THE IMPACT OF RECREATIONAL HUNTING OF WHITE-TAILED DEER ON NATIVE AND NON-NATIVE PLANT ABUNDANCE IN NORTHERN PENNSYLVANIA

By

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DEDICATION AND ACKNOWLEDGEMENTS

I would like to dedicate this paper to my best friend and fiancé Devin Shadle, who has the distinction of being one of the most kind and patient people I have ever met. While working through this paper, he not only helped me collect my field data but also reassured me during my many periods of doubt that I was capable of getting this project done.

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ABSTRACT

The introduction and spread of non-native plant species is a large area of concern for land managers and ecologists, because these species are often capable of outcompeting native species, altering vegetative community dynamics by reducing species richness and ecosystem function. Low species diversity and disturbances in community structure are factors that may be associated with facilitating the success of non-native species. The white-tailed deer (*Odocoileus virginianus*) is a large herbivorous animal that can alter plant communities through a variety of mechanisms, including trampling and grazing. At high population densities, white-tailed deer can reduce plant species richness and alter plant community interactions, potentially facilitating an increase in the abundance and richness of non-native plants. Hunting is an effective tool for controlling high deer populations. In this study, I examined the impact of the presence or absence of recreational hunting on native and non-native plant species in habitats that were known to contain deer in north central Pennsylvania. I sampled native and non-native plant species richness and percent cover within $1m^2$ quadrats placed at 10 random locations along five transects. Two transects were in non-hunted, privately owned land and three transects were in public game lands. Each transect was sampled on five different dates, except for one transect in the private land, which was sampled four times. These transects represented a variety of habitat types. Plant species richness and percentage cover in hunted and non-hunted plots were analyzed with linear and generalized linear models using R statistical software. I found that native species richness was greater and non-native species abundance was higher at the public game lands compared to the site where hunting was prohibited. The greater native species richness in hunted sites suggests that controlling deer populations densities with hunting may play a role in maintaining the health of vegetative communities. Future studies should look at vegetative data in additional habitat types and survey sites across hunted and non-hunted lands to improve the certainty of these findings.

INTRODUCTION

The anthropogenic introduction of non-native species has significantly altered plant communities throughout the world (Mack et al. 2000). Most of the novel species introduced to new habitats by humans fail to survive in their introduced habitats, and many of those that become naturalized are either neutral or even positive influences in their new communities (Mack et al. 2000). However, some of these introduced species are a cause for concern for ecologists and land managers because of their potential to outcompete native species and alter vegetative community composition (Levine et al. 2003; Vellend et al. 2017). Those species that persist and spread in these new habitats to the detriment of native plant structure and function are defined as invasive (Mack et al. 2000). As the displacement of plants throughout the world is showing no signs of abating, the study of invasive species and efforts to discover the factors that lead to their success have similarly grown (Early et al. 2016).

Introductions of non-native species can be intentional, such as in the introduction of ornamentals, or accidental. A major source of displaced species is the unintentional transport of propagules on internationally shipped goods (Hulme et al. 2008; Early et al. 2016). Once non-native plants successfully establish in an ecosystem, they may outperform native species, reducing ecosystem species richness and possibly leading to native species extinction (Levine et al. 2003). Non-native species success can also alter ecosystem function due to the relationship between ecosystem functions, which include decomposition, nutrient retention, and plant productivity, and species richness (Naeem et al. 1994; Cardinale et al. 2012). The deleterious

effects of biodiversity decline on ecosystem function accelerate as biodiversity loss continues (Cardinale et al. 2012).

There are multiple theories about what leads some non-native species to become invasive (Catford et al. 2009). An early theory proposed by Baker (1974) is that the success of invasive plants is due to the characteristics they often share with weedy species; both weedy species and successful invasive are often early successional species, possessing rapid growth and producing large amounts of viable seeds. The invasiveness of a plant can be correlated to high values of traits associated with plant performance, namely fitness, growth rate, leaf-area allocation, shoot allocation and size (van Kleuven et al. 2010). Invasiveness has also been associated with high trait plasticity (Yeh & Price 2004; Richards et al. 2006). The enemy release hypothesis, which conjectures that the success of invasive species is due to the introduction of non-natives into a habitat with no natural enemies, is another prominent theory that attributes the development of invasive species to the characteristics of the community where non-native species are introduced (Keane & Crawley 2002).

Additional characteristics of the recipient plant community that may influence the growth of non-native plant species are community biodiversity and structure. Systems with high species richness may be less susceptible to non-native introductions (Tilman 1997; Levine et al. 2004), and the identity of dominant plant species in a community also plays a role in non-native success (Emery & Gross 2006). Ecological disturbances such as climate change, physical soil disturbance, and nutrient enrichment can disrupt ecosystem structure and lead to the opening of ecological niches in plant communities that facilitate the establishment of non-native species (Lake & Leishman 2004; Milbau et al. 2013; Jauni et al. 2015). In addition, disturbance may reduce or even remove species from the community that would compete with the non-native

species (Byers 2002).

Once non-native species are introduced into an ecosystem, restoring a plant community to its former state is often difficult or unsuccessful. Eradication of a non-native species requires either the reduction of the population below sustainable levels or the removal of all individuals from the community (Myers et al. 2000). These efforts tend to be very expensive and are prone to failure (Myers et al. 2000). In addition, successful eradication of non-native species can lead to unintended negative consequences on ecosystem properties and processes, especially in instances where non-native species are numerous or they have replaced a niche previously occupied by a native species in the community (Zavaleta et al. 2001; Corbin & D'Antonio 2011). Due to the cost, logistical challenges, and possible negative effects of invasive species eradication, the most effective non-native species management strategy is preventing the establishment and spread of these introduced species. This requires an understanding of the factors that facilitate the establishment and success of exotic species invasion (Davies & Sheley 2007; Early et al. 2016).

