Effects of Prescribed Burning on Forest Understories in Northern New Jersey

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Prescribed burning is well established as a tool for forest management, especially for fire hazard reduction and enhancing habitat quality for preferred game or timber species (Wood 1988). In the pine barrens of the southern New Jersey (NJ) coastal plain, prescribed burning has been used for wildfire fuel reduction for nearly 100 years, with thousands of hectares of public and private lands burned annually by the NJ Forest Fire Service today (Clark et al. 2014). The use of prescribed burning has been more limited in the hardwood forests of northern NJ, outside of isolated campgrounds and natural areas. However, fire has been increasingly used or promoted as a tool for habitat restoration or enhancement purposes in natural lands management in these areas interest of meeting a variety of objectives, including suppressing invasive plant species, enhancing the regeneration of oaks and other desirable species, increasing habitat or species diversity, or supporting the needs of various species of conservation concern. These goals represent some of the priorities for forest restoration in northern NJ [New Jersey Department of Environmental Protection (NJDEP) 2020], but little formal research has been conducted to date to determine the effectiveness of prescribed burning in this context, where the superabundant white-tailed deer and invasive plant species, forest fragmentation and soil disturbance regimes may lead to altered trajectories of forest response (Kelly 2019, Richburg et al. 2004, Nuttle et al. 2013).

Prescribed burning has traditionally been used to enhance tree regeneration and native plant diversity in pine-dominated forests (Clark et al. 2014), and has been hypothesized to benefit the regeneration of oaks by reducing competition, litter depth, or seed predation and increasing light and nutrient availability (Abrams 1992, Brose et al. 2011). However, the benefits of fire for oaks and other hardwoods are highly variable, with the outcomes depending upon initial site conditions, species, timing, size class structure, and other factors, and may also be detrimental or counterproductive to achieving these goals (Brose et al. 2011). The increased browse pressure from elevated populations of white-tailed deer (*Odocoileus virginianus*) and their preference for oak and other select hardwood species (Latham et al. 2005) raises important questions about its utility in this in this particular context.

Native forest vegetation throughout the Mid-Atlantic region has been severely degraded by overabundant white-tailed deer, which have increased due to the extermination of natural predators, warming winters, forest fragmentation, increased food resources from agriculture and suburban landscaping, and refugia from hunting (McWilliams et al. 2018). In northern NJ, deer have contributed to 70-80% declines in tree regeneration and native shrub and herb cover since the mid-twentieth century (Kelly 2019). These declines have resulted in indirect effects on shrub and ground nesting birds (Baiser et al. 2009), invertebrates (Chips et al. 2015), and amphibians (Bucciarelli et al. 2009). Several studies have reported the lack of regeneration in response to fire in the presence of sustained browse pressure except where herbivores were excluded or reduced, even when combined with increased light from canopy gaps (Nuttle et al. 2013, Andruk et al. 2014, Miller et al. 2017, Brose et al. 2011). Tree regeneration can be suppressed with deer densities as low as 6 deer/km^2 (Russell et al. 2017), and populations in northern New Jersey are regularly in excess of 30 deer/mi^2 (Kelly 2019), suggesting that prevailing levels of browse pressure are likely to constrain potential benefits of fire for tree regeneration in these areas.

Invasive plant species have also increased dramatically during roughly the same time period as deer in New Jersey (Kelly 2019) and may also profoundly alter ecosystem structure and function in ways that are detrimental to native plants, animals, fungi and ecosystem services (Bucciarelli et al. 2014, Burghardt et al. 2010, Ashton and Lerdau 2008, Ehrenfeld et al. 2001). Invasive woody plant species currently reach their highest concentrations in the fragmented forests of the northeast corridor, from Boston to Washington D.C. (Kartesz 2015). In northern New Jersey forests, invasive woody shrubs and vines increased 11-40 x since the mid-twentieth century and currently comprise the majority of vegetative cover in these forest layers on average (Kelly 2019). In younger forests that have grown on post-agricultural soils, moreover, invasive plant species are typically several times as abundant as native species (Kelly unpublished data), facilitated by the history of soil disturbance and degradation. These post-agricultural forests currently represent the majority of forestlands in much of northern New Jersey, owing to widespread declines in agricultural activity over the past century (Russell 1988).

Few options currently exist for suppressing invasive plant species at the large scales at which they occur, as biological controls are lacking for most invasive species, and synthetic herbicides and mechanical mowing present substantial economic costs and risks of collateral damage to native flora, fauna, and soils (Ward et al. 2018). Both grazing and fire are endemic disturbances to forest ecosystems and these prescriptions can be implemented at large spatial scales in a relatively cost-effective manner, but only the latter has been utilized with any frequency on public lands in NJ to date. Although prescribed burning has been extensively used for invasive plant management in grassland and rangeland contexts (Pendergrass et al. 1998), little research has been conducted on its effects in eastern forests (Richburg et al. 2004, Faulkner et al. 1989).

Our research was initiated to address these knowledge gaps and begin to study the effects of prescribed burning on a) reducing invasive plant species and b) enhancing native understory vegetation, including tree regeneration, native shrubs, herbs and woody vines, in the hardwood forests of northern NJ. Our hope is that the results will inform the use of prescribed burning as a tool for forest management in these and other similar contexts in the future.

Methods and Study Area

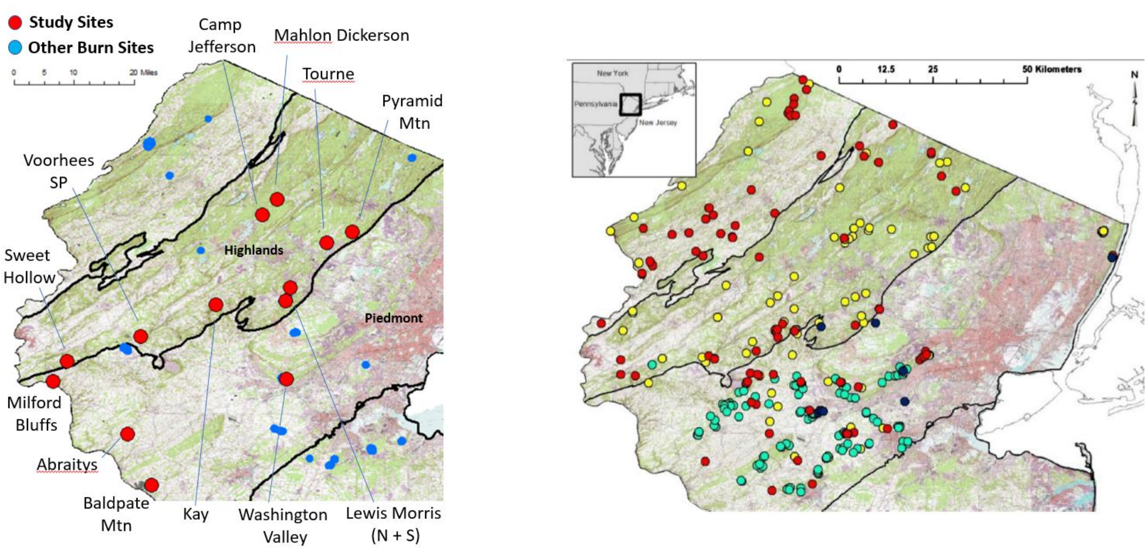
We studied vegetation conditions in a total of 60 plots in 14 locations where prescribed burning had occurred in northern NJ since 2012 (Figure 1), including representative areas scattered throughout the Piedmont and Highlands physiographic provinces. In each location, paired plots were established in adjacent a) burned and b) unburned areas with similar conditions (aspect, slope, canopy composition, etc.) for comparison. In four plots, the control data were collected 1-4 dormant seasons prior to burning. Plots were categorized according to the frequency of burns since 2012 (1 or 2-3), the number of growing seasons since the last burn (1-2 or 3-4), and the site history (post-agricultural "young" forests vs. "old" forests with no agricultural history) (Table 1).

We worked with local partners to identify suitable sites for study. GPS locations and dates of all prescribed burns conducted by NJDEP in the study region since 2012 were provided by the NJDEP Forest Fire Service (J. Webber, unpublished data), and maps of local burn sites were provided by local partners including Morris County Parks (K Kovacevic, M. Trump), Mercer County Parks (J. Stark, J. Rogers) and NJ Natural Lands Trust (R. Cartica, M. Rapp). Sites were selected for study when sufficient in size (>1 ha), with discernible land use history (based on 1930 aerial imagery and 1899 Vermeule forest map), and absence of active recreational use (playgrounds, camping, etc.). Sites on the coastal plain were excluded in order to maintain the relative consistency of geological characteristics, forest composition, and natural fire regimes between study sites. All sites were mixed oak-hardwood uplands and occupied a wide range of topographic positions, aspect, etc.

Vegetation data was collected by RVCC staff and interns with assistance from four technicians from NJDEP Office of Natural Lands Management. Parameters studied included density, diameter, height and % cover of trees (>4”dbh), saplings (1-4” dbh), and large seedlings (<1” dbh”, >1’ tall), and % cover and height of native and invasive shrubs, lianas, and herbs. Stem densities and dbh of trees were measured in 100 m^2 quadrats, with height of the tallest large seedling measured in each quadrant. Small seedlings density, cover and height, and herb cover were measured in two to four 1 m^2 plots nested within each larger quadrat. Percent cover of shrubs, lianas and large seedlings were measured using line intercept sampling along the center line of each 100 m^2 quadrat. The height of native and invasive shrubs and large seedlings were measured at the beginning, center and end of each quadrat, along the line. In most sites, five quadrats were positioned along each of three parallel 100-m transects spaced 20 m apart, except for Baldpate Mountain, where each plot included five quadrats (Figure 3). Transects were marked with metal or wooden stakes at the beginning and end in order to facilitate repeated data collection over time. Data were collected in June through early October in 2019-2021, except for control data for two sites (Washington Valley Park) which were collected in 2015.

