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Evaluating the Effects of Morrow's Honeysuckle Control on Vertebrate and Vegetation Assemblages, and Small Mammal Foraging Ecology at Fort Necessity National Battlefield

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Evaluating the Effects of Morrow's Honeysuckle Control on Vertebrate and Vegetation Assemblages, and Small Mammal Foraging Ecology at Fort Necessity National Battlefield

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ABSTRACT

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Exotic, Japanese bush honeysuckles (*Lonicera* spp.; Caprifoliaceae) are tied to a variety of impacts on wildlife and ecosystems. Morrow's honeysuckle (*Lonicera morrowii*) has become a persistent invader in eastern North America. We organized a restoration initiative at Fort Necessity National Battlefield (FONE), Pennsylvania, USA from 2004 – 2010. Concurrently, we studied the consumption of Morrow's honeysuckle fruits by small mammals from October – November 2009 and July – August 2010, and determined habitat variables that affected visitation rate to foraging stations. Areas of FONE were invaded by Morrow's honeysuckle after the land had been cleared for agriculture, and routine mowing ceased in the mid-1980s. Our restoration goals were to control honeysuckle and restore native vegetation with a plan to promote both early-successional habitat and mimic the historical conditions from the mid-1700s. Treatment and reference sites were established, and treatment sites received a combination of yearly mowing and broad-spectrum herbicides from October 2006 – August 2010. The vegetation and vertebrate communities were monitored pre-removal from 2004 – 2006, and throughout the restoration from 2007 – 2010.

Our control techniques were highly effective at reducing the presence of Morrow's honeysuckle in the treatment area. The percent cover of Morrow's honeysuckle declined dramatically from 2005 – 2010. No direct, short-term adverse impacts on the monitored vegetation and vertebrate communities occurred. In fact, most species varied as a function of time over the study, rather than because of the presence or removal of Morrow's honeysuckle. We found that small mammals were better indicators of changes in the vegetation community than were songbirds. Competitive interactions between small mammals appeared to produce an indirect negative effect of restoration. Overall, our restoration efforts were successful at controlling Morrow's honeysuckle with minimal impact on the monitored communities.

When compared to native soft mast, Morrow's honeysuckle was generally less consumed by white-footed mice (*P. leucopus*). Honeysuckle fruits had significantly less protein (0.66%) and lipids (0.67%) than all natives. Morrow's honeysuckle had one of the highest moisture contents, which was important in the use of its fruits. Despite high moisture content, Morrow's fruits are still lacking key nutrition, likely leading to its overall low consumption. Total energy always distinguished the highest selected fruits: black cherry (*P. serotina*) (0.45 kcal), and common dewberry (*R. flagellaris*) (0.36). Morrow's honeysuckle creates monocultures that exclude natives, which are the more nutritious and utilized food items. This may force small mammals to forage longer, or travel further distances with the possibility of increasing their risk of predation. This result corresponds to our finding that high visitation rate to foraging stations was negatively associated with shrub coverage in fields. The most common shrub in the field was Morrow's honeysuckle, found to be the closest shrub to 85% of stations. Since honeysuckle is less nutritious and a lesser-used food item, animals would lose energetic profit if they continued to feed in areas of honeysuckle, and it likely explains why they do not often forage in dense honeysuckle areas.

To Donald Edward Rose,
for being the one who gave me my first set of Wildlife Fact File cards,
and for all the penny-candy I could ever eat
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CHAPTER 1

REVIEW ARTICLE

Introduction, Justification for Removal of Morrow's Honeysuckle (*Lonicera morrowii*), and Previous Research Results from Fort Necessity National Battlefield, Farmington, Pennsylvania

Charneé Lee Rose[1,2]

Introduction

The economic damages associated with invasive species and their control was estimated to be about $138 billion per year in the U.S. (Pimentel 2002). Global travel and international trade have become pathways to accelerated invasion (Mack & Erneberg 2002), increasing monetary losses. In the biological context, ~40% of endangered species in the United States are at risk due to competition or predation by non-indigenous species (Wilcove et al. 1998). Although both animal and plant exotics contribute to the damages described above, exotic plants alone can spread prolifically (Manchester & Bullock 2000), deteriorate ecosystem services (Gordon 1998; Ehrenfield 2003), and negatively impact global economies (Naylor 2000; Zavaleta 2000).

Northeastern and mid-western portions of the United States have been invaded by aggressive Eurasian bush honeysuckles. Exotic honeysuckles were introduced to the U.S. through the ornamental industry in the mid-1700s, including Amur honeysuckle (*Lonicera maackii*), Morrow's honeysuckle (*Lonicera morrowii*), and Tartarian honeysuckle (*Lonicera tatarica*) (Rehder 1940; Luken & Thieret 1995). Bush honeysuckles have a strong tolerance for a

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This chapter written in the style of Restoration Ecology

broad range of soil moisture, soil types, light regimes and cover types. They grow in riparian areas, early successional habitat (McClain & Anderson 1990), forest interiors (Woods 1993), edges, and corridors. The shrubs also occupy areas of disturbed land including roadsides, railroads (Barnes & Cottam 1974), and abandoned agricultural land (Hauser 1966). Humans furthered the range of these shrubs by using them in mine reclamations (Wade 1985), shelterbelts (Herman & Davidson 1997), and for wildlife resources (Mulvihill et al. 1992; VanDruff et al. 1996). A variety of honeysuckle species were widely distributed across the northeastern United States by the early 1900s (Rehder 1903).

Large areas of Fort Necessity National Battlefield (FONE) were invaded by one of these invasive bush honeysuckles, Morrow's honeysuckle (*Lonicera morrowii*) (National Park Service 1991). According to the General Management Plan for FONE, the park will be managed to: 1) prevent damage by exotic species, 2) protect rare, threatened, or endangered species, and 3) reestablish historical vegetative conditions (National Park Service 1991). An additional plan was developed to control Morrow's honeysuckle by restoring a portion of the study site to a mature hardwood forest, and to manage the remaining area as early successional habitat for a declining game bird, the American woodcock (*Scolopax minor*). To determine the impacts of both Morrow's honeysuckle cover and control procedures we performed pre-treatment and post-treatment surveys of American woodcock, songbird, small mammal, herpetofauna, and vegetation communities. Also, we conducted a secondary study examining the effects of Morrow's honeysuckle on small mammal foraging ecology, while concurrently examining habitat affinities in 2009 and 2010.

Justification

Restoration

The support for the restoration research conducted at Fort Necessity National Battlefield comes from the need to study invasive species to develop methods for both efficient removal and subsequent reestablishment of natural ecosystems (Hartman & McCarthy 2004). Understanding trends in the invasion process, as well as the impact of control and management activities, is necessary to manage exotics (Hunter & Mattice 2002). Studies measuring the effects of exotic plant species on native communities are lacking (Tickner et al. 2001; Hejda et al. 2009). Previous to the implementation of this research project, no studies were found that examined the effect of Morrow's honeysuckle removal on native vegetation. Likewise, there is no comprehensive project known to assess the response of vertebrate populations (songbirds, small mammals, amphibians, and reptiles) to the removal of invasive Morrow's honeysuckle. Due to the aggressiveness of Morrow's honeysuckle, control is difficult; however, the removal of this exotic at local sites is a critical step in restoring the habitat and developing management practices (Hartman & McCarthy 2004). Small-scale removal of Morrow's honeysuckle is important in order to continue removal on a "site-by-site basis" across the landscape (Wiens et al. 1993; Hartman & McCarthy 2004).

Additionally, we placed special emphasis on American woodcock, as it is a popular game bird with declining populations in portions of the United States (i.e., eastern and Midwestern). This species uses wetlands and early successional areas as nesting and foraging habitat; however, both are currently at risk due to habitat destruction and afforestation (Dwyer et al. 1983; Sauer & Bortner 1991). Long-term declines (1967-2010) have taken place in woodcock populations across the eastern and central United States (Cooper & Parker 2010). Since populations of this popular game bird are known to be declining (Brown et al. 2004; Kelley 2004; Cooper & Parker

2010), it has been listed by the U.S. Fish and Wildlife Service as a Game Bird Below Desired Condition (U.S. Fish and Wildlife Service 2004). Also, it is listed on the Audubon Watchlist as a species that is in slow decline and of national conservation concern (National Audubon Society 2010). Due to this species' importance, as both a consumptive and non-consumptive species, it was critical to assess the potential habitat quality for woodock at Fort Necessity to create an area of sustainable habitat.

Small Mammal Foraging Ecology

The research conducted by Edalgo et al. (2009) highlights the need for additional studies to be conducted that further investigate how Morrow's honeysuckle alters small mammal ecology. This paper expressed the need for a study that determines if white-footed mice, as well as other small mammals, readily consume the fruit and seeds of bush honeysuckle. Additionally, a number of other studies suggest that exotic plants, especially *Lonicera* spp., have the potential to alter small mammal behaviors (Witmer 1996; Williams 1999). Bush honeysuckles outcompete and greatly reduce native vegetation (Batcher & Stiles 2000); therefore, it is likely that bush honeysuckles affect the food available to the small mammal species that serve vital roles in the ecosystem (Bellows et al. 2001).

As the spread of bush honeysuckle continues throughout the United States, it is important to study the extent to which small mammals incorporate *Lonicera* into their diets, and determine if they influence the population dynamics of exotic bush honeysuckle. This information is necessary as small mammal seed consumption has been shown to influence the spread of native plant species (Ostfeld et al. 1997); if the same is true for exotics, small mammals could experience a decrease in species diversity (Horncastle et al. 2004).

For foraging studies to successfully examine preference, it is necessary to understand the habitat characteristics and microsites that small mammals show fidelity towards. Not only is it important to determine these characteristics, but it is equally as valuable to understand how these change over various habitats and seasons. Knowledge of selected habitat characteristics and seasonal variability can increase a study's likelihood of detecting species and capturing small mammal food preference, while increasing the statistical power of their analyses.

Study Description

Study Site

We conducted both the primary (restoration response) and secondary (foraging behavior; habitat modeling) studies at Fort Necessity National Battlefield. The National Park Service established Fort Necessity National Battlefield, located in Fayette County in southwestern Pennsylvania, U.S.A, in 1933 (Fig. 1) (39º48'43" N, 84º 41'50" W). The historical park is approximately 390 ha in size (Fig. 2). Elevations throughout the park range from 535 – 710 m. The average annual temperature at Fort Necessity is 9º C, the mean winter temperature is -3º C, and the mean summer temperature is 22º C. The average precipitation level is 119 cm (National Park Service 1991). Brinkerton and Armagh silt loams characterize the soils; they are moderate to well drained, medium-textured, and moderately deep (Kopas 1973).

The restoration project sites are located west of an historical replication of Fort Necessity. The original was built by George Washington and his troops in 1754 at the onset of the French and Indian War (Figs. 3 & 4). During the mid-1700s, the hillside was predominantly an oak-hardwood forest. Core pollen samples taken from the site showed that oaks (*Quercus* spp.), red maple (*Acer rubrum*), hickories (*Carya* spp.), American beech (*Fagus grandifolia*) and birch (*Betula* spp.) comprised the forest (Kelso 1994). After the war, the land was cleared for pasture use prior to the establishment of the park in 1933 (National Park Service 1991). Until the mid-1980s, the pasture was maintained by mowing. When the agricultural practices ended, the land was passively managed and allowed to follow natural succession (Love & Anderson 2009). However, a dense cover of Morrow's honeysuckle (Fig. 5) (Love & Anderson 2009) established and dominated the study area until restoration procedures, which involved a combination of mowing and herbicide (Figs. 6 & 7), started in 2007.

The foraging ecology study sites are widely distributed across cover types throughout Fort Necessity that are inhabited by Morrow's honeysuckle (Fig. 8). We chose study locations in three available types: field, edge, and forested areas. Although forested locations in the park contain less Morrow's honeysuckle, we believed it was important to include this cover type in the study as bush honeysuckles are known to be shade tolerant and hybrids are often found in forest interiors (Woods 1993).

Objectives

The overall objectives of this project were to:

1) assess the best time (according to shrub's phenological stage) to apply herbicide or mechanically remove Morrow's honeysuckle;
2) determine the most effective and cost-efficient method to control Morrow's honeysuckle;
3) determine the species composition of shrub and herbaceous communities prior to, and following restoration procedures;
4) determine the relative abundance and location of American woodcock (*Scolopax minor*) prior to, and following restoration procedures;
5) assess the relative abundance and location of earthworms, the woodcock's major prey sources, within the study area;
6) assess the effects of Morrow's honeysuckle on the diversity and biomass of insects;
7) determine the relative abundance and richness of amphibians and reptiles within the study area prior to, and following restoration procedures;
8) determine changes in relative abundance and richness of songbirds in response to management activities;

9) assess the effects of Morrow's honeysuckle on songbird fitness level (fat class and body mass index), and nest success;

10) determine the effects of Morrow's honeysuckle on songbird territory size and density;

11) determine the relative abundance and richness of small mammals within the study area prior to and following restoration procedures;

12) assess the effectiveness of prebaiting Sherman traps within the study area dominated by Morrow's honeysuckle;

13) assess the effects of Morrow's honeysuckle on microhabitat selection of small mammals;

14) assess the species of small mammals that actively consume Morrow's honeysuckle fruits;

15) investigate if small mammals use Morrow's honeysuckle in their diet in the same quantities as native soft mast fruits;

16) determine if the magnitude of Morrow's fruit consumption remains consistent across cover types, and throughout seasonal changes of Morrow's fruiting period;

17) assess the habitat characteristics that contribute to high small mammal visitation rate to foraging stations across cover types, seasons, and spatial scales; and

18) develop a set of management options for the removal of Morrow's honeysuckle and the consequences each of these options may have on flora and fauna within the study area.

My research focuses on objectives: 3, 4, 7, 8, 11, 14, 15, 16, and 17. Based on these objectives, and subsequent literature reviews, the following hypotheses were tested:

3) determine the species composition of shrub and herbaceous communities prior to, and following restoration procedures;

   H_0: There is no difference in species composition of shrub and herbaceous species in the study plots prior to, and following restoration.
   H_a: There is a difference in composition of shrub and herbaceous species, with restoration plots showing higher species diversity.

4) determine the relative abundance and location of American woodcock (*Scolopax minor*) prior to, and following restoration techniques;

   H_0: American woodcock will use the study area indiscriminately prior to and following restoration.
   H_a: American woodcock abundance will be greater in the reclaimed plots.

7) determine the relative abundance and richness of herpetofauna prior to and following restoration;

H_0: Herpetofauna species will use the study area indiscriminately.
H_a: Herpetofauna abundance/richness will be greater in the reclaimed plots.

8) determine the relative abundance and richness of songbirds within the study area prior to and following management activities;

H_0: Songbird species will use the study area indiscriminately.
H_a: Songbird abundance/richness will be greater in the reclaimed plots.

11) determine the relative abundance and richness of small mammals prior to and following restoration procedures;

H_0: Small mammal species will use the study area indiscriminately.
H_a: Small mammal abundance/richness will be greater in the reclaimed plots.

14) assess the species of small mammals that actively consume Morrow's honeysuckle fruits;

H_0: All granivorous species present consume honeysuckle fruits.
H_a: Not all granivorous species present consume honeysuckle fruits.

15) investigate if small mammals use Morrow's honeysuckle in their diet in the same quantities as native soft mast fruits;

H_0: Small mammals utilize honeysuckle and natives indiscriminately in their diets.
H_a: Small mammals show distinct foraging preferences between Morrow's and native fruits, with native soft mast fruits showing higher consumption rates.

16) determine if the magnitude of Morrow's fruit consumption remains consistent across cover types, and throughout seasonal changes of Morrow's fruiting period;

H_0: Small mammal consumption of honeysuckle fruit remains consistent across cover types and the rate of consumption does not change throughout tested seasons.
H_a: Small mammals consume honeysuckle fruits differently depending on cover type and season tested, with foraging pressures highest in edge plots and the July study phase.

17) assess the habitat characteristics that contribute to high small mammal visitation rate to stations across cover types and between two seasons;

H_0: Small mammal visitation rate to study boxes is independent of environmental variables, and there is no difference in visitation rates.

H_a: Small mammal visitation rate to study boxes is increased with % shrub, % overhead canopy, % log and increased height of vertical vegetation depending on cover type observed and season.

Previous Research

Previous research conducted at Fort Necessity, since 2004, have provided answers to hypotheses derived from at least seven of the eighteen stated objectives above.

Total Non-structural Carbohydrates (TNC)

In 2004 and 2005, Love and Anderson (2009) conducted a field study that determined when the total non-structural carbohydrates of Morrow's honeysuckle were at their lowest. This study found that TNC levels in the shrub roots were lowest in May, after leaf and flower formation. Conversely, the TNC levels were at their highest in the roots during October. Love and Anderson (2009) concluded that managers looking to control populations of Morrow's honeysuckle should time their efforts to coincide with when the root TNC levels are at their lowest, to maximize their control efforts.

Effective Methods for Removing Morrow's Honeysuckle

In 2004 and 2005, Love and Anderson (2009) conducted a field study that tested four control methods for invasive Morrow's Honeysuckle. The four control methods tested included cut, mechanical removal, stump application of glyphosate, and foliar application of glyphosate. The study found that foliar application of herbicide and mechanical removal of shrubs was the most effective methods for controlling and reducing Morrow's honeysuckle.

Effects of Morrow's Honeysuckle on Invertebrates

From July 2004 to August 2005, Love (2006) assessed the effect Morrow's honeysuckle had on invertebrate biomass at Fort Necessity National Battlefield. This study used a modified leaf vacuum to sample invertebrates on both single and dense thickets of Morrow's honeysuckle shrubs and single Southern arrowwood (*Viburnum dentatum*) shrubs. This study found that the native shrub contained lower overall invertebrate biomass than either a single bush or thicket of honeysuckle. However, the native contained 5 times more larval leaf chewer biomass than that of the dense thickets, and 1.5 times more than that found on a single honeysuckle bush. It was concluded that lower levels of larval leaf chewers could negatively affect songbirds by increasing time spent foraging (Sample et al. 1993).

Effects of Morrow's Honeysuckle on Exotic Earthworms

The effects of Morrow's honeysuckle on earthworm abundance was studied at Fort Necessity National Battlefield, in 2004 and 2005 (Edalgo & Anderson 2009). This study found that the four species of earthworm at the site were all exotic. Although these species were found in soils underneath Morrows honeysuckle, tulip poplar (*Liriodendron tulipifera*) and black locust (*Robinia pseudoacacia*) supported higher densities (Edalgo & Anderson 2009).

Effects of Morrow's Honeysuckle on Prebaiting

The effectiveness of prebaiting small mammal Sherman traps in an invasive shrub community, Morrow's honeysuckle, was tested from 2004 – 2005 at Fort Necessity. Edalgo and Anderson (2007) found that prebaiting did not improve trapping success in Morrow's honeysuckle dominated landscapes. Based on these results, prebaiting was considered unnecessarily time consuming and costly since no difference was seen in trapping success.

Effects of Morrow's Honeysuckle on Small Mammal Microhabitat Selection

Sherman trapping and fluorescent powder tracking was used to examine the microhabitat selected by white-footed mice (*Peromyscus leucopus*) at Fort Necessity, in 2004 and 2005. This study found that mice selected paths with low exotic herbaceous vegetation, as well as paths with greater shrub (including Morrow's honeysuckle) and tree cover (Edalgo et al. 2009). Based on this research they recommend managing for shrub and tree cover, and controlling exotic species.

Literature Review

Restoration Ecology

Restoration initiatives have been practiced as a means of offsetting the deterioration of ecological systems caused by human population growth (Chew 2001). There is a variety of definitions for restoration ecology and many of them involve reversing "negative ecosystem developments" through human intervention (Van Andel & Aronson 2006). Halle and Fattorini (2004) provide the following definition: the process of assisting the recovery and management of ecological integrity. Their definition also includes the ideas of maintaining: historical ecological processes, biodiversity, and cultural practices. Additionally, restoration involves tracking the population trends of fauna and flora under their current environmental conditions (Morrison 2002).

There are a variety of reasons for restoring a degraded site: to protect ecosystem goods and services, to preserve native biodiversity, to promote economic productivity, and to reduce fragmented landscapes (Van Andel & Aronson 2006). Ecosystem services such as carbon sequestration, recreational experiences, and pollution filtration can be improved through restoration initiatives (Dabbert et al. 1998; Van Andel & Aronson 2006).

When developing a restoration project, levels of community function have to be considered in the planning process (Falk et al. 1996). The first level deals with dispersal and colonization dynamics that help determine the species composition at the study site. The second level pertains to individual environmental and habitat characteristics that further filter the species that establish. The third level deals with the interactions of biotic communities such as competition, predation, and mutualism. All three levels of community function must be considered for restoration to be successful (Hobbs 2002; Falk et al. 1996).

It is important to realize that restoration usually occurs on a small scale, and is often considered impossible at larger landscape levels because of lack of public support and land-use conflicts (Van Andel & Aronson 2006). Due to such difficulties a variety of approaches are used to narrow the scope of restoration projects. Some restoration initiatives focus on keystone species (Jones & Lawton 1995; Stone 1995; Paine 1996; Palmer et al. 1997), while others focus on endangered species (Palmer et al. 1997; Young 2000). However, projects often focus on species assemblages as good indicators of ecosystem health (Palmer et al. 1997; Young 2000; Van Andel & Aronson 2006).

In textbook terms, full restoration means that the reclaimed area is resilient; that is, it can recover from stress (Walker et al. 2000). In many studies, restoration is never finished and some level of maintenance is necessary to maintain the treated area (Falk et al. 1996; Baron et al. 2002; Suding et al. 2004). Nevertheless, restoration is often considered accomplished when: 1) the reclaimed area contains the same species as the reference sites, 2) the restored area is largely dominated by native species, 3) the restored ecosystem is capable of reproducing populations for continued stability, 4) the restored ecosystem is integrated into the larger landscape, 5) threats to the treated area have been reduced or eliminated, and 6) the treated area is equally as self-sustaining as the reference sites (Van Andel & Aronson 2006).

Characteristics of Exotic Plants

Changes in land-use, climate, and concentrations of carbon dioxide in the atmosphere can positively affect invasive plants (Vitousek et al. 1996; Dukes & Mooney 1999; Mooney & Hobbs 2000; Simberloff 2000; Dukes 2002; Kriticos et al. 2003; Weltzin et al. 2003). Nonindigenous plant species disrupt ecosystems, compete with native species and cause economic losses (DiTomaso 2000; Levine et al. 2003; Dukes & Mooney 2004; D'Antonio &

Hobbie 2005). Management of exotics is a complex problem as “novel” or “emergent” ecosystems become more common and difficult to restore (Hobbs et al. 2006).

Invasive plants have a variety of traits that allow them to successfully establish in an area, and many perform better in invaded ranges than in their native ranges (Hinz & Schwarzlaender 2004). Invasive plants often prolifically grow in a variety of habitats. Many exotic plants have deep root systems, and produce large quantities of flowers and seeds. Likewise, they tend to have staggered germination and seeds that persist in the seed bank for extended periods. Lastly, some exotic plants exhibit allelopathy and are resistant to grazing.

The establishment of invasive plant species in an area not only depends on the attributes of the invasive, but it is also highly dependent on the characteristics of the landscape. The vulnerability of a site, in terms of open growing space, can be used to predict success of an invader (Radosevich et al. 2003). It is suggested that late-successional vegetation communities are less vulnerable due to the presence of canopy cover (Woods 1993; Radosevich et al. 2003). While the above information can be applied generally when predicting landscape spread, there are known instances when plant invasives have been able to establish in forested communities (Trisel & Gorchov 1994).

Four Stages of Invasion

To form an understanding of how invasive plant species become established, a four-stage approach to invasion has been developed by Theoharides and Dukes (2007). The four stages of invasion include: transport, colonization, establishment and landscape spread.

Transport of species is happening faster than ever before due to global travel and trade (Mack et al. 2000; Reichard & White 2001; Le Maitre et al. 2004; Theoharides & Dukes 2007). Likewise, exotic plants have been introduced for aesthetic purposes since the 19th century and this is a practice that continues today (Mack & Lonsdale 2001). These species are given a significant advantage because they are intentionally cultivated, climate-matched to determine where they will grow best, and are subject to less environmental stochasticity (Mack 2000; Mack & Lonsdale 2001; Theoharides & Dukes 2007). Invasive species, introduced as ornamentals, that have escaped cultivation in the United States include: pampas grass (*Cortaderia jubata),* Japanese knotweed *(Fallopia japonica),* and Japanese honeysuckle *(Lonicera japonica)* (Theoharides & Dukes 2007). Since research has shown that humans are the primary transporters of exotic plants (Pauchard & Shea 2006), understanding and anticipating patterns of trade and travel may allow investigators to predict the species that might become invasive in the United States (Theoharides & Dukes 2007).

Colonization of exotic plants is more difficult than often imagined. Due to initial small population sizes, these species must overcome lack of genetic variability and both environmental and demographic stochasticity (Sakai et al. 2001; Theoharides & Dukes 2007). In fact, it has been noted that only 10% of exotics colonize non-native ranges (Williamson & Fitter 1996). Climate is the factor that sets limits on plant distribution and productivity (Sakai et al. 2001); therefore, invasives that are introduced to a variety of landscapes have a better chance of colonization (Lockwood et al. 2005). It is with repeated introductions that the initial small populations obtain greater genetic variability and the capacity to adapt to new environmental conditions (Theoharides & Dukes 2007).

Establishment of invasive plant species requires populations that are self-sustaining (Theoharides & Dukes 2007). Often exotics have traits that help to achieve these goals, including: secondary chemical compounds that deter grazing, allelopathy, and fast growth and reproduction rates (Dietz & Edwards 2006; Theoharides & Dukes 2007). Both the enemy release and evolution of increased competitive ability hypotheses suggest that invasives benefit from transport outside their native range because they lack natural enemies and can devote energy to growth rather than defense (Blossey & Notzold 1995; Carpenter & Cappuccino 2005).

Landscape spread refers to dispersal within a given area over long time periods, where invasive plants "exist as interacting groups at different stages of colonization and establishment" (Theoharides & Dukes 2007). The rate of spread by these species is influenced by landscape heterogeneity and fragmentation of habitat patches. Local-scale population dynamics, between invasive and native plants, are determined by resource availability and community heterogeneity (Davies et al. 2005; Melbourne et al. 2007; Theoharides & Dukes 2007). Like native species, the connectivity of suitable habitat patches heavily influences spread and population dynamics (Knight & Reich 2005; Ohlemuller et al. 2006). Research has shown that large patches promote native species, while smaller patches with increased edge tend to favor invasive spread (Timmens & Williams 1991; Parendes & Jones 2000; Harrison et al. 2001; Ohlemuller et al. 2006; Theoharides & Dukes 2007).

Exotic Bush Honeysuckles

Bush honeysuckles are in the family Caprifoliaceae, also known as the honeysuckle family. There are 16 genera in Caprifoliaceae, containing 365 species, including flowering nutmeg (*Leycesteria formosa*), old-fashioned weigela (*Weigela florida),* and Japanese Honeysuckle (*Lonicera japonica*) (Hickey & King 1997). Many of these species are valued as ornamental flowering shrubs. General characteristics of the family include: shrub or small tree, simple or pinnately compound opposite leaves, small or absent stipules, capsule, berry or drupe fruit, and bisexual flowers (Swanson 1994; Hickey & King 1997). Specifically, bush honeysuckles are deciduous shrubs that grow upright to heights of 2 – 6 m. The shrubs are multi-stemmed, oppositely branched, and produce large quantities of pink, white or yellow flowers, and later produce red or yellow soft mast (Gleason & Cronquist 1991). Morrow's honeysuckle has oblong green leaves that have a pubescent underside. Morrow's has 3-6 cm long leaves, 1-2 cm white flowers, and red fruits (Figs. 9 & 10) (Petrides 1972).

There are roughly 180 honeysuckle species worldwide, and only 20 of them are native to the United States. Many of the established honeysuckle species in the United States were introduced as ornamentals. Tartarian honeysuckle (*Lonicera tatarica*) was introduced in 1752 (Rehder 1940), while Morrow's honeysuckle and Amur honeysuckle (*Lonicera maackii*) were not cultivated until the late 1800s (Luken & Thieret 1995). When flowering, exotic bush honeysuckles can be distinguished from all native bush honeysuckles except swamp fly-honeysuckle (*Lonicera oblongifolia*) by their hirsute styles. Swamp-fly honeysuckle can be distinguished from other invasive species by examining its hairless leaves and solid white pith (Petrides 1972). Also, bush honeysuckles generally leaf-out earlier and retain their leaves longer than native species (Trisel & Gorchov 1994).

In field identification, there is considerable difficulty in distinguishing Tartarian honeysuckle, Morrow's honeysuckle, and their hybrid *Lonicera x bella*. It is possible that *L. x bella* has been misidentified as one of the parent species due to consistency in morphological characteristics (Barnes & Cottam 1974; Wyman 1977). A parent species, Morrow's honeysuckle, has leaves that are elliptic to oblong gray-green in color, the lower surface is pubescent, and 3 – 6 cm. The flowers are white to yellow, and pubescent with densely hairy peduncles. The shrub ranges up to 2 m in height with fruits that are red (Rehder 1940; Wyman 1977; Gleason & Cronquist 1991). Dissimilarly, Tartarian honeysuckle has leaves that are ovate, white to pink flowers, and longer peduncles. The height can be up to 6 m, with fruits that are red or yellow (Gleason & Cronquist 1991). The hybrid, *L. x bella*, has slightly pubescent leaves, flowers that are pink fading to yellow, with few hairs on the peduncles. The height ranges up to 6 m and has fruits that are red or yellow (Gleason & Cronquist 1991).

Morrow's honeysuckle is native to Japan and was brought to the United States by agriculturist Dr. James Morrow, where botanist Asa Gray described the species (Barnes & Cottam 1974). Morrow's escaped cultivation and is now established in many portions of the United States and Canada. Batcher and Stiles (2000) reported the establishment of Morrow's honeysuckle in the following areas: Arkansas, Colorado, Connecticut, District of Columbia, Illinois, Iowa, Kentucky, Maine, Maryland, North Carolina, Ohio, Ontario, Pennsylvania, Quebec, Rhode Island, Saskatchewan, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming.

Morrow's honeysuckle, and other species of exotic bush honeysuckles, occupy a variety of cover types across the United States. Bush honeysuckle is often found in riparian areas, early successional habitat (McClain & Anderson 1990), forest interiors (Woods 1993), edges, and corridors. These shrubs also occupy areas of disturbed land including roadsides, railroads (Barnes & Cottam 1974), and abandoned agricultural land (Hauser 1966).

Honeysuckle can tolerate a wide range of soil types, moisture levels, and light regimes (Barnes & Cottam 1974; Dirr 1990; Woods 1993). These shrubs are highly competitive when growing in shaded areas (Barnes & Cottam 1974) and retain their leaves longer than other deciduous plants (Woods 1993). Anthropogenic land disturbance and urban conditions have been found to predict the occurrence of honeysuckle species (Borgmann & Rodewald 2004).

Reproduction in bush honeysuckles is highly dependent on seed dispersal (Converse 1985); although, commercial growers have employed greenwood and hardwood cutting techniques. Amur honeysuckle seeds ripen September through November; Morrow's honeysuckle and Tartarian seeds ripen June through August (Schopmeyer 1974). Bush honeysuckles produce fruits in large quantities. A study in southwestern Ohio estimated that Amur honeysuckle and *L. xylosteum* had over 400 million fruits/ha (Ingold & Craycraft 1983). The fruit of Morrow's honeysuckle is high in sugar (76%), but low in vital nutrients such as protein (0.6%) and lipids (<2%) (Witmer & Van Soest 1998). Also, the fruits are able to persist in the environment for longer periods of time due to the high quantities of secondary compounds, such as tannins, lignins, and terpenoids, making them less palatable and resistant to grazers. (Stiles 1980; Cippollini & Levey 1997).

Wildlife and Exotic Bush Honeysuckles

Honeysuckle has been cited as an important year-round browse plant for a variety of wildlife species. However, it is given an overall poor rating as a wildlife food source by Martin et al. (1951) and White and Stiles (1992). White-tailed deer (*Odocoileus virginianus*), eastern cottontail (*Sylvilagus floridanus*), wild turkey (*Meleagris gallopavo*), northern bobwhite (*Colinus virginianus*), and songbirds eat the fruit and vegetation during the winter (Ripley et al. 1957; White & Stiles 1992; Witmer & Van Soest 1998; Vellend 2002). Williams et al. (1992) concluded that small mammals did not affect the population dynamics of Amur honeysuckle since consumption was low. It is possible that bush honeysuckles are competing with native vegetation for pollinators (Macior 1968). Bush honeysuckle blooms attract honey bees (*Apis mellifera*) in early summer (Southwick et al. 1981; Clark 1984), ruby-throated hummingbirds (*Archilochus colubris*) forage at Trumpet honeysuckle (*Lonicera sempervirens*) flowers, and spring azure butterflies use bush honeysuckles as larval habitat (*Celastrina ladon*) (Miller & Miller 1999).

