Title: Drivers of Spatial Heterogeneity in Hemlock Woolly Adelgid-Related Eastern Hemlock Declines in New York's Hudson Highlands Region

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Introduction:

Eastern hemlock (*Tsuga canadensis)* is a coniferous tree species that grows in temperate forests of eastern North America from the southern Appalachian Mountains northward into Canada and as far west as Wisconsin. It has been a common tree species across this range for much of the past ten millennia and is considered a keystone or foundation species because of the unique habitat it produces and its myriad of interspecies relationships (Ellison et al. 2005, Ellison et al. 2018). Eastern hemlock is a highly shade tolerant and long-lived tree species that is one of the only species across much of the eastern temperate forest capable of forming late-successional, self-perpetuating conifer stands; also able to grow in mixed stands with deciduous trees (Ellison et al. 2005). According to the United States (U.S.) National Vegetation Classification system, hemlock is considered the dominant or a significant common component of 14 forest associations which is more than any other individual tree species in the eastern United States (USNVC, 2023). The presence of hemlock trees can also create distinctive forest structural and microenvironment conditions which exert a strong influence over ecosystem processes and are important for biodiversity (Fassler et al. 2019). For example, the typically dense, evergreen canopies of mature hemlock trees help to keep stream temperatures cool, which is important for the survival of trout and other headwater stream species (Fassler et al. 2019). Hemlocks provide refuge from deep snow for deer and other animals and support various species of birds, salamanders, benthic invertebrates, and forest floor mesofauna (Ellison et al. 2005; Fassler et al. 2019). As is the case across much of North America, introduced species are altering the native landscape and posing threats to hemlock trees (amongst many other regionally native species) and the ecosystems they inhabit.

Hemlock Woolly Adelgid (*Adelges tsugae,* HWA*)* is an invasive aphid-like insect that is a major conservation concern for hemlock trees. Native to Japan and having coevolved with Japanese Hemlock (*Tsuga diversifolia),* HWA was first noted in western North America in the 1920s, where it does not pose a threat to Western Hemlock (*Tsuga heterophylla)* trees, the hemlock species native to that region. In 1951, HWA was reported on the east coast of the U.S. near Richmond, Virginia and has since spread to more than 19 states and much of the eastern hemlock's native range. Over the past 70 years, HWA has been causing decline of eastern hemlock and the closely related Carolina hemlock (*Tsuga caroliniana)* whose small endemic range in the southern Appalachian Mountains lies within that of the eastern hemlock. HWA reproduces parthenogenetically (without male fertilization) which contributes to its rapid spread. With two generations per year, the insects disperse by wind, birds, and human activity up to 50km per generation (Preisser et al. 2014; Lombardo, 2017). HWA is susceptible to cold winter

temperatures with population declines beginning at -25°C and full mortality of insect populations by -35°C, though warmer average winter temperatures and later seasonal frosts driven by climate change allow for HWA to remain viable later into the winter and continue their northward spread (Dukes et al., 2009). Both generations of HWA feed on hemlock needles from trees of all ages, cutting off the flow of nutrients and sugars from the needles to rest of the tree causing progressive needle loss and eventual mortality, typically within 10-20 years (Ellison et al. 2018). Hemlock mortality will change forest microclimate and temperature, increase light penetration to the forest floor, and alter nutrient cycling (Abella, 2014). It is projected that the ongoing loss of hemlock trees will reduce biodiversity and homogenize forest composition, completely changing the structure and function of forests across the Northeast (Ellison et al. 2005; Tingley et al. 2002). In forests where hemlock has already been completely eliminated by HWA, canopies have largely transitioned to native deciduous broadleaf trees such as black birch (*Betula lenta*) and red maple (*Acer rubrum)* and are increasingly colonized by opportunistic invasive plant species (Orwig et al. 1998; Ingwell et al. 2012; Zukswert et al. 2014). Forests with hemlock decline have been targeted for "salvage" logging and real estate development, which could further exacerbate the ecological impacts associated with HWA (Ellison et al. 2018).

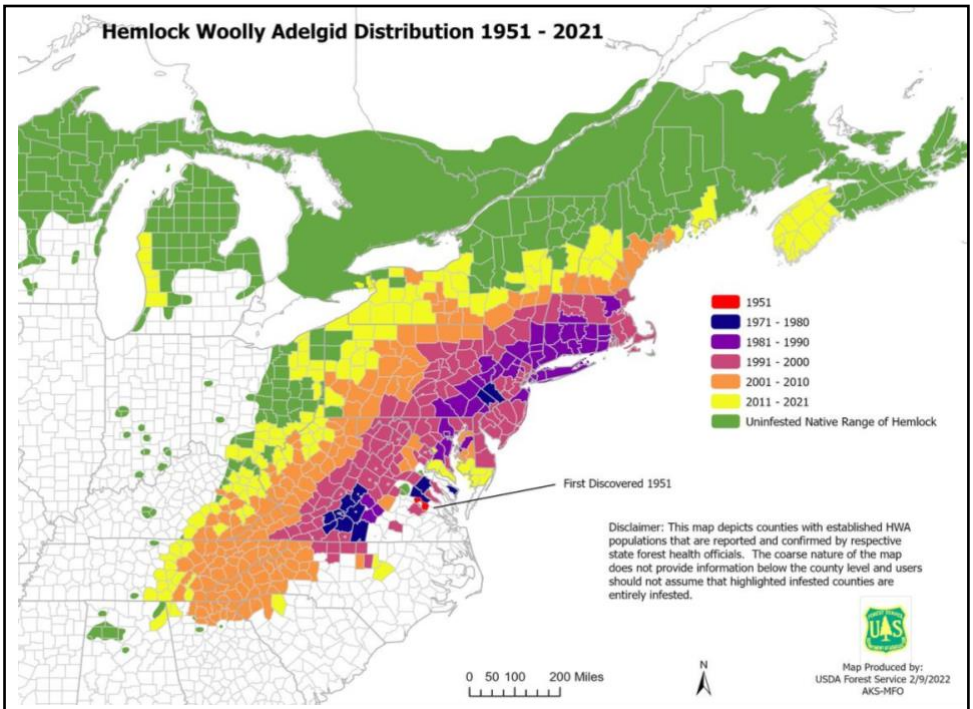

Figure 1: Current distribution of hemlock woolly adelgid across the hemlock's native range (USDA Forest Service, 2022)

As hemlocks have now been in decline for decades, some researchers have directed their interests towards site conditions and microclimate as potential drivers or inhibitors of HWA-related decline. Some studies have produced evidence that larger overstory hemlocks persist longer than smaller understory hemlocks after HWA infestation and that hemlocks in xeric environments decline faster than elsewhere (Orwig & Foster 1998, Orwig et al. 2002). Evidence from research in New England and the Delaware Water Gap suggests hemlocks on southern/western facing slopes tend to show significantly more decline than north-facing slopes across studies and that hemlock crown class may play a role in the rate of decline of trees (Vogt

et al. 2016). Some studies find no correlation between site characteristics and hemlock decline (Rentch et al. 2009). Most studies that investigate site drivers of hemlock decline have taken place in varying locations throughout the trees' native range at different parts of the timeline of HWA infestation, perhaps indicating that trends found as significant have also been subject to the unique landscapes of study.