Data were entered into MS Excel, classified according to the variables of interest, and summarized by quadrat for each year of study. Statistical analyses were conducted using generalized linear mixed effects models (GLMM) to compare the fixed effects of each factor (burn frequency, time since burn, forest age) on each response of interest (% cover and height of native vs. invasive shrub, liana and herb species, and density, cover and height of understory trees of different size classes), with random effects (quadrats, plots, stands and sites) to account for the dependency resulting from repeated measurements collected at each level. Appropriate distributions and error types were chosen for each type of data, including binomial distribution (logit error) for % cover, negative binomial (logit error) for count data, and Gamma distribution (log error) for continuous height data and cumulative % cover. All analyses were conducted using the glmmTMB package (Version 1.1.2.3) in R (Version 4.05) and R Studio (Version 1.4.1103). Data exploration and model validation followed Zuur and Ieno (2016) and Zuur et al. (2010, 2009).

Figure 1. Locations of forest study sites and other prescribed burn sites burned in northern NJ from 2010-2021 (left) and reference study sites (right, Kelly 2019). Study sites (left) and reference sites (right) are shown in red.



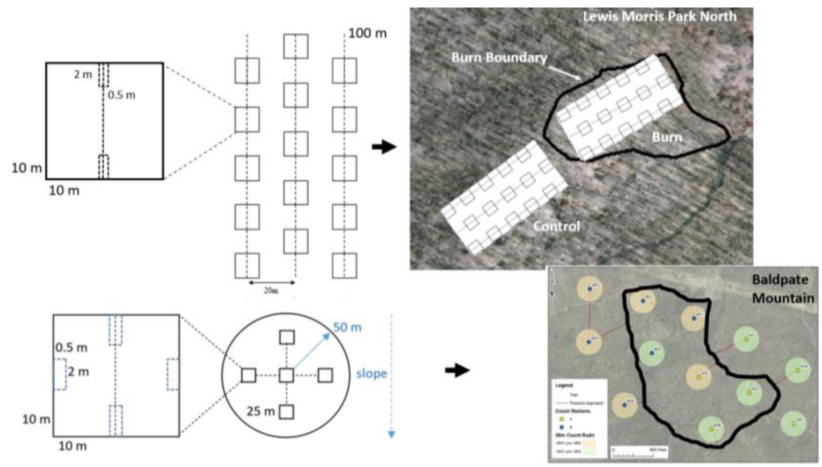
In order to confirm that the study sites were representative of northern NJ forest conditions and the associated priorities for forest restoration mentioned above, understory conditions in the control plots were first compared to reference data sets of regional and historic forest conditions in New Jersey (Kelly 2019). The reference data included 62 study sites scattered across northern NJ, all of which consisted of “old” forests with no agricultural history. The same sites were studied during two separate time periods, including a) 2014-2018, which provides reference data for regional understory conditions, and b) from 1948-1973, providing historical reference data for forest understory conditions when deer population were less than 10 deer/mi^2 (Kelly 2019). Statistical analyses were conducted in SAS-JMP 9.0 using one-sided single-sample means tests to determine whether the mean stem density of large seedlings or saplings, and cumulative % cover of shrubs, lianas and herbs differed from these benchmarks. Given the skewed distribution of the data, all analyses were conducted using non-parametric Wilcoxon ranks and Kruskal-Wallis tests at confidence intervals of 0.95, with signed-rank (Ro) test statistics and p-values to indicate significance. Variation in the data is provided in terms of the standard error of the mean.

Table 1. Characteristics of study plots including forest age, location, stand, burn frequency, time since burn, and sample size. * = control data collected in same plots prior to burn, ** = burns that took place during study period, requiring split classification of plot data characteristics by year. (WAVP3O burn plot was burned a second time, and ABRA02 control plot was burned, both in 2021).

Site	Age	Burn Frequency	Stand	Plot	Year(s) Burned	# Plots Burn	# Plots Control	Time Since Burn 1-2 yrs	Time Since Burn 3-4 yrs
Baldpate Mountain	Old	1	1	A02	2021	5	5	x	
Baldpate Mountain	Young	1	2	A03	2021	5	5	x	
Baldpate Mountain	Young	1	2	A06	2021	5	5	x	
Abraitys Pine Stand	Young	2	1	ABRA01	2018, 2019	15	12	x	x
Abraitys Pine Stand	Young	1	1	ABRA02	2021**	12	12*	x	
Baldpate Mountain	Old	1	1	B01	2021	5	5	x	
Baldpate Mountain	Old	1	1	B02	2021	5	5	x	
Baldpate Mountain	Young	1	2	B05	2021	5	5	x	
Camp Jefferson	Old	1	1	CAJEB	2019	10	10	x	x
Kay Environmental Center	Young	2	1	KAYE01B	2012, 2018	15	14	x	x
Lewis Morris Park Central	Old	1	1	LEMO0501B	2019	15	15	x	x
Lewis Morris Park North	Old	1	1	LEMON1B	2018	15	15	x	x
Lewis Morris Park South	Young	1	1	LEMOS20403B	2019	15	15	x	x
Lewis Morris Park South	Young	1	2	LEMOS20404B	2019	15	15	x	x
Lewis Morris Park South	Young	2	3	LEMOS21B06B	2012, 2018, 2019	15	15	x	x
Lewis Morris Park South	Young	2	1	LEMOS22A05B	2012, 2018, 2019	15	15	x	x
Lewis Morris Park South	Young	2	4	LEMOS22B07B	2012, 2018	15	15	x	x
Lewis Morris Park South	Old	2	5	LEMOS23B02B	2012, 2019	15	15	x	x
Mahlon Dickerson	Old	1	1	MADIB	2019	15	15	x	x
Milford Bluffs	Young	1	1	MIBL01	2019	10	10*	x	
Pyramid Mountain	Old	1	1	PYRA1B	2019	15	15	x	x
Pyramid Mountain	Old	1	2	PYRA3B	2019	15	15	x	x
Sweet Hollow Preserve	Young	1	1	SWHO02B	2019	14	14*	x	
Tourne County Park	Old	1	1	TOUR1B	2019	15	15	x	x
Tourne County Park	Young	1	2	TOUR3B	2019	15	15	x	x
Voorhees State Park	Old	1	1	VOOR03	2019	15	15	x	x
Voorhees State Park	Young	2	2	VOORNEB	2017, 2019	10	15	x	x
Washington Valley Park	Young	1	1	WAVP10N	2019	20	20*	x	x
Washington Valley Park	Old	2	2	WAVP3O	2019, 2021**	20	20*	x	
Washington Valley Park	Old	1	2	WAVP3O	2019	20	20*	x	

To assess the potential confounding effects of elevated deer browse on tree regeneration and understory responses, we conducted infrared deer surveys in the vicinity of each of the 14 study sites in 2019-2021. Surveys were conducted using a Zenmuse XT infrared sensor mounted on a DJI Inspire drone in 2019-2020, and Autel EVO II Dual drone with FLIR 640 Thermal Sensor in 2021. Surveys were conducted at night in order to improve contrasts between deer and ambient temperatures, and all surveys were conducted after the hunting season (mid-February through April) in order to provide the most conservative estimate of local deer populations; i.e., after winter mortality and prior to birthing (May). Surveys covered >3 km^2 surrounding each plot, effectively accounting for the typical range sizes and movement patterns of deer in these areas (Williams et al. 2008). All flights were conducted with an FAA-certified pilot aided by a visual observer trained and certified for night-time operations. All missions were either flown in public (Class G) airspace at ≤400 feet above ground level, or with proper authorization in Class D or other airspace where flights were restricted. All flights complied with federal regulations, and under an FAA waiver for night-time operations of small unmanned aircraft systems (sUAS). Deer density results were compared to average large seedling and sapling densities, as these exhibited the strongest relationships and responses to deer browse in previous research (Kelly 2019) and should serve as an indicator of potential browse effects on other understory vegetation. Statistical analyses were conducted using the glmmTMB package in R using generalized linear models (Gamma distribution with log error) for continuous data.

Figure 2. Sampling design for data collection. Study plots include three parallel 100-m transects spaced 20 m apart, with 100m^2 quadrats spaced 10 m apart. Line intercept was collected in the center line of each quadrat, and two to four nested 1 m^2 quadrats were nested in each. At Baldpate Mountain (only) quadrats were arranged in circular plots (50 m radius), which were evenly spaced throughout the study area. All plots were located a minimum of 20 m from the edge of the stand or treatment area when possible.



Results

Data were collected in a total of 30 burn plots in June through early October 2019-2021, including 16 in young (post-agricultural) forests, and 14 in old (intact) forests, and 26 control plots (13 young, 13 old). In each case, these data were collected in the first growing season following these dormant season burns, except for three sites (Kay Environmental Center, Lewis Morris North, Lewis Morris South – Stand 4), which were first measured in the second growing season following burning. Six young forest plots and two old forest plots were burned 2-3 times with the remainder burned once. Nineteen paired plots were measured in both 1-2 year and 3-4 year post-burn intervals in order to assess longer term responses in vegetation cover. Only one old forest plot with multiple burns was available for the 3-4 year data set.

