



Forest Dynamics in National Parks in the Eastern Rivers and Mountains Network

Natural Resource Report NPS/ERMN/NRR—2016/1182



ON THE COVER

Photograph of forested mountains in Gauley River National Recreation Area.
Photograph credit: Doug Manning

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Executive Summary

Understanding both the current condition of forest resources and how forests are changing over time is critical to sound management of parks' terrestrial ecosystems. In 2014, the Eastern Rivers and Mountains Network (ERMN) of the National Park Service (NPS) Inventory and Monitoring Program concluded its eighth season of monitoring forests in eight national parks, sampling nearly 350 plots twice over the eight years. Based on analysis of monitoring data from these plots, this report offers the latest answers to the following question and provides park-specific management recommendations developed from these findings.

“How are park forests changing over time in relation to weather, climate, landscape dynamics, invasive species, deer browse, and natural processes such as gap formation and succession?”

We compare tree mortality, recruitment and growth across the eight ERMN parks and discuss the implications for forest health. Species-specific trends in dominance, growth and mortality are presented for each park.



Photo 1. Emerald ash borer. Credit: David Cappaert, Michigan State University, Bugwood.org

Allegheny Portage Railroad National Historic Site (ALPO)

Forest composition in ALPO appears relatively stable, since the species composition of the seedling layer matches the species composition in the forest canopy. ALPO's forests exhibited moderate growth, typical of a middle-aged stands. More trees died than were recruited into the canopy, as smaller trees were out-competed by the larger canopy trees. However, the overall increase in tree basal area indicates that the forest is maturing. Red oak appears to be in good condition in the park, with a strong growth rate, solid recruitment, and no observed mortality. In contrast, white ash experienced >2% mortality as a result of the emerald ash borer, even though living trees still show a

high average growth rate. Most of the park's ash trees will likely die in the next few years, such that hazard trees in high visitor use areas should be assessed. Hemlock health is surprisingly stable, despite hemlock woolly adelgid being present in the park for several years. Hemlock mortality equals recruitment into the canopy and the growth rate for hemlocks is typical of healthy trees. Monitoring the effects of hemlock insecticide treatments and the release of biological control insects should continue.

Bluestone National Scenic River (BLUE)

Compared to other network parks, forests in BLUE have high tree mortality, low tree recruitment into the canopy, and the slowest tree growth. These forests are under stress due to a drier, warmer climate and droughty soils that create harsher growing conditions compared to other network parks. Since the park's trees are more stressed, they are more susceptible to additional impacts such as pests, pathogens, drought, and climate change. Park managers should anticipate that summer droughts and insects such as hemlock woolly adelgid, gypsy moth, and emerald ash borer may cause greater or quicker damage in BLUE than in the other network parks.



Photo 2. ERMN vegetation crew navigates rocky terrain in BLUE's mesic forest. Credit: John Wiley

In BLUE's ridgetops and upper slopes, the species composition of the forest is shifting away from oaks, hemlock, and pine, which are being replaced by red maple. The increase in red maple in dry oak-dominated forests is a region-wide trend, influenced by fire suppression in the 20th and 21st centuries and limited light availability under the closed canopy, but possibly also as a result of a land-use change, the loss of American chestnut, and browse from increased white-tailed deer populations.

Forests on BLUE's lower slopes and floodplain are diverse, containing many different tree species. The seedling layer in these forests is strongly dominated by ash, since trees infested with emerald ash borer produce abundant seeds in response to stress. These seedlings are not expected to survive to join the canopy unless a biological control is discovered for the emerald ash borer. Sugar maple, white oak, and red maple are the next most abundant tree seedlings following ash.

Delaware Water Gap National Recreation Area (DEWA)

Of the ERMN parks, DEWA contains the greatest diversity of vegetation types, including successional habitats, and mesic and xeric forests. Fast-growing, young trees are establishing in the open successional habitats, which causes the park-wide tree growth and recruitment rates to be among the highest in the network.

In mesic habitats, canopy composition appears stable, since the species with solid recruitment and growth rates are the dominant trees, including American beech, red maple, red oak, white pine, black cherry, and tuliptree. Hemlock and American elm are declining in these habitats, while muscledwood showed strong recruitment into the subcanopy. Although ash seedlings are abundant, they will not survive to canopy height unless a biological control for emerald ash borer is discovered. Emerald ash borer has not been detected in the counties that contain DEWA, however, it will likely invade the park in the next few years. Park managers should be on the lookout for signs of borer infestation and ash decline (see

<http://www.emeraldashborer.info/>), since the insect may already occur in the park but has gone undetected.

In drier forests, common canopy trees such as species in the red oak group, pignut hickory, and black birch are relatively healthy. Declining species include chestnut oak, hemlock, sassafras, pitch pine, mockernut hickory, and particularly white oak. Strong declines in white oak were likely caused by a combination of gypsy moth defoliation and droughty conditions in their preferred ridgetop habitats. Regionally, white oak has experienced



Photo 3. Red maple seedling. Credit: Doug Manning

strong declines with little recruitment over the past two centuries, as intensive land-clearing followed by fire suppression has had greater negative impact on white oak, compared with other oak species. In the coming years, red maple will likely increase in these forests. This trend has been documented region-wide, influenced by fire suppression in the 20th and 21st centuries and limited light availability under the closed canopy, but possibly also as a result of a wetter climate, land-use change, the loss of American chestnut, and browse from increased white-tailed deer populations.

Park managers should be vigilant for thousand cankers disease on black walnut, especially along the floodplain of the Delaware River in the Black Walnut Bottomland Forest. This disease complex has been found in southeastern Pennsylvania, not far from DEWA's southern boundary. See <http://www.thousandcankers.com/home.php> for additional information and resources.

Fort Necessity National Battlefield (FONE)

In general, forests in FONE experienced moderate growth, recruitment, and mortality compared to other network parks. However, major shifts in canopy composition may be expected in the future. Although oak trees comprise 25% of the canopy, oak seedlings make up only 4% of the seedling layer, which is instead strongly dominated by red maple, black cherry, and white ash. Park managers may consider the strategic use of prescribed fire and deer exclosure fencing to promote oak regeneration, especially if disturbances such as storms or pest-induced tree mortality create gaps in the forest canopy. Currently, white ash trees exhibited the fastest growth rate, however, most of the park's ash trees will likely die in the next few years from emerald ash borer infestation. Hazard ash trees in high visitor use areas should be assessed.

Friendship Hill National Historic Site (FRHI)

Much of FRHI is characterized by young forests that have established on former agricultural land or land that was timbered in the mid-1900s. These young trees are growing rapidly on rich mesic soils, giving the park the fastest growth rate in the network. These young trees are also competing with each other for a place in the future canopy, leading to a high mortality rate as some trees are out-competed. The current canopy composition, dominated by tuliptree, red maple, black cherry, sugar maple, and boxelder, is relatively stable. Pin oak showed very strong growth as a result of some trees established in the former agricultural land growing in full sun, on fertile soil, and with limited competition. Most of the park's ash trees will likely die in the next few years from emerald ash borer infestation, so hazard ash trees in high visitor use areas should be assessed.

Invasive plant species are one of the most pressing threats to forest health in FRHI. As discussed in previous monitoring reports, FRHI contains the most invasive plant species (as measured by total cover and number of invasive plant species per plot) of any network park. Without concerted management efforts, invasive plant species will continue to expand within the park, further degrading forest health.

Gauley River National Recreation Area (GARI)

In the summer of 2012, severe storms caused widespread damage throughout GARI, toppling so many trees that the standing living tree basal area decreased slightly (-0.14%) in the park. Gaps in the forest canopy created by wind storms are an important natural disturbance in eastern forests

increasing light to the forest floor and promoting seedling growth. However, tree-of-heaven, a non-native invasive tree, is a serious threat to GARI's storm-damaged forests. This prolific, fast-growing tree thrives in canopy gaps and can quickly form extensive stands, out-competing native trees. Park managers should consider using the strain of *Verticillium* wilt specific to tree-of-heaven as a biological control to contain this species.

In GARI's xeric forests, hemlock, tuliptree, a variety of oaks, and red maple dominate the canopy. Although oak recruitment into canopy was lacking in all species except red oak, it is notable that chestnut oak and white oak show similar relative abundance in the canopy and seedling layers. Abundant oak seedlings are crucial to securing oak reproduction in the storm-damaged canopy gaps.

In GARI's mesic habitats, sugar maple and red maple will likely continue to dominate in the canopy. Black birch, tuliptree and basswood are currently abundant in the canopy but will not reproduce in closed canopy forests. These species should thrive in storm-damaged canopy gaps if invasive species are kept in check. Ash and paw paw were more abundant in the seedling layer than in the canopy, but neither of these species is expected to reach the canopy due to emerald ash borer infestation and the species' growth form as a small tree, respectively.



Photo 4. Tree-of-heaven establishing in a canopy gap in GARI. Credit: Doug Manning

Johnstown Flood National Memorial (JOFL)

The young forests at JOFL are dominated by red maple and black cherry, both of which have average growth and relatively stable mortality and recruitment of canopy trees. While red maple seedlings are abundant, black cherry seedlings are scarce, suggesting that black cherry may not dominate in the future canopy without disturbance. As the forest matures, early-successional species such as black locust are declining, due to shading by the canopy trees. Though white pines are not abundant in the canopy, this species is the fastest growing in the park. Ash seedlings are much more abundant than ash canopy trees, though these seedlings will likely not survive to maturity unless a biological control is discovered for the emerald ash borer. Most of the park's ash trees will likely die in the next few years, such that hazard trees in high visitor use areas should be assessed.



Photo 5. Adult viburnum leaf beetle on a viburnum fruit.
Credit: Paul Weston

During forest monitoring in JOFL, two other important changes in forest health were noted. In 2010, severe dieback on spicebush (*Lindera benzoin*) was observed in the park. Samples were collected and analyzed at Penn State University to investigate possible pest and pathogens that might be responsible for the dieback, however, none were found. Spicebush has since begun to recover in the park with many individuals sprouting from the bases of old stems. The same year, viburnum leaf beetle (*Pyrrhalta viburni*) was discovered in the park and has been responsible for severe decline in several viburnum shrub species (*Viburnum* spp.). Viburnum leaf beetle is a non-native, invasive insect that eventually kills many viburnum species that it feeds on. For more information on viburnum leaf beetle, see <http://www.hort.cornell.edu/vlb/index.html>.

New River Gorge National River (NERI)

Compared to other network parks, the forests in NERI show moderate growth, recruitment and mortality. Some notable species-specific dynamics are occurring in both xeric and mesic habitats.

In NERI's xeric forests, the canopy species composition is shifting as the forest matures. Early successional species such as sassafras and black locust are being out-competed by the mature oaks and hickories. Ash also showed elevated mortality, due to the emerald ash borer infestations. Oak species experienced strong growth; however, oak seedlings are severely underrepresented when compared with their abundance in the canopy. Interestingly, chestnut oak seedlings are as relatively abundant as chestnut oak trees, probably as a result of mast events that produced numerous germinating acorns. Red maple seedlings disproportionately dominate the seedling layer, following the region-wide trend of increasing red maple in dry oak-dominated forests, influenced by fire suppression in the 20th and 21st centuries and limited light availability under the closed canopy.



Photo 6. Xeric oak-hickory forest with a healthy lone hemlock tree.
Credit: Doug Manning

Hemlock showed solid recruitment, no observed mortality, and average growth, suggesting that hemlocks are persisting in the park despite the presence of hemlock woolly adelgid for numerous years. Monitoring the effects of hemlock insecticide treatments and the release of biological control insects in hemlock-dominated stands should continue since successive years of drought and / or warm winters could cause significant stress to the hemlock trees.

The canopies of mesic forests are diverse. Without major disturbances, the future canopy will likely continue to be dominated by basswood, sugar maple, red oak, and black cherry, with less tulip tree and more red maple. However, the non-native exotic species tree-of-heaven has the highest recruitment and the fastest growth rate of any species present in NERI's mesic forest. This tree poses a serious threat to the park's forest, especially in disturbed stands, such as

those damaged by storms or in canopy gaps left by dying ash trees. Park managers should consider using the strain of *Verticillium* wilt specific to tree-of-heaven as a biological control to contain this species. Ash trees in NERI had the highest mortality rate of any species in any of the network parks, a result of the longer period of infestation by emerald ash borer compared to the other parks. Ash seedlings are abundant in mesic habitats, since dying ash trees produce abundant seeds in response to stress. These seedlings are unlikely to survive to maturity unless a biological control for the borer is discovered. Park managers should be on the lookout for any surviving ash trees that could assist the US Forest Service's efforts to identify ash trees that may be tolerant or resistant to emerald ash borer (see <http://www.fs.fed.us/nrs/tools/ash/>).

Predictors of Tree Growth and Mortality

To investigate the factors that influence tree growth and mortality, we used stepwise model selection procedure to model tree growth and mortality as a function of stand and soil variables. The strongest influence on tree growth was stand basal area, which was observed consistently across parks, and soil horizons. As stand basal area increases, tree growth decreases, driven by competition among trees for resources in increasingly crowded stands. No additional relationships were observed between tree growth and soil chemical variables in organic horizon soils. In mineral soils, calcium to aluminum ratio, base saturation, and carbon to nitrogen ratio were significant predictors of tree growth, though the relationships were not consistent across all parks. Fewer significant models were generated for tree mortality. Using the subset of plots from which organic horizon soils were collected, significant models were identified, although the predictors varied among the West Virginia parks (base saturation and pH) and the western Pennsylvania parks (carbon to nitrogen ratio and percent dead trees).

Acknowledgments

We are grateful for the efforts of numerous individuals who have contributed to the overall success of the forest health monitoring program and have made this report possible. We thank park superintendents, resource managers, and staff who have provided valuable logistic support to field crews. We also thank the many field crew members and botanists for their hard work and dedication to this program. We extend our thanks to the staff at numerous other Inventory and Monitoring Networks for their continued collaboration on collecting and analyzing forest monitoring data.

Introduction

Forests are dynamic ecosystems that can serve as an integrated measure of terrestrial ecosystem health by expressing information about climate, soils, and disturbance. Understanding both the current condition of forest resources and how forests are changing over time is critical to sound management of parks' terrestrial ecosystems. Both ecological and anthropogenic forces shape the forests of the eastern United States. Ecological factors (e.g., geology, topography, soil nutrient availability, weather, and disturbance patterns) and anthropogenic stressors (e.g., harvest and land-use conversion) can act independently or in concert to directly influence the structure, composition, and dynamics of forest vegetation. Many changes in forest vegetation are linked to several interacting ecological and anthropogenic factors, such as exotic invasive species, white-tailed deer (*Odocoileus virginianus*), acid and nutrient deposition, climate change, and altered disturbance regimes.

By measuring taxonomic, structural, and demographic features, certain parameters of forests can be assessed to determine whether or not the ecosystem's parameters fall within expected or accepted norms and ranges of variability (Karr and Dudley 1981). These measures serve as indicators of ecological integrity that can be explicitly linked to forest management (Tierney et al. 2009).

In particular, tree recruitment and mortality rates, as well as annual tree growth, are important indicators of forest health and vitality. Tree growth rates can decline in response to environmental factors or anthropogenic stress, and tree mortality is often preceded by some years of reduced tree growth (Ward and Stephens 1997, Pedersen 1998, Dobbertin 2005). Decreased growth or elevated mortality rates in trees of a particular species can indicate a health problem for that species (Hyink and Zedeker 1987, Duchesne et al. 2003), while altered vital rates for multiple species across a region may indicate a regional environmental stress, such as acid deposition (Steinman 2004, Dobbertin 2005).

Across fifty national park units throughout the eastern United States, the National Park Service (NPS) Inventory and Monitoring (I&M) Program has established more than 2,000 long-term forest health monitoring plots that provide a unique perspective into the forest dynamics of these protected areas. As part of this nationwide NPS I&M Program (Fancy et al. 2009), the Eastern Rivers and Mountains Network (ERMN) developed its long-term monitoring program for forests (Perles et al. 2014a) in response to the identification of vegetation communities and soils as high-priority vital signs for the network (Marshall and Piekielek 2007). The objective of the ERMN Vegetation and Soil Monitoring Program is to provide information regarding the condition of forest and soil resources in the parks, and how their condition is changing through time. One of the founding questions that guide the monitoring program is:

“How are vegetation composition, structure, and demography changing over time in relation to weather, climate, landscape dynamics, invasive species, deer browse, and natural processes such as gap formation and succession?”

Based on analysis of the most current ERMN monitoring data, this report offers the latest answers to that question and provides park-specific management recommendations developed from these findings.

Methods

The ERMN includes nine parks in New York, New Jersey, Pennsylvania, and West Virginia (Figure 1), which together encompass nearly 91,000 ha (225,000 ac) of land area and over 965 km (600 mi) of streams and rivers within the parks' authorized boundaries. The network includes four smaller parks in central and southwestern Pennsylvania that have a primarily cultural or historical focus. The cultural parks are Allegheny Portage Railroad National Historic Site (ALPO), Johnstown Flood National Memorial (JOFL), Fort Necessity National Battlefield (FONE), and Friendship Hill National Historic Site (FRHI). The five larger parks preserve segments of large rivers and generally extend to the ridge tops surrounding the river section. The river parks are Upper Delaware Scenic and Recreational River (UPDE), Delaware Water Gap National Recreation Area (DEWA), New River Gorge National River (NERI), Gauley River National Recreation Area (GARI), and Bluestone National Scenic River (BLUE). In 2007, the ERMN began monitoring vegetation communities and soil in all of its parks except UPDE, which contains insufficient publically owned land on which to conduct long-term vegetation monitoring.

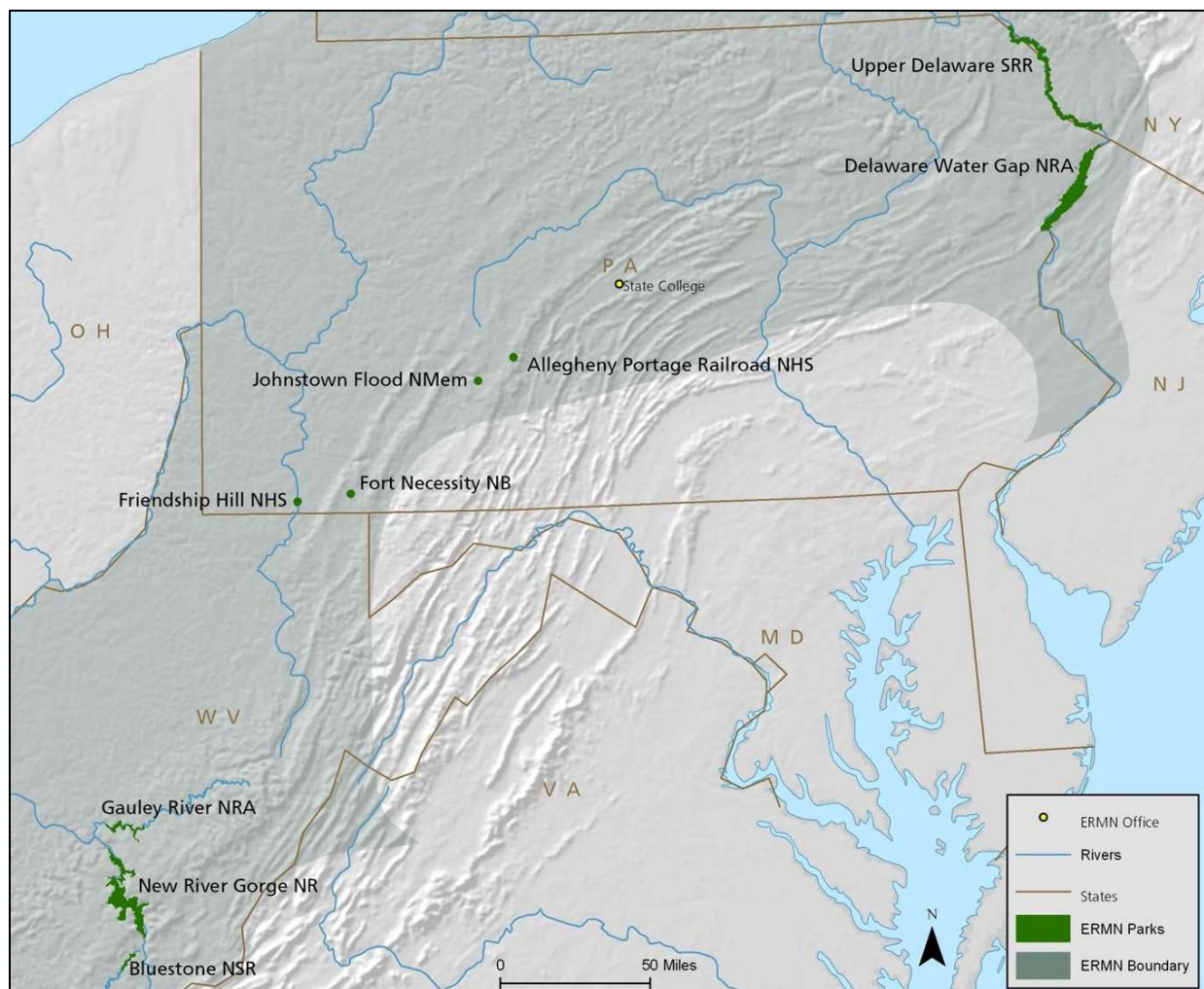


Figure 1. Locations of parks in the Eastern Rivers and Mountains Network.

Adapted from the U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) Program and the vegetation monitoring protocols of four other NPS Inventory and Monitoring programs in the eastern United States (Sanders et al. 2008, Schmit et al. 2009, Comiskey et al. 2009, Tierney et al. 2015), the ERMN forest protocol monitors a representative suite of site and vegetation measures in an extensive network of randomly located permanent plots (Perles et al. 2014a).

The sampling universe for vegetation and soil monitoring includes all area within each park's authorized boundary that is publically owned, vegetated, not intensively managed, and has a slope of less than 30° (Perles et al. 2014a). Vegetation and soil are monitored at permanent plots, since the use of permanent plots increases the power to detect trends through time (Urquhart et al. 1998). For each park, sampling locations were selected from a regular grid of potential plot locations using a generalized random-tessellation stratified (GRTS) design (McDonald 2004, Stevens and Olsen 2004). Plots are sampled on a four-year panel design, in which one panel containing one-fourth of a park's total plots is sampled each year. The 2014 field season marked the eight year of sampling the monitoring plots, and the completion of the second round of panel visits, such that all plots had been sampled twice (Table 1).

Table 1. The number and type of monitoring plots sampled in ERMN parks from 2007–2012. Plots from panel 1 (2007 and 2011) are shown in blue. Plots from panel 2 (2008 and 2012) are shown in orange. Plots from panel 3 (2009 and 2013) are shown in green. Plots from panel 4 (2010 and 2014) are shown in purple.

	NERI	DEWA	JOFL	ALPO	FONE	FRHI	GARI	BLUE	Total
2007 – Panel 1	20	26	1	3	5	5	12	12	84
2008 – Panel 2	29	26	3	6	5	5	9	9	92
2009 – Panel 3	25	25	3	6	5	5	9	9	87
2010 – Panel 4	26	25	3	6	5	5	10	10	90
2011 – Panel 1	22 ^a	26	3 ^b	6 ^b	5	5	12	12	91
2012 – Panel 2	28 ^c	26	3	6	5	5	9	9	91
2013 – Panel 3	25	25	3	6	5	5	9	9	87
2014 – Panel 4	26	25	3	6	5	5	10	10	90

^a Two plots were added in NERI to panel 1 in 2011 on newly-acquired land with the park's authorized boundary. For more information on this sampling design see the vegetation and soil monitoring protocol (Perles et al. 2014a)

^b Logistical issues prevented a full panel of plots from being established in ALPO and JOFL in 2007, so additional plots were added in 2011 to complete the panel.