Because hemlock decline and mortality typically occur within 20 years of HWA infestation, Eastern and Carolina hemlock's conservation status is near threatened and there is limited time to implement conservation measures. To date, forest managers have experimented with a wide range of control methods against the HWA with varying levels of success. Anecdotal evidence suggests that certain genetic lineages of eastern and Carolina hemlock are less susceptible to the HWA than others, motivating some practitioners to develop HWA-resistant variants (Oten et al. 2014). Various commercial insecticides have proven effective in controlling HWA on individual trees; though this method is typically costly, requires repeated applications, and creates other adverse ecological impacts making them impractical landscape-scale management tools (Frank et al. 2011). Mounting research suggest that biological controls such as HWA-eating insects can be effective control agents at forest and possibly landscape scales. In particular, introduced predators of the HWA have been shown to sustain their populations for long durations of time and reach remote areas beyond where they were initially introduced (Wiggins et al. 2016; Davis et al. 2012). *Laricobius nigrinus*, a small species of tooth-necked fungus beetle has already established populations following release in trials spanning from Georgia to Pennsylvania and two species of silver flies (*Leucopis argenticollis* and *L. piniperda*) that prey on adelgid in the northwestern U.S. have been in testing for introduction to forests in the eastern U.S. (Wiggins et al. 2016; Davis et al. 2012; Havill et al. 2018). Regardless, chemical and biological control methods are most effective when implemented relatively early in an infestation. Forest practitioners are faced with the ongoing challenges of tracking the spread of HWA to inform management decisions and for assessing the efficacy of management efforts at the stand, landscape, and regional scales.

Much of the difficulty in tracking the spread of HWA lies in the poorly navigable and rough terrain where hemlocks often grow, rendering traditional field surveys inefficient and costly. Remote sensing and geographic information systems technologies have the potential to aid practitioners in easily tracking the spread of HWA so as to implement controls in a timely manner. Remote sensing has never been easier than today with an abundance of freely available software, tools, and data repositories. Several studies have demonstrated the utility of multi-spectral satellite remote sensing and high-resolution hyperspectral airborne remote sensing for quantifying hemlock decline (Hanavan et al. 2015; Jones et al. 2015; Pontius et al. 2005). For a variety of reasons, many of these existing remote sensing models are of fairly limited applicability to practitioners; complex datasets or methods, high acquisition costs of proprietary information, and even the natural propensity of broadleaf trees to overtop hemlocks in forest

mixed stands may pose obstacles in translating this technology from the academic to practitioner setting. To help realize the potential of remote sensing for HWA monitoring and management, we have been in collaboration with practitioners from The Nature Conservancy and The Catskill Center to develop a free and user-friendly remote sensing model. Our model tracks decline in the normalized difference vegetation index (NDVI; metric of canopy chlorophyll content) from freely-available New York State airborne orthoimagery available through the Discover GIS Data NY portal (https://gis.ny.gov/orthoimagery). Notably, this imagery is collected during the early spring when broadleaf trees are leafless, so the modeled NDVI corresponds only to evergreen species, many of which are hemlock trees. The high native spatial resolution (<0.5m) of this aerial imagery provides opportunities to quantify decline at the tree-scale to find areas of declining hemlock trees, and the 4-band imagery (red, green, blue, and near-infrared) necessary for quantifying NDVI is available from 2009 onwards which enables tracking of the range of hemlock decline across New York State. While this model has proven effective in identifying conifers by means of NVDI and tracking their changes, we lack the tree-level data on temporal patterns of decline needed to properly assess how early in a hemlock's decline there is a measurable response of NDVI. Early detection is crucial for the timely implementation of management controls.

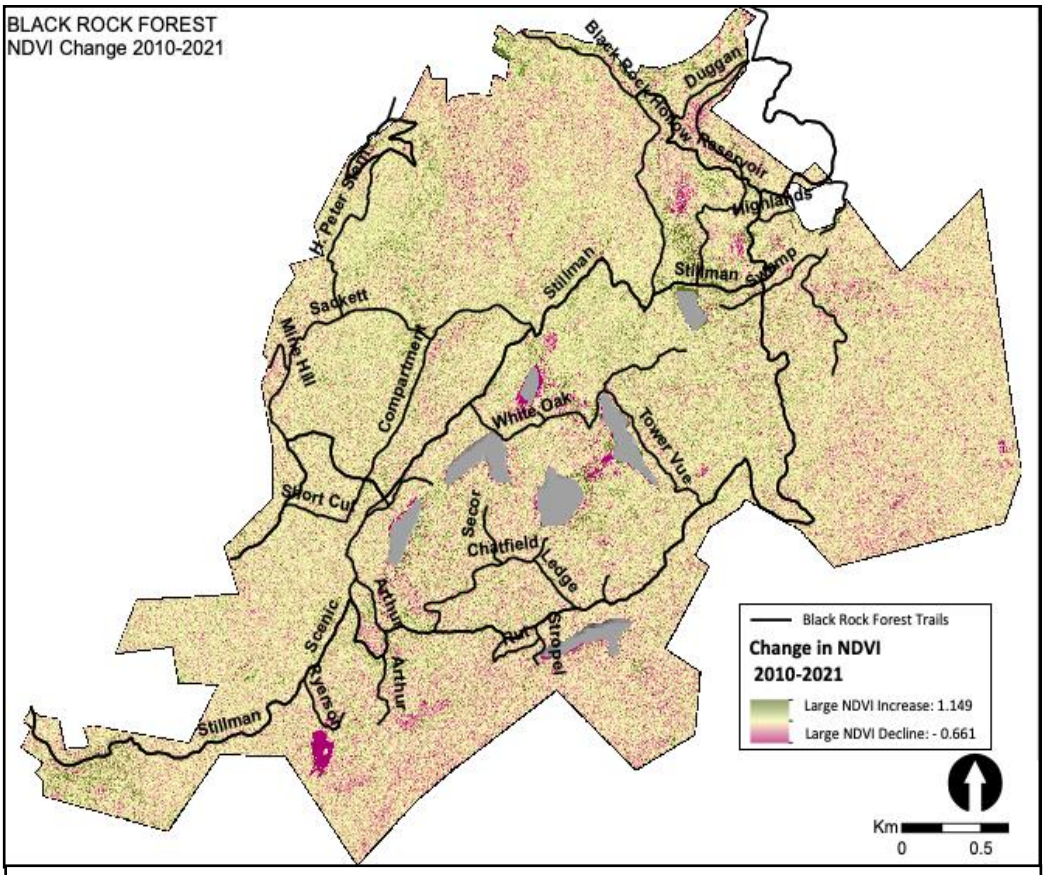

Figure 2: Map of NDVI change at Black Rock Forest, New York using Orange County orthoimagery data between the years 2010 – 2021.