The resulting data set included measurements from 1212 100 m^2 quadrats, including 678 collected 1-2 years after burning, and 534 from 3-4 years after the fire treatment. Measurements included counts of 10,1178 large seedlings and 2,297 saplings, and 557 to 840 averages of maximum height for native and invasive shrubs and large seedlings per quadrat (contingent upon their presence in each). A total of 24 woody and 16 non-indigenous species were recorded, and 57 woody and 140 herbaceous native species. Analyses of individual species responses were restricted to the most prevalent, occurring in at least 10% of sites at % cover of 1% or more. Several species were aggregated by genera during data collection, including native grapes (*Vitis* spp.), and others were lumped together later for analysis, including native *Rubus* spp. (*R. occidentalis, R. flagellaris, R. allegheniensis, R. argutus*) and *Vaccinium* spp. (*V. corymbosum, V. angustifolium, V. stramineum*, *V. pallidum*) and non-indigenous bush honeysuckles (*Lonicera morrowi, L. maackii, L. villosa*).

Baseline Comparisons

Analyses of baseline forest conditions in the control plots confirmed that the study sites exhibited similar deficits in tree regeneration as previous research (Kelly 2019)(Figure 3). Old forest controls exhibited 79% fewer large seedlings ($p < 0.0001$) and 62% less saplings ($p < 0.0001$) compared to historical conditions (Table 2). There were also major decreases in native shrub (75%) and herb cover (85%), and greater cover of invasive shrubs (74x), lianas (8x) and herbs (1.5x). Old forest controls were relatively consistent with other old forests in the region, although there were significantly less native herbs (39%), lianas (37%), and invasive herbs (54%). Densities of large seedlings and saplings in young forest controls were lower than both historic and regional benchmarks, and exhibited significantly more cover of exotic, invasive plant species. These included 328x more shrubs, 36x more lianas and 5x more herbs than historic conditions, and 7x shrubs, 3x lianas, and 2x herbs greater amounts (respectively) compared to present-day old forests in the region (Figure 2).

Figure 3. Characteristics of control plots (n = 26) relative to regional and historic data sets (Kelly 2019). These figures indicate the long-term deficits in tree regeneration, and native shrub, herb and liana cover relative to historic data sets, as well as a superabundance of invasive shrubs and vines.

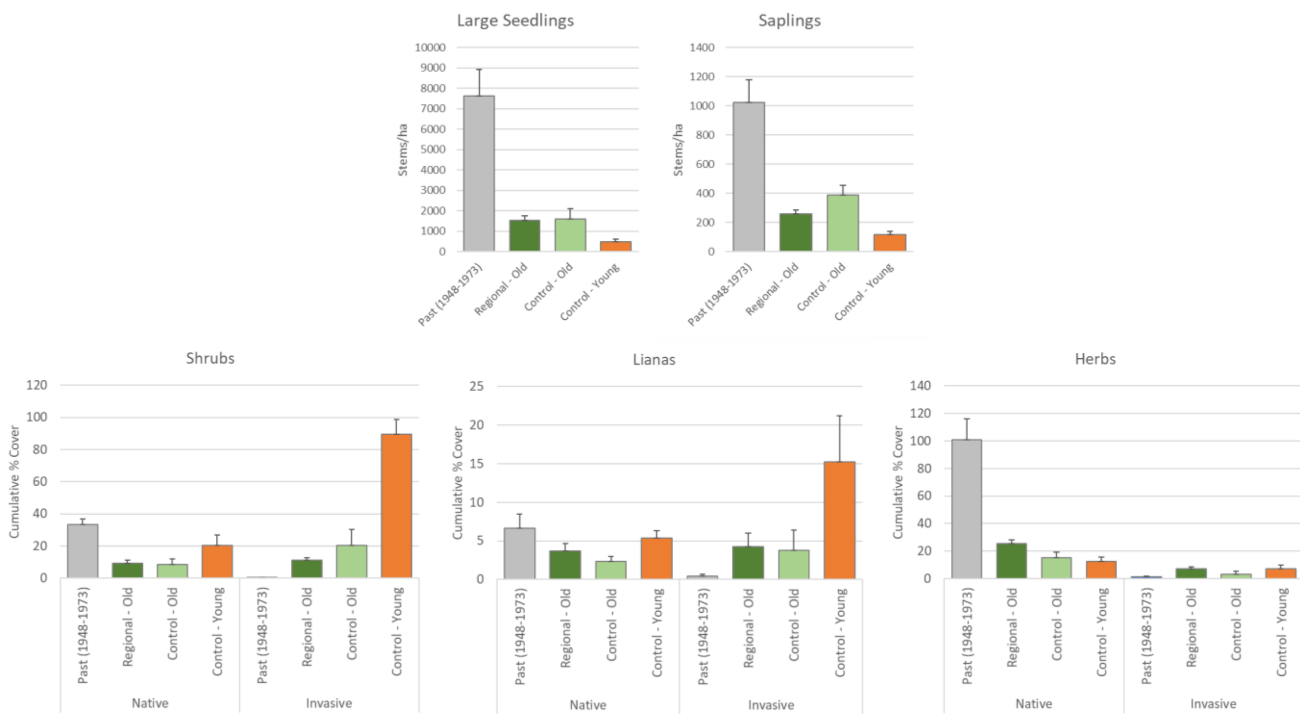
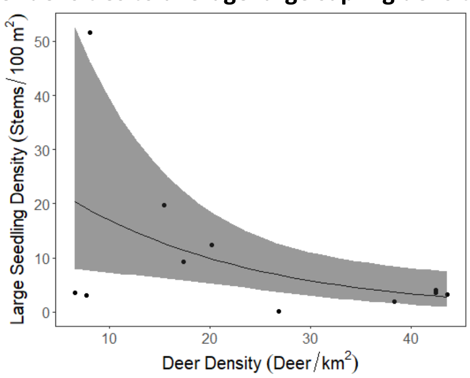


Table 2. Results of non-parametric means tests between control sites and regional and historic benchmarks (Kelly 2019)

					Benchmarks			
					Regional		Historic	
Age	Type	Mean	SE	df	Ro	P	Ro	P
Old	Large Seedlings	1597.1	508.3	12	-7.5	0.3116	-45.5	**0.0001**
	Saplings	389.9	65.1	12	26.5	0.0341	-45.5	**0.0001**
	Native Shrubs	8.5	3.3	12	-4.5	0.3934	-42.5	**0.0006**
	Native Lianas	2.3	0.7	12	-29	**0.0204**	-44.5	**0.0002**
	Native Herbs	15.4	3.7	12	-33.5	**0.0085**	-45.5	**0.0001**
	Invasive Shrubs	20.3	10.1	12	-9.5	0.2648	27.5	**0.0273**
	Invasive Lianas	3.7	2.6	12	-20.5	0.077	-20.5	0.077
	Invasive Herbs	3.3	2.0	12	-30.5	**0.0131**	-18.5	0.0985
Young	Large Seedlings	473.5	129.3	12	-45.5	**0.0001**	-45.5	**0.0001**
	Saplings	114.8	23.7	10	-21	**0.0337**	-22	**0.0269**
	Native Shrubs	20.4	6.6	12	21.5	0.0732	-32.5	**0.0107**
	Native Lianas	5.3	1.0	12	15.5	0.1527	-22.5	0.0636
	Native Herbs	12.5	3.1	12	-37.5	**0.0031**	-45.5	**0.0001**
	Invasive Shrubs	89.5	9.1	12	45.5	**0.0001**	45.5	**0.0001**
	Invasive Lianas	15.2	6.0	12	35.5	**0.0052**	44.5	**0.0002**
	Invasive Herbs	7.3	2.7	12	-7.5	0.3115	11.5	0.2224

Thermal imaging drone (sUAS) surveys of deer populations in the vicinity of the 14 study sites found an average of 22 deer/km^2 (range = 7 – 47 deer/km^2). Survey areas were an average of 10.6 km^2 in size around each study site except for Abraitys Pine Stand, Camp Jefferson, Washington Valley Park and Voorhees State Park, which occurred in adjacent areas. The results of the GLM (Gamma) analyses found strong relationships between local deer densities and average densities of large seedlings in each study site (Figure 4; intercept = 3.37, p = 0.0001; estimated slope -0.02, p = 0.01), but no relationship with sapling densities (intercept = 1.23, p = 0.008; estimated slope = 0.006, p = 0.356).

Figure 4. Relationship of local deer densities to average large sapling densities in control plots per study site



Invasive Shrub, Liana and Herb Cover

Prescribed burning yielded different responses depending upon the species type, species, frequency of burn, and time (Tables 3-5, Figures 5-7). Overall, prescribed burning significantly reduced the cumulative % cover of invasive shrubs (p = 0.0154) and lianas (p < 0.0001) after the first burn, and lianas after the 2-3 burns (p = 0.001), but led to increases in invasive herb cover, especially after multiple burns (p = 0.001) (Figure 5, Table 3). Although no effect was observed for time since burn in general, interaction effects were evident in all cases, with increased reductions of cover for invasive shrubs and lianas after multiple burns (p ≤ 0.0001). The opposite was the case for invasive herbs, which exhibited negative trends in cumulative % cover after 3-4 growing seasons following multiple prescribed burns (p = 0.001). Significant decreases in invasive shrub height were also observed following multiple burns (p < 0.0001), with positive interaction effects over time as well (Table 5).

