$; $p = 0.012$; $p < 0.001$). Likewise, mean sapling and shrub densities were lower in nest vs. random plots, though the result was non-significant. Nest plots had significantly higher proportions of water cover and bare ground ($U = 4129.5, 4469$; $z = -3.768, -2.767$; $p < 0.001$, $p = 0.006$) than random plots. Mean proportion of herbaceous cover was lower in nest vs. random plots, though the result was not significant.

Within the ravines, nest plots had a significantly lower mean tree density than random plots ($U = 997.5$, $z = -3.547$, $p < 0.001$; Fig. 47). There was a non-significant lower mean canopy coverage per plot in nest plots compared to random plots (Fig. 48). Mean canopy height was significantly higher in nest plots compared to random plots ($U = 1041.5$, $z = -3.211$, $p < 0.001$). Mean proportion of FHD “hits” was significantly higher in nest vs. random plots ($U = 618.5$, $z = -4.805$, $p < 0.001$). Mean proportion of shrub cover was significantly lower in nest vs. random plots ($U = 1180$, $z = -2.397$, $p = 0.017$). Mean sapling cover was lower in nest vs. random plots, but the difference was not significant. Mean sapling and shrub densities were both lower in nest plots compared to random plots, though the result was not significant (Figs. 49 and 50). Other mean cover proportions including water, herbaceous, and bare ground cover were all higher in nest vs. random plots, though the result was not significant.

Within the benches, there was a lower mean tree density in nest plots compared to random plots (similar to ravines), though the result was not significant (Fig. 47). Mean

proportion of canopy cover per plot was not significantly different in nest vs. random plots (Fig. 48). There was a significantly higher mean canopy height in the nest plots vs. random plots ($U = 688$, $z = -3.513$, $p < 0.001$). Mean proportion of FHD “hits” was significantly higher in nest vs. random plots ($U = 494$, $z = -4.805$, $p < 0.001$). Mean proportions of sapling and shrub cover were significantly lower in nest vs. random plots ($U = 911, 723$; $z = -2.009, -3.267$; $p = 0.045$, $p < 0.001$). Mean sapling and shrub densities were lower in nest vs. random plots, but the result was not significant (Fig. 49 and 50). Mean proportions of water cover and bare ground were significantly higher in nest vs. random plots ($U = 735, 893.5$; $z = -3.302, -2.120$; $p < 0.001$, $p = 0.034$). Mean proportion of herbaceous cover was significantly lower in nest vs. random plots ($U = 768.5$, $z = -2.960$, $p = 0.003$).

Each site showed variability in vegetation structure characteristics. There were no significant differences in mean tree density and mean canopy cover between nest and random plots at all sites (Table 17). Canopy cover over the plot center was significantly higher in nest vs. random plots at TOM ($U = 0.0$, $z = -2.160$, $p = 0.030$) and MARS ($U = 2$, $z = -2.577$, $p = 0.008$). Canopy height was significantly higher in nest vs. random plots at DUNN ($U = 76.5$, $z = -2.955$, $p = 0.002$), SPRU ($U = 82.5$, $z = -2.916$, $p = 0.004$), and SAW ($U = 57.5$, $z = -3.382$, $p < 0.001$), and significantly lower at MARS ($U = 4$, $z = -2.301$, $p = 0.024$). Proportion of FHD “hits” was significantly higher in nest vs. random plots at ADAM ($U = 2$, $z = -2.819$, $p = 0.003$), TOM ($U = 0.0$, $z = -2.152$, $p = 0.030$), SPRU ($U = 74$, $z = -3.110$, $p = 0.002$) and SAW ($U = 88$; $z = -2.537$; $p = 0.010$). No significant differences were found between nest and random plots in mean sapling density or mean proportion of sapling cover. Shrub density was significantly lower in nest vs. random plots at SAW ($U = 92$, $z = -2.571$, $p = 0.014$). Proportion of shrub cover was significantly lower in nest vs. random plots at DUNN ($U = 100.5$, $z = -2.346$, $p = 0.017$), SPRU ($U = 96$, $z = -2.590$, $p = 0.010$), and SAW ($U = 91$; $z = -2.457$; $p = 0.013$). Proportion of water cover was significantly higher in nest vs. random plots at SPRU ($U = 81$, $z = -2.973$, $p = 0.003$). Proportion of herbaceous cover was significantly lower in nest vs. random plots at SPRU ($U = 49.5$, $z = -3.693$, $p < 0.001$).

Table 16. Mean vegetation structure characteristics of each site type.

Vegetation Structure	<u>Ravine</u> (n=114)	<u>Bench</u> (n=109)	<u>Statistical Results</u> (U, z, p)
Trees/plot	20.3 ± 8.1	31.6 ± 8.4	1987.0, -8.781, <0.001
Closest tree (m)	3.1 ± 1.2	2.7 ± 1.1	5228.0, -2.693, 0.007
Saplings/plot	10.6 ± 13.3	5.4 ± 6.1	4974.0, -2.582, 0.010
Sapling cover (%)	27.8 ± 25.7	18.3 ± 17.7	5064.0, -2.392, 0.017
Shrubs/plot	9.1 ± 16.9	4.4 ± 7.1	5292.5, -1.991, 0.046
Shrub cover (%)	27.3 ± 14.1	17.2 ± 11.6	3553.0, -5.525, < 0.001
Water cover (%)	5.9 ± 9.3	13.5 ± 16.2	4557.0, -3.702, < 0.001
Herb cover (%)	30.1 ± 18.9	27.3 ± 17.7	5682.5, -1.102, 0.271
Bare cover (%)	36.7 ± 20.2	42 ± 23.4	5458.5, -1.567, 0.117
Canopy coverage/plot (%)	93.9 ± 6.3	96 ± 8.9	4440.5, -3.856, <0.001
Canopy height (m)	24.1 ± 9.7	21.5 ± 4	4799.5, -2.948, 0.003
FHD Vegetation "Hits" (%)	50.7 ± 11	51.7 ± 10.7	5794.0, -0.870, 0.384

*"Closest tree" n= 123, 107 (more than one closest tree in some plots)

Table 17. Mean vegetation structure characteristics of nest and random plots at each site. *Denotes significant difference between nest and random plots.

Site	<u>Trees/plot</u>		<u>Canopy cover/plot</u>	
	Nest	Random	Nest	Random
Ravines				
ADAM	12.8 ± 5.4	20 ± 4.8	93 ± 4.5	86.7 ± 10.3
HORN	24.6 ± 5.7	28.4 ± 6.5	94.5 ± 5.7	97.5 ± 3.5
TOM	16.5 ± 2.1	24.1 ± 6.1	100 ± 0.0	97 ± 3.5
VAN	27 ± 12.4	25.1 ± 12.6	94.3 ± 6.1	94.5 ± 5.5
DUNN	15.5 ± 4.7	17.4 ± 3.2	93.1 ± 6	95 ± 5.3
Benches				
SPRU	29.8 ± 6.2	34 ± 6	97.2 ± 4.5	97.5 ± 4.9
MARS	33.8 ± 8.5	38.5 ± 5.7	98.8 ± 2.5	97.5 ± 3.5
SAW	31.7 ± 11.1	28.5 ± 6.9	94.5 ± 9.1	91.8 ± 20.8
<i>Ravines mean</i>	<i>18.2 ± 7.6 *</i>	<i>23 ± 8.1 *</i>	<i>93.7 ± 5.8</i>	<i>94.3 ± 6.9</i>
<i>Benches mean</i>	<i>30.8 ± 8.7</i>	<i>33.5 ± 7.4</i>	<i>96.2 ± 6.9</i>	<i>95.5 ± 12.7</i>
<i>Pooled mean</i>	<i>25.1 ± 10.3</i>	<i>27 ± 9.3</i>	<i>95 ± 6.5</i>	<i>94.5 ± 9.5</i>

Table 2. cont'd.

<u>Plot center cover</u>		<u>Canopy height (m)</u>		<u>FHD vegetation "hits"</u>	
Nest	Random	Nest	Random	Nest	Random
95.1 ± 13.2	95.8 ± 3.2	27 ± 13.2	21.7 ± 2.8	61.3 ± 8.9*	39.2 ± 8.7*
96.2 ± 3.2	95.6 ± 3.6	22.8 ± 4.4	23.2 ± 1.5	54 ± 9.5	46.1 ± 8.1
99.7 ± 0*	95.4 ± 3.3*	21 ± 0.0	20.7 ± 3.7	65.9 ± 3.4*	43.3 ± 9*
95.7 ± 3.1	95.4 ± 3.1	22.9 ± 4.4	21.5 ± 3.9	50.1 ± 7.7	42.5 ± 7.3
94.4 ± 5.7	95.2 ± 4.4	27.5 ± 14.9*	21.1 ± 2.8*	56.2 ± 10	49.5 ± 9.8
96.6 ± 2.7	97 ± 2	21.9 ± 4.1*	17.9 ± 1.2*	54.3 ± 10.6*	43.3 ± 6.3*
99.3 ± 1.9*	96.9 ± 2.3*	18.5 ± 1.9*	22 ± 2.2*	53.6 ± 10.2	44.4 ± 6.9
96.6 ± 3.3	91.2 ± 21.3	23.5 ± 4.1*	18.5 ± 2.5*	55.4 ± 8.5*	44.7 ± 13.8*
95 ± 4.9	95.4 ± 3.4	26 ± 12.4*	21.6 ± 3*	55.9 ± 9.8*	44.1 ± 9*
96.8 ± 4.1	94.9 ± 12.7	22.4 ± 4.1*	19.4 ± 2.7*	54.7 ± 9.6*	44.2 ± 9.5*
96 ± 4	95.2 ± 8.2	24 ± 9*	20.8 ± 3.1*	55.2 ± 9.7*	44.2 ± 9.1*

Table 17. cont'd.

Site	<u>Sapling density/plot</u>		<u>Shrub density/plot</u>		<u>Sapling cover/plot</u>		<u>Shrub cover/plot</u>	
	Nest	Random	Nest	Random	Nest	Random	Nest	Random
Ravines								
ADAM	15.8 ± 18.5	25.6 ± 19.5	3.2 ± 7.2	8.8 ± 11.8	36.4 ± 45.4	53.8 ± 23.8	23.6 ± 18.7	39.1 ± 14.2
HORN	11.5 ± 9.8	8.2 ± 7.6	2.5 ± 4.2	11.3 ± 27.9	30.6 ± 20.2	23.2 ± 18.1	23.6 ± 10.9	31.7 ± 18.2
TOM	35.5 ± 13.4	10.6 ± 13.8	0.5 ± 0.7	12.9 ± 26.3	65 ± 28.3	27 ± 30	44 ± 1.4	32.9 ± 17.3
VAN	1.9 ± 1.8	7.8 ± 12.1	18.1 ± 28.1	3.9 ± 4.6	8.7 ± 10.6	22.3 ± 19.4	20.6 ± 16.4	20.2 ± 9.8
DUNN	8.7 ± 12	5.8 ± 5.8	8.9 ± 13.3	15.6 ± 19.3	25.1 ± 25.9	21.8 ± 15.4	24 ± 11.6*	33 ± 8.4*
Benches								
SPRU	5.2 ± 5	5.8 ± 4.8	3.6 ± 5.7	9.6 ± 10.3	15.4 ± 14.5	24.8 ± 14.4	14.6 ± 8.9*	25.3 ± 12*
MARS	13 ± 18.2	6.1 ± 4	2.5 ± 5	0.1 ± 0.3	36 ± 36.7	23.5 ± 20	17.5 ± 11.5	16.9 ± 6.6
SAW	4.6 ± 6.4	5 ± 3.8	3.6 ± 5.9 *	10.3 ± 10.2 *	15.4 ± 17.8	21.1 ± 17.6	14.9 ± 12.3*	26.6 ± 15.5*
<i>Ravines mean</i>	<i>9.9 ± 12.6</i>	<i>11.6 ± 14.2</i>	<i>8.1 ± 14.4</i>	<i>10.5 ± 19.6</i>	<i>26.4 ± 26.8</i>	<i>29.6 ± 24.4</i>	<i>24.1 ± 12.7*</i>	<i>31.4 ± 14.9*</i>
<i>Benches mean</i>	<i>5.4 ± 6.8</i>	<i>5.6 ± 4.1</i>	<i>3.5 ± 5.7</i>	<i>6.8 ± 9.4</i>	<i>16.5 ± 17.7*</i>	<i>23.1 ± 17*</i>	<i>14.9 ± 10.5*</i>	<i>23.1 ± 12.5*</i>
<i>Pooled mean</i>	<i>7.4 ± 10</i>	<i>9.3 ± 11.8</i>	<i>5.6 ± 10.8</i>	<i>9.1 ± 16.5</i>	<i>20.9 ± 22.7*</i>	<i>27.1 ± 22*</i>	<i>19 ± 12.4*</i>	<i>28.2 ± 14.5*</i>

Table 17 cont'd.