To investigate the status of and potential drivers of hemlock decline on a local scale, we designed a study at a research forest in the Hudson Highlands region of New York. Our goals were to (1) further our understanding of temporal and spatial declines in hemlock health across

varying site conditions, and (2) use dendrochronological analyses to correlate tree-level declines with our remote sensing model to assess its efficacy as a tool for practitioners. We explored the roles of canopy class, basal area, and water availability as mediators of HWA-related hemlock decline; also building tree core extraction into our protocol to build chronologies of hemlock growth over time. Based on published literature that studies hemlock declines in nearby regions, we expect to find a more advanced state of hemlock decline in xeric than mesic environments, and hypothesize a positive correlation between hemlock size and health across the range of sites at BRF. Knowing that HWA reached the lower New York region in the mid 1990's, we also expect our core-based growth chronologies to evidence a decline in growth around that time.

Methods:

This study was conducted at Black Rock Forest (BRF), a 1,590-hectare research forest located in Cornwall, New York (41.4343° N, 74.0354° W) about 75 miles north of the New York metropolitan area. Canopy hemlock trees ranging from intermediate to dominant crown classifications (henceforth "focal trees") were selected across upland/lowland areas (Figure 3). To inform our selection, NDVI was modeled across the entirety of BRF using New York State orthoimagery data for Cornwall County (https://gis.ny.gov/orange-county-orthoimagery-downloads) available freely online and changes in NDVI were calculated between the years 2010 and 2021. Positive changes in NDVI over this period indicate growth of evergreen species, whereas negative changes in NDVI indicate declines in evergreen species (Figure 2). Areas of the forest that exhibited notable declines in NDVI were highlighted for field survey, which was aided by local knowledge of BRF staff regarding whether or not these areas in fact contained hemlock stands. 51 of the total 75 selected focal trees were cored to create hemlock growth chronologies. From each focal tree, two cores were extracted at 90° angles from one another so as to account

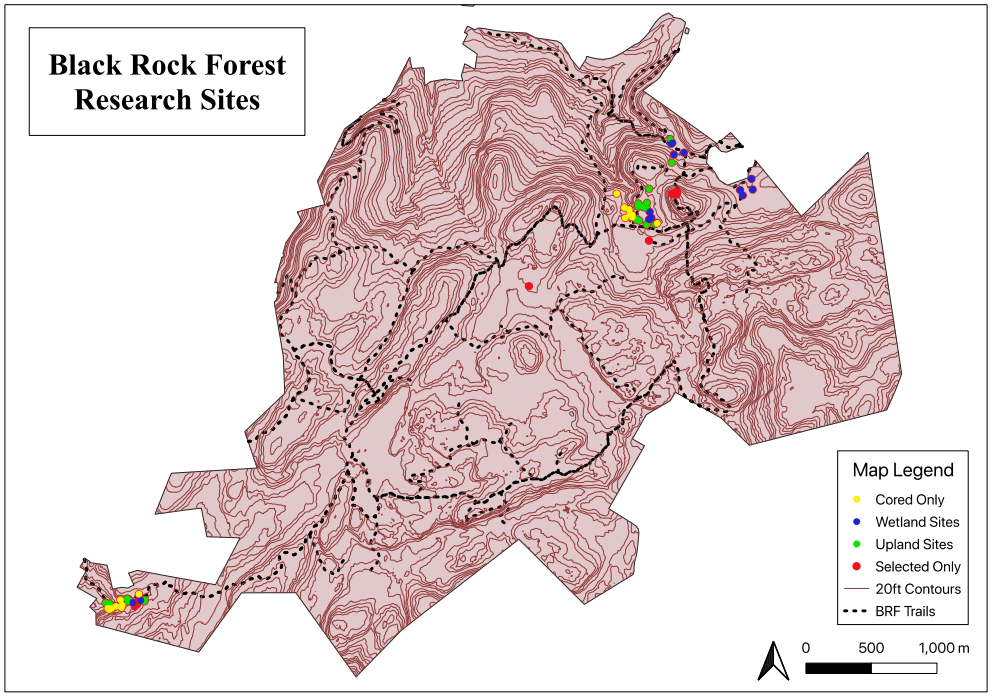

Figure 3: Study Sites Across Black Rock Forest. Sites are clustered nearby traversable paths and known areas of eastern hemlock presence as informed by our remote sensing model and Black Rock Forest Staff knowledge.

for the uneven radial growth of trees (resulting in 102 total cores to process). Site assessments were conducted at 34 of the focal trees that were cored.

Methods: Site Assessments

Within site assessments we identified each woody stem (>5cm DBH) to species that was within a 10-m radius of the focal tree and quantified their DBH and canopy position: overtopped (crown of tree is below the general level of the forest canopy), intermediate (crown of tree extends to the bottom of the forest canopy but the tree does not , co-dominant and dominant). Distance between our focal stem and every other within our plots was calculated with a hypsometer. The health of each hemlock tree contained within each plot was assessed by estimating live crown ratio (LCR) and crown foliar density on a 6-category classification system (0: Dead, 1: 1-20% LCR or density, 2: 21-40% LCR or density, 3: 41-60% LCR or density, 4: 61-80% LCR or density, 5: 81-100% LCR or density). Live crown ratio was defined as the percentage of a tree's total height with living branches. Foliar density refers to the percentage of those living branches with foliage, ranging from the typical density of a healthy hemlock tree to an entirely defoliated (dead) branch. An aggregated health score was calculated for each tree by averaging its respective LCR and density scores. LCR and crown foliage density estimates were made from ground level in the field by pairs or groups of researchers in efforts to eliminate biases. All further data analysis and tree core processing was undertaken at the CUNY Advanced Science Research Center located in New York City.