Individual species responses were also highly variable, but most exhibited negative trends in % cover following both single or multiple burns (Figure 5, Table 3). Japanese barberry (*Berberis thunbergia*) and Japanese honeysuckle (*Lonicera japonica*) exhibited the most significant negative trends, with increases in negative responses in the 3-4 growing seasons following burning (p < 0.0001). These were followed by multiflora rose (*Rosa multiflora*), burning bush (*Euonymus alatus*), and oriental bittersweet (*Celastrus orbiculatus*), with *R. multiflora* exhibiting significant interaction effects in the 3-4 growing season as well (p < 0.0001). The declines exhibited by privet (*Ligustrum vulgare*) were not significant; however, interaction effects led to significant reductions after 3-4 growing seasons following burning. Positive effects were observed with Japanese Stiltgrass (*Microstegium vimineum*), but were not significant on average at 95% confidence (p = 0.0745), and exhibited negative interaction effects over time as with invasive herbs overall. Significant positive responses were observed with bush honeysuckle (Lonicera spp.) after one burn (p = 0.0001) and Japanese wineberry (*Rubus phoenicolasius*) after multiple burns (p < 0.0001).

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Figure 5. Effects of prescribed burning on % cover of the most common invasive shrub, liana and herb species. Gray bar indicates mean values, with outlier box plots indicating median (center line), 25% and 75% quartiles (white boxes), interquartile range (whiskers) and outliers. Graphs compare old, intact forests to young, post agricultural forests, and fire frequencies of 0 (controls), 1, and multiple (2) burn treatments with 2-3 burns.

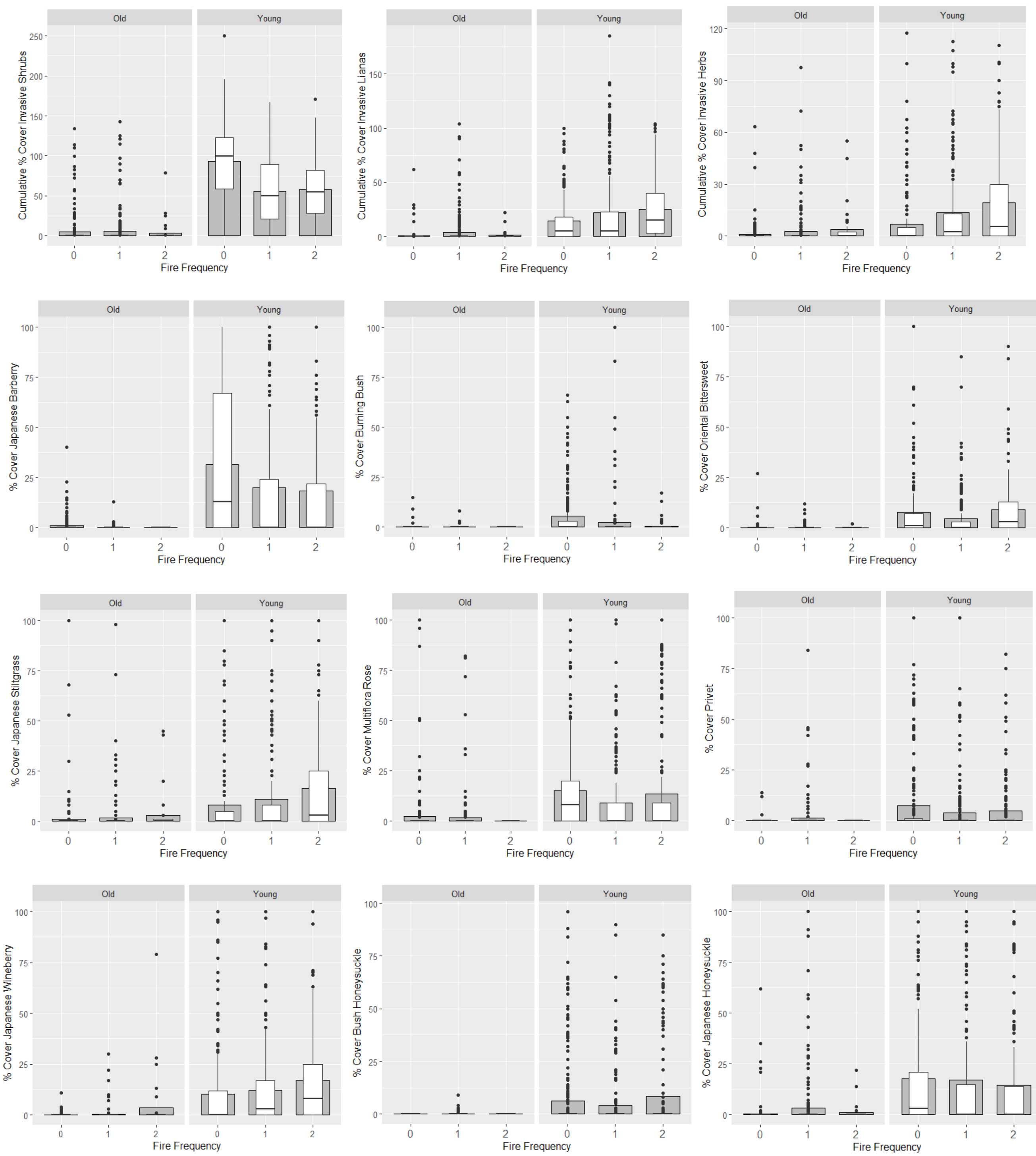


Table 3. Comparisons of changes in cumulative percent cover in older, intact forests compared to young, post-agricultural forests. Statistical significance is indicated according to: p < 0.10*, p < 0.05**, p < 0.01 ***.

			One Burn		2-3 Burns		Young		Time 2		OneBurnxTime2		2-3BurnsxTime2	
Response	**intercept**	**p-value**	**vs Control**	**p-value**	**vs Control**	**p-value**	**vs Old**	**p-value**	**vs T1**	**p-value**	**vs Control**	**p-value**	**vs Control**	**p-value**
Total Invasive Shrubs	-0.88	0.0296	**-0.33**	**0.0154**	-0.01	0.9775	**4.85**	**<0.0001**	-0.01	0.8682	0.08	0.4004	**0.42**	**0.0001**
Total Invasive Lianas	-0.68	0.1220	**-1.04**	**<0.0001**	**-2.60**	**<0.0001**	**3.27**	**<0.0001**	-0.06	0.5650	0.21	0.1700	**0.81**	**<0.0001**
Total Invasive Herbs	-1.56	0.0001	0.31	0.0822	**1.06**	**0.0010**	**2.01**	**0.0002**	0.10	0.2747	**-0.45**	**0.0010**	0.30	0.0828
Japanese Barberry	**-10.93**	**<0.0001**	**-0.34**	**<0.0001**	**-4.90**	**<0.0001**	**6.48**	**<0.0001**	-0.01	0.8470	**0.56**	**<0.0001**	**1.18**	**<0.0001**
Japanese Honeysuckle	**-11.68**	**<0.0001**	**-2.44**	**<0.0001**	**-4.56**	**<0.0001**	**8.38**	**<0.0001**	**0.23**	**0.0114**	0.06	0.6267	**2.04**	**<0.0001**
Multiflora Rose	**-10.16**	**<0.0001**	**-0.39**	**<0.0001**	**-1.84**	**0.0441**	**6.41**	**<0.0001**	**-0.19**	**0.0094**	0.04	0.7128	**0.47**	**<0.0001**
Burning Bush	**-13.62**	**<0.0001**	**-1.77**	**0.0113**	**-2.00**	**0.0391**	**2.27**	**0.0135**	0.19	0.0688	-0.03	0.8710	-0.26	0.3766
Oriental Bittersweet	**-9.86**	**<0.0001**	**-1.10**	**<0.0001**	**-1.83**	**0.0008**	**5.40**	**<0.0001**	**0.20**	**0.0489**	0.13	0.3557	0.06	0.6109
Privet	**-13.19**	**<0.0001**	-0.05	0.9617	-1.23	0.4853	**2.86**	**0.0273**	**-1.01**	**0.0069**	**1.35**	**0.0004**	**1.59**	**<0.0001**
Japanese Stiltgrass	**-11.81**	**<0.0001**	0.00	0.9573	0.30	0.0745	**5.57**	**<0.0001**	**1.42**	**<0.0001**	**-3.08**	**<0.0001**	**-1.34**	**<0.0001**
Bush Honeysuckle	**-16.70**	**<0.0001**	**1.68**	**0.0001**	-1.23	0.4137	**6.72**	**0.0028**	**2.03**	**<0.0001**	**-2.20**	**0.0001**	**-1.64**	**0.0001**
Japanese Wineberry	**-11.13**	**<0.0001**	**-0.56**	**<0.0001**	**1.02**	**<0.0001**	**5.30**	**<0.0001**	**-0.26**	**0.0019**	**0.21**	**0.0447**	**0.62**	**<0.0001**

Native Understory Vegetation and Tree Regeneration

Native shrub, liana and herb species exhibited similar trends and variation compared to invasive species, but with more muted responses overall (Tables 4-5, Figures 6-7). Cumulative % cover of native lianas declined (p = 0.0289) and herbs increased (p = 0.0039) after multiple burns, but significant trends (including interaction effects) were otherwise lacking overall. Significant declines occurred with native shrub height (p = 0.001) with significant interaction effects following both single and multiple burns (Table 4). In terms of individual species responses, significant declines occurred for spicebush (*Lindera benzoin*) after multiple burns (p = 0.0232), and grapes (*Vitis* spp.) following both single (p < 0.0001) and multiple burns (p = 0.048). The latter also exhibited significant but variable interaction effects with time following 1-2 or 3-4 growing seasons after burning. Maple leaf viburnum (*Viburnum acerifolium*) also exhibited declines after the first burn, and interaction effects resulting in declines in the 3-4 growing seasons after multiple burns.

