Eastern Hemlock Decline in Riparian Areas from Maine to Alabama

Daniel M. Evans, W. Michael Aust, C. Andrew Dolloff, Ben S. Templeton, and John A. Peterson

ABSTRACT

Eastern hemlock (*Tsuga canadensis*) in the Appalachian mountain range is threatened by the introduced hemlock woolly adelgid (*Adelges tsugae*). Potential impacts on riparian systems are great because of eastern hemlock's role as a foundation species that influences site soil, vegetation, and stream characteristics. We installed permanent research sites at 49 locations in riparian areas, from Maine to Alabama, to survey eastern hemlock health, measure stand dynamics, and predict near-term forest composition without eastern hemlock. This report summarizes the initial stand measurements from summer of 2008. We found hemlock woolly adelgid present at 25 of 49 stands from Massachusetts to Georgia, and all of these stands had some degree of hemlock decline. New England states, Ohio, western Pennsylvania, Kentucky, and Alabama had good hemlock health and no sign of hemlock woolly adelgid. Eighteen of the 49 sites had no nonhemlock conifer species in the overstory, and 30 of 49 sites had less than 5 $m^2 ha^{-1}$ of nonhemlock conifers. Without eastern hemlock, 25 of the stands would have more than 90% hardwood in the overstory, many of which are in the mid-Atlantic and southern states at sites dominated by shrubs in the understory such as *Rhododendron maximum*. Competition from shrubs may hinder stand regeneration after disturbance by hemlock woolly adelgid. On the basis of the abundance of hardwood species and lack of conifer species present in the overstory at many infested hemlock-dominated stands, these sites may convert to hardwood-dominated stands, which will affect terrestrial and aquatic ecosystem dynamics.

Keywords: eastern hemlock, hemlock woolly adelgid, forest insect pests, forest disturbance, shrub competition

Decline of individual tree species because of introduced diseases and insect pests has occurred before in the forests of eastern North America. In the early 1900s, chestnut blight (*Cryphonectria parasitica*) functionally removed the American chestnut (*Castanea dentata*) from the overstory of the Appalachian Mountains. In the 1930s, Dutch elm disease fungus (*Ophiostoma ulmi*) decimated American elm (*Ulmus americana*) across its range. Currently, the hemlock woolly adelgid (HWA) (*Adelges tsugae*) has the potential to equal these disturbances and may remove eastern hemlock (*Tsuga canadensis*), as well as the lesser known Carolina hemlock (*Tsuga caroliniana*) from much of their eastern ranges (Orwig and Foster 1998, Orwig et al. 2002). This small aphidlike insect was first introduced to eastern North America around Richmond, Virginia, in the 1950s (Souto et al. 1996) and has spread across much of the range of *T. canadensis* from New England to North Carolina (US Forest Service 2008).

T. canadensis is considered a foundation species that defines ecosystem structure and mediates hydrologic regimes and nutrient cycling (Ellison et al. 2005). It has been found to benefit wildlife such as salamanders, fish, and freshwater invertebrates that are intolerant of seasonal drying (Snyder et al. 2002), as well as songbirds and mammals (Yamasaki et al. 2000, Tingley et al. 2002). It provides a unique habitat structure, has a high leaf area index that is known to moderate daily temperature fluctuations and seasonal understory light levels, helps to stabilize stream base-flows (Rogers 1980, Canham et al. 1994, Ellison et al. 2005), and has a high level of transpiration (Ford and Vose 2006). In addition, its coniferous, recalcitrant litter creates soil conditions that suppress decomposition and slow nutrient turnover and release (Finzi et al. 1998, Jenkins et al. 1999, Yorks et al. 2003).

Evidence from the mid-Atlantic and New England states suggests that the decline of *T. canadensis* is having effects on overstory dynamics with a shift to hardwood species, which is altering forest ecosystems because of functional differences between evergreen conifers and deciduous hardwood trees. In Connecticut, Orwig and Foster (1998) observed prolific establishment of birch (*Betula lenta*) and opportunistic plants such as *Ailanthus altissima* following *T. canadensis* decline, which they attribute to more light on the forest floor. Small et al. (2005) saw a similar shift in overstory dominance to mixed hardwoods, including black oak (*Quercus velutina*); sassafras (*Sassafras albidum*); American beech (*Fagus grandifolia*); and *Betula* species, such as *B. lenta.* Eschtruth et al. (2006) found increases of understory vegetation and species richness in Pennsylvania and New Jersey, which they also attribute to increases in understory light levels. In southern New England, Jenkins et al. (1999) found increased *B. lenta* regeneration and increases in net nitrogen (N)

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This article uses metric units; the applicable conversion factors are: centimeters (cm): 1 cm = 0.39 in.; meters (m): 1 m = 3.3 ft; square meters (m^2): 1 m^2 = 10.8 ft^2; hectares (ha): 1 ha = 2.47 ac.