The White-Tailed Deer

Animals are believed to play a major role in altering the abundance of non-native species due to their ability to shape the diversity and relative abundance of plants within their communities. For example, large herbivores can alter forest communities by crushing plants (Heckel et al. 2010), spreading propagules as they move throughout their range (Vellend et al. 2003; Castellano et al. 2013), or through grazing (Côté et al. 2004; Averill et al. 2018). In particular, the grazing preferences of large herbivores for certain palatable plant species may influence species diversity by altering the competitive interactions within the community, resulting in changes in species diversity (Côté et al. 2004). The movement and grazing of large

herbivores also cause ecosystem disturbances, which facilitate the establishment and non-native species by providing a niche space in the plant community (Jauni et al. 2015).

One species that can have a considerable effect on plant diversity and structure is the white-tailed deer (*Odocoileus virginianus*), hereafter referred to as WTD. WTD are one of the more widespread and abundant large herbivores in North America. As opposed to the populations and territory sizes of most other large native herbivores, such as bison (*Bison bison*) and elk (*Cervus canadensis*), WTD populations and ranges increased significantly throughout the 20th century (Laliberte & Ripple 2004). In mixed forest and agricultural habitats that are especially conducive for WTD success, WTD densities may be as great as 60 deer/$km^{2,}$, many times greater than estimated WTD populations before the European arrival in North America (Porter et al. 1994). Much of the change in WTD populations can be attributed to anthropogenic factors such as increases in forage availability due to agriculture, the elimination of native predators, such as wolves (*Canis lupus*) and cougars (*Puma concolor)*, and the establishment of laws by the government designed to protect wildlife (Porter et al. 1994).

WTD presence is one of the largest drivers determining the structure and configuration of the understories of forests, even to the extent that WTD are sometimes considered “ecosystem engineers” (Côté et al. 2004). At high population densities, WTD are associated with a decrease in the cover of woody plants and forbs along with an increase in the cover of ferns and graminoids (Côté et al. 2004; Eschtruth & Battles 2009). Under certain conditions, WTD populations can easily exceed the carrying capacity of their habitat. In these situations, the WTD density may cause long-term changes in ecosystem function and structure that persist even after WTD are removed (Côté et al. 2004; Nuttle et al. 2011).

In Pennsylvania's Allegheny Plateau, increases in WTD populations in the early 20th century coincided with the near elimination of eastern hemlock (*Tsuga canadensis*) and witch hobble (*Viburnum lantanoides*) from the landscape, concurrent with decreases in the abundance of small saplings across many species (Leopold et al. 1943; Hough 1965). In the same period, the prevalence of species not typically consumed by WTD such as American beech (*Fagus grandifolia*), hay-scented fern (*Dennstaedtia punctilobula*) and New York fern (*Parathelypteris noveboracensis*) increased significantly (Hough 1965). Overall vegetative biodiversity in this period declined (Rooney & Dress 1997). As the abundance of both WTD and exotic species grew simultaneously in the eastern United States in the last century, some ecologists suggested that the increasing WTD populations were directly facilitating the spread of invasive species (Vavra et al. 2007; Averill et al. 2018). Several WTD exclusion/access studies using vegetation plots have suggested the existence of a direct link between WTD presence and invasive species abundance (Eschtruth & Battles 2009; Knight et al. 2009; Andrea Relva et al. 2010).

Hunting: Controlling WTD Populations

As WTD populations have increased and the influence of an overabundance of these animals has become better understood, their management has become an essential priority. The primary mechanism for controlling WTD populations in ecosystems where there are few natural predators is recreational hunting (Woolf & Roseberry 1998). WTD populations are significantly higher in areas where hunting is prohibited (Frost et al. 1997), and may be higher in privately owned lands compared to public lands (Small et al. 1991). The Pennsylvania Game Commission (PGC) uses recreational hunting as its main tool in managing the state's WTD population (2019). The PGC manages the state's WTD populations by determining management goals using a variety of metrics, including WTD and forest health (**Figure 1)**. Field data showing high WTD

impacts or low vegetation regeneration in forested test plots are an indication to the agency that WTD populations are too high. As a result, the PGC may choose to increase the hunting allocation for antlerless WTD for the subsequent hunting season with the aim of reducing population levels (Pennsylvania Game Commission 2019).