Methods: Tree Core Processing and Data Analysis

102 total tree cores were dried under a fume hood to reduce the likelihood of molding, glued into pre-routed wooden boards, and sanded using 80, 220, 400, and 600 level grits of sandpaper to obtain a smooth finish for ease of scanning and digitization. Cores were scanned using an Epson Expression 12000XL scanner for analysis in WinDendro software, (Regent Instruments, 2017). WinDendro is used for the precise annotation of tree growth, allowing us to create a growth chronology of our cores to the nearest tenth of a millimeter. As a quality control measure, tree core data were visually crossdated by cross referencing growth between these pairs of cores.

All data collected were analyzed using the Analysis Toolpak within Excel Version 16.54 (Microsoft, 2021). Total basal area of all stems and only hemlock stems was compared between upland and lowland sites – 20 sites were located in upland environments and 14 located in low-lying areas in close proximity to standing/flowing water. Total plot basal area, stem density, and focal tree DBH were compared to plot average LCR, crown density, and aggregate health score using linear regression analysis to determine if there was any correlation between the size of trees and the state of hemlock decline. We also used linear regression to test for correlation between our health metrics and hemlock trees that fell into three distinct canopy classes – dominant/codominant, intermediate, and overtopped. All linear regression analyses used a p value of $P=0.10$ for significance.

Results:

Total basal area of all stems and only hemlock stems was compared between upland and lowland sites – 20 sites were located in upland environments and 14 located in low-lying areas in close proximity to standing/flowing water. Total plot area ranged from 1.89m² to 0.74m² in upland environments, and from 1.67m² to 0.77m² in lowland environments (Figure 4). Average plot basal area across all upland sites was 1.38m², and 1.20m² for lowland sites. Hemlock basal area ranged from 1.08m² to 0.16m² across upland sites, and from 0.79m² to 0.19m² across lowland sites, averaging 0.51m² and 0.41m² respectively.

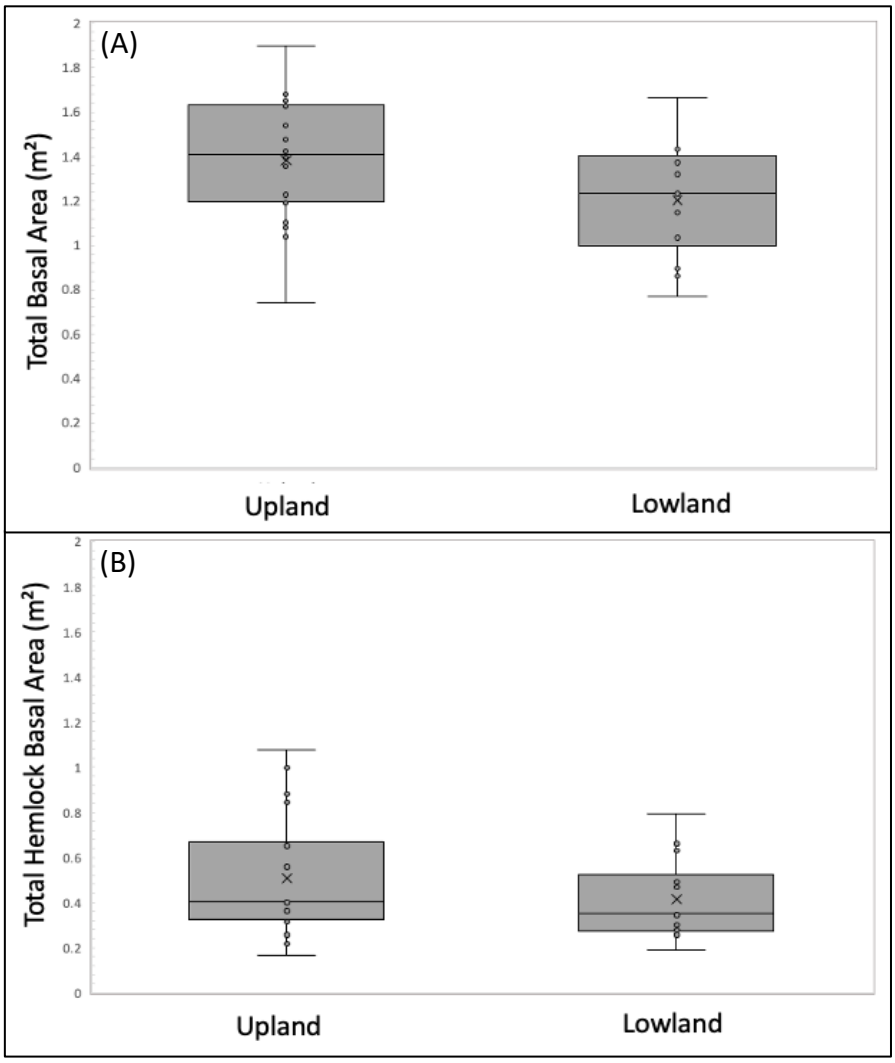

Figure 4: Comparison of total basal area of all stems >5 cm DBH per site between upland and lowland areas (A), and comparison of total hemlock basal area per site between upland/lowland areas of study (B).

We find a higher median plot basal area and more variation for our upland sites than the lowland sites (Figure 4). The same pattern exists when filtering for hemlock stems only.

We use total plot basal area as a means of estimating resource competition on the plot scale. Based on our linear regression there was no significant correlation between live crown ratio and plot basal area (p= 0.133; Figure 5). However, crown density and average health scores were significantly correlated with stand basal area (p=0.021 and p=0.013; Figure 5). However, these correlations are likely of limited ecological importance given the very low r^2 values (R^2 = 0.01 and R^2 = 8E-6; Figure 5). Linear regression analysis comparing stem density with live crown ratio, crown density, and aggregate health score determined no correlation, yielding p values of p=0.868, p=0.597, and p=0669 (Figure 6).