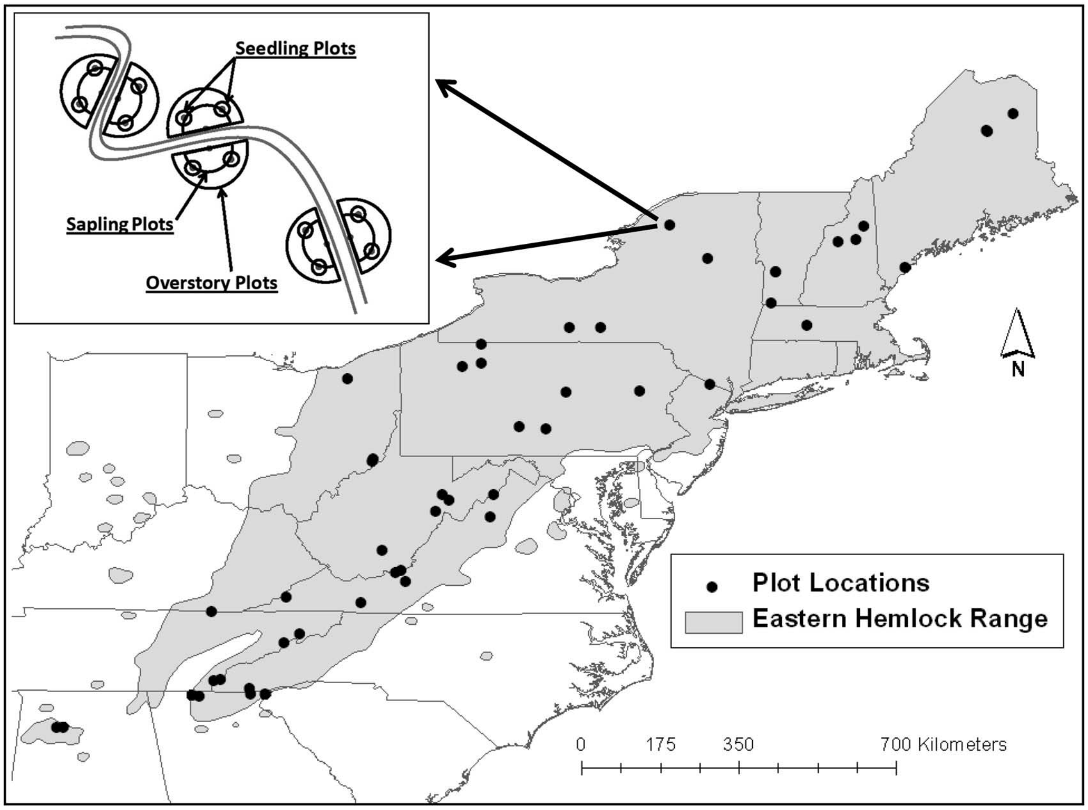

Figure 1. Plot locations within the range of *T. canadensis*. Map inset shows plot layout for overstory, sapling, and seedling measurements along the length of a representative stream.

mineralization, nitrification, and N turnover at sites with hemlock mortality. In a review article addressing the effect of *T. canadensis* mortality on nutrient cycling, Yorks et al. (2000) concluded that hemlock mortality will lead to increases in leaching of nitrogen and cations from terrestrial systems into aquatic systems and depletion of site capital of these nutrients. Over time, these processes can lead to increased acidification of soil and streams and to mobilization of toxic metals, such as aluminum, into streams (Stoddard 1994, Yorks et al. 2000).

The extent of impact on *T. canadensis* populations due to HWA infestation and the subsequent impacts on surface water quality across the Appalachian Mountains is largely unknown. Extensive surveys have been conducted addressing the spread of HWA (US Forest Service 2008), but little is known about the health and mortality of *T. canadensis* across its range. Research addressing overstory and understory responses to hemlock mortality in the southern Appalachian Mountains is also largely lacking, with little large-scale published research in the southern states, such as Tennessee, North Carolina, Virginia, and Georgia. We expect differences in forest response to HWA decline in the southern states because of climatic differences compared with more northern states, because HWA are sensitive to very cold winter temperatures (Parker et al. 1998, Skinner et al. 2003).

To address questions about the continental-scale patterns of *T. canadensis* decline and its effects on forest and aquatic systems, we installed permanent plots at 49 riparian sites ranging from Maine to Alabama. This report summarizes the initial stand measurements at these sites and has three primary objectives: (1) to survey the health of *T. canadensis* stands in riparian areas across its range, (2) to examine relationships between *T. canadensis* decline and site variables, and (3) to describe stand characteristics, including overstory, saplings, and seedlings, and predict immediate replacement of *T. canadensis*.

Methods

We selected potential *T. canadensis* riparian stands using a geographic information system (GIS) (ESRI, ARCMap) by generating target sampling points within the range of eastern hemlock in the Appalachian mountain range that are located on public lands. A random point generator was used in the GIS to select four target points within each degree of latitude along eastern hemlock's range. We then selected the nearest perennial stream to each target point with *T. canadensis* making up at least 10% of the total basal area (BA), over a 100-m stream length, using a combination of aerial photos, satellite images, local land managers' input, and ground truthing. A total of 49 riparian stands in 15 states (Figure 1), ranging from 19 to 1,050 m in elevation (Table 1), were selected for measurement. At each site, presence/absence surveys were conducted for HWA on the lower branches of the measured *T. canadensis* trees, as well as all trees along the length of the *T. canadensis* stand. If HWA was found anywhere in the stand, the entire stand was noted as infested for our analysis.

The length of each *T. canadensis* stand along the length of the stream was measured, and three measurement points were placed 25, 50, and 75% of the stand length from the edge of the stand. At each point, we established a 0.02-ha circular plot in the summer of 2008 (Figure 1). Plots were split in half, with two half-circle subplots on each side of the riparian zone and plot centers at a bank-full

Table 1. Site locations, stand density, and *T. canadensis* decline at riparian stands from Maine to Alabama.