Table 4. Comparisons of changes in cumulative percent cover in older, intact forests compared to young, post-agricultural forests. Statistical significance is indicated according to: p < 0.10*, p < 0.05**, p < 0.01 ***.

			One Burn		2-3 Burns		Age		Time		OneBurnxTime2		2-3BurnsxTime2	
Species	**intercept**	**p-value**	**vs Control**	**p-value**	**vs Control**	**p-value**	**vs Old**	**p-value**	**vs. T1**	**p-value**	**Freq1xTime**	**p-value**	**Freq2xTime**	**p-value**
Total Native Shrubs	0.78	0.1757	0.21	0.2381	0.48	0.1319	0.70	0.2058	**0.20**	**0.0402**	-0.20	0.1627	0.02	0.9191
Total Native Lianas	**-1.18**	**0.0001**	0.01	0.9600	**-0.72**	**0.0289**	**1.66**	**<0.0001**	-0.19	0.0745	0.06	0.6943	0.28	0.1282
Total Native Herbs	**1.86**	**<0.0001**	0.26	0.1446	**0.84**	**0.0039**	0.11	0.7701	0.18	0.1788	-0.17	0.3787	0.18	0.4567
Spicebush	**-12.13**	**<0.0001**	-0.10	0.2558	**-3.25**	**0.0232**	**5.67**	**0.0051**	0.15	0.2662	-0.13	0.5924	-0.08	0.6955
Grape	**-12.48**	**<0.0001**	**-0.62**	**<0.0001**	**-1.15**	**0.0480**	**1.24**	**0.0421**	**-0.79**	**<0.0001**	**0.37**	**0.0492**	**0.81**	**<0.0001**
Maple Leaf Viburnum	**-7.75**	**<0.0001**	**-1.37**	**0.0433**	1.19	0.2711	-2.41	0.0857	**-0.15**	**0.0166**	0.07	0.5192	**-0.95**	**0.0002**
Black Haw Viburnum	**-15.42**	**<0.0001**	0.20	0.2560	-0.24	0.8001	**2.83**	**0.0034**	**0.97**	**<0.0001**	**0.98**	**0.0000**	**-0.94**	**0.0005**
Blueberry	**-8.42**	**<0.0001**	0.55	0.4660	1.68	0.1528	**-4.01**	**0.0108**	**-6.18**	**<0.0001**	**3.61**	**0.0004**	-15.19	0.9966
Blackberry	**-12.36**	**<0.0001**	**0.47**	**0.0004**	1.34	0.0935	0.61	0.3072	-0.32	0.1599	**1.30**	**0.0000**	**0.81**	**0.0010**

Table 5. Comparisons of changes in cumulative percent cover in older, intact forests compared to young, post-agricultural forests. Statistical significance is indicated according to: p < 0.10*, p < 0.05**, p < 0.01 ***.

			One Burn		2-3 Burns		Age		Time		OneBurnxTime2		2-3BurnsxTime2	
Species	**intercept**	**p-value**	**vs Control**	**p-value**	**vs Control**	**p-value**	**vs Old**	**p-value**	**vs. T1**	**p-value**	**Freq1xTime**	**p-value**	**Freq2xTime**	**p-value**
Sapling Density	**0.73**	**0.0070**	-0.30	0.0628	**-0.62**	**0.0045**	**-1.21**	**0.0007**	0.09	0.2518	**-0.39**	**0.0020**	0.11	0.6085
Lg. Seedling Density	**1.63**	**<0.0001**	-0.16	0.5622	0.03	0.9480	**-1.04**	**0.0293**	**0.32**	**0.0446**	-0.38	0.1025	0.50	0.0645
Lg. Seedling Cover	**-4.71**	**<0.0001**	-0.15	0.6190	**-2.90**	**<0.0001**	**-1.65**	**0.0073**	**-0.13**	**0.0027**	**-0.76**	**<0.0001**	**1.27**	**<0.0001**
Lg. Seedling Height	**4.79**	**<0.0001**	**-0.28**	**0.0222**	**-0.55**	**0.0003**	-0.20	0.0884	**0.12**	**0.0122**	-0.07	0.3123	**0.22**	**0.0106**
Native Shrub Height	**3.70**	**<0.0001**	-0.29	0.1574	**-0.80**	**0.0019**	**0.77**	**0.0010**	0.06	0.4202	**0.26**	**0.0152**	**0.32**	**0.0133**
Invasive Shrub Height	**4.05**	**<0.0001**	-0.24	0.0994	**-0.72**	**<0.0001**	**0.98**	**<0.0001**	0.02	0.7879	0.00	0.9813	**0.22**	**0.0140**

Increases were observed in blueberry species (*Vaccinium* spp.) following burns, which were not significant except for the interaction effects that increased these responses in the 3-4 growing seasons following the first burn. Blackberries (*Rubus* spp.) also increased after one burn ($p = 0.0004$), with positive interaction effects in the 3-4 growing seasons following multiple burns. Both *Vaccinium* spp. and *Rubus* spp., were the least abundant of any individual species analyzed, with an average cover of 0.5-1% per site, including sites with zero % cover.

All measures of tree regeneration (% cover, density, height) exhibited negative responses to prescribed burning (Figure 7, Table 5), especially after multiple burns ($p < 0.0001$ - 0.0045). Many of these trends also exhibited enhanced responses via interaction effects over time. The only exception was large seedling densities, which exhibited no significant response. In terms of deer herbivory, identifiable browse damage was recorded in 50-87% of quadrats and 26-52% of large seedling stems per year.

Figure 6. Effects of prescribed burning on % cover of the most common native shrub, liana and herb species.