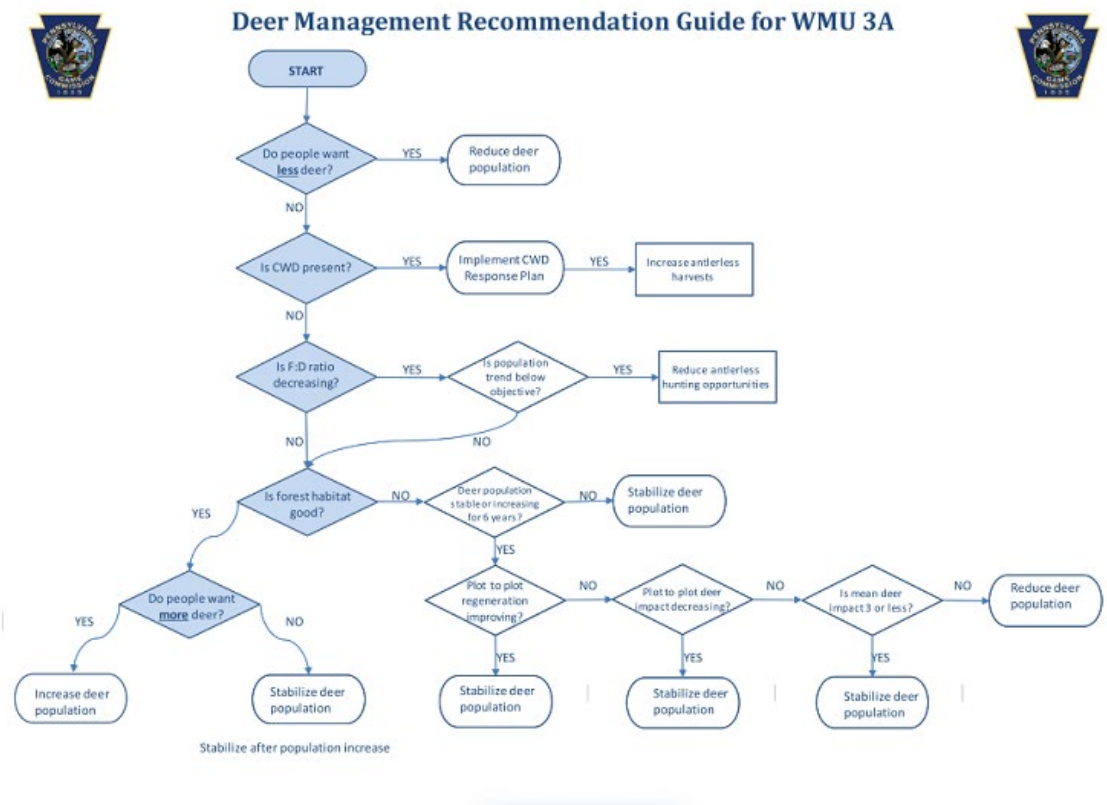

Figure 1) Deer Management Recommendation Guide (Pennsylvania Game Commission 2019)

The effectiveness of hunting for managing WTD populations is dependent on a variety of factors, the first of which is a sufficiently large cohort of hunters. Hunters generally are uninterested in harvesting more than two or three animals in a season regardless of how many WTD they are allowed to hunt (Stedman et al. 2008), so giving a small population of hunters unlimited freedom to hunt is likely to be an ineffective population management strategy.

Unfortunately, from a WTD management standpoint, hunter numbers in the United States are on the decline in most of the country, and these declines are steepest in the Mid-Atlantic (Enck et al. 2000). Overall sales of hunting licenses have declined since their peak in 1982 (U.S. Fish and Wildlife Service), and hunting participation is predicted to continue to decrease due to aging hunter populations, a decrease in the recruitment of younger hunters, and the increasing urbanization of the United States (Winkler & Warnke 2013).

Another major component leading to potential decreases in the efficacy of WTD management efforts is the decline in the amount of land available for hunting. The access and availability of potential private hunting land has declined steadily in recent decades due to development and the management decisions of private landowners, who often are not aware of the adverse effects of WTD overpopulation (Jagnow et al 2006; Stedman et al. 2008). Not having places available to hunt increases the likelihood that hunters will cease participating in the sport (Stedman et al. 2008). In addition, the posting of private land against hunters prevents effective control of WTD populations by limiting hunter's access to lands inhabited by WTD (Hygnstrom et al. 2011). The lack of hunter access to WTD inhabited areas, such as posted private property, leads to the establishment of areas of refugia with extremely high WTD population densities (Frost et al. 1997; Brown et al. 2000; Williams et al. 2013).

Hunters can also play a role in affecting WTD population density by altering their movement patterns through their habitats. WTD will alter their movement as a response to hunting pressures, moving towards areas of greater cover and away from areas with high hunter density (Root et al. 1988; Kilgo et al. 1998; Little et al. 2016). Hunters preferentially hunt close to roads and in areas of lower slope, and hunter density in a given location can be predicted by its slope and distance from the nearest road (Diefenbach et al. 2005). High hunter densities may

cause movement of WTD into areas of the landscape with lower hunter density (Diefenbach et al. 2005). The actions of hunters on moving WTD may cause alterations in populations in hunted areas at a magnitude greater even greater than harvested numbers (Brown et al. 2000; Martin & Baltzinger 2002). Once WTD enter refugia, their populations can remain stable or increase despite high harvest rates in adjacent areas (Brown et al. 2000). The plant communities in these WTD refugia may be more strongly affected by the vegetative disturbances characteristic of WTD activity, and this WTD pressure may facilitate the establishment of exotic species introduced to these ecosystems (Martin & Baltzinger 2002).

The aim of my study is to investigate the potential relationship between the abundance of non-native species and WTD in areas where hunting is used to control WTD populations compared to sites where hunting is prohibited. I predicted that vegetation would differ between sites that are hunted and those that are not, because WTD, particularly at high population densities, have a huge capacity to shape understory vegetation, and their presence can facilitate the establishment of invasive species and decrease overall understory biodiversity. Hunting is a highly effective WTD population management tool (Frost et al. 1997) and hunting may lead to higher WTD mortality and lower populations on public lands (Small et al. 1991), so it was assumed that WTD populations were higher in areas where WTD hunting is prohibited. I expected that the presumed increase in WTD impact at the non-hunted site would be evidenced by higher richness and percent cover of non-native species, concurrent with lower richness and cover of native species. In addition, I predicted overall species richness would be lower at the non-hunted transects than at the hunted transects.

METHODS

Site Description

This study took place in Pennsylvania Wildlife Management Unit (WMU) 3A of the Allegheny Plateau region of the Appalachian Mountains in north central Pennsylvania (**Figure 2**). WMU 3A consists of 3900 km^2 of regions of McKean, Potter, Tioga, and Bradford counties that lie north of U.S. Route 6. These counties and this WMU are part of a larger region of the state known as the Pennsylvania Wilds, which consists of nearly a quarter of the state's land but only 4% of its population (Hill n.d). Portions of this area were glaciated during the Wisconsinan glaciation approximately 25,000 to 12,000 years ago. The glacier's removal of much of the topographical relief in this region is a large factor differing the terrain of the Allegheny Plateau from most of the rest of the Appalachian Mountain Range (Sevon & Fleeger 1999).