Site name	State	Latitude	Longitude	Elevation (m)	Total BA[a]	Overstory *Tsca* BA	Healthy	Light decline	Moderate decline	Severe decline	Dead
					. . .(m^2 ha^{-1})	. . .	. . . .	. . . .(%)	. . . .	. . . .	
Squapan Lake[b]	ME	46.59287	−68.23920	201	38.5	4.5	100				
Ireland Pond Creek[b]	ME	46.25071	−68.76800	272	38.2	10.2	95	5			
Scraggly Lake[b]	ME	46.23397	−68.75670	234	61.1	17.0	96	1	4		
Plumb Creek[b]	NY	44.34207	−75.10780	274	63.7	54.8	31	38	13	4	13
Culhane Brook[b]	NH	44.33019	−71.22410	444	43.4	21.4	100				
Whiteface Creek[b]	NH	44.07038	−71.38450	458	50.7	24.3	95	5			
Walker Creek[b]	NH	44.01427	−71.74700	372	44.1	17.2	99				1
Forks Brook[b]	NY	43.67310	−74.35830	559	44.2	17.9	26	62	12		
Rachel Carson[b]	ME	43.49615	−70.40360	19	46.3	26.2	96				3
Texas Brook[b]	VT	43.40719	−72.98640	253	40.9	9.8	88	9	3		
Homerstone Brook[b]	VT	43.40700	−72.98590	251	47.6	24.8	57	28	5	1	10
Roaring Brook[b]	VT	42.78260	−73.08630	491	51.6	35.0	91	7	1		1
Cadwell Creek	MA	42.34253	−72.37270	200	67.2	34.5	4	33	43	18	3
Sexton Creek[b]	NY	42.30570	−77.10670	481	42.8	24.6	20	43	23	5	9
Danby Creek[b]	NY	42.28445	−76.48960	340	32.7	7.3	30	60	9		1
Coffee Run[b]	PA	41.96861	−78.88980	474	59.0	35.0	55	22	21	1	1
Eastern Branch[b]	PA	41.56960	−78.87970	477	48.9	34.8	97	2			2
Lamentations Run[b]	PA	41.50881	−79.26640	273	55.9	34.1	95	1			3
Buckeye Trail[b]	OH	41.26978	−81.57240	229	51.9	14.6	97	3			
Wanaque River	NJ	41.15387	−74.31350	205	36.5	9.8	1	36	47	14	2
Hickory Run	PA	41.02476	−75.70850	347	62.1	43.2		21	50	27	1
Rapid Run	PA	40.99449	−77.18060	490	60.0	30.3	12	65	12	6	5
Trough Creek	PA	40.31385	−78.12940	296	66.4	37.6	2	31	53	13	1
Roaring Run	PA	40.27397	−77.57820	280	53.3	14.5		14		60	26
Dismal Creek[b]	OH	39.66671	−81.04610	232	28.8	0.7	100				
Rockcamp Creek[b]	OH	39.60772	−81.07590	290	16.0	1.1	100				
Condon Run	WV	38.94319	−79.66690	924	38.3	20.5	6	66	21	1	6
Little Stony Creek	VA	38.93893	−78.64440	488	52.2	16.8			3	49	47
White Branch	WV	38.84457	−79.52360	925	29.8	16.2			31	58	11
Little River	WV	38.61807	−79.78910	874	51.0	23.4	38	47	11	5	
Fridley Run	VA	38.49165	−78.70270	596	36.1	20.7				77	23
Meadow Creek	WV	37.83583	−80.87390	538	48.3	14.5		25	53	21	
White Rocks Branch	VA	37.43295	−80.49092	918	48.8	15.8		2	18	79	1
Laurel Creek	VA	37.38000	−80.59300	1022	48.0	17.6		27	22	30	21
War Spur	VA	37.20520	−80.40630	654	42.3	11.9		9	56	34	1
Big Stone Gap[b]	VA	36.88242	−82.79070	542	26.9	15.6	71	20	8	1	
Middle Creek	VA	36.76967	−81.29300	815	49.4	13.4		14	68	18	
Rock Creek[b]	KY	36.60535	−84.27730	373	54.5	24.6	78	14	8		
Clarks Creek	TN	36.14720	−82.52850	526	48.5	20.9	39	47	10		4
Paint Creek	TN	35.97655	−82.84160	655	48.4	26.7	21	61	6		12
Garett Creek	NC	35.22779	−84.10550	684	48.4	27.3		16	51	25	7
Shuler Creek	NC	35.21470	−84.24260	495	36.6	15.0		7	72	21	
Nantahela River	NC	35.06770	−83.52270	1050	53.5	27.0			51	48	1
Persimmon Creek	GA	34.95394	−83.49260	742	35.6	24.1			40	57	3
Overflow Creek	GA	34.95158	−83.20850	489	27.5	13.5		9	67	13	12
Sumac Creek	GA	34.92073	−84.67190	420	40.6	9.6	51	49			
Burke Branch	GA	34.91116	−84.52490	735	60.4	36.4	39	60	1		
Sipsey River[b]	AL	34.28308	−87.39590	181	43.7	30.9	99	1			
Rush Creek[b]	AL	34.27448	−87.25160	227	52.4	14.4	99				1

[a] BA, basal area; *Tsca*, *T. canadensis*.
[b] No sign of hemlock wooly adelgid at site.

location. Data from the subplots were added to produce plot-scale summaries and means. At each subplot, a complete measurement of the overstory was taken for all trees and shrubs greater than 10.2 cm dbh (1.37 m). *T. canadensis* trees were also measured for their level of decline using a 5-point scale: 1 = healthy (<10% dieback), 2 = light decline (11–25% dieback), 3 = moderate decline (26–50% dieback), 4 = severe decline (>51% dieback), 5 = dead. We did not include intermediate and suppressed *T. canadensis* trees in these measures to avoid overestimation of decline due to site resource competition effects on lower-story trees. An index of hemlock decline, based on the mean of a site's overstory dominant and codominant *T. canadensis* dieback scores, was calculated for descriptive purposes. Within the overstory plots, smaller 0.004-ha plots were used to measure the midstory (saplings) of trees and shrubs that were greater than 1.37 m but with a dbh less than 10.2 cm (Figure 1). Two 0.0004-ha plots were nested within the midstory for measurement of all plants with heights less than 1.37 m (seedlings).