^c Due to a resurveying of the park boundary, one plot from panel 2 that was established in 2008 was no longer within the park boundary in 2012; therefore, this plot was removed from the monitoring program.

Field Methods

At each plot, the ERMN monitors a suite of vegetation and soil variables. The plot design includes several embedded sampling units (Figure 2). Details of the sampling methods, in addition to Standard Operating Procedures for data collection, can be found in the Vegetation and Soil Monitoring Protocol (Perles et al. 2014a). Tree, stand, and site measurements are collected within fixed-area, circular plots, 15 meters in radius. For this protocol, a tree is defined as specific woody species (see

Perles et al. 2014a) with a diameter-at-breast-height greater than or equal to 10 cm. Tree regeneration and shrub measurements are collected on four 2-m radius circular microplots embedded within each plot. Data on coarse woody debris are collected using line intersect sampling along six 15-m (49.2-ft) transects. Data on understory plant composition are monitored using twelve 1-m² quadrats set along the six transects. A photograph of the plot is taken from the plot's southern edge to document change in vegetation structure through time.

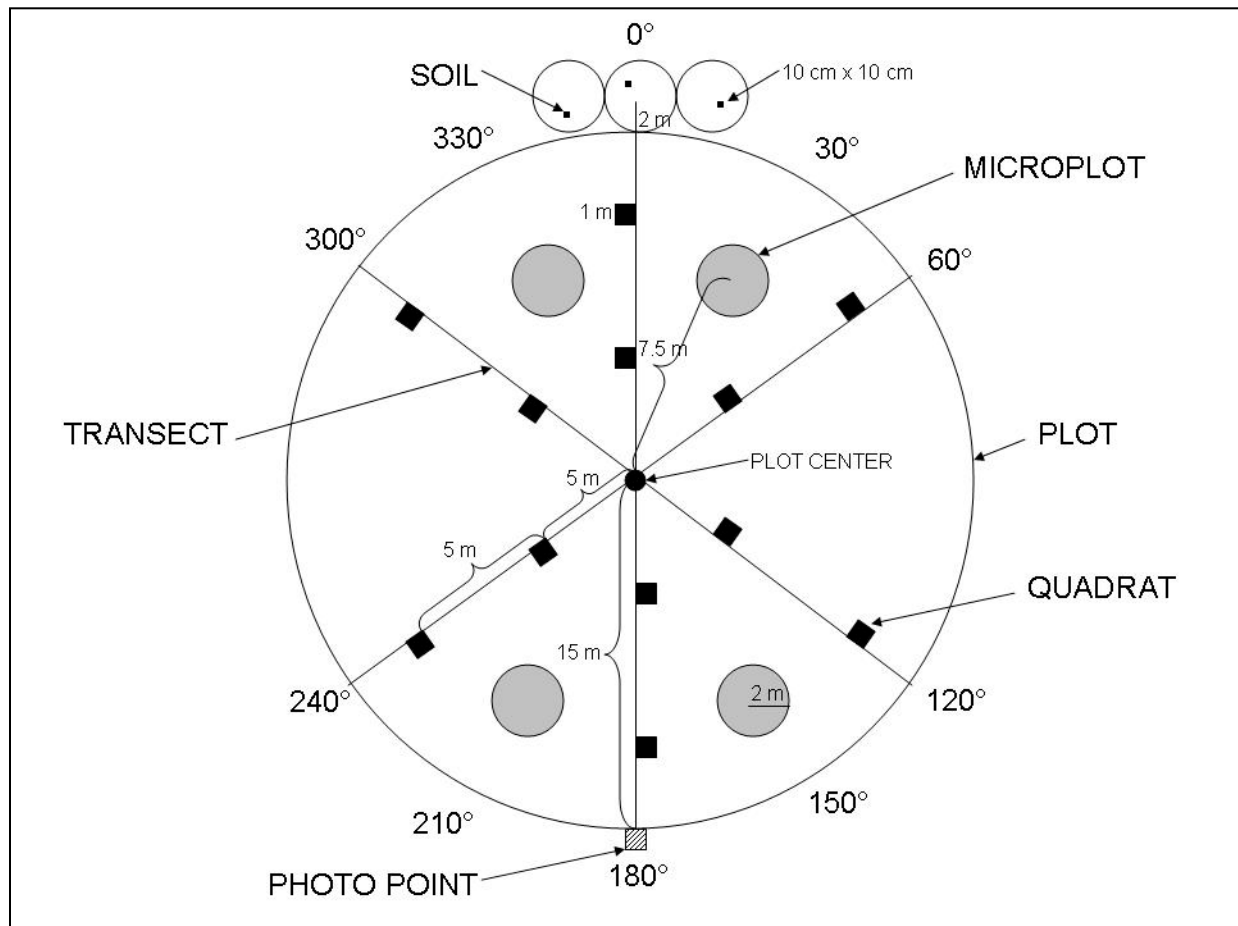


Figure 2. Plot design for Eastern Rivers and Mountains Network Vegetation Monitoring protocol. Tree, stand, and site measurements are collected within the plot. Tree regeneration and shrub measurements are collected in the microplots. Data on coarse woody debris are collected along transects. Data on understory plant composition and the diversity of understory species are collected in the quadrats. A photograph of the plot is taken from the plot's southern edge. Three soil samples are collected from sampling frames located adjacent to the plot's northern edge.

In addition, at the plot's first census, three soil samples are collected from sampling frames located adjacent to the plot's northern edge (Figure 2). One sample is collected from each of three 2-m radius circle sampling frames. A 10×10-cm quadrat is used to identify the sampling area within the 2-m radius sampling frame. Then the soil within the quadrat is removed and collected by genetic horizons (e.g., O, A, and/or B), storing soil from each horizon separately. Soil is typically collected to a depth of <15 cm.

Laboratory Analysis

Soil samples were air dried and then chemically analyzed at the Analytical Lab at the University of Maine. Soil pH was measured in distilled water. Organic matter was measured by loss on ignition (LOI) at 550°C. Total nitrogen and carbon were measured by combustion analysis at 1,350°C. Exchangeable acidity was extracted in potassium chloride and measured by titration. Exchangeable cations and phosphorous were extracted in ammonium chloride and measured by inductively coupled plasma with optical emission spectrometry (ICP-OES). All values for cations reported here are extractable concentrations, not total soil concentrations. From the chemical analysis, several additional variables were calculated, including effective cation exchange capacity (ECEC), percent base saturation (BS), molar calcium to aluminum ratio (Ca:Al), carbon to nitrogen ratio (C:N), and nitrogen to phosphorus ratio (N:P).

Data Analysis

Mortality, recruitment, and growth rates are calculated by comparing data collected on the same trees during two different sampling visits. The rates reported here were computed by comparing data from 2007 - 2010 with data from 2011 - 2014, respectively. Data were collected from over 14,400 trees within 346 plots among the eight parks.

Mortality and Recruitment

Annual mortality rate (MR; % trees/year) is calculated from successive observations as the percent of trees that died between sample events, converted to an annual basis. Mortality is calculated as:

$$MR = \frac{N_d \times 100}{N_o \times t}$$

where N_o is the number of trees in the previous census, and N_d is the number of trees that died after t years.

Similarly, annual recruitment rate (RR; % trees/year) is calculated as the percent of trees that grew such that their diameters exceeded the minimum threshold for trees (10.0 cm). These individuals “became” trees between the two sampling events. Recruitment is calculated as:

$$RR = \frac{N_r \times 100}{N_o \times t}$$

where N_o is the number of trees in the previous census, and N_r is the number of recruited trees after t years.

To compare parks across the network, recruitment and mortality rates were calculated by plot and then averaged within each park. To compare recruitment and mortality rates for tree species within a park, rates were calculated using all observed trees of that species within the park. Each individual tree was treated as a sample unit, which does not account for spatial autocorrelation (i.e., correlation of individuals on the same plot). Therefore, the species-level standard error is likely an underestimate of the true error of the population, and is not reported here. Species-specific rates were calculated

only for species that had 10 or more observed individuals and occurred in more than 3 plots within a park.

Tree Growth

Before calculating tree growth metrics, trees whose diameter at breast height (DBH) differed more than 6 cm between censuses were removed, since these differences were likely the result of human error. These records accounted for less than 1% of the over 14,400 observed trees.

One standard measure of tree growth is mean annual diameter increment (MADI), calculated as the sum of the annual diameter increment (in cm) for all stems of a species or in a plot:

$$MADI = \frac{\sum(DBH_2 - DBH_1)/t}{n}$$

where DBH_1 is the diameter of the tree in the first census, DBH_2 is the diameter of the tree in the next census, with a census interval of t years, and n equals the total number of trees. MADI rates were calculated by plot and then averaged within each park. MADI was also calculated for many tree species by averaging the annual diameter increment for individual trees within each park for species with greater than 5 observed individuals within the park. Each individual tree was treated as a sample unit, which does not account for spatial autocorrelation (i.e., correlation of individuals on the same plot). Therefore, the species-level standard error reported herein is likely an underestimate of the true error of the population.

An alternative measure of tree growth, change in total plot live tree basal area (% basal area/year) is calculated as:

$$\Delta BA = \frac{(BA_2 - BA_1) \times 100}{BA_1 \times t}$$

where BA_1 is the total plot live tree basal area in the first census, BA_2 is the total plot live tree basal area in the next census, and a census interval of t years. Basal area change is calculated for each plot and then averaged within each park.

Species Composition

To investigate how the species composition of the forest may change in the future, we compared the current species composition of the tree canopy with the species composition of the seedling layer within each park. In the four larger parks, species composition was compared within broad vegetation domains by analyzing plots in xeric habitats (i.e., dry, usually upper slopes and ridgetops) and plots in mesic habitats (i.e., moist, usually lower slopes and valleys) separately (Table 2).

Table 2. Number of plots sampled by park and vegetation domain.

Park	Mesic	Xeric	Early Successional	Total
ALPO				24
BLUE	22	18		40
DEWA	46	50	6	102
FONE				20
FRHI				20
GARI	15	25		40
JOFL				12
NERI	41	60		101

Relative abundance measures for the canopy and seedling layers were calculated from the latest visit to each plot during 2011 - 2014. For canopy trees, relative basal area was calculated by summing the basal area across plots for each species and dividing by the total basal area for all species from all plots. Basal area was chosen over density because basal area provides a better measure of dominance in a forest stand. For seedlings, relative density was calculated by summing the number of seedlings across plots for each species and then dividing by the total number of seedlings for all species from all plots. Comparison of the relative abundance of species in the canopy and seedling layers provides information on the likely longer term shifts in species composition in forest stands.

Although nearly all live trees and most seedlings are identified to species, many dead trees and some seedlings cannot be reliably identified to species. In these cases, the genus is recorded in the field. For this analysis, genus-level records (e.g., *Fraxinus* spp.) are treated separately from species-level records (e.g., *Fraxinus americana*) since the genus-level records could include more than one species (e.g., *F. americana* and *F. pennsylvanica*).

Predictors of Tree Growth and Mortality

To investigate the factors that influence tree growth and mortality, we modeled plot-level MADI and MR as a function of stand and soil variables. For this analysis, growth and mortality data were filtered in the following ways:

- removed 16 woodland plots with $< 11\text{m}^2/\text{ha}$ total basal area at first census, since growth rates in these early successional habitats are not representative of the majority of the parks' forests;
- removed six plots with high mortality that received severe storm damage that impacted greater than 25% of the plot in between censuses.

Models were built using park-specific data for DEWA, or using data from geographically-related parks, such that data were grouped for NERI, BLUE, and GARI, and for ALPO, JOFL, FONE, and FRHI. Separate models were built for O horizon soil data and A horizon soil data.

For each park-horizon-specific data set, we fit a full regression model using the following variables listed in Table 3. These soil variables are accepted as the key indicators of soil condition that affect

forest health (Perles et al. 2014b). The natural log of Ca:Al was used to adjust for the strong skew in the Ca:Al data. Outliers greatly exceeding two standard deviations around the mean were removed from the analysis. Because several of the soil variables are correlated with each other, we calculated the variance inflation factor (*vif*) and removed variables with high *vif* values that were not significant in the global model. The remaining variables were submitted to a stepwise model selection by AIC, calculated in the MASS library in R 3.1.3 (Venables and Ripley 2002, R Core Team 2015). We examined adjusted R^2 values and linear fits between observed and predicted values of response variables to describe the quality of model fits.

Table 3. Stand and soil chemistry variables used in model of tree growth and mortality.

Predictor Variable	Abbreviation	Formula
Tree basal area at time 1	BA	Sum basal area for each living tree within a plot
Percent dead standing tree at time 1	%Dead	Percent of standing trees that are dead within a plot
pH	soil pH	Direct measure
Aluminum (mg/kg)	Al	Direct measure
Effective cation exchange capacity ¹	ECEC	(Ca + K + Mg + Na + acidity)
% Base Saturation ¹	BS	((Ca + K + Mg + Na) / ECEC) *100
Calcium : Aluminum ratio	Ca:Al	(Ca / 40.078) / (Al / 26.981)
Carbon : Nitrogen ratio	C:N	TC / TN

¹ For these variables, cations were first converted to meq / 100g using this formula:
Cation (mg/kg) / ((Cation atomic weight / valence)*10).

Results and Discussion

Forest Dynamics

First, we compare tree mortality, recruitment and growth across the eight parks in ERMN and discuss the implications for forest health in the parks. Next, park-specific trends in tree species composition and the growth and mortality of the parks' common tree species will be presented for each park. Note that the term “canopy” is used in this section to refer to trees that are greater than 10 cm at DBH, and includes trees that are in subcanopy or overtopped crown positions in the stand.

In nearly all the ERMN parks, mortality rate is much higher than the recruitment rate (Figure 3), which is expected in middle-aged, closed-canopy forests. Two-thirds of the observed mortality is in sub-canopy and overtopped trees growing in shade below the canopy that are out-competed by larger canopy trees. Mortality rates in ERMN parks are typical of rates reported for other eastern forests, ranging from 0.3–1.6% (Lorimer 1980, Runkle 1998, Runkle 2000, Busing 2005). The only park in which tree recruitment exceeded mortality was DEWA. Approximately 10% of the park is in open early successional habitats in which trees are quickly establishing, leading to a high recruitment rate. The rapidly growing trees in these habitats also contribute to the higher increase in basal area observed in the park (Figure 4).



Figure 3. Mortality and recruitment rates (percent / year) for trees in ERMN parks. Error bars represent 1 standard error above and below the mean.

Average tree growth rates in ERMN parks (Figure 5) also fall within the expected ranges for Appalachian forests of 0.25 – 0.40-cm per year (Runkle 1998, 2000). Average annual increases in total live tree basal area in ERMN parks (Figure 4) typically exceed of the reported values of -0.22 –

0.45% for old growth forests in our region (Runkle 2000), which is expected since the young and middle-aged forests in ERMN parks would accrue basal area faster than old-growth stands.

However, FRHI, GARI and BLUE are notable exceptions when looking at tree growth across the network. Much of FRHI is characterized by young forests that have established on former agricultural land. These young trees, growing in full sun on fertile soils, are exhibiting rapid growth, the highest in the network (Figures 4, 5). The severe storms in GARI in 2012 toppled so many trees that, on average, the standing living tree basal area decreased by -0.04% in the park (Figure 4), compared with an average increase in basal area of 1.11% during the same time period across the whole network. For trees in GARI that survived the storms, growth is typical (Figure 5). BLUE shows consistently low tree growth, the third highest mortality rate, and second lowest recruitment rate (Figures 3, 4, and 5). The average growth rate is the lowest of the network for all tree species averaged together, as well as for three of the five common tree species across the network (Figure 6). These conditions result from the relatively harsher environment in BLUE compared to the other parks, primarily drier and warmer conditions that leave the trees more susceptible to stressors (see BLUE section below for further discussion).

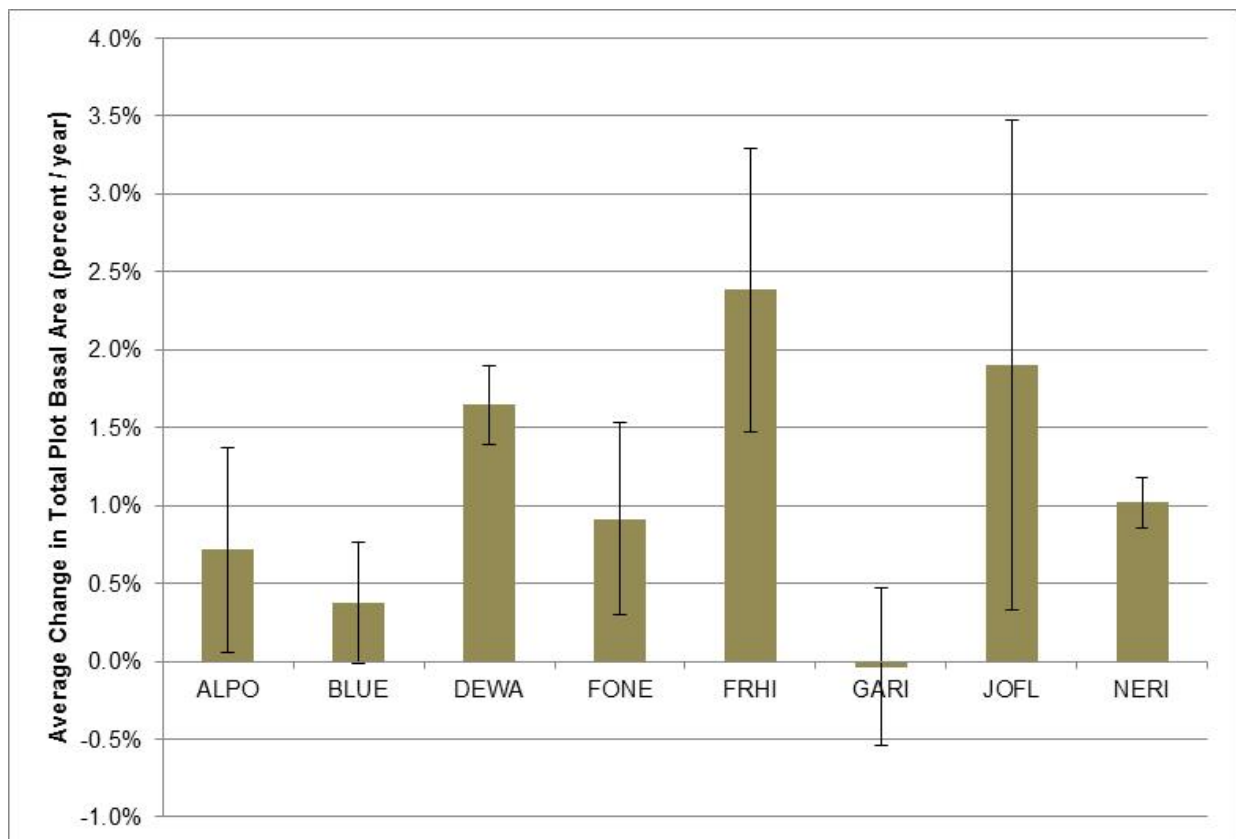


Figure 4. Average change in total plot live tree basal area (percent / year) in ERMN parks. Error bars represent 1 standard error above and below the mean.

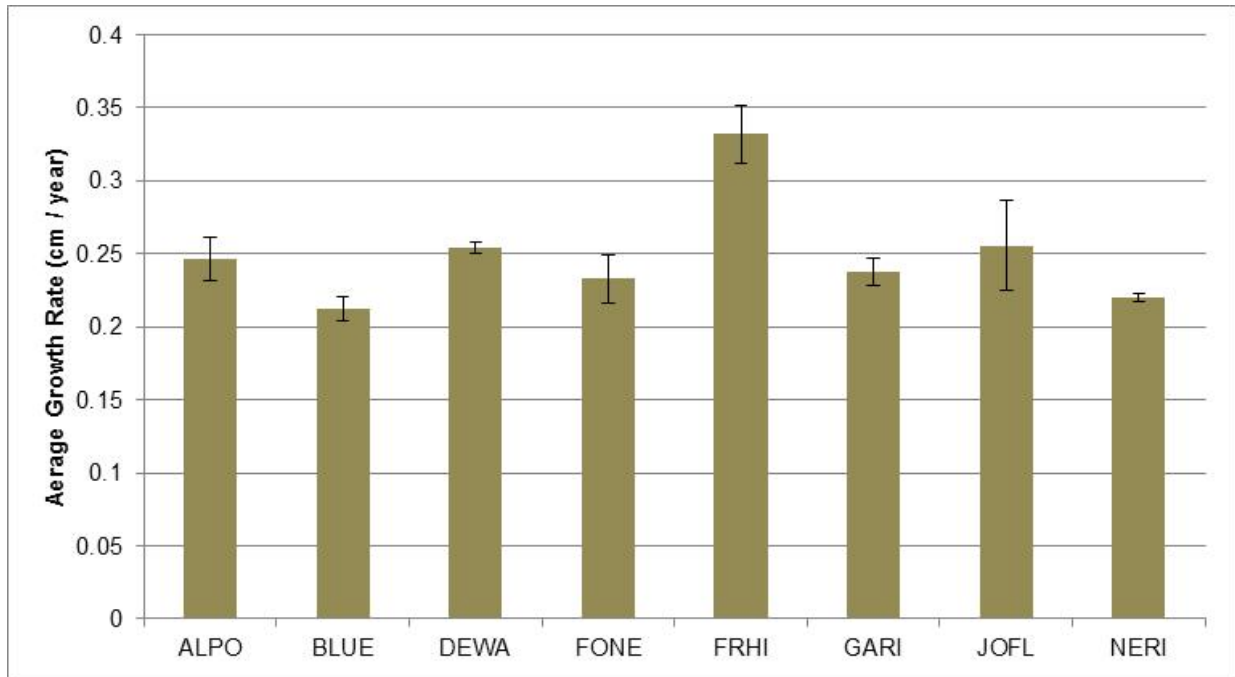


Figure 5. Average growth rate (cm / year) for live trees in ERMN parks. Error bars represent 1 standard error above and below the mean.