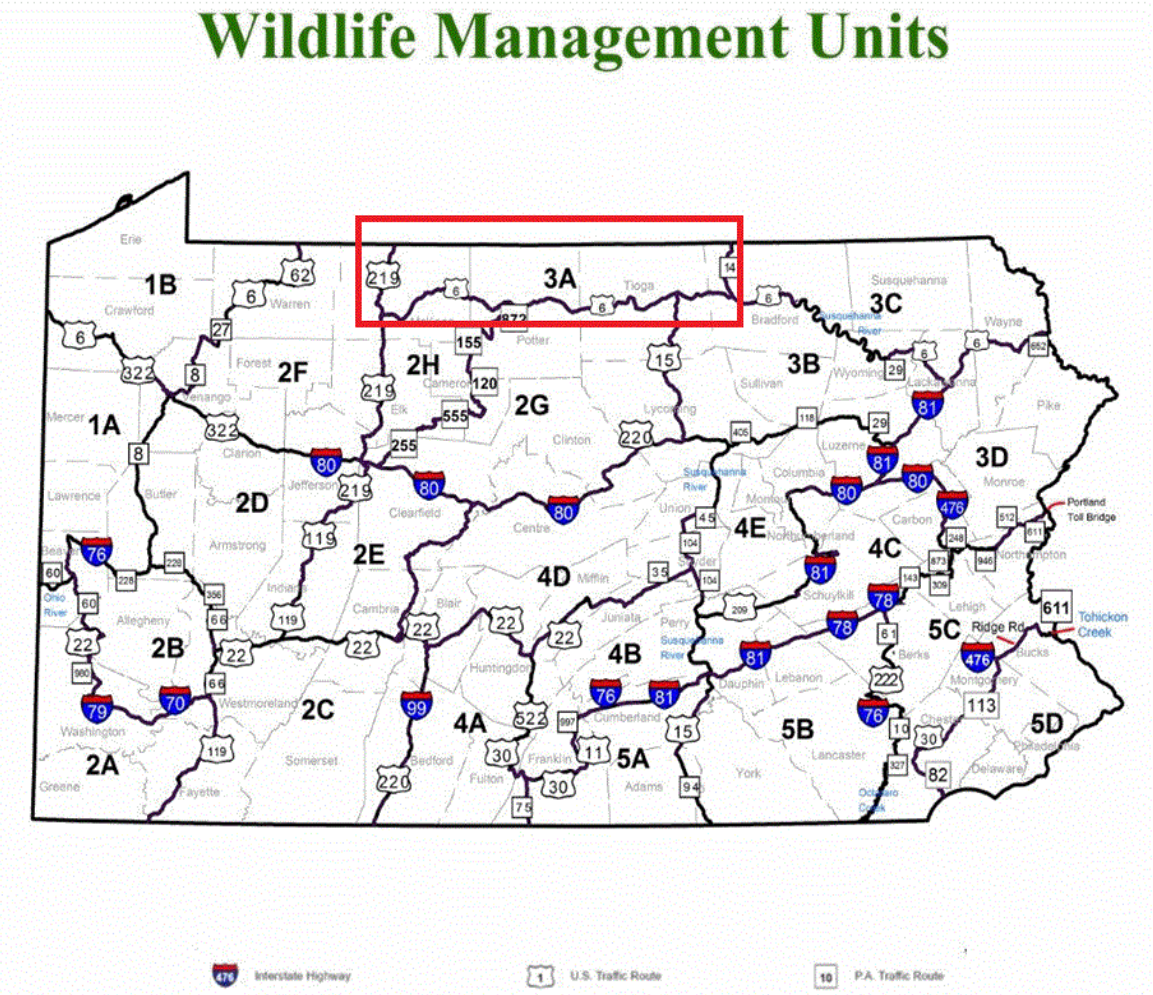

Figure 2) Map of Pennsylvania Wildlife Management Units, WMU 3A Indicated. (Image Source: Pennsylvania Game Commission)

WMUs are determined by the Pennsylvania Game Commission according to habitat and human-related land characteristics such as population, land ownership and use, and the presence or absence of major roads. Current land use in WMU 3A is classified primarily as forest (78%), agriculture (17%) and developed (2%) (PA Game Commission 2019). WMU 3A differs from surrounding WMUs due to its higher agricultural land use. The majority of WMU 3A consists of mixed deciduous/coniferous forest. Most of the forests in WMU 3A are secondary-growth

dominated by hardwood trees such as oaks (*Quercus* spp.) maples (*Acer* spp.), and black cherry (*Prunus serotina*). The old-growth forest, which consisted primarily of eastern hemlock (Tsuga canadensis), eastern white pine (*Pinus strobus*) and American beech (*Fagus grandifolia*), was harvested in the late 19th and early 20th centuries (Blangger 1997). The timbering was so extensive that nearly every tree, regardless of size, was cleared (O'Driscoll 2015). The Köppen classification of this region in north central Pennsylvania is humid continental climate (Beck et al. 2018).

Two sites were selected for vegetation data collection within WMU 3A. These sites were selected to represent an area with a hunted and non-hunted status, respectively. The hunted site was a small portion of Pennsylvania Game Lands (SLG) 64. SGL 64 is a 32.6 km^2 densely wooded plot of land consisting of two separate tracts of land owned by the State of Pennsylvania located in eastern Potter County (41.770833, -77.610899) in Pike and Ulysses Townships (Pennsylvania Game Commission 2014). The forest age distribution in SGL 64 is mainly in the 85-125 year range, older than most forests in this area, largely due to terrain steepness and lack of easy access (PA Game Commission 2014). Dominant habitat types in the area are oak (44%) northern hardwood (43%) and coniferous (7%) forests with some pockets of wetlands and early successional forest (Pennsylvania Game Commission 2014). Surrounding areas are also heavily forested. SGL 64, as the name implies, is open to the public for all game seasons and for appropriate recreation by the public. SGL 64 is typically accessed through parking sites established by the Pennsylvania Game Commission along Phoenix Road (State Route 1005) a minor roadway (Department of Transportation 2009a). Public game lands were chosen to represent the hunted site because the number of hunters who hunted on SGL 64, particularly

around parking sites, was presumed to be higher than at privately owned sites with unknown hunter density.