Site	<u>Water cover/plot</u>		<u>Herb. cover/plot</u>		<u>Bare cover/plot</u>		<u>Number of plots</u>	
	Nest	Random	Nest	Random	Nest	Random	Nest	Random
Ravines								
ADAM	3.2 ± 5	5.2 ± 9.5	20 ± 10.5	29 ± 12.9	53.2 ± 26.2	26.7 ± 20.2	5	10
HORN	7.2 ± 10.8	5 ± 11	37.1 ± 23.8	20.6 ± 15.4	32.2 ± 16.8	42.5 ± 20.8	11	10
TOM	0.0 ± 0.0	7.2 ± 9.7	4 ± 2.8	37 ± 25.7	52 ± 1.4*	22.9 ± 15.5*	2	10
VAN	2.6 ± 6.8	3.7 ± 8.4	23.1 ± 14.6	22.6 ± 18.1	53.7 ± 25.8	53.5 ± 24.6	7	10
DUNN	8 ± 10.1	3.1 ± 6	33.1 ± 18.1	37.3 ± 16.1	35 ± 15.8	26.6 ± 9.5	39	10
Benches								
SPRU	21.2 ± 16.6*	4.2 ± 9.2*	28.9 ± 15.5*	52.5 ± 10.7*	35.3 ± 16.8*	18 ± 12.5*	41	10
MARS	0.0 ± 0.0	1.8 ± 3.9	8.3 ± 3.6	25.2 ± 26.2	74.3 ± 13.1	56.1 ± 23.8	4	10
SAW	13.2 ± 17.4	9.8 ± 11	18.6 ± 10.6*	33.8 ± 16.1*	53.4 ± 23.8*	29.7 ± 17.3*	33	11
<i>Ravines mean</i>	<i>6.6 ± 9.6</i>	<i>4.8 ± 8.8</i>	<i>30.7 ± 19.1</i>	<i>29.3 ± 18.8</i>	<i>38.5 ± 19.1</i>	<i>34.4 ± 21.6</i>	<i>64</i>	<i>50</i>
<i>Benches mean</i>	<i>16.7 ± 17.3*</i>	<i>5.4 ± 9.1*</i>	<i>23.5 ± 14.5*</i>	<i>37.1 ± 21.4*</i>	<i>44.9 ± 22.7*</i>	<i>34.5 ± 23.9*</i>	<i>78</i>	<i>31</i>
<i>Pooled mean</i>	<i>12.2 ± 15.2*</i>	<i>5.1 ± 8.9*</i>	<i>26.8 ± 17</i>	<i>32.3 ± 20.1</i>	<i>42 ± 21.3*</i>	<i>34.4 ± 22.4*</i>	<i>142</i>	<i>81</i>

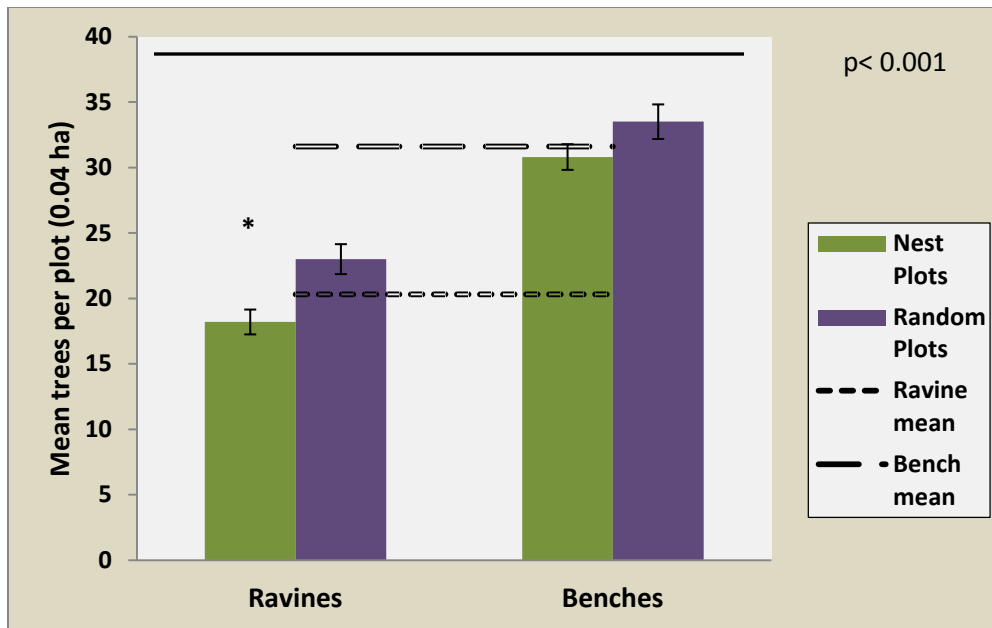


Figure 47. Mean trees per plot between ravines and benches and between nest and random plots within each site type. Bar denotes a significant difference in mean tree density between ravines and benches ($p < 0.001$). Asterisk denotes a significant difference in tree density per plot within ravines between nest and random plots ($p < 0.001$).

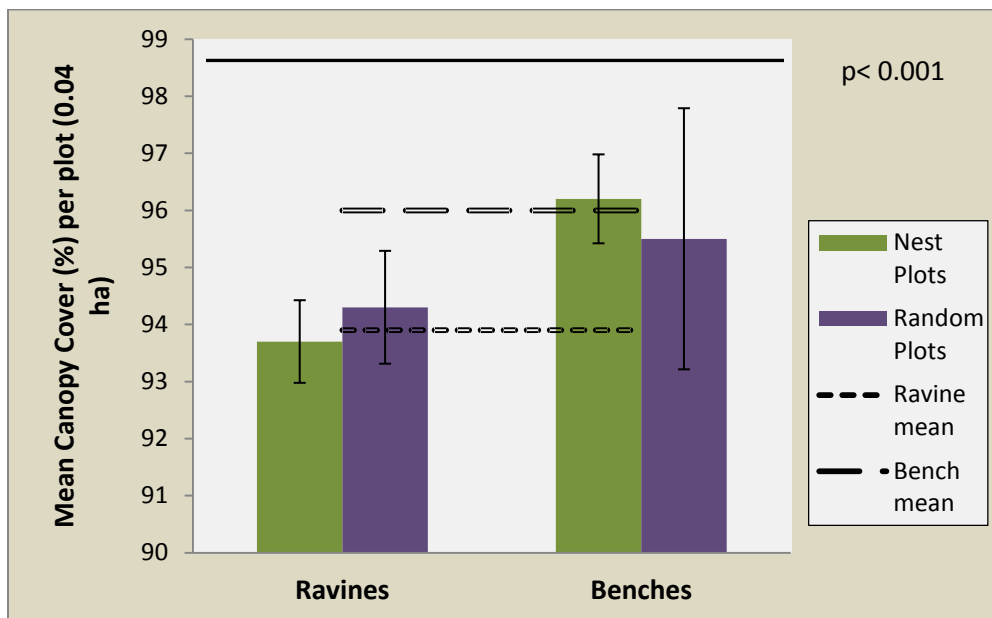


Figure 48. Mean canopy cover (%) per plot between ravines and benches and between nest and random plots within each site type. Bar denotes a significant difference in mean canopy cover between ravines and benches ($p < 0.001$).

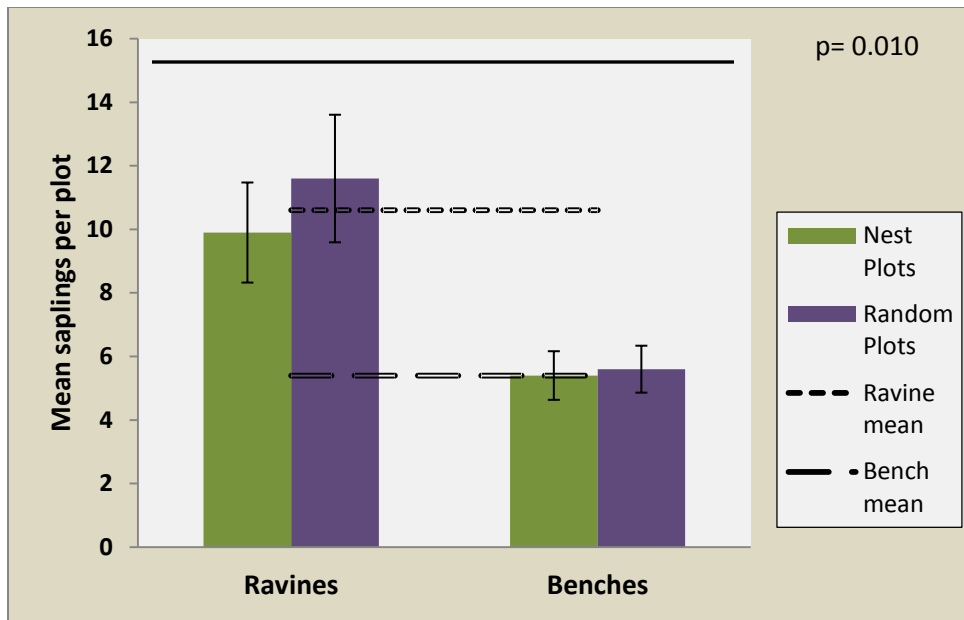


Figure 49. Mean saplings per plot between ravines and benches and between nest and random plots within each site type. Bar denotes a significant difference in mean sapling density between ravines and benches ($p=0.010$).

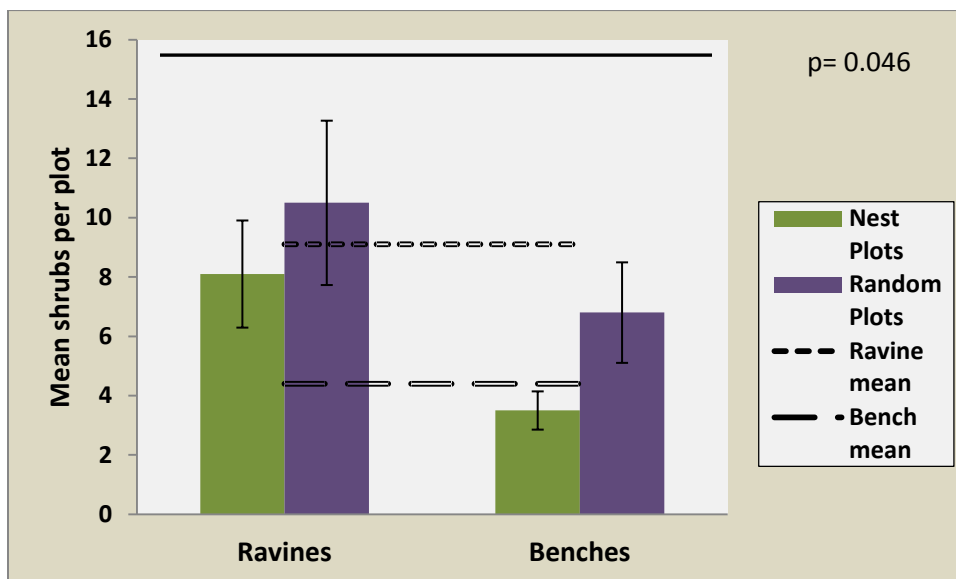


Figure 50. Mean shrubs per plot between ravines and benches and between nest and random plots within each site type. Bar denotes a significant difference in mean shrub density between ravines and benches ($p=0.046$).

and SAW ($U= 71.5$; $z= -2.984$; $p= 0.002$). Proportion of bare ground was significantly greater in nest vs. random plots at TOM ($U= 0.0$, $z= -2.156$, $p= 0.030$), SPRU ($U= 83$, -2.896 , $p=0.004$), and SAW ($U= 83$, $z= -2.672$, $p= 0.007$).