Impacts of Exotic Bush Honeysuckles

Exotic, Japanese bush honeysuckles are tied to a variety of impacts on wildlife and ecosystems. These species have been shown to affect habitat preferences of small mammals, which are vital seed dispersers across landscapes. Edalgo et al. (2009) found that white-footed mice (*Peromyscus leucopus*) avoided areas where cover was largely provided by exotic plants. While studies have shown that some small mammal species will consume honeysuckle fruits, it appears their selection is too minimal to have any impact on the shrubs' population dynamics (Williams et al. 1992).

Likewise, songbirds have been found to consume fruits of honeysuckle species; however, the red carotenoid pigments found in the fruits can cause feather discoloration, as in cedar waxwings (*Bombycilla cedrorum*), potentially influencing reproductive fitness (Burley et al. 1982). While a number of native bird species nest in honeysuckle shrubs, many of these species are exposed to increased nest predation compared to nests located in native shrubs (Fig. 11) (Schmidt & Whelan 1999; Borgmann & Rodewald 2004). For instance, American robins (*Turdus migratorius*) nesting in Amur honeysuckle are at increased risk due to the open branching architecture of the shrub (Schmidt & Whelan 1999).

Local herpetofaunal biodiversity and body condition are negatively influenced by the density of Amur honeysuckle shrubs (McEvoy & Durtsche 2004). Buddle et al. (2004) found that spider diversity was reduced in areas of honeysuckle. Additionally, Love (2006) conducted research that concluded that the abundance, biomass, and diversity of invertebrate species are lower in dense thickets of Morrow's honeysuckle than in native species (Love 2006). Many of these species may be important prey items to native reptiles and amphibians.

Since the introduction of bush honeysuckles as ornamentals and for various habitat restoration procedures, they have become pervasive invaders across portions of the United

States. These species are known to create monocultures, which displace native vegetation and prevent forest regeneration through direct competition or through allelopathy (Woods 1993; Trisel 1997). Amur honeysuckle reduces tree seedling diversity and density (Hutchinson & Vankut 1997). In early successional habitat, Tartarian honeysuckle has been found to reduce species richness (Woods 1993).

Management of Exotic Bush Honeysuckles

Although studies have found that clipping or cutting honeysuckle shrubs can be a successful means of control, the procedure has to be carried out repeatedly (Luken 1990; Nyboer 1992), works best if shrubs are growing under shade (Luken & Mattimiro 1991), should be avoided during the winter, (Batcher & Stiles 2000) is labor intensive, and dulls power-tool blades (Nyboer 1992). Although labor intensive, large Morrow's honeysuckle shrubs were effectively eliminated through the use of tractors and chains (C. Ranson 2004, Fort Necessity National Battlefield, Farmington, PA, personal communication). The hand-pulling of smaller shrubs after rain was found to be successful but also labor intensive (Todd 1985; Batcher & Stiles 2000). During the growing season prescribed burns can be used, but require follow-up treatments due to the resprouting of some honeysuckle species (Nyboer 1992).

Herbicides represent an alternative to clipping, cutting, pulling or burning, and have been used to control honeysuckle spread (Batcher & Stiles 2000). For foliar applications, a combination of 2% glyphosate solution and a surfactant (applied between August and October) produced the best results (Miller 2003). Love and Anderson (2009) found that foliar application of herbicide and mechanical removal of shrubs were the most effective methods for controlling and reducing Morrow's honeysuckle. Although both spring and fall treatments of Amur honeysuckle (with 1% foliar application of glyphosate) were effective in reducing the shrub's presence, there was seasonal variability in the effect on native shrub species, which were largely dormant (and therefore unaffected) during fall treatment (Conover & Geiger 1993).

The use of cut-stump treatments combined with herbicides is a popular method of exotic plant control. Using a cut-stump treatment or stem injection, Hartman and McCarthy (2004) found that Amur honeysuckle could be reduced by over 94%. Kline (1981) conducted a study in Wisconsin that found that both 20% and 50% solutions of Roundup® applied to cut-stumps could control Bell's honeysuckle (*Lonicera x bella*). To control bush honeysuckles, many resource managers use a 20% solution of glyphosate combined with cut-stump treatments, which work best in early fall or late summer (Nyboer 1992; Batcher & Stiles 2000; Miller 2003).

Small Mammal Ecosystem Function and Foraging Ecology

In *The Biology of Small Mammals,* Merritt (2010) defined the term "small mammal" as a mammal weighing less than 5 kg (11 lb). In 1991, Heusner defined a small mammal by a weight of ≤ 20 kg (44 lb). For the foraging study, a "small mammal" is defined as being no larger than 120 g (4.2 oz) (Delany 1974).

Small mammals forage in order to acquire key nutrients for growth, maintenance, and activity (Merritt 2010). Due to their high metabolic demands many small mammal species have adapted both physically and behaviorally to exploit available resources (Merritt 2010). Insectivores like shrews and moles eat a wide range of high-energy foods such as insects; while others eat low-energy food sources like grasses (Merritt 2010). Although the Northern short-tailed shrew (*Blarina brevicauda*) is generally referred to as an insectivore, during the winter months, this species is known to consume both fruits and seeds (Eadie 1944). However, it is more common that a large portion of the order Rodentia feed primarily on fruits and seeds (Merritt 2010). These small mammal species are called frugivores and granivores respectively, and will be the focus of the foraging ecology study.

Given the ability of bush honeysuckles to outcompete native shrub and herbaceous species (Batcher & Stiles 2000), it is likely honeysuckle affects the food available to the small mammal species that serve vital roles in the ecosystem (Bellows et al. 2001). Small mammals influence the following aspects of an ecosystem: (1) vegetation, (2) soils, and (3) other animals (Sieg 1988). Plant species composition, primary productivity and decomposition of plant materials are all affected by small mammal activity (Taylor 1935; Sieg 1988; Gibson et al. 1990; Ostfeld et al. 1997). Small mammals create caches, which can alter the distribution of plant species, and often dispersal of foraged seeds can increase germination potential (Sieg 1988). Mast cached by small mammals can be moved to a better germination location so that they are not in direct competition with the parent plant (Reichman 1979; Sieg 1988); this mode of dispersal could be important in the establishment of both native and invasive species (Stapanian & Smith 1986; McAuliffe 1990; Vander Wall 1994). Likewise, small mammals influence the chemical and physical properties of soils, particularly by adding nitrogen to the soil (Taylor 1935; Sieg 1988). Small mammals act as important consumers of insects and seeds (Sieg 1988). Likewise, they provide a stable food source for many carnivores and population fluctuations can have direct impacts on predator reproduction potential (Sieg 1988).

Previous Small Mammal Food Trials

Exotic plant invasions can affect the behavioral ecology of small mammal species (Witmer 1996; Williams 1999; Edalgo et al. 2009). Williams et al. (1992) conducted field food bioassays using Amur honeysuckle (*L. maackii*) and concluded that deer mice (*Peromyscus maniculatus*) will consume the fruits despite the bitter pericarp; rodents have taste receptor cells capable of detecting bitter taste (Caicedo & Roper 2001). Additionally, Williams et al. (1992) concluded that small mammals had little influence on the population dynamics of Amur honeysuckle since their consumption of fruits were low when compared to the large fruit crops produced (Ingold & Craycraft 1983). However, this study placed Petri dishes in fields and did little to control for the possibility of other wildlife species besides small mammals that were consuming the fruits. Therefore, it is difficult to conclude that small mammals were the only consumers of the seeds used in this study. In addition, this study did not conduct cache inventories to conclusively state that small mammals did not disperse Amur honeysuckle fruits and subsequently the seeds.

Shahid et al. (2009) conducted small mammal food bioassays in the fall of 2004 in Madison County, New York. They used the dried seeds of three exotics, Morrow's honeysuckle, buckthorns (*Rhamnus cathartica*) and multiflora rose (*Rosa multiflora*) to assess how small mammals use invasive woody plants. Morrow's honeysuckle seeds were only infrequently or never exhausted (1 out of 27 trials). They found that of the invasives used, *Rhamnus cathartica* had the highest rate of consumption (5 out of 27 trials), but was still lower than native *Cornus amomum* (silky dogwood) (17 out of 27 trials). The study was conducted across three cover types: maple-beech forest, old-field, and conifer plantation. Three plots were established, 30 x 50-m, and subdivided into three smaller parts with each receiving a Petri dish containing all fruits. The study had a sample size of n=3 per cover type. Dr. McCay expressed the need for a study to be conducted which uses the whole fruit of *L. morrowii* and, if possible, a larger sample size across cover types (Dr. Timothy McCay, Colgate University, personal communication).

Species Detection with Video Monitoring and Fluorescent Powder Tracking

Fluorescent powder tracking is a technique that has been tested numerous times on small mammals (Sheppe 1967; Lemen & Freeman 1985; Longland & Clements 1995; McCay 2000; Menzel et al. 2000; Edalgo et al. 2009). The pigment saturates the fur of the animals, which allow the researcher to return the next night with an ultraviolet 6 Watt Long Wave Lamp/Flashlight to follow the fluorescent trail (Edalgo et al. 2009), and identify species' tracks.

The tracking powder does not inhibit movement and is low in toxicity (Stapp 1994). After studying the pathological effects of the fluorescent powders, Stapp (1994) concluded that they were both ethical and safe to use during the trapping of small mammal species.

In addition to Sherman trapping, camera monitoring has been used in a variety of behavioral studies involving small mammals. Studies have used time-lapse video monitoring of songbird nests to determine predation risks by small mammals (Thompson & Burhans 2004; Stake et al. 2004). Likewise, trail cameras (Ivan & Swihart 2000) and time-lapse video monitoring units have been used to determine small mammal species composition in a study area, and their foraging preferences (Jansen et al. 2004; Pons & Pausas 2007).

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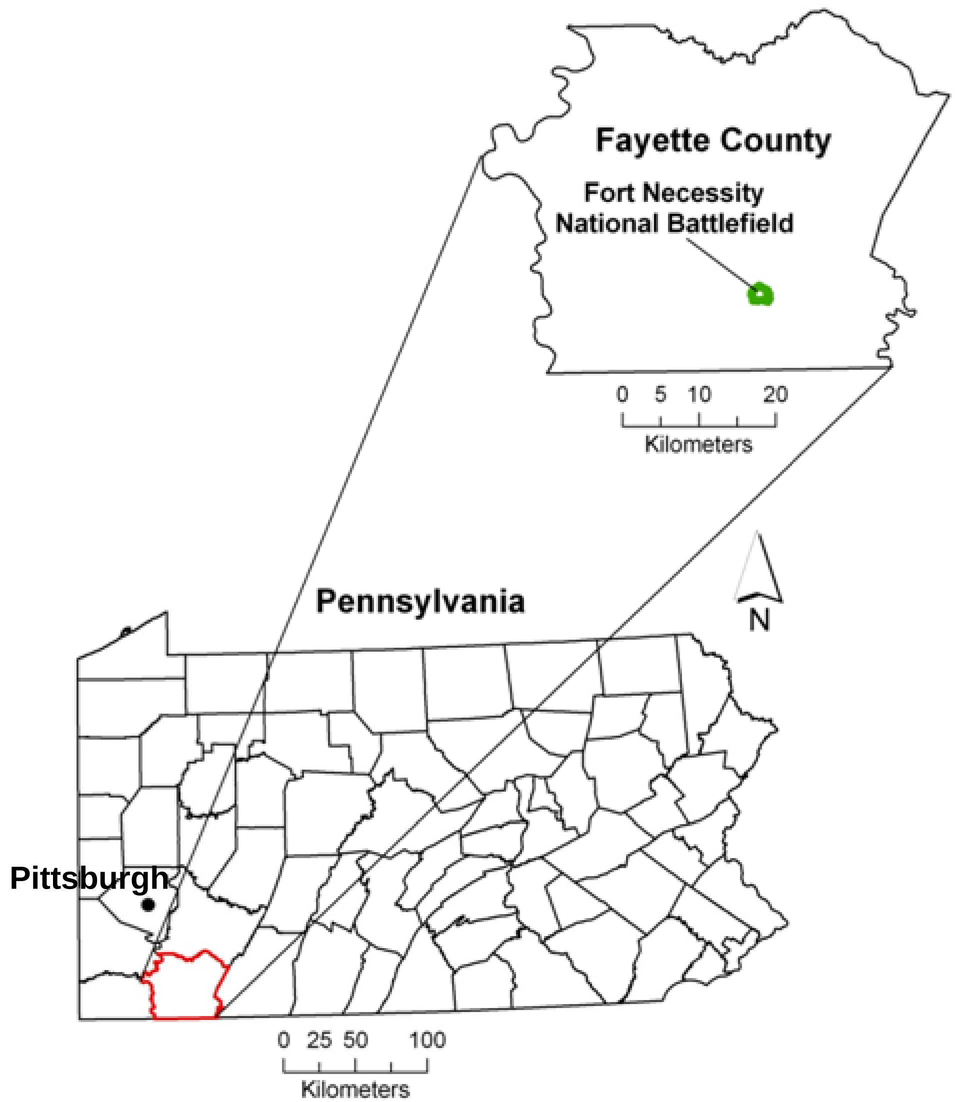

Figure 1. Fort Necessity National Battlefield lies in Fayette County, Pennsylvania, USA.

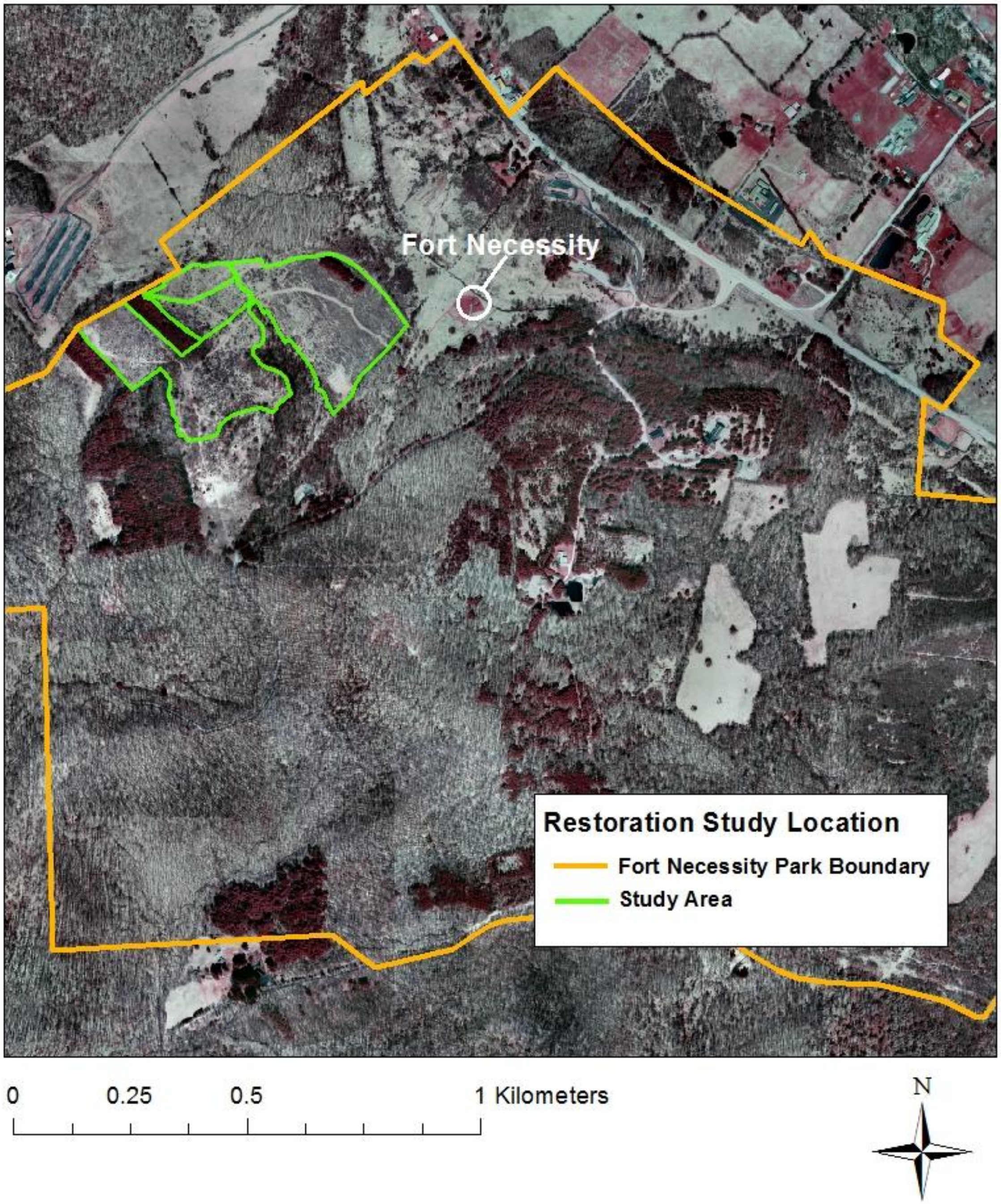

Figure 2. Project study sites are located within the 390 ha boundary of Fort Necessity National Battlefield, Pennsylvania, USA, from 2004-2010.

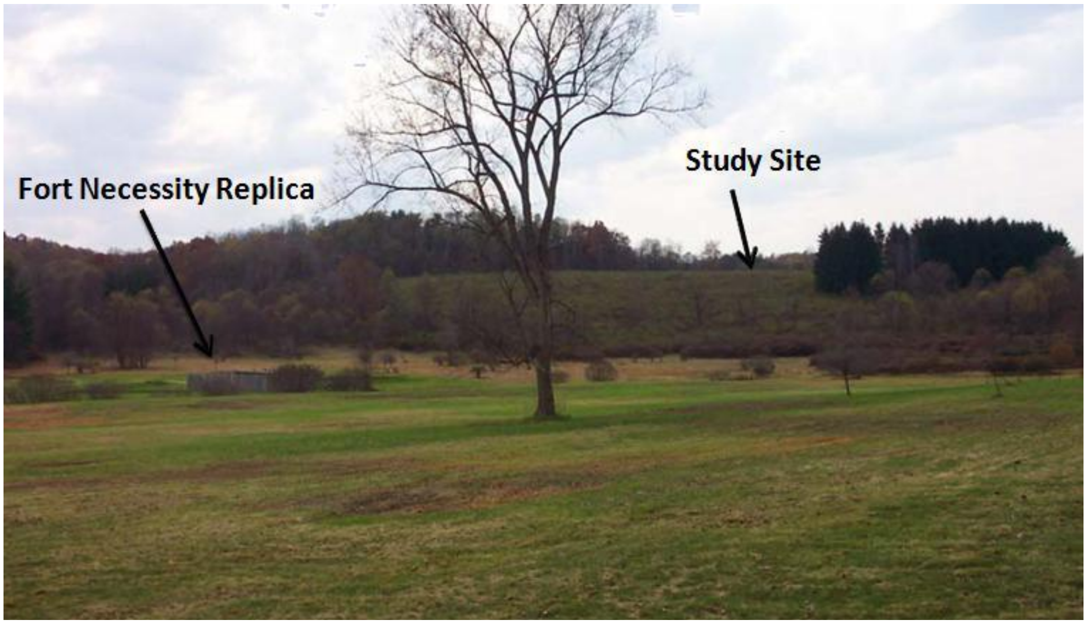

Figure 3. The study site is adjacent to the replica of Fort Necessity, at Fort Necessity National Battlefield, Pennsylvania, USA.

Figure 4. Historical re-creation of Fort Necessity at Fort Necessity National Battlefield, Pennsylvania, USA. The original was constructed by George Washington and his troops at the onset of the French and Indian War.

Figure 5. The study site was characterized by a monoculture of Morrow's honeysuckle (*Lonicera morrowii*) before treatment in 2003, at Fort Necessity National Battlefield, Pennsylvania, USA.

Figure 6. Previous graduate researcher, Jason P. Love, applying herbicide treatment to selected honeysuckle plots at Fort Necessity National Battlefield, Pennsylvania, USA, in 2005.

Figure 7. September 2007 application of Arsenal® (imidazole) to the treatment area via all-terrain vehicle, at Fort Necessity National Battlefield, Pennsylvania, USA.

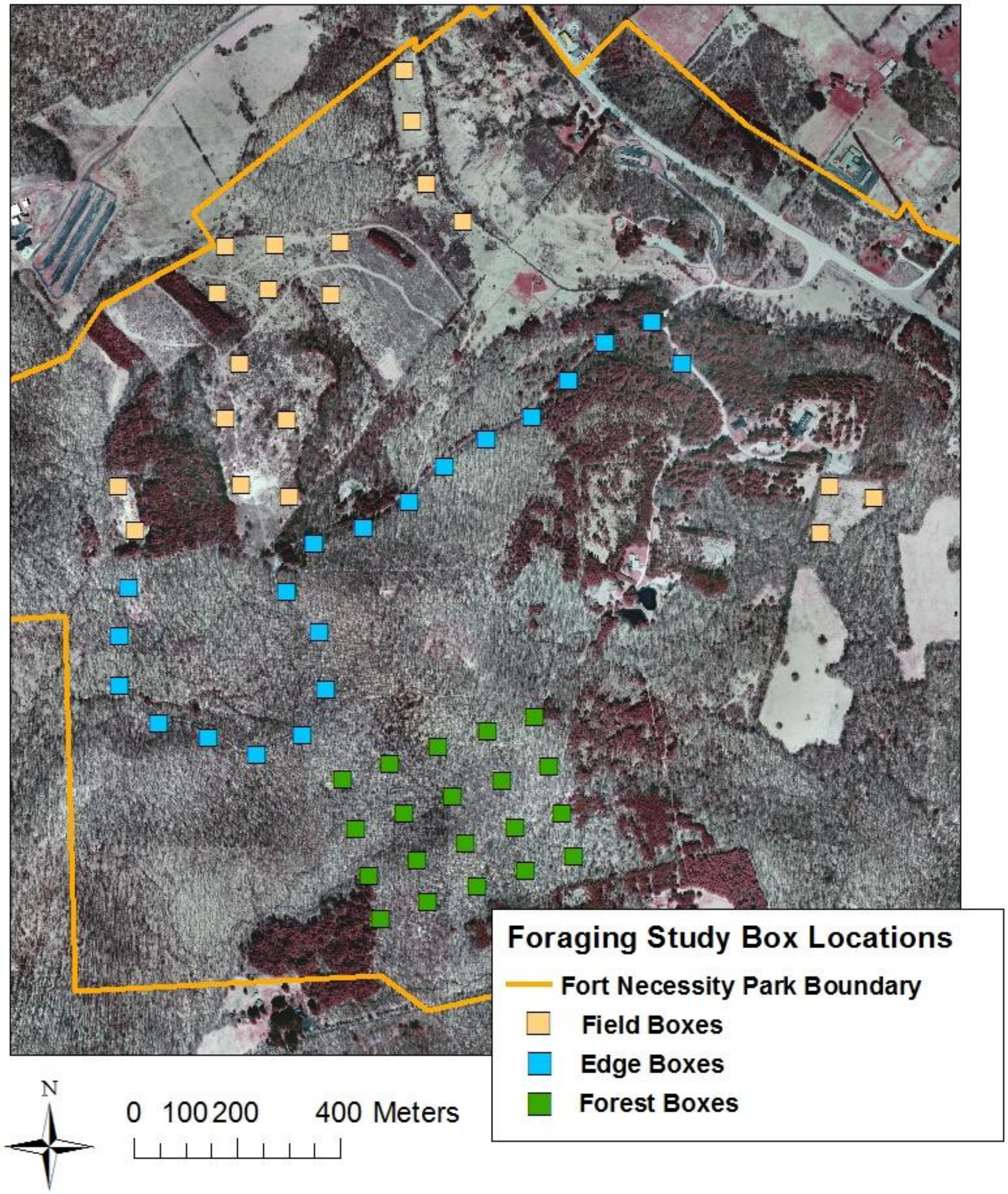

Figure 8. Foraging ecology study box locations throughout the 390 ha boundary of Fort Necessity National Battlefield, Pennsylvania, USA, 2009-2010.

Figure 9. The flowers of Morrow's honeysuckle (*Lonicera morrowii*) bloom from May-June.

Figure 10. Morrow's honeysuckle (*Lonicera morrowii*) begins fruiting in June and can carry its paired red fruits through autumn.

Figure 11. The branch architecture of Morrow's honeysuckle (*Lonicera morrowii*), and absence of thorns, leaves nesting birds open to predation; although difficult for human navigation, predators like raccoons (*Procyon lotor*) can easily move through the less dense understory of the shrub.

CHAPTER 2

RESEARCH ARTICLE

Response of Vertebrate and Vegetation Communities to the Control of Morrow's Honeysuckle (*Lonicera morrowii*) in Southwestern Pennsylvania

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Abstract

Exotic bush honeysuckles are persistent invaders in the eastern United States. Restoration initiatives involving honeysuckle removal have been conducted, but the potential consequences of chosen procedures have not been well documented for native communities. We conducted a 7– year study, with the long-term goals of controlling Morrow's honeysuckle (*Lonicera morrowii* Gray) and restoring the landscape to historical conditions. Short-term, we examined the impacts of Morrow's honeysuckle cover and the removal procedures on the biotic communities before (2004-2006) and during (2007-2010) control. Treatment sites received a combination of yearly mowing and broad-spectrum herbicides and reference sites received no treatment. Morrow's honeysuckle cover was reduced 89% following treatment. Plant species richness, percent cover of native shrubs, and floristic quality index did not vary between treatment and reference plots either before or after treatment. American woodcock (*Scolopax minor*) abundance

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This chapter written in the style of Restoration Ecology

varied among years but not between treatment and reference plots. Small mammals were better indicators of changes in the vegetation community than were songbirds at the scale of the study. Meadow voles (*Microtus pennsylvanicus*) increased over 900% and white-footed mice (*Peromyscus leucopus*) decreased 68% following restoration. Overall, our restoration efforts were successful at controlling Morrow's honeysuckle, and appeared to have minimal short-term impacts on the communities monitored.

Key words: American woodcock, early-successional habitat, exotic species, Fort Necessity National Battlefield, herbicide, invasive species, restoration, small mammal, songbird

Introduction

Restoration initiatives often involve the removal of non-indigenous plant species as they are considered undesirable compared to native species (Antonio & Meyerson 2002). However, without testing management scenarios to clearly understand the consequences of removal to native fauna and flora, it is possible that chosen practices can be harmful to the ultimate restoration goals. Hence, studies evaluating response of plant and animal communities to invasive plant species removal are greatly needed.

Morrow's honeysuckle (*Lonicera morrowii*) and other Eurasian bush honeysuckles, create 'novel' or 'emergent' ecosystems through pervasive invasion (Milton 2003; Hartman & McCarthy 2004). In 1875 Morrow's honeysuckle was introduced to the United States from Japan as an ornamental (Rehder 1940), and was later used in mine reclamation (Wade 1985) and shelterbelt formation (Herman & Davidson 1997). This species is tolerant of varying light regimes, soil types and moisture levels. Due to the indiscriminate spread of Morrow's honeysuckle (*Lonicera morrowii*) it is now found in most of the northeastern and mid-Atlantic states, as well as south-central and southeastern Canada (Batcher & Stiles 2000).

Although research has been conducted on the consequences of honeysuckle invasions (Schmidt & Whelan 1999; Daehler 2003; McEvoy & Durtsche 2004), few studies have examined the potential impacts of restoration practices used to remove these species (Hejda et al. 2009). Besides our study, only Love and Anderson (2009) have examined the effects of honeysuckle removal on early-successional herbaceous species, but their study only evaluated small experimental plots. Additionally, we found no comprehensive project assessing the response of vertebrate and vegetation communities to Morrow's honeysuckle removal. Our project evaluated the floral and faunal community response to Morrow's honeysuckle removal at a larger scale restoration event.

Our ultimate restoration goals are to control Morrow's honeysuckle by restoring a portion of the study site to a mature hardwood forest, and to manage the remaining area as early successional habitat for American woodcock (*Scolopax minor*). The objectives of this study were to determine the short-term effects of Morrow's honeysuckle removal on the following before (2004 - 2006) and after (2007 - 2010) removal: (1) the species composition of shrub and herbaceous communities; (2) the location and relative abundance of American woodcock; and (3) the relative abundance and richness of songbird, small mammal, and herpetofauna communities. For the early phase of this restoration to be considered successful we expected a decline in the percent cover of Morrow's honeysuckle in treated compared to non-treated (reference) areas, and minimal changes to the monitored communities due to control procedures.

Methods

Study Site

Research was conducted at Fort Necessity National Battlefield (FONE), located in Farmington, Pennsylvania, U.S.A. (39º48'43" N, 84º 41'50" W), in the Laurel Highlands portion of the Allegheny Mountains (Yahner et al. 2004). Elevations throughout the 390-ha park range from 535 to 710 m. The mean annual temperature is 9º C (-3º C in the winter, and 22º C in the summer). The average precipitation level is 119 cm (National Park Service 1991). Brinkerton and Armagh silt loams characterize the soils; they are moderate to well drained, medium-textured, and moderately deep (Kopas 1973).

The onset of the French and Indian War occurred at FONE in 1754, and since then the landscape has undergone many changes. During the war it was dominated by an oak-hardwood forest (Kelso 1994), but was cleared for pastureland prior to the park's establishment in 1933. The agricultural land was routinely mowed until 1985, when the area was allowed to revegetate naturally to mimic historical forested conditions (Love & Anderson 2009). However, this allowed Morrow's honeysuckle to overrun the disturbed area (Fig. 1) (National Park Service 1991). Prior to restoration the mean density of honeysuckle stems was 176,000 ± 9,960 stems/ha with the mean number of shrubs totaling 67,920 ± 4,480 shrubs/ha in fields at FONE (Love & Anderson 2009).

Restoration Procedures

To examine the impact of Morrow's honeysuckle cover and removal procedures, the 13.6–ha disturbed area was sectioned into reference (7.64–ha, where Morrow's honeysuckle would remain) and treatment (6.04–ha, where it would be removed) areas (Fig. 2). Our removal procedures were adapted from research that determined the best method for cost-effective control of Morrow's honeysuckle that would leave the most intact native vegetation (Fig. 3) (Love & Anderson 2009). Initial mowing took place in October of 2006, after 3 years of baseline data had been collected, and again in spring 2007 before the sampling season. This allowed for removal of vegetative cover, re-growth for effective herbicide application, and for survey of the community response shortly after procedures. In September of 2007, the Northeast Region Exotic Plant Management Team (NER EMPT) applied the herbicide Arsenal® (BASF Corporation, Florham Park, NJ) via all-terrain vehicle to the treatment area. Native shrubs were identified and marked to avoid mowing or spraying.

Before monitoring was conducted in 2008, we repeated the mowing process during the spring. Likewise, NER EMPT performed spot-treatments of persistent honeysuckle in September

2008 using Roundup Pro® (Monsanto Company, St. Louis, Missouri). The treated area was mowed again in the spring of 2009 and honeysuckle re-growth was treated in July 2009 by NER EMPT with Garlon® 4 (designed to treat persistent woody plants) during the sampling season (Dow AgroSciences LLC, Indianapolis, Indiana). In September 2009, native shrubs were planted in clumps and along hiking trails found in the treatment area (Fig. 4) (Appendix Ia). Spot-treatment of persistent honeysuckle was conducted in August 2010 by NER EMPT using the herbicide Accord® (Dow AgroSciences, Indianapolis, Indiana).

American Woodcock

We recorded the singing grounds of American woodcock from 2004–2010 during the breeding season (February–May) (Dwyer et al. 1983). A random starting position was chosen each week, and 10 minutes were spent in each study area. We performed surveys once a week, mapping the location of the singing grounds using a Global Positioning System (GPS) unit.

Songbirds

We used 50–m radius point counts to assess songbirds at six stations, three in reference and three in treatment (Appendix IIa) (Ralph et al. 1995, Hamel et al. 1996). Point counts were ≥ 175 m apart (Pendleton 1995) to reduce dependence of point counts but maximize the number of stations. Point counts lasted for 5 minutes at each station (Ralph et al. 1993). In 2004 (pre-removal) and 2010 (post-removal), surveys took place twice during the breeding season (May and June), and once during 2008 in June (post-treatment).

Small Mammals

Collapsible Sherman live traps (Small Folding Galvanized (SFG), 5 x 6.4 x 16.5-cm), were used to sample small mammals in four study grids throughout Fort Necessity from 2004-2010 (Appendix IIa). An equal number of trapping grids were established in both treatment and reference plots. We used 10 transects with 15 traps spaced at 8–m intervals (80 x 120–m grids).

Traps were baited with peanut butter and rolled oats wrapped with wax paper. Traps were set and checked for four consecutive days in each study plot, once a month, on three separate occasions between the months of May and August. We deducted 0.5 trap nights for each trap tripped without a capture and each with a non-target species (Beauvais & Buskirk 1999; Edalgo & Anderson 2007). Paired reference and treatment plots were trapped simultaneously to account for temporal variation. Every mouse and vole received a #1005-1 model ear tag (National Band and Tag Company, Newport, Kentucky 41072-0430). Shrews and moles were toe clipped. Animals were released at the trap station where they were caught.