The non-hunted location is an 0.12 km^2 parcel of privately owned land in Sweden Township (41.770833, -77.610899), hereafter referred to as Haskins in reference to the owner's last name. This site is posted and fenced to prevent hunting and recreational vehicle traffic. The Haskins site is primarily hardwood forest of an unknown age with some meadow habitat, while surrounding land use is largely agricultural. This property is accessed by a private dirt drive branching from another dirt road by the name of Ellis Hollow Road (Township Route 337; Pennsylvania Department of Transportation 2009b). This site was selected to represent the non-hunted survey area because the fence surrounding the property ensured hunters were prevented from access to the property, while WTD, which were able to easily jump the fence, continued to freely move throughout their habitat.

Data Collection

Plant community data was recorded during May and June 2020. A total of five 100 m transects were set across the two sites; three at the State Game Lands and two at the non-hunted property. The transects chosen were representative of the habitats present near parking sites. The choice to select only five transects was determined by concerns about adequate time and resources for gathering field data and identifying plant species.

The locations of transects at the State Game Lands consisted of a riparian area along Phoenix Run, hardwood forest, and coniferous forest (**Figure 3**). The transects in the private land consisted of meadow and hardwood forest sites (**Figure 4)**. Habitat characteristics were recorded at each transect. Ten sampling plots with an area of 1 m^2 were randomly placed along

each 100 m transect at locations determined by randomly generating 10 numbers between 1 and 100 for each transect. To observe seasonal turnover in plant community composition, these transects were visited a total of five times across May and June, except for the Haskins 2 transect which was set during the second surveying date, and thus visited only four times over the course of the data collection period. A complete list of transect names, site habitats, and visit dates are listed in **Table 1**.

Table 1) Description of surveyed transects including hunted status, habitat and visit dates

Transect ID	Hunted Status	Habitat	Visit 1	Visit 2	Visit 3	Visit 4	Visit 5
Haskins 1	Non-hunted	Hardwood	May 23, 2020	June 3, 2020	June 10, 2020	June 16, 2020	June 24, 2020
Haskins 2	Non-hunted	Meadow	N/A	June 3, 2020	June 10, 2020	June 16, 2020	June 24, 2020
State Game Lands 1	Hunted	Riparian	May 19, 2020	May 27, 2020	June 9, 2020	June 16, 2020	June 24, 2020
State Game Lands 2	Hunted	Hardwood	May 19, 2020	May 27, 2020	June 9, 2020	June 16, 2020	June 24, 2020
State Game Lands 3	Hunted	Coniferous	May 19, 2020	May 27, 2020	June 9, 2020	June 16, 2020	June 24, 2020

Figure 3) Location of transects in State Game Lands 64 (Image Credit: Google Earth)

Figure 4) Location of transects on private, non-hunted land. (Image Credit: Google Earth)

I identified nearly all vascular plants observed in sampling plots, and each species was designated as either native or non-native. Plants in these plots were assigned species according to Peterson & McKenny (1996), Cobb et al. (2005), Delaware Department of Natural Resources (2018), and Penn State Extension (2013). When identification wasnewb not possible in the field, photographs and samples were taken for later identification. Native status was determined using the Pennsylvania Invasive Species List (Pennsylvania DCNR) and U.S. Department of Agriculture PLANTS Database (USDA, NRCS 2020). The percent groundcover of each species was also recorded. When plant identification was not possible to at least the genus level, the native status was designated as "Unknown", and each unique unknown species was numbered. The cover and presence of each of these unknown species was recorded along with all identified species.

Species richness and relative percent cover were calculated and analyzed to determine differences in native and non-native abundance between hunted sites and sites that were not hunted. Overall richness for hunted and non-hunted transects was determined by summing all species observed (native, non-native, and unknown) at hunted and non-hunted sites. This richness data was used to make general comparisons between species richness at hunted and non-hunted sites. Species richness by native status was determined by separately summing the number of species classified as native, non-native, and unknown in each plot. This metric was used to make comparisons of richness by native status across hunted and non-hunted sites. Relative cover for native, non-native, and unknown species was calculated by first determining total percentage cover in each plot. Then native, non-native, and unknown cover was expressed as a proportion of the total cover for each plot. This allowed comparisons of cover between

hunted and non-hunted sites that controlled for the range of variation in total cover that occurred across plots, transects, and sites.

Statistical Analysis

All analyses were done using R Statistical Software version 4.0.2 (R Core Team 2020). Linear models were used to assess relationships between richness data and hunted status. Relative cover was analyzed using a generalized linear model with a "logit" link. A generalized linear model was used to analyze relative cover data due to a violation of the assumption of a normal distribution in the relative cover data. The generalized linear model was fit to a quasibinomial distribution to account for overdispersion in the variance of the response variable relative to the means. Results for the generalized linear model were reported as odd ratios after backtransforming from the log odds scale. The independent variables in the richness and relative cover models were: native status (three levels: native, non-native, or unknown) hunted status (two levels: hunted or non-hunted), habitat (4 levels: coniferous, hardwood, meadow, riparian), and sampling visit number (5 levels: 1-5). A native status x hunted status interaction term was included in both models to determine if differences in hunted and non-hunted sites were dependent on native status.