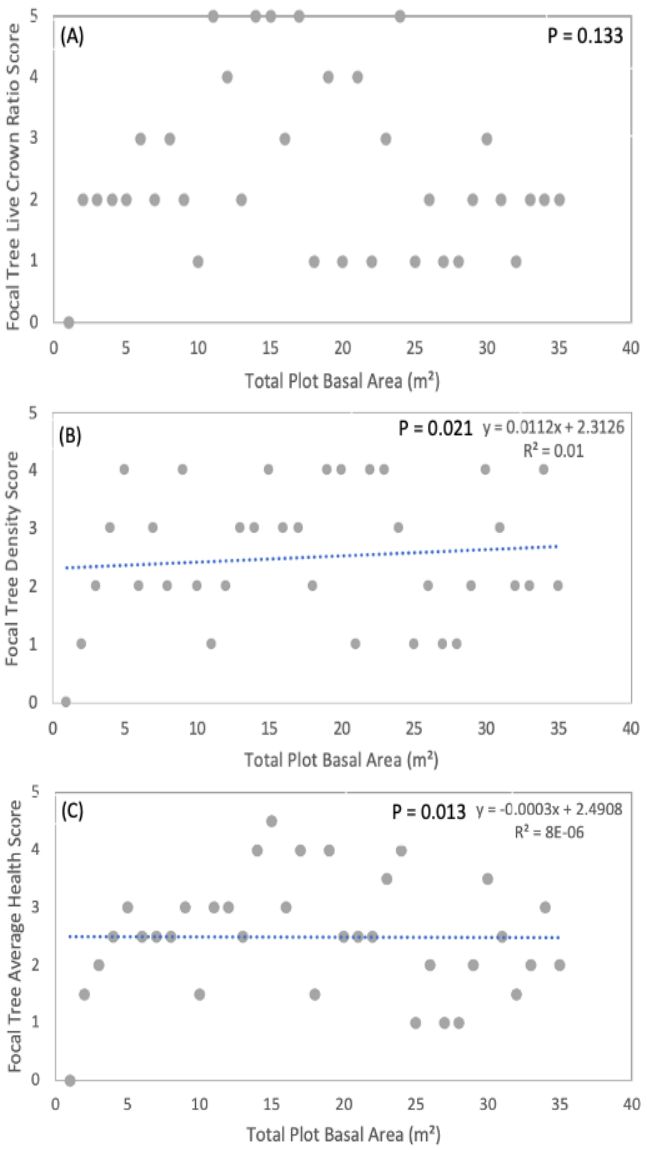

Figure 5: There is no significant correlation between focal tree live crown ratio and total plot basal area (A), though there is statistical correlation when comparing total plot basal area to focal tree density (B) and average health score (C).

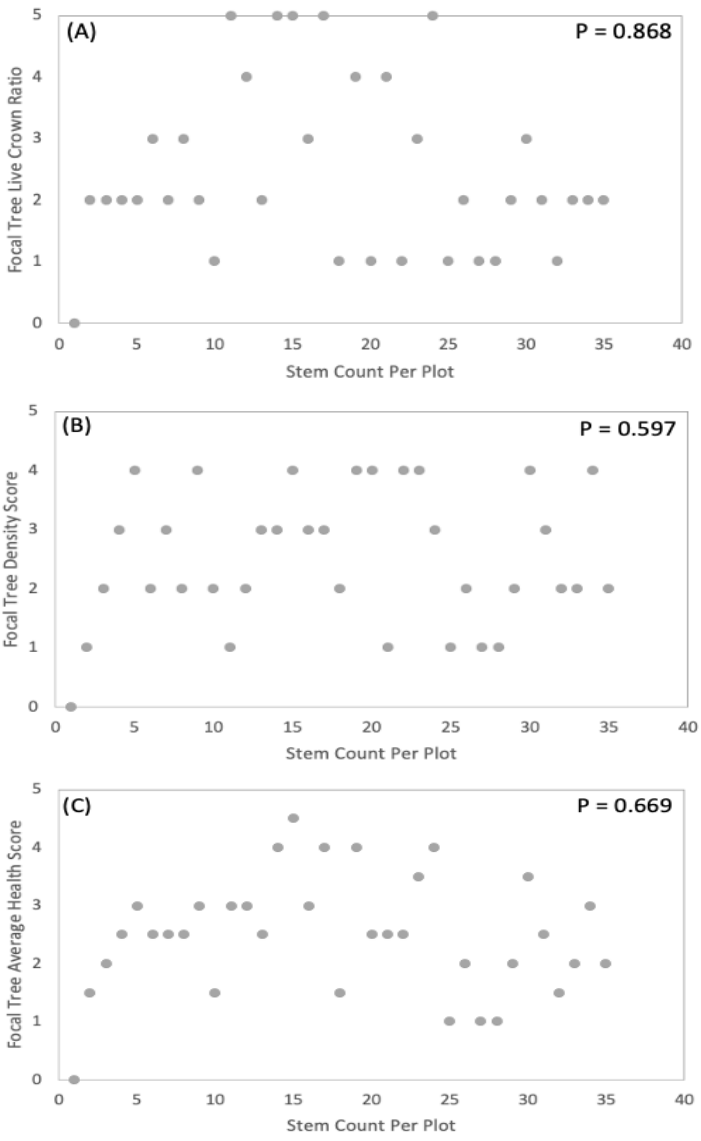

Figure 6: When comparing focal tree health metrics to their respective plot stem densities (A: Live Crown Ratio, B: Crown Density, C: Average Health Score), there is no significant correlation found.

We also compare our three metrics of hemlock health across three canopy classes; dominant/codominant (which have been grouped together here due to their similar canopy position), intermediate, and overtopped. Linear regression comparing live crown ratio to crown classes provides no significant correlation (p=0.322 for dominant/codominant trees, p=0.883 for intermediate trees, and p=0.919 for overtopped trees, Figure 7). Based on linear regression analyses there is also no correlation between crown density score and hemlock canopy class, yielding p=0.273 for dominant/codominant trees, p=0.303 for intermediate trees, and p=0.954 for

overtopped trees (Figure 8). Similar results are found when running linear regression analyses for aggregate health score and hemlock health metrics – comparing aggregate health to the dominant/codominant class yields p=0.195, comparing with the intermediate class yields p=0.699, and comparing with the overtopped class p=0.976 (Figure 9). Overall, we found no significant correlation when comparing all hemlock health metrics against hemlock canopy class. When comparing these health metrics across canopy classes, we also looked at the variation within our data by measuring coefficients of variation (standard deviation divided by mean) for each distinct canopy class tree dataset (dominant and codominant are separated in this analysis, Figure 10). We find that the dominant canopy class exhibits the most variation within live crown ratio scores. For density and average health scores, variation increases as one decreases in canopy class size from the codominant strata to the overtopped strata. The intermediate canopy class had the highest average live crown ratio score (>2.5), whereas the dominant class had the highest foliage density and aggregate health scores (>2.5, Figure 10).

When comparing aggregate health score between upland and lowland plots, upland site health scores ranged from 1 to 4 whereas lowland site aggregate health scores ranged from 1.5 to 4.5 (Figure 11). We also find a higher median and mean aggregate health score in lowland plots than upland plots (Figure 11).