Amphibians and Reptiles

We used pitfall traps combined with cover boards to increase the chances of sampling a diverse array of reptiles and amphibians. There were six pitfall drift-fence arrays installed throughout Fort Necessity, three each in treatment and reference plots (Appendix IIa). Each array consisted of three, 3-m long, 50-cm high silt-fence arms arranged in a triad design (Gibbons & Semlitsch 1981). Five-gallon (20-liter) plastic buckets were installed in the center, and at the end of each silt-fence arm array. Pitfall arrays were open every year, from 2004 – 2010, for four consecutive nights for three trapping periods between the months of May – August. Total body length and snout-vent length were recorded for salamanders and snakes, and individuals were toe clipped or caudal scale clipped for future identification. Small mammals captured in the pitfalls were ear tagged or toe clipped.

Thirty-six cover boards designed to sample reptiles and amphibians were located in the study area. The cover objects were 5 cm thick, and 30.5 cm in diameter. Three cover boards were placed 10 m north of the center bucket of the pitfall array and three were placed 10 m south of the center bucket. The cover boards were separated by 8 m to eliminate bias based on recapture distances for several species of salamanders (Mathis et al. 1995). The boards were checked twice, once at the beginning and once at the end of each pitfall trapping period.

Vegetation

Vegetation sampling was conducted through stratified random plots from the four small mammal trapping grids. There were a total of three 5 x 5-m shrub plots per trapping grid (*n* = 12). Three trap positions were chosen at random to conduct vegetation plot surveys: G2, J6, and D11 (Appendix IIa). These three locations were used throughout all study plots in both reference and treated areas. We estimated shrub cover class at each plot. Cover classes were determined using the scale designed by Daubenmire (1959), as follows: 1 (0-5%), 2 (> 5-25%), 3 (> 25-50%), 4 (> 50-75%), 5 (> 75-95%), and 6 (> 95-100%). We estimated the cover class of each species in each herbaceous vegetation plot. Vegetation surveys were conducted in 2005 (pre-removal), 2008 and 2010 (post-removal).

Data Analysis

We used Shannon–Wiener diversity index (H') (Shannon & Weaver 1949), species richness (S), and Pielou (1966) evenness index (J') to evaluate the communities. We used a floristic quality index (FQI) developed for West Virginia plants to identify the coefficient of conservatism (C) value for each species to determine mean C and FQI scores for each quadrat (Rentch & Anderson 2006). Cover class was transformed into percentage values by taking the midpoint of each class (Kercher et al. 2003).

American woodcock singing-ground locations were mapped in ArcGIS 10.0 (ESRI 2010) so that the breeding grounds could be displayed as a layer on an aerial photo of the park for each year of the study. Additionally, we calculated the following statistics for both treatment and reference plots: total number of males heard calling per year, mean number of males per survey day, and highest number of males per survey day.

Abundance of the two most recorded songbird species was assessed as the number of individuals of the species recorded during each point count survey. Since the abundance of individual songbirds was often too low for comparison, we calculated the proportion of species that belonged to a given habitat guild (early-successional, generalist, or late-successional) (Whitcomb et al. 1981; Ehrlich et al. 1988; McDermott 2007). In years where multiple point-counts were conducted, we averaged the counts for each station (Nur et al. 1999).

For small mammal and herpetofauna trapping data, we calculated total relative abundance and species relative abundances. Relative abundances were calculated as captures/100 trap nights. For small mammals, individual species representing 2% or more of total captures were included in analysis. Due to the low number of captures for all herpetofauna species, no quantitative analyses, besides relative abundance calculations, were performed on these data. Although pitfall arrays had four buckets and six cover boards, when calculating relative abundances all buckets and all cover boards were condensed into one open trap night each, due to the connectivity of the traps (Appendix IIIa).

For vegetation, songbird, small mammal, and American woodcock data we used a repeated measures, mixed effect analysis of variance (ANOVA) using PROC MIXED (SAS version 9.1.3; SAS Institute, Inc., Cary, NC, U.S.A.) to determine changes in the community indices and individual abundances over study plots, years, and their interaction. Different covariance structures were tested to determine which best fit the data based on the lowest Akaike Information Criteria (AIC) score. The lower the AIC score the better the goodness of fit for the model being tested (Burnham & Anderson 2002).

Multiple comparisons were determined using Tukey’s least-square means. We tested all variables for normality and homogeneity of variances. We approximated parametric assumptions through transformation of the following variables: number of shrubs, native percent cover, vegetation richness, field sparrow (*Spizella pusilla*) abundance, and Sherman trap and pitfall total relative abundance were log transformed; all Sherman trap and pitfall species abundances were square root transformed.

After Bonferroni corrections, to control Type 1 error rate (Williams et al. 2007), tests were considered significant at $p < 0.05$ ($\alpha = 0.05/1$ test) for woodcock, $p < 0.006$ ($\alpha = 0.05/8$ tests) for vegetation, $p < 0.007$ ($\alpha = 0.05/7$ test) for songbirds, and $p < 0.003$ ($\alpha = 0.05/20$ tests) for Sherman live-trapped and pitfall-trapped small mammals. The *F*-statistics and *p*-values for variables with statistically significant *F*-tests, in plot type, year, or the interaction, are given in the results section. Variables without significant *F*-tests are reported with corresponding *F*-statistics and *p*-values in Appendix IVa.

We visually summarized the vegetation, small mammal, and songbird communities using the multivariate technique nonmetric multidimensional scaling (NMDS) in Program R (version 2.11.1). The solution was determined using a Bray-Curtis dissimilarity matrix and multiple random starts (Clarke 1993) in 3-dimensions. Following the Wisconsin method, the data were square root transformed and double standardized (Oksanen et al. 2009). We correlated variables through vector fitting, using 999 permutations, and determined the strength (r^2) of each vector. We used average weighted abundances to add species to the ordination (Oksanen et al. 2009). It is important to note that songbird communities from a pre-removal survey conducted in 2004 were correlated to pre-removal vegetation variables measured in 2005. Permutational multivariate analysis of variance (PERMANOVA) (Program R version 2.11.1) was used to test whether the visual grouping of sites was statistically different ($\alpha = 0.05$) (Marchetti et al. 2010). Due to insufficient sample sizes to calculate permutational *p*-values, we made comparisons between pre-removal plots, and between post-removal plot types, post-removal years, and the interaction.

Results

Vegetation

We identified 12 native and three exotic shrub species at FONE (Appendix Va & VIa). An interaction showed mowing and herbicide treatments greatly reduced Morrow's honeysuckle cover and controlled the reemergence of this species, while reference plots did not change across years ($F_{[2,19]} = 9.25$, $p < 0.001$) (Fig. 5). After 5 years of treatment, percent cover of Morrow's honeysuckle was lowered by 89% when compared to the reference plots in 2010 (Appendix VIIa). An interaction showed percent cover of native shrubs did not change in treatment plots or between plot types within years, but 2008 and 2010 reference were higher than 2005 reference plots ($F_{[2,19]} = 15.13$, $p < 0.001$) (Fig. 5). Shrub species richness across all plots in 2008 was roughly twice that of 2005 ($F_{[2,19]} = 10.67$, $p < 0.001$) (Fig. 5; Appendix VIIa).

We identified 93 herbaceous species, of which 29% were exotic (Appendix Va & VIIIa). Measures of community quality were calculated with both shrub and herbaceous species. Mean C ($F_{[2,19]} = 37.17$, $p < 0.001$), mean FQI ($F_{[2,19]} = 19.99$, $p < 0.001$) and species richness ($F_{[2,19]} = 14.54$, $p < 0.001$) varied temporally (Fig. 6; Appendix VIIa). A significant interaction showed that species evenness ($F_{[2,19]} = 11.26$, $p < 0.001$) (Fig. 7) in 2010 treatment plots was about 35% higher than 2010 reference. Additionally, it showed evenness in 2010 treatment plots was 15% higher than in 2005; evenness in 2010 reference plots was nearly 15% lower than in 2008 reference. There were no differences in diversity or exotic richness by year, plot type, or the interaction (Appendix IVa).

Treatment and reference sites showed strong separation in NMDS ordination space (Fig. 8). Pre-removal (2005) reference and treatment plots clustered, and were not different ($F_{[1,2]}$ = 0.99, p = 0.674) based on the vegetation community. Post-removal treatment plots (2008, 2010) clustered, and differed ($F_{[1,4]}$ = 6.79, p = 0.004) from post-removal reference plots. Sampling years also separated vertically in ordination space ($F_{[1,4]}$ = 2.98, p = 0.033), with no interaction effect ($F_{[1,4]}$ = 1.62, p = 0.156). Neither of the post-removal plot types overlapped with the pre-removal plots. Morrow's honeysuckle percent cover, number of exotics, diversity, richness and evenness were correlated with plot groupings (Table 1). Post-removal treatment plots (2008, 2010) were negatively associated with honeysuckle cover and positively associated with increasing diversity, evenness, and vegetation richness (2008). Treatment plots in 2005 and 2008 were positively associated with exotic species.

American Woodcock

We recorded 219 displaying male woodcock (Appendix IXa). There was a significant interaction between treatment and year ($F_{[6,18]}$ = 2.82, p = 0.041) (Fig. 9). The number of males was numerically highest in treatment plots, with the highest overall mean in the post-removal treatment plots. The treatment plots during 2007 and 2010 had higher counts than 2009, which had the lowest count of any year within the treatment (Appendix Xa). There were no significant differences within reference plots across years or between plot types within specific years. Woodcock were located adjacent to mowed areas (Appendix XIa - XVIIa), particularly along the Outer and Inner Meadow Trails in the treatment area (Appendix Ia).

Songbirds

We identified 35 songbird species with 388 individual sightings (Appendix XVIIIa & XIXa). The proportion of early-successional species ranged from 77 to 92%. The majority of the remaining observations were generalist species (Appendix XVIIIa). There was a significant increase in species diversity ($F_{[2,8]}$ = 12.96, p = 0.0031) and richness ($F_{[2,8]}$ = 13.91, p = 0.0025) (Fig. 10) over time: both were higher during post-removal years (2008 and 2010) than pre-removal (2005). Richness increased by 65% and diversity increased by 30% between 2004 and 2010. Neither varied across plot type or in the interaction effect (Fig. 10). There were no differences in early-successional and generalist species proportions, species evenness, or in field sparrow (*Spizella pusilla*) and eastern towhee (*Pipilo erythrophthalmus*) abundances (Appendix IVa & XXa).

Pre-removal (2004) plots did not cluster, leading to dissimilarity across plots (Fig. 11), large within group variation, and no difference in plot types ($F_{[1,2]}$ = 0.54, p = 1.000). Both plot types during post-removal (2008 and 2010) clustered ($F_{[1,4]}$ = 1.75, p = 0.139), with separation between each year ($F_{[1,4]}$ = 3.71, p = 0.009), and no interaction effect ($F_{[1,4]}$ = 1.30, p = 0.264). Both post-removal plot types overlapped with pre-removal. Mean FQI, and songbird species diversity and richness were correlated with the ordination (Table 1). Post-removal reference plots generally had decreased mean FQI (Fig. 11), while post-removal treatment plots had higher diversity and richness (Appendix XXa).

Small Mammals

A total of 48,000 Sherman trap nights were attempted with 38,906 after deductions, yielding 2,285 captures, with 1,445 distinct individuals (Appendix XXIa). Eleven species were recorded (Appendix XXIIa): white-footed mice (*Peromyscus leucopus*), meadow voles (*Microtus pennsylvanicus*), short-tailed shrews (*Blarina brevicauda*), meadow jumping mice (*Zapus hudsonius*), and masked shrews (*Sorex cinereus*) occurred in large enough numbers (> 2% of captures) to analyze separately (Appendix XXIa).

Total relative abundance ($F_{[6,12]}$ = 0.22, p < 0.001), and meadow jumping mouse abundance ($F_{[6,12]}$ = 8.17, p = 0.001) varied by year with no difference in plot type or interaction

(Fig. 12). For both plot types, total abundance directly after treatment (2007 and 2008) was less than 2005, 2009, and 2010; and lower in 2006 than 2005 and 2010. There was no difference in jumping mouse abundance between 2004 and 2005, but these were roughly 89% higher than all other years. There was an interaction effect for white-footed mice ($F_{[6,12]}$ = 7.95, p = 0.001) and meadow voles ($F_{[6,12]}$ = 12.26, p < 0.001) (Figs. 12 & 13). In 2009 and 2010, white-footed mouse abundance was 80% higher in post-removal reference verses post-removal treatment plots (Appendix XXIIIa). Within treatment plots, abundance was higher in 2005 than 2008 and 2009. Reference plots were higher in 2009 than 2004, 2007, and 2008; 2008 was also lower than 2006 and 2010. Meadow vole abundance was roughly 2070% higher in 2010 treatment plots versus the mean for all other plots across years. Diversity, richness, evenness, and all other species' abundances were not significant by plot type, year, or interaction (Appendix IVa).

In the NMDS ordination (Fig. 14), pre-removal plots (2005) clustered in the top left, with no difference between types ($F_{[1,2]}$ = 1.35, p = 0.651) based on the small mammal community. The post-removal community was grouped by plot type ($F_{[1,4]}$ = 5.86, p = 0.015), year ($F_{[1,4]}$ = 10.60, p < 0.001), and the interaction effect ($F_{[1,4]}$ = 5.93, p = 0.012). During 2010, the reference and treatment plots showed the largest differences, stratified horizontally. Post-removal treatment and reference plots occupied different areas of the ordination space from the pre-removal plots. Mean C, total relative abundance, and small mammal species diversity and evenness were correlated with the solution (Table 1). Mean C and total relative abundance were higher in 2010 plots, with treatment plots generally having higher total abundance. Species diversity and evenness were negatively associated with 2010 treatment plots (Fig. 14). In the ordination space, meadow voles were strongly positioned with 2010 treatment plots, and white-footed mice (a generalist species) was in the center of the ordination.

We further examined the small mammal community with pitfall traps. We attempted 504 trap nights with 349 individual captures (Appendix XXIVa). We caught 10 species (Appendix XXIIa): white-footed mice, meadow voles, short-tailed shrews, meadow jumping mice, masked shrews, smoky shrews (*Sorex fumeus*), and southern bog lemmings (*Synaptomys cooperi*) occurred in large enough numbers (> 2% of captures) to analyze separately (Appendix XXVa). Sherman traps were more efficient at capturing white-footed mice (32.2% of Sherman captures, 3.5% of pitfall captures), and meadow voles (28.7%, 9.3%). Pitfalls were more efficient at capturing masked shrews (10.4%, 60.6%), and smoky shrews (1%, 12.2%) (Appendix XXVa).

Meadow vole abundance varied across years ($F_{[6,24]} = 4.63$, $p = 0.003$), with no plot type or interaction effect (Fig. 13). Voles had higher abundance in 2009 than 2006 and 2008. Total relative abundance, diversity, richness, evenness, and abundances of all other species were not significant by plot type, year, or interaction effect (Appendix IV).

The sites had weak grouping in NMDS ordination space based on pitfall captures (Fig. 15). Pre-removal (2005) plots clustered at the top of the ordination, with no differences ($F_{[1,2]} = 5.35$, $p = 1.000$). While treatment plots were found on the lower half of the ordination, and reference on the top half, no differences were found between post-removal plots ($F_{[1,4]} = 1.302$, $p = 0.427$), years ($F_{[1,4]} = 0.67$, $p = 0.708$) or the main effects interaction ($F_{[1,4]} = 0.32$, $p = 0.993$). Post-removal reference plots overlapped with the pre-removal plots. Post-removal treatment plots (2008, 2010) were associated with higher mean FQI values. Masked shrews appeared to be associated with both post-removal reference and treatment plots (2008), and Northern short-tailed shrews were most common in pre-removal plots.

Amphibians and Reptiles

We captured 9 species of amphibians and 2 species of reptiles in pitfalls (Appendix XXVIa), from 504 trap nights, with 54 unique captures (Appendix XXVIIa). Using cover boards, we caught 4 species of amphibians and 5 species of reptiles (Appendix XXVIa), from 252 trap nights, with 74 unique captures (Appendix XXVIIIa). Pitfalls were a better method of sampling anurans, while cover boards were more efficient at detecting snake species. In total, between the two methods, we observed 10 species of amphibians and 5 species of reptiles. The most common species (combining methods) were redback salamanders (*Plethodon cinereus*) (*n* = 48, 37.5%), American toads (*Anaxyrus americanus*) (*n* = 20, 15.6%), Fowler's toads (*Anaxyrus fowleri*) (*n* = 15, 11.7%), smooth green snakes (*Opheodrys vernalis*) (*n* = 10, 7.8%) and green frogs (*Lithobates clamitans melanotus*) (*n* = 3, 2.3%) (Appendix XXIXa & XXXa).

Discussion

Vegetation

Control techniques for Morrow's honeysuckle were highly effective at reducing the shrub's presence in treated areas. The percent cover of honeysuckle declined dramatically, even though treatment areas experienced resprouting in 2008 due to root and stem sprouts (Love & Anderson 2009). Restoration was able to reduce honeysuckle cover without negatively impacting native shrub cover, shrub species richness, or herbaceous diversity and richness in treated areas. During pre-removal treatment surveys we noted a number of rare species including adderstongue (*Ophioglossum vulgatum*) and slender wheatgrass (*Elymus trachycaulus*), which we did not relocate during post-removal surveys. Instances such as these, and others, have likely lead to an increase in evenness as low abundance species are weeded out (Mulder et al. 2004) by competition or possibly restoration procedures. Mean C only ranged from 2.4–4.4. This range of values indicates that the study site generally supported species that were wide spread or were associated with degraded habitat (Rentch & Anderson 2006). The results of the floristic quality index correspond to mean C, but scores sites with similar mean C values higher than others if they have fewer exotic species. Since the treatment area is still recovering from management practices, and a monoculture of honeysuckle still persists in the reference, it is not surprising that low values are represented in both plot types.

The vegetation community experienced a shift in species composition following removal procedures. Restoration procedures opened a large amount of growing space in the treatment area for pioneer species (Denslow 1980) (those likely resistant to herbicide treatments and/or those with large quantities of seeds in the seedbank), while the control area remained dominated by honeysuckle. The result of removal was the creation of a large field consisting of primarily grass and herbaceous species. Many of the species are indicators of early-successional habitat and some are also exotic. Although a number of exotic herbaceous species persisted in the field, there was no one species dominating the area, and with community heterogeneity comes more sufficient use of resources (Davies et al. 2005).

American Woodcock

Overall, the study area served as important habitat for American woodcock. Sepik and Derleth (1993) concluded that nearby, suitable habitat for nesting had more influence on use than the vegetation cover in the singing grounds. Both the reference and treatment areas were surrounded by adequate brood-rearing habitat (young to mixed-age hardwoods) (Sepik & Derleth 1993), this is likely the reason there were no differences in singing males between the plot types.

Songbirds

The honeysuckle removal procedures revealed no short-term adverse impacts on songbird community indices or composition. These results are consistent with the findings of McCusker et al. (2010), who found no differences in avian community structure when comparing areas with and without *Lonicera* species. We are likely seeing these results as songbirds are generally better indicators of habitat conditions at the landscape scale than at smaller localized sites (Carignan &

Villard 2001). Additionally, these species are territorial (Brown 1969) and when habitat is limited, they can establish territories in non-preferred habitat (Van Horne 1983). Therefore, songbird abundances can be misleading when surveying for habitat quality or preference.

Small Mammals

The honeysuckle removal procedures showed no direct, negative impacts on small mammal community indices or relative abundances. Sullivan (1990) also found that treatment of honeysuckle with a glyphosate herbicide had little effect on recruitment of *Peromyscus* spp. and *Microtus* spp. young. While the ability to recolonize an area is unknown for many small mammal species (McShea et al. 2003), meadow voles (*Microtus pennsylvanicus*) recolonized the post-removal treatment plots in higher numbers than any other small mammal species. Restoration procedures produced critical habitat for this early-successional species (Manson et al. 1999). Meadow voles are competitive and can decrease the abundances of species like white-footed mice (*Peromyscus leucopus*) (Boonstra & Hoyle 1986) and meadow jumping mice (*Zapus hudsonius*) (Anthony et al. 1981). Likewise, species diversity is negatively correlated with high meadow vole density (Anthony et al. 1981; Manson et al. 1999). We had similar results, noting the absence of white-footed and meadow jumping mice from the treated area. The meadow vole populations spiked so dramatically in the treated plots that it likely produced a potential indirect negative effect of the restoration.

Amphibians and Reptiles

The most common amphibian captured was the redback salamander (*Plethodon cinereus*). The treatment site lacks a well-developed litter layer, and has little coarse woody debris (both of which are important for habitat requirements and moisture retention) (Ash 1995; Petranka 1998). A number of the herpetofauna species monitored will likely benefit as portions of the restoration site return to a reforested condition.

Conclusions

In general, individual abundances and indices varied temporally, and did not change appreciably due to the removal of honeysuckle. It is likely that honeysuckle cover is not the only factor affecting the monitored communities at FONE. We consider the restoration successful as there was a significant decline in the percent cover of Morrow's honeysuckle. Likewise, after testing this management scenario we observed minimal negative impact to the monitored communities and conclude that chosen procedures were not harmful to the ultimate restoration goals (Table 2). Given this result, we recommend a continued maintenance program, follow-up treatments and planting of native herbaceous cover and seedlings, to promote the persistence of the treated areas and encourage the establishment of native plant species.

Implications for Practice

- Mowing and broad-spectrum herbicides were effective at reducing the cover of Morrow's honeysuckle.
- Mowing and herbicides had minimal negative impacts on the biotic communities.
- Songbird abundances appeared to be poor indicators of localized habitat quality at the scale of this research.
- When managing habitat for American woodcock it is important to provide adequate early-successional habitat for singing-grounds but also to promote nearby forested cover for nesting and brood-rearing.
- Continued maintenance, including the planting of native herbaceous and woody vegetation and follow-up treatments, is necessary to promote a healthy native community.

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Table 1. Vector relations to nonmetric multi-dimensional scaling (NMDS) ordination of biotic communities at Fort Necessity National Battlefield, Pennsylvania, USA, 2005, 2008, and 2010. A significance of 0.1 is denoted by (^{O}), 0.05 by (*), 0.01 by (**), and 0.001 by (***).

Community Matrix	Variable	Vector r^2	*p*-value
Vegetation Surveys	Morrows honeysuckle cover	0.76	0.003**
	Native shrub cover	0.01	0.968
	Number of exotics	0.57	0.026*
	Number of shrubs	0.07	0.703
	(C) Coefficient of Conservatism	0.12	0.539
	(FQI) Floristic Quality Index	0.24	0.321
	Species diversity (H)	0.51	0.037*
	Species richness (S)	0.57	0.026*
	Species evenness (J)	0.55	0.024*
Songbird Point Count Surveys	LONMOR (honeysuckle cover)	0.01	0.985
	Native shrub cover	0.14	0.481
	Number of exotics	0.13	0.547
	(C) Coefficient of Conservatism	0.37	0.132
	(FQI) Floristic Quality Index	0.51	0.047*
	Early-successional spp. (E)	0.25	0.254
	Generalist spp. (G)	0.24	0.272
	Species diversity (H)	0.72	0.006**
	Species richness (S)	0.68	0.013*
	Species evenness (J)	0.09	0.652
Mammal Sherman Traps	LONMOR (honeysuckle cover)	0.45	0.083^{O}
	Native shrub cover	0.07	0.713
	Number of exotics	0.13	0.509
	(C) Coefficient of Conservatism	0.54	0.036*
	(FQI) Floristic Quality Index	0.47	0.058^{O}
	CPUE (captures/100 trap nights)	0.67	0.002**
	Species diversity (H)	0.94	< 0.001***
	Species richness (S)	0.24	0.299
	Species evenness (J)	0.84	0.002**
Mammal Pitfall Arrays	LONMOR (honeysuckle cover)	0.29	0.233
	Native shrub cover	0.04	0.831
	Number of exotics	0.24	0.279
	(C) Coefficient of Conservatism	0.42	0.086^{O}
	(FQI) Floristic Quality Index	0.51	0.032*
	CPUE (captures/100 trap nights)	0.18	0.405

Table 1. Continued

Variable	Vector r^2	*p*-value
Species diversity (H)	0.09	0.656
Species richness (S)	0.04	0.844
Species evenness (J)	0.52	0.069^{O}

Table 2. Impact of restoration removal procedures on various biotic communities sampled during the restoration process at Fort Necessity National Battlefield, Pennsylvania, USA from 2004-2010.

Community	Impact	Type	Reason
Herpetofauna	NA	NA	Undetermined, too few captures for analysis
Vegetation (herbaceous)	No	Neutral	None apparent, could have been masked by yearly variation across entire study site
Vegetation (native shrub)	No	Neutral	None apparent, natives were marked for avoidance during restoration procedures
American woodcock	Yes	Positive	Increased habitat that is critical during breeding season
Songbird	No	Neutral	None apparent, likely due to the large scale habitat remaining intact
Small mammal	Yes	Indirect	Increased habitat that is preferred by a competitive small mammal species

Figure 1. The Fort Necessity National Battlefield, Pennsylvania, USA, study site was characterized by a dense monoculture of Morrow’s honeysuckle in 2004 before treatment.

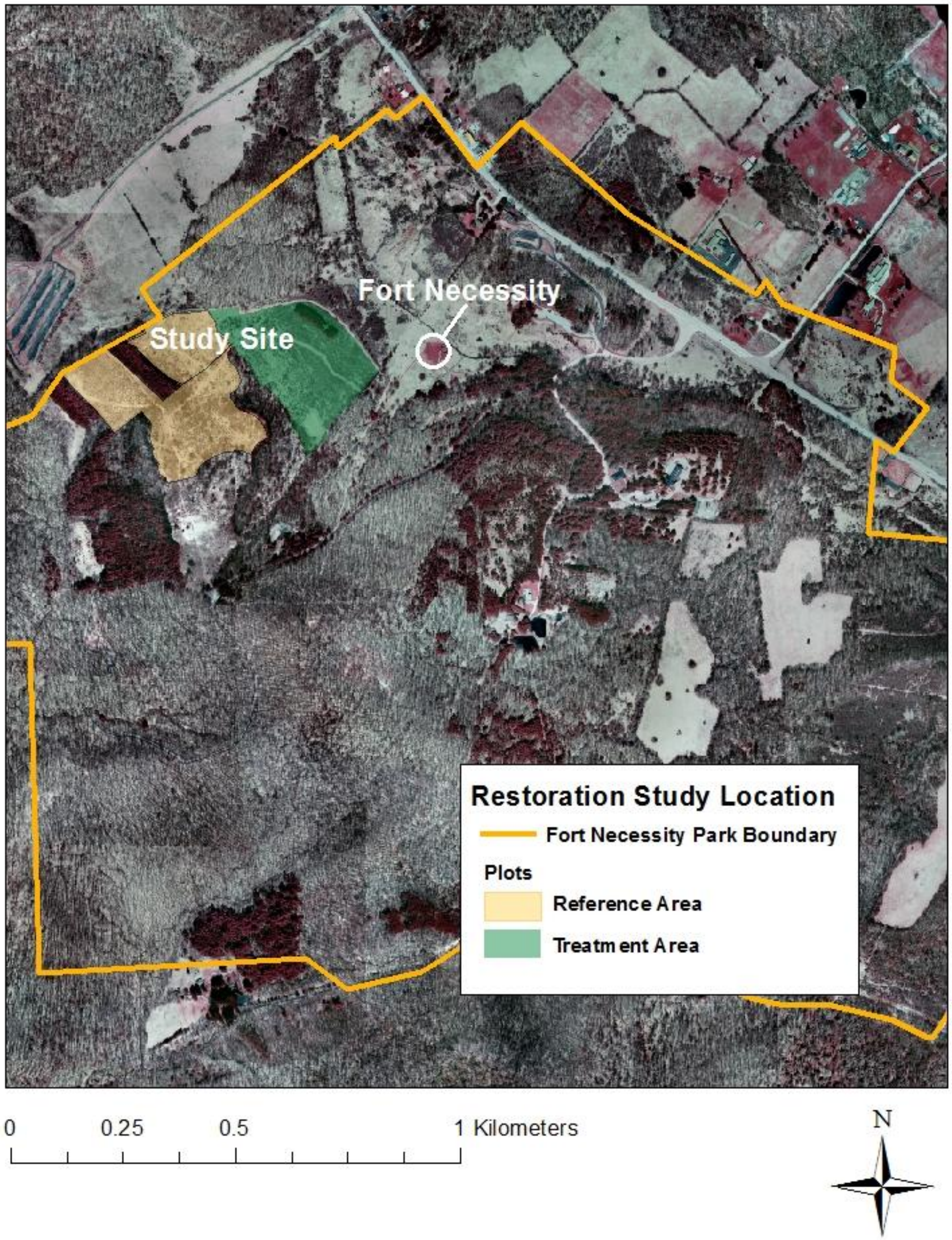


Figure 2. Project study sites were located within the 390 ha boundary of Fort Necessity National Battlefield, Pennsylvania, USA, from 2004-2010.

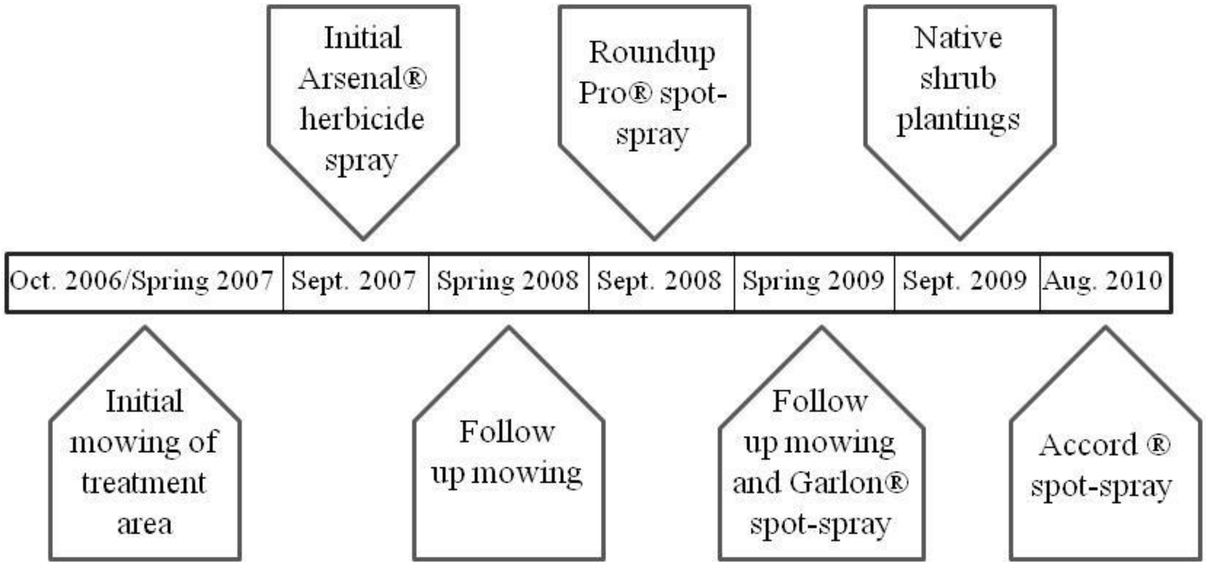


Figure 3. Timeline of major restoration procedures from October 2006 – August 2010 at Fort Necessity National Battlefield, Pennsylvania, USA.

Figure 4. The post-removal treatment site at Fort Necessity National Battlefield, Pennsylvania, USA, after September 2009 planting of native shrubs.