ANOVA tables were used to initially assess the effects of each explanatory variable and interaction on response variables. After assessing ANOVA tables, the emmeans package was used to conduct post-hoc tests of differences between mean native, non-native, and unknown species richness and relative percent cover between hunted and non-hunted transects and to estimate effect size.

The assumptions of equal variance and normality associated with linear models were visually assessed using plots of residuals against fitted values and normal Q-Q plots. No data transformations were deemed necessary beyond the logit transformation in the generalized linear model. Figures for this study were generated using the ggplot2 package in R.

RESULTS

Species Richness

Across all plots, a total of 187 vascular plant species were present. There were a 154 species at the hunted sites, while 66 species were found at the non-hunted transects (**Table 2)**. These numbers include unique unknown species. Overall richness varied between sites; species richness was greatest at the riparian State Game Lands 1 site, and lowest at the coniferous State Game Lands 3 transect. No non-native species were identified at either Haskins 1 or the State Game Lands 3 transects, and every species identified at the State Game Lands 3 site was native. Overall species richness at the two transects with hardwood habitats, Haskins 1 and State Game Lands 2, also differed, with 3.5 times greater species richness at State Game Lands 2.

Table 2) Species richness at each surveyed transect

Transect ID	Native Richness	Non-native Richness	Unknown Richness	Total
Haskins 1	19	0	3	22
Haskins 2	36	13	4	53
State Game Lands 1	73	20	10	103
State Game Lands 2	62	11	3	77
State Game Lands 3	17	0	0	17

After accounting for the effects of habitat and sampling date, there is strong evidence that the difference in species richness between hunted and non-hunted sites depended on native status (**Table 3)**. On average, native vegetation was 2.04 ± 0.27 (mean ± SE) species greater at hunted sites than at the non-hunted sites (t_{338}=7.9 p< 0.0001; **Figure 5**). When accounting for the same factors, there is no evidence of an observed difference in non-native species richness between hunted and non-hunted sites (p=.2862; **Figure 5**). In addition, there is no evidence of a difference in overall richness between hunted and non-hunted sites (p=0.2799; **Figure 5**).

Table 3) ANOVA table of linear models fitted to species richness

RICHNESS	**DF**	**F**	***p*-VALUE**
NATIVE	2	94.98	<0.001
HUNTED	1	47.09	<0.001
HABITAT	3	66.72	<0.001
VISIT NUMBER	1	5.81	0.016
NATIVE:HUNTED	2	31.92	<0.001

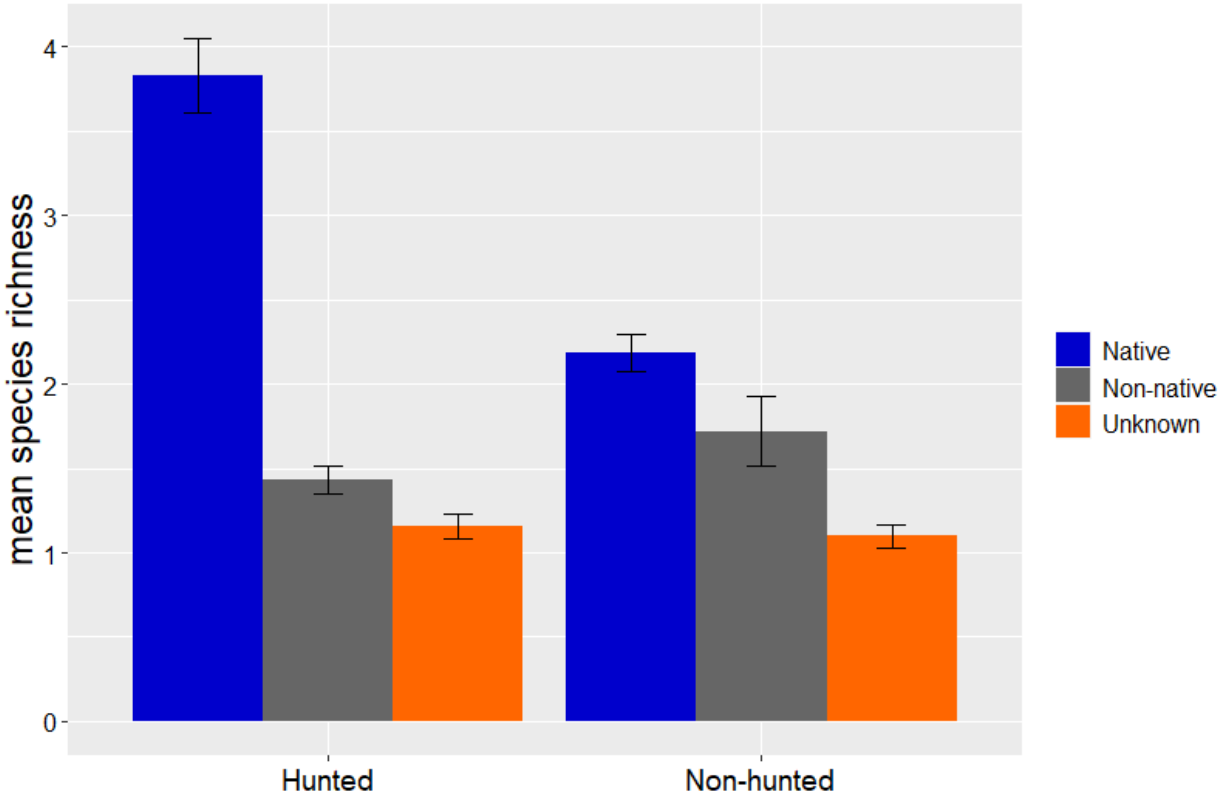

Figure 5) Mean species richness by per plot by native status across hunted and non-hunted sites

Relative Percentage Cover

The difference in relative percentage cover between hunted and non-hunted sites depended on native status (**Table 4)**. There is no evidence that the relative cover of native species differed between hunted and non-hunted sites (Z=-1.376 p=0.1689; **Figure 6**). However, there is also marginal evidence that non-native species accounted for 1.99 ± 0.75 times more of the total observed cover at hunted sites compared to non-hunted sites (Z=1.839 p=0.0659; **Figure 6**).