Principal component analysis (PCA) was used to explore associations between site location variables, site quality measures, and hemlock decline in stands that are infested by HWA. The primary goal of this analysis was to identify site characteristics that have relationships with hemlock decline that could focus future research efforts or assist forest management decisions. All sites with *T. canadensis* in the overstory (47) were included in this analysis. The following independent variables were selected that had possible mechanistic relationships to tree health and growth and are readily available to forest managers: latitude (LAT), longitude (LONG), elevation (ELEV), slope (SLOPE) annual precipitation (PRECIP), mean overstory stand height (HEIGHT), stand BA (TOTALBA), *T. canadensis* BA (TSCABA), and the percentage of the stand that

Table 2. Eigenvalues and eigenvectors for two principal components at riparian *T. canadensis* stands from Maine to Alabama.[a]

Component axis	1	2
Eigenvalue	3.539	2.378
Percentage of total	32.178	21.622
Eigenvectors (unrotated)		
Elevation	0.353	0.005
Percentage stressed	0.291	−0.016
Tsca BA	−0.083	0.509
Latitude	−0.495	0.050
Longitude	−0.471	0.020
Stand height	0.348	−0.001
Precipitation	0.286	0.058
Slope	0.128	−0.170
Total BA	−0.083	0.509
% *Tsca* BA	0.012	0.545
TRI	0.230	0.079

[a] $n = 47$. *Tsca*, *T. canadensis*; BA, basal area; TRI, topographic radiation index. Percentage stressed = % *T. canadensis* dead + moderate/severe decline.

was *T. canadensis* (%TSCA). In addition, we calculated an aspect-based topographic radiation index (TRI) (Roberts and Cooper 1989). This index ranges from 0 to 1, with 0 indicating generally cool and wet north- and northeast-facing slopes and 1 indicating warm, dry, south- or southwest-facing slopes. We used the sum of the percentage of *T. canadensis* BA that was dead or was in moderate or severe decline to create a continuous measure of stressed hemlock (hemlock decline [HD]) for use as a dependent variable. The elbow technique was used within the PCA to select the components with Eigenvalues that represented significant variability (Zuur et al. 2007). Analysis was conducted with SAS 9.1 (SAS Institute, Cary, NC).

Results

Overstory *T. canadensis* Decline

Strong gradients of *T. canadensis* decline were apparent across the study range. HWA was not detected in the northern New England states (Maine, Vermont, and New Hampshire), and the majority of the *T. canadensis* was healthy (Table 1), with the exception of one site in Massachusetts. In addition, there was no sign of HWA and little *T. canadensis* decline in northwestern Pennsylvania, Ohio, Alabama, and Kentucky. All other sites showed signs of HWA and *T. canadensis* decline, with two exceptions at Big Stone Gap (Virginia) and Rock Creek (Kentucky). The highest levels of *T. canadensis* decline were in the band running along the Allegheny Mountains to the Smoky Mountains, from south-central Pennsylvania to the western tip of North Carolina. Decline indexes were greatest in the mid-Atlantic and southern states, with the highest statewide mean decline index of 3.3 out of 5 in Virginia ($n = 7$) and North Carolina ($n = 3$). West Virginia and New Jersey had the next highest mean decline index, 2.8 ($n = 4$ and 1 respectively). Little Stony Creek (Virginia) had the highest mean stand decline index level, 4.4, with Roaring Run (Pennsylvania) and Fridley Run (Virginia) having the next highest mean decline index, 4.1. Across all sites a mean of 36.9% of the *T. canadensis* BA had moderate decline or worse, with 13 mid-Atlantic and southern sites having more than 75% of the *T. canadensis* BA in moderate decline or worse (Table 1).

The elbow technique within the PCA selected two axes or components (Table 2) that describe significant variability. In the first component, low LAT and LONG and high ELEV and HEIGHT correspond to each other, demonstrated by the large loading for these variables. Hemlock decline has a slight relationship with this component. In the second component, PRECIP, TSCABA, %TSCA, and TOTALBA correspond positively to each other.

Overstory Stand Description

T. canadensis was found in a variety of forest types across the study range, from spruce-fir ecosystems in its northern range to diverse hardwood-dominated, mixed mesophytic ecosystems in the mid-Atlantic and southern states. Total BA of overstory trees ranged from 16.0 to 67.2 $m^2\ ha^{-1}$ (Table 1). Overstory *T. canadensis* BA ranged from 0.7 to 54.8 $m^2\ ha^{-1}$, which represented between 2.6 and 86.1% of overstory BA at all sites. *T. canadensis* made up more than 50% of the overstory BA at 23 of the 49 stands, and only 3 stands had less than 20% *T. canadensis* in the overstory. A mean of 5% of the standing *T. canadensis* BA was dead, with a maximum of 47% dead at Little Stony Creek (Virginia) (Table 1). Three stands in Virginia and one stand in Pennsylvania had more than 20% of *T. canadensis* BA dead. We did not observe any Carolina hemlocks at any of our sites.

Coniferous replacements for *T. canadensis* were generally scarce and not diverse. Mean overstory nonhemlock conifer BA was 6.4 $m^2\ ha^{-1}$ (Table 3). Eighteen of the 49 sites had no nonhemlock conifers in the overstory, and 30 of 49 sites had less than 5.0 $m^2\ ha^{-1}$ of nonhemlock conifers. However, there were notable exceptions to this trend. In Maine, New Hampshire, Vermont, and northern New York, there were substantial components of nonhemlock conifer, including *Abies balsamea*, *Thuja occidentalis*, and *Picea* spp. (Table 4). In contrast, southern and mid-Atlantic states had less diversity of nonhemlock conifer species, and many sites had no nonhemlock conifers. *Pinus strobus* was found at 19 stands, ranging from northern Maine to Georgia, with a mean of 2.5 $m^2\ ha^{-1}$ for all sites. *Picea rubens* was found at three West Virginia and Virginia sites with relatively high BA at Little River (West Virginia) and War Spur (Virginia). Other conifers were limited, with the majority being relatively small BA of other *Pinus* spp. and one stand of *Pinus taeda* at Rush Creek (Alabama).