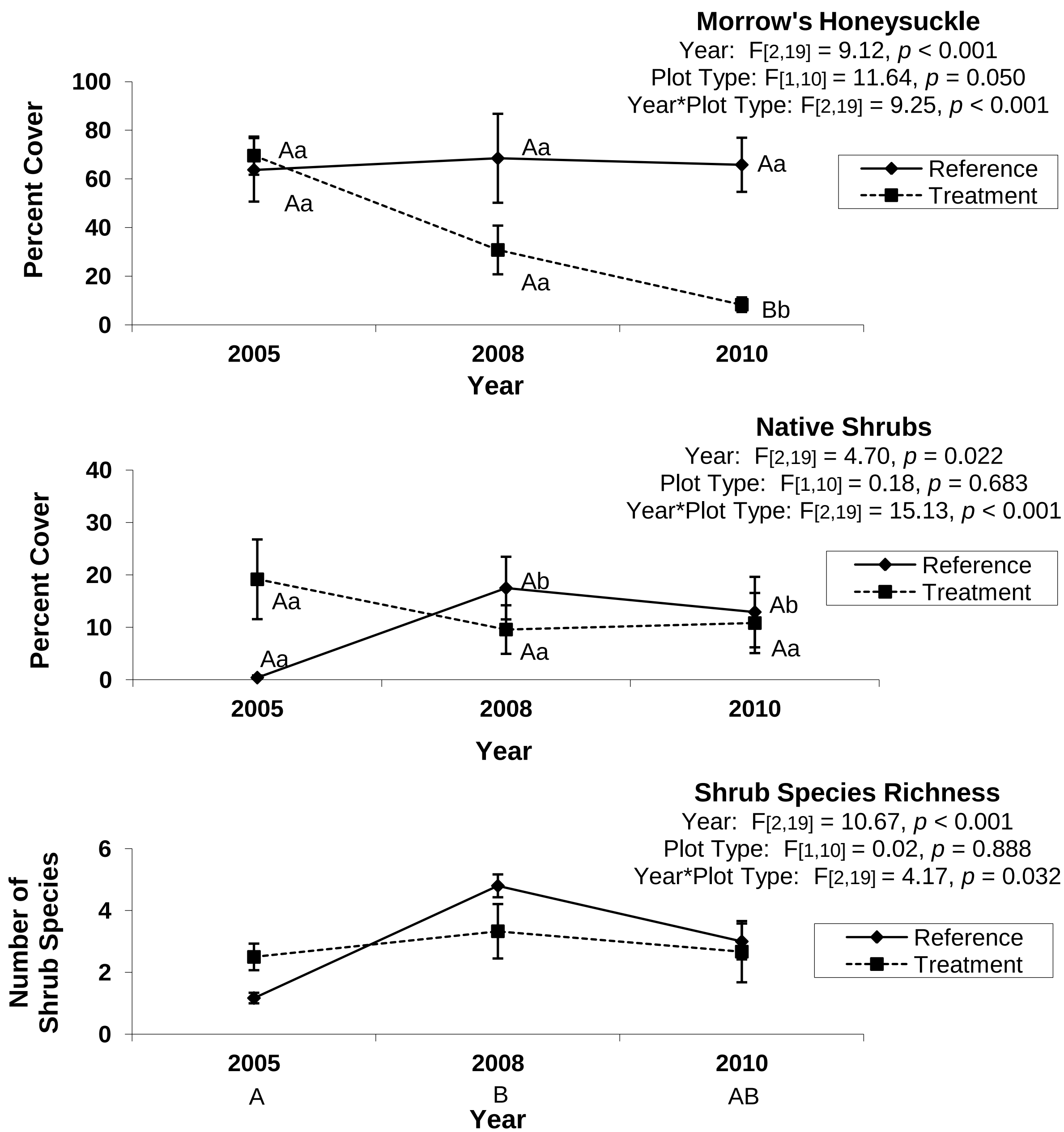


Figure 5. Mean (± SE) change in percent cover of Morrow's honeysuckle and native shrubs and shrub species in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2005), and post-removal (2008, 2010). Different lowercase letters indicate differences ($p < 0.05$) within plot types across years. Different capital letters indicate differences ($p < 0.05$) within a year, between plot types. Different capital letters below years indicate differences ($p < 0.05$) between years.

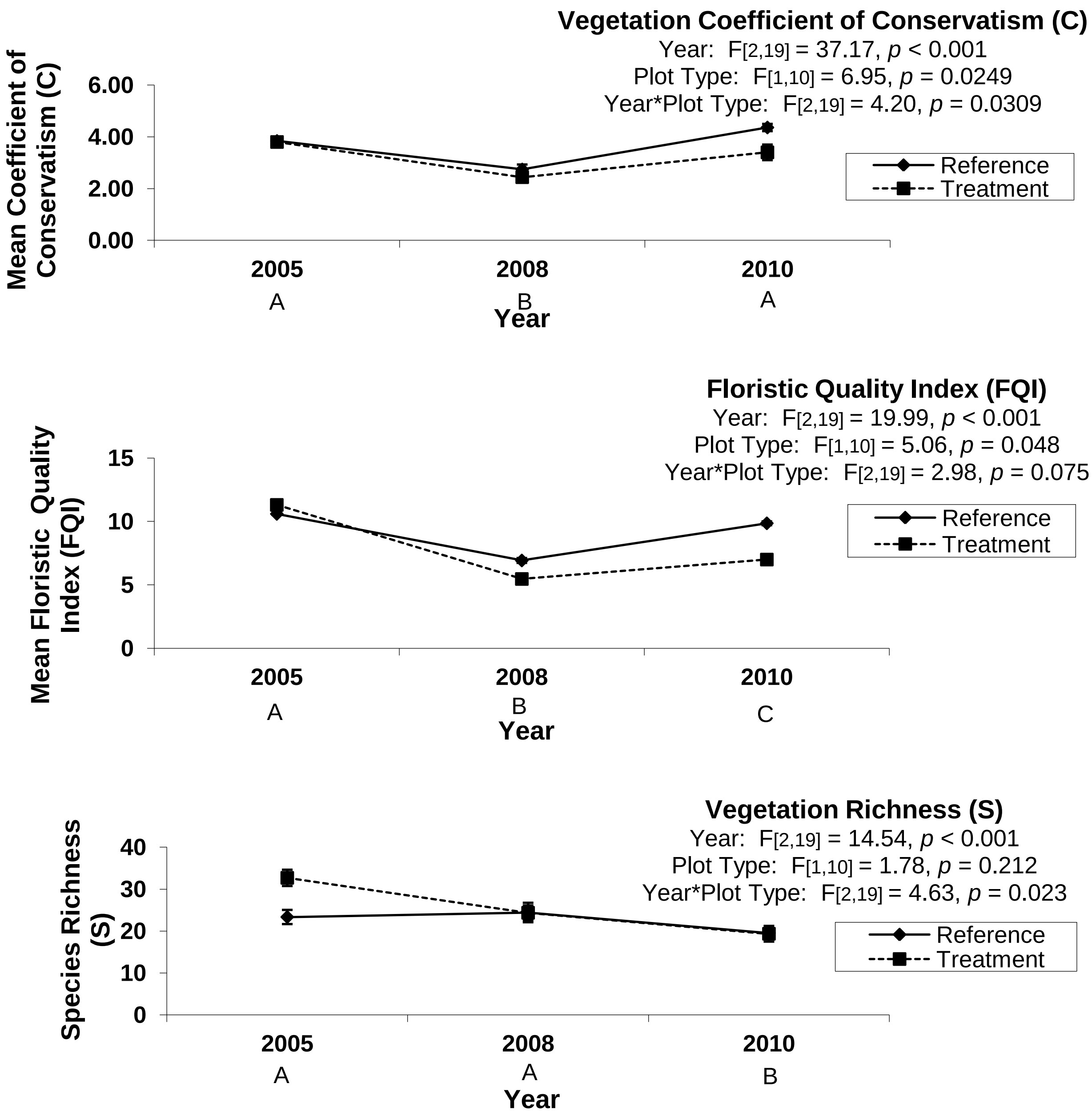


Figure 6. Mean (± SE) change in vegetation richness, coefficient of conservatism and floristic quality in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2005), and post-removal (2008, 2010). Different capital letters below years indicate differences ($p < 0.05$) between years.

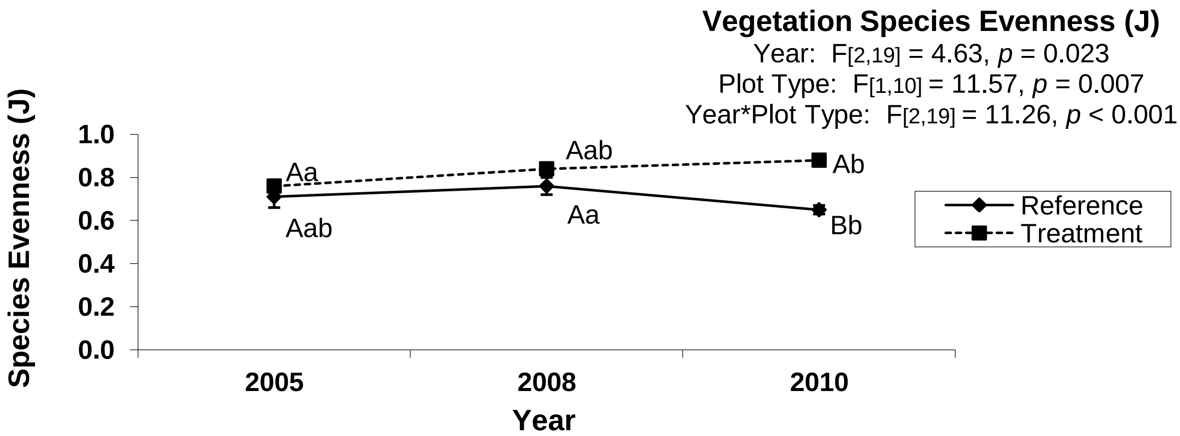


Figure 7. Mean (± SE) change in vegetation evenness in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2005), and post-removal (2008, 2010). Different lowercase letters indicate differences ($p < 0.05$) within plot types across years. Different capital letters indicate differences ($p < 0.05$) within a year, between plot types.

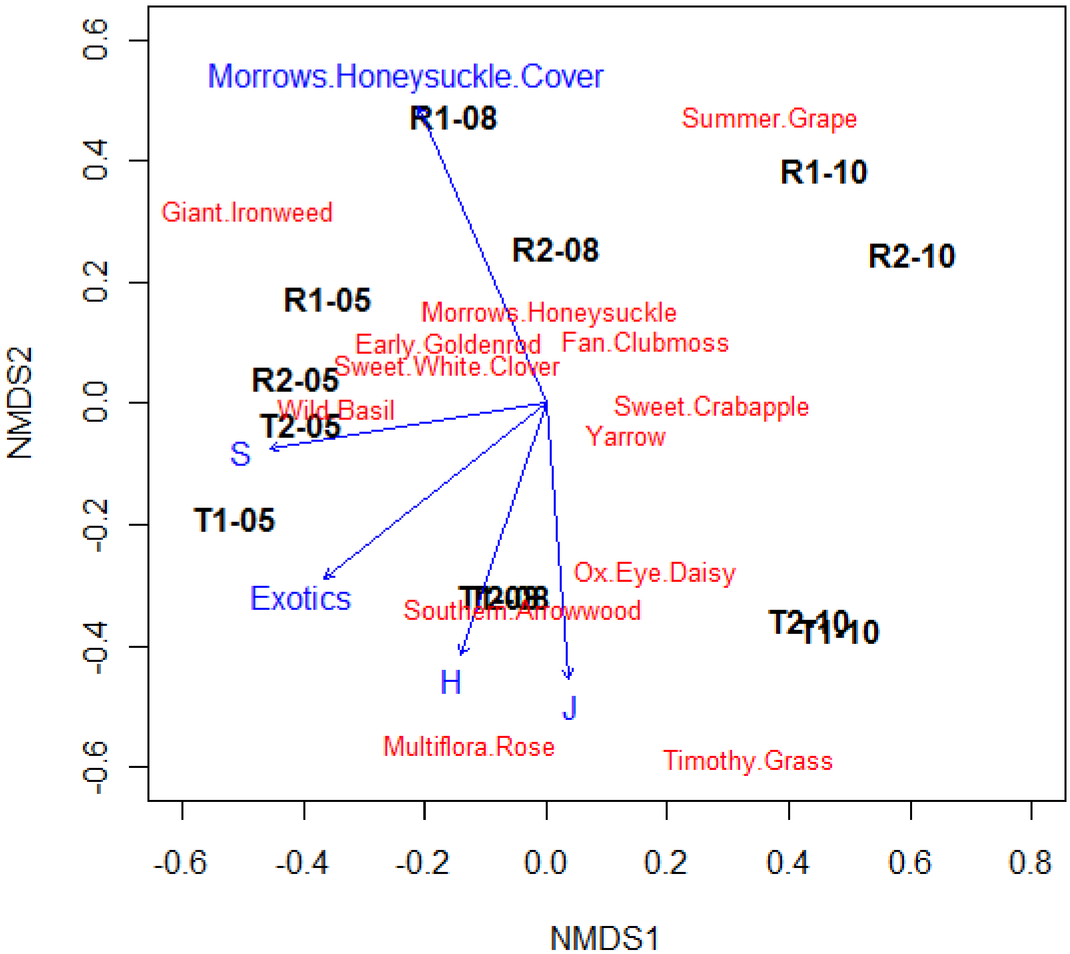


Figure 8: Nonmetric multidimensional scaling (NMDS) ordination of vegetation surveys (Bray–Curtis matrix) conducted at Fort Necessity National Battlefield, Pennsylvania, USA, in 3 dimensions showing sites labeled by type (T = Treatment, R = Reference), years (2005, 2008, 2010), habitat vectors, and weighted means positions of selected species that have strong correlation to the ordination. Pre-removal surveys were in year 2005, post-removal surveys were in years 2008 and 2010. Stress = 7.0 in the 3-dimensional solution. Vectors are significant at p = 0.05. Exotics stands for the average richness of exotic species.

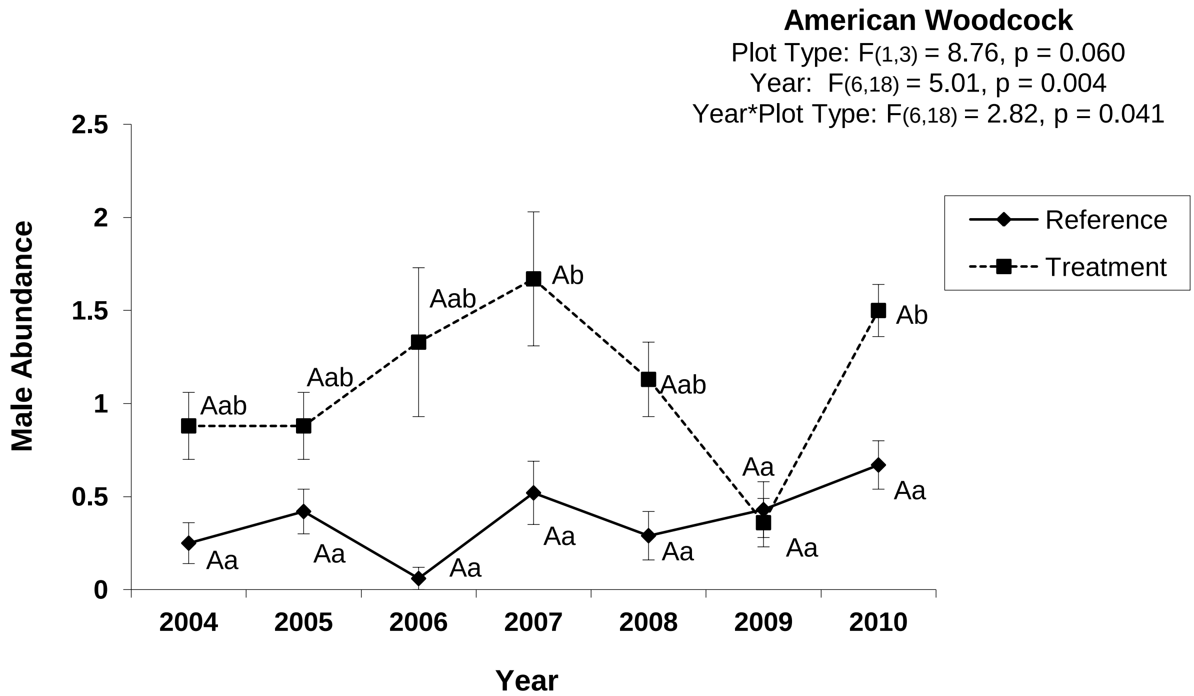


Figure 9. Mean (± SE) change in American woodcock hear calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2004 – 2006), and post-removal (2007 – 2010). Different lowercase letters indicate differences (p < 0.05) within plot types across years. Different capital letters indicate differences (p < 0.05) within a year, between plot types.

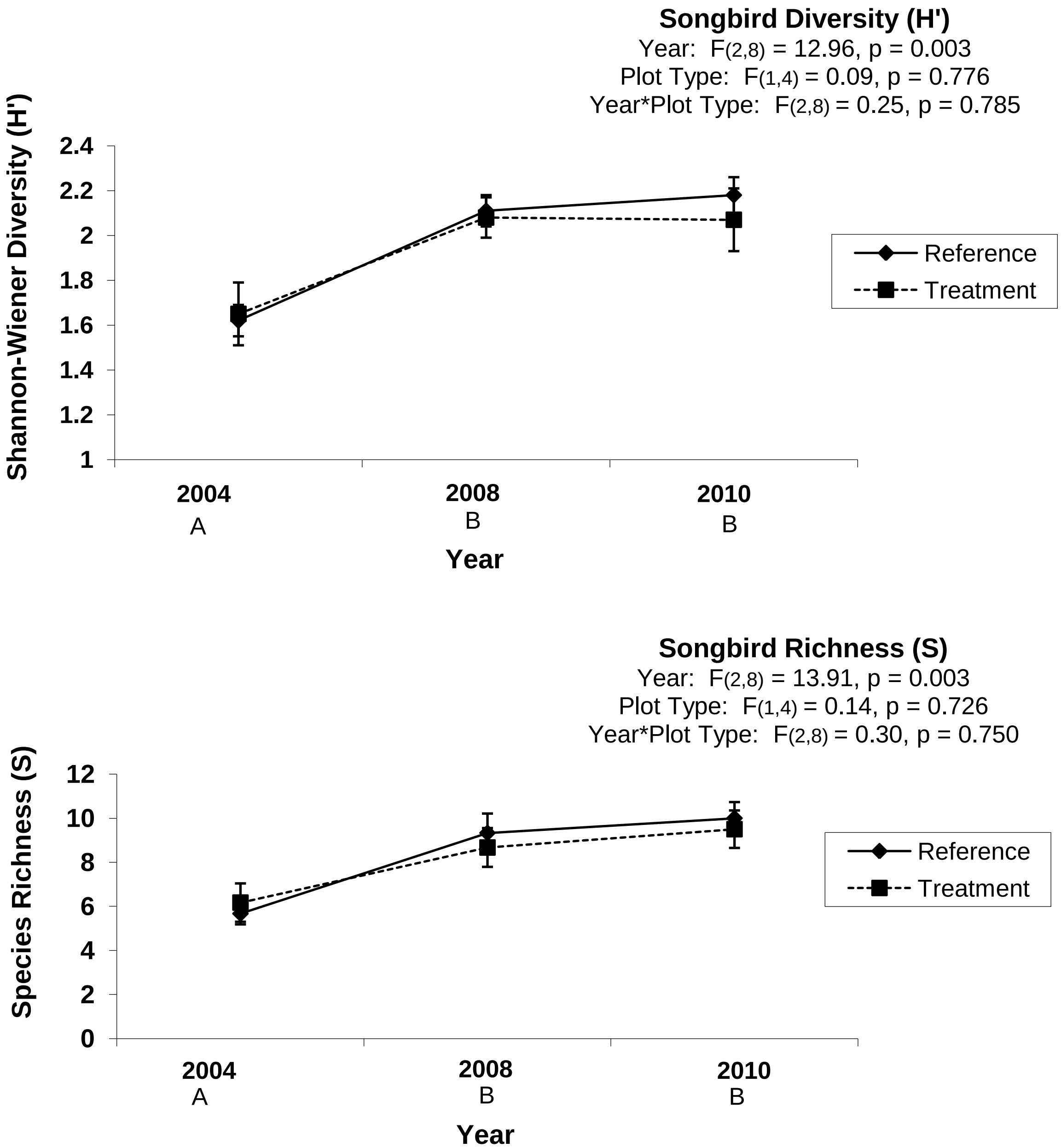


Figure 10. Mean (± SE) change in songbird diversity and richness in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2004), and post-removal (2008, 2010). Different capital letters below years indicate differences (p < 0.05) between years.

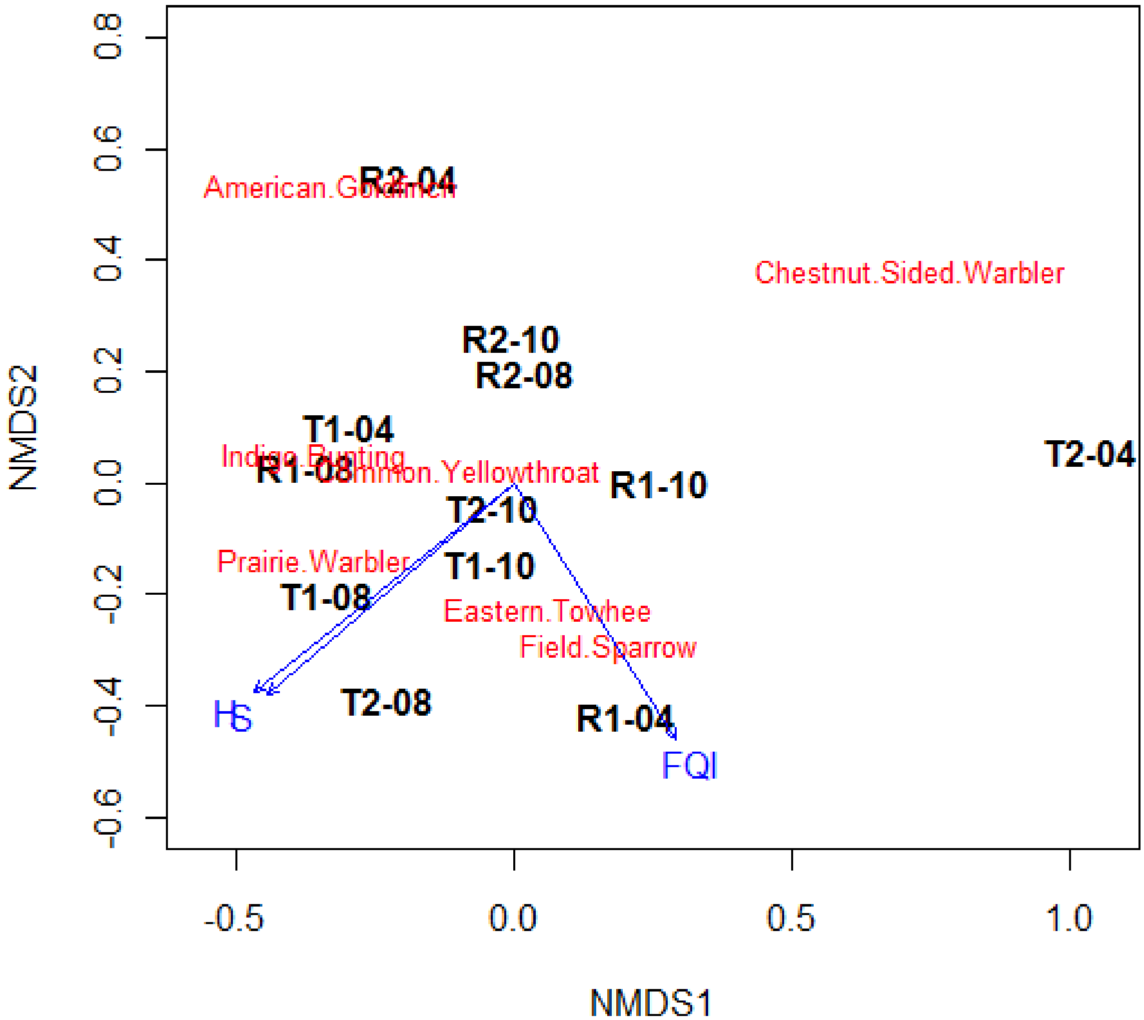


Figure 11. Nonmetric multidimensional scaling (NMDS) ordination of songbird point count surveys (Bray–Curtis matrix) from Fort Necessity National Battlefield, Pennsylvania, USA, in 3 dimensions showing sites labeled by type (T = Treatment, R = Reference), year (2004, 2008, 2010), habitat vectors, and weighted means positions of selected species that have strong correlation to the ordination. Pre-removal surveys were in year 2005, post-removal surveys were in years 2008 and 2010. Stress = 5.7 in the 3-dimensional solution. Vectors are significant at p = 0.05. FQI is code of the plant floristic quality index, H is for songbird species diversity, and S is for songbird species richness.

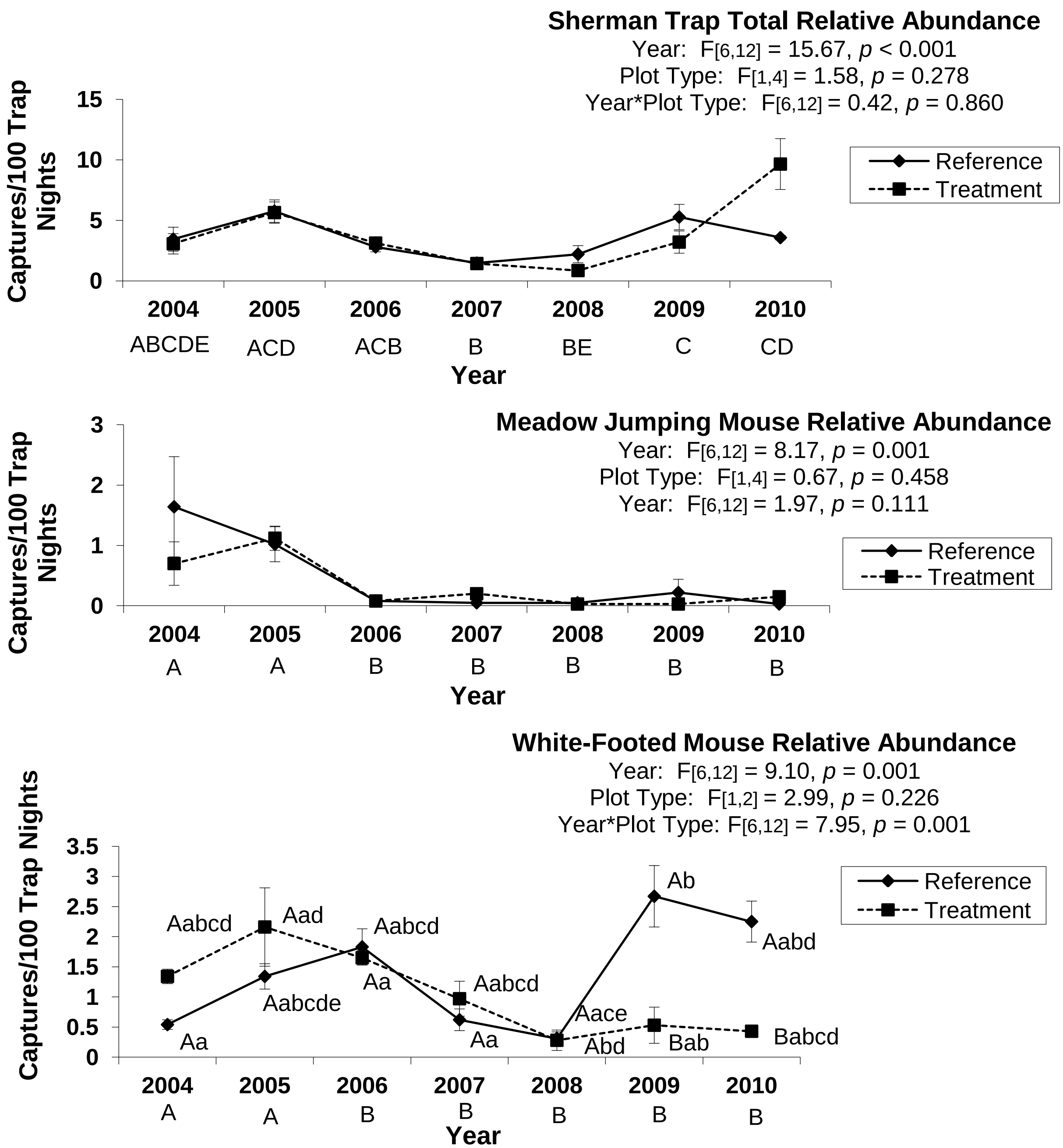
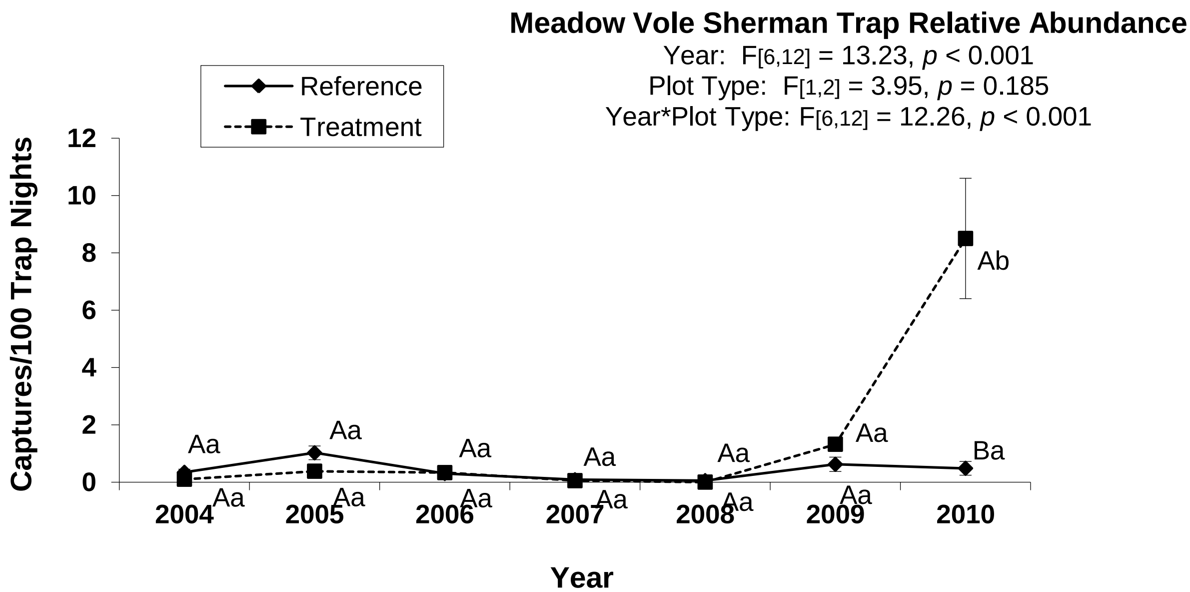


Figure 12. Mean (± SE) change in total relative abundance, meadow jumping mouse, and white-footed mouse densities, in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2004 – 2006), and post-removal (2007 – 2010). Different lowercase letters indicate differences ($p < 0.05$) within plot types across years. Different capital letters indicate differences ($p < 0.05$) within a year, between plot types. Different capital letters below years indicate differences ($p < 0.05$) between years.



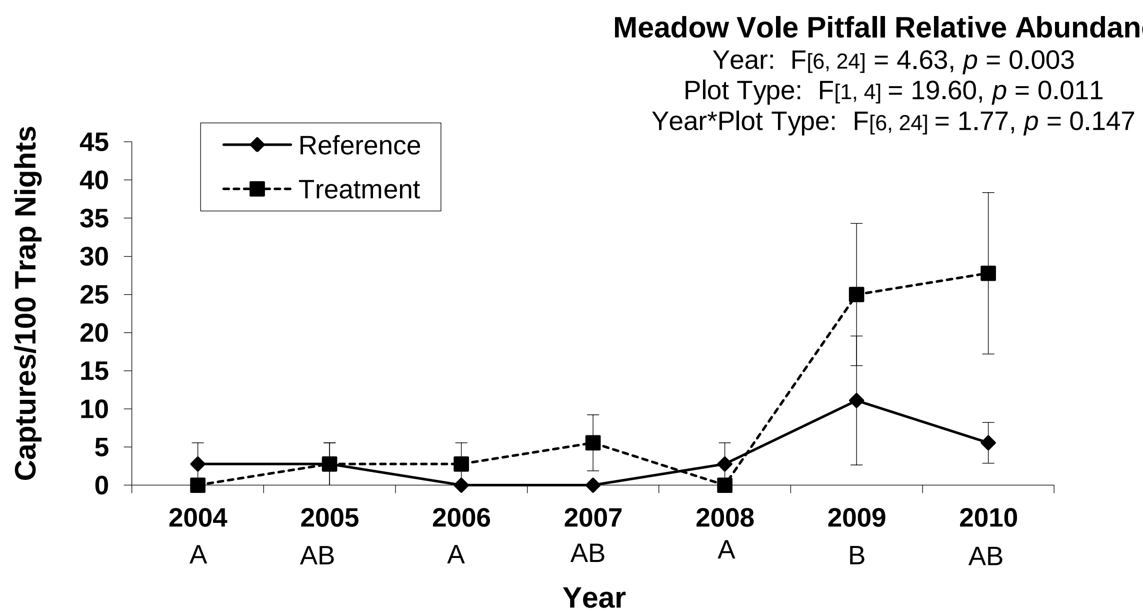


Figure 13. Mean (± SE) change in meadow vole densities from Sherman and pitfall traps, in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during pre-removal (2004 – 2006), and post-removal (2007 – 2010). Different lowercase letters indicate differences ($p < 0.05$) within plot types across years. Different capital letters indicate differences ($p < 0.05$) within a year, between plot types. Different capital letters below years indicate differences ($p < 0.05$) between years.