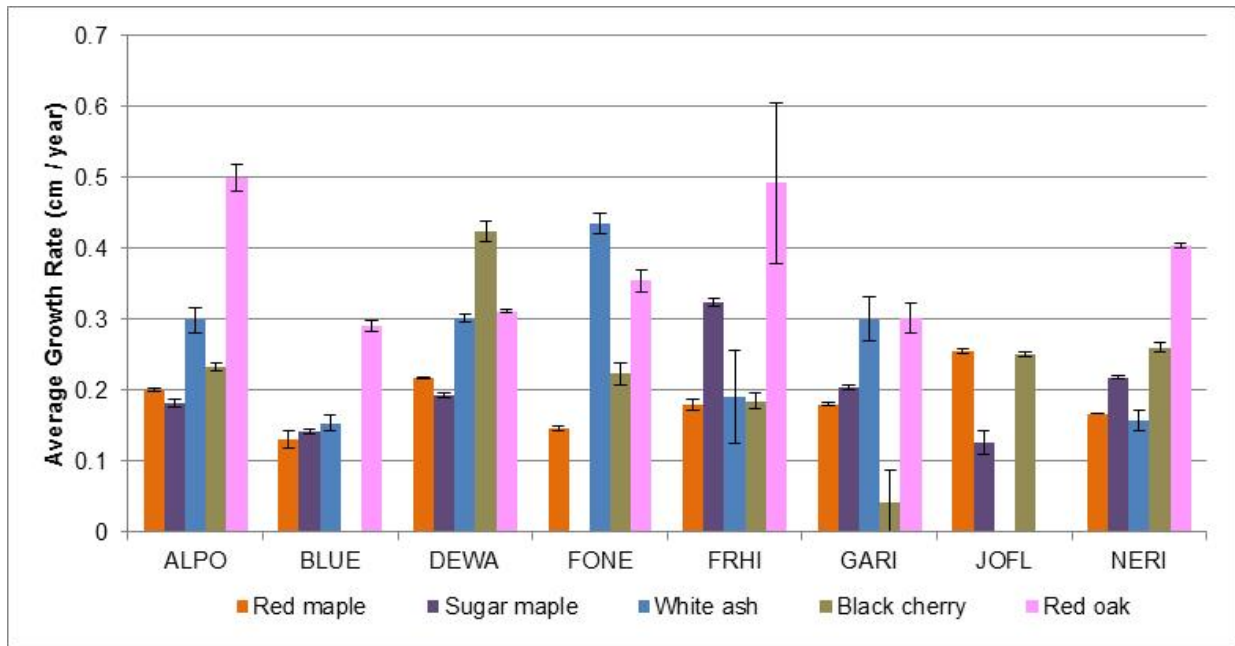


Figure 6. Average growth rate (cm / year) for common tree species in ERMN parks. Error bars represent 1 standard error above and below the mean.

Allegheny Portage Railroad National Historical Site

Forests in ALPO are dominated by red maple (*Acer rubrum*), black cherry (*Prunus serotina*), American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and red oak (*Quercus rubra*). This forest composition appears relatively stable, since the seedling composition matches the canopy composition for the dominant species (Figure 7). However, hemlock (*Tsuga canadensis*) and sugar maple are underrepresented in the seedling layer, indicating that these species may be less common in the future canopy. More striped maple (*Acer pensylvanicum*) and ash (*Fraxinus* spp.) seedlings are present in the seedling layer than in the canopy. The ash seedlings are unlikely to survive to maturity because of the emerald ash borer (*Agrilus planipennis*), an invasive exotic insect, but striped maple is likely to be more common in the future canopy.

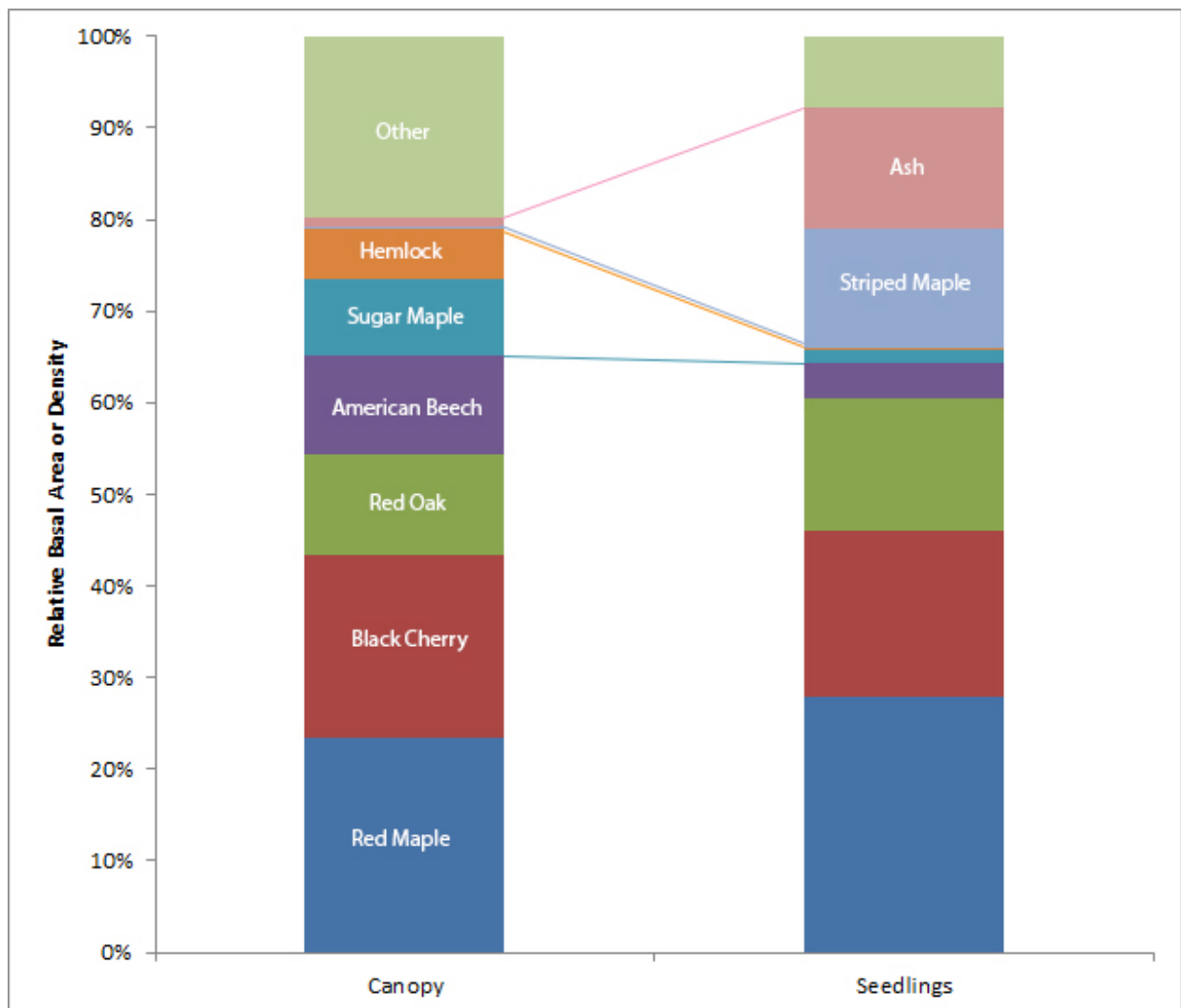


Figure 7. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in monitoring plots in ALPO. Colored lines highlight notable shifts in species composition.

Forests in ALPO show moderate growth, typical of a middle-aged forest. Despite the fact that more trees died than were recruited, the overall increase in basal area indicates that the forest is maturing (Figures 3 and 4) as smaller trees are out-competed by the larger canopy trees.

The fastest growing species in the park are red oak, white ash (*Fraxinus americana*), hemlock, and American beech (Figure 8). Red oak appears to be in good condition in the park, with a strong growth, solid recruitment, and no observed mortality (Figure 9). In contrast, white ash experienced >2% mortality, despite the high average growth rate. Emerald ash borer is most likely responsible for the elevated mortality. Although hemlock woolly adelgid (*Adelges tsugae*) have been present in the park for several years, hemlock mortality equals recruitment and the growth rate for hemlocks is typical of healthy trees. Monitoring the effects of hemlock insecticide treatments and the release of biological control insects in hemlock-dominated stands should continue since successive years of droughts and / or warm winters could cause significant stress to the hemlock trees.

Several common understory trees, apples (*Malus* spp.), hawthorns (*Crataegus* spp.), serviceberry (*Amelanchier arborea*), and black locust (*Robinia pseudoacacia*) are the slowest growing species in the park (Figure 8), which is typical of small trees persisting in the shade of the forest canopy. Black locust and apple also have high mortality rates (Figure 9), as these early successional species are out-competed in the middle-aged forest.

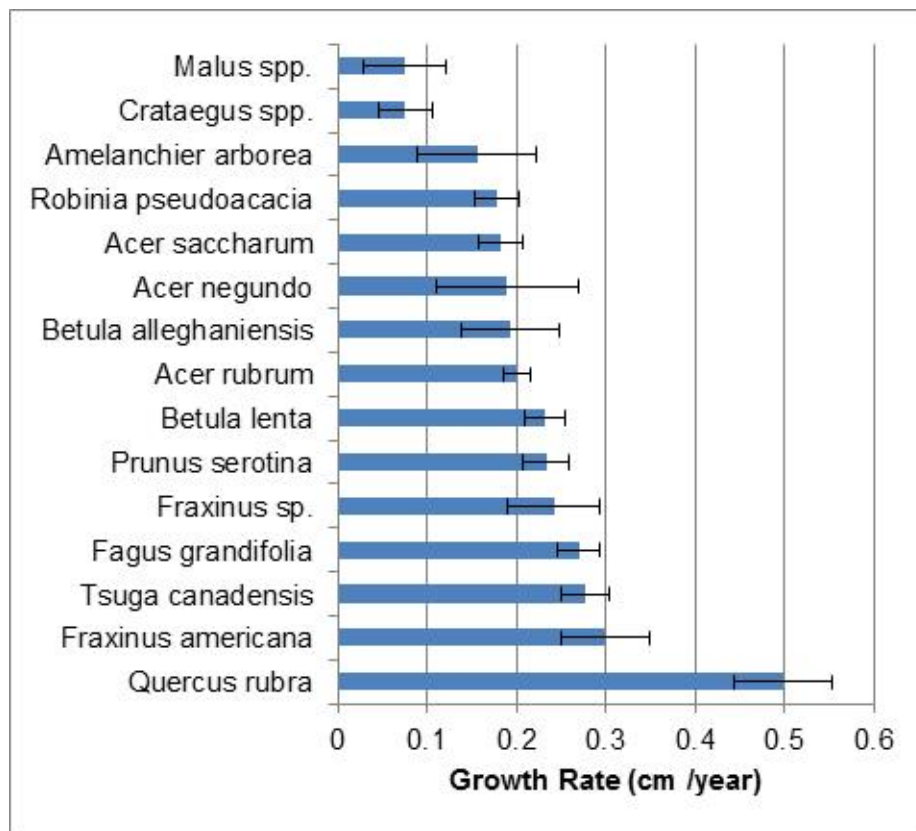


Figure 8. Growth rate of common tree species in ALPO. Error bars represent 1 standard error above and below the mean.

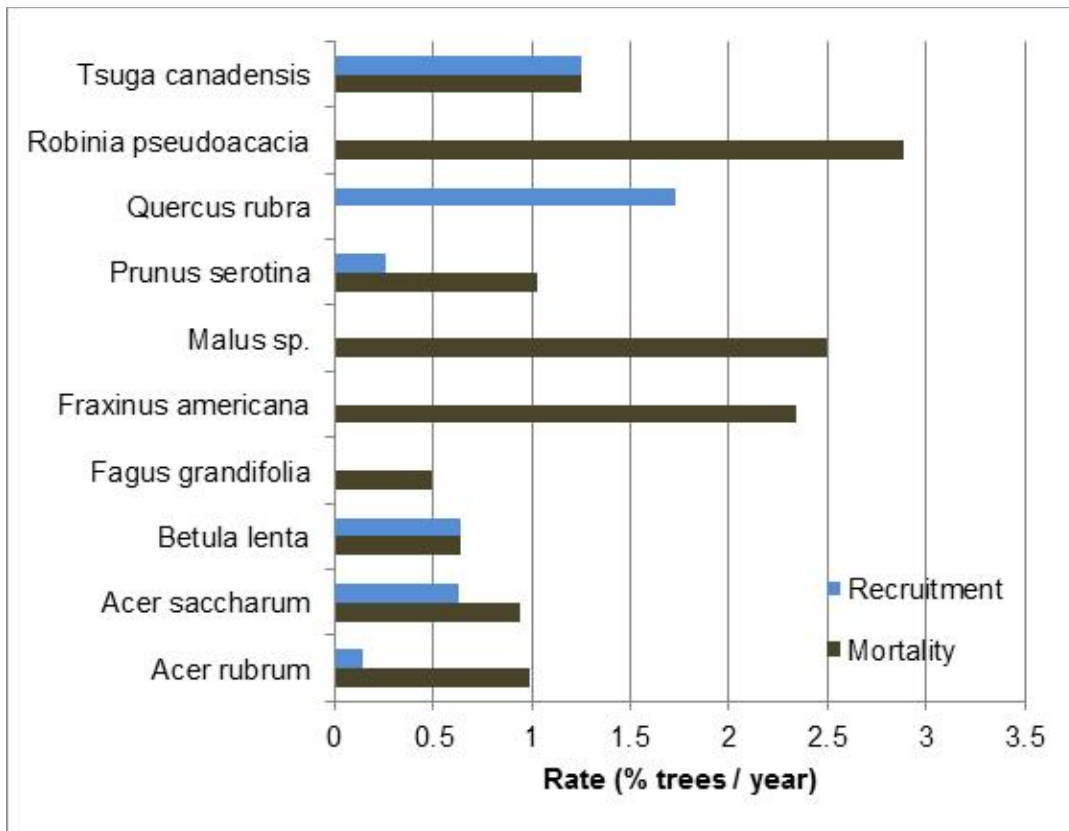


Figure 9. Mortality and recruitment rates (percent / year) for common tree species in ALPO.

Bluestone National Scenic River

Compared to other network parks, BLUE has the third highest tree mortality rate and second lowest recruitment rate (Figure 3). Tree growth in BLUE is the lowest of the network for all tree species averaged together (Figure 5), as well as for three of the five common tree species across the network (Figure 6). The average basal area change was also the lowest of the network with the exception of the storm-impacted GARI (Figure 4). These conditions result from the relatively harsher environment in BLUE compared to the other parks, primarily drier and droughty conditions that leave the trees more susceptible to stressors.

Although BLUE is nearly adjacent to NERI, BLUE falls in a separate climatic zone than NERI and GARI (NOAA 2015). Precipitation is the most critical aspect of the climatic differences for the parks' forests. Annual precipitation in BLUE was 36-40 inches for the 30 year normal from 1981-2010, whereas the rest of the ERMN parks received 40-50 inches annually (Figure 10, Oregon State University 2015). BLUE also has a slightly higher average annual temperature, based on 30-year normal, compared to the other network parks (Oregon State University 2015).

BLUE is in a different physiographic province based on the underlying geomorphology (Fenneman and Johnson 1946). It is the southernmost ERMN park in the Eastern Ridge and Valley physiographic province, while most other ERMN parks occur on the Western Allegheny Plateau.

Although DEWA and part of ALPO also lie within the Eastern Ridge and Valley, those two experience cooler temperatures than BLUE (Oregon State University 2015).

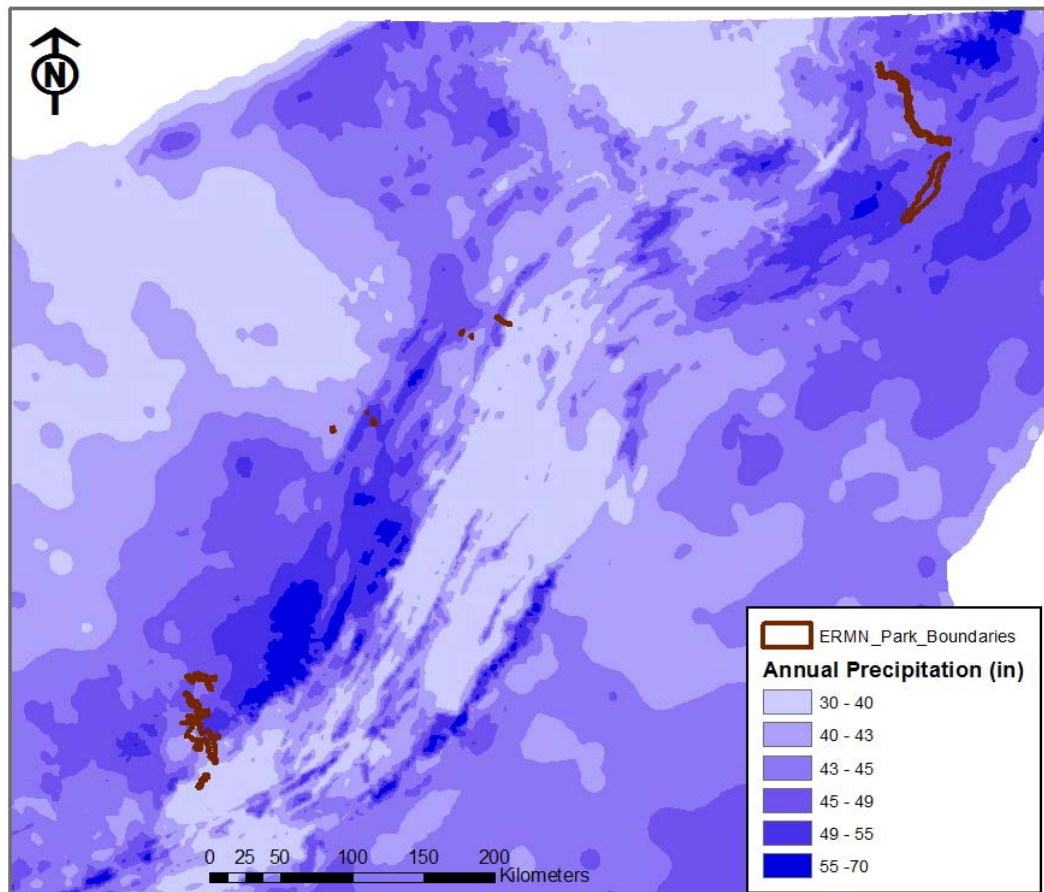


Figure 10. Annual precipitation in inches based on 30-yr normals (Oregon State University 2015). BLUE is the southern-most park, receiving less than 40 inches of precipitation annually.

BLUE soils are primarily Cateache and Pipestem channery silt loams and the Gilpin-Highsplint-Berks complex, which are well-drained, droughty soils that tend to lack a substantial organic horizon (USDA NRCS and DOI NPS 2013). When soil samples were collected at ERMN vegetation monitoring plots, very few plots in BLUE contained organic horizons in the soil profile. Only 27% of the plots contained any collectable O horizon soil among the three soil sampling frames and only 19 out of a possible 120 samples had O horizons greater than 0.5 cm thick. One of the important functions of the organic horizon is to increase water retention and availability. The organic horizon also insulates the mineral soil from extreme temperatures, provides nutrients, and influences the mineral soil structure, further influencing the mineral soil's water holding capacity (Binkley and Fisher 2012).

The drier, warmer climate and the droughty soils lead to harsher growing conditions for BLUE's forest. Since the park's trees are generally more stressed, they are more susceptible to additional impacts such as pests, pathogens, drought, and climate change. Insects such as hemlock woolly

adelgid, gypsy moth, and emerald ash borer may cause greater or quicker damage in BLUE than in the other parks. Similarly, forests in BLUE will likely experience higher mortality from severe droughts.

Although the soils and geology of the southeastern portion of NERI are very similar to those of BLUE, there are only six monitoring plots in that section of NERI because much of the land is privately owned in that section of the park. The forests in the southeastern portion of NERI are likely experiencing similar stresses as the forests in BLUE, but these effects are masked by the majority of the plots in NERI. This section of NERI is therefore also likely at risk of negative impacts from other additional stressors.

Focusing specifically on the xeric forests in BLUE, the dying trees in BLUE's xeric habitats are primarily oaks (*Quercus* spp.), hemlock, and white pine (*Pinus strobus*) (Figure 11). These trees are the dominant species in the xeric forest, and elevated mortality with minimal recruitment foreshadows an uncertain future for this forest type. Red maple, sugar maple, mockernut hickory (*Carya alba*), and sourwood (*Oxydendron arboreum*) are the only species in xeric habitats for which recruitment equals or exceeds mortality, suggesting these species may be a larger component of the future canopy. The dominant oaks, pine and hemlock are also underrepresented in the seedling layer when compared with the canopy composition (Figure 12). Red maple and white ash are dominant in the seedling layer, which is a large increase over their portion of the canopy trees. Since ash seedlings are unlikely to reach maturity due to emerald ash borer, red maple will likely become a larger component of the future forest canopy. The increase in red maple abundance in xeric oak-dominated forests is a region-wide trend (Abrams 1998), influenced by the restriction of fire in the 20th and 21st centuries (Brose et al. 2013), but possibly also as a result of climate changes, land-use changes, the loss of foundation and keystone species, and browse from larger white-tailed deer populations (McEwan et al. 2011). Fortunately, the current oak trees exhibit typical growth, which typically indicates good health (Figure 13).

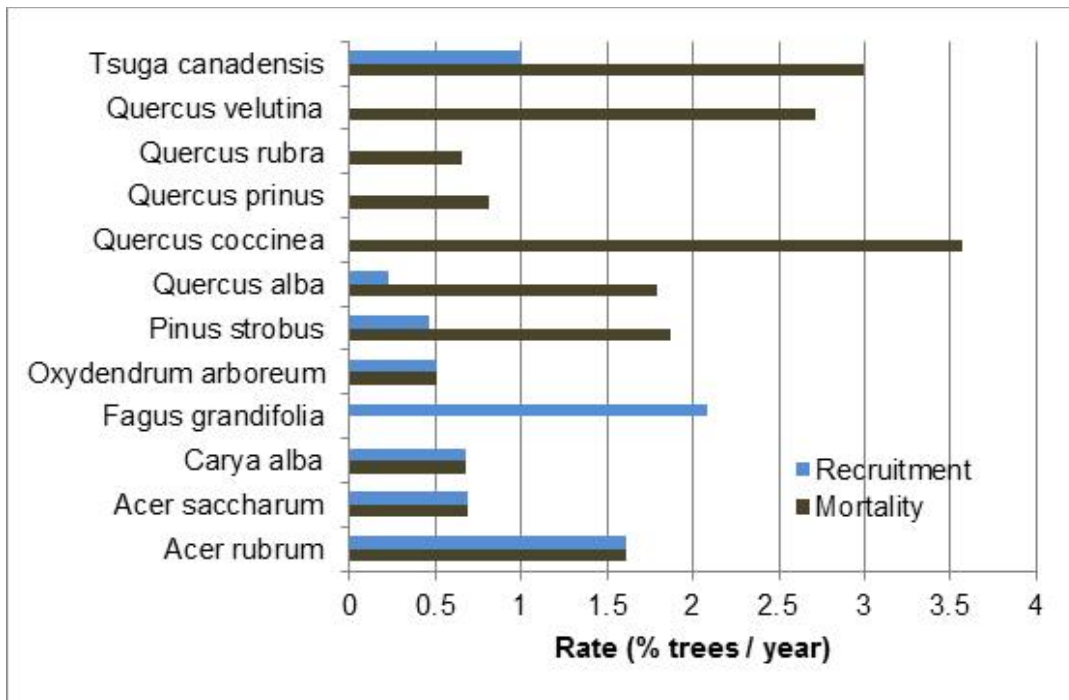


Figure 11. Mortality and recruitment rates (percent / year) for common tree species in xeric habitats in BLUE.