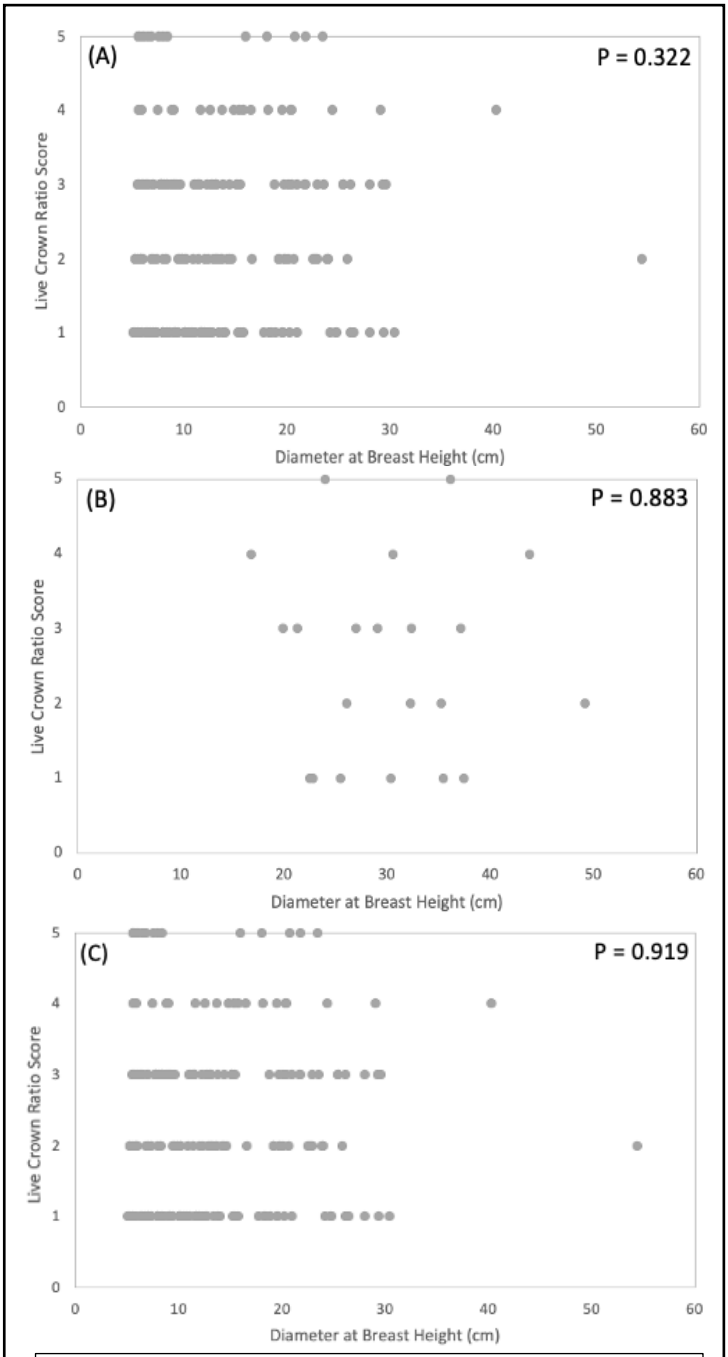

Figure 7: Linear regression analyses comparing individual hemlock live crown ratio scores with diameter at breast height, stratified across three canopy classes: Dominant/codominant (p=0.322), intermediate (p=0.883), and overtopped (p=0.919).

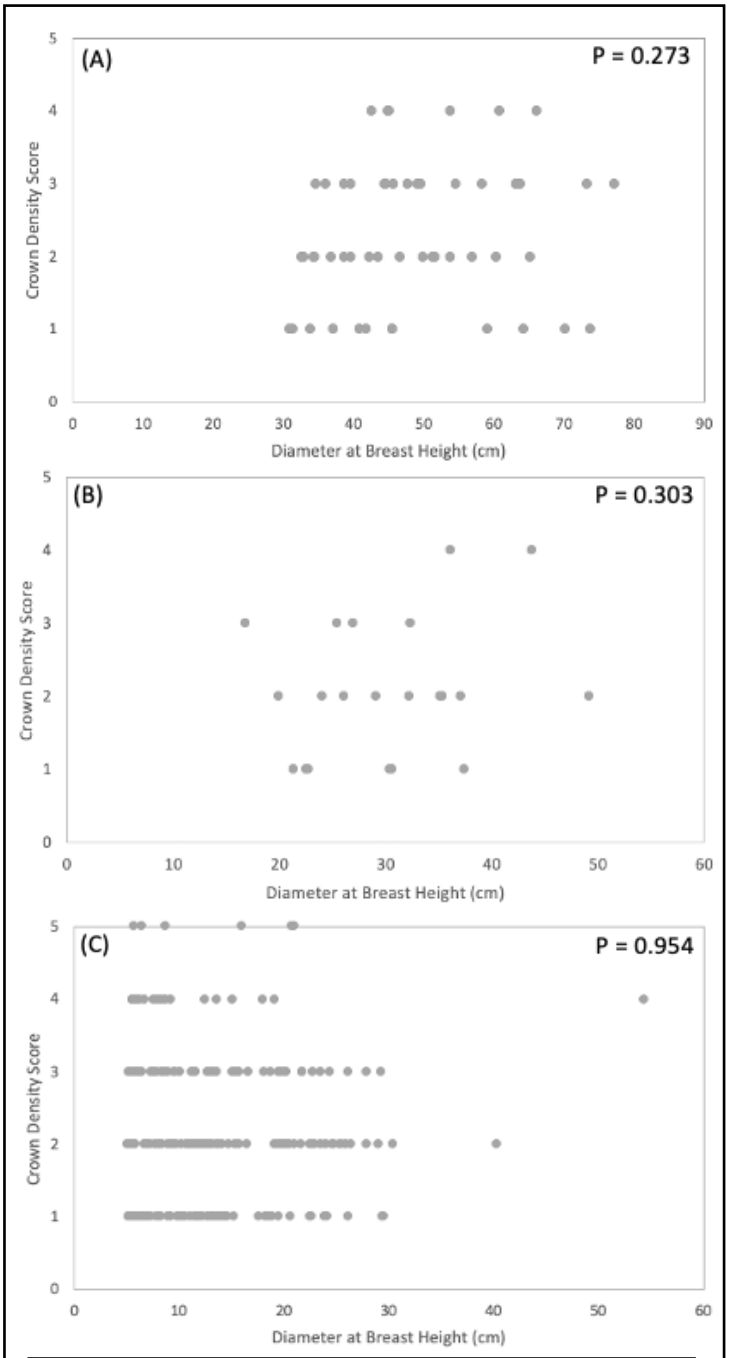

Figure 8: Linear regression analyses comparing individual hemlock crown density scores with diameter at breast height, stratified across three canopy classes: Dominant/codominant (p=0.273), intermediate (p=0.303), and overtopped (p=0.954).