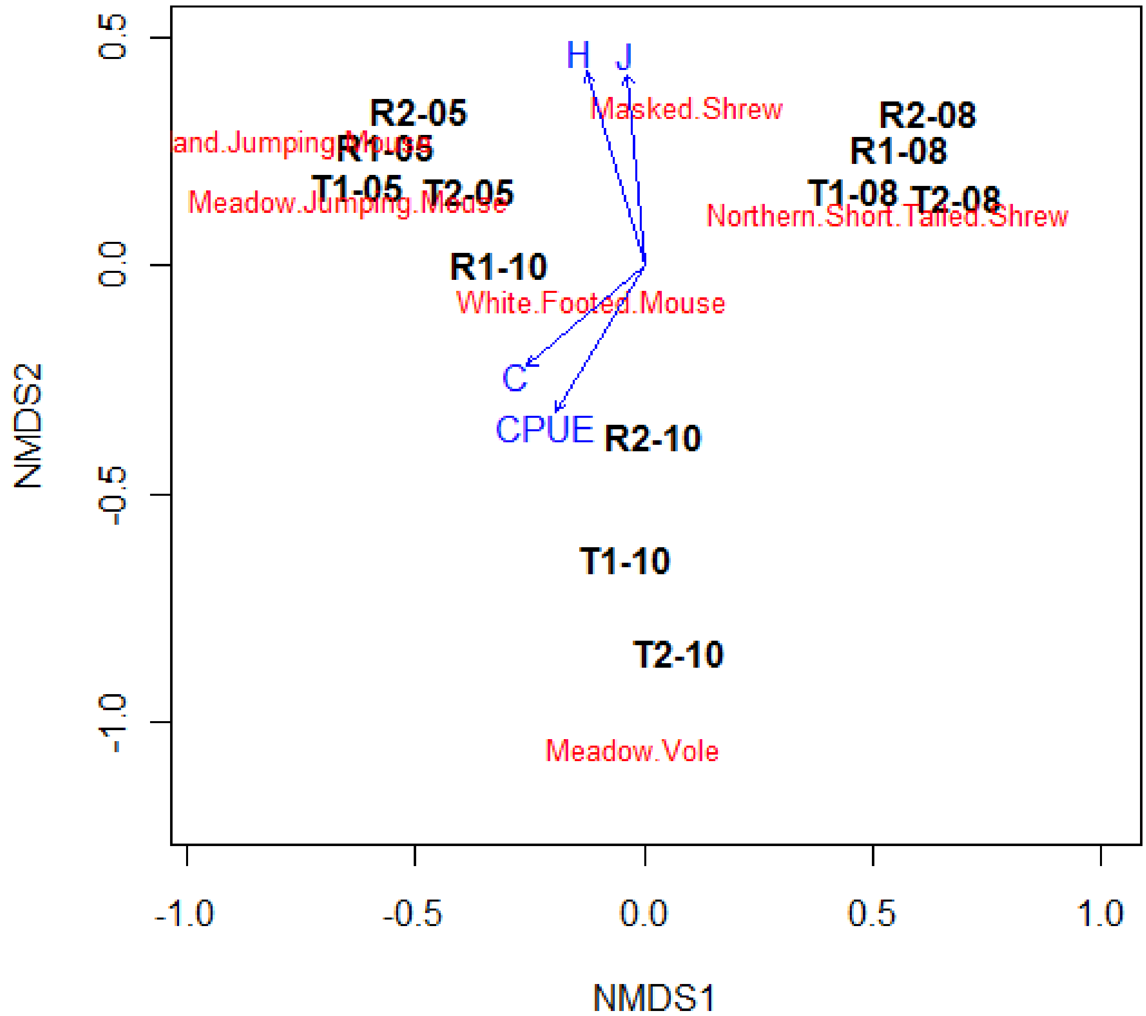


Figure 14. Nonmetric multidimensional scaling (NMDS) ordination of small mammal Sherman trapping (Bray–Curtis matrix) from Fort Necessity National Battlefield, Pennsylvania, USA, in 3 dimensions showing sites labeled by type (T = Treatment, R = Reference), year (2005, 2008, 2010), habitat vectors, and weighted means positions of selected species that have strong correlation to the ordination. Pre-removal surveys were in year 2005, post-removal surveys were in years 2008 and 2010.Stress = 2.7 in the 3-dimensional solution. Vectors are significant at $p \leq 0.05$. Code is as follows: CPUE = captures/100 trap nights, C = plant coefficient of conservatism, H = small mammal species diversity, J = small mammal species evenness.

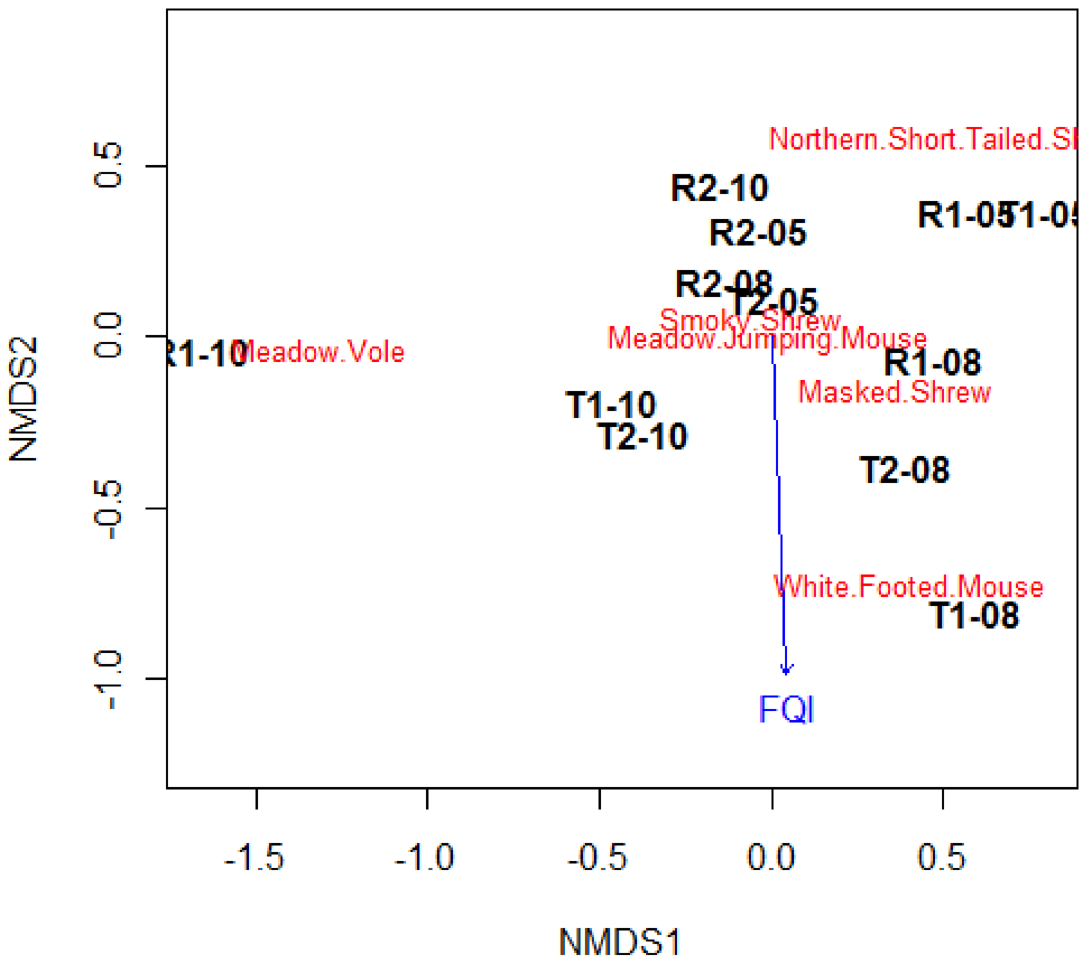


Figure 15. Nonmetric multidimensional scaling (NMDS) ordination of small mammal pitfall arrays (Bray–Curtis matrix) from Fort Necessity National Battlefield, Pennsylvania, USA, in 3 dimensions showing sites labeled by type (T = Treatment, R = Reference), year (2005, 2008, 2010), habitat vectors, and weighted means positions of selected species that have strong correlation to the ordination. Pre-removal surveys were in year 2005, post-removal surveys were in years 2008 and 2010. Stress = 6.2 in the 3-dimensional solution. All vectors shown are significant at $p = 0.05$. Code is as follows: FQI = plant floristic quality index.

CHAPTER 3

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RH: Mammal use of exotic honeysuckle fruits

White-footed mouse (*Peromyscus leucopus*) selection of invasive bush honeysuckle fruits in the Northeastern United States

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White-footed mice (*Peromyscus leucopus*) are influential in the consumption and distribution of seeds and fruit of native plants; however, little is known about their interactions with exotic shrubs. Depending on foraging activity, this species could represent an essential element of resistance against exotic plants, contribute to their spread, or have no impact. Use of invasive Morrow's honeysuckle (*Lonicera morrowii*) and 5 native soft mast species was studied across 3 cover types (forest, field, and edge) and 2 survey rounds (October – November 2009 and July – early August 2010) in southwestern Pennsylvania, USA. Feeding stations, containing equal quantities of each species, were randomly placed in each of the 3 cover types (n = 20). Honeysuckle was always present, but native species differed based on availability. In addition,

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nutrient composition, total energy, seed number, mass, and handling time were measured. Fruit consumption was non-random based on compositional analysis ($P < 0.05$). Honeysuckle was consumed over native staghorn sumac (*Rhus typhina*) in July – early August. Otherwise, it was consumed less than all natives. Honeysuckle fruits had significantly less protein (0.66%) and lipids (0.67%) than all natives ($P < 0.05$). Total energy was important in distinguishing the highest selected fruits: black cherry (*Prunus serotina*) (0.45 kcal), and common dewberry (*Rubus flagellaris*) (0.36). Use of fruits beyond the first chosen was inconsistent and varied based on moisture, protein, lipids and carbohydrates. Average seeds and mass per fruit, and handling time had no influence on use. White-footed mice show plasticity in their diets with the ability to optimize trade-offs in nutrient content across seasons. Natives in the invaded landscape appear to experience higher consumption pressures on their fruits, and we conclude that Morrow's honeysuckle creates a monoculture of a lesser-used, and likely a lesser-preferred, food item.

Key words: compositional analysis, diet selection, foraging preference, invasive species, soft mast

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Widespread attention has been paid to global trade and travel as potential pathways of invasive species spread (Hulme 2009). However, ecological pathways of persistence (such as wildlife spread of exotic plant seeds) are rarely studied (Edalgo et al. 2009; Shahid et al. 2009; Williams et al. 1992). Generalist granivore species, *Peromyscus*, are widely distributed across the United States (Hall 1980; Kantak 1983). Small mammals, including white-footed mice (*Peromyscus leucopus*) are the dominant consumers of fruit and seeds in both forests (Schnurr et al. 2002) and fields (Bowers and Dooley 1993; Manson et al. 1999; Ostfeld et al. 1997). Schnurr et al. (2004) and Ostfeld et al. (1997) found that *Peromyscus* influence the spread and survival of seedlings into a number of habitats. The diet of white-footed mice in northern latitudes is heavily dominated by fruits and seeds, especially during colder months (Wang et al. 2009). During autumn months, *Peromyscus* cache fruits in a manner that increases germination potential (Vander Wall et al. 2001). It is possible, given dietary needs and preference, that *Peromyscus* could assist in the spread of invasive plants or contribute little to their population dynamics.

Since being introduced from Japan as an ornamental (Luken and Thieret 1995; Rehder 1940), Morrow's honeysuckle (*Lonicera morrowii*) has become a persistent invasive shrub in the northeastern United States. Morrow's honeysuckle occupies a wide range of cover types including riparian areas, early successional fields (McClain and Anderson 1990), forest interiors (Woods 1993), edges, and corridors. This shrub also occupies areas of disturbed land including roadsides, railroads (Barnes and Cottam 1974), and abandoned agricultural land (Hauser 1966).

A variety of studies have examined the impacts of invasive plants on native ecosystems. Although studies are lacking that examine the effect of exotic plants on small mammal populations, Edalgo et al. (2009) conducted a study on the effects of Morrow's honeysuckle on small mammal microhabitat selection. This study found that white-footed mice selected areas with shrub cover, including that from Morrow's honeysuckle, when compared to random trails. If white-footed mice are selecting areas with Morrow's honeysuckle cover, then this could equate to foraging activity in these areas, and potentially indicate consumption of this invasive species. It is important to determine consumption rates (compared to native species as references) as they could be a factor in the pervasive spread of this invasive species. Also, because Morrow's honeysuckle is known to outcompete both native herbaceous and shrub species (Batcher and Stiles 2000) it is critical to determine if it is used as a replacement food source or if it creates vast monocultures of a less consumed food source. The objectives of our study were to: 1) determine the rate of consumption of Morrow's honeysuckle when compared to that of available native species; 2) assess the magnitude of selection across cover types; 3) and identify fruit characteristics (nutritional content, seed number, mass, and handling time) which may affect use.

MATERIALS AND METHODS

Study site. – Research was conducted at Fort Necessity National Battlefield (FONE), located in Fayette County in southwestern Pennsylvania, U.S.A. (39º48'43" N, 84º 41'50" W). FONE is situated in the Laurel Highlands portion of the Allegheny Mountains subregion of the Appalachian Plateau. Elevations throughout the 390 ha park range from 535 – 710 m. The average annual temperature is 9º C, the mean winter temperature is -3º C, and the mean summer temperature is 22º C. The average precipitation level is 119 cm (National Park Service 1991). Deep, poor to moderately drained soils (Philo silt loams) characterize the low laying areas within the park. Moderately deep, moderate to well drained soils (Brinkerton and Armagh silt loams, Cavode silt loams, and Gilpin channery silt loams) characterize the upland sites within the park (Kopas 1973).

The FONE landscape has undergone a variety of alterations. The park was once dominated by an oak-hardwood forest in the mid-1700s. The land was cleared for livestock grazing prior to the establishment of FONE in 1933. After acquisition of the area the National Park Service actively managed by mowing until the mid-1980s. It was believed that if mowing ceased it would allow the pasture to return to forested conditions. However, with passive management, the area became dominated by a dense cover of Morrow's honeysuckle and other exotic species (National Park Service 1991). During a survey of FONE fields, mean percent frequency of 93 plant species was calculated from 225 1 x 1 m vegetation samples and showed that Morrow's honeysuckle occurred at 92% of the sites (Love 2006). This frequency was greater than any native or invasive plant species known at the study site.

Cover types on FONE include field, wet meadow, oak-hardwood forest, coniferous forest, wetland, ephemeral stream, and natural and induced edge. This study was conducted across 3 of the dominant cover types: oak-hardwood forest, induced edge, and field. Edges were forested areas located within 10 m of park roads. The forest interior were forested areas 100 m in from the woodland edge or from any human-created opening (Laurance et al. 2001). The forest was comprised of mixed-hardwoods dominated by northern red oak (*Quercus rubra*) and yellow poplar (*Liriodendron tulipifera*). Fields were early successional areas dominated by various grass species (*Dactylis* spp., *Phleum* spp.), sedges (*Carex* spp.), and goldenrods (*Solidago* spp.).

Foraging stations. – We used cafeteria-style food boxes (36 x 21 x 21 cm), constructed from 1.5 cm thick plywood and 1.5 cm aperture wire mesh to allow for camera monitoring of feeding activity (Fig. 1). Each box had a single 5 x 5 cm square hole at ground level to provide access to small mammals (Shahid et al. 2009). Small mammal movements were not confined during feeding trials; individuals could visit multiple stations or forage freely outside of the stations. Four weigh boat dishes (4.6 x 4.6 cm) (Avogadro's Lab Supply Inc., Miller Place, New York) were housed in each box. All foraging stations were staked to the ground with 15 cm lawn staples and metal connecting plates to avoid disturbance by raccoons (*Procyon lotor*) or other non-target species. Sixty stations were randomly placed based on cover type (strata) with 20 in each. Each station was spaced ≥ 100 m apart to ensure statistical independence (Fig. 2) (Pearson et al. 2001; Williams 2002).

Fruit selection. – This study took place during 2 rounds: October 20 – November 14, 2009 at a time when soft mast natives were abundant in the environment (Round 1), and July 12 – August 6, 2010 when native soft mast was limited (Round 2). Natives were chosen at random from 9 species during Round 1, and from 3 species during Round 2, using a random number generator in Microsoft Excel (Appendix Ib). When a fruit species could not be located in sufficient numbers another species was randomly selected as a substitute. During the early summer sampling period the native fruit mast available was limited. Therefore, there were only 2 native species available for use: northern dewberry (*Rubus flagellaris*), and staghorn sumac (*Rhus typhina*). During the fall sampling period the native fruit mast consisted of black cherry (*Prunus serotina*), southern arrowwood (*Viburnum dentatum),* and winter grape (*Vitis cinerea*). Fruits were picked when ripened, from at least 3 plants, and within a 3-day period of the first pick. Fruit samples were frozen at -20º C at the FONE Natural Resources Research Station (until food trials), and at West Virginia University (until processed for nutrient composition).

Consumption monitoring. – Each of the fruit selection rounds in the study lasted a total of 20 days, excluding a 5-day pre-baiting period when rolled oats and peanut butter were placed in each box (Shahid et al. 2009) to encourage small mammal visitation. We placed the fruits (n = 7 per species) in separate dishes in a random order nightly. Cafeteria boxes were checked every 24 h. The number and condition of fruits and seeds remaining were recorded, cleared from the site and replenished. All berries were replaced regardless of non-consumption or condition. This was done to eliminate bias of selection based on the time individual fruits had been placed in the box.

Nutrient composition and metrics. – For each fruit species we measured percent moisture, ash, fat, crude protein (CP), neutral detergent fiber (NDF) (cellulose, hemicelluloses, and lignin), and nonfiber carbohydrates (NFC) (starches, sugars, fructans, and pectins). Each of the following procedures was performed in 3 rounds consisting of subsamples taken from a larger 10 – 20 g sample depending on moisture content (n = 30 – 600 fruits per sample). Percent moisture was determined from weights before and after drying a 10 – 20 g sample. Fat content was directly measured using a 0.50 g sample of each fruit placed in 26 x 60 mm Whatman® Cellulose

Extraction Thimbles (Whatman Incorporated, Piscataway, New Jersey) and refluxed with petroleum ether in a Tecator^{TM} Soxtec Apparatus (Rose Scientific, Alberta, Canada) (Dobush et al. 1985). Nitrogen content was measured using a 0.50 g sample of each fruit placed in a digestion tube (250 ml) in a Tecator^{TM} Digestion System (Rose Scientific, Alberta, Canada), followed by a Kjeldahl Auto 1030 Analyzer (Foss Tecator, Hoganas, Sweden) with automatic distillation and titration. We then estimated crude protein content as: [% nitrogen x 4.4] (Smith et al. 2007; Witmer 1998). Neutral detergent fiber was directly measured using a 0.25 g sample for each fruit following the Ankom Analyzer procedure, using an Ankom^{200} Fiber Analyzer (ANKOM Technology, Fairport, New York) (Getachew et al. 2004). Total nonfiber carbohydrate (TNC) was calculated as: [100% - (% fat + % ash + % CP + % NDF)] (Smith et al. 2007). Calculating NDF allowed us to distinguish between structural and nonstructural carbohydrates. Samples were then placed in a 600º C furnace for 2 h to oxidize all organic matter, allowing us to weigh the resulting inorganic residue to determine ash. Total energy (kcal) was calculated for each species as: [((% NFC x 4) + (% CP x 4) + (% fat x 9)) x fruit mass] (Atwater and Bryant 1900).

We dissected 100 fruits of each species to determine the average seed number per fruit (Williams et al. 1992). Using a Scout Pro^{TM} SP202 portable bench scale (capacity 200 x 0.01 g) (Totalcomp Scales & Balances, Fair Lawn, New Jersey) we weighed 100 samples of fruits to the nearest gram to establish the wet mass of each fruit species. To reduce scale error, all species were weighed in subsamples of 25 (except staghorn sumac, n = 50) due to the low mass of individual fruits.

Small mammal surveys. – Collapsible Sherman live traps (Large Folding Galvanized (LFG), 7.5 x 8.9 x 23 cm), were used to determine small mammal species composition. Trapping began after the final day of the selection study, and spanned over 4 consecutive days. Rolled oats and peanut butter were placed in each trap (Shahid et al. 2009), along with cotton to minimize exposure to weather extremes. Traps were opened daily at 1600 h, and checked and closed at 0600 h, to allow for detection of both diurnal (e.g., eastern chipmunks, *Tamias striatus*) (Aschoff 1966) and nocturnal species while minimizing mortality and stress (Sikes et al. 2011). Every mouse and vole received a #1005-1 model ear tag (National Band and Tag Company, Newport, Kentucky 41072-0430). Shrews were toe clipped. Animals were released at the trap station where they were caught.

Sherman traps were associated with each of the 60 foraging stations; 1 trap placed directly at the box, 1 trap spaced 33 m and another 66 m away from each box. This trap spacing makes the second trap spaced 66 m away from the first box, but only 33 m away from the next box (if present), keeping the trap spacing consistent (n = 60/per cover type). This system was employed to determine if the boxes were separated through the captures of individual mammals.

Camera monitoring units were used to confirm live trapping results, as well as to survey the foraging times and preferences of soft mast. A camouflaged camera with articulating arm, and a 12-volt deep-cycle marine battery were used in each cover type, nightly, randomly assigned to 1 of the 20 cafeteria box positions. The units consisted of a video camera connected to a recorder in a weatherproof case (Fuhrman Diversified, Seabrook, Texas) by a 20 m cable. The system used infrared light to allow filming during nocturnal foraging bouts without disrupting the animals' natural behaviors.

The articulating arm was used to position the cameras on the ground, 20 cm from the wire-mesh wall of the box, in clear view of the seed dishes and to allow for adequate lighting. Where possible, the video recorder and battery were placed behind habitat structures (logs, trees, rocks, etc.) and up to 10-15 m away from the box. We changed the videotape daily and replaced the battery every 24 – 48 h. Monitoring was started between 1800 – 2000 h, at the beginning of an active foraging period for the white-footed mouse (Williams et al. 1992). Monitoring extended until 0200 to 0400 h as tapes have an 8 h maximum recording capacity. Although other studies have had success with using time-lapse recording to extend their taping efforts a full 24 h (Stake et al. 2004; Thompson and Burhans 2003), we did not alter the standard recording frames in order to get a reliable calculation for handing time of each fruit species. Handling time was calculated for all fruits through visual monitoring of foraging activity, and was defined as the time it look a small mammal to completely consume a chosen fruit. The video footage obtained from the camera monitoring was viewed for the following information: 1) species identification, 2) time the individual entered, 3) type and number of each fruit selected, 4) handling time for each fruit, and 5) time individual exited the station.

The opening of the cafeteria boxes were saturated in fluorescent tracking powder (Edalgo et al. 2009). The pigment saturated the fur of the animals (2-4 μm; Radiant Color, Inc), and allowed us to return with an ultraviolet 6 Watt Long-Wave Lamp (Edalgo et al. 2009) to identify species' tracks. This allowed us to fine-tune the analysis and exclude any trial nights for a particular station that had been used by a non-target species (any mammal larger than 120 g, songbirds, etc.).

Research was conducted under the National Park Service (NPS) research permit FONE-2009-SCI-0002. Animal handing followed protocols approved by the Animal Care and Use Committee at West Virginia University, protocol number 09-0905. This study met guidelines of the American Society of Mammalogists (Sikes et al. 2011) for proper handling of study species.

Diet analysis. – We used compositional analysis (CA) and multivariate techniques to determine if the small mammal species surveyed were using mast resources more frequently than would be expected by chance given their relative availability. We used $\alpha = 0.05$ for all tests unless otherwise noted. We calculated the proportions of use for each fruit species, and a category of unconsumed fruits (UNCON), for each of the 60 boxes, after pooling the data from all trial days. Because all fruits were available in equal proportions in the study, selection probability was standardized across all species, and the UNCON category (100/number of foraging categories). The UNCON category was necessary to examine consumption differences among habitats. For a given study box, the consumption of the various foraging categories are not considered independent since if the consumption of one fruit increases, the consumption of another decreases (Aitchison 1986; Pendleton et al. 1998). Therefore, the analysis converted the proportions to log-ratios, and in instances when proportions equaled zero a small constant was added (0.01) (Aebischer et al. 1993). Our analysis compared use and availability of each fruit species to an arbitrary reference class by differencing log-transformed ratios of species proportions for each foraging box (Aitchison 1986; Aitchison 1994), following the methods described in Dickson and Beier (2002). A matrix ranking foraging categories was constructed when mast use was found to be nonrandom, and *t*-tests were performed to compare use between foraging categories.

Morrow's honeysuckle and UNCON were used as reference categories during multivariate analysis of variance (MANOVA) testing. The MANOVA was used to test if mean numbers of fruits consumed were consistent across cover types. When a significant MANOVA was observed, a series of univariate analysis of variance (ANOVA) tests were performed.

When the ANOVA *F*-test was significant, pairwise comparisons of cover types were examined through Tukey's multiple comparisons procedure. Residual diagnostics were performed to assess the validity of the model assumptions. In some instances, permutation tests were used to confirm previous results when a departure from the normality assumption was noted.

Nutrient and mammal community composition analysis. – We used MANOVA to determine if there were significant differences in nutrient composition, seed number, and mass among the various fruit species. This same method was used to determine if there were differences between total relative abundance, measured as captures/100 trap nights (CPUE), and the relative abundance of the 2 most dominant small mammals captured across cover types. We deducted 0.5 trap nights for each trap tripped without a capture and each with a non-target species (Beauvais and Buskirk 1999; Edalgo and Anderson 2007).

Following a significant MANOVA result, we performed a series of ANOVA tests. When the ANOVA *F*-test was significant, pairwise comparisons were examined through Tukey's multiple comparisons procedure. Percent moisture, fat, and crude protein composition of fruits and relative abundance of meadow voles (*Microtus pennsylvanicus*) (during all study rounds) were found to be in violation of parametric assumptions and were square root transformed.

Based on the camera monitoring we calculated the following descriptive statistics: total species abundances, total of each fruit species selected, average time foraging began, and average time foraging ended. Additionally, we calculated average handling time for fruit species by individual small mammal species. When the same individual was seen consuming multiples of a species we averaged the times for that individual. Due to low and unequal sample sizes, we used a Kruskal-Wallis one-way analysis of variance test to determine differences in handling time for each fruit species. Handling time was only analyzed for white-footed mice due to few observations of other species at foraging stations. Given a significant result, pairwise comparisons were examined through a series of Bonferroni corrected ($\alpha = 0.002$) Mann-Whitney U tests.

RESULTS

Small mammal community composition. – When pooling all trapping efforts between rounds there were a total of 1,330 trap nights after deductions, with 289 total captures. Out of 211 unique individuals, 187 were white-footed mice (88.63%). Other species captured included: meadow vole (n = 11 captures, 5.21%), northern short-tailed shrew (*Blarina brevicauda*, n = 5, 2.37%), woodland vole (*Microtus pinetorum*, n = 4, 1.90%), meadow jumping mouse (*Zapus hudsonius*, n = 2, 0.95%), masked shrew (*Sorex cinereus*, n = 1, 0.47%), and southern bog lemming (*Synaptomys cooperi*, n = 1, 0.47%). White-footed mice were also the dominant species observed during camera monitoring (Table 1). White-footed mice comprised 77% of the camera observations in Round 1, and 84% in Round 2. The average length of a visit to a foraging station by a white-footed mouse was 1.67 min. ± 0.19 in Round 1, and 2.22 ± 0.51 in Round 2. The eastern chipmunk (n = 3) was the only species observed with cameras that was not captured through Sherman live trapping. We never identified songbird species in the fluorescent powders, and we did not capture any on camera footage. Although, raccoons were captured on film, they were not seen disturbing the foraging boxes, so no trial nights for any station needed to be excluded.

Capture rates varied among cover types in Round 1 (Wilks' $\lambda_2 = 0.13$, $F_{6,14} = 4.20$, $P = 0.013$) and Round 2 ($\lambda_2 = 0.07$, $F_{6,14} = 6.42$, $P = 0.002$) (Table 2). There were no differences in total captures across cover types in Round 1 ($F_{2,9} = 1.29$, $P = 0.321$) or in Round 2 ($F_{2,9} = 1.94$, $P = 0.199$). There was no difference in white-footed mouse relative abundance across cover types in Round 1 ($F_{2,9}$ = 0.55, P = 0.596); however, in Round 2 fewer mice were captured in fields when compared to edges ($F_{2,9}$ = 5.47, P = 0.028). In Round 1 meadow voles had higher relative abundance in fields when compared to all other cover types ($F_{2,9}$ = 9.00, P = 0.007); the same relationship was seen in Round 2 ($F_{2,9}$ = 21.14, P < 0.001).

Fruit use. – Fruit use was non-random based on a global test of selection during Round 1 (Wilks' λ_4 = 0.12, P < 0.001), and Round 2 (λ_3 = 0.19, P < 0.001); foraging categories were ranked in order of preference for both rounds (Table 3):

Round1: Unconsumed > Cherry > Arrowwood > Grape > Honeysuckle; and

Round 2: Unconsumed > Dewberry > Honeysuckle > Sumac.

These same patterns of use were also seen, for each round, through camera monitoring at the foraging stations (Table 1). In Round 1, black cherry comprised 51% of the observed consumption, southern arrowwood 35%, winter grape 14% and Morrow's honeysuckle < 0.5%. In Round 2, northern dewberry comprised 56% of the observed consumption, Morrow's honeysuckle 29%, and staghorn sumac 15%.

Although the pattern of use did not differ across cover types during any round, consumption within a species (compared to total unconsumed fruits) and the magnitude of use (i.e., the difference between the mean numbers of fruits consumed compared to honeysuckle) for species varied across cover types in Round 2 (Wilks' λ_2 = 0.60, $F_{6,110}$ = 5.26, P < 0.001 for both tests due to functional dependency) (Table 4). Less honeysuckle was consumed in the field than in edge or forest boxes ($F_{2,57}$ = 10.15, P < 0.001), with no difference between edge and forest. Northern dewberry experienced less consumption in the field compared to other cover types ($F_{2,57}$ = 5.31, P = 0.008). Staghorn sumac also experienced less consumption in the field when compared only to edge boxes ($F_{2,57}$ = 4.96, P = 0.010), with no differences between edge and forest, and field and forest. The use of Morrow's honeysuckle over staghorn sumac was increased in the forest when compared to both edge and field ($F_{2,57}$ = 7.95, P < 0.001), with no difference between edge and field. There was no variation in the magnitude of use between northern dewberry and Morrow's honeysuckle across cover types ($F_{2,57}$ = 1.86, P = 0.165). There was no variation in consumption within a species or magnitude of use between a native and honeysuckle across cover types in Round 1 (λ_2 = 0.76, $F_{8,108}$ = 1.99, P = 0.055) (Table 4).

Fruit species characteristics. – There was variation ($\lambda_5 \approx 0.00$, P < 0.001) in the nutritional composition ($F_{35,28}$ = 397.61, P < 0.001), seed number and mass ($F_{10,1186}$ = 7699.20, P < 0.001). Among fruit species there were differences in all characteristics measured: fat ($F_{5,12}$ = 650.05, P < 0.001), moisture ($F_{5,12}$ = 674.19, P < 0.001), NDF ($F_{5,12}$ = 193.88, P < 0.001), ash ($F_{5,12}$ = 8.77, P = 0.001), crude protein ($F_{5,12}$ = 208.43, P < 0.001), NFC ($F_{5,12}$ = 58.43, P < 0.001), total energy (kcal) ($F_{5,12}$ = 1042.20, P < 0.001), seeds per fruit ($F_{5,594}$ = 10675, P < 0.001), mass per fruit ($F_{5,594}$ = 23769, P < 0.001), and handling time (χ^2_5 = 27.19, P < 0.001) (Table 5). During Round 1, black cherry was consumed highest; it had the highest kcal, protein, and non-fiber carbohydrates. Morrow's honeysuckle, used least often, had the highest moisture content, seed count, and mass, but the lowest fat, protein, and kcal. Handling time was not different between species. During Round 2, northern dewberry was consumed highest; it had the highest kcal, handling time, seed number and mass. The most utilized items, dewberry and honeysuckle, had statistically identical moisture content, which was higher than the least consumed fruit (staghorn sumac). Sumac had the highest protein, but the lowest kcal and handling time.

DISCUSSION

Characteristics affecting use. – We determined that the fruits of Morrow's honeysuckle were generally least consumed when compared to native species; although, it is as important to understand the characteristics about the fruits (other than invasiveness) that may have contributed to use. Our study verified that white-footed mice did not forage randomly, but individuals still consumed portions of all fruits available. Vickery et al. (1994) found a similar result and concluded that deer mice (*Peromyscus maniculatus*) appeared to sample a variety of foods on a daily basis to determine which had the greatest nutritional value. Although white-footed mice may exhibit similar behavior when foraging, certain fruit characteristics did stand out during the study. The highest consumed fruit from each round was always a native species, and had the highest total energy content available. Energy gained from consuming fat, protein, and carbohydrates is used for thermoregulation, growth, movement, and reproduction (Bryant and Tatner 1991). Lewis et al. (2001) found that white-footed mice used the highest-energy foods regardless of the amount of protein found in them. This result was seen in our study, with the addition that the amount of lipids and carbohydrates also appeared to be secondary to total energy.

High use also seemed to coincide with a need for water resources. Many animals rely on water in food because drinking water may be scarce and drinking may increase other risks to survival such as depredation (Maloney and Dawson 1998; Withers 1992). Our study was conducted within the breeding season for white-footed mice, which occurs from March until November (Wolff 1985). Breeding peaks in this species appear to correspond to instances where water intake and dietary protein were important; water and protein losses occur during lactation when water is lost during milk production (Barboza et al. 2009).