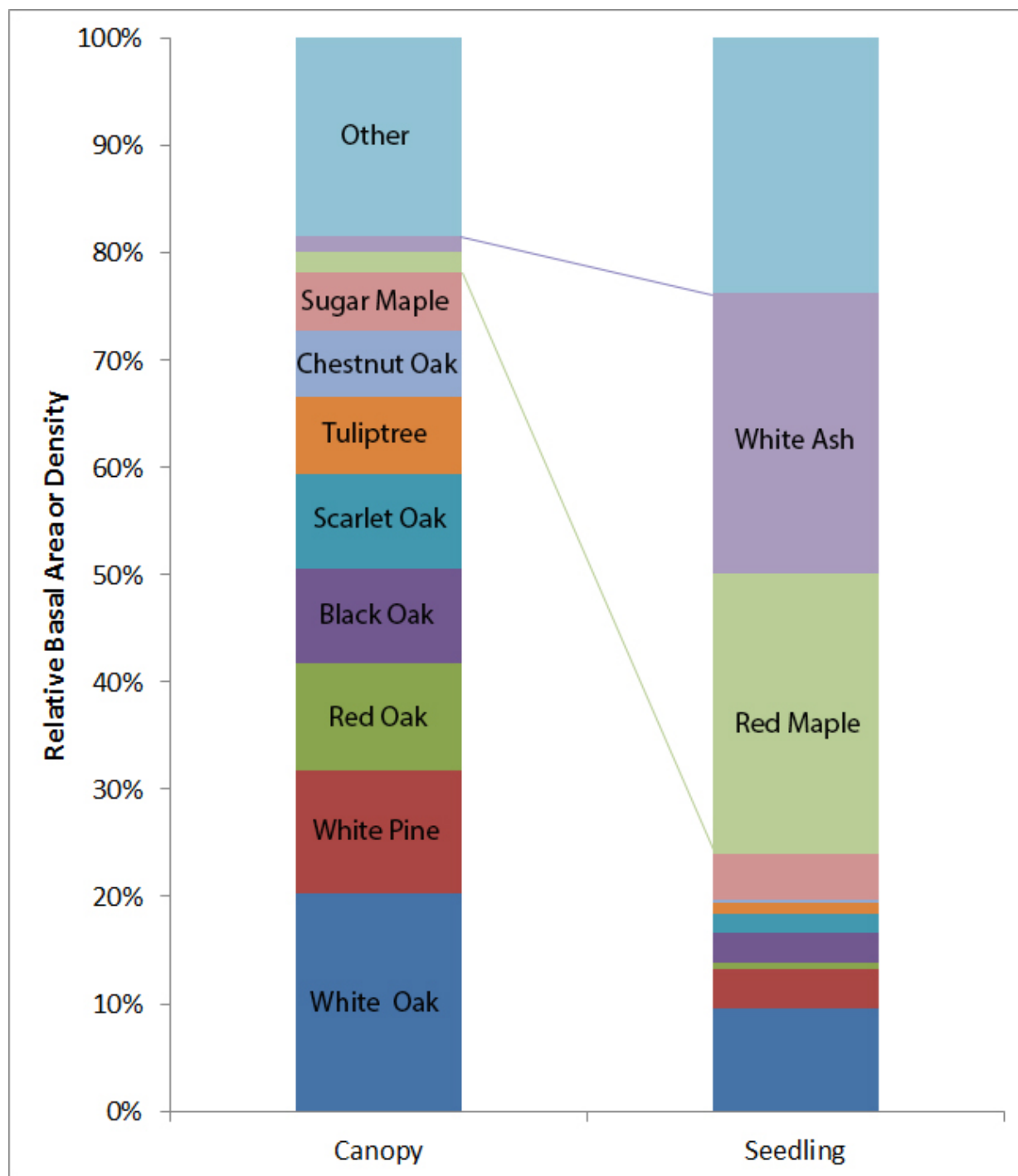


Figure 12. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in xeric habitats in BLUE. Colored lines highlight notable shifts in species composition.

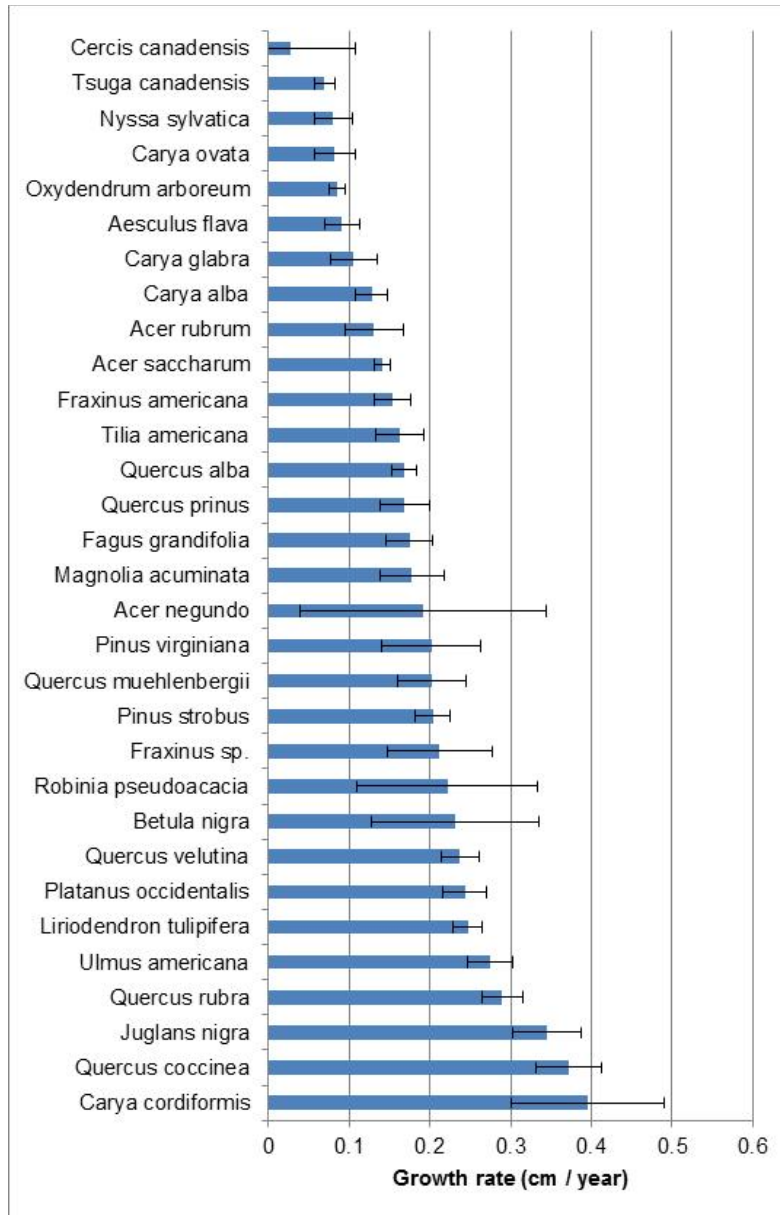


Figure 13. Growth rate of common tree species in BLUE. Error bars represent 1 standard error above and below the mean.

Mesic forests on the lower slopes and floodplains of BLUE contain varied canopy species composition (Figure 14). Since mortality greatly exceeds recruitment for nearly all common canopy species (Figure 15), canopy composition may shift in coming years away from red oak, black oak (*Quercus velutina*), white pine, white ash, and basswood (*Tilia americana*). White oak appears stable in the canopy and is well represented in the seedling layer. Although sycamore (*Platanus occidentalis*) and river birch (*Betula nigra*) showed no recruitment, these species are likely to persist in the floodplain along the Bluestone River since they are adapted to disturbance from flooding. The high mortality rate for black locust is a result of competition from canopy trees in the maturing forest as black locust is an early successional, shade intolerant tree.

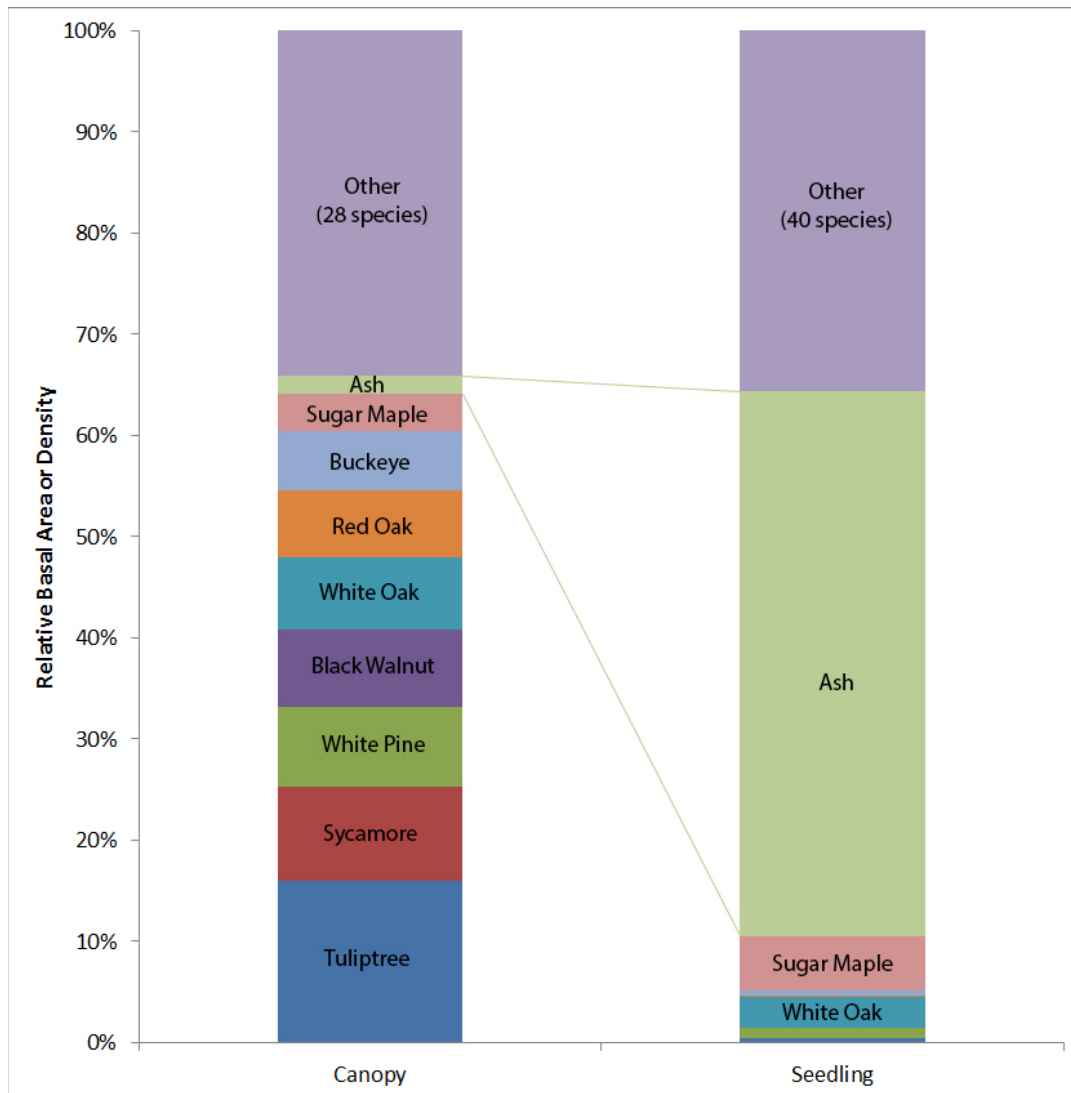


Figure 14. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in mesic habitats in BLUE. Colored lines highlight notable shifts in species composition.

Sugar maple, yellow buckeye (*Aesculus flava*), and tuliptree (*Liriodendron tulipifera*) will likely continue to dominate the canopy in the short term, since tree recruitment equaled or exceeded mortality for these species and sugar maple seedlings are common. Tuliptree is currently the most abundant canopy tree; however, this species needs sunlight to regenerate, so seedlings and saplings do not survive without disturbance to the canopy. The seedling layer is strongly dominated by ash, although these seedlings are not expected to survive to join the canopy unless a biological control is discovered for the emerald ash borer.

The fastest growing tree species in mesic forests along BLUE's lower slopes are bitternut hickory (*Carya cordiformis*) and tuliptree. Along the floodplain, black walnut (*Juglans nigra*), American elm (*Ulmus americana*), and sycamore exhibited higher growth rates. Hemlocks have the slowest growth rate of any canopy tree species in the park, likely due to infestation of hemlock woolly adelgid (Figure 13).

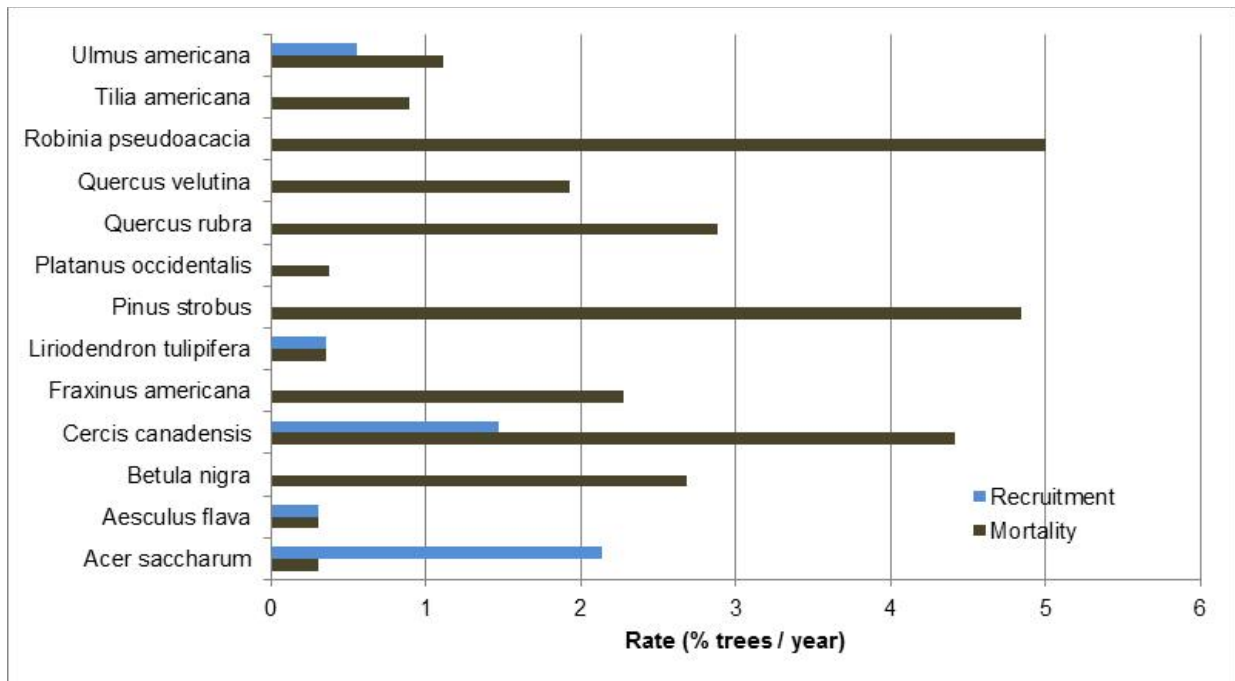


Figure 15. Mortality and recruitment rates (percent / year) for common tree species in mesic habitats in BLUE.

Delaware Water Gap National Recreation Area

Of the ERMN parks, DEWA contains the greatest diversity of vegetation types, including riparian, successional, and mesic and xeric forests, with small pockets of rare vegetation communities dispersed throughout the park.

Approximately 10% of the park contains non-forested early successional habitats in which trees are currently establishing. The addition of trees to early successional plots contributes to the park's high recruitment rate (Figure 3). The rapid growth of these trees also causes the greater increase in basal area observed in the park (Figure 4). The fastest growing trees in these habitats are black birch (*Betula lenta*), eastern red cedar (*Juniperus virginiana*), white pine, black cherry, and black walnut (Figure 16). Due to the small number of plots in successional habitats (n=6) and the diversity of species contained in those plots, species-specific recruitment and mortality rates could not be reliably calculated.

Park managers should be vigilant for thousand cankers disease on black walnut, especially along the floodplain of the Delaware River. This disease complex is native to the western United States and is the result of the combined activity of a fungus (*Geosmithia morbida*) and the walnut twig beetle (*Pityophthorus juglandis*). Thousand cankers disease has been found in southeastern Pennsylvania, not far from DEWA's southern boundary. The park's Black Walnut Bottomland Forest community would be strongly affected by this disease complex. See <http://www.thousandcankers.com/home.php> for additional information and resources.

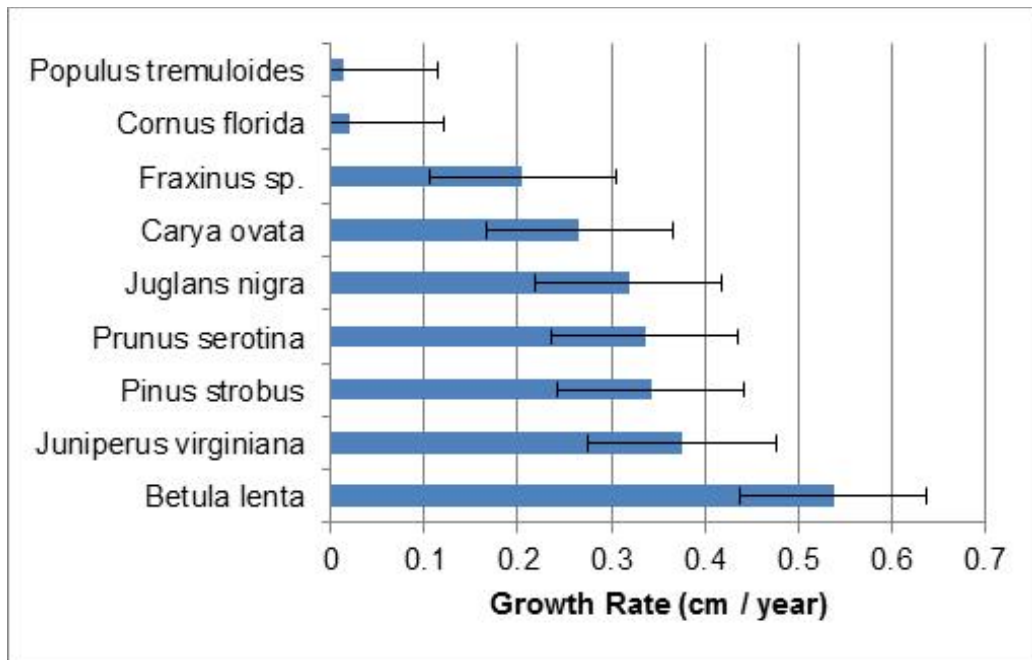


Figure 16. Growth rate of common tree species in early successional habitats in DEWA. Error bars represent 1 standard error above and below the mean.

In mesic habitats, canopy composition appears stable, since the species with solid recruitment and growth rates are the dominant trees, including American beech, red maple, red oak, white pine, black cherry, and tuliptree (Figures 17 and 18). In the longer term, sugar maple may become more prominent in these habitats because sugar maple and ash species comprise a large majority of the seedling layer and the shade tolerant sugar maple seedlings are adapted to survive under closed canopy forest. The ash seedlings are not predicted to survive to canopy height unless a biological control for emerald ash borer is discovered. Emerald ash borer has not been detected in the counties that contain DEWA, however, it will likely invade the park in the next few years, or it may already occur in the park but has gone undetected. Park managers should be on the lookout for signs of borer infestation and ash decline (see <http://www.emeraldashborer.info/>).

Black cherry experienced elevated mortality much higher than the recruitment (Figure 18), likely a result of shading by the maturing canopy and periodic defoliation from eastern tent caterpillars (*Malacosoma americanum*). Hemlock and American elm are declining in these habitats due to a lack of recruitment and elevated mortality caused by hemlock woolly adelgid, elongate hemlock scale (*Fiorinia externa*), Dutch elm disease and elm yellows, respectively. Musclewood (*Carpinus caroliniana*) had the highest recruitment rate, driven strongly by the recruitment of three musclewood trees into one plot, along with lower recruitment in other plots. This understory species typically persists as a small tree, and numerous individuals crossed the threshold to be tallied as trees between the two censuses.

Other understory trees such as flowering dogwood (*Cornus florida*), eastern red cedar, and hop hornbeam (*Ostrya virginiana*) had the slowest growth rates, which is typical given their shaded location under the canopy (Figures 18 and 19).



Figure 17. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in mesic habitats in DEWA. Colored lines highlight notable shifts in species composition.

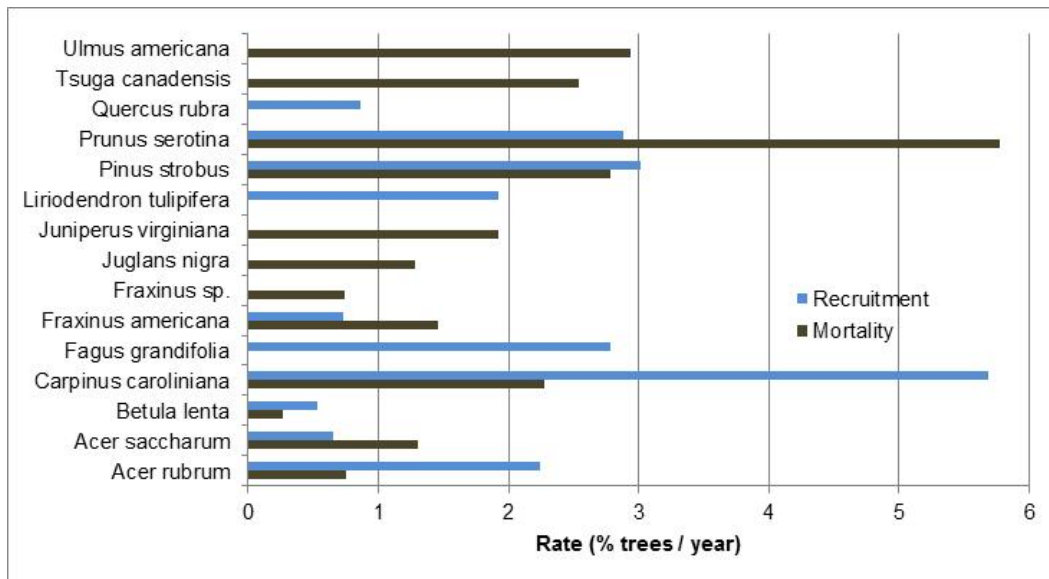


Figure 18. Mortality and recruitment rates (percent / year) for common tree species in mesic habitats in DEWA.