Hardwood species in the overstory were diverse and extensive (Table 3). Hardwoods constitute a mean of 43.0% of the total BA across all sites, with a range from 7.6% to 97.4%. However, if *T. canadensis* were removed from the stands, hardwood would constitute a mean of 78.0% of the BA at all stands. Without *T. canadensis*, 25 of the stands would have greater than 90% hardwood in the overstory, many of which are in the mid-Atlantic and southern states. All 49 stands had an *Acer* species, and 44 stands had a *Betula* species. *Acer* made up a mean of 4.3 $m^2\ ha^{-1}$ across all the sites and was dominated by *A. rubrum* and *A. saccharum* with lesser amounts of *A. pensylvanicum*. *Betula* constituted a mean of 4.2 $m^2\ ha^{-1}$ and comprised *B. lenta*, *B. papyrifera*, *B. alleghanienis*, and *B. nigra*. *Acer* and *Betula* spp. made up a mean of 9.6% of total overstory BA for all sites. If *T. canadensis* were removed from the overstory, *Acer* and *Betula* spp. would make up a mean of 36.2% of total BA. *Quercus* was also well represented across our study range, with a mean of 3.6 $m^2\ ha^{-1}$. Twenty-nine sites had *Quercus* present, with a majority being *Q. alba* and the remainder being a mix of *Q. coccinea*, *Q. rubra*, *Q. prinus*, and one *Q. velutina*. *Fagus grandifolia* was found on 29 sites spread across the study, and *Liriodendron tulipifera* was found on 24 sites, primarily in the stands in Ohio and south, with mean BA of 1.6 and 2.4 $m^2\ ha^{-1}$, respectively. Twenty-nine other hardwood species were identified, which represented an additional 2.6 $m^2\ ha^{-1}$ of hardwood BA in the overstory.

Table 3. Overstory description at riparian *T. canadensis* stands from Maine to Alabama.

		Conifers						Hardwoods						
Site name	State	*Tsca*[a]	*Picea* spp.	*Pinus strobus*	*Abies balsamea*	Other conifers	Total	*Acer* spp.	*Betula* spp.	*F. grandifolia*	*L. tulipifera*	*Quercus* spp.	Other hardwood	Total
		(m^2 ha^{-1})												
Squapan Lake	ME	4.5	14.5	2.3	1.8	11.4	34.6	1.5	2.3					3.9
Ireland Pond Creek	ME	10.2	15.9		2.0	4.6	32.7	3.0	2.5					5.5
Scraggly Lake	ME	17.0	9.6		1.7	28.2	56.4	0.4	4.0				0.3	4.7
Plumb Creek	NY	54.8	3.3				58.1	1.0	1.9	0.2			2.5	5.6
Culhane Brook	NH	21.4	4.0	4.9	0.7		31.0	5.8	6.1	0.4				12.4
Whiteface Creek	NH	24.3	9.0		2.6		35.9	8.5	4.4	0.5			1.4	14.8
Walker Creek	NH	17.2	2.6				19.8	9.3	6.5	5.2		0.6	2.7	24.3
Forks Brook	NY	17.9	5.7		3.0		26.7	5.3	12.2					17.5
Rachel Carson	ME	26.2	2.9	12.2			41.3	3.9	1.1					5.0
Texas Brook	VT	9.8	4.5		0.8		15.2	7.5	14.3	2.1		1.6	0.2	25.7
Homerstone Brook	VT	24.8					24.8	2.8	1	2.4		6.9	0.7	22.8
Roaring Brook	VT	35.0	3.8		2.1		40.9	3.6	6.8	0.1			0.2	10.6
Cadwell Creek	MA	34.5		17.4			51.9	6.3	6.6			1.9	0.5	15.3
Sexton Creek	NY	24.6		2.1			26.7	8.8	2.2			1.9	3.1	16.1
Danby Creek	NY	7.3					7.3	7.8	5.6				12.0	25.5
Coffee Run	PA	35.0					35.0	5.9	10.2	0.1		1.2	6.6	24.0
Eastern Branch	PA	34.8					34.8	10.4	1.0	1.1			1.7	14.2
Lamentations Run	PA	34.1					34.1	8.6	3.3	2.9		5.5	1.5	21.8
Buckeye Trail	OH	14.6					14.6	0.9	0.1	0.3	0.5	33.0	2.4	37.3
Wanaque River	NJ	9.8					9.8	14.4	4.5	1.2	0.3	3.2	3.1	26.7
Hickory Run	PA	43.2		9.9			53.0	5.2	3.7				0.2	9.0
Rapid Run	PA	30.3		9.7			40.0	9.5	2.7			5.3	2.5	20.0
Trough Creek	PA	37.6		6.6			44.1	5.7			6.6	4.6	5.3	22.3
Roaring Run	PA	14.5		1.0			15.6	1.8	1.9	17.9	10.5	3.3	2.2	37.7
Dismal Creek	OH	0.7					0.7	9.1	1.1	1.8	6.2	1.8	8.1	28.1
Rockcamp Creek	OH	1.1		2.2			3.4	3.7		0.7	0.6	5.7	1.9	12.7
Condon Run	WV	20.5	0.7				21.2	8.1	6.6				2.5	17.1
Little Stony Creek	VA	16.8		0.3		0.6	17.6	2.2	0.1	0.1	2.7	29.2	0.1	34.5
White Branch	WV	16.2					16.2	3.0	4.6				5.9	13.6
Little River	WV	23.4	15.2				38.6	3.8	4.0	1.1		1.5	1.9	12.4
Fridley Run	VA	20.7		0.1		0.3	21.2	0.6	1.4	0.2	7.3	2.8	2.7	14.9
Meadow Creek	WV	14.5					14.5	3.5	6.1	6.1	3.1	8.5	6.7	33.9
White Rocks Branch	VA	15.8					15.8	3.9	5.5		9.0	13.9	0.8	33.1
Laurel Creek	VA	17.6					17.6	3.0	24.0				3.4	30.4
War Spur	VA	11.9	10.4				22.3	3.3	8.6	3.4		3.3	1.5	20.0
Big Stone Gap	VA	15.6					15.6	3.4	3.3		1.8		2.8	11.3
Middle Creek	VA	13.4		21.8			35.2	0.2	3.5		2.8	6.1	1.5	14.2
Rock Creek	KY	24.6					24.6	2.7	0.1	20.1	3.1		3.9	29.9
Clarks Creek	TN	20.9		4.5			25.3	1.8	1.4	0.7	1.6	5.3	12.4	23.2
Paint Creek	TN	26.7		13.0			39.7	1.1	3.3		2.2		2.1	8.7
Garett Creek	NC	27.3					27.3	3.2	6.4		7.0	3.3	1.3	21.1
Shuler Creek	NC	15.0		1.4		0.6	16.9	1.8	1.4	1.4	5.8	0.8	8.5	19.7
Nantahela River	NC	27.0					27.0	2.8	2.6	0.5	0.5	15.6	4.6	26.5
Persimmon Creek	GA	24.1					24.1	0.1	1.2	0.2	7.2	0.2	2.5	11.5
Overflow Creek	GA	13.5		5.0			18.5	1.7			2.5	1.1	3.8	9.1
Sumac Creek	GA	9.6		8.8		1.4	19.8	3.2	2.0	0.3	6.6		8.6	20.8
Burke Branch	GA	36.4		0.6			37.0	3.1	2.3	0.4	15.8	0.7	1.2	23.4
Sipsey River	AL	30.9					30.9	2.7		3.3	1.2		5.6	12.8
Rush Creek	AL	14.4				6.8	21.2	0.8		5.6	11.1	6.0	7.8	31.2

