Effects of White-Tailed Deer

on vegetation structure and woody seedling composition at Manassas National Battlefield Park, Virginia

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Introduction

Manassas National Battlefield Park is located on the northern tip of the Piedmont Plateau, within the Culpepper Basin, a large Mesozoic trough extending north from the central Piedmont (Fleming and Weber 2003). The park is located approximately 4 km (2.5 mi) northwest of Manassas, Virginia, and 42 km (26 mi) west of Washington, DC. The park comprises 2,111 ha (5,212 acres) of which 839 ha (2,073 acres) are forest, varying from early successional stands of Virginia pine (*Pinus virginiana*) to relatively mature oak-hickory and bottomland hardwood forests (Fleming and Weber 2003). Hay fields, abandoned fields, and a high-use administrative area account for 1,215 ha (3,000 acres) of the park. The mosaic of woodlands and fields is ideal habitat for white-tailed deer (*Odocoileus virginianus*, fig. 1). Lands adjacent to the southern and eastern boundaries of the park consist of

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residential areas; for now, lands adjacent to the western and northern boundaries of the park remain relatively undeveloped. Development is proceeding at a rapid pace and the battlefield park is becoming an oasis. White-tailed deer densities in the park are high, at about 1 deer per 4 acres (63.4 ± 7.7 deer/km^2, estimated in fall 2000–2004 using spotlight counts with distance sampling; Bates 2005). All forests within the park have a prominent browse line (fig. 2).

types, white-tailed deer can negatively affect forest stand development by reducing growth and survival rates of tree seedlings and saplings, and preventing adult recruitment into tree populations (see review by Russell et al. 2001). Research also suggests that white-tailed deer may cause irreversible shifts in successional stable-state forest communities by altering the species composition of plant communities (Stromayer and Warren 1997, Augustine et al. 1998).

Figure 2. A prominent browse line is evidence of a dense population of white-tailed deer at Manassas. Browsing by deer may be affecting shrubs and herbs in the forest interior to levels that may be detrimental to wildlife species that depend on thick understory vegetation to thrive. NPS/BRYAN GORSIRA

Background

White-tailed deer are at historically high densities in most of the eastern United States (McCabe and McCabe 1984). In some areas, deer densities are estimated to be two to four times higher than pre-European settlement densities (Redding 1995, Van Deelen et al. 1996). Deer density in the park is approximately 63.4 deer/km^2 (~~24.5~~ deer/mi^2) (Bates 2005), which greatly exceeds the estimated carrying capacity of 15.4 deer/km^2 (~~5.9~~ deer/mi^2) for the Virginia Piedmont (Whittington 1984).

The primary factor determining the magnitude of white-tailed deer effects on vegetation is density of deer in an area (Tilghman 1989, Stromayer and Warren 1997, Russell et al. 2001). A substantial amount of research evidence suggests that for some community

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Effects of browsing, however, do not appear to be consistent across the range of white-tailed deer (Russell et al. 2001). For example, some studies detected no effects of white-tailed deer on plant survival and growth, or found only sporadic effects during some years and seasons, for particular sites, or for some deer densities (Russell et al. 2001). The tolerance of a plant community to browsing may vary within community types and among physiographic regions because of differing abiotic and biotic factors of the environment (Mladenoff and Stearns 1993, Augustine and McNaughton 1998, Liang and Seagle 2002). The objectives of our study were to investigate and compare the effects of browsing by white-tailed deer on the vegetation structure and woody seedling composition in three forest types. In particular, we were interested in the effects deer might be having on the succession of the forests in Manassas National Battlefield Park.

Figure 1 (left). Abundant in many parts of the eastern United States, white-tailed deer are a management concern at Manassas National Battlefield Park in Virginia. Recent research at the park measured the effects of white-tailed deer on three forest plant communities, documenting changes in forest structure and succession. NPS/BRYAN GORSIRA

Deer exclosures

We compared the effects of deer browsing on three forest types in the park for five years from 2000 to 2004. The forest types studied were oak-hickory, Virginia pine-eastern red cedar (*Juniperus virginiana*) successional, and Piedmont/mountain bottomland, as described by Fleming and Weber (2003).