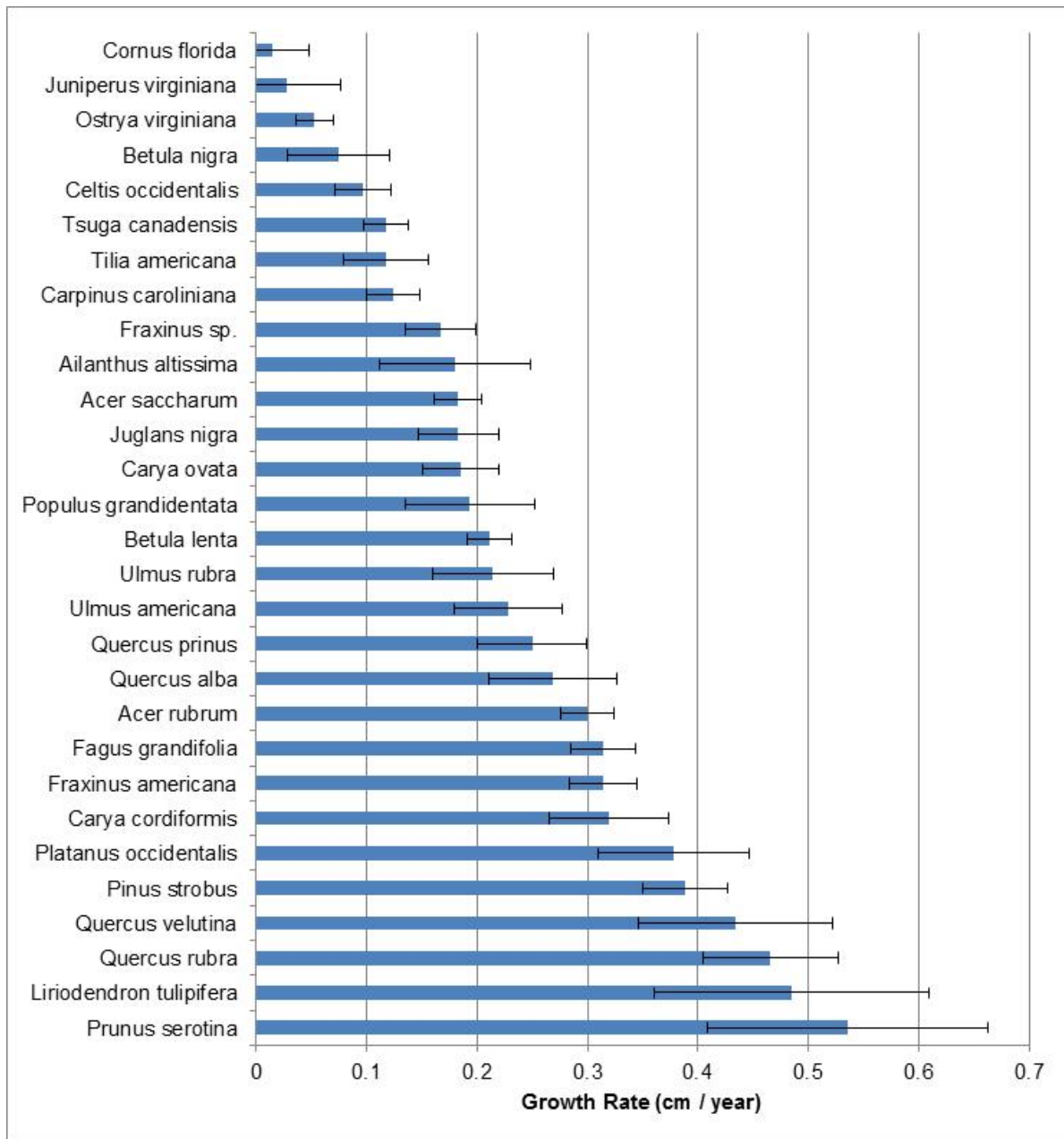


Figure 19. Growth rate of common tree species in mesic habitats in DEWA. Error bars represent 1 standard error above and below the mean.

In DEWA's xeric habitats, black birch, white pine, and scarlet oak (*Quercus coccinea*) are increasing in abundance in the canopy, with much higher recruitment than mortality (Figure 20). Red maple, black oak, red oak, and pignut hickory (*Carya glabra*) appear stable, with slightly higher recruitment than mortality. Red, black, and scarlet oak also show strong growth rates (Figure 21). The dominant canopy trees are represented in the seedling layer; however, red maple seedlings are disproportionately abundant (Figure 22). The increase in red maple abundance in xeric oak-dominated forests is a region-wide trend (Abrams 1998), influenced by the restriction of fire in the 20th and 21st centuries (Brose et al. 2013), but possibly also as a result of wetter climate, land-use

change, the loss of foundation and keystone species, and browse from larger white-tailed deer populations (McEwan et al. 2011).

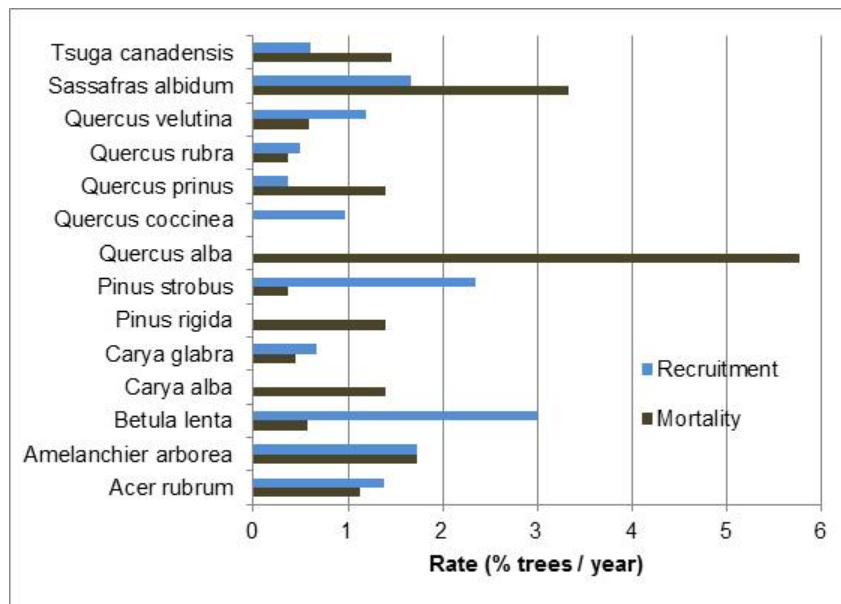


Figure 20. Mortality and recruitment rates (percent / year) for common tree species in mesic habitats in DEWA.

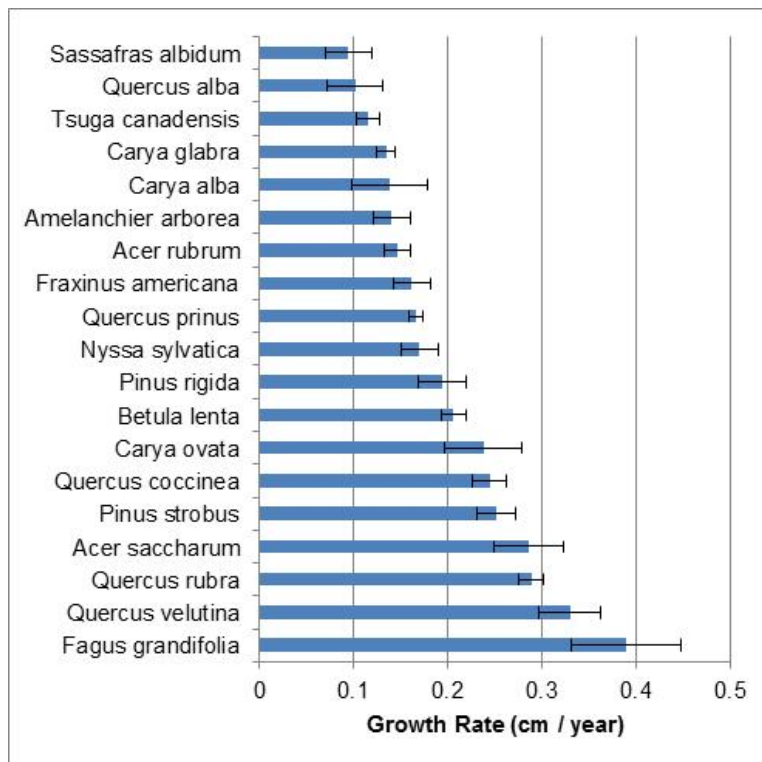


Figure 21. Growth rate of common tree species in xeric habitats in DEWA. Error bars represent 1 standard error above and below the mean.

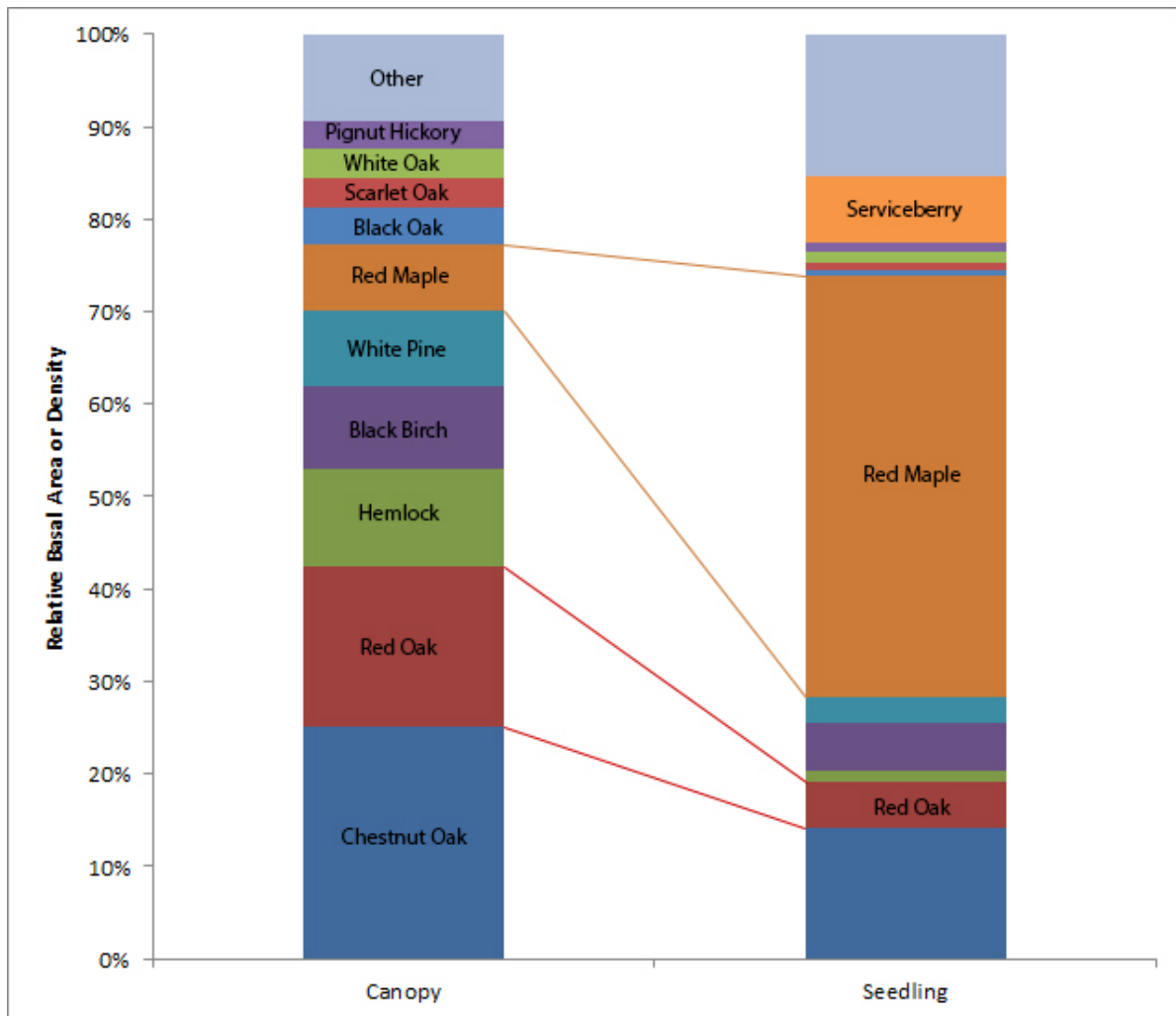


Figure 22. Growth rate of common tree species in xeric habitats in DEWA. Error bars represent 1 standard error above and below the mean.

Species that showed declines (i.e., mortality > recruitment) include white oak (*Quercus alba*), sassafras (*Sassafras albidum*), hemlock, chestnut oak (*Quercus prinus*), pitch pine (*Pinus rigida*), and mockernut hickory (Figure 20). White oak's mortality rate is likely inflated by a single plot in which all three white oaks died, though lower rates of white oak mortality occurred in nearly half of the xeric plots that contained white oak. In general, white oak mortality is likely a result of several years of defoliation by gypsy moth caterpillars (*Lymantria dispar dispar*) in combination with the droughty conditions on ridgetops where white oaks are typically dominant. White oak is the only oak species that lacked recruitment and also exhibited the second slowest growth of common trees in xeric habitats. Regionally, white oak has showed strong declines with little recruitment over the past two centuries, as intensive land-clearing followed by fire suppression had greater negative impact on white oak, compared with other oak species (Abrams, 2003). Hemlock's slow growth rate (Figure 21) is likely caused by infestations of hemlock woolly adelgid and elongate hemlock scale (*Fiorinia externa*). Sassafras exhibited the slowest growth rate (Figure 21) as a result of their subcanopy or overtopped positions in the forest. This shade intolerant tree does not thrive when growing under

other canopy trees. The strongest growth rate occurred in American beech. These beech likely occur on the more mesic of the xeric sites, and the increased moisture and soil fertility lead to stronger tree growth than on typical xeric sites.

Fort Necessity National Battlefield

In general, forests in FONE experienced moderate growth rate and basal area change, moderate mortality, and the lowest recruitment rate compared with other network parks (Figure 3-5). The increase in tree basal area indicates that the forests are maturing. As is typical in middle-aged forests, tree mortality will exceed recruitment as smaller stems are out-competed by larger canopy trees.

However, the only common tree species with recruitment are white ash, black birch, and red maple (Figure 23). Although oak trees comprise 25% of the canopy, oak seedlings make up only 4% of the seedling layer, which is instead strongly dominated by red maple, black cherry, and white ash (Figure 24). Without major disturbances, it is likely that the future canopy will be dominated by red maple, black birch, and black cherry, as opposed to oaks. Park managers can consider the strategic use of prescribed fire and deer exclosure fencing to promote oak regeneration, especially if disturbances such as storms or pest-induced tree mortality create gaps in the forest canopy.

Currently, white ash trees exhibit the fastest growth rate (Figure 25), however, their growth will likely slow (and eventually cease) as emerald ash borer infests the parks' trees. It is also unlikely that the abundant white ash seedlings will develop into canopy trees, unless a biological control for the emerald ash borer is discovered. The slowest growing tree species are understory species such as hawthorn (*Crataegus* sp.) and serviceberry (*Amelanchier arborea*), which are overtopped by mature canopy.

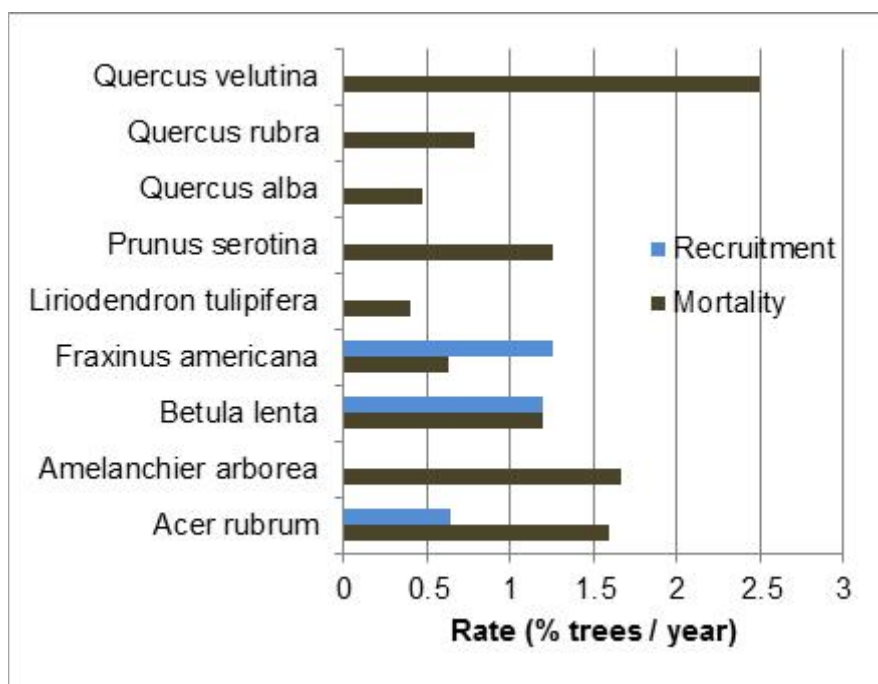


Figure 23. Mortality and recruitment rates (percent / year) for common tree species in FONE.

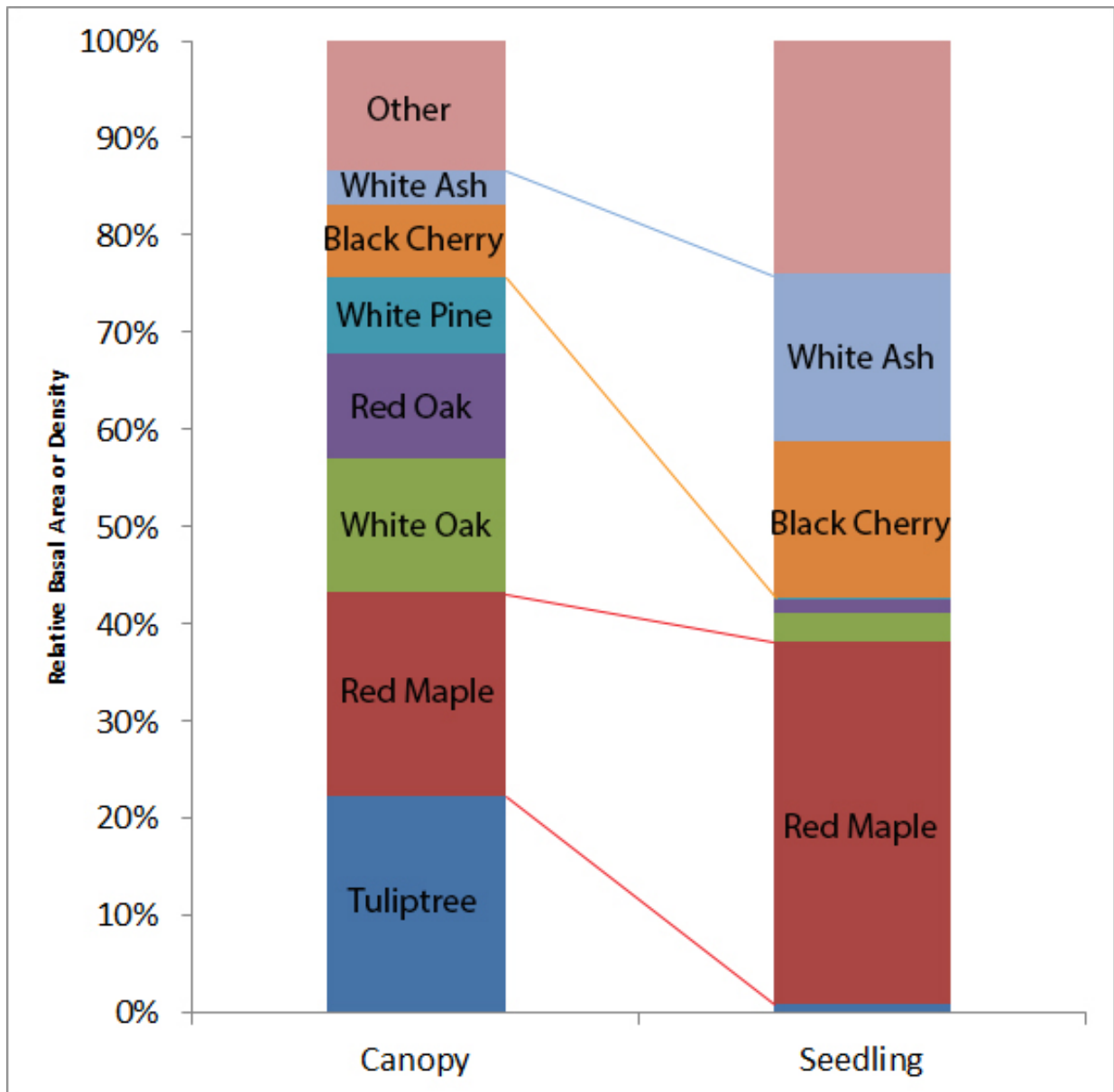


Figure 24. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in FONE. Colored lines highlight notable shifts in species composition.

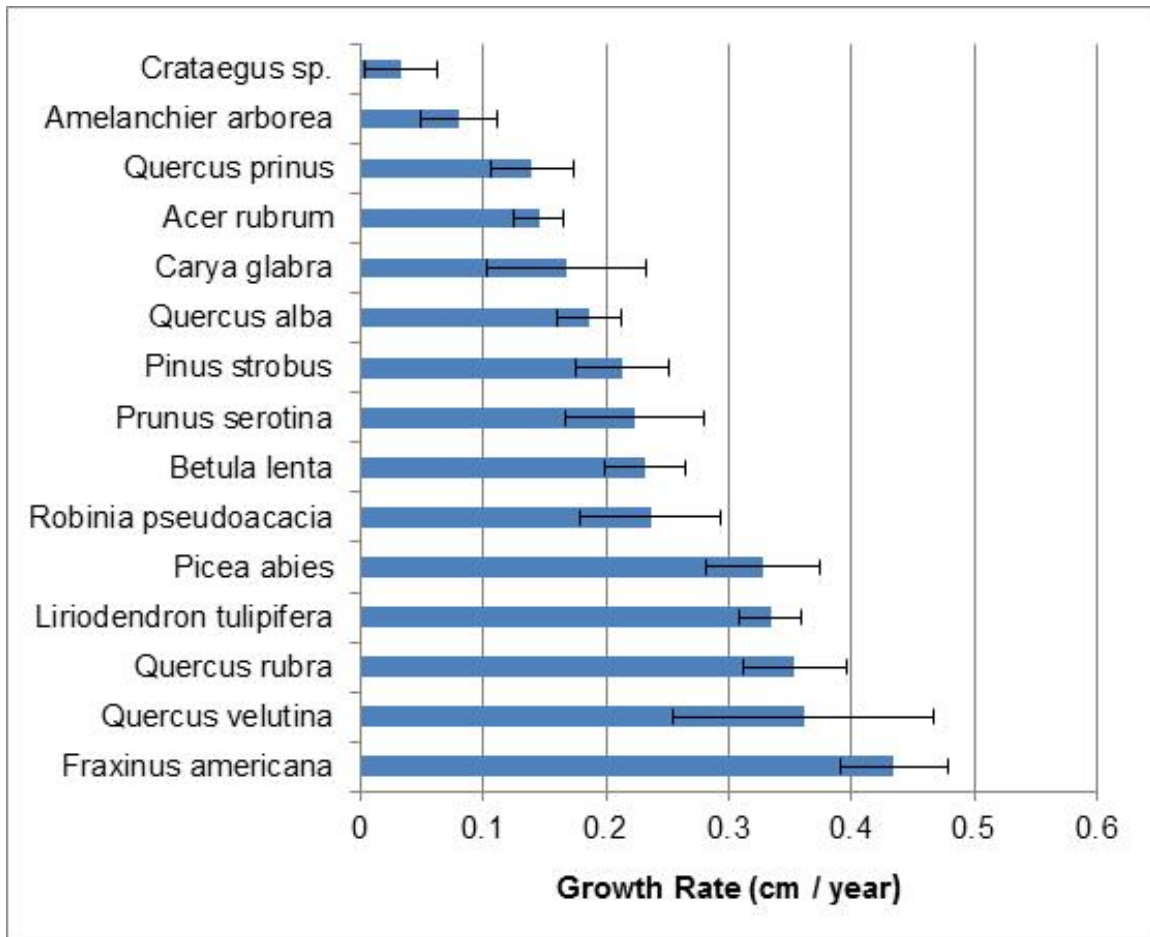


Figure 25. Growth rate of common tree species in FONE. Error bars represent 1 standard error above and below the mean.