Table 4) ANOVA table of generalized linear models fitted to relative cover

COVER	DF	DEV RESID	DF RESID	DEV	*p-VALUE*
NULL			653	619.1	
NATIVE	2	406.14	651	212.9	<0.0001
HUNTED	1	0	650	212.9	1
VISIT NUMBER	1	0	549	212.9	1
HABITAT	3	0	646	212.9	1
NATIVE:HUNTED	2	3.43	644	209.5	0.018

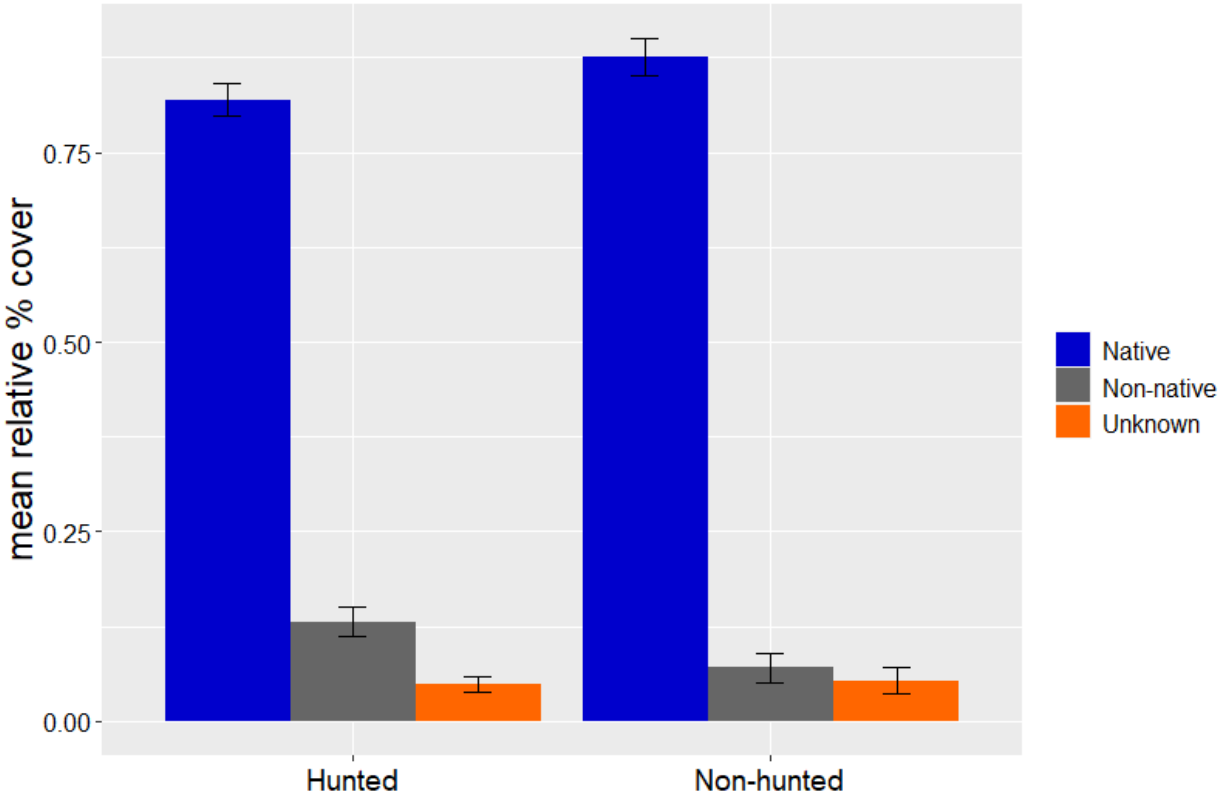

Figure 6) Mean relative percentage cover per plot by native status across hunted and non-hunted sites

DISCUSSION

I hypothesized that richness and abundance of native species would be lower along with higher richness and abundance of non-native species in sites where hunting was prohibited. This prediction was formed because existing research links an overabundance of WTD to the success of some invasive species (Coté et al. 2004; Eschtruth & Battles 2009; Knight et al. 2009; Andrea Relva et al. 2010), and hunting is the main mechanism through which WTD populations are managed (Frost et al. 1997; Woolf & Roseberry 1998; Pennsylvania Game Commission 2019). This study was conducted under the assumption that WTD populations differed between the State Game Lands and the private Haskins site because of the differences in hunted and land ownership status (Small et al. 1991; Frost et al. 1997). My results indicated that hunting may have an indirect effect on native and non-native richness; sites where WTD hunting was permitted exhibited greater native species richness, although these hunted sites also had a greater abundance of non-native species compared to sites where hunting was prohibited.

I believe the higher richness of non-native plants at hunted sites is related to differences in WTD densities between hunted and non-hunted sites. WTD overabundance can cause significant changes in competitive interactions between plants and cause disturbances in vegetative communities (Côté et al. 2004). These factors have been associated with alterations of plant community structure, possibly facilitating the success of non-native species (Côté et al. 2004; Jauni et al. 2015). The higher native species richness at hunted sites may have been the result of lower WTD density, as hunters can impact WTD populations by both removing deer from the population through harvesting and altering WTD movement patterns towards areas of lower hunter density (Frost et al. 1997; Diefenbach et al. 2005; Little et al. 2016). Hunter densities are highest in public lands near roads, so WTD populations may have been significantly

lower in the surveyed area than areas farther from the road, potentially reducing the potential WTD influence on vegetative communities where plant surveying data was collected (Diefenbach et al. 2005).