We collected vegetation data from 10 exclosures (2 x 6 m; 6.6 x 19.7 ft) and 10 control plots in each forest type from June to August each year of the study. Exclosures (fig. 3) were constructed in June 2000 and consisted of welded wire fence, 2 m (6.6 ft) tall, with mesh openings (5 x 10 cm; 2 x 4 in) that facilitated the passage of small mammals. Within the center of each exclosure, we established a vegetation plot (1 x 4 m; 3.3 x 13.1 ft) using metal stakes at each corner. An adjacent control plot (1 x 4 m; 3.3 x 13.1 ft) was paired with each exclosure and located 1 m (3.3 ft) from the side opposite the exclosure entrance. All exclosures were randomly located in and among forest types using a random location generator within ArcView 3.1 (Environmental Systems Research Institution, Redlands, California).

Methods

We estimated ground cover using the point-intercept method (Hays et al. 1981) in three corners of each plot. A

frame (0.5 x 0.5 m; 1.6 x 1.6 ft) with a 10-cm (3.9-in) interval grid (16 points) was placed on the corner. We recorded the type of ground cover below each point (48 points per plot) using the following categories: litter, forb (i.e., all broadleaf plants, including seedlings), grass, fern, moss, and soil. We excluded the corner nearest the exclosure and control entrance because of possible bias from vegetation being trampled.

We estimated vertical plant cover using a vegetation profile board (0.5 x 1.5 m; 1.6 x 4.9 ft; Nudds 1977). The profile board was divided into three 0.5-m (1.6-ft) sections (with 25 squares in each section) and placed at one end of the plot. The number of squares not obstructed by vegetation for each section was then recorded by an observer at the opposite end of the plot. The procedure was repeated with the profile board and observer shifted to obtain counts for the other half of the plot.

We determined survival rates of woody plant seedlings by tagging at least one representative seedling of each species in each plot at the beginning of the study. Because of difficulties identifying species, ash (*Fraxinus* spp.), blueberry (*Vaccinium* spp.), hickory (*Carya* spp.), and oak (*Quercus* spp.) were identified only to genus. Oaks were further divided into either red or white categories based on the presence of bristles on the leaves (Petrides 1972). The height and status (alive or dead) of each tagged seedling was recorded each year of the study. We tagged more than 450 seedlings.

Annual seedling survival rates were consistently significantly lower in the controls than in the exclosures, with few exceptions.

Results

We analyzed only forb cover because of too few numbers in the other ground-cover categories. At the start of the study (2000) the amount of forb cover among treatments (controls vs. exclosures) was not different for any of the forest types ($P = 0.136$). In the bottomland hardwood forest, which flooded in fall 2002 and spring 2003 with declines in forb cover in 2003, overall (control and exclosure combined) forb cover tended to decrease over time (fig. 4b); in the oak-hickory and Virginia pine-red cedar forests overall forb cover tended to increase with time (figs. 4a and 4c, respectively). Forb cover in the controls remained relatively stable compared to the exclosures where forb cover clearly increased (with the exception of noted flood impacts on the bottomland forest) (fig. 4).

Figure 3 (left). A deer exclosure in pine habitat at Manassas. Exclosures like this are used to measure the vegetation response from excluding deer. NPS/BRYAN GORSIRA