Vegetative Composition. – Select species of tree, understory tree, sapling and shrub relative densities were compared between ravines and benches (Table 18). Those compared were among the highest densities at each site type. Within the ravines, eastern hemlock was the most dense tree species, followed by yellow birch and black birch. Eastern hemlock was also the most dense tree on the benches (more so than in ravines), followed by yellow birch and sugar maple. There was a higher relative density of snags in the ravines than on benches. The understory layer in the ravines was dominated by witch-hazel, American beech and maple saplings with rhododendron and Japanese barberry, while the benches contained a higher density of birch saplings and serviceberry. Ravines and benches had similar densities of EAHE saplings, with the ravines having a slightly higher density.

The ecological important values (EIV; tree density + dominance + frequency) of nest tree species were compared between nest and random plots overall, between site types, and study sites (Table 19). Overall, within ravines and benches and within each study site, EAHE had the highest EIV of all nest tree species. Between nest and random plots, EAHE had higher EIVs overall and within both ravines and benches in nest vs. random plots. More variability was found within each site with ADAM, HORN and MARS having lower EIVs in nest vs. random plots, and the other sites having the opposite trend. Between nest and random plots, Yellow birch EIVs were lower overall and within both ravines in benches in nest vs. random plots. Again more variability was found between sites, with ADAM, HORN and SPRU having higher EIVs in nest vs. random plots, with the opposite trend at all other sites. Overall and within ravines, other species including black birch, American beech and sugar maple all had higher EIVs in nest vs. random plots with black gum EIV slightly lower in nest vs. random plots. Within

benches, black birch, sugar maple, and black gum had lower EIVs in nest vs. random plots, with American beech having a higher EIV in nest vs. random plots. For these four tree species, much site-level variability was found between nest and random plots. Sites with a relatively low sample size of nest plots (TOM, ADAM, MARS) must be interpreted carefully. Differences in sapling and shrub specific species mean stem densities were observed in nest and random plots.

Foliage Height Diversity. – The total proportion of foliage height diversity “hits” were compared at six height intervals between ravines and benches (Table 22). There was a significantly higher proportion of vegetation hits in intervals 0.5-3 m and 24+ m (greater than 24 m) in the ravines compared to the benches. Proportion of hits in interval 18-24 m was greater in the ravines than in the benches but not significantly. Benches had significantly higher proportions of hits in intervals 6-12 m and 12-18 m when compared to ravines. Proportion of hits in intervals 3-6 m and pooled intervals were non-significantly greater on the benches than ravines.

Foliage height diversity intervals and pooled intervals were also compared overall and within each site type between nest and random plots (Table 23). Overall, the proportion of vegetation hits was significantly greater in nest plots in intervals 6-24+ m and pooled intervals than random plots. Proportion of hits in intervals 0.5-3 m and 3-6 m was not significantly different in nest vs. random plots. Within the ravines, nest plots had a significantly higher proportion of vegetation hits in the 6-12 m, 18-24 m, 24+ m, and pooled intervals compared to random plots. Proportion of hits in intervals 0.5-3 m and 12-18 m were not significantly different between nest and random plots. Similarly, within benches, the proportion of hits was significantly greater in nest vs. random plots in intervals 18-24 m, 24+ m, and pooled intervals. Proportion of hits in intervals 0.5-3 m, 3-6 m, 6-12 m and 12-18 m were not significantly different in nest vs. random plots.

Table 18. Relative density (%) between ravines and benches.

Veg. Composition Type	Vegetation Composition	<u>Ravine</u>	<u>Bench</u>
		(n=114)	(n=109)
Tree	Eastern Hemlock	33.6	39.4
	American Beech	5.2	1.0
	Black Birch	7.5	3.2
	Yellow Birch	18.4	15.0
	Red Maple	2.4	4.7
	Sugar Maple	5.3	12.3
	Black Gum	1.0	0.5
	Basswood	0.7	4.2
	White Oak	1.4	3.9
	Snag	9.4	5.9
Understory Tree	Witch-hazel	17.4	38.7
	Rhododendron	5.6	0.0
Sapling	Eastern Hemlock	12.1	11.7
	American Beech	43.4	18.4
	Birch spp.	16.8	26.9
	Red Maple	2.2	1.2
	Striped Maple	6.9	0.2
	Sugar Maple	11.5	7.9
Shrub	Japanese Barberry	45.8	4.1
	Spicebush	10.3	11.7
	Serviceberry	11.2	31.7

Table 19. Ecological importance value of tree species.

Site	<u>Eastern Hemlock</u>		<u>Yellow Birch</u>		<u>Black Birch</u>	
	Nest	Random	Nest	Random	Nest	Random
Ravines						
ADAM	114.8	134.5	51.7	50.8	25.6	0.0
HORN	86.9	112.9	65.4	28.8	27.9	18.8
TOM	113.6	98.2	0.0	63.1	43.5	3.9
VAN	126.6	97.3	7.4	37.1	44.1	39.3
DUNN	67.3	52.9	53.5	60.0	25.1	41.1
Benches						
SPRU	93.3	45.8	62.7	47.4	9.2	9.3
MARS	90.6	115.7	4.3	25.1	23.2	0.0
SAW	113.2	73.0	32.6	35.9	9.4	24.7
<i>Ravines mean</i>	<i>101.8</i>	<i>99.2</i>	<i>35.6</i>	<i>48.0</i>	<i>33.2</i>	<i>20.6</i>
<i>Benches mean</i>	<i>113.2</i>	<i>73.0</i>	<i>32.6</i>	<i>35.9</i>	<i>9.4</i>	<i>24.7</i>
<i>Pooled mean</i>	<i>100.8</i>	<i>91.3</i>	<i>34.7</i>	<i>43.5</i>	<i>26.0</i>	<i>17.1</i>

Table 19 cont'd.

<u>American Beech</u>		<u>Sugar Maple</u>		<u>Black Gum</u>		<u>Number of plots</u>	
Nest	Random	Nest	Random	Nest	Random	Nest	Random
41.3	21.6	51.2	18.7	0.0	3.4	5	10
15.0	17.4	29.5	11.7	0.0	3.0	11	10
54.8	9.2	0.0	22.3	0.0	3.1	2	10
8.4	7.3	21.8	20.9	0.0	0.0	7	10
16.9	11.3	16.5	7.2	10.3	3.4	39	10
2.4	5.7	34.8	37.9	1.6	0.0	41	10
5.4	12.4	41.3	42.8	5.2	3.0	4	10
4.7	1.8	33.6	34.3	1.4	5.8	33	11
27.3	13.4	23.8	16.2	2.1	2.6	64	50
4.7	1.8	33.6	34.3	1.4	5.8	78	31
18.6	10.8	28.6	24.5	2.3	2.7	142	81

Table 20. Relative density (%) of sapling species.

Site	<u>Eastern Hemlock</u>		<u>American Beech</u>		<u>Birch spp.</u>	
	Nest	Random	Nest	Random	Nest	Random
Ravines						
ADAM	0.0	0.0	50.6	32.4	25.3	43.4
HORN	20.5	40.2	60.6	54.9	7.9	0.0
TOM	21.1	14.2	78.9	39.6	0.0	25.5
VAN	38.5	62.8	38.5	6.4	15.4	7.7
DUNN	0.9	1.7	46.2	25.9	4.7	20.7
Benches						
SPRU	1.9	1.7	15.0	1.7	40.7	20.7
MARS	73.1	23.0	1.9	27.9	7.7	0.0
SAW	3.9	10.9	30.9	20.0	31.6	14.5
<i>Ravines mean</i>	<i>16.2</i>	<i>23.8</i>	<i>55.0</i>	<i>31.8</i>	<i>10.7</i>	<i>19.5</i>
<i>Benches mean</i>	<i>26.3</i>	<i>11.9</i>	<i>15.9</i>	<i>16.5</i>	<i>26.7</i>	<i>11.7</i>
<i>Pooled mean</i>	<i>20.0</i>	<i>19.3</i>	<i>40.3</i>	<i>26.1</i>	<i>16.7</i>	<i>16.6</i>

Table 20 cont'd.

<u>Red Maple</u>		<u>Striped Maple</u>		<u>Sugar Maple</u>		<u>Number of plots</u>	
Nest	Random	Nest	Random	Nest	Random	Nest	Random
0.0	1.2	3.8	2.0	7.6	17.2	5	10
3.1	0.0	0.0	0.0	0.0	0.0	11	10
0.0	0.0	0.0	0.0	0.0	9.4	2	10
0.0	2.6	0.0	12.8	0.0	0.0	7	10
4.9	1.7	12.8	37.9	23.3	0.0	39	10
0.0	1.7	0.5	0.0	5.1	17.2	41	10
0.0	1.6	0.0	0.0	1.9	19.7	4	10
0.0	9.1	0.0	0.0	4.6	10.9	33	11
<i>1.6</i>	<i>1.1</i>	<i>3.3</i>	<i>10.5</i>	<i>6.2</i>	<i>5.3</i>	<i>64</i>	<i>50</i>
<i>0.0</i>	<i>4.1</i>	<i>0.2</i>	<i>0.0</i>	<i>3.9</i>	<i>15.9</i>	<i>78</i>	<i>31</i>
<i>1.0</i>	<i>2.2</i>	<i>2.1</i>	<i>6.6</i>	<i>5.3</i>	<i>9.3</i>	<i>142</i>	<i>81</i>

Table 21. Relative density (%) of shrub species.

Site	<u>Witch-hazel*</u>		<u>Rhododendron*</u>		<u>Japanese barberry</u>		<u>Spicebush</u>		<u>Serviceberry</u>		<u>Number of plots</u>	
	Nest	Random	Nest	Random	Nest	Random	Nest	Random	Nest	Random	Nest	Random
Ravines												
ADAM	12.5	6.8	0.0	0.0	0.0	56.8	0.0	11.4	87.5	18.2	5	10
HORN	29.6	2.7	0.0	0.0	0.0	35.4	18.5	45.1	51.9	16.8	11	10
TOM	100.0	14.7	0.0	7.8	0.0	46.5	0.0	31.0	0.0	0.0	2	10
VAN	11.0	20.5	0.8	30.8	47.2	38.5	0.0	0.0	33.9	0.0	7	10
DUNN	15.6	42.1	7.4	5.7	63.3	18.9	3.6	0.0	0.8	0.0	39	10
Benches												
SPRU	17.0	41.7	0.0	0.0	0.0	0.0	57.8	27.1	2.7	3.1	41	10
MARS	100.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4	10
SAW	36.1	61.1	0.0	0.0	16.8	0.0	30.3	6.2	14.3	29.2	33	11
<i>Ravines mean</i>	<i>33.7</i>	<i>17.4</i>	<i>1.6</i>	<i>8.9</i>	<i>22.1</i>	<i>39.2</i>	<i>4.4</i>	<i>17.5</i>	<i>34.8</i>	<i>7.0</i>	<i>64</i>	<i>50</i>
<i>Benches mean</i>	<i>36.1</i>	<i>61.1</i>	<i>0.0</i>	<i>0.0</i>	<i>5.6</i>	<i>0.0</i>	<i>29.4</i>	<i>11.1</i>	<i>5.7</i>	<i>10.8</i>	<i>78</i>	<i>31</i>
<i>Pooled mean</i>	<i>40.2</i>	<i>36.2</i>	<i>1.0</i>	<i>5.5</i>	<i>15.9</i>	<i>24.5</i>	<i>13.8</i>	<i>15.1</i>	<i>23.9</i>	<i>8.4</i>	<i>142</i>	<i>81</i>

*WIHA and RHODO were associated with other shrub densities due to the majority of these spp. having < 10 cm dbh and < 2 m height.