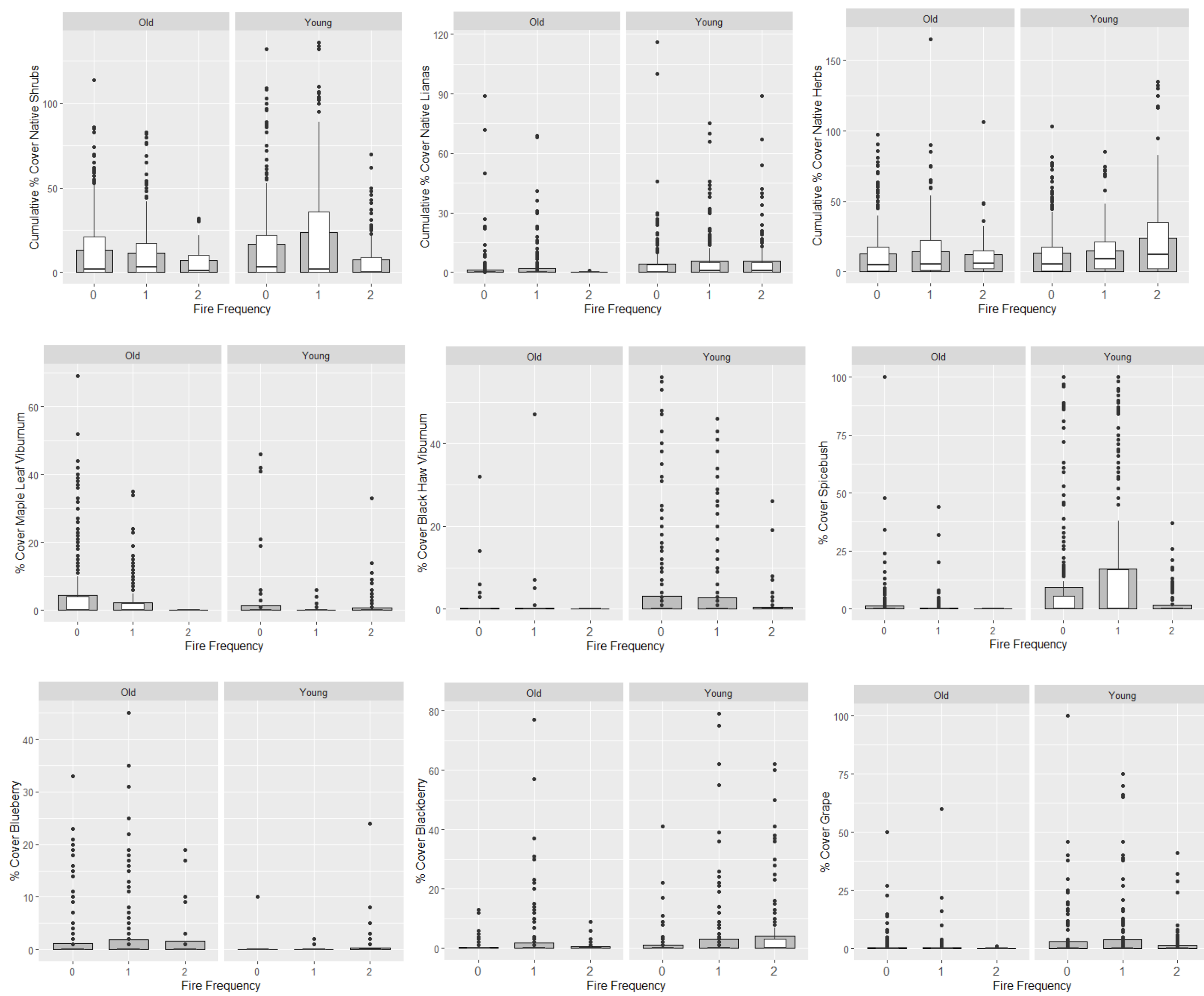
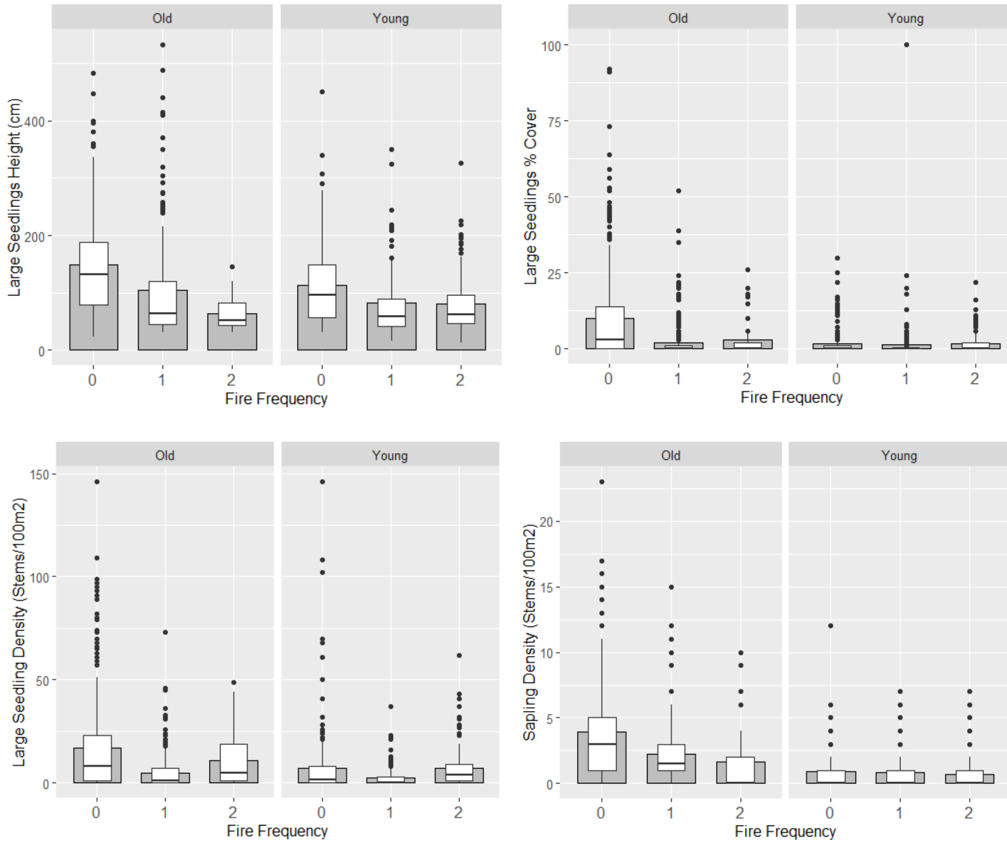


Figure 7. Effects of prescribed burning on large tree seedling density, cover and height, and sapling density.



Discussion

The results of this study demonstrate the potential utility of prescribed burning for suppressing non-indigenous invasive shrubs and lianas, especially in the younger, post-agricultural forests where these species are most abundant. The majority of prevalent species declined as a result of prescribed burning, especially after multiple burns, and the positive interaction effects with time suggests that these reductions are compounded by delayed responses in the 3-4 growing seasons following a burn treatment. Those species most affected appeared to be those with the thinnest bark, including Japanese barberry (*Berberis thunbergii*), multiflora rose (*Rosa multiflora*), and Japanese honeysuckle (*Lonicera* japonica). The impacts on barberry in particular were immediate and striking, with the bright yellow sap boiling out of the cambium after fire moved through (Figure 8). Although few formal studies are available for comparison, similar declines in biomass and fuel loads in response to fire occurred with Japanese honeysuckle (*Lonicera japonica*), and no response with privet (*Ligustrum vulgare*), in Tennessee and Georgia, by Faulkner et al. (1989).

As significant as these declines were in some cases, greater reductions may be possible through the use of growing season burns. Richburg et al. (2004) found that dormant season burns did not deplete carbohydrate reserves of invasive species such as multiflora rose (*Rosa multiflora*), Japanese barberry (*Berberis thunbergii*) and bush honeysuckle (*Lonicera morrowi*), which exhibited rapid replenishment and growth compared to growing season treatments. It is also worth noting that in many cases the primary goals of the prescribed burns we studied were fuel and brush reduction in the vicinity of campgrounds. Further reduction of invasive species may be possible if practitioners work with this particular goal in mind; i.e., applying their working knowledge of fire behavior, site and weather conditions, to focus on woody or other species reduction rather than leaf litter and fuel per se.

Figure 8. Photos of initial responses to burning. Top row = Lewis Morris Park before (left) and after (right) prescribed burning in April 2019, Second row = prescribed burning effects on Japanese barberry (*B. thunbergii*), showing sap boiling out of cambium at Milford Bluffs in winter 2020;

While these results are promising, significant variation in individual species trends was observed, including positive responses of several invasive shrubs such as Japanese wineberry (*Rubus phoenicolasius*) and herbs such as Japanese stiltgrass (*Microstegium vimineum*). This suggests that attention must first be paid to local conditions in order to determine whether prescribed burning is an appropriate tool for management. If burning is conducted on sites where these species are already prevalent, it may be counterproductive, leading instead to increased invasive cover. This may also be the case in older forests with heavy loads of latent invasive propagules existing in the seed banks. We observed several examples where increased light and disturbance on ridges and south-facing slopes resulted in increased prevalence of invasive species such as *R. phoenicolasius*, Japanese angelica tree (*Aralia elata*), and garlic mustard (*Alliaria petiolata*), which was facilitated by the dieback of trees and understory shrubs such as witch hazel (*Hamamelis virginiana*) following fire.

Sensitivity to local contexts and initial conditions is all the more important given the negative effects of prescribed burning observed for various native components of the understory flora, including individual shrub and liana species cover, overall native shrub height and cover, and tree regeneration. Nearly all measures of tree regeneration (large seedling % cover & height, sapling densities) as well as shrub cover and height showed negative responses to burning (Figure 9). The recovery of these understory elements represents a significant priority for forest restoration in many eastern forests, where overabundant deer, competition from invasives, and isolation and reduction of local seed sources from forest fragmentation have led to drastic reductions in the landscape (Kelly 2019, Almendinger et al. 2020, McWilliams et al. 2018). Although blackberry (*Rubus* spp.) and blueberry (*Vaccinium* spp.) species increased in response to fire, many important understory species such as spicebush (*Lindera benzoin*) and maple leaf viburnum (*Viburnum acerifolium*) decreased, often drastically. Other less abundant species appeared to follow similar trends including increases by huckleberry (*Gaylussacia* spp.) and decreases by shadbush (*Amelanchier arborea*), witch hazel (*Hamamelis virginiana*), and ironwood (*Carpinus caroliniana*), among others.

Figure 9. Photos of responses to prescribed burning after 3-4 growing seasons. Left– impacts of prescribed burning on native understory growth, three growing season after single burn at Voorhees State Park; Right = Reductions of invasive cover in young, post-agricultural forest at Lewis Morris Park, three growing seasons after burn. Left of dashed line= control, right = burn)

Of particular importance are the negative effects on sapling (1-4” dbh) densities, given the significant length of time required for this component of forest understories to develop or recover. In one local study, for example, sapling densities did not reach levels of historic benchmarks until > 15 years, and eventually did so only with intensive long-term deer management (Almendinger et al. 2020, Kelly and Ray 2020). Decreases in sapling densities were not necessarily indicative of mortality in this study, as ample resprouting was observed in many cases; however, it may take decades for these individuals to return to equivalent pre-burn heights, whether from stump sprouts or seeds. None of the sites studied exhibited significant tree mortality as a result of dormant season burning; however, one such case was observed at Lewis Morris Park, with a prescribed burn in 2012 leading to large-scale mortality of beech (*Fagus grandifolia*). Because this was the only site available for study at 10-12 years after burning, we excluded it from our analyses due to insufficient sample sizes. Future research of these and other sites in northern New Jersey should help address the long-term impacts of fire on both understory vegetation and canopy trees.