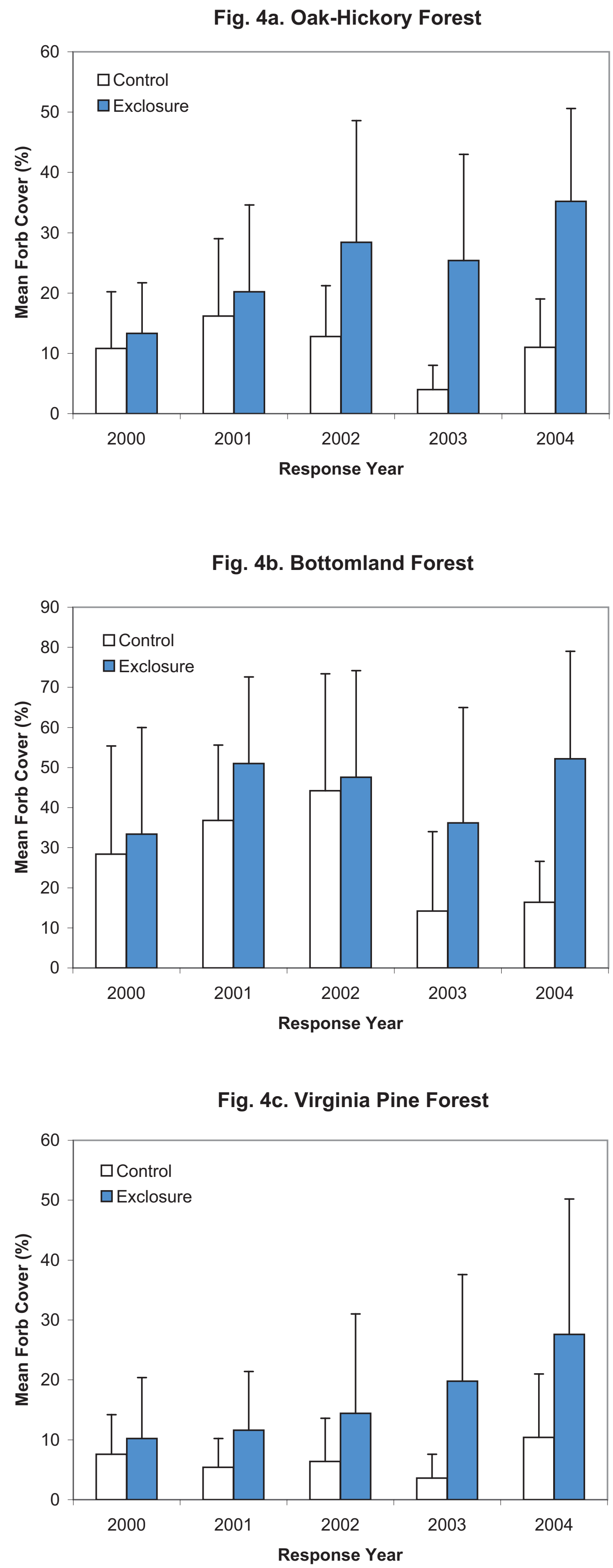

Figure 4. Mean percentage by year of forb cover inside and outside deer exclosures in three forest types—oak-hickory, bottomland, and Virginia pine—at Manassas National Battlefield Park.

We analyzed vertical plant cover in the three forest types at three height intervals: bottom (0–0.5 m, 0–1.6 ft), middle (0.6–1.0 m, 2.0–3.3 ft), and top (1.1–1.5 m, 3.6–4.9 ft) (table 1). At the start of the study we found a significant effect of forest type on the bottom interval for overall vertical plant cover ($P < 0.001$). Similar to results for forb cover, we also noted a greater amount of overall vertical cover of the bottom interval of the bottomland hardwood forest at the start of the study than in the other two forest types (table 1).

Trends in overall vertical plant cover from year 1 to year 5 of the study differed among forest types for the bottom interval ($P = 0.021$), though we found no differences in trends among forest types for the other intervals ($P = 0.941$ and $P = 0.348$). Flooding decreased vertical plant cover of the bottom interval of the bottomland hardwood forest in 2003 (table 1).

A significant treatment-by-forest-type interaction occurred at each interval of vertical cover (all $P < 0.026$). At the beginning of the study, vertical cover at each height interval was greater in the controls than in the exclosures in the Virginia pine-eastern red cedar forest, while the opposite was true for the bottomland hardwood forest; vertical cover in the oak-hickory forest was greater in the controls in the top and bottom intervals, but lesser in the middle interval (table 1). There is a significant treatment-by-year interaction at each height interval (all $P < 0.042$); trends in vertical cover at all heights were consistently less in controls than in exclosures (table 1).

With few exceptions, annual seedling survival rates were consistently significantly lower in the controls than in the exclosures (table 2). Canopy species displaying the greatest mortality from year 1 to year 5 of the study (control treatments) included ashes, hickories, red maple (*Acer rubrum*), and red and white oaks (*Quercus rubra* and *Q. alba*, respectively; table 2). Of the shrub and subcanopy species, boxelder (*Acer negundo*), black hawthorn (*Crataegus* spp.), and spicebush (*Lindera benzoin*) had the greatest mortality from year 1 to year 5 in the control treatments; mortality was not statistically significant for blueberry and redbud (*Cercis canadensis*; table 2). Seedling heights were not analyzed because of high seedling mortality.

Discussion

Herbivory by deer severely impacted forb cover in all three forest types. At the beginning of our study, forb cover was similar between treatments in each of the forest types. By the fifth year forb cover in the exclosures was at least 30% greater than in the controls (see fig. 4). Forbs constitute a large proportion of the white-tailed deer's diet and are heavily consumed in late spring and early summer (McCullough 1985). On the Piedmont Plateau, forbs account for 55.9% of a deer's diet during spring and summer (Whittington 1984).