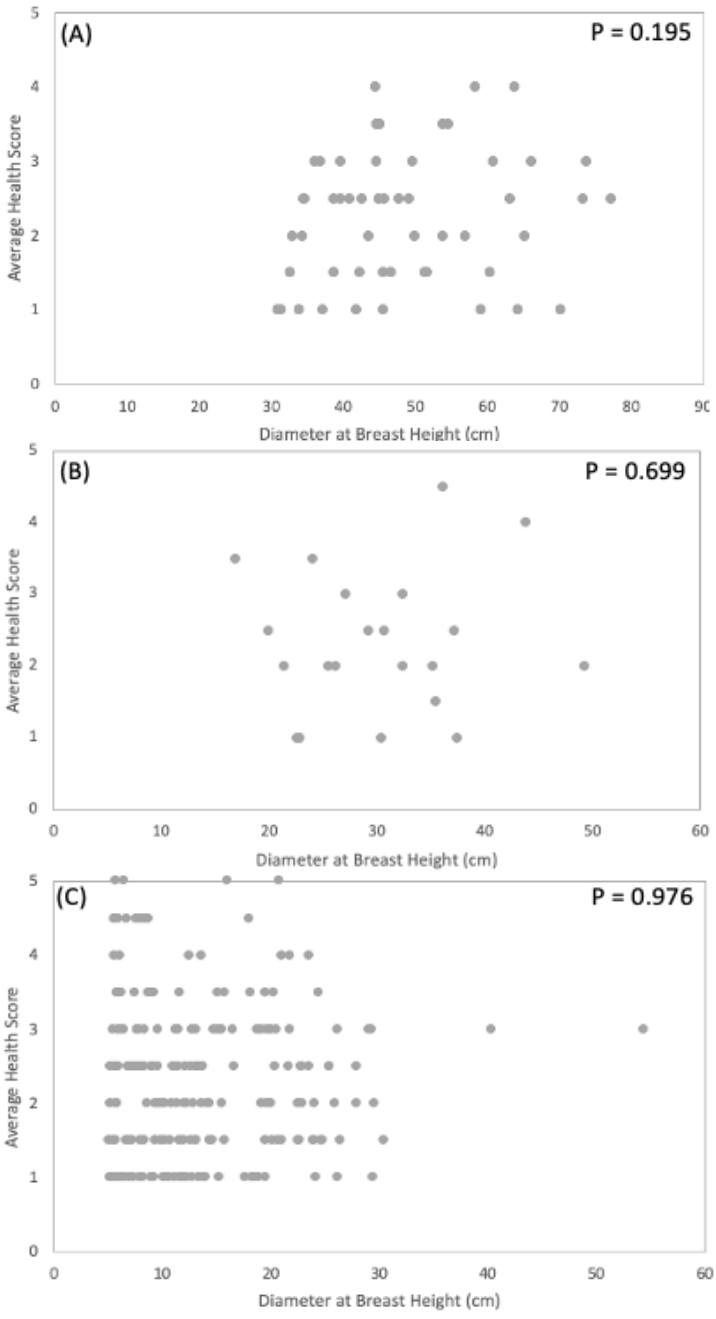

Figure 9: Linear regression analyses comparing aggregate health scores (averaged LCR and density) with diameter at breast height, stratified across three canopy classes: Dominant/codominant (p=0.195), intermediate (p=0.699), and overtopped (p=0.976).

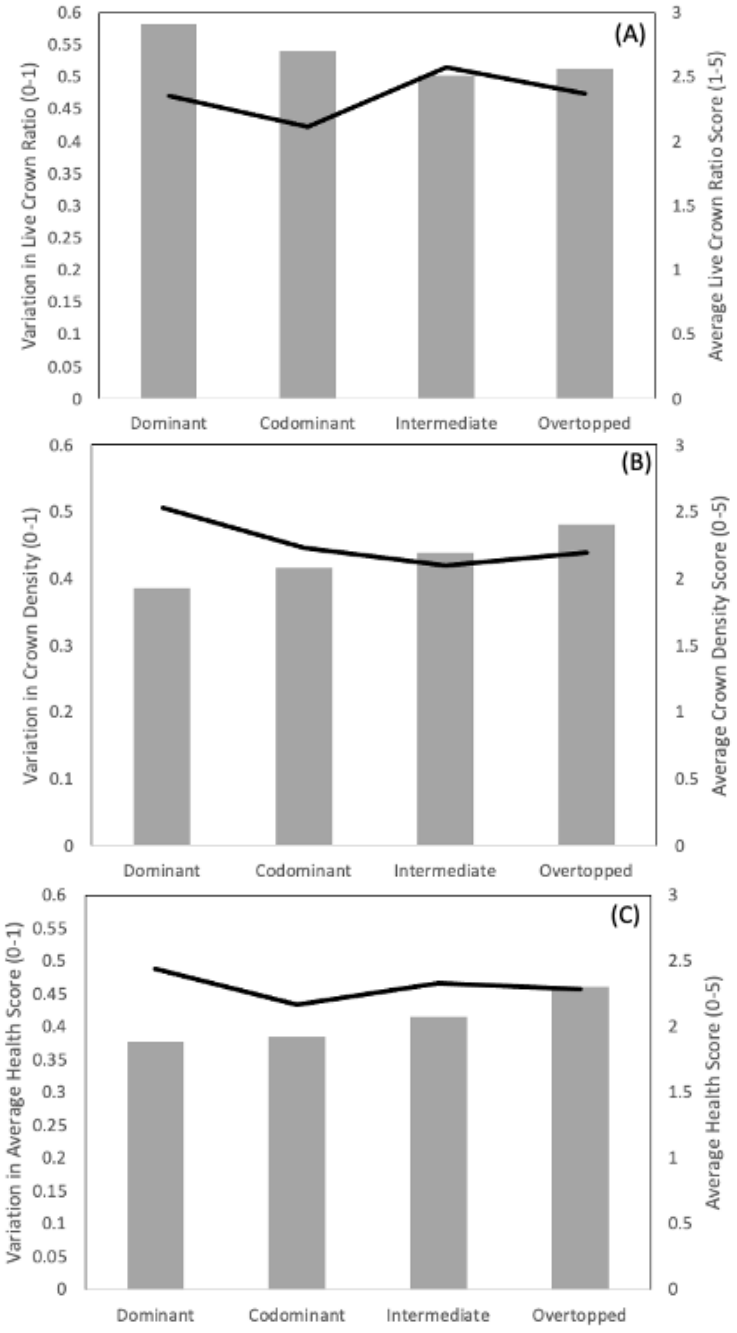

Figure 10: When comparing variation of health metrics within canopy classes (gray bars), we find that the dominant class of hemlock trees exhibits the most variation in live crown ratio, while the overtopped class exhibits the most variation in density and average health scores. Health class scores (black lines) are superimposed over variation.