Friendship Hill National Historic Site

Much of FRHI is characterized by young forests that have established on former agricultural land or land that was timbered in the mid-1900s. These young trees are growing rapidly on rich mesic soils, giving the park the highest growth rate in the network (Figures 4, 5). Many of these young stands are in the stem exclusion phase, where young trees compete with each other for a place in the future canopy, leading to a high mortality rate as some stems are out-competed.

The current canopy is dominated by tuliptree, red maple, black cherry, sugar maple, and boxelder (*Acer negundo*) (Figure 26). Of these species, only box elder and sugar maple are well represented in the seedling layer and have current tree recruitment (Figure 27). Black cherry and red maple have abundant seedlings, but lack current tree recruitment. These species, along with American elm, are likely to persist as dominant in FRHI's forest. Tuliptree, a pioneer tree that reproduces poorly under a closed canopy, is very underrepresented in the seedling layer and has mortality greater than recruitment. Tuliptree are long lived but will not reproduce without future disturbance. Pin oak (*Quercus palustris*) and sycamore are also underrepresented in the seedling layer. Ash seedlings are disproportionally abundant in the seedling layer compared to the canopy, but these seedlings are likely not to survive to maturity due to the emerald ash borer.

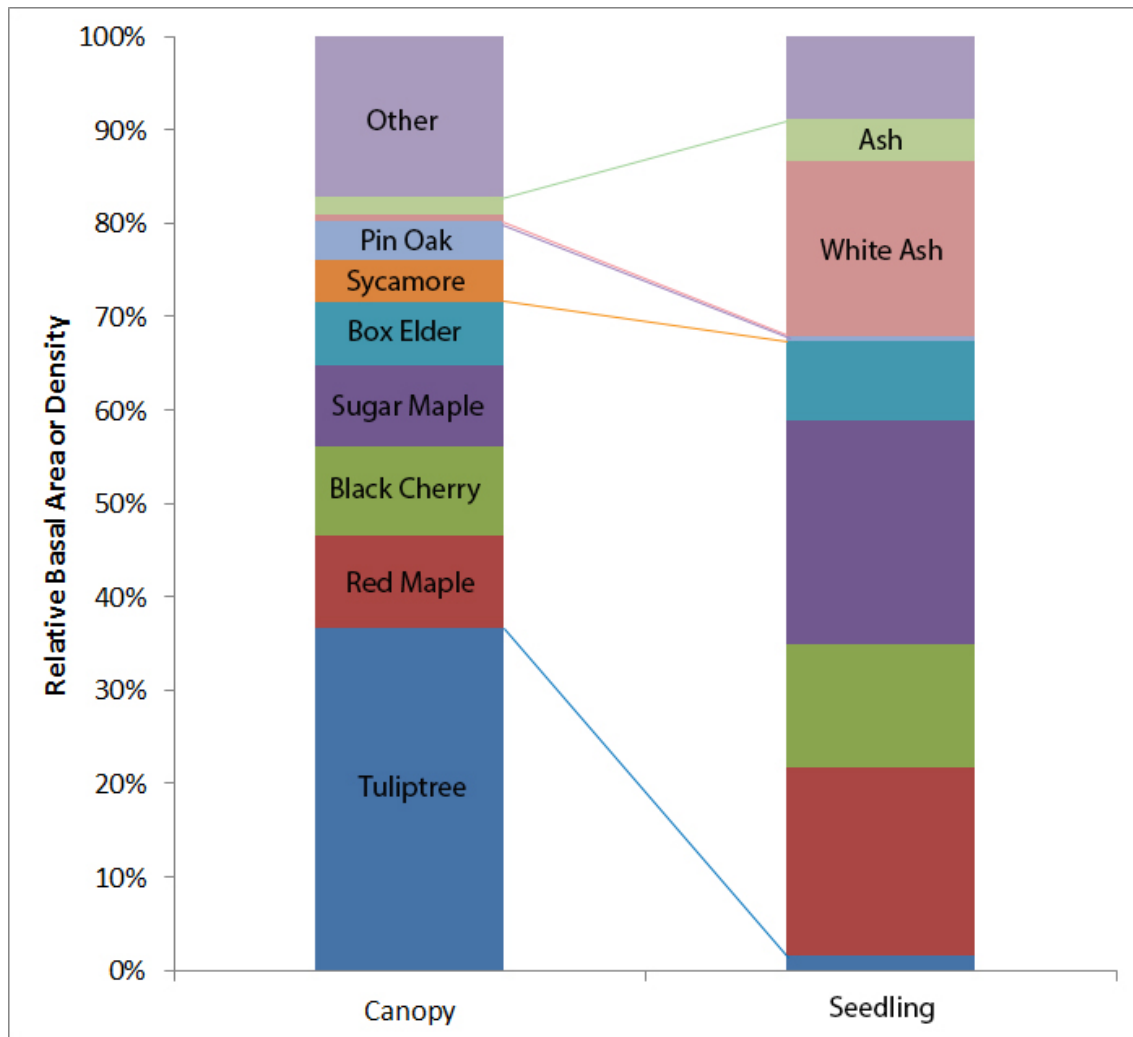


Figure 26. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in FRHI. Colored lines highlight notable shifts in species composition.

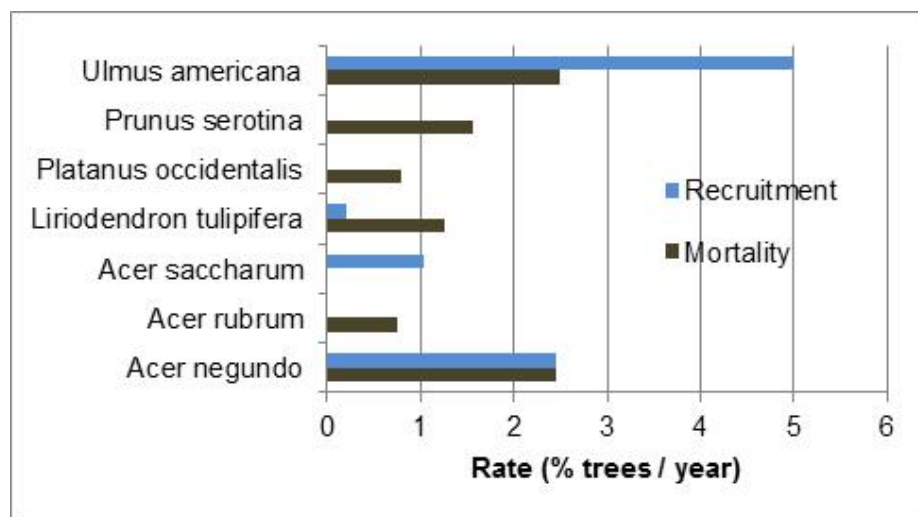


Figure 27. Mortality and recruitment rates (percent / year) for common tree species in FRHI.

Pin oak (*Quercus palustris*) showed very strong growth (Figure 28) due to some open grown trees (n=11) established in the former agricultural land. This species is typically fast growing, and in addition, these individuals are growing in full sun, on fertile soil, and with limited competition. Red oak, tuliptree, American elm, and sycamore exhibited higher growth rates, while slippery elm (*Ulmus rubra*), white oak, red maple, black cherry, American beech, and white ash showed slower growth (Figure 28).

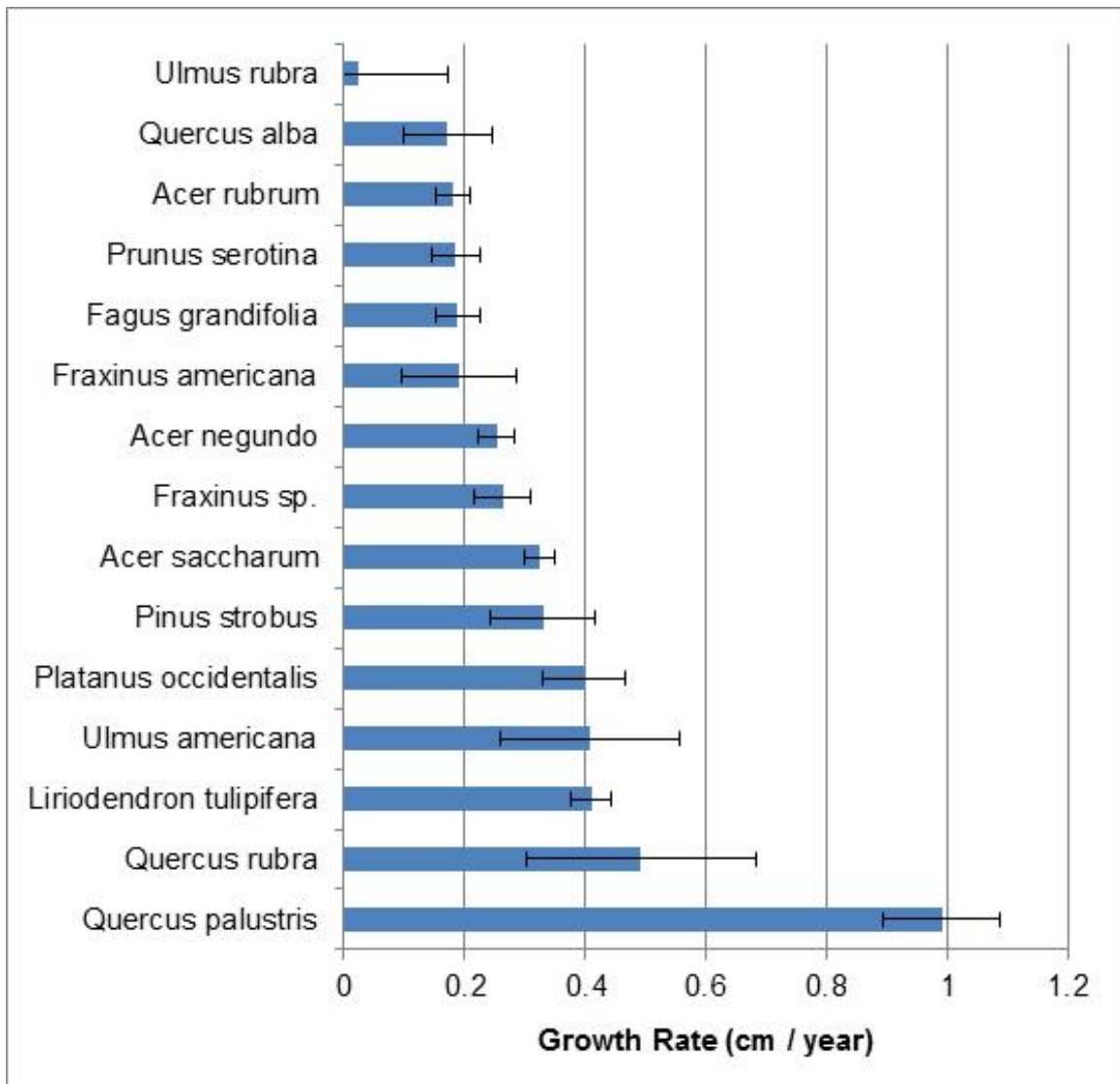


Figure 28. Growth rate of common tree species in FRHI. Error bars represent 1 standard error above and below the mean.

Invasive plant species are one of the most pressing threats to forest health in FRHI. As discussed in the previous monitoring report (Perles et al. 2014c), FRHI contains the most invasive plant species (as measured by total cover and number of invasive plant species per plot) of any network park. Without concerted management efforts, invasive plant species will continue to expand within the park, further degrading forest health.

Gauley River National Recreation Area

In the summer of 2012, severe storms caused widespread damage throughout GARI, toppling so many trees that the standing living basal area decreased by -0.14% in the park (Figure 4), compared with an average increase in basal area of 4.45% across the whole network. For those trees that survived the storms, growth is typical for the ERMN network (Figure 5). GARI has the second highest recruitment rate in the network (Figure 3), driven primarily by sourwood recruitment into the subcanopy in xeric habitats (Figure 29).

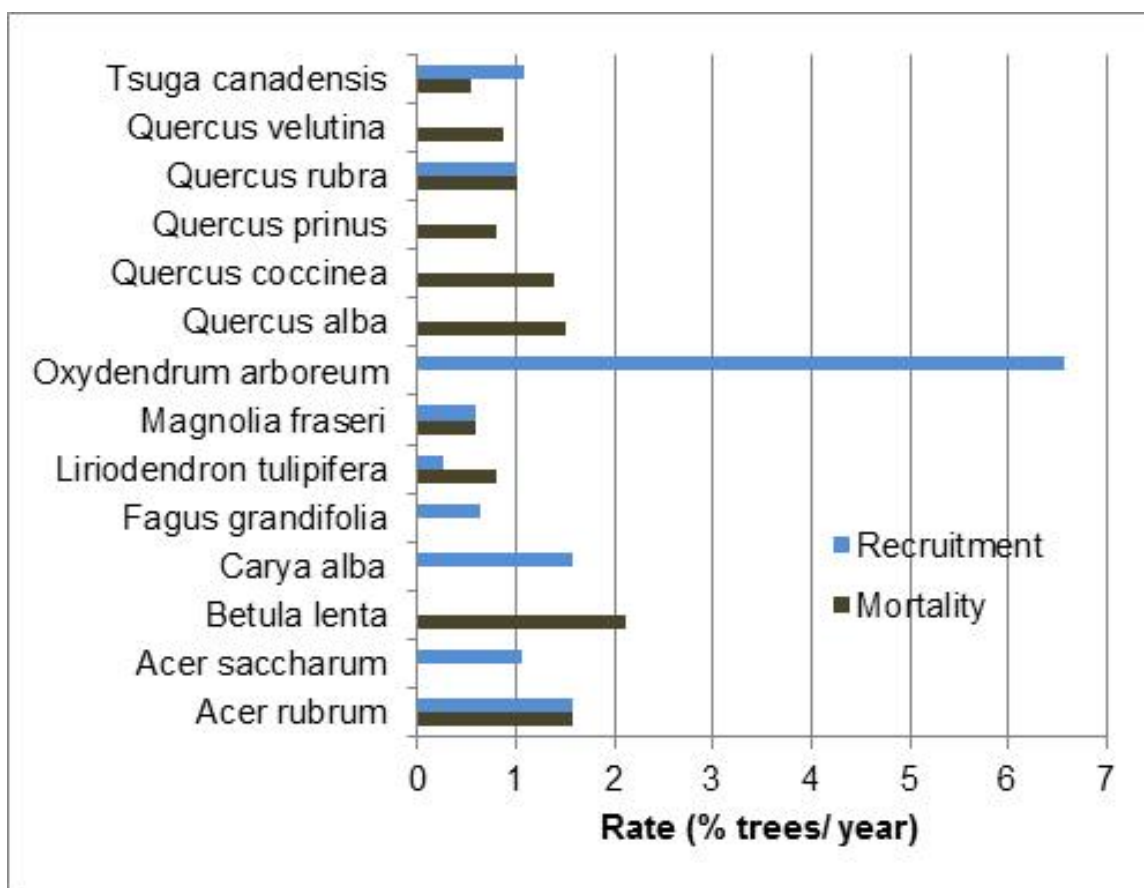


Figure 29. Mortality and recruitment rates (percent / year) for common tree species in xeric habitats in GARI.

In xeric habitats, hemlock, tuliptree, a variety of oaks, and red maple dominate the canopy (Figure 30). Although oak recruitment has been lacking over the past four years in all oak species except red oak (Figure 29), it is notable that chestnut oak and white oak show similar relative abundance in the canopy and seedling layers. Abundant oak seedlings are crucial to oak reproduction in the storm-damaged canopy gaps. Growth rates for oaks range from average for white oak (0.18 cm /year) to high for scarlet oak (0.49 cm / year) (Figure 31).

Red maple is much more abundant in the seedling layer than the canopy (Figure 30), and had recruitment equal to mortality (Figure 29), suggesting that this species will be more abundant in coming years. Over the past four years, mockernut hickory, hemlock, sugar maple, and American

beech recruited trees into the canopy, although none of these species are well represented in the seedling layer. Tuliptree and black birch experienced higher mortality than recruitment and are also underrepresented in the seedling layer. Sourwood had the highest recruitment rate, driven strongly by the recruitment of three sourwood trees into one plot, along with lower recruitment in other plots. This understory species typically persists as a small tree, and numerous individuals crossed the threshold to be included in the canopy between the two censuses.

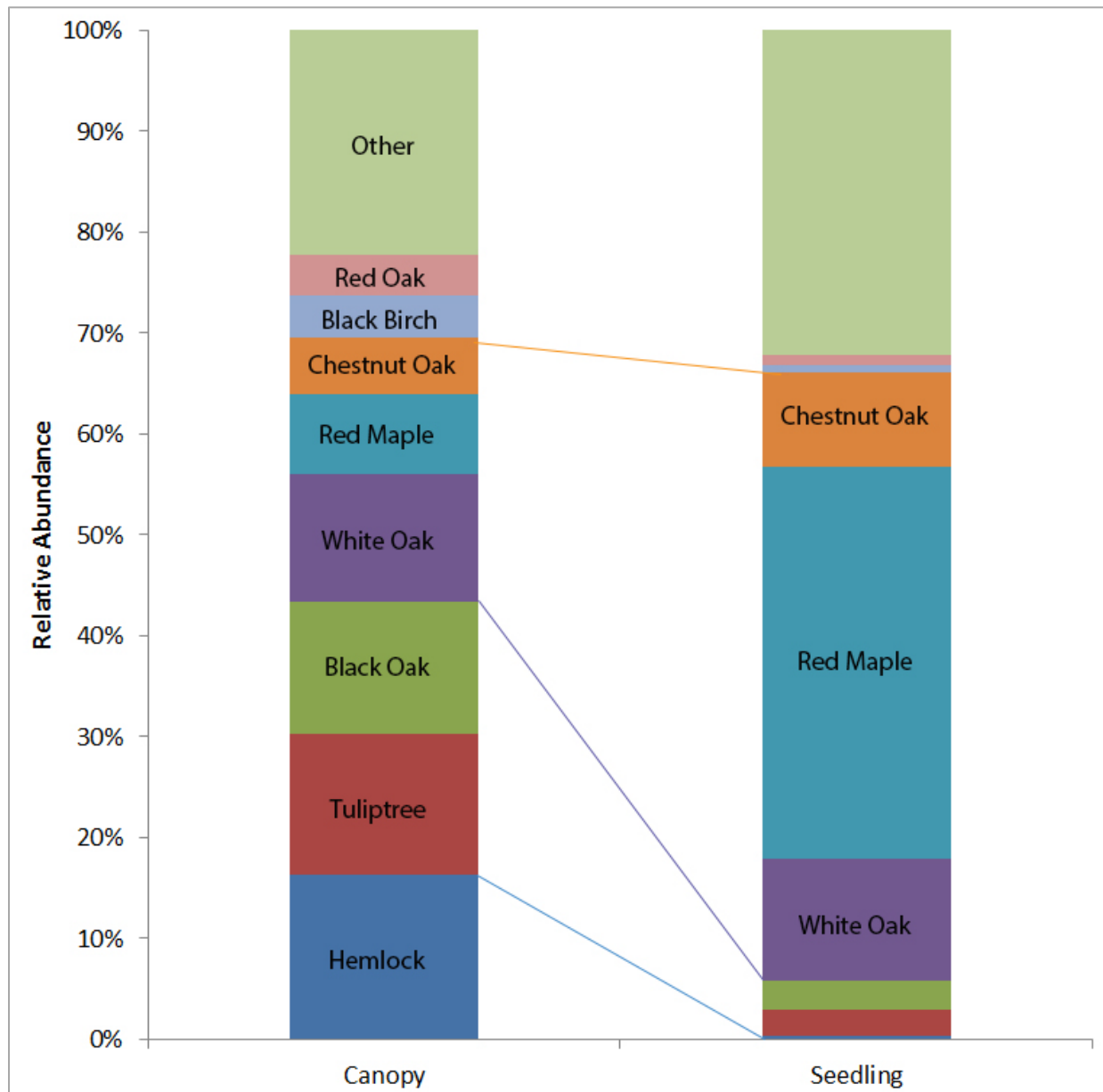


Figure 30. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in xeric habitats in GARI. Colored lines highlight notable shifts in species composition.

Of the common canopy trees, tree-of-heaven (*Ailanthus altissima*) demonstrated considerably faster growth compared with the other species (Figure 31). This non-native invasive species occurred in three plots, one of which (GARI.003) contained 15 tree-of-heaven individuals. The fast growth and prolific reproduction of this species makes it a significant threat to GARI's forests, especially given the wide-spread disturbances caused by the 2012 storms. If tree-of-heaven establishes in the canopy gaps created by the storms, it will out-grow the native trees and form extensive stands of tree-of-heaven. Fortunately, a promising biocontrol for tree-of-heaven has been discovered in the mid-Atlantic region to effectively eliminate tree-of-heaven from forest stands (Kasson et al 2014). A specific-strain of Verticillium wilt, caused by the fungus *Verticillium nonalfalfae*, causes 100% mortality in tree-of-heaven, and spreads throughout tree-of-heaven colonies via root graphs. The most effective method to inoculate a tree-of-heaven stand with the fungus is to wound the root collar of a healthy tree-of-heaven and then place logs from infected trees on top of the wounds (Davis 2014). Rigorous testing of the impacts of this strain of Verticillium wilt on native woody species has shown zero to negligible impacts on native woody species (Kasson et al. 2015). Further testing on agricultural crops and other forest plants is being conducted by a US Forest Service laboratory in Ohio (Joanne Rebbeck, personal communication). This biocontrol could be crucial to ensuring regeneration of native trees in the vast storm-damaged areas of GARI.

Of the common native trees in GARI, hickories (*Carya* spp.) had the slowest growth rates, along with black gum (*Nyssa sylvatica*) and black cherry (Figure 31). Hickories are typically slow-growing, while black gum is often a subcanopy tree, and growth is limited by competition. Slow growth in black cherry is likely a result of shading by the maturing canopy and periodic defoliation from eastern tent caterpillars.

In GARI's mesic habitats, sugar maple and red maple will likely continue to dominate in the canopy, based on their recruitment exceeding mortality (Figure 32) and their strong representation in the seedling layer (Figure 33). Black birch may remain dominant in the short term, due to the typical recruitment and zero mortality, but very low representation in the seedling layer indicates its long term dominance is questionable. Tuliptree and basswood experienced higher mortality than recruitment, as well as very low representation in the seedling layer, suggesting that these trees will not be as important in the future canopy without disturbance that would increase light availability to the shade-intolerant seedlings of these species. Ash and paw paw (*Asimina triloba*) were more abundant in the seedling layer than in the canopy, but neither of these species is expected to reach the canopy due to emerald ash borer and the species' growth form as a small tree, respectively.