Herbaceous cover is an important habitat requisite to many species of wildlife, including small mammals (Rossell and Rossell 1999) and ground-nesting birds such as the golden-winged warbler (*Vermivora chrysoptera*; Rossell et al. 2003). Only a few studies, however, have investigated the impacts of deer on herbaceous cover. In a 20-year photographic study, Hough (1965) reported that deer herbivory progressively decreased herbaceous cover in a virgin hemlock hardwood forest in northwestern Pennsylvania. Tilghman (1989) in contrast, found no significant impacts on herbaceous cover at five different deer densities (0–30 deer/km^2 or ~~0–11.6~~ deer/mi^2) in uncut stands of Allegheny hardwood forests on the Allegheny Plateau in northwestern and north-central Pennsylvania.

Deer browsing suppressed vertical plant cover in each forest type in a manner similar to forb cover. The impacts on vertical cover were particularly pronounced during the last two years of our study as substantial differences accrued among treatments (table 1). Vertical plant cover is an important habitat attribute to understory bird species. It has been positively correlated with the abundance and species richness of breeding birds (McShea and Rappole 1992) and the abundance and species diversity of wintering birds (Zebehazy and Rossell 1996). It also has been negatively correlated with predation rates of artificial ground nests (Greenberg et al. 2002). To our knowledge only McShea and Rappole (1992) included vertical plant cover as part of an investigation of deer impacts on vegetation. In their study they concluded that browsing by white-tailed deer (90 deer/km^2; ~~35~~ deer/mi^2) reduced vertical plant cover to the point that it adversely affected the understory bird community in an oak forest of northern Virginia. Other studies throughout the eastern United States, although not directly measuring vertical plant cover, also have reported that deer browsing negatively affects the understory structure of a forest by reducing woody stem densities and heights (e.g., Hough 1965, Alverson et al. 1988, Tilghman 1989, Healy 1997).

Conclusive results that deer reduce survival rates of woody plant seedlings are rare because few studies have monitored individual plants (Russell et al. 2001). In our study we tracked survival of tagged seedlings representing each woody species in each plot over a five-year period. Results clearly indicate that deer browsing adversely affected seedling survival rates of all species except for hackberry (*Celtis occidentalis*), blueberry, and redbud (table 2). In the only

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other study that quantified seedling survival rates, Liang and Seagle (2002) found that deer (20–30 deer/km^2; ~~7.7–11.6~~ deer/mi^2) significantly reduced survival of green ash (*Fraxinus pennsylvanica*) and American hornbeam (*Carpinus caroliniana*), and generally reduced survival of red maple, sweetgum (*Liquidambar styraciflua*), American beech (*Fagus grandifolia*), and tulip poplar (*Liriodendron tulipifera*).

Table 1. Vertical plant cover in three forest types at Manassas National Battlefield Park, Virginia