Table 22. Foliage height diversity between site types.

FHD Interval (m)	Ravine (n=114)	Bench (n=109)	Statistical Results (U, z, p)
0.5-3	6.9 ± 4.0	4.3 ± 3.5	3858.5, -4.901, <0.001
3-6	6.8 ± 3.8	7.3 ± 3.3	5670.5, -1.129, 0.259
6-12	9.1 ± 3.7	12.6 ± 2.5	2741.5, -7.226, <0.001
12-18	12.2 ± 3.1	13.8 ± 2.6	3995.5, -4.633, <0.001
18-24	11.0 ± 4.6	9.7 ± 5.3	5468.5, -1.549, 0.121
24+	4.9 ± 5.3	3.9 ± 5.2	5289, -1.966, 0.049
Pooled intervals	50.7 ± 11.0	51.7 ± 10.7	5794, -0.870, 0.384

Table 23. Foliage height diversity between nest and random plots.

Site Type	FHD Interval (m)	Nest Plots (n=142)	Random Plots (n=81)	Statistical Results (U, z, p)
Ravines	0.5-3	7.2 ± 3.9	6.5 ± 4.1	1434.5, -0.947, 0.344
	3-6	6.9 ± 3.9	6.5 ± 3.8	1508.5, -0.524, 0.600
	6-12	9.7 ± 3.5	8.2 ± 3.8	1240.5, -2.058, 0.040
	12-18	12.5 ± 2.9	11.8 ± 3.3	1462.5, -0.789, 0.430
	18-24	12.5 ± 3.9	9.0 ± 4.6	877, -4.139, <0.001
	24+	7.1 ± 5.8	2.0 ± 2.6	765.5, -4.830, <0.001
	Pooled intervals	55.9 ± 9.8	44.1 ± 9.0	618.5, -5.607, <0.001
Benches	0.5-3	4.3 ± 3.5	4.5 ± 3.6	1165.5, -0.294, 0.769
	3-6	7.2 ± 3.3	7.4 ± 3.6	1163.5, -0.307, 0.759
	6-12	12.6 ± 2.6	12.9 ± 2.5	1140, -0.466, 0.641
	12-18	14.0 ± 2.3	13.3 ± 3.2	995.5, -1.452, 0.147
	18-24	11.3 ± 4.9	5.8 ± 4.2	487, -4.865, <0.001
	24+	5.3 ± 5.6	0.4 ± 0.9	491.5, -5.029, <0.001
	Pooled intervals	54.7 ± 9.6	44.2 ± 9.5	494, -4.805, <0.001
Pooled	0.5-3	5.6 ± 4.0	5.8 ± 4.0	5601.5, -0.323, 0.746
	3-6	7.1 ± 3.5	6.9 ± 3.7	5559.5, -0.414, 0.679
	6-12	11.3 ± 3.3	10.0 ± 4.0	4719.5, -2.232, 0.026
	12-18	13.3 ± 2.7	12.4 ± 3.3	4767, -2.137, 0.033
	18-24	11.8 ± 4.5	7.8 ± 4.7	2952.5, -6.052, <0.001
	24+	6.1 ± 5.7	1.4 ± 2.3	2839, -6.440, <0.001
	Pooled intervals	55.2 ± 9.7	44.2 ± 9.1	2312.5, -7.423, <0.001

Surface Water and Terrain. – Nest and random plot terrain zones and mean distances to water (m) were also compared. Proportions of each plot type (nest vs. random) were compared between each terrain zone including; (1) stream; (2) floodplain; and (3) slope (Table 24). Overall, there were significantly more nest plots than random plots located in the stream zone ($\chi^2 = 8.032$, $df = 2$, $p = 0.018$). Fewer nest plots were found in the floodplain and slope zones than random plots. Within the ravines, nest and random plots were most commonly found in the slope zone, with a relatively low proportion of nest plots found in the stream zone. Within the benches, there were significantly more nest plots than random plots located in the stream zone ($\chi^2 = 12.3$, $df = 2$, $p = 0.002$). Fewer nest plots than random plots were located in the floodplain and slope zones. Significant differences in proportion of nest and random plots were found at two sites; (1) SPRU had significantly more nest than random plots occurring in the stream zone and significantly less nest than random plots occurring in the slope zone ($\chi^2 = 6.215$, $df = 2$, $p = 0.045$), and (2) SAW had significantly less nest than random plots occurring in the floodplain zone ($\chi^2 = 6.044$, $df = 2$, $p = 0.049$).

The mean distance to water (m) was compared between nest and random plots overall, within site types and at each site (Fig. 51). Overall, nest plots were placed significantly closer to water than random plots with a mean distance of 14.8 m vs. 25.3 m ($U = 3562$, $z = -4.733$, $p < 0.0011$). Similarly, within both the ravines and benches nest plots were located significantly closer to water than random plots with means of 15.8 m vs. 23.9 m and 14.0 m vs. 27.4 m, respectively ($U = 1145, 654$; $z = -2.599, -3.764$, $p = 0.009$, $p < 0.001$). Significant differences in mean distance to water between nest and random plots occurred at three sites (Fig. 52) including; (1) DUNN, nest plots were significantly closer to water than random plots with a mean of 13.6 m vs. 27.7 m ($U = 93.5$, $z = -2.519$, $p = 0.010$); (2) SPRU, nest plots were significantly closer to water than random plots with a mean of 5.4 m vs. 27.2 m ($U = 51$, $z = -3.752$, $p < 0.001$); and (3) MARS, nest plots were significantly farther away from water than random plots with a mean of 104.4 m vs. 34.9 m ($U = 4$, $z = -2.263$, $p = 0.024$). ADAM, HORN, VAN and SAW all

had nest plot mean distances that were closer to water than random plots, while at TOM the mean distance to water in random plots was shorter than nest plots.

Relationship of Acadian Flycatcher Densities to Eastern Hemlock Health

Two response variables were chosen; (1) pair density and (2) fledgling density. Nest survival was not chosen as a response variable because there were no significant differences in survival between studies or ravines and benches. Variables that were determined to be the most important potential predictors of these response variables were; (1) Site type; ravines and benches; (2) eastern hemlock health rating; (3) eastern hemlock relative density; and (4) mean understory density. As a result of the Pearson product-moment correlation coefficient test, the predictor variables site type, eastern hemlock rating and mean understory density revealed a high linear correlation to the response variables (Table 25). Eastern hemlock relative density showed a low to very low negative correlation with both response variables. Ravines had a moderate to strong negative correlation with pair and fledgling densities, while benches have the inverse correlation. Eastern hemlock rating had a moderate to strong positive correlation with pair and fledgling densities (the higher the EAHE rating the higher the densities). Understory density had a strong negative correlation with pair and fledgling densities. Among the two predictor variables EAHE rating and understory density, there was a Pearson r value of -0.632 which indicates a moderate to moderately-high negative correlation between these two variables (the higher the understory density the lower the EAHE rating).

Linear regression tests were performed to determine if significant relationships exist between these variables, and if we can use site type (ravines and benches), eastern hemlock health, and understory density to predict ACFL pair and fledgling density. The step-wise linear regression showed that multicollinearity existed between site type and EAHE health rating. This was because EAHE were rated significantly higher on benches

and significantly lower in ravines, thus high EAHE rating= benches and low EAHE rating= ravines. Therefore, both of these predictor variables did not need to be included in the linear regression test; EAHE health rating was chosen instead of site type. Results of the linear regressions showed a significant relationship between understory density and EAHE health with both pair ($F= 35.848$, adj. $r^2= 0.833$, $p< 0.001$) and fledgling densities ($F= 34.840$, adj. $r^2= 0.829$, $p< 0.001$; Tables 26 and 27). Adjusted r^2 values show that 83.3% and 82.9% variability in pair density and fledgling density are influenced by the combination of these predictor variables, respectively. Standardized beta weights for mean understory density were higher than EAHE health beta weights, therefore, mean understory density is a generally a somewhat larger contributor to the variability in both pair and fledgling densities (also reflected in Pearson's r coefficient values).

Table 24. Proportion of plots (%) in each terrain zone.

	<u>Stream zone</u>		<u>Floodplain zone</u>		<u>Slope zone</u>	
	Nest	Random	Nest	Random	Nest	Random
Ravines						
ADAM	0.0	20.0	20.0	10.0	80.0	70.0
HORN	18.2	10.0	27.3	10.0	54.5	80.0
TOM	0.0	20.0	0.0	20.0	100.0	60.0
VAN	0.0	10.0	14.3	10.0	85.7	80.0
DUNN	7.7	0.0	23.1	10.0	69.2	90.0
Benches						
SPRU	48.8	10.0	26.8	30.0	24.4	60.0
MARS	0.0	0.0	0.0	40.0	100.0	60.0
SAW	33.3	9.1	18.2	54.5	48.5	36.4
<i>Ravine</i>	<i>7.8</i>	<i>12.0</i>	<i>21.9</i>	<i>12.0</i>	<i>70.3</i>	<i>76.0</i>
<i>Bench</i>	<i>39.7</i>	<i>6.5</i>	<i>21.8</i>	<i>41.9</i>	<i>38.5</i>	<i>51.6</i>
<i>Pooled</i>	<i>25.4</i>	<i>9.9</i>	<i>21.8</i>	<i>23.5</i>	<i>52.8</i>	<i>66.7</i>

Table 25. Pearson's r coefficients and significance between variables.

Predictor	<u>Pairs/km</u>		<u>Fledglings/km</u>	
	Pearson's r	p	Pearson's r	p
Site Type	-0.649	<0.001	-0.782	<0.001
EAHE rating	0.779	<0.001	0.823	<0.001
EAHE relative density	-0.317	0.249	-0.153	0.587
Understory density	-0.880	<0.001	-0.845	<0.001

Table 26. Linear regression results of response variable pair density.

	Stand. Beta	S.E.	t	p	Tolerance
Understory density	-0.646	0.174	-4.576	0.001	0.600
EAHE Health	0.370	0.817	2.625	0.022	0.600
Understory density X EAHE Health	F	adj. r²	p	S.E.	
	35.848	0.833	< 0.001	1.23043	

Table 27. Linear regression results of response variable fledgling density.

	Stand. Beta	S.E.	t	p	Tolerance
Understory Density	-0.541	0.452	-3.785	0.003	0.600
EAHE Health	0.481	2.119	3.369	0.006	0.600
Understory density X EAHE Health	F	adj. r²	p	S.E.	
	34.840	0.829	< 0.001	3.19277	

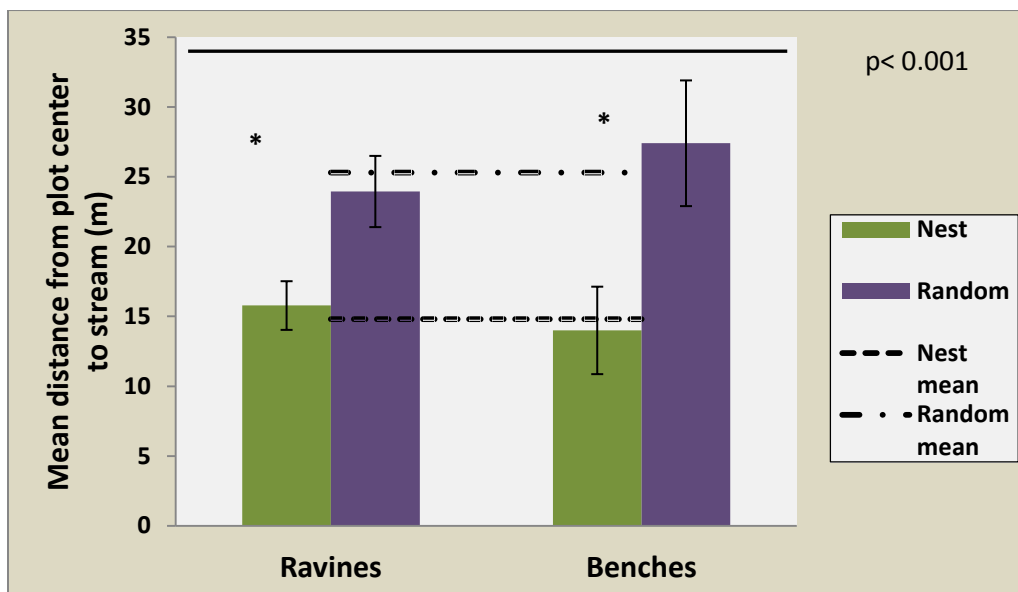


Figure 51. Mean distance to water (m) in nest and random plots at each site type. Bar denotes a significant difference in overall mean distance to water between nest and random plots ($p < 0.001$). Asterisks denote significant differences in mean distance to water between nest and random plots within both the ravines ($p = 0.009$) and benches ($p < 0.001$).