[a] *Tsca*, *T. canadensis*.

Understory Description

T. canadensis saplings were found at 45 of the 49 sites, with a mean of 273 stems ha^{-1} (Table 4). Only 15 sites had other conifer saplings, and these sites were primarily in the northern sites. Forty two sites had hardwood saplings with a mean of 307 stems ha^{-1}. Thirty-eight sites had shrubs in the sapling size class, with a mean of 734 stems ha^{-1} with a clear trend toward more shrubs in the mid-Atlantic and southern states (Table 4). *Rhododendron maximum* had a mean of 431 stems ha^{-1} and made up the majority of shrubs in the sapling size class, particularly in the southern states.

T. canadensis seedlings were found at 33 sites, with a mean of 2,228 stems ha^{-1} (Table 4). Thirty sites had no coniferous replacement for *T. canadensis* in the seedling size class, whereas all sites but one had hardwoods in the seedling size class, with a mean of 8,161 stems ha^{-1}. Shrubs were also abundant in the seedling size class with a mean of 8,935 stems ha^{-1}, which were again dominated by *R. maximum* in the southern states (Table 4).

On a site level ($n = 49$), there was a strong positive correlation between HD and *R. maximum* stems ha^{-1} in the sapling ($r = 0.296$, P-value $= 0.038$) and seedling ($r = 0.425$; $P = 0.0023$) size classes. However, there was not a significant relationship between sapling and seedling sized *R. maximum* stems ha^{-1} and sapling and seedling-sized *T. canadensis* stems ha^{-1} (four-comparisons $P \geq 0.087$).

Discussion

Eastern Hemlock Health

We found *T. canadensis* to be in decline in riparian areas from Massachusetts to Georgia and that 9 of 49 sites had lost at least 10%

Table 4. Understory description for saplings and seedlings at riparian stands from Maine to Alabama.