Height interval	Forest type	Treatment	Mean % vertical plant cover (SD) Year 1: 2000	Year 2: 2001	Year 3: 2002	Year 4: 2003	Year 5: 2004	% Change
Bottom (0–0.5 m)	Oak-Hickory	Control	44.8 (21.5)	37.6 (20.9)	42.4 (19.9)	35.4 (23.5)	37.8 (13.6)	-7.0
		Exclosure	43.0 (17.5)	53.8 (17.5	54.0 (19.8)	63.8 (19.1)	61.0 (23.4)	18.0
	Bottomland hardwood	Control	79.3 (14.1)	65.4 (28.1)	82.9 (24.7)	43.4 (28.2)	58.0 (21.0)	-21.3
		Exclosure	83.0 (19.4)	82.6 (17.6)	71.6 (26.1)	65.6 (33.5)	82.9 (24.9)	-0.1
	Virginia pine-Eastern red cedar successional	Control	52.8 (24.9)	55.6 (17.6)	52.8 (26.8)	31.8 (21.7)	39.2 (24.2)	-13.6
		Exclosure	24.2 (19.1)	33.2 (23.3)	48.5 (20.7)	37.0 (23.4)	53.8 (18.7)	29.6
Middle (0.6–1.0 m)	Oak-Hickory	Control	12.0 (16.4)	15.0 (16.0)	14.2 (20.6)	7.6 (17.0)	7.6 (15.9)	-4.4
		Exclosure	13.4 (16.5)	17.4 (16.8)	12.7 (16.0)	21.2 (22.3)	17.6 (22.1)	4.2
	Bottomland hardwood	Control	19.3 (23.9)	25.0 (31.4)	24.2 (38.2)	6.0 (8.0)	9.4 (14.9)	-9.9
		Exclosure	25.8 (29.8)	36.8 (38.2)	40.2 (39.9)	36.0 (35.2)	38.6 (29.0)	12.8
	Virginia pine-Eastern red cedar successional	Control	42.2 (31.0)	49.2 (27.2)	50.8 (29.5)	21.2 (26.7)	22.2 (25.0)	-20.0
		Exclosure	11.8 (16.1)	21.6 (23.8)	20.3 (28.1)	14.2 (25.4)	25.4 (23.9)	13.6
Top (1.1–1.5 m)	Oak-hickory	Control	20.6 (25.6)	17.6 (18.6)	12.7 (13.6)	1.6 (3.9)	12.6 (20.0)	-8.0
		Exclosure	15.6 (16.9)	14.6 (20.9)	24.0 (23.7)	15.6 (18.1)	10.2 (12.1)	-5.4
	Bottomland hardwood	Control	10.4 (20.4)	21.2 (23.6)	17.3 (32.3)	8.8 (15.4)	6.0 (9.0)	-4.4
		Exclosure	12.2 (23.7)	16.4 (17.1)	36.4 (34.2)	26.8 (40.1)	27.7 (32.9)	15.5
	Virginia pine-Eastern red cedar successional	Control	46.6 (29.6)	46.6 (35.7)	61.3 (35.3)	22.2 (26.8)	27.2 (31.2)	-19.4
		Exclosure	18.0 (32.9)	26.6 (32.7)	25.5 (34.0)	15.8 (27.6)	23.2 (25.3)	5.2

Note: Percentage of vertical plant cover was estimated in 10 control plots and 10 exclosures for three forest types—oak-hickory, bottomland hardwood, and Virginia pine-eastern red cedar successional—at three height intervals. Control plots and exclosures measured 1 x 4 m each.

Table 2. Survival of tree and shrub seedlings at Manassas National Battlefield Park, Manassas, Virginia

		Year 1 (2000)	Year 2 (2001)		Year 3 (2002)		Year 4 (2003)		Year 5 (2004)		
Species[1]	Treatment	# Seedlings	Survival rate	*P*-value	Survival rate	*P*-value	Survival rate	*P*-value	Survival rate	*P*-value	# Seedlings
Green and white ash	Control	51	0.314	<0.001	0.216	<0.001	0.176	<0.001	0.118	<0.001	5
	Exclosure	54	0.889		0.833		0.630		0.574		32
Black cherry	Control	23	0.696	0.934	0.261	0.026	0.217	0.026	0.130	0.001	3
	Exclosure	15	0.667		0.600		0.467		0.467		7
Boxelder	Control	15	0.467	0.751	0.067	0.006	0.000		0.000		0
	Exclosure	19	0.421		0.211		0.105		0.053		1
Black hawthorn	Control	14	0.643		0.500	0.073	0.286	0.022	0.214	0.015	3
	Exclosure	11	1.000		0.909		0.909		0.818		9
Vaccinium spp.	Control	7	0.429	0.066	0.429	0.117	0.429	0.213	0.429	0.213	4
	Exclosure	12	0.917		0.833		0.750		0.750		9
Hackberry	Control	22	0.545	0.021	0.409	0.601	0.182	0.322	0.091	0.081	2
	Exclosure	19	0.842		0.526		0.316		0.316		6
Hickory	Control	11	0.273	<0.001	0.091	0.020	0.000		0.000		0
	Exclosure	9	0.889		0.778		0.667		0.667		6
Red maple	Control	16	0.125	0.021	0.000		0.000		0.000		0
	Exclosure	13	0.615		0.462		0.462		0.385		5
Rebud	Control	7	0.429	0.087	0.429	0.424	0.429	0.999	0.286	0.555	2
	Exclosure	15	0.800		0.600		0.400		0.400		6
Red oak group	Control	18	0.333	<0.001	0.333	0.018	0.167	0.023	0.111	0.023	2
	Exclosure	17	0.882		0.765		0.588		0.529		9
Spicebush	Control	12	0.583		0.500	0.006	0.083	0.005	0.083	0.086	1
	Exclosure	10	1.000		0.900		0.600		0.300		3
White oak	Control	9	0.556	0.131	0.444	0.246	0.111	0.033	0.111	0.033	1
	Exclosure	11	0.909		0.727		0.727		0.727		8

Note: Seedlings were tagged in 2000, the first year of the study. Seedling survival was monitored 2000–2004 in 10 control plots and 10 exclosures in three forest types: oak-hickory, bottomland hardwood, and Virginia pine-eastern red cedar successional. Control plots and exclosures measured 1 x 4 m each. *P*-values could not be calculated for treatments having survival rates of 0 or 1.