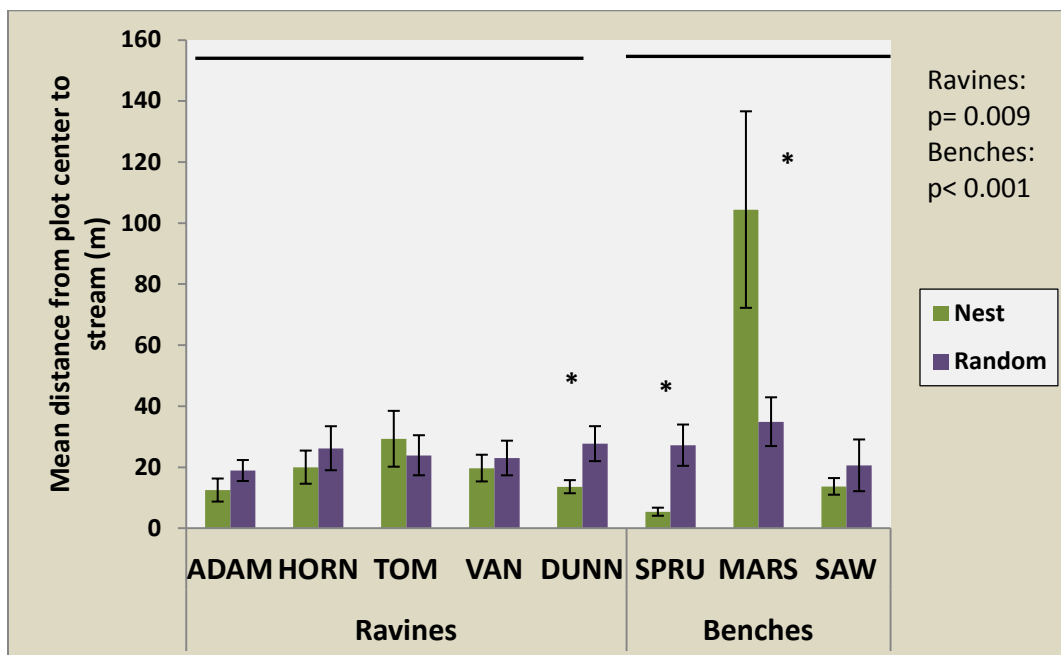


Figure 52. Mean distance to water (m) in nest and random plots at each site. Bars denote significant differences in overall mean distances to water between nest and random plots at both ravines ($p = 0.009$) and benches ($p < 0.001$). Asterisks denote significant differences in mean distance to water between nest and random plots within DUNN, SPRU, and MARS ($p < 0.05$).

Chapter IV

DISCUSSION

Eastern Hemlock Conditions

Eastern hemlock health rating, height of 1st live branch, health rating of 1st live branch and terminal measurements varied greatly between studies, ravines and benches, and among sites. Overall hemlock health has deteriorated in the 12 years since the 2002 survey, especially in ravines.

Eastern Hemlock Health 2014. – Results of the HWA survey performed in 2014 revealed that 86% of eastern hemlock trees on benches were rated as being fair-good with respect to their condition compared to only 34% in ravines (see Fig. 14, Table 2a). Specific measures of health, including height of 1st live branch, health of 1st live branch and mean terminal measurements per branch corroborate the overall health condition (see Figs. 18,21, 25, 26, 27 28; Tables 3a, 4) . On benches, branches on hemlocks were significantly lower and healthier than in ravines. Terminal metrics, which revealed less bare and HWA infested terminals per branch on benches than in ravines, also support this finding. Thus, bench hemlocks are unquestionably healthier than ravine hemlocks.

Comparing eastern hemlock health between study sites revealed that streams within a site type (ravines or benches) tended to be similar to each other. SPRU, SAW and MARS (bench sites) had similar EAHE condition metrics as did ADAM, HORN, TOM, VAN and DUNN (ravine sites). SPRU and SAW had the healthiest hemlock trees with 95% and 92%, respectively, in fair to good health, while ADAM (28%), TUWA (21%) and TOM (8%) had the fewest trees rated in fair to good health. DUNN (34%), HORN (55%), VAN (64%), and MARS (72%) fell between these other five sites. Other hemlock metrics (hgt. 1st live branch; health of 1st live, terminal metrics) varied between sites, but generally showed similar trends with SPRU and SAW having more low and healthy branches, and ADAM, TOM and TUWA having more high and unhealthy branches; and DUNN, HORN, and VAN were intermediate with regard to these metrics.

Morin et al. (2009) reported that HWA adelgid populations spread more rapidly in areas with greater EAHE basal area and abundance. Basal area of EAHE in ravines totaled 91.56 compared to 93.55 m² on benches but relative dominance was higher in ravines at 40.4% compared to 33.3% on benches. Thus, EAHE composed proportionally more basal area on benches (larger trees) compared to ravines but was not as dominant due to higher basal areas of other tree species in ravines. Abundance of EAHE was higher on benches with a total of 1,353 trees counted compared to 769 in ravines; yet relative density was similar at 39.4% on benches and 33.6% in ravines (Table 18). EAHE was more densely packed on benches compared to ravines which had a higher density of other deciduous tree species. According to Morin et al. (2009), it seems there should have been a more rapid or at least an equal rate of EAHE decline on benches as in ravines but obviously this is not the case. The rate of spread of HWA and its effects on hemlocks is likely driven by other factors discussed below.

Deterioration of Eastern Hemlock Health from 2002-2014. – Twelve years elapsed between HWA surveys performed in 2002 and 2014. Overwhelmingly, the 2002 survey had a significantly higher proportion of healthy trees and fewer trees in poor condition than the 2014 survey. Of all EAHE trees surveyed in 2002, 72% were rated

from good-pristine, compared to 2014 when only 11% of all trees rated from good-pristine. In 2002, only 9% of all EAHE trees surveyed were rated as very poor-poor compared to 2014 when over half of the EAHE trees surveyed (51%) were rated as very poor-poor (see Fig. 16, Table 2a)

Ravines showed a much more apparent decline than benches between surveys. In 2002, ravines already contained more unhealthy EAHE trees compared to benches, as reported in this study. In both surveys, no pristine trees were found in ravines, and virtually no very poor trees were found on the benches except for a single tree in 2014. In ravines, 87% of EAHE trees in 2002 were rated as being in fair to good condition, compared to only 37% in 2014. On benches in 2002, there were no EAHE trees rated very poor-poor, thus 100% of all trees surveyed were in fair-good condition and 93% were rated in good-pristine condition. In 2014, benches still had 83% of EAHE trees rated in fair-good condition, though only 32% of these trees were rated in good condition and there were no pristine trees.

Each comparable study site (ADAM, HORN, TOM, VAN , DUNN, SPRU, MARS) had significantly fewer healthy EAHEs and more unhealthy EAHEs in 2014 compared to 2002 (see Figure 17). ADAM showed the biggest difference in the proportion of very poor trees surveyed with 0% in 2002 and 25% in 2014. TOM had the greatest difference in proportion of tree in poor condition with 14% in 2002 and 88% in 2014. DUNN and MARS also showed significant changes in trees in poor condition with DUNN increasing from 5% in 2002 to 58% in 2014 and MARS increasing from 0% in 2002 to 27% in 2014. Half of the trees at SPRU (52%) and MARS (50%) were rated in fair condition in 2014 compared to 5% and 10% in 2002, respectively. Proportion of EAHE trees rated as good dropped significantly between 2002 and 2014 at all sites, with ravines sites showing decreases of 50-70% and benches showing decreases of 51-69%.

Mean height of the 1st live branch significantly increased by 2.7 meters between 2002 and 2014 surveys (see Figs. 19, 20). Ravines showed a significant increase of 3.7 m

between surveys, but the mean height on benches remained similar with a difference of only 0.06 m between 2002 and 2014. All ravines sites showed an increase in 1st live branch height. One bench, SPRU, also showed a significant increase. Interestingly, MARS was the only site to show a mean decrease in 1st live branch height, but it was not significant. MARS follows the trend in overall EAHE health decline between 2002 and 2014. It is likely that EAHE trees at MARS did not experience a lot of lower branch break-offs, but instead were rated poorly because poor health ratings of the 1st live branch (MARS had 0% of 1st live branches rated as poor in 2002 compared to 30% in 2014).

Following the trend of deteriorating EAHE condition, there were significantly more 1st live branches rated as poor and significantly less rated as good in 2014 compared to 2002 (see Fig. 23, Tables 3a,b). This was seen at both ravines and benches and among all study sites, with the exception of TOM, which had more branches rated as poor in 2002 than 2014, though the difference was not significant (see Fig. 24).

Mean number of terminals per branch significantly decreased between 2002 and 2014 by 17.9 terminals. Between 2002 and 2014, ravines experienced a significant decrease of 14.9 terminals per branch and benches a decrease of 22.8 terminals per branch. Because there was a significant difference in total terminals between studies, results of comparisons the number of non-new, new and infested terminals must be interpreted with caution. Overall and between ravines and benches, there were significantly fewer terminals of each type (non-new, new and HWA infested) in 2014 compared to 2002, with the exception of mean new terminals in ravines, which was lower but not significantly so (see Figs. 30,31; Table 4). With more dead/bare terminals in 2014 than 2002 there are fewer non-new terminals remaining on the branches, thus fewer terminals can be infested with HWA and there is less possibility for new growth.

Summary. - Between 2002 and 2014 there was a dramatic decline in EAHE health. Lower branch die-off and fullness/suppleness (good health) of branches has decreased significantly, and the deterioration of needles has increased significantly.

Eastern hemlocks at ravines show more deterioration than benches. It is important to note that population densities of adelgids fluctuate in response to density-dependent changes in the nutritional quality of hemlock (McClure et al. 2001). HWA densities may increase rapidly while newly infested trees are still healthy, and then drop sharply when trees begin to decline (McClure et al. 2001). Bench sites have experienced some hemlock deterioration since 2002, but still have healthier EAHE than ravines. Bench sites have the potential to experience this rapid HWA population increase on their still fairly healthy EAHE trees, so with time we may see an equally dramatic deterioration as we have seen in the ravines. Ravines had an average of 2.8 HWA infested terminals per branch (range: 0.0-32.0) which is only 4.7% of the mean total terminals on a branch. It is possible that the ravines are in this period of sharp decline and we may yet see some ravine areas experience increases in new growth. However, HWA densities are known to resurge if infested trees respond to a population drop with new growth (McClure et al. 2001). Continued monitoring of EAHE health on both ravines and benches is necessary to further determine rate of spread and potential mechanisms behind the rate of spread of HWA.