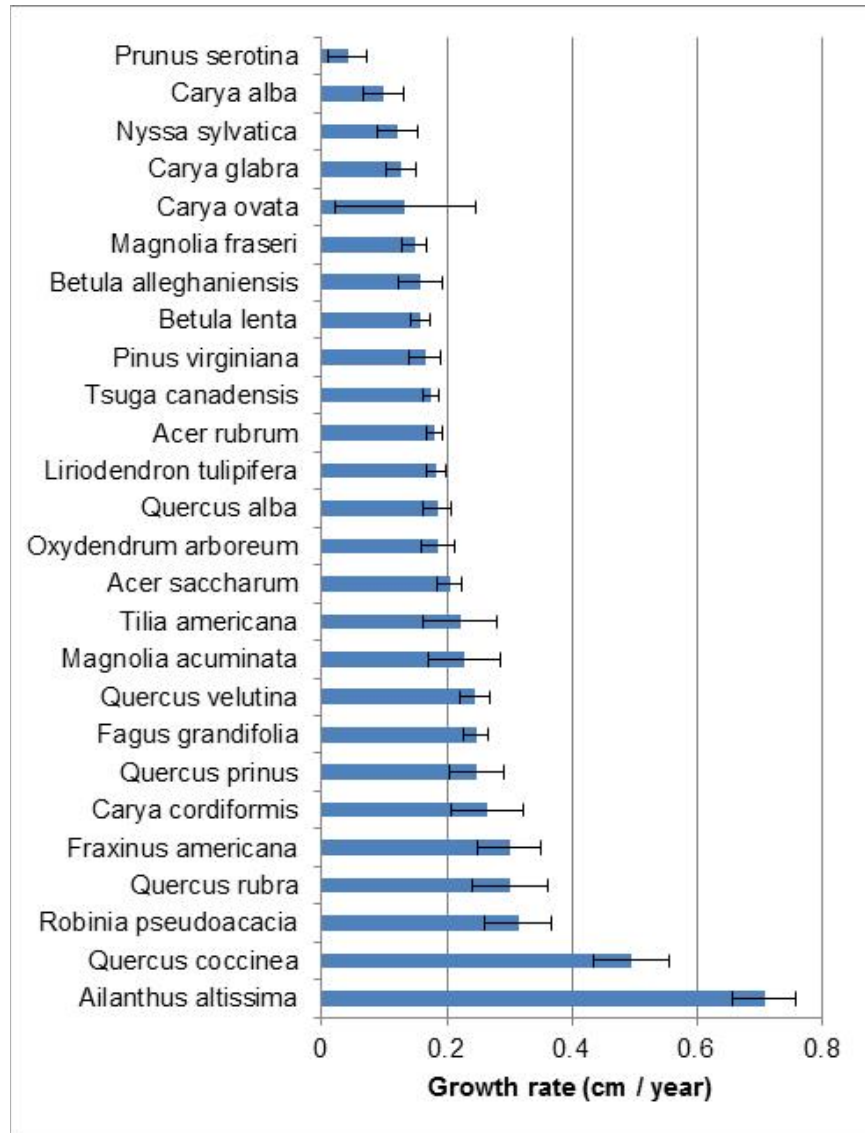


Figure 31. Growth rate of common tree species in GARI. Error bars represent 1 standard error above and below the mean.

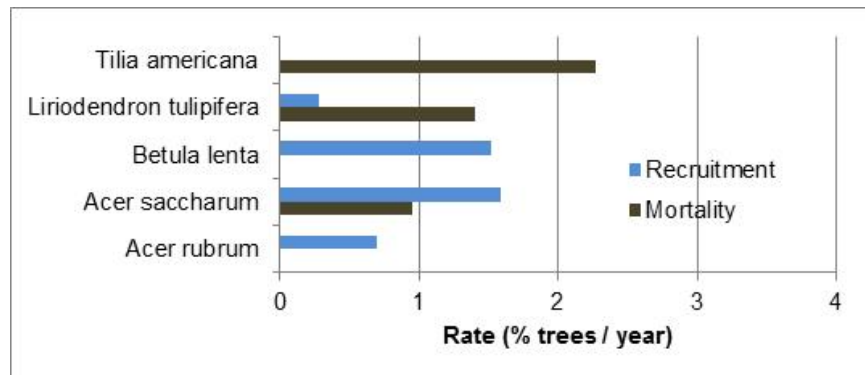


Figure 32. Mortality and recruitment rates (percent / year) for common tree species in mesic habitats in GARI.

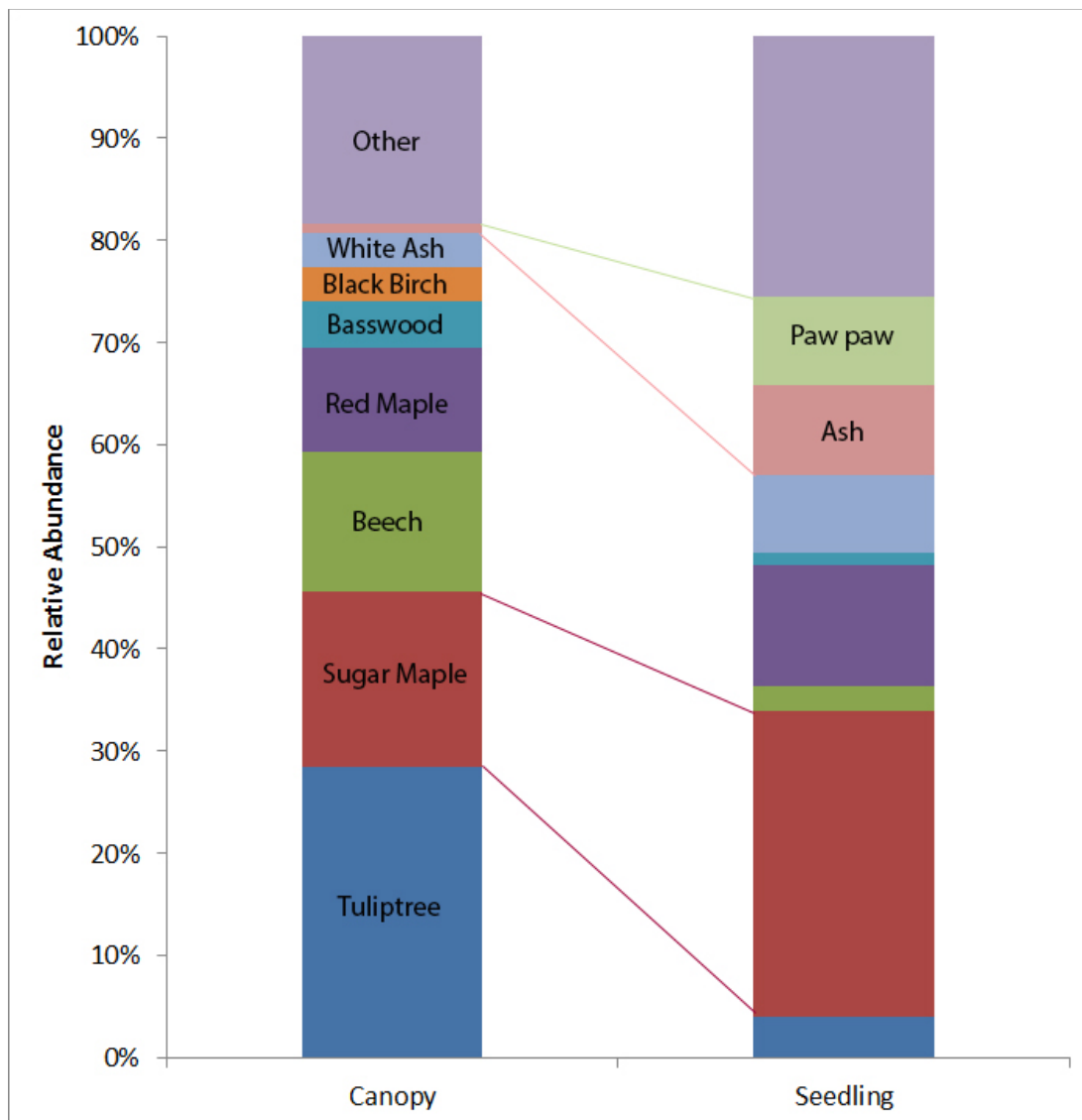


Figure 33. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in mesic habitats in GARI. Colored lines highlight notable shifts in species composition.

Johnstown Flood National Memorial

The young forests at JOFL are dominated by red maple and black cherry, both of which have average growth and relatively stable recruitment and mortality (Figures 34-36). While red maple seedlings are abundant, black cherry seedlings are scarce, suggesting that black cherry may not dominate in the future canopy without disturbance.

As the forest matures, early-successional species such as black locust (*Robinia pseudoacacia*) decline, as shown by the species' slow growth, high mortality and the absence of black locust seedlings (Figures 34-36). In two plots, apple trees (*Malus* sp.) are maturing such that they are now counted as trees, leading to high recruitment, although these individuals will remain small trees, overtopped by the canopy. Hawthorn (*Crataegus* sp.), black birch, and sugar maple experienced typical mortality, with recruitment lacking or altogether absent.

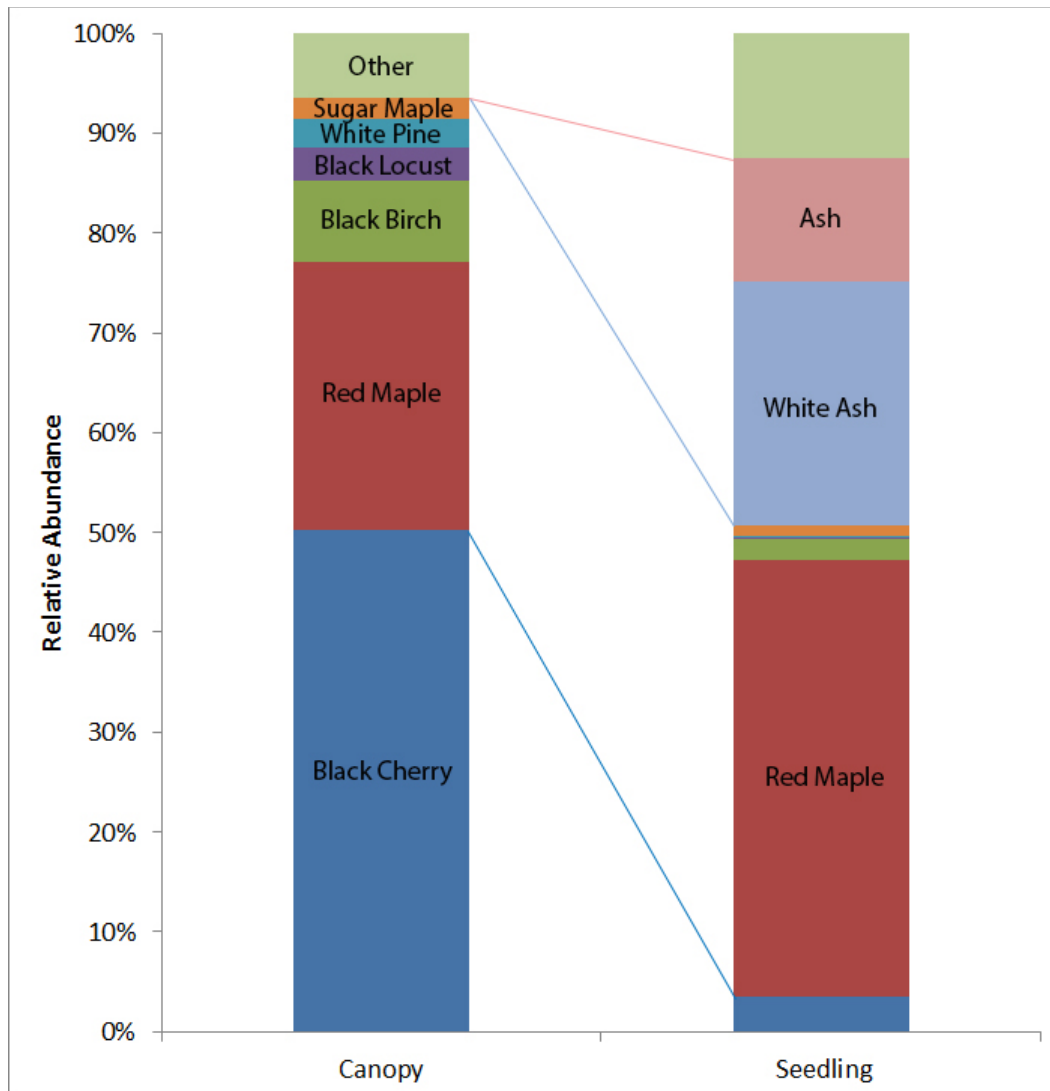


Figure 34. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in JOFL. Colored lines highlight notable shifts in species composition.

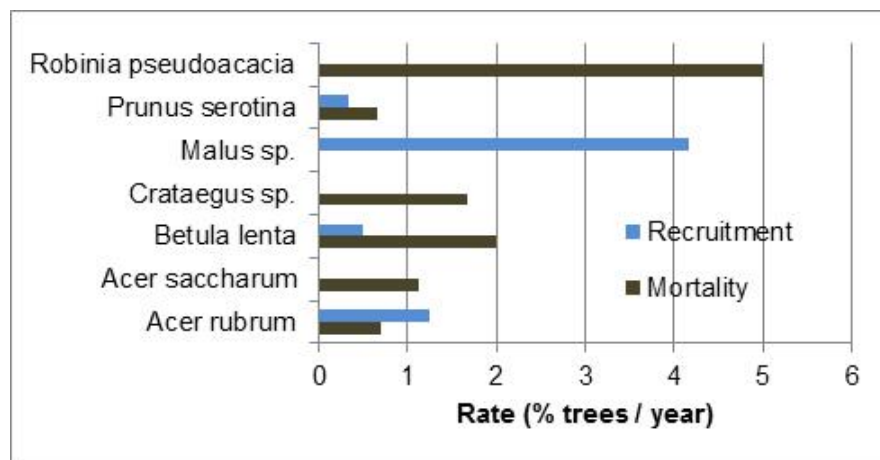


Figure 35. Mortality and recruitment rates (percent / year) for common tree species in JOFL.

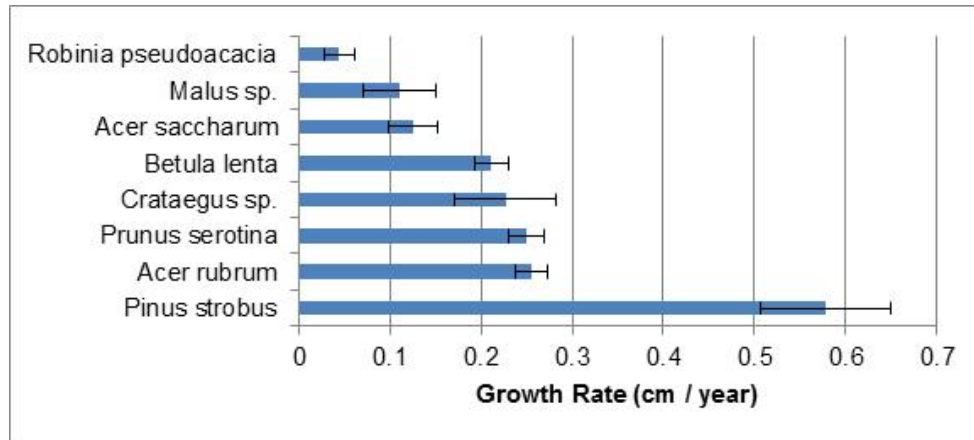


Figure 36. Growth rate of common tree species in JOFL. Error bars represent 1 standard error above and below the mean.

Though white pines are not abundant in the canopy, this species is the fastest growing in the park (Figure 36). White ash and ash species seedlings are much more abundant than ash canopy trees (Figure 34), these seedlings will likely not survive to maturity unless a biological control is discovered for the emerald ash borer.

During forest monitoring in JOFL, two other important changes in forest health were noted. In 2010, severe dieback on spicebush (*Lindera benzoin*) was observed in the park. Samples were collected and analyzed at Penn State University to investigate possible pest and pathogens that might be responsible for the dieback; however, none were found. Spicebush has since begun to recover in the park with many individuals sprouting from the bases of old stems. The same year, viburnum leaf beetle (*Pyrrhalta viburni*), was discovered in the park and has been responsible for severe decline in several viburnum species (*Viburnum* spp.) Viburnum leaf beetle is a non-native, invasive insect that eventually kills many viburnum species that it feeds on. For more information on viburnum leaf beetle, see <http://www.hort.cornell.edu/vlb/index.html>.

New River Gorge National River

Compared to other network parks, the forests in NERI show moderate growth, recruitment and mortality (Figure 3-5). Some notable species-specific dynamics are occurring in both mesic and xeric habitats.

In mesic habitats, the non-native exotic species tree-of-heaven has the highest recruitment and the fastest growth rate (Figures 37 and 38). This tree poses a serious threat to the park's forest, especially in disturbed stands, such as those damaged by storms or in canopy gaps left by dying ash trees. A strain of the fungus *Verticillium nonalfalfae* is a promising biological control for tree-of-heaven which is discussed further in the previous GARI section.

Ash (*Fraxinus* spp.) trees, including white ash (*Fraxinus americana*) in NERI had the highest mortality rate of any species in any of the network parks, a result of the longer period of infestation by emerald ash borer compared to the other parks. Ash seedlings are abundant in mesic habitats (Figure 39); however, these individuals are unlikely to survive to maturity unless a biological control

for the borer is discovered. The mortality rates for box elder and hemlock were also elevated (4%), but recruitment equaled mortality for hemlock. For most species in NERI's mesic habitats, mortality exceeded recruitment; however, this is typical of middle-aged closed canopy forests and doesn't indicate a forest health concern.

Aside from tree-of-heaven, species in mesic habitats with higher growth rates include black walnut (*Juglans nigra*), red oak, slippery elm, black oak, and tuliptree, while mockernut hickory, musclewood, cucumber tree (*Magnolia acuminata*), and black gum had the slowest growth rates (Figure 38). Both white ash and hemlock showed moderate growth rates despite the stress from invasive exotic pests.

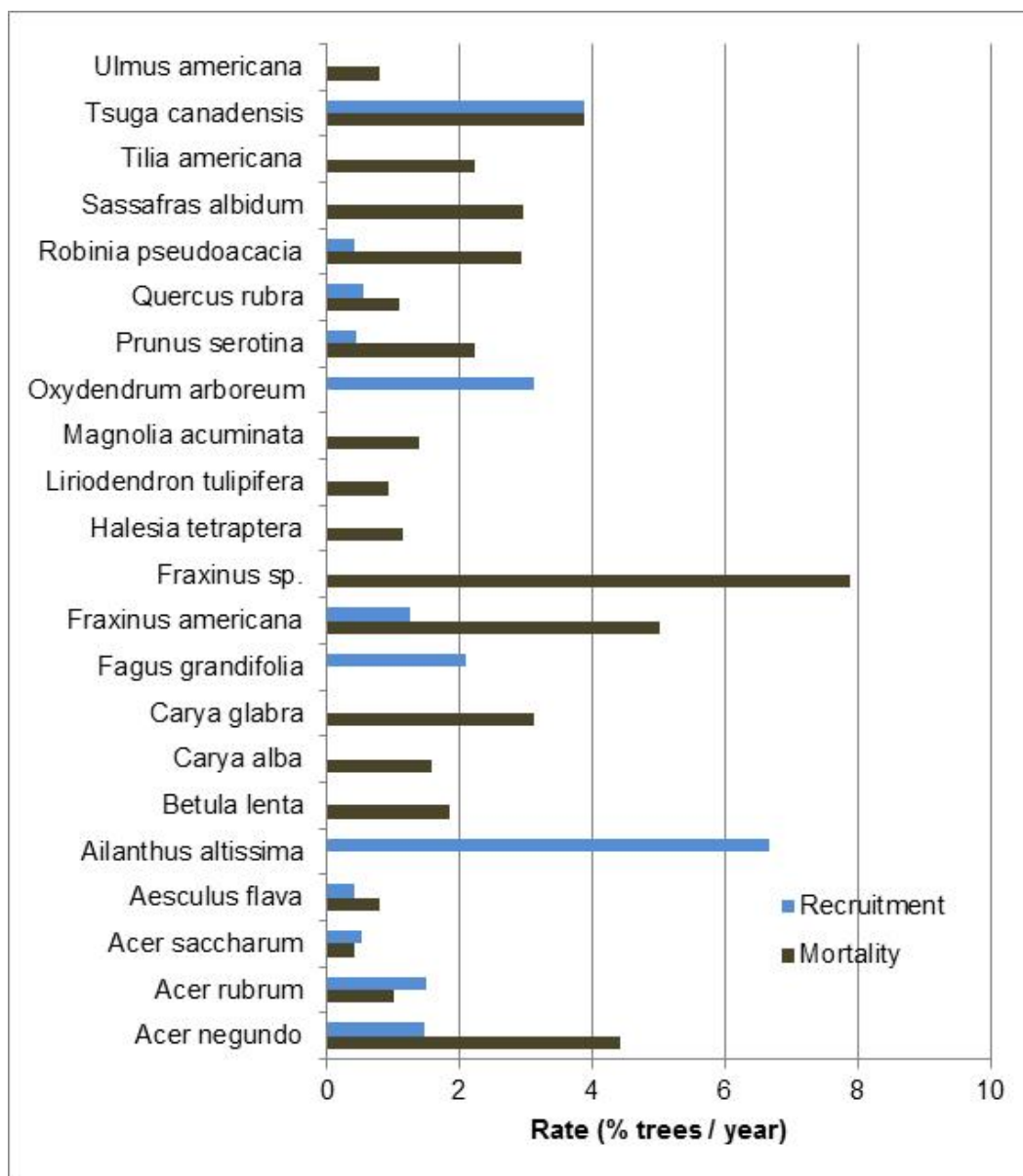


Figure 37. Mortality and recruitment rates (percent / year) for common tree species in mesic habitats in NERI.

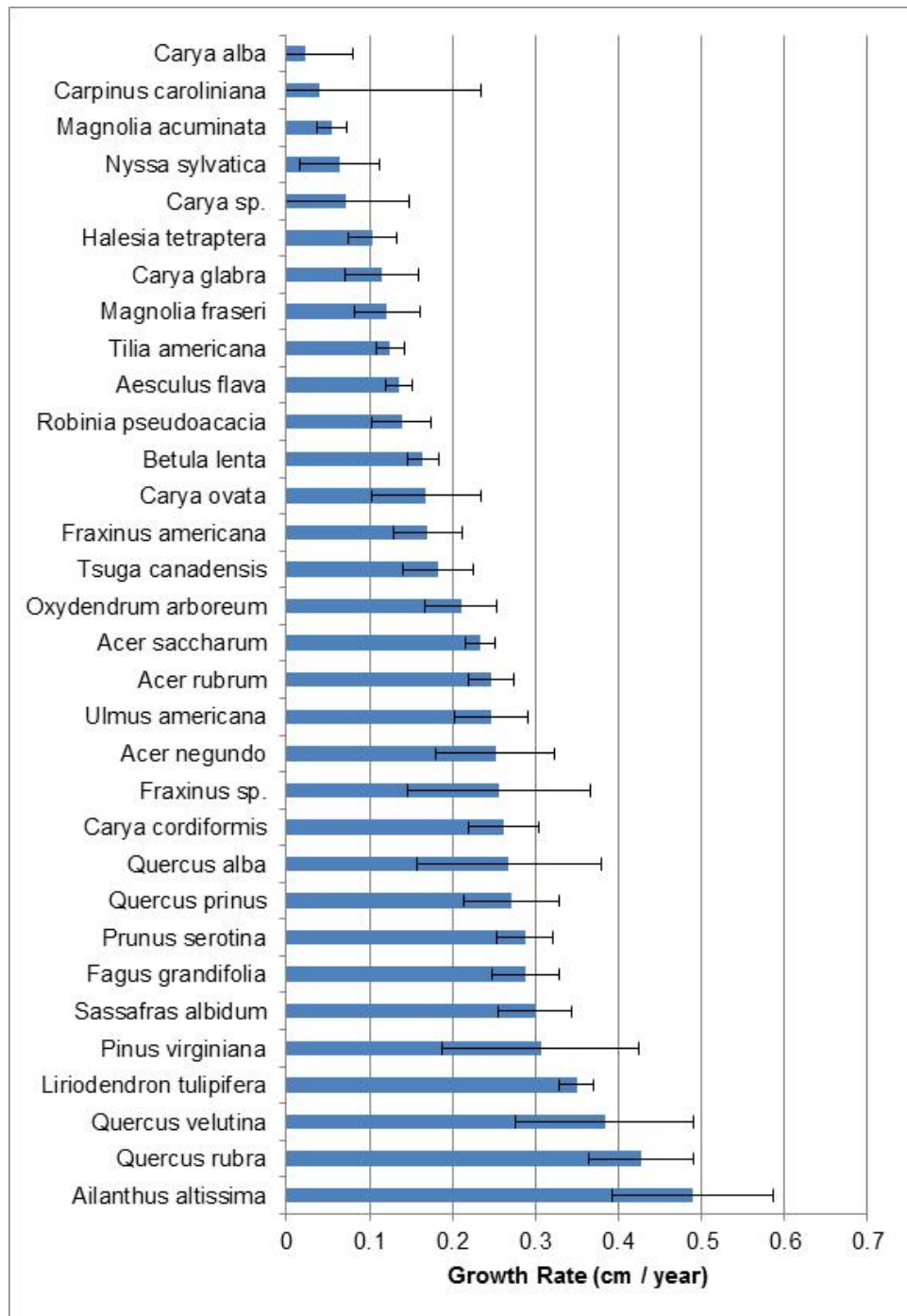


Figure 38. Growth rate of common tree species in mesic habitats in NERI. Error bars represent 1 standard error above and below the mean.