The lack of positive tree regeneration and native understory response in general appears to be the result of excessive deer browse in our study, as has been reported elsewhere (Nuttle et al. 2013). Observable browse damage was recorded in 50-87% of quadrats and 26-52% of large seedling stems per year. Although initial increases were observed in small seedling densities (<1’ tall), few of these grew to larger size classes as a result of browse. Suppression of native regeneration would be expected given the average deer densities we observed were 22 deer/km^2 (range = 7-47 deer/km^2) in the vicinity of our study sites. Nuttle et al. (2013), for example, found fire reduced understory diversity and density when exposed to deer browse at densities of 12-18 deer/km^2, but increased when browse pressure was eliminated. Andruk et al. (2014) found no effect of fire in stimulating oak regeneration after three years, but found increases in response to deer management and fire combined, and increased recruitment of saplings from seedlings with deer management alone. Russell et al. (2017) found significant regional declines in large seedling size classes when deer populations exceeded 6 deer/km^2. Similarly, both Kelly (2019) and Bradshaw and Waller (2016) found sustained high deer densities to result in long-term sapling declines as well. Prescribed burning is a traditional tool for deer management (Lashley et al. 2011, Turner et al. 2020, Wood 1988), serving to order attract, sustain or increase deer populations by enhancing forage quantity or quality. It appears that its use in contexts where deer populations are already elevated, however, results in the absence of any positive understory responses in native vegetation for most species or overall.

Given the high numbers of deer and absence of healthy native understories throughout much of northern New Jersey, it seems advisable to avoid prescribed burning in areas where regeneration or shrub layers are in good condition, at least without effective, long-term deer management. Conversely, in areas where invasive plant species that are susceptible to fire suppression are superabundant, fire appears to offer a valuable tool for enhancing forest understory conditions. This is especially the case in many post-agricultural forests, where tree regeneration is significantly lower on average, except for those that have increased native shrub layers comprised of fire prone species.

Future Research

This study was limited by the lack of available study sites where prescribed burning had occurred more than four years prior or where multiple burns had taken place, especially in old forests. The results were also restricted to dormant season burns. Additional data is needed on the effects of prescribed burns (Richburg et al. 2004) in combination with mechanical or other treatments in the growing season (Miller et al. 2017), as well as deer management for improving understory responses to fire (Nuttle et al. 2013, Ward et al. 2018). This study provides a preliminary coarse-grained analyses of the data, focusing on dominant species, cumulative understory structure and compositional responses to fire. Multivariate approaches investigating shifts in overall species composition and diversity are needed (Eales et al. 2018). Lastly, research is needed to assess how variables affecting fire behavior such as stand and meteorological conditions, may be utilized to enhance results for suppressing invasive species and improving native species responses.

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Appendix I. Site summaries of % cover data for most prevalent species. Species names are abbreviated with four letter codes (e.g., BETH = Berberis thunbergia, LOSP = Lonicera spp.)

Site	Plot	# Burns	Age	Last Burn Year	BETH		CEOR		EUAL		LIVU		LOJA		LOSP		ROMU		RUPH		MIVI		LIBE		RUSP		VACC		VIAC		VIPR		VITI	
					1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
Abraitys Pine Stand	ABRA01	2	Young	2019	0	0	0	0	0	0	3	2	34	60	0	0	64	64	2	3	11	16	5	3	18	22	0	0	0	0	0	0	2	0
	ABRA02	0	Young		1		0		0		0		80		0		54		5		18		15		11		0		0		0		2	
	ABRA02	1	Young	2021	1		0		0		0		51		2		52		3		19		12		15		0		0		0		0	
Baldpate Mountain	A01	0	Young	2021	14		1		0		0		7		0		47		0		0		3		0		0		0		12		5	
	A02	1	Old	2021	0		0		0		2		5		0		6		0		15		4		1		0		0		0		8	
	A03	1	Young	2021	7		0		0		0		4		3		22		0		8		19		0		0		0		14		1	
	A04	0	Young	2021	31		0		0		3		2		0		29		0		0		12		0		0		0		0		6	
	A05	0	Young	2021	17		2		3		1		6		0		35		0		0		14		0		0		0		18		0	
	A06	1	Young	2021	0		0		0		0		2		0		10		0		1		15		0		0		0		0		0	
	B01	1	Old	2021	1		0		2		2		8		0		65		0		0		3		7		0		0		0		0	
	B02	1	Old	2021	0		0		0		0		0		0		2		0		0		0		0		0		0		0		0	
	B03	0	Old		3		0		0		0		13		0		18		0		0		8		0		0		0		0		1	
	B04	0	Old		1		0		0		2		0		0		13		0		23		1		0		0		0		0		5	
	B05	1	Young	2021	4		6		0		0		7		0		19		0		0		32		0		0		0		0		8	
	B06	0	Old		0		9		0		3		22		0		87		0		28		9		3		0		0		0		2	
Camp Jefferson	CAJEB	1	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	16	0	9	4	0	0	0	0
	CAJEC	0	Old		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	3	0	10	13	0	0	0	0
Kay Enviro. Center	KAYE01B	2	Young	2018	0	0	27	23	1	2	20	30	0	0	41	50	0	1	5	17	0	3	0	0	1	4	0	0	0	1	0	0	5	5
	KAYE02C	0	Young		4	5	3	12	6	7	43	47	7	4	40	42	12	13	1	1	0	0	0	4	0	1	0	0	0	0	0	2	1	0
Lewis Morris S1	LEMO0501B	1	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
	LEMO0502C	0	Old		1	2	0	0	1	2	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Lewis Morris North	LEMON1B	1	Old	2018	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
	LEMON1C	0	Old		1	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2
Lewis Morris South 2	LEMOS20403B	1	Young	2019	78	88	1	0	0	0	0	1	10	12	0	0	5	2	17	10	7	5	0	0	0	0	0	0	0	0	0	0	1	0
	LEMOS20404B	1	Young	2019	24	28	1	1	1	0	0	0	0	0	0	0	1	2	2	5	3	0	0	0	0	0	0	0	0	0	0	1	0	1
	LEMOS20404C	0	Young		74	66	5	3	7	7	0	0	2	0	0	0	10	7	3	3	1	0	2	2	0	0	0	0	0	0	0	0	3	0
	LEMOS21B06B	2	Young	2018	16	37	5	12	0	0	0	0	0	9	0	0	1	2	15	18	34	16	0	0	0	0	0	0	0	0	2	2	0	2
	LEMOS22A05B	2	Young	2019	7	34	2	6	0	0	0	0	0	38	0	0	3	8	48	53	19	35	0	0	1	1	0	0	0	0	0	0	0	0
	LEMOS22A05C	0	Young		91	92	3	7	2	2	0	0	7	8	0	0	13	12	2	1	2	8	1	0	0	0	0	0	0	0	6	17	0	2
	LEMOS22B07B	2	Young	2018	27	32	13	15	0	0	0	0	6	13	0	0	1	1	17	12	26	24	0	0	0	0	0	0	0	0	0	0	1	0
	LEMOS22B07C	0	Young		79	83	6	8	5	7	1	1	10	14	0	2	11	11	32	29	0	1	0	0	0	0	0	0	0	0	0	1	7	1
	LEMOS23B02B	2	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	5	0	0	0	0	0	0	0
	LEMOS23B02C	0	Old		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0	0
Mahlon Dickerson	MADIB	1	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9	0	4	8	0	0	0	0
	MADIC	0	Old		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	11	0	5	2	0	0	0	0
Milford Bluffs	MIBL01	1	Young	2020	29		1		0		0		4		0		8		29		34		4		1		0		0		3		7	
	MIBL01	0	Young		34		2		0		0		32		0		15		27		29		1		0		0		0		5		7	
Pyramid Mountain	PYRA1B	1	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0
	PYRA2C	0	Old		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	0
	PYRA3B	1	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	PYRA4C	0	Old		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0
Sweet Hollow	SWHO02B	0	Young		22		4		0		0		3		0		7		5		4		91		0		0		0		5		0	
	SWHO02B	1	Young	2020	17		0		0		0		0		0		2		1		6		89		0		0		0		4		0	
Tourne County Park	TOUR1B	1	Old	2019	0	0	0	1	1	0	10	13	8	15	1	0	4	2	1	2	2	2	3	3	9	19	1	0	0	0	0	4	5	0
	TOUR2C	0	Old		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	5	8	2	1	1	1
	TOUR3B	1	Young	2019	0		11		17		11		8		0		2		3		1		1		18		0		1		3		3	
	TOUR4C	0	Young		0		4		32		14		3		3		1		0		2		1		0		0		12		3		7	
Voorhees State Park	VOOR03	1	Old	2019	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1	0	5	3	9	7	0	0	0	0
	VOORNEB	2	Young	2019	1		1		0		0		1		3		5		4		0		12		1		0		1		0		0	
	VOORNEC	0	Young		18		2		2		0		0		11		17		0		1		13		1		1		0		0		0	
	VOORNWC	0	Old		4	5	0	0	0	0	0	0	0	0	0	0	2	2	1	0	0	0	8	10	0	0	2	0	28	19	0	0	0	0
Washington Valley Park	WAVP10N	1	Young	2019	1		16		1		12		58		0		17		28		38		0		0		0		0		1		7	
	WAVP10N	0	Young		3		34		0		9		47		0		11		33		29		0		0		0		0		3		5	
	WAVP3O	1	Old	2019	0		1		0		0		24		0		0		4		13		0		0		0		0		0		1	
	WAVP3O	2	Old	2021	0		0		0		0		3		0		0		10		8		0		0		0		0		0		0	
	WAVP3O	0	Old		1		0		0		0		0		0		1		0		3		1		0		0		0		0		0	
			Control (AVG)		**15**	**20**	**3**	**2**	**2**	**2**	**3**	**4**	**9**	**2**	**2**	**3**	**15**	**3**	**4**	**3**	**5**	**1**	**7**	**1**	**1**	**0**	**1**	**0**	**2**	**3**	**2**	**2**	**2**	**0**
			Burn (AVG)		**7**	**14**	**3**	**4**	**1**	**0**	**2**	**3**	**8**	**9**	**2**	**3**	**10**	**5**	**6**	**7**	**8**	**6**	**7**	**0**	**3**	**3**	**1**	**0**	**1**	**1**	**1**	**1**	**2**	**1**