Temporal variation in characteristics. – White-footed mouse diets during the summer may be predominantly arthropod-based (high-protein foods), and during this season consumption of fruits and green vegetation (high-water foods) are considerably higher than during autumn when hard-mast consumption is increased (Wolff et al. 1985). Protein catabolism produces the solute urea, which increases urinary water loss in attempts to remove this toxic byproduct (Barboza et al. 2009), increasing the need for water during the summer. Although the most protein-rich fruit was not highly used during Round 2 (summer), high protein ingestion from the environment may explain the consumption of fruits such as dewberry and honeysuckle, which were high in water content.

Variation in the environment can change the availability and demand of both water and nutrients (Barboza et al. 2009). Although the pattern of use did not indicate that they were choosing fruits based solely on individual amounts of protein, fat, carbohydrates, etc., these nutrients appear to be important during specific rounds. Protein, fat, and carbohydrates seemed particularly important in Round 1 (autumn), while moisture content was more prominent in Round 2. This indicates that during the autumn, when high foraging activity was taking place in preparing for winter, easily digested sugars (quick energy) were important, and during breeding peaks lipids containing more energy and protein providing nitrogen for antibodies were important (Barboza et al. 2009).

Characteristics with low influence on use. – There were a number of characteristics that seemed to provide no influence on use. Average seeds and mass per fruit lacked importance in the study with the exception of the highest consumed species, dewberry, in Round 2. The use of this species is likely most attributed to it being highest in total energy (Lewis et al. 2001), and its high seed count is a potential side-effect. Likewise, Morrow's honeysuckle was second in mass and seeds only to dewberry and it experienced the lowest consumption when compared to all other fruits, except staghorn sumac. In an effort to consider the trade-off between energy expended during consumption and nutrients obtained we calculated the handling time for each species. However, any discernible pattern of use seemed to have little to do with this variable. Additionally we looked at patterns between total energy per gram of fruit, and total energy when compared to energy expenditure, and no additional information was revealed. Previous studies have also found no correlation between the diet of white-footed mice and the seed size, number, mass or hardness (Ivan and Swihart 2000; Kaufman and Kaufman 1989; Shahid et al. 2009), and when a highly used food item was removed individuals increased use of non-utilized food items. This ability to use a previously lower consumed food source could explain the use of honeysuckle in Round 2. There were fewer natives in the environment during this time (early summer), and this corresponds to what appears to be the highest consumption of this invasive species.

Additional factors affecting foraging. – All of the species used in the study were available in the environment outside of the foraging boxes. Due to aggressive spread, Morrow's honeysuckle was the most common study species in field habitats. We don't think that honeysuckle saturation in particular cover types confounded the results, leading to greater use of less-common natives. The consumption pattern of native fruits over honeysuckle fruits was consistent between the 3 cover types, including the forest where honeysuckle was nearly absent.

Fruits and seeds can constitute a large proportion of small mammal diets (Martin et al. 1961). Therefore, the number of fruits left unconsumed in the study boxes throughout the rounds was surprising. With closer inspection of the data, we found that there were instances when specific foraging boxes were rarely (≤ 2) or never visited by small mammals. The ability to forage may change due to weather conditions or threat of depredation (Barboza et al. 2009; Dutra et al. 2011). In addition, animals need to care for offspring, rest, and interact with other components of their environment (Caro 2005). Time for feeding activity could have been restrained and have affected the unconsumed totals depending on favorable environmental conditions. Along with the study species, there was a variety of alternative food sources available outside of the study boxes. It is therefore likely that study stations were in competition with other surrounding food sources in the environment. This directly relates to field boxes experiencing lower consumption rates during Round 2, as many were inundated with grass seed. It is likely that the saturation of grass seed is responsible for low consumption rates, due to a competing food source. Since predation risk is often regarded as one of the most pervasive factors affecting foraging activity of small mammals (Ebersole and Wilson 1980; Manson and Stiles 1998), it may be of interest to study the environmental variables that contribute to high visitation success to foraging stations.

Morrow's honeysuckle fruit characteristics. – Although honeysuckle is classified as a low quality browse food (White and Stiles 1992), Morrow's honeysuckle was in the mid-range of values for non-fiber carbohydrates and total energy available, while having one of the highest moisture contents of study fruits. Its moisture content appears to be important in its use by white-footed mice, a result not previously documented due to the availability of only dried seeds in prior feeding trials. Otherwise, the fruits are lower in fat and protein than any other species used in the study, results consistent with Witmer and Van Soest (1998). This means they are lacking key nutrition for survival and reproduction. The fruits of Morrow's honeysuckle contain compounds, such as iridoid glucosides (Ikeshiro et al. 1992), which can deter grazing herbivores. Although our study has already identified a number of factors that both favor and deter selection of honeysuckle, it is likely that defensive compounds play a role in avoidance of this species, and should be further evaluated.

Dominant species and monitoring techniques. – Through multiple sampling techniques we determined that the dominant small mammal species was the white-footed mouse. Evidence does not support that consumption of fruit was confounded by the community composition of small mammals due to the low abundances of other species and the generally consistent relative abundance of white-footed mice across cover types. While there was a decrease observed in the captures of white-footed mice during Round 2 in fields when compared to other cover types; this also coincided with a general decrease in consumption rates in that cover type. Therefore, we are confident in attributing fruit consumption to this species. They are habitat generalists (Adler and Wilson 1987; Dueser and Shugart 1978) known to have a greater dietary breadth than many granivores, which explains the ready consumption of all fruit species in the study (Lackey et al. 1985). We used foraging stations that would likely not alter mammal behavior or impede their movements to attain a more accurate picture of consumption (Connors et al. 2005). Our spacing of ≥ 100 m between boxes generally sufficed for independence; although some individuals were captured at more than 1 box, or closer than 33 m to a neighboring box. In the few instances where cameras captured other species foraging (specifically eastern chipmunks and meadow voles), their consumption of honeysuckle appeared to match those of white-footed mice. Other species captured on the monitoring units did not consume fruits from the foraging boxes. In nearly all instances seen on film, small mammals consumed the whole fruit (not just the seeds) while in the box; few were carried outside of the box. This confirmed our decision to use whole fruits at the stations, and to conduct nutritional analysis on the entire fruit.

Nutrition and energy expenditure. – The relation between food abundance and consumption can determine an animal's energy intake; the more abundant a food source is, the more likely it is to be discovered during foraging activity (Barboza et al. 2009). Animals with high metabolic rates, such as small mammals, may be unable to sustain prolonged searches for food, especially if the food source has low abundance or is of low quality (Barboza et al. 2009). This appears to be the case with invasive Morrow's honeysuckle, as it is low in vital nutrients, and creates vast monocultures reducing natives from the environment, which are the higher consumed food sources. This in turn may force small mammals, like white-footed mice, to forage for longer periods of time, or travel further distances and increase their risk of depredation. While white-footed mice showed plasticity in their diets with the ability to optimize (utilize the fruits with the highest total energy) between trade-offs in nutrient content across seasons, it is likely that this species is pressured by large energy expenditures to find suitable food sources. Natives in the invaded landscape experience higher consumption rates on their fruits, and although this shrub provides cover for small mammals (Dutra et al. 2011; Edalgo et al. 2009), we conclude that Morrow's honeysuckle creates a monoculture of a less nutritious and less used food item.

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Table 1. Small mammal species identified, time observed, and fruit consumed during camera trials at Fort Necessity National Battlefield, Pennsylvania, USA, October 2009 – August 2010. The standard error (SE) for time is reported in minutes. Successful nights were determined for each cover type out of 20 trials. Success was defined as a night when at least one small mammal was observed.

Round	Number Observed			Time Entering		Time Leaving	
Species	Edge	Field	Forest	$\overline{X}$	SE	$\overline{X}$	SE
Round 1							
White-footed mouse	9	9	9	21:25	28	00:30	30
Meadow vole	1	0	0	03:30	0	03:36	0
N. short-tailed shrew	2	0	0	01:11	11	01:12	10
Meadow jumping mouse	1	0	0	20:30	0	02:30	0
Masked shrew	2	1	1	22:06	38	22:07	37
Morrow's honeysuckle	0	0	1				
Black cherry	52	46	46				
Southern arrowwood	40	31	28				
Winter grape	7	16	16				
Successful nights	15 (75%)	10 (50%)	10 (50%)				
Round 2							
White-footed mouse	13	5	8	21:31	16	23:17	20
Meadow vole	0	2	0	19:44	34	20:23	5
Eastern chipmunk	2	0	1	18:47	35	19:11	27
Morrow's honeysuckle	14	5	10				
Northern dewberry	25	11	21				
Staghorn sumac	6	2	7				
Successful Nights	12 (60%)	7 (35%)	8 (40%)				

Table 2. Captures per 100 trap nights (CPUE) by small mammal species and total at Fort Necessity National Battlefield, Pennsylvania, USA, during October 2009 – August 2010, using Sherman live traps. Means in a row with different uppercase letters are significantly different at $P < 0.05$, based on Tukey's multiple comparisons. Only CPUE, white-footed mouse and meadow vole abundances were tested statistically due to low number of captures for other species.

Round	Edge		Field		Forest	
Species	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Round 1 (Oct. – Nov. 2009)						
White-footed mouse	10.00 A	2.97	14.17 A	3.76	13.33 A	1.92
Meadow vole	0.00 B	0.00	1.25 A	0.42	0.00 B	0.00
Woodland vole	0.00	0.00	0.00	0.00	0.83	0.83
N. short-tailed shrew	0.83	0.83	0.42	0.42	0.00	0.00
Meadow jumping mouse	0.00	0.00	0.42	0.42	0.00	0.00
Total	10.83 A	2.20	16.25 A	3.15	14.17 A	1.60
Round 2 (July – Aug. 2010)						
White-footed mouse	23.82 A	3.67	6.44 B	1.13	18.21 AB	5.33
Meadow vole	0.00 B	0.45	3.55 A	1.64	0.00 B	0.00
Woodland vole	0.00	0.00	0.00	0.00	0.92	0.53
N. short-tailed shrew	0.00	0.00	0.93	0.54	0.00	0.00
Meadow jumping mouse	0.00	0.00	0.42	0.42	0.00	0.00
Southern bog lemming	0.00	0.00	0.48	0.48	0.00	0.00
Masked shrew	0.00	0.00	0.51	0.51	0.00	0.00
Total	23.82 A	3.67	12.33 A	3.13	19.13 A	5.33

Table 3. Simplified ranking matrix of foraging boxes based on comparing foraging categories during each round at Fort Necessity National Battlefield, Pennsylvania, USA, from October 2009 – August 2010. Matrices of log-ratio differences were constructed for each box based on pooled observations. A species in a row was used significantly ($P < 0.05$) more (+ + +) or less (- - -) compared to the column headings. Single signs (+ or -) indicate a numerical, but not significant, difference. The number of positive values correspond to the rank for each foraging category, with the highest ranked item being the most consumed.

Round Species					
Round 1 (Oct. – Nov. 2009)	Honeysuckle	Cherry	Arrowwood	Grape	Rank
Morrow's honeysuckle	0	---	---	---	0
Black cherry	+++	0	+++	+++	3
Southern arrowwood	+++	---	0	+++	2
Winter grape	+++	---	---	0	1
Unconsumed	+++	+++	+++	+++	4
Round 2 (July – Aug. 2010)	Honeysuckle	Dewberry	Sumac		
Morrow's honeysuckle	0	---	+++		1
Northern dewberry	+++	0	+++		2
Staghorn sumac	---	---	0		0
Unconsumed	+++	+++	+++		3

Table 4. Mean (± SE) number of fruits consumed per box by cover type and overall average at Fort Necessity National Battlefield, Pennsylvania, USA. Original data are provided for ease of interpretation, while significances ($P < 0.05$) are based on log-ratio differences in statistical tests. Different uppercase letters in the column "Overall" indicate differences among species in a round. Different uppercase letters behind means of native species under a cover type represent a significant change in the magnitude of use (i.e., the difference between the mean numbers of fruits consumed) for a native species compared to honeysuckle across cover types within a round. Different lower case letters indicate a significant difference in consumption of a fruit across cover types within a round. Differences are based on Tukey's multiple comparisons.

Round	Edge		Field		Forest		Overall	
Species	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Round 1 (Oct. – Nov. 2009)								
Morrow's honeysuckle	0.10 a	0.03	0.39 a	0.08	0.33 a	0.06	0.27 E	0.03
Black Cherry	1.71 Aa	0.16	3.53 Aa	0.20	3.06 Aa	0.18	2.77 B	0.11
Southern arrowwood	1.65 Aa	0.16	2.97 Aa	0.19	2.21 Aa	0.17	2.28 C	0.10
Winter grape	0.57 Aa	0.09	1.55 Aa	0.14	0.99 Aa	0.11	1.04 D	0.07
Unconsumed	23.96	0.36	19.57	0.51	21.42	0.43	21.65 A	0.26
Round 2 (July – Aug. 2010)								
Morrow's honeysuckle	1.94 a	0.16	0.94 b	0.12	2.55 a	0.17	1.81 C	0.09
Northern dewberry	2.86 Aa	0.18	1.36 Ab	0.14	2.79 Aa	0.18	2.33 B	0.10
Staghorn sumac	0.82 Aa	0.09	0.44 Ab	0.07	0.43 Bab	0.06	0.56 D	0.04
Unconsumed	15.38	0.34	18.26	0.28	15.24	0.32	16.29 A	0.19

Table 5. Nutrient composition and physical characteristics of all fruit species used during foraging trials from October 2009 – August 2010 at Fort Necessity National Battlefield, Pennsylvania, USA. Means in a column with different uppercase letters are significantly different at $P < 0.05$, based on Tukey's multiple comparisons, except for handling time, which is significant at $P < 0.002$, based on Bonferroni corrected Mann – Whitney U tests.

	Seeds (no.)		Mass (g)		Moisture (%)		NDF (%)		Ash (%)	
Species	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Morrow's honeysuckle	4.59 B	0.12	0.32 B	0.00	81.61 A	0.46	1.73 E	0.16	0.64 BCD	0.01
Southern arrowwood	1.00 C	0.00	0.12 E	0.00	54.90 C	0.20	10.97 C	0.67	1.52 ABC	0.02
Black cherry	1.00 C	0.00	0.30 C	0.00	48.14 D	0.61	19.33 A	0.59	1.92 A	0.29
Northern dewberry	44.03 A	0.39	0.66 A	0.00	81.54 A	0.25	6.35 D	0.22	0.82 BC	0.09
Staghorn sumac	1.00 C	0.00	0.04 F	0.00	62.14 B	0.92	18.78 A	0.19	1.38 ABCD	0.26
Winter grape	1.40 C	0.07	0.23 D	0.00	61.60 B	0.21	13.22 B	0.76	1.62 A	0.05

	Fat (%)		Protein (%)		NFC (%)		kcal/fruit		Handling Time (sec.)	
	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Morrow's honeysuckle	0.67 E	0.06	0.66 D	0.02	14.69 C	0.32	0.216 D	0.006	41.91 B	4.62
Southern arrowwood	16.01 A	0.25	1.78 B	0.02	14.83 C	0.85	0.253 C	0.003	57.11 AB	3.73
Black cherry	5.45 B	0.15	3.83 A	0.20	21.33 A	0.59	0.449 A	0.002	55.04 AB	3.76
Northern dewberry	1.84 D	0.14	0.96 C	0.04	8.49 D	0.23	0.359 B	0.005	111.45 A	23.47
Staghorn sumac	1.51 D	0.16	1.62 B	0.02	14.57 C	0.72	0.031 E	0.001	16.09 C	1.94
Winter grape	3.89 C	0.15	1.94 B	0.10	17.73 B	0.32	0.262 C	0.007	50.98 ABC	9.38

Figure 1. Foraging boxes that were placed throughout Fort Necessity National Battlefield, Pennsylvania, USA, from October 2009 – August 2010.

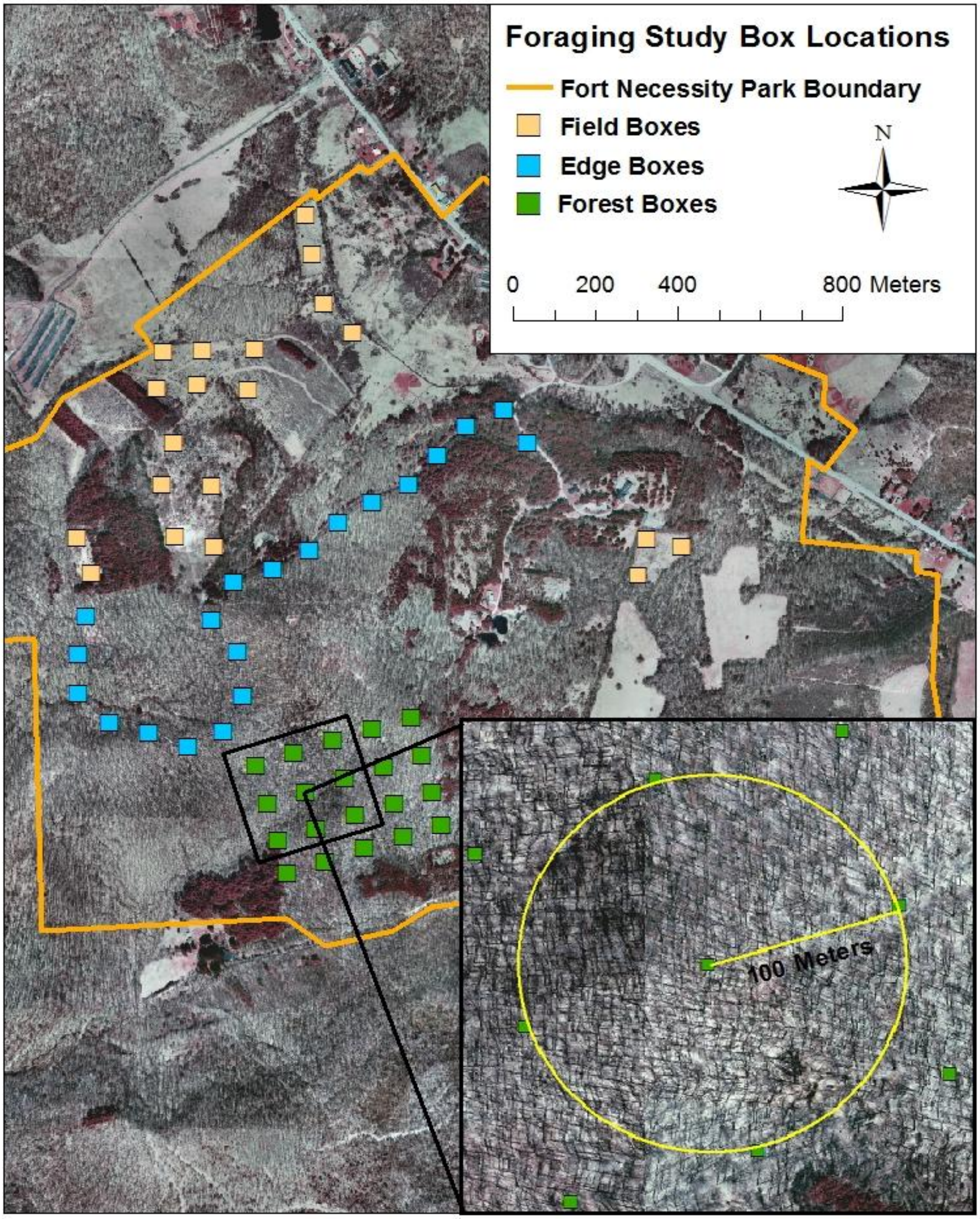

Figure 2. Foraging box locations throughout the 390 ha boundary of Fort Necessity National Battlefield, Pennsylvania, USA, October 2009 – August 2010, with emphasis on the 100 m separation between boxes.

CHAPTER 4

Habitat Characteristics that Affect White-footed Mouse (*Peromyscus leucopus*) Visitation Rate to Foraging Stations

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ABSTRACT. – Habitat use by white-footed mice (*Peromyscus leucopus*) is well documented, but no studies found have modeled the environmental characteristics that contribute to increasing visitation rate to foraging stations that don't impede natural movements. Quantifying these variables would allow greater detection in future trapping or foraging studies. Likewise, the patterns of seed consumption and dispersal may be predictable, based on the habitat structure preferences exhibited by this species. Visitation rate (no. days foraging occurred/no. days monitored) of white-footed mice to 60 foraging stations was recorded during fall (October – November) 2009 and summer (July – August) 2010, across three cover types (field, forest, edge) in southern Pennsylvania. In addition, environmental variables were collected at two spatial scales (100 m^2 and 400 m^2) to determine the relative importance of these factors and their interactions in explaining increasing visitation rate. The response variable, visitation rate, was modeled by season, cover type, and scale using general linear models. Seasons showed larger differences in key variables than scales within a cover type. We found

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This chapter written in the style of the American Midland Naturalist that edge and forest were most similar in the variables that predicted increasing visitation rate. These two cover types required areas of high structural complexity (increased canopy cover, logs, and shrubs). In contrast, fields experienced low levels of visitation in areas of high structural complexity. Variables of importance included increased rocks around foraging stations, with decreased amounts of forb, grass, and shrub coverage. Given these results, it is likely that the field cover type is the most vulnerable of the three to invasions from exotic plant species as the open space could be ideal for seed germination. These areas could be at risk from both forgotten seed caches and post-gut viability of invasive seeds. Variables that were favored appeared to reduce energy expenditure and foraging time, while providing refuge from associated predators in particular cover types (aerial in forest and edge, and terrestrial in fields). These identified areas are likely well-traveled and have higher rates of foraging activity and seed dispersal. Therefore, study stations in these areas are likely to have increased visitation rate that could lead to successful detection of species, and potentially the increased consumption of one food source over another.

INTRODUCTION

White-footed mice are key consumers of plant materials and pest insects (Smith and Campbell, 1978; Elkinton *et al*., 1996). Additionally, they provide a valuable prey base for mammalian, avian, and herpetofaunal predators (Klimstra, 1959; Hockman and Chapman, 1983; Livezey, 2007). The distribution of white-footed mice is greatly affected by habitat availability (Cummings and Vessey, 1994), and this species has been the subject of numerous research efforts centered on habitat selection and response to anthropogenic disturbances (McComb and Rumsey, 1982; Clark *et al*., 1987; Planz and Kirkland, 1992; Cummings and Vessey, 1994; Dooley and Bowers, 1996; Nupp and Swihart, 1998; Wolf and Batzli, 2002; Jorgensen, 2004).

The conclusion drawn is that this species is a habitat generalist (Kaufman *et al*., 1983; Lackey *et al*. 1985). Since white-footed mice are widely distributed, habitat use has been documented across a variety of cover types (Kaufman *et al*., 1983), and due to variation between patches (Bowman *et al*., 2001) environmental variables have been studied at different spatial scales (Nupp and Swihart, 1996).

Seed consumption and dispersal patterns (Drickamer, 1970; De Steven, 1991; Myster and Pickett, 1993; Manson and Stiles, 1998; Shahid *et al*., 2009) have also been documented for this species, which has been found to be the dominant consumer of seeds in old fields of eastern North America (Bowers and Dooley, 1993; Ostfeld *et al*., 1997; Manson *et al*., 1999). White-footed mice, together with other granivores, may strongly influence the seed survival and distribution of both native and invasive trees and shrubs into field cover types (Gill and Marks, 1991; Ostfeld *et al*., 1997). White-footed mice adjust their diets seasonally in response to food availability and dietary needs (Hamilton, 1941; Rose, 2011: Chapter 3), and studies have examined the effects of seed species, edge distance, and patch fragmentation on visitation rate to feeding or track stations. Some have even correlated white-footed mouse visitation rate to the feces of other species, such as raccoons (*Procyon lotor*) (Page *et al*., 1998).

While no studies found have modeled the environmental characteristics that contribute to increasing visitation rate to foraging stations, similar modeling has been done at Sherman traps (Silva *et al*., 2005). Since our stations don't impede movement by trapping the animal, and don't deter revisitation (Anthony *et al.*, 2005), it is important to study the variables that could lead to higher foraging activity to these stations, particularly by white-footed mice. Quantifying these variables could allow greater detection in both trapping and foraging studies, since both rely on feeding activity at trap stations. Increased trap visitation could influence the accuracy of markrecapture studies (Tull and Sears, 2007). Low trap success (defined as poor visitation rate), could cause studies to inadequately represent the diversity of a small mammal community, or their foraging preferences due to lack of time, field assistance or funds (Tull and Sears, 2007). Lastly, patterns of small mammal consumption and seed dispersal may be predictable based on observed habitat preferences.

The objectives of this study were to determine the relation between habitat variables and visitation rates at foraging stations. We also wanted to assess if the key variables remained consistent among cover types, and if not, which cover types were more closely related through habitat selection. Likewise, we evaluated if these variables differed between two spatial scales (100 m^2 and 400 m^2), and two seasons (summer and fall).

MATERIALS AND METHODS

STUDY SITE

This study was conducted in October 2009 and July 2010 at the National Park Service's Fort Necessity National Battlefield (FONE), located in Fayette County in southwestern Pennsylvania, U.S.A. (39º48'43" N, 84º 41'50" W). The average annual temperature is 9º C, the mean summer temperature is 22º C and the mean winter temperature is -3º C. Elevations range from 535 to 710 m with an average precipitation level of 119 cm (National Park Service, 1991). There are a variety of cover types present, including: field, wet meadow, oak-hardwood, conifer, wetland, and intermittently-flowing stream. Philo silt loams (deep, poor to moderately drained soils) characterize the low laying areas within the park. Brinkerton and Armagh silt loams, Cavode silt loams, and Gilpin channery silt loams (moderately deep, moderate to well drained soils) characterize the upland sites within the park (Kopas, 1973).

This study was conducted across three of the dominant cover types: forest interior, edge, and field. Interior forests were at least 100 m from any natural or made-made opening (Laurance 2001). The forest was comprised of mixed-hardwoods, and included species such as northern red oak (*Quercus rubra*), yellow poplar (*Liriodendron tulipifera*), and black cherry (*Prunus serotina*); with a ground covering of greenbrier (*Smilax* spp.), club-moss (*Lycopodium* spp.), and winter grape (*Vitis cinerea*). Edges were forested areas within 10 m of a park road, and again dominated by red oak, poplar, and cherry, often with an understory of eastern white pine (*Pinus strobus*) and Norway spruce (*Picea abies*). Ground cover consisted of greenbrier and shrub species, including Japanese barberry (*Berberis thunbergii*) and Morrow's honeysuckle (*Lonicera morrowii*). Fields were primarily early-successional areas, dominated by grasses (*Dactylis* spp., *Phleum* spp.), goldenrods (*Solidago* spp.), and shrub species, including Morrow's honeysuckle, Southern arrowwood (*Viburnum dentatum*), and sweet crabapple (*Malus coronaria*).

VISITATION RATE

In a companion study, we compared the selection for Morrow's honeysuckle fruits to the fruits of five native soft mast species across three cover types (forest, field, and edge), and two fruiting rounds at FONE (Rose, 2011: Chapter 3). Seven of each fruit species (the species used during each round varied based on availability in the environment) were placed in a random order within each of 60 wooden boxes (foraging stations), 20 in each cover type. Each day, stations were visited and the numbers of berries consumed were counted. All unconsumed fruits were removed, and fresh fruits of each species were added. Foraging occurrence was confirmed when possible by camera monitoring units and fluorescent tracking powder at the stations (Rose, 2011: Chapter 3). Although we had a sample size of 60, not all foraging stations yielded optimal data due to low visitation rate by mice. We calculated the visitation rate as the proportion of study days that foraging took place (number out of 14 days).

HABITAT SURVEYS

To help explain visitation rate we conducted a habitat survey at each station. Habitat variables were measured in four 10 x 10 m quadrats, centered around each station, forming a 400 m^2 survey area. Secondarily, habitat variables were measured in four 5 x 5 m quadrats nested within the larger plots, forming a 100 m^2 survey area (Fig. 1). These two spatial scales were used as the smaller represents nightly movements, and the larger represents potential home range movements (Wolff, 1985). Individual quadrats were used to facilitate accuracy, and values were averaged. We estimated the proportion (to the nearest 5%) of each quadrat's area covered by the following variables: grass (GS), sedge (SE), forb (FB) (broad-leaved herbaceous), fern (FN), moss (MS), shrub (SB), tree (TE), green (GN) (living plant material, composed of variables listed above), leaf litter (LF), log (LG), rock (RK), water (WR), road (RD) and dead plant material (DPM). Due to various strata, these proportions are overlapping, and therefore can sum to values over 100%. Logs were defined as being woody, non-rooted, horizontal, and having a diameter $\geq$ 7.5 cm (Tinker and Knight, 2001). Shrubs were defined as woody vegetation having a height of < 5 m, and multiple stems. Shrubs were also defined functionally; that is, when a thick, continuous patch of vine with a minimum height of 0.5 m (Holway, 1991) was observed it was considered a shrub (Roth, 1976). Only rocks at least 15 cm wide and 10 cm high were considered in quadrat proportions. To determine visual obstruction of vegetation, we obtained two robel pole measurements (Robel *et al*., 1970) at 0, 1, 5 and 10 m from the box in each cardinal direction: tallest vegetation touching the pole (tallest sight, TS) and first visible interval on the pole (first sight, FS). All measurements were recorded at 4 m from the pole and at a vertical height of 1 m. To estimate the total overhead canopy cover (CC) we used a spherical densiometer to take measurements at 0, 1, 5 and 10 m from the box in each cardinal direction. Slope (SL) of the ground was determined in each cardinal direction, at 5 m from each station, using a percent scale clinometer (Forestry Suppliers, Inc., Jackson, MS). Soil moisture (SM) was measured, at 5 m, on a scale of 0 (dry) to 10 (saturated) with a Soil Moisture Meter (Lincoln Irrigation, Inc., Lincoln, NE). The species of shrub located closest to the box was identified, and its distance from the box (SD), stem number (SN), height and width and perpendicular width were measured. The latter three variables were used to calculate conical shrub volume (SV) (Jiménez-Lobato and Valverde, 2006). The species of tree located closest to the box was identified, and its distance from the box (TD), diameter at breast height (DBH), height (TH), and crown width (CW) were measured. Variables that were measured once per station (SL, SM, SD, SN, SV, TD, DBH, TH, and CW) and applied to both spatial scales were included in candidate sets to determine their relative influence at each scale. The mean ± SE, with minimum and maximum ranges can be found for all variables measured in Appendix Ic.

DATA ANALYSIS

We used general linear modeling in Program R (version 2.11.1), with visitation rate as the response variable and a variety of predictor variables to create candidate models (Guisan and Zimmermann, 2000; Armstrong and Ewen, 2002; McKenzie *et al.*, 2007), based on Chamberlin's multiple working hypothesis approach (Chamberlin, 1931). This method avoids assumptions and biases of traditional stepping model selection procedures (Anderson *et al.*, 1994). We used the Gaussian family and identity link function due to normally distributed model errors, and calculated an adjusted Pearson's correlation coefficient (unbiased R^2) for each model.

Visitation rate was considered separately for each cover type sampled, without including cover type as a categorical variable in the model. We were interested in determining influential environmental variables within a cover type and less concerned with the effect of cover type on overall visitation rate. Additionally, each cover type had 10 *a priori* models, which were generally unique to that specific cover type. *A priori* models were based on peer-reviewed literature for each cover type and factors we believed would influence white-footed mouse foraging behavior. In instances where too few papers addressing environmental variables were found for white-footed mice, papers based on deer mice (*Peromyscus maniculatus*) were used instead. Variable correlation was checked visually using a scatterplot matrix and statistically with a Pearson's correlation test, and one of the correlated pair was removed if $| r | > 0.75$. Percent plot coverage by shrubs, and percent green were highly correlated in forest plots (0.79); since shrub was a common variable in literature models percent green was removed. Percent plot coverage by brush, and visual obscurity (first sight) were highly correlated in edge plots (0.83); since visual obscurity was a common variable in literature models percent brush was removed.

The following models for visitation rate (Y) were considered for edge cover types, at both spatial scales and each season:

1. Y = FS + SB + TE + TS, suggested by Manson *et al*. (1999) for white-footed mice;
2. Y = CC + FB + SB, suggested by Van Deusen and Kaufman (1977) for white-footed mice;
3. Y = CC + FB + FN + GS + SB, suggested by Manson and Stiles (1998) for white-footed mice;
4. Y = CC + FS + TS, suggested by Stancampiano and Schnell (2004) for white-footed mice;

5. Y = LF + LG + SB + TD;
6. Y = FS + SD + TD + TS;
7. Y = FN + MS + SB + SM;
8. Y = CC + LF;
9. Y = FS + RD + RK + SL; and
10. Y = SV + SN + DBH + TH + CW.