Site name	State	Saplings (10.3 cm > dbh > 0.1 cm)					Seedlings (height < 1.37 m)				
		Tsca[a]	Non-*Tsca* conifer	Hardwood	Shrubs	*Rhma*	*Tsca*	Non-*Tsca* conifer	Hardwood	Shrubs	*Rhma*
		(stems ha^{-1})									
Squapan Lake	ME		2,389	1,544			1,236	19,562	1,648		
Ireland Pond Creek	ME	844	165	350			8,237	10,502	6,384	618	
Scraggly Lake	ME	618	927	432			4,530	12,973	9,267	4,325	
Plumb Creek	NY	103		227			206	1,854	15,238		
Culhane Brook	NH	680	329	268			6,590	7,001	9,267	2,883	
Whiteface Creek	NH	535	906	82	535		9,267	9,472	10,296	2,883	
Walker Creek	NH	597	144	227	247		4,325	7,001	4,942		
Forks Brook	NY	618	288	165	247		3,707	1,854	5,766	5,766	
Rachel Carson	ME	227	62	41			7,207	3,089	17,297	10,296	
Texas Brook	VT	309	371	782	41		206	4,736	8,443		
Homerstone Brook	VT	309		124			412		5,560		
Roaring Brook	VT	288	185	844	41		4,942	3,089	20,798	412	
Cadwell Creek	MA	185					618	618	2,883	2,265	
Sexton Creek	NY	82		288					11,943	2,883	
Danby Creek	NY			21	412				2,471	12,149	
Coffee Run	PA	124		62					10,090	824	
Eastern Branch	PA	82		144	62		2,059		4,119	824	
Lamentations Run	PA	62					618		7,001		
Buckeye Trail	OH	412		309	288				2,471	2,059	
Wanaque River	NJ	165		432	844	206			3,501		
Hickory Run	PA	21		82	62	62	10,296		4,942	5,354	3,501
Rapid Run	PA	62			247		15,856		5,766	2,677	
Trough Creek	PA	41		62	515	62	206	412	7,207	618	
Roaring Run	PA	41		618	124		412		28,005	2,265	
Dismal Creek	OH	1,112		494	247		1,030	206	13,179	18,121	
Rockcamp Creek	OH	62	21	288	268		618		9,472	13,179	
Condon Run	WV	82	62	21	2,780	2,739	1,030		5,354	8,443	8,237
Little Stony Creek	VA	494	41	21	844		4,325	618	824	53,127	16,473
White Branch	WV			21	41				18,739	2,265	
Little River	WV	62		185	144		2,059	9,061	23,063	3,501	
Fridley Run	VA	432		329	1,009				20,798	27,387	
Meadow Creek	WV	103		206	782	762			2,265	1,030	206
White Rocks Branch	VA	556			577	474	412		2,677	618	618
Laurel Creek	VA	103		62	4,221	4,221			1,030	9,884	9,884
War Spur	VA	62	41	144	1,791	844			11,326	1,854	1,854
Big Stone Gap	VA	103		103	2,162	1,812	206		3,501	4,942	2,265
Middle Creek	VA	227		206	1,750	1,462		206	2,677	6,178	4,736
Rock Creek	KY	329		3,562	185		412		15,856	16,062	
Clarks Creek	TN	391		185	659	391	206		7,413	18,327	824
Paint Creek	TN	21		247	1,874	1,112	412		4,530	23,269	8,237
Garett Creek	NC	515			1,359	1,153			3,913	16,268	12,767
Shuler Creek	NC	103		21	2,080	906			1,030	17,709	8,031
Nantahela River	NC	62			1,338	1,215			5,354	31,917	9,266
Persimmon Creek	GA				1,380	1,380			2,059	22,857	12,355
Overflow Creek	GA	82		206	2,203	1,400	2,265	1,030	2,883	38,301	4,119
Sumac Creek	GA	638	21	515	1,153	824	5,972	206	18,945	9,267	2,883
Burke Branch	GA	577		62	2,327	947				11,326	9,472
Sipsey River	AL	268		865	906		8,649		6,178	6,590	
Rush Creek	AL	577		206	206		618		11,532	16,268	

[a] *Tsca*, *T. canadensis*; *Rhma*, *R. maximum*.

of their overstory *T. canadensis*. Eighteen stands in the mid-Atlantic and southern states had at least 50% of their *T. canadensis* in moderate to severe decline. Without progress on suppression of HWA, it is likely many of these stands will lose part or all of their overstory *T. canadensis*. With our single sampling of current decline of *T. canadensis,* we cannot determine how fast the species will be removed from our study stands, but we can see that much of the *T. canadensis* is stressed and that portions of it have already succumbed to HWA. However, we must note that our data come from public lands. Differences may exist in the way public lands are responding to HWA compared with private lands, because of past management or salvage harvesting activities.

Our survey of *T. canadensis* decline, conducted in the summer of 2008, generally agrees with US Forest Service data describing the range of the HWA in 2008 (US Forest Service 2008). The range of infestation, according to the US Forest Service maps, runs along the western edge of the mainstem of the Appalachian Mountains and curves east into southern New England. The range of HWA appears to be limited in the north by temperature (Parker et al. 1998, Skinner et al. 2003), and it is limited in the South by its only suitable, widespread southern host species, *T. canadensis* (McClure 1987). The insect appears to have almost reached the southern terminus of its possible range, as only a few dozen counties in the range of *T. canadensis* do not have HWA in Georgia and Alabama. The US Forest Service infestation maps indicate that the HWA has not reached very far into New Hampshire, Vermont, Maine, and northern New York. It has also not spread far enough west to affect stands in northwestern Pennsylvania or the isolated *T. canadensis* stands in

Ohio and Alabama. Our data support this, showing healthy stands of *T. canadensis* in New Hampshire, Vermont, Maine, Ohio, Alabama, northern New York, and western Pennsylvania.

However, there are a few discrepancies in terms of *T. canadensis* health within the estimated range of the HWA. Our data show a finer scale of *T. canadensis* decline over its range compared with the US Forest Service maps. The US Forest Service maps are produced from data at a county-level scale. Many counties that have the insect may have the majority of their stands uninfested and may have generally healthy trees. This is particularly true for newly infected counties because the insect is able to jump into new counties through a diverse range of vectors, such as logging, trails, roads, and birds (McClure 1990). Our data confirm that there is variability in infestation and *T. canadensis* health in counties that are labeled as infested.

Implications of Losing *T. canadensis*

The implications of riparian systems in the Appalachian Mountains losing overstory *T. canadensis* are far-reaching because of its role as a foundation species that controls forest and aquatic ecosystem dynamics (Ellison et al. 2005). This is particularly true in the mid-Atlantic and southern states, where we found few conifer replacements for *T. canadensis* and many stands with no nonhemlock conifers. If *T. canadensis* succumbs to the HWA, our data indicate that there will be immediate, short-term shifts to hardwood-dominated overstories. We found that many stands have substantial hardwood trees in the overstory. These trees have the capability to capture resources and expand their size and stand position if *T. canadensis* trees die. This agrees with the modeling work of Spaulding and Rieske (2010), who demonstrated that many stands in their study range would convert to hardwood forest types with the introduction of HWA and subsequent loss *of T. canadensis* in the overstory. The ability of the existing *T. canadensis* seedlings and saplings to grow and compete with existing hardwoods, in the presence of HWA infestation, is unknown. The positive relationship between shrubs and HD indicates that as *T. canadensis* is removed, heavy shrub cover in both the sapling and seedling size classes may suppress regeneration of trees, hardwood or conifer. The specific role of *R. maximum* and density of *T. canadensis* was addressed by a study in the Smoky Mountains of North Carolina and Tennessee (Roberts et al. 2009). The authors found that *R. maximum* could inhibit tree regeneration and that response to *T. canadensis* decline was site-specific. Forest response to *T. canadensis* decline will be conditional on the amount of *T. canadensis* in a stand and the ability of tree seedlings and saplings to grow into the overstory and replace the canopy position of the *T. canadensis.*