The question remains as to why eastern hemlocks are in better health on benches than in ravines currently? HWA is known to be spread by wind, passive movement of migratory birds, and humans (McClure and Cheah 1999). Among potential reasons could be physical isolation, on a landscape scale benches appear to be more isolated from other stands of hemlock than ravines and relative isolation from human activity compared to ravine hemlock stands. The five ravine sites in DEWA are all heavily hiked by recreationists, especially in warmer seasons, when HWA crawlers are active (April-May and June-July). Bench sites SPRU and SAW in DSF have trails, although they experience considerably less traffic compared to the ravine sites. MARS is located on private property and therefore hosts few or no hikers (albeit some dedicated graduate students). Benches also have narrower stands of hemlock along the streams that are surrounded by deciduous forest, while ravines have hemlock stands that extend a

considerable way upslope away from the stream. More extensive surrounding deciduous forest may somewhat protect the hemlock stands on benches. Ravine structure may facilitate more rapid spread of the HWA because trees are staggered elevationally. This may make it easier for HWA to spread from one tree to another. Wind dispersal may also be more of an issue in ravines than on benches. Finally, there is evidence that hemlocks closer to water fare better in the face of the adelgid onslaught than trees growing in dryer conditions (*****). Hemlocks are more likely to be growing in more favorable wet conditions on benches by virtue of the braided and meandering nature of bench streams flowing across a flat landscape.

Reproductive Success

Acadian flycatcher pair density, pair productivity, nest productivity all varied greatly between site types (ravines and benches) and study sites. Pair density differences were also observed between studies. Nest survival did not show much variability between studies, site types (ravines and benches), and among study sites.

Pair Density. – Breeding densities across all sites averaged 3.4 pairs/km (range: 0.0-9.0 pairs/km), with benches supporting significantly higher pair densities of 6.4 pairs/km (range: 1.6-9 pairs/km) compared to ravines with only 1.9 pairs/km (range: 0-4.6 pairs/km) (Figs. 33-34). Benches sustain healthier EAHE than ravines, as stated earlier, so it is not surprising that 90%, 94%, and 100% of nests at SPRU, SAW and MARS, respectively, were located in EAHE and that benches would have a higher pair density than ravines. Ravine pairs also showed a high preference for nesting in EAHE (see **Nest Preference** below), however, the poor condition of EAHE in ravine sites is likely the reason why pair density is significantly lower in the ravines.

Between study sites, pair densities varied but were not significantly different with SPRU having the highest pair density at 9.0 pairs/km and TOM having the lowest with only 0.68 pairs/km (Fig. 33). Study sites that stood out as anomalies included MARS and DUNN. MARS had fairly healthy EAHE throughout the study stream reach, yet had a very low pair density of only 1.6 pairs/km. This may be an anomaly because there is only one season of data from this stream. Nevertheless, it is strange to see complete abandonment of ACFL activity in an area that still supports relatively healthy EAHE. Other sites, such as TOM, ADAM, VAN and HORN also had low pair densities, however, EAHE health was generally poor at these sites so low pair densities were expected. DUNN, which also had poor EAHE health, supported 4.6 pairs/km, which was the highest of the ravines and the 3rd highest overall (below SPRU and SAW). Low EAHE densities and poor condition should dictate DUNN having a lower pair density but ACFL have increased their use of deciduous trees on this stream compared to other streams during the current study and compared to 2001/2002 (see **Nest Preference** below).

Mean pair density significantly decreased by 1.7 pairs/km across comparable pooled sites (HORN, TOM, VAN, DUNN, SPRU, MARS) between 2001/02 and 2013/14 (Table 5a-b). Ravines also showed a significant decrease of 1.9 pairs/km, while benches decreased by 1.1 pairs/km though this decrease was not significant. Decreases in pair density can be attributed to the decline in EAHE health. When comparable sites were compared individually, no significant decreases/increases were found, which would attribute to a lower sample size being compared. When ravine and bench sites are pooled together this creates a larger sample size for statistical analyses. Therefore, with continued study on ACFL pair densities it is very likely that eventually significant decreases and perhaps increases (likely SPRU and/or SAW due to potential crowding effect) will be found within a study site.

Between 2001/02 and 2013/14 decreases in mean pair density that were not significant were seen at TOM, DUNN, MARS and VAN (range: 1.8-4.9), while SPRU and HORN, had increases in mean pair density of 2.1 and 0.6 pairs/km, respectively.

Although EAHE health significantly decreased at HORN, a decline in mean pair density did not occur, it rose slightly. A shift in nesting preference did not occur at this site (unlike DUNN), so perhaps ACFL remain steady here because over half of the EAHE trees are still in fair health and perhaps other contributing factors make this site desirable for breeding such as higher food density, lower predation rates, etc. TOM had 0 pairs in 2013 and only 1 pair in 2014 compared to 3 and 4 pairs in 2001/02, respectively. Decline in ACFL breeding at this site can likely be attributed to the significant decline in EAHE health; lower branch die-off at the site was so great that a terminal survey couldn't even be done because branches retaining needles couldn't be reached. This was also the case at TUWA which had no ACFL activity in 2013/14. Thus, we may be witnessing the progression of complete ACFL abandonment of some sites as EAHE health declines. VAN pair density decreased by 4.9 pairs/km, the largest decrease of any site since 2002 although during the earlier study VAN was studied for only one year. Thus, the 7 pairs observed in 2002 may not have represented the typical pair density on this stream. MARS was only surveyed during one season during the current study so the 4.2 pairs/km decline since 2001/2002 may also not be representative of this stream. DUNN, as explained above, also produced some interesting results. EAHE health significantly declined as did pair density but the decline was not significant and in spite of this, DUNN had the third highest pair density during the current study. Unlike at any other ravine site, data suggests that pairs at DUNN are shifting their nesting preference to deciduous trees when compared to the 2001/2002 study. Although involving only one site, this is one of the most important and encouraging results from the current study since it suggests that ACFL have a capacity to alter nesting site preference. It is not necessarily a surprising result since ACFL are known to prefer deciduous trees in other parts of their range but actually demonstrating/seeing the switch occur is dramatic.

Nesting activity was patchy on virtually all streams with some areas supporting pairs while other portions of the stream reaches were devoid of nests. In fact, nests appeared to be clustered on some stream reaches including SPRU and SAW. On some

streams such as ADAM and VAN, EAHE were in poorer condition where nesting activity was absent. As with this study, Sheehan (2003) and Allen (2008) did not show much variability in pair density between consecutive field seasons. Thus, the result of an overall significant pair density between 2001/02 and 2013/14 does represent an actual decrease in pair density and not just a fluctuation between seasons.

Nest Survival. – The nest survival rate (Mayfield) for all sites pooled was 51.1%. Benches had a higher survival rate than ravines with 55.3% vs. 48.7%, respectively, although the difference was not significant (Table 6a; Fig. 36). There were no significant differences in survival rate between study sites. Of nests placed in EAHE, reproductive success was not influenced by health of the tree. There were a higher proportion of nests placed in very poor-poor hemlocks in ravines and inversely, a higher proportion of nests placed in healthier EAHE on benches. These placement patterns are a direct reflection of the number of poor vs. healthy trees at each site type. In general, reproductive success was also not influenced by surrounding vegetation structure, nest concealment, or nest placement variables. This is similar to results from the study conducted in southwestern Pennsylvania by Allen (2008) in pristine hemlock habitat and in Arkansas by Wilson and Cooper (1998a).

Of the 60 nests that failed in this study, 95% failed due to predation, with the other 5% (three nests) failing due to bad weather during one week of a particularly rainy, windy and cold period in 2013). No predation events were witnessed, so responsibility cannot be attributed to particular species. The most likely candidates, based on anecdotal observations of their abundance, include blue jay (*Cyanocitta cristata*), gray squirrel (*Sciurus carolinensis*), red-bellied woodpecker (*Melanerpes carolinus*), sharp-shinned hawk (*Accipiter striatus*), barred owl (*Strix varia*), eastern chipmunk (*Tamias striatus*), and black rat snake (*Elaphe obsoleta*). High numbers of blue jays and gray squirrels were observed at SPRU and DUNN in 2014, which (coincidentally?) had considerably higher failure rates than during the previous field season. Because nest concealment variables in this study and in Allen (2008) were

determined to have no effect on reproductive success it has been suggested by Wilson and Cooper (1998a) that there are other variables that may be important in concealing nests from predators. Further research employing remote cameras would help to clarify site specific predators of ACFL nests.

On comparable sites (HORN, TOM, VAN, DUNN, SPRU, MARS), the Mayfield survival rate in 2001/02 of 57.8% was higher than the Mayfield survival rate of 50.4% in 2013/14, though not significantly (Table 6b, Figs. 37-38). Ravines in 2013/14 had a lower survival rate of 43.5% compared to 62.8% in 2001/2002 but the difference was not significant. Benches in 2013/14 had a higher survival rate of 56.1% than the 2001/02 survival rate of 51.7% but this again was not significant. There were no significant differences found in the survival rates between study sites and nest survival rates in EAHE trees and hardwood trees were also not significantly different between studies. The overall nest survival rate of 51.1% is somewhat higher than in other areas of Pennsylvania. Allen (2008) reported a mean survival rate of 39% in southwestern Pennsylvania in pristine hemlock habitat, and Becker et al. (2008) reported a survival rate of 43% in central Pennsylvania.

Pair Productivity. - Mean fledgling density varied greatly between site types and among study sites (range: 0.0-22.6) (Table 11; Fig. 41) Benches produced significantly more fledglings/km with a mean of 16.2 fledglings/km compared to ravines with only 3.5 fledglings/km. Benches also had higher pair densities and generally a higher number of nests (with the exception of DUNN). Studies involving the Louisiana waterthrush (*Parkesia motacilla*) by Ernst (2012) and Barnes (2014) also found higher pair and fledgling densities on benches compared to ravines in on some of the same streams used in this study.

Site-level productivity is determined by abundance and nest survival. So, by understanding the factors that influence abundance (pair density) and nest survival (seemingly driven by predators) one would be able to predict productivity. This study

has provided evidence that abundance/density of ACFL in DSF and DEWA is influenced by the health of the eastern hemlock trees at a site. Other variables, including vegetation structure and composition, may also influence abundance/density at a site. However, nest survival does not appear to be influenced by EAHE health or vegetation structure/composition so further study on factors driving nest survival is needed.

Mean number of fledglings per pair was 2.1, and varied slightly between site types and among sites, though no significant differences were found. Benches produced a mean of 2.7 fledglings/pair, while ravines only produced a mean of 1.2 fledglings/pair (Table 11; Fig. 42). The mean at the benches is similar to the mean number of ACFL fledglings (2.8) per pair reported by Whitehead and Taylor (2002). Fledglings produced per pair should be independent of pair density per site, and should be more directly related to site-level nest survival rates. Allen (2008) reported fledglings/pair to be independent of pair density at study sites. In this study, there were higher survival rates (though not significantly) at three of the ravine sites (ADAM, HORN, VAN) compared to the bench sites, yet ravines produced a lower number of fledglings per pair. Thus, pairs at these sites were not producing as many fledglings not because nests were failing, but because of other reasons. A potential explanation is that the lower number of fledglings is the result of lower resource abundance in ravines. Also, instances of eggs not hatching occurred more in ravines sites than on the bench sites.

Interesting and valuable topics for further study on site-level and per-pair productivity can be explored by examining habitat/territory characteristics such as insect resources, predation rates and site fidelity. The latter has been studied by Mulvihill et al.(2008) who found site fidelity to be lower on lower quality, acid-impacted streams.

Nest Productivity. – Whitehead and Taylor (2002) reported that ACFLs commonly attempt to renest after a first nest failure as do most neotropical migrants. More than half of ACFL pairs (63.4%) that experienced a first nest failure attempted to

renest in this study (Table 12a). Similar levels of renesting attempts occurred after a first nest failure in both ravines and benches. The incidence of first nest failure was also similar between ravines and benches. Renesting attempts in this study were higher than reported by Allen (2008) who had only 24-33% of pairs attempting a renest after first nest failure in southwestern Pennsylvania.