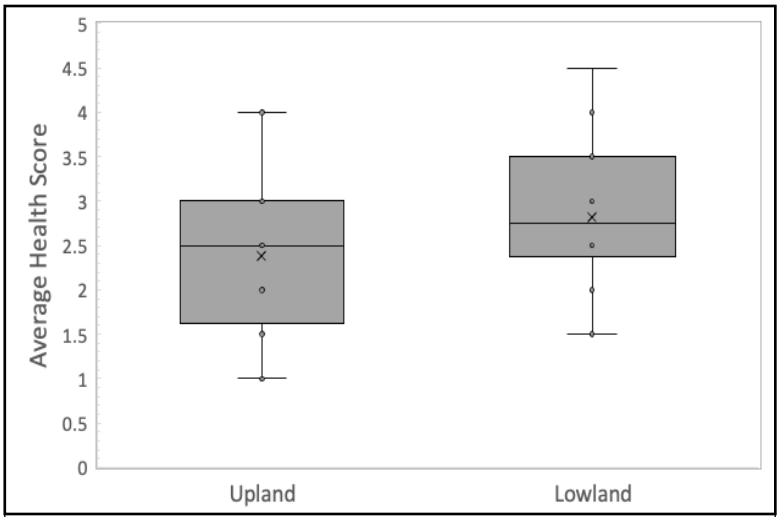

Figure 11: Box and whisker plots of aggregate (average) health score (density and live crown ratios) between upland and lowland sites.

Beyond analyzing stand characteristics and their correlation to hemlock health, we also have evidence regarding trends in temporal tree decline looking back in time based on preliminary growth chronologies assembled from our tree cores.

We see a general decline in growth beginning around the mid 1990's (concurrent with the northward spread of HWA into the New York/New England region, Figure 1), though some trees exhibit a gradual decline following infestation (e.g., focal tree 51, Figure 12) while others show a rapid decline following infestation (i.e focal tree 10, Figure 12).

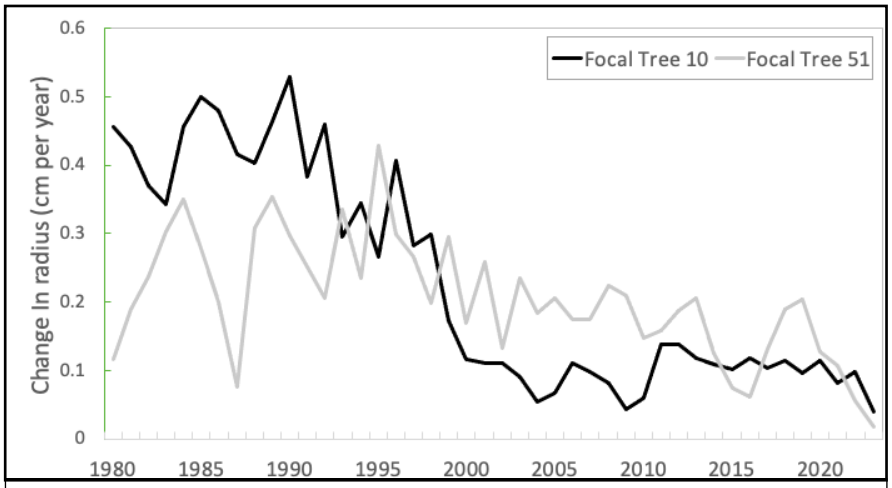

Figure 12: Example growth chronologies of selected cores. Each chronology is based on an average of yearly radial growth between A/B cores for each respective individual tree.

Discussion: Assuming that all hemlocks at BRF were infested with HWA at relatively the same time (Figure 1), some of our data points towards confirming findings from the canon of hemlock decline literature that hemlocks in xeric environments decline at a faster rate than their counterparts in more mesic or wet environments (Orwig & Foster 1998, Orwig et al. 2002, Vogt et al. 2016). We find an overall higher aggregate health score of hemlocks in lowland areas, despite their lower proportion of basal area (Figures 4, 11). These patterns may be due to higher resource competition associated with higher basal area at upland plots, lower water availability at upland plots, or a combination of both.

Though we did not find any significance between crown class and hemlock health using linear regression analyses, more general trends within our data are in line with literature that suggests a weak association between canopy class (and furthermore tree size) and state of health. Studies

have found that trees in lower canopy classes have less vigor and decline faster than those in more dominant positions (Orwig & Foster 1998, Orwig et al. 2002), which our data partially agrees with as the dominant class has higher average foliar density and aggregate health scores than all other classes (Figure 11). Trees that fall below the dominant and codominant classes receive no direct light from overhead, have smaller root systems, and are under a higher degree of resource competition than their dominant counterparts, which may explain these trends (Vogt et al. 2016).

It is worth noting our relatively small dataset (n=34) and singular type of correlation analysis (linear regression), whereas much of the published literature uses more robust data sets and complex mortality modeling (Vogt et al. 2016). Our data also only represents a snapshot of hemlock decline in the year 2023; a longer sample period would provide the opportunity to look at time as a function of decline and to determine varying rates of decline. Many studies point towards slope and aspect as drivers of hemlock decline, finding that steeper slopes and south/west aspects are associated with a higher level of decline (Vogt et al. 2016). Other studies find no correlation between such slope and aspect on decline (Rentch et al. 2009). Adding slope and aspect to our protocol could provide meaningful data, especially considering the variety of steep-sloped environments and range of tree aspects across the mountainous Hudson Highlands where our study takes place.

As it stands today, our work provides a detailed assessment of hemlock health across BRF and may inform management decisions for practitioners by identifying where healthier stands of hemlock exist. We plan to build upon this work by comparing our tree core growth chronologies with our remote sensing model to determine how we can assess temporal declines from freely available orthoimagery. If our model is successful at predicting areas of high hemlock decline across BRF, we can further extrapolate it across New York State thus providing a useful tool across a much larger range of hemlock decline – HWA has not been detected in the most northen parts of New York State (Figure 1). There is also the possibility of gathering more field data such as slope, aspect and canopy light to bolster our existing analyses and extending our field season another year to investigate trends in hemlock decline over time. Our current data analyses will also be expanded upon using more statistical tests of correlation and variance.

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