The canopy of mesic forests is diverse, dominated by tuliptree, basswood, sugar maple, red oak, yellow buckeye, and red maple. Without major disturbances, the future canopy will likely continue to be dominated by basswood, sugar maple, red oak, and black cherry, with less tuliptree and more red maple, based on the composition of the seedling layer (Figure 39). Although paw paw seedlings are currently abundant, this species' growth form limits it from becoming a dominant canopy tree.

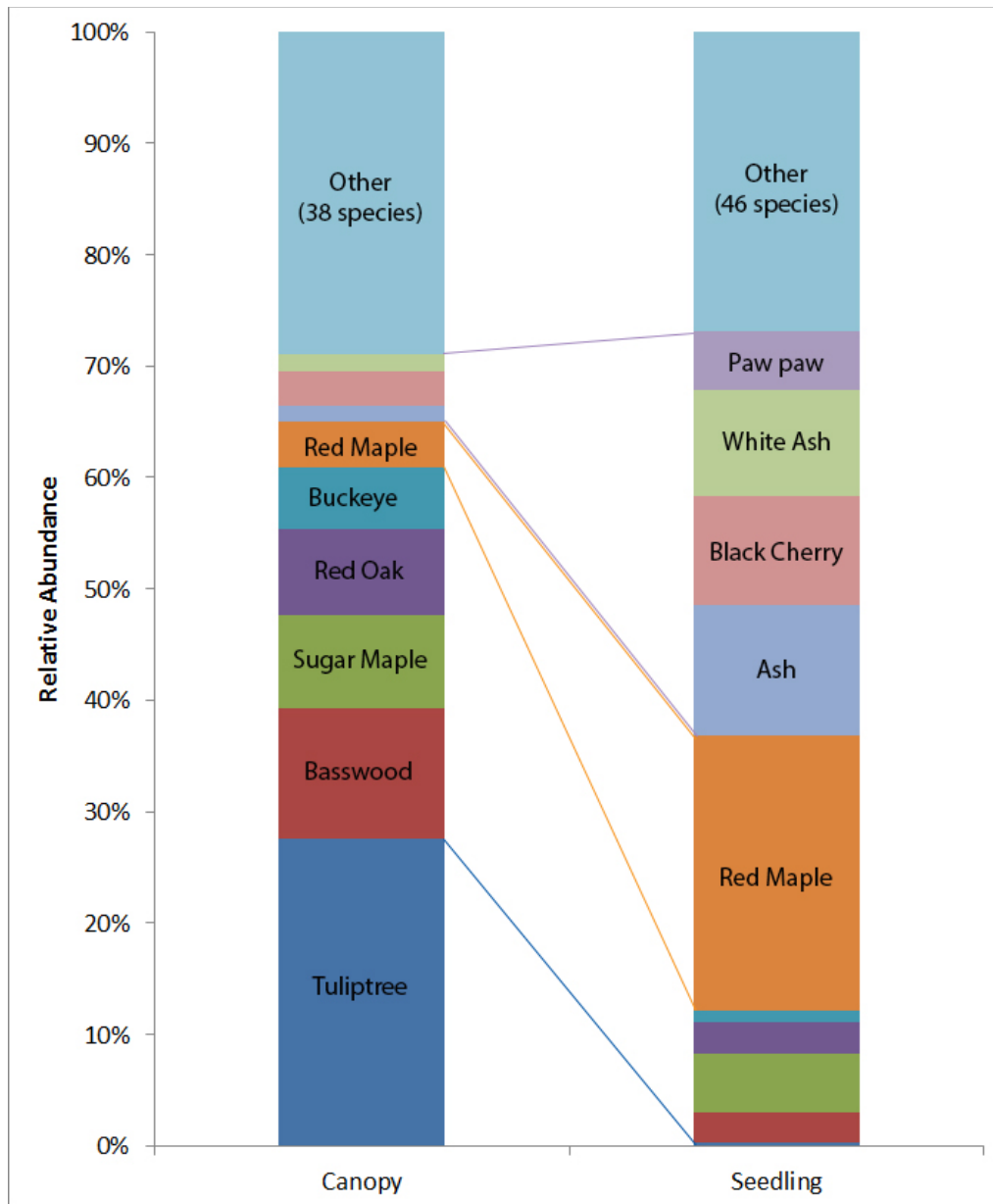


Figure 39. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in mesic habitats in NERI. Colored lines highlight notable shifts in species composition.

In NERI's xeric habitats, the canopy species composition is shifting as the forest matures. Early successional species are being out-competed by the mature oaks and hickories. Sassafras and black locust experienced very high mortality (Figure 40), and black locust had the slowest growth rate (Figure 41), evidence of their decline due to shading. Ashes (including white ash) also showed elevated mortality, due to the emerald ash borer infestations. Black cherry mortality was also higher than expected due to shading by the maturing canopy and periodic defoliation from eastern tent caterpillars. American beech, shagbark hickory and serviceberry had strong recruitment (Figure 40), though serviceberry will persist in the subcanopy due to the species' slow growth rate (Figure 41) and short growth form.

Oak mortality exceeded recruitment (Figure 40), but was not higher than expected for middle-aged forests. The growth rate of oaks ranged from average to high (Figure 41). However, oak seedlings are severely underrepresented when compared with their abundance in the canopy, while red maple seedlings disproportionately dominate the seedling layer (Figure 42). The increase in red maple abundance in xeric oak-dominated forests is a region-wide trend (Abrams 1998), influenced by the restriction of fire in the 20th and 21st centuries (Brose et al. 2013), but possibly also as a result of wetter climate, land-use change, the loss of foundation and keystone species, and browse from larger white-tailed deer populations (McEwan et al. 2011).

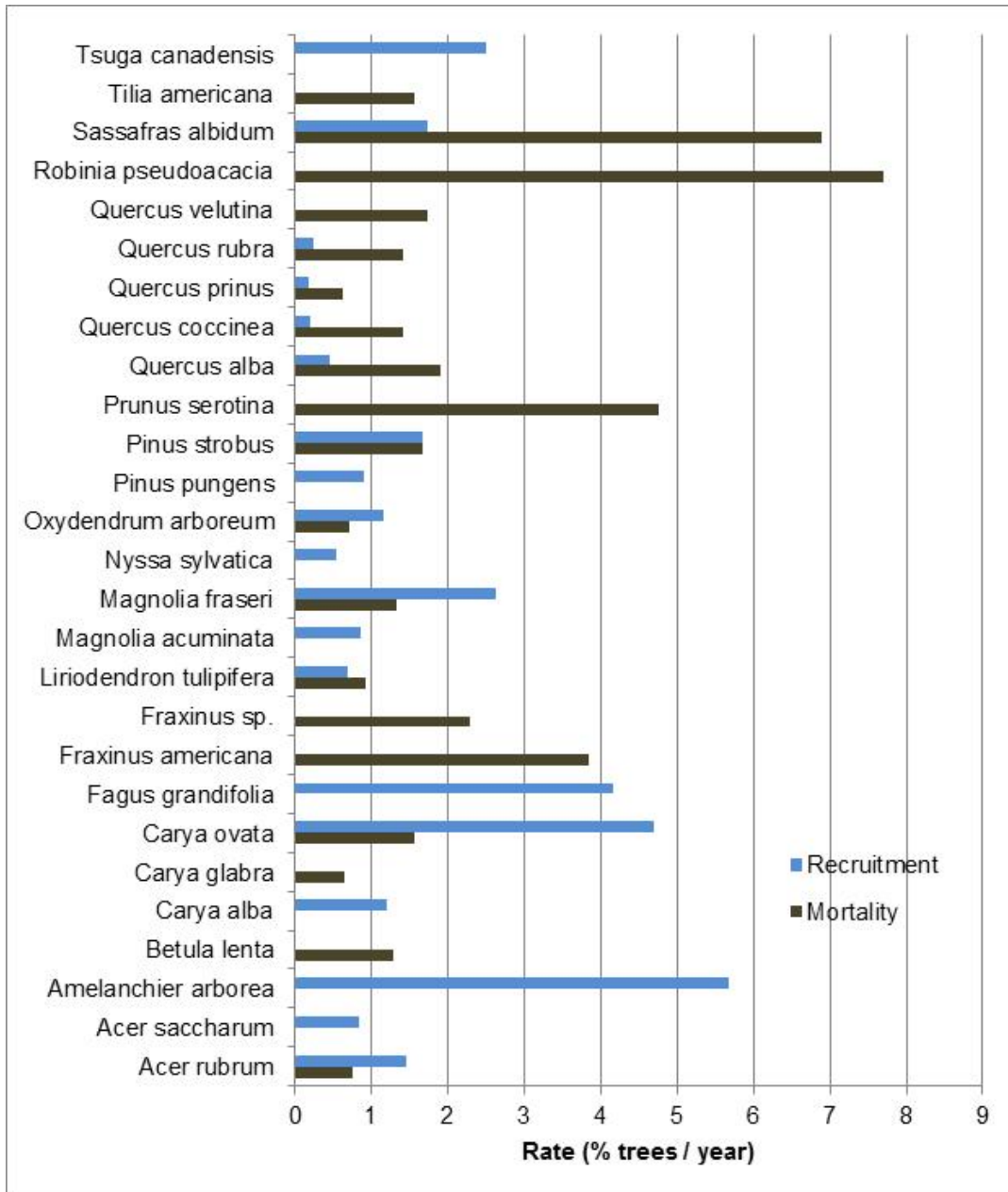


Figure 40. Mortality and recruitment rates (percent / year) for common tree species in xeric habitats in NERI.

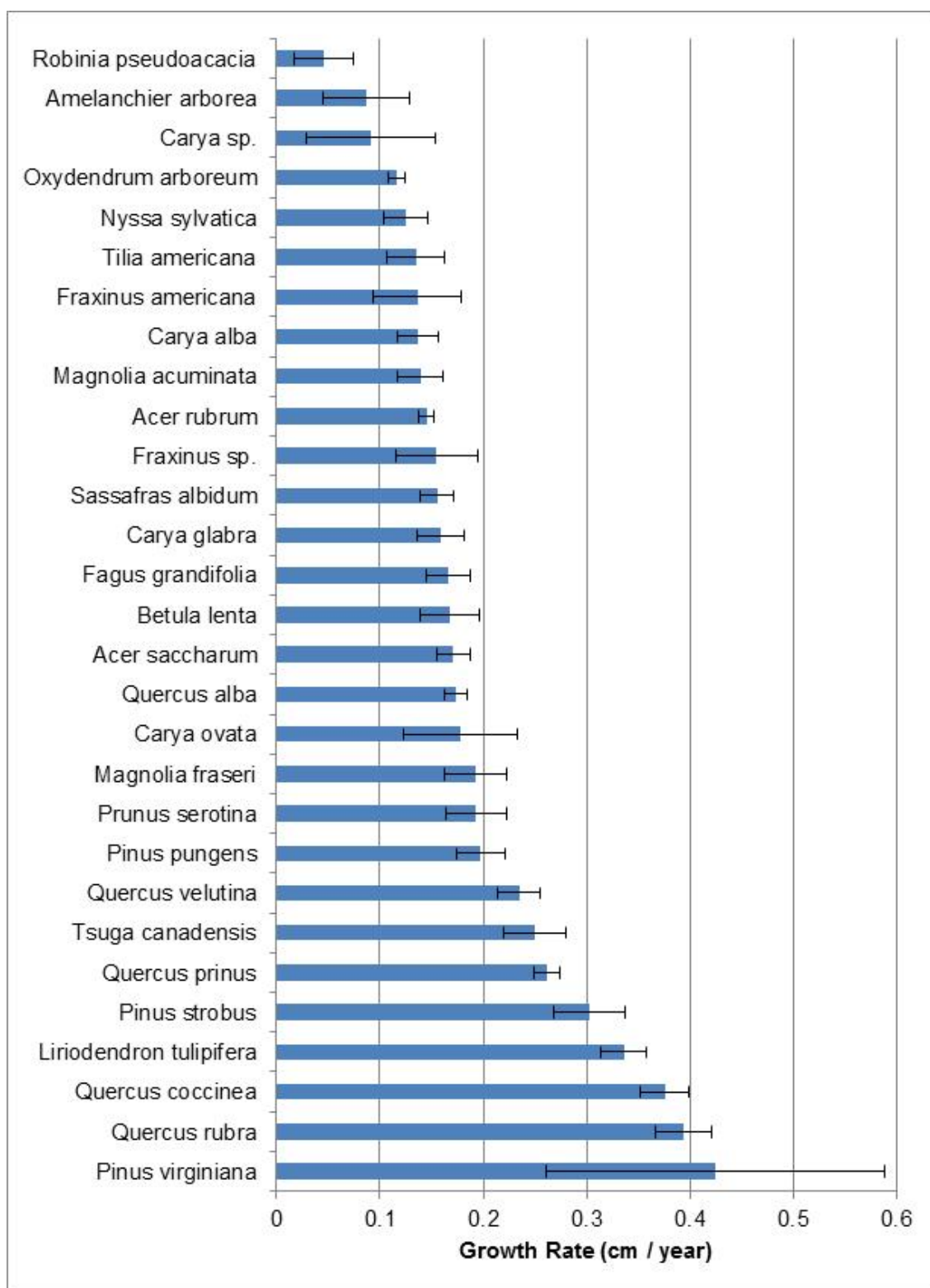


Figure 41. Growth rate of common tree species in xeric habitats in NERI. Error bars represent 1 standard error above and below the mean.

Interestingly, chestnut oak seedlings are as relatively abundant as chestnut oak trees (Figure 42), probably as a result of mast events that produced numerous germinating acorns. Sassafras seedlings were more abundant than sassafras trees (Figure 42); however, these shade-intolerant seedlings will not likely grow to dominant canopy trees without some disturbance to the canopy.

Hemlock showed solid recruitment, no observed mortality, and average growth (Figures 40 and 41), suggesting that hemlocks are persisting in the park despite the presence of hemlock woolly adelgid for numerous years. Monitoring the effects of hemlock insecticide treatments and the release of biological control insects in hemlock-dominated stands should continue since successive years of droughts and / or warm winters could cause significant stress to the hemlock trees.

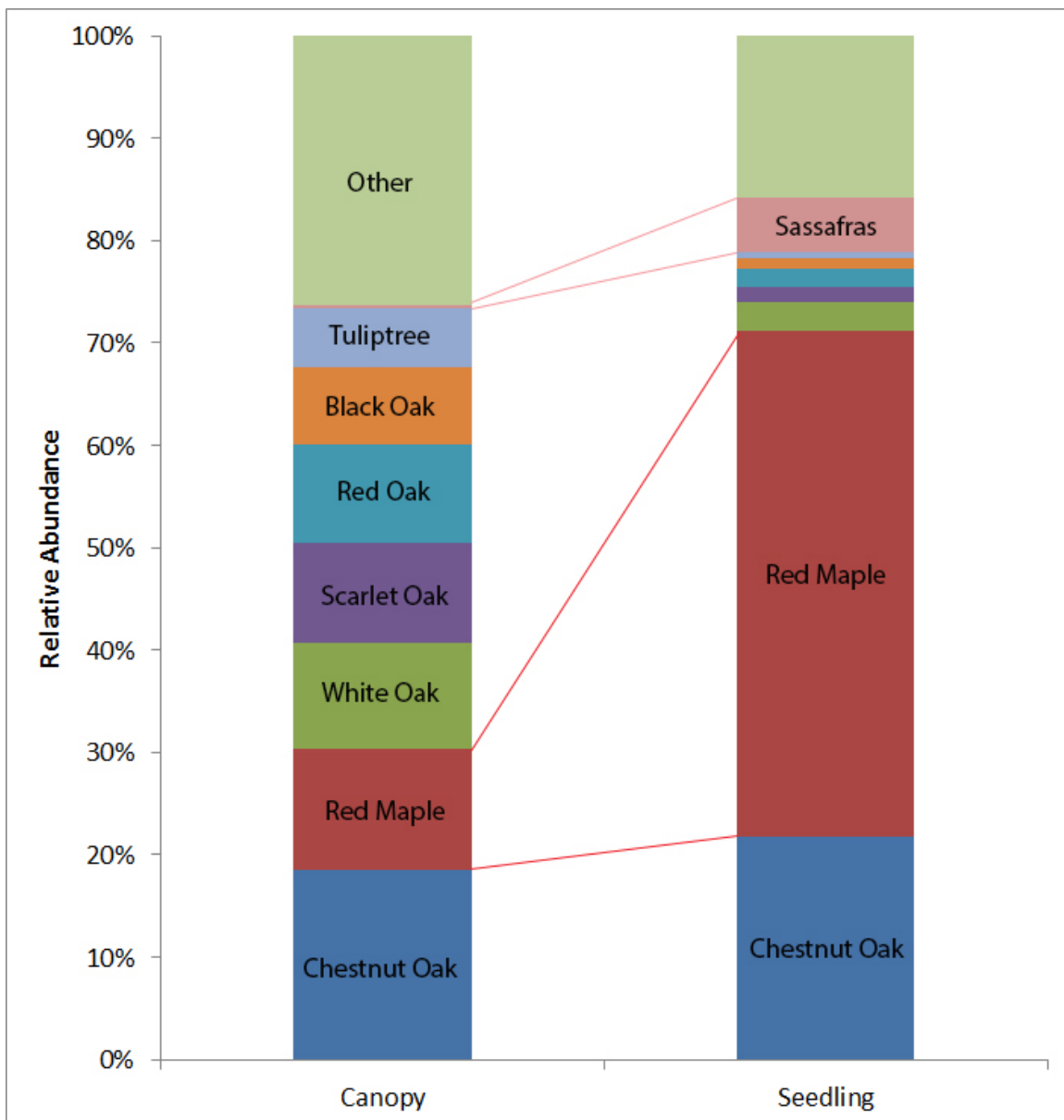


Figure 42. Relative basal area of tree species in the canopy and relative density of tree species in the seedling layer in xeric habitats in NERI. Colored lines highlight notable shifts in species composition.

Predictors of Tree Growth and Mortality

Using the stepwise model selection procedure, we identified the best models for predicting tree growth and mortality in ERMN parks. The strongest influence on tree growth was stand basal area, observed consistently across parks and soil horizons (Table 4). As stand basal area increases, tree growth decreases, which is driven by competition among trees for resources in increasingly crowded stands. No additional relationships were observed between tree growth and soil chemical variables in organic horizon soils. In A horizon soils, Ca:Al, BS, and C:N were significant predictors of tree growth, though the relationships were not consistent across all parks (Table 4). The ratio of calcium to aluminum was a significant predictor of tree growth in NERI, GARI, BLUE, and DEWA; however, the models indicated a positive relationship between Ca:Al and growth in the West Virginia parks and a negative relationship in DEWA.

Using A horizon soil variables in all ERMN parks or O horizon soil variables in DEWA, no significant models were generated to predict tree mortality (Table 5). Using the subset of plots from which organic horizon soils were collected (in all parks except DEWA), significant models were identified, although the predictors varied among the West Virginia parks and the western Pennsylvania parks. In NERI, GARI, and BLUE, mortality increased with decreasing organic horizon C:N, but also with increasing percentage of standing dead trees, and decreasing total basal area. Mortality rate might be lower in stands with more basal area because of the age of many forest stands in the parks. The younger stands with less basal area are closer to the stem exclusion phase of stand development where mortality is high as young trees compete for a position in the future canopy. In older forest stands with more basal area, competition is not as strong since the canopy is established and mortality rates are lower. In the western Pennsylvania parks, especially JOFL and FRHI, younger forests are closer to the stem exclusion phase, in which increasing basal area yields increased mortality as competition intensifies. Mortality also increased with decreasing pH and increased base saturation (Table 5).

Table 4. Coefficient values for regression models predicting tree growth (mean annual diameter increment) in groups of ERMN parks. Bold values are significant at the 0.05 level. Slashes indicate variable was not included in the model or no significant model was returned. See Table 3 for explanation of soil variable abbreviations.

	BA	%Dead	AI	Log Ca:AI	BS	Soil pH	C:N	ECEC	Adjusted R ²	p value
A Horizon										
West Virginia Parks	-0.0045	-	-	0.0208	-	-	-	-0.0022	0.2209	<0.01
DEWA	-0.0016	-	-	-0.0767	0.00355	-	-	-	0.2285	<0.01
Western PA Parks	-0.0035	0.183	-	-	-	-	0.015	-	0.3232	<0.01
O Horizon										
West Virginia Parks	-0.0034	-	-	-	-	-	-	-	0.1908	<0.01
DEWA	-0.0024	-	-	-	-	-	-	-	0.0928	0.01
Western PA Parks	-	-	-	-	-	-	-	-	-	-

Table 5. Coefficient values for regression models predicting tree mortality rate in groups of ERMN parks. Bold values are significant at the 0.05 level. Slashes indicate variable was not included in the model or no significant model was returned. See Table 3 for explanation of soil variable abbreviations.

	BA	%Dead	AI	Log Ca:AI	BS	Soil pH	C:N	ECEC	Adjusted R ²	p value
A Horizon										
West Virginia Parks	-	-	-	-	-	-	-	-	-	-
DEWA	-	-	-	-	-	-	-	-	-	-
Western PA Parks	-	-	-	-	-	-	-	-	-	-
O Horizon										
West Virginia Parks	-0.0248	6.3547	-	-	-	-	-0.0995	-	0.1705	<0.01
DEWA	-	-	-	-	-	-	-	-	-	-
Western PA Parks	0.0382	-	0.002	-	0.0503	-2.1008	0.1059	-	0.2861	0.03

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