Occurrence of double-brooding in this study may be among the highest reported for ACFL (Table 12b). Out of 50 first successful nests over both study years, 21 (42%) double-brooding attempts were observed. Rates of 2.5%, 7.6%, and 13.6% have been reported in Indiana (Whitehead and Taylor 2002), but other double-brooding reports have been poorly documented in Pennsylvania (Mumford 1964, Wilson and Cooper 1998b). Allen (2008) reported that 19-24% of successful pairs attempted to double-brood. Benches showed higher occurrence of double-brooding (46.7%) than ravines (35%) but the difference was not significant. Double-brooding has also been documented to be higher in benches than ravines in Louisiana waterthrush by Ernst (2012) and Barnes (2014), who together reported only one pair out of 28 (3.6%) attempting to double-brood in ravines, while nine out of 31 (29%) attempted to double-brood on benches.

Nest Placement

Nest Placement Variables by Site Type. - Variability of nest placement and concealment factors was observed between ravines and benches (Table 13) Nest distance to tip of branch significantly differed by about 10 cm, with nests at benches being placed more toward the middle of a branch than in ravines. Associated with this, nests on benches were placed 20 cm closer to the bole than in ravines though this value was not significantly different. This variability is likely influenced by other vegetation factors around nest site such as foliage height diversity, canopy cover over nest, lower

branch availability, and species of nest tree. Nest trees on benches were larger on average than in ravines. Larger trees could also mean longer lower branches, with more linear space to place a nest. Surprisingly, nests on benches had a significantly higher height (m) than nests in ravines. With EAHE health worse and lower branch die-off higher in ravines compared to benches, the opposite result was expected. Likely other site-specific factors influence nest placement between ravines and benches such as surrounding vegetation and predator species/predation rates.

Other variables that were significantly different between ravines and benches but not unexpected were; (1) nest distance to water was closer on benches; (2) nests on benches were placed overhanging the stream more frequently (larger stream width, more stream channels and narrower EAHE stands) and; (3) canopy cover was greater over nest bench plots than ravines (canopy was denser on benches because of healthier EAHE and trees occurring closer together)

Nest Placement Variables by Nest Tree. – In this study, eastern hemlocks composed 79.6% of tree species chosen for nesting (n= 113), while the other 20.4% of nests were found in various hardwood tree/shrub species (n= 29). More explanation on preferred nest tree species can be found below in **Nest Preference**. With low sample sizes for each individual hardwood nest tree species, all hardwood tree species were pooled together to form one category of tree type for analysis. Variability was seen in some nest placement and concealment variables between EAHE and hardwood species including nest tree height, dbh, canopy cover over plot, and nest overhang (**Table 14a-b**). EAHE nest trees were significantly taller and larger than hardwood nest trees, as are most EAHE trees in DSF and DEWA in general. EAHE nest trees also provided nest sites with a significantly higher amount of canopy cover than did hardwood nest trees. Nests that were placed in EAHE overhung streams significantly more often than nests in hardwood trees, and inversely, nests in hardwood trees overhung slopes and trails significantly more often than EAHE nests. ACFL are noted for nesting over open areas

such as those provided by streams and trails (Whitehead and Taylor 2002, Sheehan 2003, Allen 2008).

Mean nest height in EAHE trees was compared between 2001/02 and 2013/14 among the comparable sites of HORN, TOM, VAN, DUNN, SPRU and MARS (Figs. 43-44). There was a significant increase in height of nests placed in EAHE overall and in ravines. Nest height in EAHE trees increased by 1.1 m overall and by 1.3 m in ravines. There was also an increase of EAHE nest heights on benches of 0.7 m. There were increases in EAHE nest heights at HORN, TOM, VAN and SPRU (range: 0.3-1.7), however, these increases were not significant. DUNN EAHE nest heights significantly increased by 2.0 m, and interestingly, MARS EAHE mean nest heights actually decreased by 2.0 m. All of these increases reflect the significant increases in mean height of the 1st live branch from HWA survey on most streams. This shows that ACFL that nest in EAHE trees are adapting to lower branch die-off by nesting in higher branches.

Across all sites, the mean height of EAHE nests was 7.4 m. Besides being significantly higher than the mean height of 5.8 m reported by Sheehan (2003), this height was also higher than Allen (2008) who reported mean nest heights in pristine EAHE of 6.2 m and Walkinshaw (1966) who observed a mean height of only 3.8 m for EAHE nests in Michigan. Mean nest heights reported from various studies range from 3.99-6.5 m (Whitehead and Taylor 2002). The highest EAHE nest found in this study was 26 m on DUNN.

Nest Tree Preference. – Within DSF and DEWA it was evident that EAHE is an important and preferable nest tree species with 79.6% of nests (n= 113) from this study placed in EAHE trees and only 20.4% placed in various hardwood species (n= 29) (Figs. 46a-n). Of hardwood species selected for nesting, birches (Yellow birch= 8, Black birch= 1) were selected most often with 31% use, followed closely by American beech with 27.6% use.

An important question that remains to be fully answered is why ACFLs choose to nest in EAHE? Does EAHE improve nesting success and provide other nesting benefits or is this tree simply chosen because it is the most common tree on ravines and benches? There are three specific reasons EAHE may benefit ACFL including; (1) camouflage; (2) structure/microhabitat; and (3) foraging opportunities.

EAHE may provide ACFL with protection from predators because EAHE typically collect forest debris in branches and their flimsy nests often look like additional debris (camouflage). Their nests often incorporate nest tails composed of debris, perhaps an effective form of camouflage in the tropics where flycatchers evolved, but apparently not in temperate deciduous woodlands (Master et al. 2012). This study found that EAHE nest plots did provide significantly more canopy cover than hardwood nest plots (95.4% vs. 93.4%) although the sample size for hardwoods is small. Allen (2008) found no difference in cover above the nest between EAHE and hardwood species (0-1 m above nest), but he did not examine total canopy cover between the two nest substrates. EAHE has a unique structure of flattened needles extending laterally from the twigs that may be more desirable for building the hammock-like nest of ACFL. ACFL prefer to nest in areas near water (where EAHE generally are located) and areas where the understory and shrub layers are generally clear of vegetation growth, which EAHEs do provide, so nesting ecology does support EAHE preference. There was a higher mean shrub density in hardwood nest plots than EAHE nest plots (4.7 vs. 8.8 per plot), as was also reported by Allen (2008), and significantly more EAHE nests near streams than hardwood nests. No other nest placement or concealment variables were significantly different.

EAHE ecological importance value was the highest among all tree species for all sites in this study (more **Vegetation Composition**), so we know that EAHE was the most available individual tree species at all sites, providing the logical reason of why so many nests were placed in EAHE. However, when comparing EAHE to all other hardwood species pooled together, there was more hardwood availability than EAHE (availability of EAHE vs. hardwood was calculated using relative density not EIV). Observed hemlock

use exceeded expected use (Table 15a-b) in relation to habitat availability at all sites indicating a preference for nesting in EAHE. So, although this study has shown that EAHE health has significantly declined, a preference for nesting in EAHE is still evident. Preference for nesting in EAHE was also reported by Sheehan (2003) with 91.8% of all nests being placed in EAHE trees (n= 97). This preference was significantly higher than the 74% preference observed in this study (comparable sites only). This of course also implies that nesting in hardwood species increased significantly from 2001/02 to 2013/14 but this trend was primarily because of increased use of hardwoods at DUNN.

DUNN was unique among all sites in terms of abundance of hardwood species for nesting. We know that EAHE health has significantly declined at this site, as it has at all the sites, but ADAM and TOM had sharper declines in EAHE health than DUNN although DUNN did have the highest proportion of poorly rated 1st live branches and mean number of bare and infested terminals. In spite of the decline at ADAM and TOM there were no nests in hardwoods but relatively few nests were found at these sites. ADAM was not studied by Sheehan (2003) so no comparison can be made with past conditions. TOM had a very low pair density and also no observation of hardwood nesting. Compared to Sheehan (2003) which had 100% of all nests in EAHE, this study found more than half (53.8%) of nests in various hardwood species (Figs. 46m-n; EAHE nesting= 46.2%). So it seems that ACFL are gradually abandoning heavily impacted sites such as TOM and ADAM, while uniquely in DEWA, at DUNN they are transitioning to nesting in hardwood species.

DUNN has the lowest density of EAHE compared to all other sites (18% vs. 32-45%), and the highest hardwood density. This suggests that the availability of hardwoods at DUNN may be responsible for the shift to hardwood nesting seen there. However, TOM has an EAHE relative density of only 32%, leaving 68% of the site available for nesting in hardwood species. Evidence suggests that ACFL may be very selective about the hardwood trees they use. At DUNN, black gum had a relative density of only 2%, yet was one of the hardwood specs chosen for nesting. Hardwood species

commonly chosen for nesting include American beech, witch-hazel (Mumford 1964, Walkinshaw 1966, Bisson et al. 2000, Allen 2008), sugar maple, flowering dogwood (Christy 1942, Mumford 1964, Shearer 1990), and rhododendron (Sheehan 2003, Allen 2008). Interestingly, none of these studies included birch species as a nesting substrate, while in this study birch species was the most frequently used hardwood species being used nine times for nesting (31% of hardwood nests). There are obviously other factors such as predation, microhabitat/structure, and food resources that are in part driving the preference for nesting trees.

Nest Reuse. – Nest reuse was uncommon in this study, but was observed both within and between field seasons. It was reported by Allen (2008) but not reported by Sheehan (2003). Between-season nest reuse has been reported also by Mumford (1964), but within-season reuse has not been well documented. In this study, and in Allen (2008), nest reuse between seasons was observed twice relatively early in the breeding season (June/early July). In Allen (2008) there were two separate females who exhibited this behavior, and in this study, a single female reused two different nests. Same season nest reuse was only observed once in this study and three times by Allen (2008) at the end of the breeding season (late July). All nest reuse between seasons and within seasons, for both Allen (2008) and this study, involved refurbishing of the nest by adding streamers and material to the lining, cup and rim.

While interspecific nest reuse was not observed in this study, it was reported once by Allen (2008). Switching of nest tree species for renesting or double-brooding by a pair in the same season was not reported by Allen (2008) however did occur 16 times in the current study. Most instances occurred at DUNN, which had the highest use of hardwood trees for nesting.

Habitat Selection

Vegetative Structure. – When comparing ravines and benches, there was significant variability found in numerous structural variables (Table 16). Benches had higher tree densities, higher proportions of water (not surprising due to the greater stream width of benches), and greater mean canopy cover than ravines. Since EAHE are the dominant species on benches, greater mean canopy cover implies that EAHEs are healthier on benches compared to ravines which in turn can explain why pair/fledgling densities are higher on benches. Ravines had higher sapling and shrub densities and cover than benches. EAHE in poor condition in the ravines results in increased light penetration to the forest floor which in turn, increases the density of sapling and shrub growth. Nesting ecology of ACFLs shows that nests are not commonly placed over areas with dense undergrowth, so increasing sapling/shrub growth due to declining EAHE this may affect the desirability of ravines as nesting sites and may indicate why ACFL pair densities are lower in ravines compared to benches.

Overall, nest vs. random plot comparisons show broad scale nest site preferences of ACFLs while comparison between ravines and benches and individual study sites show finer scale, site specific preferences (Table 17). Overall, nest plots had greater mean canopy height (significant) and mean canopy cover (not significant) than random plots. Nest plots also had lower proportions of shrub and sapling cover (significant) and sapling/shrub densities (not significant) than random plots.

Within both ravines and benches, nest plots had lower mean tree densities, lower sapling/shrub densities and cover, higher mean canopy height, and higher proportion of both water cover and bare ground cover than random plots. So even though there are vegetation structural differences between ravines and benches, these are variables that ACFL appear to be using to determine suitable nesting areas. Several studies have found similar results with ACFL preferring lower understory growth (Wilson 1997, Bakerman and Rodewald 2006, Sheehan 2003, Allen 2008).