The following models were considered for field cover types, at both spatial scales and each season:

11. Y = FS + GS + RK + TS, suggested by Pearson *et al*. (2001), for deer mice;
12. Y = FS + FB + GS + SB + TS, suggested by Manson and Stiles (1998) for white-footed mice;
13. Y = FB + GS, suggested by Dooley and Bowers (1996) for white-footed mice;
14. Y = DPM + FS + GN + TS, suggested by Morris (1979) and Kantak (1996) for white-footed mice;
15. Y = FB + GS + SB;
16. Y = GN + SL + SM;
17. Y = FB + RK + SM;
18. Y = FS + SD + TS + TD;
19. Y = FB + GS + RK; and
20. Y = SV + SN + DBH + TH + CW.

The following models were considered for forest cover types, at both spatial scales and each season:

21. Y = FB + LG + SB, suggested by Bellows *et al*. (2001) for generalist species including white-footed mice;
22. Y = CC + RK + SB, suggested by Johnston and Anthony (2008) and Coppeto et al. (2006) for deer mice;
23. Y = FN + FB + GS + SB + TE, suggested by Yahner (1986) for white-footed mice;
24. Y = LF + RK + TS, suggested by Kaminski *et al*. (2007) for white-footed mice and deer mice;
25. Y = DPM + LG + RK + SL;
26. Y = LF + LG + SB + TD;
27. Y = FN + LG + MS + SM;
28. Y = SD + TS + TD;
29. Y = FN + LG + TE; and
30. Y = SV + SN + DBH + TH + CW.

We selected models based on Akaike's Information Criterion, corrected for small sample bias and overfitting (AIC_c) (Burnham and Anderson, 2002). The best model is that with the lowest AIC_c value indicating the model with the least information lost. We calculated delta AIC_c (Δ_i = AIC_c lowest - AIC_{ci}) and Akaike weights (w_i) for each model. To determine the best candidate model given the data, we used Akaike weights, and the relative support for other models is indicated by their delta AIC_c (Burnham and Anderson, 1998). We considered alternative models to have tentative support if their delta AIC_c was < 2. Therefore, all models with Δ_i < 2 were averaged (with coefficients weighted based on w_i) to generate a final approximating model (Burnham and Anderson, 2002).

RESULTS

The mean values of the most influential variables, those with substantial support for predicting visitation rate to foraging stations, varied widely among cover types, seasons, and scales (Table 1). Visitation rate was best predicted at edge foraging stations using percent canopy cover and leaf cover as variables at both spatial scales in the summer (Table 2). These variables were seen at both scales in the fall (Table 2); however, the final model for 400 m^2 was based on two models with substantial support, and had first sight and tallest sight as added variables of importance. Therefore, model averaging was applied to obtain the following final model for 400 m^2 (Table 3):

$$Y_{edge\ (400m^2,\ fall)} = 0.094 + 0.008(CC) - 0.230(LF) + 0.028(FS) + 0.002(TS).$$

Visitation rate was best predicted at field foraging stations using percent forb, grass, rock, and soil moisture as variables at both spatial scales in the summer (Table 2). Two models were averaged to obtain both of the final models (Table 3):

$$Y_{field\ (100m^2,\ summer)} = 0.150 - 0.118(FB) + 0.032(GS) + 7.349(RK) + 0.020(SM);$$

$$Y_{field\ (400m^2,\ summer)} = 0.170 - 0.188(FB) + 0.047(GS) + 6.066(RK) + 0.058(SM).$$

For field foraging stations in the fall, visitation rate was best predicted by percent forb, grass, rock and shrub at 100 m^2 (Table 2). These same variables were found in the final model for 400 m^2, as well as first sight and tallest sight. Three models were averaged to obtain each of the final models (Table 3):

$$Y_{field\ (100m^2,\ fall)} = 1.013 - 0.442(FB) - 0.475(GS) + 0.931(RK) - 0.087(SB);$$

$$Y_{field\ (400m^2,\ fall)} = 0.885 - 0.287(FB) - 0.484(GS) + 2.314(RK) - 0.075(SB) - 0.001(FS) - 0.0002(TS).$$

For foraging stations in the forest during the summer, visitation rate was best predicted by percent forb, log, and shrub at 100 m^2 (Table 2). These same variables were found in the final model for 400 m^2, as well as fern and tree. Two models were averaged to obtain a final model for 400 m^2 (Table 3). The final model is as follows:

$$Y_{forest\ (400m^2,\ summer)} = 0.451 - 1.805(FB) + 6.000(LG) + 0.060(SB) + 0.570(FN) + 0.156(TE).$$

Visitation rate was best predicted at forest foraging stations using percent log, shrub, fern, tree, leaf and distance to the closest tree as variables at both spatial scales in the fall (Table 2). Both of the final models were averaged from two models with substantial support (Table 3):

$$Y_{forest\ (100m^2,\ fall)} = 0.580 + 4.920(LG) + 0.102(SB) + 1.185(FN) + 0.382(TE) - 0.220(LF) - 0.031(TD);$$

$$Y_{forest\ (400m^2,\ fall)} = 0.358 + 4.558(LG) + 0.062(SB) + 2.437(FN) + 0.857(TE) - 0.062(LF) - 0.013(TD).$$

DISCUSSION

SEASONAL, INFLUENTIAL ENVIRONMENTAL VARIABLES

The model variables that were best at predicting visitation rate to foraging stations likely reflect an overall need to reduce time spent foraging, and risk of predation from respective predators in the different cover types. A number of factors influence foraging behavior in small mammals; however, optimal foraging and predation risk are often cited (Ebersole and Wilson, 1980; Manson and Stiles, 1998). Increasing visitation rate at edge stations, across both scales and seasons, was best predicted by an increase in canopy cover and a decrease in leaf cover. When leaf cover is reduced, it is likely that white-footed mice are more effective at preying on insects or locating fallen fruits and seeds (Pearson *et al.*, 2001). Additionally, nocturnal predators of small mammals often hunt using sound, and mice traveling through paths heavily dominated by leaf-litter could be more easily detected (Roche *et al*., 1999). The edge cover type had a dense understory of deciduous saplings, which contributed to the canopy cover measurements. Avian predators such as barred owls (*Strix varia*), common in forests and edge cover types (Nicholls and Warner, 1972), also use vision-based cues when hunting (Conrader and Conrader, 1965). According to model variables, white-footed mice navigated habitat with high canopy cover to reach foraging stations, which was largely composed of saplings that provided visual obscurity without providing proper perches for owls.

Many of these conclusions can be drawn for the forest models as well. The lack of the canopy cover parameter in forest models may be attributed to the lack of a sapling understory due to differing light regimes. Increasing visitation rate to foraging stations, across both scales and seasons, was best predicted by an increase in log plot coverage and shrub plot coverage. Multiple studies have cited these variables as important to habitat selection in this species (Kaufman *et al*., 1983; Manson *et al*., 1999). Ferns and trees (percent cover and distance to nearest tree), among the only persistent cover-providing vegetation during colder seasons, were more important in fall than summer models. Forbs within the forest were rarely large enough to provide cover from predators, and instead may have negatively influence foraging time (Pearson *et al*. 2001). As in the edge, navigation through leaf cover would increase risk of predation by owls. The noise caused by leaves was reduced in the summer due to decomposition and increased precipitation, likely leading to the absence of the leaf parameter in summer models and its presence in fall models (when leaves had freshly fallen).

Increasing visitation rate to field foraging stations, across both scales and seasons, was best predicted by an increase in rock plot coverage and a decrease in forb coverage (as seen in the forest). A number of studies have found that *Peromyscus* spp. tend to select open sites in fields (Korpimaki *et al*., 1996; Elliott *et al*., 1997; Pearson *et al*., 2001). The use of open areas could indicate that predation risk from mammalian predators such as weasels (Korpimaki *et al*. 1996) or snakes such as black racers (*Coluber constrictor*) (Klimstra, 1959), all of which favor heavy ground cover, exceeds that from avian predators. Since structural elements such as logs were lacking in the fields, rocks provided escape cover in an otherwise herbaceous dominated cover type. These rocks were small enough to conceal small mammals without hiding predators.

Percent coverage of grass was seen in all field models across seasons and scales. Grass was positively associated with visitation rate in the summer, likely due to providing grass seed as a source of food. However, in the fall grass was negatively associated with visitation rate, likely because it provided cover for predators without providing a food source. This is an important change in a habitat characteristic to note, which could affect visitation rate, depending on the time frame of a research project. As forbs and grasses decreased in the fall, shrubs appeared to be more important to predators as cover, which would explain why this parameter is only important in fall models.

SHRUB SPECIES IMPACT ON VISITATION

Shrub species had a notable impact on visitation rate. Visitation rate was negatively associated with shrub coverage in the field stations, but positively associated in forest stations. The most common shrub species in the field was exotic Morrow's honeysuckle, which was found to be the closest shrub 85% of the time. Honeysuckle has an open understory, due to its branch architecture, which is navigable by larger predators (Schmidt and Whelan, 1999). In contrast, native common greenbrier (*Smilax rotundifolia*) was the closest shrub 95% of the time. Greenbrier was classified as a shrub based on a functional interpretation, as it occurred in large thickets. Unlike honeysuckle, this species provides a dense cover of protective thorns.

Additionally, because honeysuckle was found to be less nutritious and a lesser preferred food item when compared to native species at FONE (Rose, 2011: Chapter 3), marginal value theorem (Charnov, 1976) could also explain lower visitation rate to stations near this shrub in the field. Low food abundance and nutritional content may reduce the energy an animal gains, and therefore increase foraging time (Charnov, 1976). Given that an animal loses its profit if it continues to feed in such an area, it is likely it will not forage near dense areas of honeysuckle. This may be a more plausible explanation than predator avoidance as other studies have found that white-footed mice did not always avoid areas of high honeysuckle cover (Edalgo *et al*., 2009; Dutra *et al*., 2011).

SPATIAL INFLUENCE ON VISITATION

Although studies that only consider one spatial scale may overlook important aspects of the habitat (Dueser and Shugart, 1978), there is large discrepancy in what is considered microhabitat, macrohabitat, and landscape scales (Stapp, 1997; Bellows *et al*., 2001; Silva *et al*., 2005; Trainor *et al*., 2007). There is also discrepancy in what is considered a microhabitat verses a macrohabitat feature. Typically, studies define microhabitat and macrohabitat scales, and then examine different variables within those defined limits. Silva *et al*. (2005) defined microhabitat by examining 1 x 1 m plots and recording percent ground cover. Likewise, macrohabitat was examined at 5 x 5 m plots and variables included tree height and diameter at breast height. We attempted to measure variables (whether they be considered micro- or macrohabitat) consistently at both 100 m^2 and 400 m^2 and determine which variables were most important at predicting increasing visitation rate at these two scales.

Models for 400 m^2 surveys generally contained more variables than 100 m^2 surveys. As we believe visitation rate is largely related to predation pressure, the 400 m^2 models should contain variables linked to vulnerability approaching and leaving the foraging stations. In the forest and edge stations, high structural complexity around the box would provide protection from aerial, vision-based predators. The foraging stations themselves may provide cover, and therefore structural complexity variables are less likely to manifest at $\leq$ 5 m from the stations. In forest and edge models, variables such as fern and tallest sight are positively related to visitation rate at distances of $\leq$ 10 m. In field stations, where scent-based predators such as snakes predominate, structural complexity in the vicinity of the stations may obscure the presence of such predators. Variables such as first sight and tallest sight were negatively related to visitation rate in the fall at 400 m^2.

CONCLUSIONS

Although white-footed mice may be considered habitat generalists, this does not mean that they don't specialize in particular habitat features of a certain cover type; especially when travel routes are important, such as to and from a known foraging site. When comparing cover types, we found that edge and forest were most similar in the variables that predicted increasing visitation rate. These two cover types required areas of high structural complexity. In contrast, the field cover type experienced low levels of visitation in areas of high structural complexity. Given these results, it is likely that the field cover type is the most vulnerable of the three to invasions from exotic plant species. The highly used stations were characterized by large amounts of open area, which could be ideal for seed germination. These areas could be at risk from both forgotten cached invasive seeds (Abbot and Quink, 1970) or even post-gut viability of invasive seeds (Williams *et al*., 2000). Spatial scales within a season were nearly identical, other than 400 m^2 often requiring model averaging to include more variables. This difference at the large scale was attributed to the addition of, or decrease in structural variables that could affect predator risk to and from stations. When comparing differences across measured spatial scales or seasons, seasons showed the largest differences within a cover type in regards to the best model based on the data. This difference is especially important to note as white-footed mice are known to change their diets seasonally and we found that they also changed their habitat use based on season. By following these seasonal differences, as well as other microsites documented here, it may be possible to increase detection of foraging preference to foraging stations based on facilitating increasing visitation rate.

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Table 1. Mean (± SE) for the environmental variables found in the best models for predicting visitation rate to foraging stations at Fort Necessity National Battlefield, Pennsylvania, USA in October – November 2009 (Fall) and July – August 2010 (Summer). All values were averaged over 20 stations within each cover type (total *N* = 60).

	Edge								Field			
	Summer				Fall				Summer			
	100		400		100		400		100		400	
Variables	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Canopy cover (%)	79.93	3.22	79.11	3.29	31.73	2.91	32.04	2.97	0.25	0.25	1.06	0.68
Fern cover (%)	7.84	2.76	9.15	2.73	4.35	2.17	6.44	2.55	0.00	0.00	0.00	0.00
First sight (cm)	5.99	0.48	6.90	1.03	6.31	0.45	6.56	0.50	23.88	5.14	31.19	7.13
Forb cover (%)	17.41	4.00	15.70	3.28	8.55	4.36	9.21	4.28	70.81	6.35	69.77	6.26
Grass cover (%)	19.32	3.53	18.73	3.27	10.57	5.66	10.29	5.37	61.08	7.52	60.50	7.37
Green cover (%)	47.04	6.42	43.38	5.71	19.79	3.72	21.54	3.58	94.19	0.91	93.25	1.33
Leaf cover (%)	28.93	4.15	29.11	4.11	72.00	7.04	70.31	6.83	3.88	1.35	3.56	1.26
Log cover (%)	2.27	0.37	1.88	0.28	4.43	1.56	4.51	1.39	0.04	0.04	0.00	0.00
Rock cover (%)	1.36	0.18	1.40	0.19	1.28	0.18	1.26	0.18	3.00	0.62	3.24	0.60
Shrub cover (%)	9.29	2.48	8.01	1.88	11.77	2.26	15.20	2.99	17.34	6.09	20.30	5.61
Soil moisture (%)	1.31	0.24	1.31	0.24	1.36	0.27	1.36	0.27	1.90	0.21	1.90	0.21
Tallest sight (cm)	22.12	3.00	25.69	3.83	16.81	4.59	19.31	6.24	76.25	2.90	95.31	4.43
Tree cover (%)	3.48	0.20	3.37	0.26	3.72	1.17	3.94	1.12	0.15	0.08	0.54	0.21
Tree distance (closest) (m)	0.94	0.21	0.94	0.21	1.49	0.61	1.49	0.61	15.19	2.79	15.19	2.79

Table 1. Continued

	Field				Forest							
	Fall				Summer				Fall			
	100		400		100		400		100		400	
Variables	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Canopy cover (%)	1.00	1.00	2.35	1.62	81.31	2.96	81.25	1.80	31.87	4.53	29.43	2.83
Fern cover (%)	0.11	0.08	0.20	0.16	3.57	0.79	3.59	0.70	1.79	0.46	2.10	0.49
First sight (cm)	7.69	5.74	10.74	9.11	5.00	0.00	5.00	0.00	6.25	0.36	6.25	0.36
Forb cover (%)	46.02	5.61	44.85	5.26	2.67	0.57	2.61	0.53	0.67	0.24	0.67	0.24
Grass cover (%)	47.63	5.71	47.54	5.58	11.45	0.34	11.63	0.38	1.05	0.31	1.04	0.30
Green cover (%)	28.61	3.22	27.51	3.22	37.87	5.36	39.06	5.27	21.41	4.70	22.61	5.02
Leaf cover (%)	3.96	1.44	5.00	1.32	51.00	3.17	50.88	3.13	84.86	3.09	84.36	3.13
Log cover (%)	0.00	0.00	0.09	0.09	4.52	0.76	4.28	0.68	4.62	0.71	4.89	0.78
Rock cover (%)	5.39	0.87	5.95	0.87	3.86	0.72	3.67	0.53	1.98	0.25	1.99	0.24
Shrub cover (%)	21.76	6.78	28.98	7.00	26.21	4.50	26.56	4.43	16.54	3.56	16.62	3.63
Soil moisture (%)	1.90	0.21	1.90	0.21	0.29	0.11	0.29	0.11	23.25	0.06	23.25	0.06
Tallest sight (cm)	42.81	0.68	70.00	1.92	13.43	2.45	30.62	4.21	17.43	3.39	20.00	2.87
Tree cover (%)	0.23	0.16	0.65	0.30	2.49	0.29	2.81	0.19	4.66	0.16	4.61	0.15
Tree distance (closest) (m)	15.19	2.83	15.19	2.83	1.71	0.35	1.71	0.35	2.61	0.58	2.61	0.58

Table 2. *A priori* models for visitation rate to foraging stations in edges, fields, and forests at Fort Necessity National Battlefield, Pennsylvania, USA in October – November 2009 (Fall) and July – August 2010 (Summer) at 100 m^2 and 400 m^2 scales. The best model is chosen by Akaike's Information Criterion for small sample sizes (AICc), with small values indicating a better model fit. Model variables include: percent plot coverage of living plants (GN), grass (GS), forb (FB), fern (FN), moss (MS), shrub (SB), tree (TE), leaf (LF), log (LG), rock (RK), road (RD) and dead plants (DPM); visual obscurity measurements (tallest sight, TS and first sight, FS), canopy cover (CC); ground slope (SL), soil moisture (SM); closest shrub (SD) and tree (TD) distances to box, closest shrub stems (SN) and volume (SV); closest tree height (TH), diameter at breast height (DBH), and crown width (CW).

Model Cover Type (Scale, Season) Model Structure	Model No.	K [a]	AIC_c	Δ_i [b]	w_i [c]	Adj. R^2 [d]
Y_{edge} (100m^2, summer)						
Y = 0.659 - 1.029(LF) + 0.455(CC)	8	4	-1.550	0.000	0.974	0.5240
Y = 1.066 + 0.362(SB) - 0.055(TD) + 0.157(LG) - 1.131(LF)	5	6	6.362	7.912	0.019	0.4573
Y = 0.566 - 0.021(SD) - 0.117(TD) + 0.045(FS) + 0.004(TS)	6	6	9.124	10.674	0.005	0.3769
Y = -0.167 + 0.809(CC) + 0.005(FS) + 0.009(TS)	4	5	10.548	12.098	0.002	0.2271
Y = 0.728 + 0.109(SM) + 0.846(SB) - 7.356(MS) + 0.127(FN)	7	6	14.406	15.956	0.000	0.1886
Y = 0.159 + 0.042(SB) + 0.247(FB) + 0.649(CC)	2	5	15.912	17.462	0.000	-0.0106
Y = 0.700 + 2.060(RD) - 7.172(RK) - 0.014(SL) + 0.062(FS)	9	6	17.246	18.796	0.000	0.0649
Y = 0.362 + 0.372(SB) + 1.885(TE) + 0.007(TS) + 0.174(FS)	1	6	18.304	19.854	0.000	0.0140
Y = -0.133 + 0.882(CC) + 0.008(SB) + 0.251(FB) + 0.224(GS) + 0.827(FN)	3	7	20.743	22.293	0.000	0.0646
Y = 1.419 + 0.001(SN) - 0.119(SV) + 0.001(DBH) - 0.032(TH) - 0.037(CW)	10	7	20.875	22.425	0.000	0.0584
Y_{edge} (400m^2, summer)						
Y = 0.726 - 1.055(LF) + 0.387(CC)	8	4	-0.753	0.000	0.936	0.5046
Y = 1.077 + 0.562(SB) - 0.063(TD) - 0.378(LG) - 1.138(LF)	5	6	5.772	6.524	0.036	0.4731
Y = -0.017 + 0.625(CC) - 0.001(FS) + 0.001(TS)	4	5	7.536	8.289	0.015	0.3352

Table 2. Continued

Model Cover Type (Scale, Season) Model Structure	Model No.	K [a]	AIC_c	Δ_i [b]	w_i [c]	Adj. R^2 [d]
Y = 0.661 - 0.018(SD) - 0.034(TD) - 0.002(FS) + 0.007(TS)	6	6	9.335	10.088	0.006	0.3703
Y = 0.814 + 0.074(SM) + 1.983(SB) - 11.372(MS) + 0.066(FN)	7	6	9.985	10.738	0.004	0.3495
Y = 0.644 + 1.749(RD) - 5.848(RK) - 0.014(SL) + 0.018(FS)	9	6	11.897	12.650	0.002	0.2843
Y = 0.374 + 0.590(SB) + 2.076(TE) + 0.001(TS) - 0.001(FS)	1	6	13.580	14.333	0.001	0.2214
Y = 0.308 + 0.327(SB) + 0.175(FB) + 0.460(CC)	2	5	16.960	17.712	0.000	-0.0649
Y = 1.419 + 0.001(SN) - 0.119(SV) + 0.001(DBH) - 0.032(TH) - 0.037(CW)	10	7	20.875	21.628	0.000	0.0584
Y = 0.239 + 0.458(CC) + 0.194(SB) - 0.018(FB) + 0.506(GS) + 0.178(FN)	3	7	24.301	25.054	0.000	-0.1175
Y_{edge} (100m^2, fall)						
Y = 0.662 - 0.575(LF) + 0.005(CC)	8	4	1.585	0.000	0.927	0.4947
Y = 1.018 + 0.162(SB) - 0.032(TD) + 0.697(LG) - 0.836(LF)	5	6	7.008	5.422	0.062	0.4914
Y = 0.159 + 0.008(CC) - 0.016(FS) + 0.006(TS)	4	5	12.305	10.720	0.004	0.2343
Y = -0.053 + 0.745(SB) + 0.341(FB) + 0.011(CC)	2	5	12.635	11.050	0.004	0.2216
Y = 0.246 + 0.171(SM) - 0.721(SB) + 0.476(MS) + 0.099(FN)	7	6	13.853	12.268	0.002	0.2838
Y = 0.088 + 0.007(CC) + 0.136(SB) - 0.147(FB) + 0.698(GS) + 0.928(FN)	3	7	14.897	13.311	0.001	0.3663
Y = 0.473 + 0.001(SD) + 0.022(TD) - 0.035(FS) + 0.008(TS)	6	6	18.710	17.124	0.000	0.0870
Y = 0.460 + 0.163(SB) - 0.137(TE) + 0.008(TS) - 0.029(FS)	1	6	19.666	18.080	0.000	0.0423
Y = 0.270 - 0.041(RD) + 3.820(RK) + 0.006(SL) + 0.007(FS)	9	6	24.912	23.326	0.000	-0.2449
Y = 0.219 - 0.008(SN) + 0.045(SV) + 0.001(DBH) - 0.015(TH) + 0.050(CW)	10	7	25.933	24.348	0.000	-0.1003
Y_{edge} (400m^2, fall)						
Y = 0.621 - 0.567(LF) + 0.006(CC)	8	4	1.585	0.000	0.490	0.4947
Y = -0.268 + 0.009(CC) + 0.048(FS) + 0.004(TS)	4	5	1.658	0.073	0.472	0.5504
Y = 1.009 + 0.241(SB) - 0.034(TD) + 0.243(LG) - 0.836(LF)	5	6	8.791	7.206	0.013	0.4440
Y = 0.149 - 0.695(SB) - 1.025(TE) + 0.007(TS) + 0.043(FS)	1	6	8.947	7.363	0.012	0.4396
Y = -0.100 + 0.751(SB) + 0.303(FB) + 0.012(CC)	2	5	10.158	8.573	0.007	0.3123
Y = 0.024 + 0.004(SD) + 0.015(TD) + 0.038(FS) + 0.006(TS)	6	6	11.040	9.455	0.004	0.3778

Table 2. Continued

Model Cover Type (Scale, Season) Model Structure	Model No.	K [a]	AIC_c	Δ_i [b]	w_i [c]	Adj. R^2 [d]
Y = 0.237 + 0.174(SM) - 0.399(SB) + 0.159(MS) - 0.047(FN)	7	6	14.904	13.319	0.001	0.2452
Y = -0.017 + 0.008(CC) + 0.421(SB) + 0.014(FB) + 0.515(GS) + 0.691(FN)	3	7	15.128	13.544	0.001	0.3590
Y = 0.082 - 0.889(RD) - 0.040(RK) - 0.001(SL) + 0.062(FS)	9	6	19.748	18.163	0.000	0.0383
Y = 0.219 - 0.008(SN) + 0.045(SV) + 0.001(DBH) - 0.015(TH) + 0.050(CW)	10	7	25.933	24.349	0.000	-0.1003
Y_{field} (100m^2, summer)						
Y = 0.112 + 7.535(RK) - 0.086(FB) + 0.035(SM)	17	5	-10.431	0.000	0.546	0.7104
Y = 0.199 + 7.105(RK) + 0.073(GS) - 0.159(FB)	19	5	-9.926	0.505	0.424	0.7030
Y = 0.098 + 7.796(RK) + 0.001(FS) - 0.001(TS) + 0.068(GS)	11	6	-4.494	5.937	0.028	0.6626
Y = 0.535 + 0.340(GS) - 0.519(FB)	13	4	1.708	12.140	0.001	0.4007
Y = 0.578 + 0.326(GS) - 0.552(FB) - 0.063(SB)	15	5	5.236	15.667	0.000	0.3661
Y = -0.813 + 0.046(SL) + 0.156(SM) + 0.823(GN)	16	5	10.995	21.427	0.000	0.1546
Y = -3.152 + 5.48(DPM) + 4.301(GN) - 0.010(FS) - 0.005(TS)	14	6	11.012	21.443	0.000	0.2675
Y = 0.297 - 0.035(SB) + 0.580(GS) - 0.486(FB) - 0.002(TS) + 0.009(FS)	12	7	11.783	22.214	0.000	0.3607
Y = 1.015 - 0.028(SD) + 0.004(TD) - 0.001(FS) - 0.007(TS)	18	6	13.872	24.303	0.000	0.1549
Y = 0.515 - 0.025(SN) + 0.001(SV) - 0.008(DBH) + 0.011(TH) + 0.023(CW)	20	7	21.143	31.575	0.000	-0.0411
Y_{field} (400m^2, summer)						
Y = 0.108 + 6.533(RK) - 0.138(FB) + 0.081(SM)	17	5	-3.994	0.000	0.642	0.5966
Y = 0.338 + 4.797(RK) + 0.176(GS) - 0.321(FB)	19	5	-1.993	1.999	0.236	0.5541
Y = 0.529 + 0.364(GS) - 0.536(FB)	13	4	0.150	4.144	0.081	0.4402
Y = 0.314 + 3.862(RK) + 0.003(FS) - 0.004(TS) + 0.314(GS)	11	6	2.995	6.989	0.019	0.5047
Y = 0.531 + 0.364(GS) - 0.537(FB) - 0.002(SB)	15	5	3.769	7.763	0.013	0.4053
Y = 0.635 - 0.096(SB) + 0.463(GS) - 0.305(FB) - 0.004(TS) + 0.002(FS)	12	7	4.877	8.871	0.008	0.5430
Y = 0.283 + 0.044(SL) - 0.166(SM) - 0.360(GN)	16	5	10.986	14.980	0.000	0.1537
Y = 0.861 - 0.007(SD) + 0.001(TD) - 0.002(FS) - 0.004(TS)	18	6	12.908	16.901	0.000	0.1869
Y = 0.442 + 0.013(DPM) + 0.441(GN) - 0.003(FS) - 0.004(TS)	14	6	13.299	17.292	0.000	0.1799
Y = 0.515 - 0.025(SN) + 0.001(SV) - 0.008(DBH) + 0.011(TH) + 0.023(CW)	20	7	21.143	25.137	0.000	-0.0411

Table 2. Continued

Model Cover Type (Scale, Season) Model Structure	Model No.	K [a]	AIC_c	Δ_i [b]	w_i [c]	Adj. R^2 [d]
Y field (100m^2, fall)						
Y = 1.063 - 0.460(GS) - 0.498(FB)	13	4	-3.008	0.000	0.342	0.5073
Y = 0.812 + 2.361(RK) - 0.400(GS) - 0.292(FB)	19	5	-2.924	0.084	0.328	0.5614
Y = 1.210 - 0.576(GS) - 0.575(FB) - 0.260(SB)	15	5	-2.589	0.419	0.278	0.5539
Y = 0.675 + 3.448(RK) - 0.323(FB) - 0.051(SM)	17	5	1.612	4.620	0.034	0.4497
Y = 0.621 + 3.435(RK) + 0.001(FS) + 0.001(TS) - 0.432(GS)	11	6	3.570	6.578	0.013	0.4746
Y = 1.085 - 0.277(SB) - 0.517(GS) - 0.617(FB) + 0.001(TS) + 0.011(FS)	12	7	5.435	8.443	0.005	0.5157
Y = 0.178 + 0.015(SN) - 0.009(SV) + 0.002(DBH) + 0.044(TH) - 0.022(CW)	20	7	12.034	15.042	0.000	0.3263
Y = 0.621 + 0.022(SL) + 0.017(SM) - 0.335(GN)	16	5	15.207	18.215	0.000	-0.0860
Y = 0.125 + 0.026(SD) + 0.009(TD) + 0.034(FS) + 0.001(TS)	18	6	16.915	19.923	0.000	-0.0239
Y = 0.707 - 0.084(DPM) - 0.472(GN) + 0.014(FS) - 0.001(TS)	14	6	19.385	22.393	0.000	-0.1585
Y field (400m^2, fall)						
Y = 0.715 + 3.108(RK) - 0.382(GS) - 0.232(FB)	19	5	-6.795	0.000	0.395	0.6385
Y = 1.278 - 0.629(GS) - 0.633(FB) - 0.279(SB)	15	5	-5.558	1.237	0.213	0.6155
Y = 0.793 + 3.308(RK) - 0.004(FS) - 0.001(TS) - 0.534(GS)	11	6	-5.219	1.576	0.180	0.6441
Y = 1.094 - 0.502(GS) - 0.537(FB)	13	4	-4.733	2.062	0.141	0.5491
Y = 0.533 + 4.678(RK) - 0.188(FB) - 0.059(SM)	17	5	-3.154	3.641	0.064	0.5664
Y = 1.352 - 0.162(SB) - 0.721(GS) - 0.512(FB) - 0.002(TS) - 0.001(FS)	12	7	1.326	8.121	0.007	0.6056
Y = 0.178 + 0.015(SN) - 0.009(SV) + 0.002(DBH) + 0.044(TH) - 0.022(CW)	20	7	12.034	18.830	0.000	0.3263
Y = 0.552 + 0.025(SL) + 0.011(SM) - 0.076(GN)	16	5	15.826	22.621	0.000	-0.1201
Y = 0.640 - 0.006(SD) + 0.007(TD) - 0.002(FS) - 0.001(TS)	18	6	18.588	25.383	0.000	-0.1133
Y = 0.936 - 0.196(DPM) - 0.366(GN) - 0.005(FS) - 0.001(TS)	14	6	19.345	26.140	0.000	-0.1562
Y forest (100m^2, summer)						
Y = 0.464 + 0.026(SB) - 2.041(FB) + 5.711(LG)	21	5	-28.740	0.000	0.615	0.8110
Y = 0.360 + 4.488(LG) + 1.138(FN) + 2.301(TE)	29	5	-26.644	2.096	0.216	0.7901
Y = 0.379 - 0.004(SL) + 5.178(LG) + 0.600(DPM) + 1.455(RK)	25	6	-24.427	4.313	0.071	0.8069
Y = 0.743 + 0.304(SB) - 0.025(TD) + 5.122(LG) - 0.375(LF)	26	6	-24.315	4.425	0.067	0.8058