Appendix II. Site Summary Data for Cumulative Cover, Height and Density of Understory Vegetation

Site	Plot	# Burns	Age	Last Burn Year	Shrubs (% Cover) Native 1	Shrubs Native 2	Shrubs Invasive 1	Shrubs Invasive 2	Lianas (% Cover) Native 1	Lianas Native 2	Lianas Invasive 1	Lianas Invasive 2	Herbs (% Cover) Native 1	Herbs Native 2	Herbs Invasive 1	Herbs Invasive 2	Shrub Height (cm) Native 1	Shrub Height Native 2	Shrub Height Invasive 1	Shrub Height Invasive 2	Lg. Seedling (% Cov) 1	Lg. Seedling 2	Lg. Seed. Height (cm) 1	Lg. Seed. Height 2	Lg. Seed. (#/100 m²) 1	Lg. Seed. 2	Sapling (#/100 m²) 1	Sapling 2
Abraitys Pine Stand	ABRA01	2	Young	2019	23	25	71	70	2	1	33	60	8	10	12	16	80	106	119	116	0	0	71	83	3	6	2	1
	ABRA02	0	Young		26		61		5		80		3		18		71		135		1		81		4		1	
	ABRA02	1	Young	2021	27		59		3		51		7		19		87		126		1		62		5		1	
Baldpate Mountain	A01	0	Young	2021	14		92		6		8		10		1		413		175		4		127		10		2	
	A02	1	Old	2021	5		13		8		5		50		8		280		119		2		69		1		3	
	A03	1	Young	2021	33		43		1		4		18		4		201		119		0		56		2		4	
	A04	0	Young	2021	24		128		6		2		0		0		146		273		0		124		0		3	
	A05	0	Young	2021	32		88		4		8		3		1		174		209		0		125		8		1	
	A06	1	Young	2021	15		24		0		2		14		1		53		56		0		174		0		0	
	B01	1	Old	2021	10		111		4		9		23		3		197		201		4		115		2		1	
	B02	1	Old	2021	0		2		1		0		7		1				30		0		60		2		1	
	B03	0	Old		8		62		2		13		15		1		98		148		3		197		4		5	
	B04	0	Old		1		73		5		0		44		23		101		210		0		72		2		1	
	B05	1	Young	2021	32		33		8		13		22		3		139		132		0		46		2		0	
	B06	0	Old		12		96		7		30		29		13		168		199		0		66		0		1	
Camp Jefferson	CAJEB	1	Old	2019	29	15	0	0	0	0	0	0	0	16	0	0	39	26			11	4	54	208	36	11	2	1
	CAJEC	0	Old		20	32	0	0	0	0	0	0	0	11	0	0	80	79			4	8	134	180	15	19	5	4
Kay Enviro. Center	KAYE01B	2	Young	2018	2	4	68	100	23	21	27	25	0	21	0	16		37		173	3	3		93		10		1
	KAYE02C	0	Young		1	8	109	118	1	7	10	20	0	5	0	1		62		232	0	7		158		9		2
Lewis Morris S1	LEMO0501B	1	Old	2019	4	9	0	0	0	0	0	0	3	3	0	0	20	69	13		0	2	215	106	6	1	1	1
	LEMO0502C	0	Old		2	2	3	4	0	0	0	0	7	4	0	0	17	14	68	21	36	26	221	219	37	10	3	3
Lewis Morris North	LEMON1B	1	Old	2018	13	0	1	1	0	0	0	1	2	0	0	0	49				13	1	58	72	5	3	1	1
	LEMON1C	0	Old		0	0	1	2	2	2	0	0	2	1	0	0			75	70	2	1	93	83	11	2	3	4
Lewis Morris South 2	LEMOS20403B	1	Young	2019	0	0	92	103	3	1	19	13	6	8	10	5	58		143	126	8	2	90	188	1	1	1	1
	LEMOS20404B	1	Young	2019	0	4	29	37	2	2	1	1	42	30	3	1	25	500	67	67	3	2	128	78	3	5	1	1
	LEMOS20404C	0	Young		2	2	88	86	7	1	10	3	41	33	1	0	95	173	107	108	6	5	99	139	15	46	0	1
	LEMOS21B06B	2	Young	2018	2	3	32	59	2	6	6	21	49	31	34	19	26	51	56	76	1	3	47	62	2	10	0	0
	LEMOS22A05B	2	Young	2019	1	1	58	95	0	1	2	44	49	30	28	42	63	110	75	113	0	2	115	150	8	6	0	0
	LEMOS22A05C	0	Young		8	17	111	107	0	2	10	14	6	11	2	8	399	377	148	144	0	0	38	86	0	0	1	0
	LEMOS22B07B	2	Young	2018	0	0	30	47	6	2	36	28	29	35	26	27		87	63	71	1	2	45	60	3	5	0	0
	LEMOS22B07C	0	Young		0	2	136	146	10	5	15	21	15	8	1	1		157	139	177	1	1	90	117	6	11	0	0
	LEMOS23B02B	2	Old	2019	9	11	0	0	0	0	0	0	8	8	0	0	28	44			0	8	41	74	14	16	0	0
	LEMOS23B02C	0	Old		2	2	1	1	0	0	0	0	4	9	0	0	31	53	50	76	24	15	154	147	50	24	2	2
Mahlon Dickerson	MADIB	1	Old	2019	18	45	0	0	0	0	0	0	21	31	0	0	23	49			0	1	53	86	7	3	3	2
	MADIC	0	Old		41	46	0	0	1	0	0	0	20	28	0	0	66	69			19	23	133	148	52	56	5	4
Milford Bluffs	MIBL01	1	Young	2020	6		114		7		34		17		30		232		231		1		42		2		0	
	MIBL01	0	Young		8		105		7		5		14		34		97		227		1		55		7			
Pyramid Mountain	PYRA1B	1	Old	2019	4	5	0	0	0	0	0	0	6	12	0	0	16	26			0	0	295	60	0	0	3	2
	PYRA2C	0	Old		2	1	0	0	0	0	0	0	19	17	0	1		18			7	4	182	141	4	2	9	5
	PYRA3B	1	Old	2019	0	0	0	0	1	1	0	0	9	12	0	0		9			1	0	246	132	1	1	2	2
	PYRA4C	0	Old		1	1	0	0	4	0	0	0	14	6	0	0	13	27		41	1	1	155	211	2	1	4	4
Sweet Hollow	SWHO02B	0	Young		96		34		0		7		13		5		172		92		0		31		0		0	
	SWHO02B	1	Young	2020	93		21		0		1		13		7		179		96		0		45		0			
Tourne County Park	TOUR1B	1	Old	2019	13	27	26	18	18	9	8	16	37	27	15	11	79	96	82	125	3	3	132	134	6	5	4	2
	TOUR2C	0	Old		15	22	0	0	5	6	0	0	11	14	0	0	46	54	50	38	4	5	70	81	5	15	2	2
	TOUR3B	1	Young	2019	10	23	29	39	6	8	4	18	22	15	3	4	54	76	148	162	0	1	92	70	5	6	1	1
	TOUR4C	0	Young			16		50		11		7		26		12		121		186		3		96		2		1
Voorhees State Park	VOOR03	1	Old	2019	17	11	0	0	0	0	0	0	16	10	0	0	22	33			2	1	44	59	4	10	2	1
	VOORNEB	2	Young	2019	22	16	13	19	2	1	2	2	4	11	0	0	30	50	37	69	3	3	60	87	17	13	2	3
	VOORNEC	0	Young			15		77		5		2		15		1		72		192		1		150		5		2
	VOORNWC	0	Old		38	31	6	7	1	1	0	0	30	25	0	0	75	84	81	85	9	5	162	159	20	14	3	3
Washington Valley Park	WAVP10N	1	Young	2019	2	1	66	64	9	3	74	59	3	5	42	17	119	410	148	118	0	0	56	49	0	1	1	1
	WAVP10N	0	Young		20		81		0		0		9		20						0				0		2	
	WAVP3O	1	Old	2019	1		5		1		25		13		14		27		57		3		74		3		5	
	WAVP3O	2	Old	2021	1		10		0		3		21		12		27		86		0		75		3		5	
	WAVP3O	0	Old		0		0		1		0		11		2						5				6		8	
			Control (AVG)		**16**	**13**	**53**	**40**	**3**	**3**	**8**	**5**	**13**	**14**	**5**	**2**	**126**	**97**	**140**	**114**	**5**	**7**	**115**	**141**	**11**	**14**	**3**	**2**
			Burn (AVG)		**13**	**11**	**32**	**34**	**4**	**3**	**12**	**15**	**17**	**17**	**9**	**8**	**83**	**105**	**100**	**110**	**2**	**2**	**92**	**97**	**5**	**6**	**2**	**1**