Regardless of what trees replace *T. canadensis* and how fast this process occurs, stands with high *T. canadensis* BA and heavy hemlock decline will have changes in their stand composition and structure that may affect adjacent aquatic system function. If stands convert to hardwood, they may have a suite of changes to terrestrial and aquatic ecosystem dynamics. Hardwood litter has more available N and less lignin than conifer litter and is therefore of a higher quality for microbial decay (Fogel and Cromack 1977, Melillo et al. 1982, Van Cleve and Ericson 1993, Scott and Binkley 1997), which can lead to increases in decomposition of soil organic matter and increases in stream nutrient loads, with possible site capital losses of nutrients and cations (Jenkins et al. 1999, Yorks et al. 2000). Conifers generally have higher leaf area index than hardwoods (Teske and Thistle 2004). Hemlock decline and conversion to hardwood can lead to increased understory light levels (Eschtruth et al. 2006), increased stream temperatures, decreased winter transpiration, and increased storm discharge (Rogers 1980, Canham et al. 1994, Ellison et al. 2005, Ford and Vose 2006). The cumulative impact of degraded water quality in headwater systems on higher-order streams and rivers is of particular concern when the continental scale of HWA spread is considered. High quality water in forested headwater streams has the ability to dilute nutrient or temperature loads from agricultural, industry, and development downstream. This ability to dilute downstream pollutant loads may be reduced if headwater streams' pollutant loads increase because of conversion from *T. canadensis* to hardwoods or stand regeneration is stalled, particularly across large landscapes as our data suggest.

Management

The two components that our PCA identified are intuitive and help to organize our data into useful categories. The first component had strong loadings for variables that associate with location across the range of our study, such as ELEV and HEIGHT. The second component includes variables that are generally measures of stand density. This component shows that the TOTALBA, *TSCABA,* and %TSCA correspond to each other. We did not find an association between TRI and HD or clear evidence that stands on higher quality sites were less susceptible to decline, as others have at a statewide scale (Mayer et al. 2002), nor did we find that HD strongly associated with either component. Other site variables, such as foliar chemistry, may prove more helpful in addressing how *T. canadensis* responds to infestation by HWA (Pontius et al. 2006). A limitation of our analysis is the lack of information about the amount of time each stand has been infested with HWA. Some stands may be more resistant to the HWA. However, because our study stands were infested for varying amounts of time, it is difficult to determine whether stands that appear healthy are more resistant or have simply had HWA for less time. We will be able to address this issue further when we return to the sites and remeasure decline at year 5.

The ability of other conifer species to provide the ecosystem functions of *T. canadensis* is unknown. Some alternate conifers, such as *Picea* spp. or *Abies* spp., may have high enough leaf area index to provide the shade required to moderate stream temperatures and support shade tolerant understory species cohorts. Alternate conifers may also create acidic soil conditions, slow decomposition of soil organic matter, and reduce leaching of nutrients and site capital losses of nutrients and cations. However, there may also be some species-specific ecosystem functions that cannot be replaced by alternate conifers.

Land managers who have a goal of maintaining short-term aquatic ecosystem functions should maintain what conifers they do have in riparian management zones. In areas where *T. canadensis* cannot be saved, contingency plans should be crafted with a focus on replacing *T. canadensis* with other conifers. This is particularly true where land management goals are to protect water quality. Our study demonstrated that some conifers do coexist with *T. canadensis*. Species such as *P. strobus* are well represented and could perform some of the ecological functions conducted by *T. canadensis*. Additional efforts should address successful regeneration and restocking of riparian areas with seedlings and saplings. Intensive silvicultural strategies may be needed to release trees above the vigorous shrub layer that we found at many sites.

Conclusions

The positive side of our survey of *T. canadensis* health is that this species is well represented in both the seedling and sapling size class at many of our study sites. In addition, 24 of 49 sites do not have signs of HWA. In the stands that do have HWA, much of the overstory *T. canadensis* BA is still alive. This is good news for efforts addressing how to slow the spread of HWA, reduce the damage done to *T. canadensis*, or remove HWA entirely from the Appalachian Mountains. Although the insect has been in the region for more than 50 years, there are many places where *T. canadensis* is still thriving, and even in the stands that have been functionally replaced, we found live seedlings and saplings that would survive if the insect were brought under control. Though we focused this study on the decline of *T. canadensis* and found that many stands are on the verge of losing *T. canadensis*, the potential remains for the return of *T. canadensis* to these stands. However, serious challenges remain. If the spread and vigor of HWA continues, management will have to address the loss and replacement of *T. canadensis*. Intensive and expensive silvicultural strategies may be required to grow new cohorts of trees above vigorous shrub layers in the mid-Atlantic and southern states. Research addressing suitable replacements for *T. canadensis* under differing landowner goals will be needed. Additional research is needed addressing how privately owned forests have been affected by HWA, the magnitude of preemptive and salvage harvesting of *T. canadensis*, and the effects of harvesting infested stands on forest regeneration dynamics.

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