As previously mentioned, WTD grazing preferences can determine the structure of vegetative communities (Coté et al. 2004). An avenue of research not included in this study that would be a valuable topic in future research was the grouping of identified plant species into categories (i.e. forbs, graminoids, woody species, etc.). This data could have been easily recorded during plant identification, and it could have served as another metric for measuring potential WTD effects on plant community composition. Existing research indicates that high levels of WTD impact tends to be associated with an increase in the abundance of ferns and graminoids concurrent with declines in woody species and forbs (Côté et al. 2004; Eschtruth & Battles 2009). In particular, one of the dominant species in the hardwood Haskins 1 site was the hay-scented fern (*Dennstaedtia punctilobula*), a species that is not grazed by WTD that can become a "native invasive" at high abundance, inhibiting sapling growth and forest regeneration (De la Cretaz et al. 1999). The browsing of other understory plants by WTD is a dominant factor leading to the overabundance of this species.

There were several shortcomings of this study design that need to be addressed. In this study, only the hardwood transects, Haskins 1 and State Game Lands 2 were analogous habitats. The variation in habitats makes drawing meaningful comparisons between hunted and non-hunted sites more difficult and limited the statistical tests I was able to conduct. Some of the lack of consistency between transects habitats was the result of limitations of my surveying sites; my non-hunted site had neither coniferous nor riparian area. I encourage future studies of additional

habitat types in additional sites representing hunted and non-hunted lands to increase the rigor of these findings.

Another factor that may have influenced the results of this study was the differences in the habitats surrounding the two surveyed locations. While SLG 64 consisted of 32.6 km^2 of predominantly heavily forested area (PA Game Commission 2014) surrounded by still more forested area, the Haskins site was largely surrounded by agricultural land in which corn (*Zea mays* subsp. *Mays)* and green beans (*Phaseolus vulgaris)* were the predominant species grown. WTD will preferentially consume corn over most foods found throughout their habitats, which is why the use of corn in WTD baiting or supplemental feeding is so common (Newbolt 2020). This readily available and highly palatable food source around the Haskins site may have altered the grazing patterns of the WTD in this area, potentially leading them to preferentially graze crops over the species found in the forested Haskins area, and reducing the effects of potential WTD overabundance on the plants in the surveyed area.

The human influences varied significantly between the Haskins site and SLG 64. The State Game Lands are public lands available for recreational use, namely hunting, fishing, and trapping during their appropriate seasons. The fenced-in, privately owned Haskins property experiences no human effects outside of those that may be caused by the property owners. The surveyed portions of SLG 64 were also easily accessible from Phoenix Run Road. This road is not very frequently travelled but experiences a great deal more traffic than the private drive through the Haskins property. As humans enter and travel through the State Game Lands, they may introduce and spread propagules of non-native species (Hulme et al. 2008). The direct evidence of this human influence at SLG 64 was occasionally visible in the form of garbage during transect visits, particularly in the riparian area along Phoenix Run where anglers would

likely have been present. The potential for non-native plant species introduction into the Haskins property is much lower than the State Game Lands simply because there are fewer potential human vectors. The lack of human influence at the Haskins property may have been a factor leading to the lower non-native species abundance at this site compared to SLG 64.

The higher native species richness at hunted sites may indicate possible positive effects of WTD hunting on the composition of plant communities. This finding supports the merit of promoting recreational WTD hunting as an indirect method of managing non-native vegetative species. Hunting land availability and the overall number of hunters are both declining throughout the United States due to a variety of factors (Winkler & Warnke 2013) at the same time that non-native species are continuing to spread through plant communities (Vellend et al. 2017). Efforts to increase the awareness and participation in recreational WTD hunting may be a strategy to slow down or halt the spread of non-native species in ecosystems inhabited by WTD by supporting native species diversity (Tilman 1997; Levine et al. 2004). This is beneficial from an ecosystem management standpoint, because the simplest and most cost-effective strategy for managing non-native or invasive species is preventing their establishment and spread, due to the difficulty of returning an ecosystem to pre-invasion conditions once introduced species have become problematic (Myers et al. 2000; Zavaleta et al. 2001).

Aims to reduce WTD populations to maintain plant community structure and function need to be balanced against public perceptions about WTD and the opinions of hunters themselves. Hunters want high enough WTD populations to ensure WTD harvesting success. The chance of harvest success is a major factor determining the effort and time hunters are willing to put into WTD hunting (Stedman et al. 2008). In addition, the public may have differing opinions than ecosystem managers about ideal WTD populations. In Pennsylvania

WMU 3A, WTD populations have increased in recent years, causing concerns that impacts on the area's vegetation will soon increase above acceptable levels. However, this concern is at odds with residents of this WMU who reported that they believe WTD populations in the area are too low (PA Game Commission 2019). Bridging the gap between public beliefs about adequate WTD populations and the appropriate number of WTD for vegetative community health is essential to ensure the continued support and success of WTD population management efforts.

Finally, there may also be value in increasing educational efforts to improve the understanding of private landowners about the negative effects of WTD overabundance. Private landowners who chose to post their land against hunters tend to be younger, more educated, and more likely to be from urban backgrounds compared to landowners who chose not to post (Jagnow et al. 2006). Landowners from urban backgrounds are more likely to have negative feelings towards hunting activities as a whole than their counterparts from rural backgrounds (Jagnow et al. 2006), presumably due to a lack of understanding about the importance of hunting for species with no natural predators. Providing educational opportunities to these landowners about the benefits of hunting WTD may change their perception about hunting and cause them to alter their posting decision.

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