[1]Green ash (*Fraxinus pennsylvanica*) and white ash (*F. Americana*), black cherry (*Prunus serotina*), boxelder (*Acer negundo*), black hawthorn (*Crataegus* spp.), *Vaccinium* spp. (deerberry and lowbush blueberry), hackberry (*Celtis occidentalis*), hickory (*Carya* spp.), red maple (*Acer rubrum*), redbud (*Cercis canadensis*), red oak group (*Quercus rubra* and *Quercus* spp.), spicebush (*Lindera benzoin*), and white oak group (*Quercus alba* and *Quercus* spp.).

In our study seedling survival rates varied among species, suggesting that deer selectively browse across forest types. Selective browsing often alters species composition of a forest stand or ecosystem when preferred species are removed (Augustine and McNaughton 1998, Russell et al. 2001, Liang and Seagle 2002). By the fourth year of our study boxelder, hickory, and red maple seedlings were completely eliminated from control plots, while red and white oak seedlings were severely reduced (see table 2). In addition, ash, black cherry (*Prunus serotina*), and hackberry were the most abundant species at the beginning of our study and continued to be the most abundant at the end (see table 2). These results suggest deer browsing is directing succession of forests toward stands with fewer species and a greater dominance of ash, black cherry, and hackberry, particularly in the oak-hickory and bottomland hardwood forests that are currently dominated by species impacted in our study. Our supposition is supported by Tilghman (1989), who found that browsing at deer densities of 30 deer/km^2 (~~11.6~~ deer/mi^2) caused a dramatic shift in species composition of Allegheny hardwood forests, favoring black cherry. Healy (1997) also found that deer browsing (10–17 deer/km^2 or ~~3.9–6.6~~ deer/mi^2) interrupted oak regeneration in oak forests of Massachusetts, and predicted that future stands would be dominated by white pine (*Pinus strobus*), red maple, and sweet birch (*Betula lenta*).

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Conclusions

This study investigated the effects of deer browsing on three forest types common to the northern Piedmont Plateau. Results indicate that white-tailed deer are having a significant impact on the structure and woody seedling composition of forests in Manassas National Battlefield Park and are changing the forest successional process. In each forest type, forb cover and vertical plant cover were suppressed, and species richness and seedling survival rates were reduced. No differences in browsing effects were apparent among the forest types, which may be indicative of the browsing intensity during our study. At 63.4 deer/km^2 (~~24.5~~ deer/mi^2), deer density in the park greatly exceeds the estimated carrying capacity of 15.4 deer/km^2 (~~5.9~~ deer/mi^2) for the Virginia Piedmont (Whittington 1984). Thus, browsing levels in our study were likely too high to discern any potential differences in browsing tolerances among forest types.

The findings in this ecological study justify the need to begin actively managing the deer population in Manassas National Battlefield Park. Browsing by white-tailed deer may be impacting the herb and shrub layers in the forest interior to levels that may be detrimental to wildlife species that are dependent on a thick understory to thrive. In addition, we predict that the future composition of forests in the park, particularly in the oak-hickory and bottomland hardwood types where the greatest number of current dominants is most affected, will shift toward stands with fewer species and a greater dominance of ash, black cherry, and hackberry.

The dilemma for managers at Manassas National Battlefield Park and at the many other parks with high deer numbers is how to deal with this situation. Should high deer numbers be treated as a population fluctuation that will resolve itself naturally, or should we employ heavy-handed management treatments such as population reductions or contraception? Either scenario has the potential for as yet unknown repercussions, including trophic-level responses resulting from population reductions and behavioral changes from heavy use of contraception. In addition, parks have porous borders that, without cooperation from adjacent landowners to control deer populations outside the park, could reduce the effectiveness of treatments within the park. For now, this research information, combined with the results of the distance sampling, has prompted us to develop a request for funding of an environmental impact statement to investigate these and other management options. In the meantime, we continue to collect data from the deer exclosure and control plots with plans to analyze plant species richness and diversity.

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