Vegetative Composition. – Eastern hemlock was found to be the most dense tree species by far on both ravines and benches, with EAHE being more dense on benches (Table 18). The second most dense tree species at both ravines and benches was yellow birch, it was more dense in ravines than on benches. This may be the reason that some ACFL decided to nest in this species. It would be interesting to know if yellow birch densities have increased significantly since the 2001/02 study at these study sites. Yellow birch is generally slow growing, but sapling growth rate can increase by up to 8 cm per decade with increased light penetration decreasing competition with neighboring tree species such as EAHE (USDA 1965b). ACFL were observed nesting in species with low relative densities such as black gum and rhododendron. It is likely that a combination of availability, species-specific traits, and local site traits influence ACFL choice for nesting substrate. Black birch was the third most dense tree species in ravines, and was used once as a nesting site in ravines. It was not as dense on benches. Sugar and red maples were more dense on benches than in ravines, with sugar maple being more dense than red maple at both site types. Sugar maple was observed as a nest tree species five times in this study but was not used in 2001/02. Red maple was observed once as a nesting site in 2001/02 but not during this study.

It has been documented that birch and maple species are among the most common saplings to grow in areas of EAHE decline (Orwig and Kizlinski 2002, Brantley et al. 2013). This coincides with results from this study that combined species of maple (sugar, red and striped) relative densities were higher in ravines than on benches. American beech sapling relative density was also considerably higher in ravines than on benches. Birch sapling relative densities however, were higher on benches than in ravines, which is likely because variety of other sapling such as was lower than ravines. The invasive Japanese barberry was considerably more abundant in ravines than on benches, which can also be attributed to greater light penetration due to the decline in EAHE.

Foliage Height Diversity. - Greater shrub and sapling densities were found in ravines because the poorer hemlock condition allowed greater light penetration to the forest floor. Therefore, it was not surprising to see significantly more vegetation hits in the 0.5-3 m interval in ravines compared to benches (Table 22). There was significantly more vegetation in the 6-12 and 12-18 m intervals on benches compared to ravines, because hemlocks were in better condition on benches with more lower and mid-level branches retaining needles. Nest plots, in both ravines and benches, had significantly more vegetation in intervals spanning 6-24+ m compared to random plots (Table 23). This may suggest that ACFL place their nests in areas that have more live vegetation over their nests. This also makes given preferred nest placement mentioned above. Coinciding with results from this study, Allen (2008) reported that hemlock sites and nest plots in general had significantly fewer vegetation hits in interval 0-2 m than did hardwood sites and random plots.

Surface Water and Terrain. - Given ACFL affinity for nesting close to water it was not surprising that nest plots were found significantly more frequently in the stream terrain zone and had a mean distance closer to water than random plots (Table 24, Fig. 51-52). Nest plots on benches were found more often in the stream zone than nest plots in ravines. Over half of the bench nests found (53.5%) were located within 10 m of the stream but this is more likely given the flat terrain and braided channels characteristic of benches. Other studies have documented ACFL affinity for streams and that ACFL abundance decreases with increasing distance from water (Christy 1942, Brauning 1992, Murray and Stauffer 1995, Allen 2008). All sites, except TOM and MARS, had a shorter mean distance to water in nest plots than random plots. This difference can be explained with low nest sample size at these sites. One interesting observation was at MARS, as distance from stream increased the health of EAHE trees increased. This effect was most evident at MARS, but moderately observed at SAW and SPRU as well. The distance of EAHE to water was not measured in the HWA survey, but this would be an

interesting variable in which to conduct further research to see if this phenomenon is going to be a more frequent occurrence as EAHE health continues to decline.

Acadian Flycatcher Response and Implications for HWA Management

The result of the Pearson's r correlation and linear regression tests show that the EAHE health rating variables and mean understory density can be accurate predictors of ACFL pair and fledgling density on streams (Tables 25-27). Multicollinearity found between site type and EAHE health rating demonstrates that benches have better EAHE health than ravines. Benches, with their combination of lower understory density and higher EAHE health ratings yielded significantly higher pair and fledgling densities. Ravines, with higher understory densities, due to increased light penetration, and lower EAHE health rating had lower pair and fledgling densities. It was not surprising that relative EAHE density was not significantly correlated with these response variables because EAHE dominated all sites and EAHE density was relatively similar between ravines and benches (34% vs. 39%, respectively).

It is important to note that the regression tests were performed using broad-scale analysis to increase sample size and test reliability. Much variability was seen among some sites, particularly Dunnfield Creek, which was often an outlier for many metrics. Increasing understory density and declining EAHE health has not affected pair and fledgling density at DUNN to the same extent observed on other ravines. Yes, pair density was lower at DUNN in 2013/14 compared to 2001/2002, but of all the ravines studied, this site still supported relatively high numbers of ACFL. My data suggest that the apparent switch in preference from nesting solely in EAHE trees to using various hardwood species (primarily birch spp.) is responsible for maintaining relatively high ACFL densities and productivity. This is encouraging from many perspectives including

the fact that birch species, especially black birch, are the likely successors to hemlocks in the northeast (Orwig and Kizlinski 2002, Brantley et al. 2013).

Conversely, TOM, with very poor EAHE and relatively high shrub density, also supports a generally high density of various hardwood species, including birch spp., yet no evidence of adaptation to nesting in hardwoods was observed at this site. In fact, it appears as though this site will follow the fate of TUWA and eventually be abandoned altogether. An additional ravine site in DSF, Devil's Hole, was explored for possible inclusion in this study during the peak breeding season in 2014. This site supports few EAHE (in fair to good health) and extensive hardwoods including American beech, birch spp., maple spp., and rhododendron. No ACFL activity was observed at this site. Thus, it appears that in northeastern Pennsylvania, ACFL prefer breeding in sites with a certain threshold of EAHE. When this threshold is crossed, whether with regard to EAHE abundance and/or health decline, ACFL appear to either abandon or adapt to the environment. Why adaptation is occurring only at DUNN is a central question raised by this thesis. Perhaps the fact that hemlocks have always been less dense in this ravine has made it easier to make the transition to nesting in hardwoods as hemlock condition has deteriorated. Further investigation of site-specific preferences of ACFL certainly needs to be done to answer this intriguing and enigmatic question. Interestingly, Ernst (2012) and Barnes (2014) both reported Louisiana waterthrush at TUWA, though this ravine site did have lower pair densities than bench sites. Several pairs of Louisiana waterthrush were observed breeding at DUNN during the course of this study and it will be interesting to see if, like the ACFL, Louisiana waterthrush will remain at DUNN but abandon TUWA as EAHE continues to decline.

The fact that the consequences of EAHE decline are reflected in ACFL metrics (especially pair and fledgling densities) and behavior (significantly increasing nest height, switching to nesting in hardwoods) indicates that ACFL is a robust bioindicator of the effects of hemlock deterioration. The studies of Tingley et al. (2002), who reported that ACFL were found only in non-infested sites in Connecticut, Sheehan (2003) who

reported lower pair densities in the most infested sites, and Becker et al. (2008) who also reported ACFL nest sites were always located in areas of low infestation, have also demonstrated this. What sets the current study apart is that it provides much more insight into exactly what nesting habitat/hemlock features ACFL actually respond to, how they respond and how the species may adapt in the future to the complete disappearance of hemlock in the forests of the northeastern United States.

Effectiveness as a bioindicator and evidence linking hemlock health to ACFL reproductive success suggest we can trust what ACFL are telling us about the relative ecological value of hemlock ravines and benches. Benches support higher densities of healthier hemlocks and, consequently, higher ACFL pair and fledgling densities and a higher occurrence of double-brooding. These results corroborate those of Ernst (2012) and Barnes (2014) using the Louisiana waterthrush as an alternative bioindicator of habitat health. Thus, based on these results, the bench environment appears more ecologically productive and capable of supporting higher densities of ACFL and LOWA chicks, presumably because of higher resource levels. Benches, therefore, should receive higher priority for HWA management/treatment over ravines. Ravines such as TOM, TUWA and ADAM are likely too degraded with respect to hemlock health to recover, even with management assistance. This is not to say that ravine sites in DEWA should be completely overlooked. They provide habitat for a variety of wildlife and are aesthetically pleasing with their fast moving streams tumbling over rocks and waterfalls as they descend from surrounding highlands to the Delaware River.

Currently, the most effective method of HWA control is chemical treatments, but these can only be applied to relatively small areas due to practical and economic considerations. They also need to be repeated making them even more expensive (USDA 2014). Other, so far less effective, biological methods of control are being explored and include various insect predators including; (1) Japanese lady beetle (*Sasajiscymnus tsugae*, formerly *Pseudoscymnus tsugae*); (2) Chinese lady beetle (*Scymnus* spp.) and; (3) British Columbia Derodontid beetle (*Laricobius nigrinus*) (Cheah

et al. 2004). Another biological treatment being explored is HWA-killing fungi (79 strains) which have been reported as having good potential as a control agent for HWA (Cheah et al. 2004). Chemical and biological treatments work best when used on hemlocks that are still in relatively good health, so it may be imperative to commence treatment first with these methods on benches followed by biological control. Ravines sites such as HORN and VAN that appear to still have recovery potential would be next in line. *Sasajiscymnus tsugae* has previously been used at VAN and ADAM by the National Park Service in 1999-2000 and effectiveness was monitored for three consecutive years (Evans 2003). Unfortunately, these beetles have had seemingly little effect on keeping HWA at bay. Bench sites in DSF are among the most valued eastern hemlock stands that remain in Pennsylvania and until an effective and lower-cost method of controlling the HWA is developed, protection of these habitats is essential to maintaining the eastern hemlock ecosystems and the unique communities that reside within them.

Evidence supports that due to the higher abundance of healthier hemlocks on benches we see higher pair and fledgling densities and higher occurrences of double-brooding, therefore as shown by Ernst (2012) and Barnes (2014) with the Louisiana waterthrush, productivity is higher on the benches than in ravines. When considering HWA management, because benches have higher productivity levels they would seem to have a higher ecological value, and therefore, should be a higher priority for HWA treatment than ravines. Ravines such as TOM and ADAM seem “too far gone” in regard to EAHE health and likely would not experience as high a benefit from treatment as the bench sites. This is not to say that ravine sites in DEWA should be completely overlooked. Ravine sites still provide habitat for various wildlife and are a great pleasure to many people aesthetically. As HWA continues to spread through DSF, DEWA, and the entire range of eastern hemlocks, many studies predict that eventually this essential tree will disappear and be replaced with mixed deciduous forest resulting in drastic ecological changes and shifts in biocommunities. Currently, the most effective method

of HWA control are chemical treatments, but can only be used in relatively small areas with continued reapplications making them very expensive (USDA 2014). Other seemingly less effective biological methods being explored are through use of insects including; (1) Japanese lady beetle (*Sasajiscymnus tsugae*, formerly *Pseudoscymnus tsugae*); (2) Chinese lady beetle (*Scymnus* spp.) and; (3) British Columbia Derodontid beetle (*Laricobius nigrinus*) (Cheah et al. 2004). Another biological treatment being explored is HWA-killing fungi (79 strains) which have been reported as having good potential as a control agent for HWA (Cheah et al. 2004). Chemical and biological treatments work best when used on hemlocks that are still in relatively good health, so it may be imperative to commence treatment on benches and ravines sites such as HORN and VAN that do not show as low EAHE health conditions as say, ADAM or TOM. It should be noted that the use of *Sasajiscymnus tsugae* has previously been used at VAN and ADAM by the National Park Service in 1999-2000 and was monitored for three consecutive years (Evans 2003), however these beetles have seemingly had little effect on keeping HWA at bay. Bench sites in DSF are among the most valued eastern hemlock stands that remain in Pennsylvania and until an effective and lower-cost method of controlling the HWA is developed, protection of these habitats is essential to maintaining the eastern hemlock ecosystem and the unique communities that reside within them.

Chapter V.

CONCLUSIONS

(Might make things clearer if I add bulleted conclusions)

Eastern Hemlock Conditions

Reproductive Success

Nest Placement

Habitat Selection

Summary

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Appendices

A.

B.