Table 2. Continued

Model Cover Type (Scale, Season) Model Structure	Model No.	K [a]	AIC_c	Δ_i [b]	w_i [c]	Adj. R^2 [d]
Y = 0.448 - 0.016(SM) + 4.333(LG) - 0.368(MS) + 1.186(FN)	27	6	-22.743	5.997	0.031	0.7792
Y = -0.029 + 0.122(SB) + 5.963(TE) - 0.871(FB) + 3.170(FN) + 3.661(GS)	23	7	1.093	29.833	0.000	0.3893
Y = 0.749 + 2.051(RK) - 0.005(TS) - 0.195(LF)	24	5	2.127	30.867	0.000	0.1154
Y = 0.359 + 0.256(CC) + 2.812(RK) + 0.057(SB)	22	5	2.394	31.134	0.000	0.1035
Y = 0.880 - 0.053(SD) - 0.044(TD) - 0.007(TS)	28	5	2.810	31.550	0.000	0.0847
Y = 0.432 - 0.011(SN) + 0.451(SV) + 0.001(DBH) + 0.007(TH) + 0.009(CW)	30	7	12.161	40.901	0.000	-0.1936
Y_{forest} (400m^2, summer)						
Y = 0.473 + 0.082(SB) - 2.482(FB) + 6.413(LG)	21	5	-27.833	0.000	0.546	0.8034
Y = 0.392 + 4.897(LG) + 2.089(FN) + 0.571(TE)	29	5	-26.852	0.981	0.335	0.7832
Y = 0.427 - 0.025(SM) + 4.400(LG) - 0.342(MS) + 1.997(FN)	27	6	-23.730	4.103	0.070	0.7898
Y = 0.314 - 0.001(SL) + 5.711(LG) + 1.252(DPM) + 1.523(RK)	25	6	-21.997	5.836	0.030	0.7708
Y = 0.725 + 0.342(SB) - 0.019(TD) + 5.796(LG) - 0.370(LF)	26	6	-21.131	6.702	0.019	0.7606
Y = -0.150 + 0.768(CC) + 4.521(RK) + 0.081(SB)	22	5	-1.643	26.190	0.000	0.2674
Y = 0.499 + 4.970(RK) - 0.001(TS) + 0.014(LF)	24	5	0.073	27.906	0.000	0.2018
Y = 0.262 + 0.128(SB) + 3.690(TE) - 0.700(FB) + 4.816(FN) + 0.921(GS)	23	7	2.652	30.485	0.000	0.3398
Y = 0.795 - 0.027(SD) - 0.040(TD) - 0.001(TS)	28	5	5.510	33.343	0.000	-0.0476
Y = 0.432 - 0.011(SN) + 0.451(SV) + 0.001(DBH) + 0.007(TH) + 0.009(CW)	30	7	12.161	39.994	0.000	-0.1936
Y_{forest} (100m^2, fall)						
Y = 0.725 + 0.147(SB) - 0.044(TD) + 5.004(LG) - 0.317(LF)	26	6	3.528	0.000	0.391	0.4707
Y = 0.251 + 4.731(LG) + 3.862(FN) + 1.246(TE)	29	5	4.247	0.720	0.273	0.3662
Y = 0.799 - 0.026(SD) - 0.050(TD) - 0.001(TS)	28	5	5.690	2.162	0.133	0.3520
Y = 0.335 + 0.205(SB) - 2.140(FB) + 5.233(LG)	21	5	6.447	2.919	0.091	0.2925
Y = 0.126 + 0.016(SL) + 4.293(LG) + 0.094(DPM) + 3.721(RK)	25	6	7.065	3.537	0.067	0.3683
Y = 0.270 + 0.033(SM) + 4.974(LG) + 0.149(MS) + 4.246(FN)	27	6	7.951	4.423	0.043	0.3397
Y = 0.613 + 7.616(RK) + 0.003(TS) - 0.263(LF)	24	5	14.402	10.874	0.002	-0.0531
Y = 0.489 - 0.183(CC) + 6.113(RK) + 0.270(SB)	22	5	14.494	10.966	0.002	-0.0579

Table 2. Continued

Model Cover Type (Scale, Season) Model Structure	Model No.	K [a]	AIC_c	Δ_i [b]	w_i [c]	Adj. R^2 [d]
Y = 0.254 + 0.148(SB) + 6.271(TE) - 1.734(FB) + 5.937(FN) - 1.907(GS)	23	7	21.471	17.944	0.000	-0.1013
Y = 0.604 + 0.030(SN) - 0.051(SV) - 0.001(DBH) + 0.001(TH) - 0.003(CW)	30	7	24.106	20.579	0.000	-0.2695
$Y_{forest\ (400m^2,\ fall)}$						
Y = 0.244 + 4.474(LG) + 3.596(FN) + 1.265(TE)	29	5	3.454	0.000	0.372	0.3908
Y = 0.598 + 0.196(SB) - 0.039(TD) + 4.734(LG) - 0.193(LF)	26	6	4.328	0.874	0.240	0.4491
Y = 0.815 - 0.026(SD) - 0.049(TD) - 0.001(TS)	28	5	5.599	2.145	0.127	0.3549
Y = 0.328 + 0.216(SB) - 1.411(FB) + 4.951(LG)	21	5	5.798	2.344	0.115	0.3151
Y = 0.119 + 0.015(SL) + 3.710(LG) + 0.071(DPM) + 5.904(RK)	25	6	6.507	3.053	0.081	0.3857
Y = 0.258 + 0.024(SM) + 4.757(LG) + 0.159(MS) + 3.973(FN)	27	6	7.070	3.615	0.061	0.3682
Y = 0.349 + 0.122(CC) + 8.809(RK) + 0.216(SB)	22	5	14.089	10.635	0.002	-0.0368
Y = 0.418 + 8.867(RK) + 0.003(TS) - 0.065(LF)	24	5	14.112	10.657	0.002	-0.0379
Y = 0.310 + 0.049(SB) + 4.459(TE) + 0.407(FB) + 5.332(FN) - 2.518(GS)	23	7	19.175	15.720	0.000	-0.1266
Y = 0.604 + 0.030(SN) - 0.051(SV) - 0.001(DBH) + 0.001(TH) - 0.003(CW)	30	7	24.106	20.652	0.000	-0.2695

a. K = Estimable parameters
b. Δi = |AICc lowest - AICci| for the ith model in comparison
c. wi = Akaike weights
d. Adj. R^2 = Unbiased Pearson’s correlation coefficient

Table 3. Parameter estimates (± SE) for models with substantial support for predicting visitation rate at edge, field and forest foraging stations at Fort Necessity National Battlefield, Pennsylvania, USA in October – November 2009 (Fall) and July – August 2010 (Summer) at 100 m^2 and 400 m^2 scales. Model variables included: percent plot coverage of grass (GS), forb (FB), fern (FN), shrub (SB), tree (TE), leaf (LF), log (LG), rock (RK); visual obscurity measurements (tallest sight, TS and first sight, FS), canopy cover (CC); soil moisture (SM) and closest tree (TD) distance to box.

Scale, Season	No.	Model (Parameter Estimate ± SE)
$Y_{edge\ (100m^2,\ summer)}$	8	Y = (0.659±0.274) - (1.029±0.243)LF + (0.455±0.313)CC
$Y_{edge\ (400m^2,\ summer)}$	8	Y = (0.726±0.268) - (1.055±0.249)LF + (0.387±0.310)CC
$Y_{edge\ (100m^2,\ fall)}$	8	Y = (0.662±0.210) - (0.575±0.165)LF + (0.005±0.004)CC
$Y_{edge\ (400m^2,\ fall)}$	8	Y = (0.621±0.210) - (0.567±0.171)LF + (0.006±0.004)CC
	4	Y = (-0.268±0.211) + (0.009±0.004)CC + (0.048±0.0023)FS + (0.004±0.002)TS
$Y_{field\ (100m^2,\ summer)}$	19	Y = (0.199±0.134) - (0.159±0.146)FB + (0.073±0.119)GS + (7.105±1.661)RK
	17	Y = (0.112±0.168) - (0.086±0.145)FB + (0.035±0.039)SM + (7.535±1.399)RK
$Y_{field\ (400m^2,\ summer)}$	17	Y = (0.108±0.204) - (0.138±0.172)FB + (0.081±0.045)SM + (6.533±1.672)RK
	19	Y = (0.338±0.157) - (0.321±0.174)FB + (0.176±0.149)GS + (4.797±2.076)RK
$Y_{field\ (100m^2,\ fall)}$	13	Y = (1.063±0.105) - (0.498±0.182)FB - (0.460±0.178)GS
	15	Y = (1.210±0.133) - (0.575±0.179)FB - (0.576±0.183)GS - (0.260±0.156)SB
	19	Y = (0.812±0.174) - (0.292±0.208)FB - (0.400±0.172)GS + (2.361±1.342)RK
$Y_{field\ (400m^2,\ fall)}$	19	Y = (0.715±0.190) - (0.232±0.213)FB - (0.382±0.164)GS + (3.108±1.362)RK
	15	Y = (1.278±0.133) - (0.633±0.177)FB - (0.629±0.173)GS - (0.279±0.141)SB
	11	Y = (0.793±0.216) - (0.004±0.005)FS - (0.534±0.209)GS + (3.308±1.212)RK - (0.001±0.001)TS
$Y_{forest\ (100m^2,\ summer)}$	21	Y = (0.464±0.045) + (2.041±0.855)FB + (5.711±0.627)LG + (0.026±0.109)SB
$Y_{forest\ (400m^2,\ summer)}$	21	Y = (0.473±0.047) - (2.482±0.945)FB + (6.414±0.724)LG + (0.082±0.113)SB
	29	Y = (0.392±0.086) + (2.089±0.893)FN + (4.897±0.928)LG + (0.571±2.820)TE

Table 3. Continued

Scale, Season	No.	Model (Parameter Estimate ± SE)
$Y_{forest\ (100m^2,\ fall)}$	26	Y = (0.725±0.318) - (0.317±0.331)LF + (5.004±1.388)LG + (0.147±0.283)SB - (0.044±0.017)TD
	29	Y = (0.251±0.327) + (3.862±2.362)FN + (4.731±1.580)LG + (1.246±7.168)TE
$Y_{forest\ (400m^2,\ fall)}$	29	Y = (0.244±0.324) + (3.596±2.169)FN + (4.474±1.393)LG + (1.265±6.962)TE
	26	Y = (0.598±0.301) - (0.193±0.328)LF + (4.734±1.313)LG + (0.196±0.280)SB - (0.039±0.018)TD

scales.

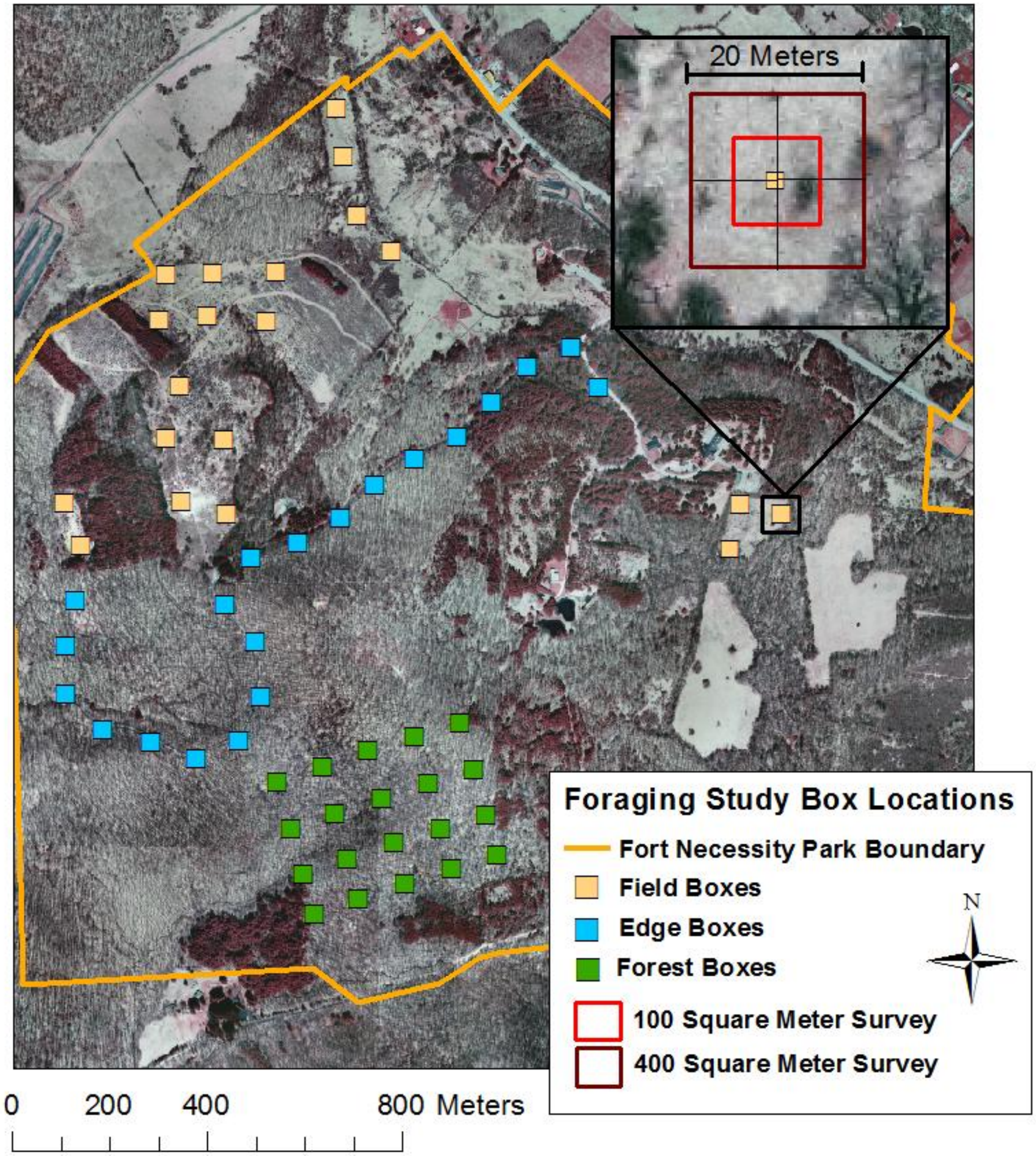

Figure 1. Foraging station locations throughout the 390 ha boundary of Fort Necessity National Battlefield, Pennsylvania, USA, from 2009-2010 with emphasis on the habitat survey spatial scales.

CHAPTER 5

REVIEW ARTICLE

Review of Conclusions and Management Implications for Fort Necessity National Battlefield, Farmington, Pennsylvania, in Relation to the Control of Morrow's Honeysuckle (*Lonicera morrowii*)

Charneé Lee Rose[1,2]

Introduction

Invasive plant species alter ecosystem services, negatively impact the diversity of native species, and cause considerable financial losses (DiTomaso 2000; Levine et al. 2003; Dukes & Mooney 2004; D'Antonio & Hobbie 2005). Restoration ecology has been practiced as a means of offsetting the deterioration of ecological systems caused by human population growth and exotic species (Ludwig et al. 1993; Chew 2001). Understanding the spread and establishment of invasive plant species, as well as the outcomes of control and management activities, is necessary to manage exotics. Continued research in the field of invasive species ecology is important for determining what habitat features support invasive species establishment, for detailing which characteristics identify potential invasive species, and for developing models to predict the invasion process (Hunter & Mattice 2002; Hartman & McCarthy 2004). By conducting restoration initiatives, we can gain valuable knowledge of how to reestablish ecosystem processes, and remove exotic species in a cost-effective manner while maintaining the integrity of the native community (Hartman & McCarthy 2004).

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This chapter written in the style of Restoration Ecology

In the mid-1980s Fort Necessity National Battlefield (FONE), located in Fayette County Pennsylvania, was invaded by an aggressive exotic shrub species, Morrow's honeysuckle (*Lonicera morrowii*). Bush honeysuckle invasion has been tied to a variety of negative ecosystem and wildlife impacts. *Lonicera* spp. suppress native vegetation (Pysek & Pysek 1995; Daehler 2003), increase the nest predation of songbirds (Schmidt & Whelan 1999), greatly reduce the body mass of herpetofaunal species (McEvoy & Durtsche 2004), and alter the habitat use of small mammals (Edalgo et al. 2009). Studies measuring the community-level effects of exotic plants are rare (Tickner et al. 2001; Hejda et al. 2009), and no studies found have examined the response of vertebrate and vegetation communities to the removal of Morrow's honeysuckle, or the small mammal foraging preference of Morrow's fruits when compared to native species. To better understand the effects of an aggressive invader, Morrow's honeysuckle, we conducted a 7-year study evaluating the presence of honeysuckle and the effects of the control methods used to remove the species on the biotic communities sampled. Additionally, we conducted a study on the consumption of honeysuckle fruits by small mammals, and the associated habitat characteristics that affected visitation rate to study stations.

Objectives

The restoration study examining the effects of Morrow's honeysuckle and its removal was conducted across 13.6 ha of FONE from May 2004 to August 2010 (Rose 2011: Chapters 1 & 2). The study objectives and hypothesis are listed below:

1) To determine the species composition of shrub and herbaceous communities prior to, and following restoration procedures.

    H_0: There is no difference in species composition of shrub and herbaceous species in the study plots prior to, and following restoration.

H_a: There is a difference in composition of shrub and herbaceous species, with restoration plots showing higher species diversity.

2) To assess the relative abundance and location of American woodcock (*Scolopax minor*) prior to, and following restoration techniques.

    H_0: American woodcock will use the study area indiscriminately prior to and following restoration.
    H_a: American woodcock abundance will be greater in the reclaimed plots.

3) To determine the relative abundance and richness of herpetofauna prior to and following restoration.

    H_0: Herpetofauna species will use the study area indiscriminately.
    H_a: Herpetofauna abundance/richness will be greater in the reclaimed plots.

4) To measure the relative abundance and richness of songbirds within the study area prior to and following management activities.

    H_0: Songbird species will use the study area indiscriminately.
    H_a: Songbird abundance/richness will be greater in the reclaimed plots.

5) To assess the relative abundance and richness of small mammals prior to and following restoration procedures.

    H_0: Small mammal species will use the study area indiscriminately.
    H_a: Small mammal abundance/richness will be greater in the reclaimed plots.

The small mammal foraging and habitat use study was conducted across edge, field, and forest cover types throughout FONE from October 2009 to September 2010 (Rose 2011: Chapter 3 & 4). The study objectives and hypothesis are listed below:

1) To assess the species of small mammals that actively consume Morrow's honeysuckle fruits.

    H_0: All granivorous species present consume honeysuckle fruits.
    H_a: Not all granivorous species present consume honeysuckle fruits.

2) To investigate if small mammals use Morrow's honeysuckle in their diet in the same quantities as native soft mast fruits.

H_0: Small mammals utilize honeysuckle and natives indiscriminately in their diets.

H_a: Small mammals show distinct foraging preferences between Morrow's and native fruits, with native soft mast fruits showing higher consumption rates.

3) To determine if the magnitude of Morrow's fruit consumption remains consistent across cover types, and throughout seasonal changes of Morrow's fruiting period.

    H_0: Small mammal consumption of honeysuckle fruit remains consistent across cover types and the rate of consumption does not change throughout tested seasons.

    H_a: Small mammals consume honeysuckle fruits differently depending on cover type and season tested, with foraging pressures highest in edge plots and the July study phase.

4) To assess the habitat characteristics that contribute to high small mammal visitation rate across cover types and between two seasons;

    H_0: Small mammal visitation rate to study boxes is independent of environmental variables, and there is no difference in visitation rates.

    H_a: Small mammal visitation rate to study boxes is increased with % shrub, % overhead canopy, % log and increased height of vertical vegetation depending on cover type observed and season.

Results

Impacts of Honeysuckle and Restoration Procedures

Vegetation. – We found that a combination of yearly mowing and applications of broad-spectrum herbicides were highly effective at reducing Morrow's honeysuckle in treatment areas (Rose 2011: Chapter 2). The percent cover of Morrow's honeysuckle declined significantly ($p < 0.05$) from 2005 (pre-removal) to 2010 (post-removal), despite treatment areas experiencing resprouting in 2008 due to root and stem sprouts. Previous studies (Webster et al. 2007; Love & Anderson 2009) examining the effectiveness of honeysuckle removal methods also experienced vigorous resprouting during control efforts. Our restoration procedures were able to reduce the plot coverage of Morrow's honeysuckle without negatively affecting either the native shrub cover or the shrub species richness. Additionally, there seemed to be little negative impact on the herbaceous vegetation with respect to diversity, richness or evenness. Any negative variation seen in this assemblage was due to significant ($p < 0.05$) yearly variation, and perhaps discrepancies in observer detectability.

While an early-successional community of grasses and herbaceous species now characterizes the treatment area, the mean coefficients of conservatism (C) and the floristic quality indices (FQI) showed no significant differences between the post-removal reference and treatment plots. The two indices are used to quantify both restoration success and the relative health of study sites based on their species composition (Northern Great Plains Floristic Quality Assessment Panel 2001; Rentch & Anderson 2006). The values of C ranged only from 2.4 to 4.4 across study plots and years. This range of values indicates that the study site generally supported species that were wide spread or were associated with degraded habitat (Rentch & Anderson 2006). Since the treatment area is still recovering from management practices, and a monoculture of honeysuckle still persists in the reference, it is not surprising that low values are represented in both plot types.

As the treatment area continues to recover from restoration procedures it is likely that the vegetation community will change as new species establish. Colonization of native species is likely to be hastened in the treatment plots due to the planting of meadow species and shrubs designed to enhance native establishment, scheduled for spring 2011 by the National Park Service. We believe with continued management and establishment of native species, the treatment area will be successfully maintained as quality early-successional habitat.

American Woodcock. – The overall study site serves as an important singing area for American woodcock, and removal procedures did not negatively affect this species. While there were no statistical significances ($p > 0.05$) between reference and treatment plots, numerically the treatment had the highest number of males throughout study years. There is likely not a greater discrepancy in use due to the habitat that is adjacent to the singing grounds. Sepik and Derleth (1993) found that the vegetation cover of singing grounds had less effect on use than the availability of nearby habitat for nesting. Across FONE, both the reference and treatment areas were surrounded by adequate brood-rearing habitat and this could have influenced the distribution. Singing males were most often located along mowed trails, and close to nearby wooded areas; these locations had open space for displaying and had protective herbaceous and forested cover nearby. In order to maintain the treatment area for American woodcock, dense plantings should be avoided (Dessecker & McAuley 2001). Additionally, we recommend that the forested areas (young to mixed-aged corridors) surrounding the study plots be left intact to complete necessary habitat needed for this species throughout the breeding cycle (Mendall & Aldous 1943).

Songbirds. – Morrow's honeysuckle removal procedures revealed no short-term adverse impact on songbird species composition, diversity or richness. When looking at individual species abundances and examining the songbird community as a whole there were no detectable differences between pre- or post-removal plot types, only significant yearly variation ($p < 0.05$). These results are consistent with the finding of McCusker et al. (2010), who found no differences in avian community structure when comparing areas with and without *Lonicera* species. Songbirds are generally better indicators of habitat conditions at the landscape scale than at smaller localized sites (Carignan & Villard 2001). Also, the life history of songbirds makes them a less than ideal assemblage to use for determining either quality of habitat or preference. These species are territorial (Brown 1969) and when habitat is limited, they can establish territories in non-preferred habitat (Van Horne 1983). This may explain their presence in the reference plots characterized by dense thicket of Morrow's honeysuckle when *Lonicera* spp. are known to cause higher rates of predation for songbird nests (Schmidt & Whelan 1999). At 77-92%, early-successional species made up the majority of songbird observations. We believe as the treatment area is maintained and native herbaceous and shrub species continue to be planted the treatment area will produce critical habitat for these species.

Small Mammals. – Like the other assemblages monitored, small mammal species appeared to experience no direct, negative impacts from honeysuckle removal procedures. This corresponds to previous research by Sullivan (1990), which showed that treatment of *Lonicera* spp. with a glyphosate herbicide had little or no effect on recruitment of various small mammal species. There were only significant differences ($p < 0.05$) in total relative abundance due to natural, yearly population fluctuation (MacCracken et al. 1985). While the ability to recolonize an area is unknown for many small mammal species (McShea et al. 2003), meadow voles (*Microtus pennsylvanicus*) recolonized the post-removal treatment plots in higher numbers than any other small mammal species. Restoration procedures produced treatment plots that provided critical habitat for this early-successional species (Manson et al. 1999). It is from this result that we discover potential indirect effects of restoration on the small mammal community. Meadow voles are competitive and can decrease the abundances of species like white-footed mice (*Peromyscus leucopus*) (Boonstra & Hoyle 1986) and meadow jumping mice (*Zapus hudsonius*) (Anthony et al. 1981). Our results are consistent with these studies: as meadow voles increased in the post-removal treatment plots we saw a decrease in white-footed mice and an absence of meadow jumping mice.

The capture rate of small mammals can often be maximized in early-successional habitats that have high structural complexity, such as dense shrubs and protective vine cover (Healy and

Brooks 1988). White-footed mice were the most frequently encountered small mammals in our study area, although they were captured at the lowest quantities in the post-removal treatment plots. Although this is partly due to an increase in competition by meadow voles, it is also likely attributed to the lack of shrub cover, as the treatment area is currently characterized by native grasses and forbs. As native shrub cover is established in the treatment area, we expect this species' population to increase. Likewise, improvements in shrub cover should help retain moisture and facilitate the recruitment of shrew species (McCay & Storm 1997). There should be a continued persistence of the meadow vole population, and likely an increase in the meadow jumping mouse population once voles reach their cyclical decline (Boonstra & Hoyle 1986).

Herpetofauna. – We did not capture herpetofaunal species in large enough numbers for statistical analysis. Among the amphibian species captured, redback salamanders (*Plethodon cinereus*) were the most common. Additionally, we caught five snake species in the pitfall and cover board arrays. The management plan for FONE suggests that portions of the treatment area will be maintained as early-successional habitat, while other areas will be reforested (National Park Service 1991). This management strategy should benefit the variety of herpetofaunal species captured. Forested conditions will facilitate moisture retention and the creation of both woody debris and leaf litter for terrestrial salamanders (Ash 1995; Petranka 1998) and associated predators such as the ring-necked snake (*Diadophis punctatus edwardsii*). The habitat requirements for the remaining snake species observed should be adequately represented in the areas that will remain as early-successional habitat (Conant & Collins 1998; Hulse et al. 2001).

Small Mammal Foraging Study

Fruit Use. – The consumption of invasive Morrow's honeysuckle and five native soft mast species was studied across three cover types (forest, field, and edge) and two fruiting rounds (October – November 2009 and July – early August 2010) at FONE (Rose 2011; Chapter 3). Diet use was determined to be non-random during both study rounds ($p < 0.05$). Morrow's honeysuckle was chosen last over all native species used, except for one trial. Honeysuckle fruits experienced higher rates of consumption over native staghorn sumac (*Rhus typhina*) during Round 2 (July- early August), which coincided with what appeared to be the highest point of honeysuckle fruit use throughout both rounds. This round was also the only instance where the magnitude of use differed across cover types, as field plots generally experienced the lowest total consumption versus all other cover types. This is likely due to the high availability of grass seed in the field study plots during this round.

Nutritional analysis revealed that honeysuckle fruits had significantly less protein (0.66%) and lipids (0.67%) than all natives ($p < 0.05$). This result is consistent with Witmer and Van Soest (1998), and leads to its classification as a low quality food for birds (White & Stiles 1992). Nevertheless, Morrow's honeysuckle was found to be in the mid-range of values for non-fiber carbohydrates (quick energy sugars) and total energy (kcal) available per fruit, while having one of the highest moisture contents of study fruits. We believe its moisture content appears to be important in the use of its fruits, a result not previously documented due to the use of only dried seeds in prior feeding trials (Shahid et al. 2009). In the only instance that honeysuckle was consumed more than a native species, that species (sumac) had significantly lower moisture, while having higher fat and protein content.

Despite high moisture content, Morrow's fruits are still lacking key nutrition (protein and lipids) for survival and reproduction, likely leading to its overall low use. Total energy was important in distinguishing the highest selected fruits: black cherry (*Prunus serotina*) (0.45 kcal), and common dewberry (*Rubus flagellaris*) (0.36). Consumption of fruits beyond the first chosen was inconsistent and varied based on moisture, protein, lipids and carbohydrates. Other variables measured, such as average seeds, mass, and handling time (time a small mammal took to consume a whole fruit) had no influence on use.

Effects on White-footed Mice. – Based on a variety of monitoring techniques, white-footed mice made up the majority of small mammals captured during study rounds (88% based on Sherman live trapping). Due to low abundances of all other species captured, we attributed the results of fruit use to this species. The relation between food abundance and consumption can determine an animal’s energy intake; the more abundant a food source is, the more likely it is to be discovered during foraging activity (Barboza et al. 2009). Animals with high metabolic rates, such as small mammals, may be unable to sustain prolonged searches for food, especially if the food source has low abundance or is of low quality (Barboza et al. 2009). This is a concern with Morrow’s honeysuckle, as it is low in vital nutrients, and creates monocultures out competing natives from the environment, which are the more nutritious and used food source. This in turn may force small mammals, like the white-footed mouse, to forage for longer periods of time, or travel further distances and increase their risk of predation. Although Morrow’s honeysuckle provides cover for small mammals (Dutra et al. 2011; Edalgo et al. 2009), we conclude that Morrow’s honeysuckle creates a monoculture of a less nutritious and less used food item.

Although white-footed mice showed plasticity in their diets with the ability to optimize between trade-offs in nutrient content across seasons, it is likely that this species is pressured by large energy expenditures to find suitable food sources at FONE. We recommend the continued removal of bush honeysuckle from the study site, and that resource managers promote the highly consumed native species documented in this research.

Small Mammal Habitat Use

Variables Affecting Visitation. – We compared white-footed mouse consumption of Morrow’s honeysuckle to five native soft mast species (Rose, 2011: Chapter 3). Although we had a sample size of 60, not all foraging stations yielded optimal data due to low visitation rate by mice. Therefore, we calculated the visitation rate (the proportion of study days that foraging took place out of 14 days), and used habitat measurements to model visitation rate across seasons (fall, October-November 2009 and summer, July-August 2010), cover types (field, forest, and edge) and scales (100 m^2 and 400 m^2) (Rose, 2011: Chapter 4).

Although white-footed mice may be considered habitat generalist (Adler & Wilson 1987; Dueser & Shugart 1978), they may still specialize in particular habitat features of a certain cover type; especially when travel routes are important, such as to and from a known foraging site. When comparing cover types, we found that edge and forest were most similar in the variables that predicted high visitation rate. These two cover types required areas of high structural complexity. Variables of importance included increased overhead canopy cover, and percent logs, shrubs and ferns. In contrast, the field cover type experienced low levels of visitation rate in areas of high structural complexity. Variables of importance included increased rocks around foraging stations, with decreased cover of forbs, grass, and shrubs. Given these results, it is likely that the field cover type is the most vulnerable of the three to invasions from exotic plant species. The highly used stations in the field were characterized by large amounts of open space and could be ideal areas for seed germination. These areas could be at risk from both forgotten cached invasive seeds (Abbot & Quink 1970) or even post-gut viability of invasive seeds (Williams et al. 2000).

Shrub Species Impact. – High visitation rate was negatively associated with shrub coverage in the field stations, but positively associated in forest stations. The most common shrub species in the field was invasive Morrow’s honeysuckle, accounting for a large percentage of shrub cover, and found to be the closest shrub 85% of the time. Honeysuckle has an open understory, due to its branch architecture, which is navigable by larger predators (Schmidt & Whelan 1999). In contrast, native common greenbrier (*Smilax rotundifolia*) was the most common shrub at forest stations, found to be the closest shrub 95% of the time. Unlike honeysuckle, this species provides a dense cover of protective thorns. Additionally, since honeysuckle was found to be less nutritious and a lesser preferred food item when compared to native species at FONE (Rose, 2011: Chapter 3), marginal value theorem could also explain lower visitation rate to stations near this shrub in the field. Low food abundance and nutritional content may reduce the energy an animal gains, and therefore increase foraging time (Charnov, 1976). Given that an animal loses its profit if it continues to feed in such an area, it is likely it will not forage near dense areas of honeysuckle.

Spatial and Seasonal Differences. – Spatial scales within a season were nearly identical, other than 400 m^2 often requiring more variables to explain visitation. This difference at the large scale was attributed to the addition of or decrease in structural variables that could affect predator risk when traveling to and from stations. When comparing differences across measured spatial scales or seasons, seasons showed the largest differences within a cover type in regards to the best model based on the data. Ferns and trees, among the only persistent cover-providing vegetation during colder seasons, were more important in fall than summer models for the forest. Additionally, percent grass in the field was positively associated with visitation rate in the summer, likely due to providing grass seed as a source of food. However, in the fall grass was negatively associated with visitation rate, likely because it provided cover for predators without providing a food source. As forbs and grasses decreased in the fall, shrubs appeared to be more important to predators as cover, which would explain why this parameter is only important in fall models at helping to predict visitation rate. These differences are especially important to note as

Peromyscus change their diets seasonally and we found that important habitat variables (those strongly affecting visitation rate) also differed based on season. We recommend following these seasonal differences, as well as other microsites documented here, as it may be possible to increase detection of species, and foraging preferences to study stations based on facilitating high visitation rate.

Management Implications

This study illustrates the importance of monitoring ecological communities over the long-term, while choosing appropriate study organisms. It is difficult to fully understand the effects of alterations to a habitat without continued monitoring. We surveyed the biotic communities in our study for seven years; although studies even of this limited length are not common in restoration research (Falk et al. 2006), due to lack of study funds. Had we not continued with several consecutive years of data collection we may have attributed population fluctuations in several of the study species to the restoration procedures, when in fact they were likely due to yearly variation. Additionally, we found that, at the scale of our restoration initiative, certain species assemblages (such as songbirds) were poor indicators of habitat quality. Future studies noting this result could conserve valuable resources (funding and field personnel) by monitoring different organisms or expanding the project to an appropriate scale.

As the spread of invasive species is accelerated due to human population growth, developments in global travel and international trade (Mack and Erneberg 2002), so must our understanding of the invasion process accelerate to keep ahead of the potential threats. Our results demonstrate the wide-ranging effects of invasive Morrow's honeysuckle, while providing protocols for removing and managing this aggressive exotic species. Resource managers looking to promote early-successional habitat for declining species such as the American woodcock, or pivotal seed dispersers such as the white-footed mouse, could follow our removal procedures to achieve effective Morrow's honeysuckle control.

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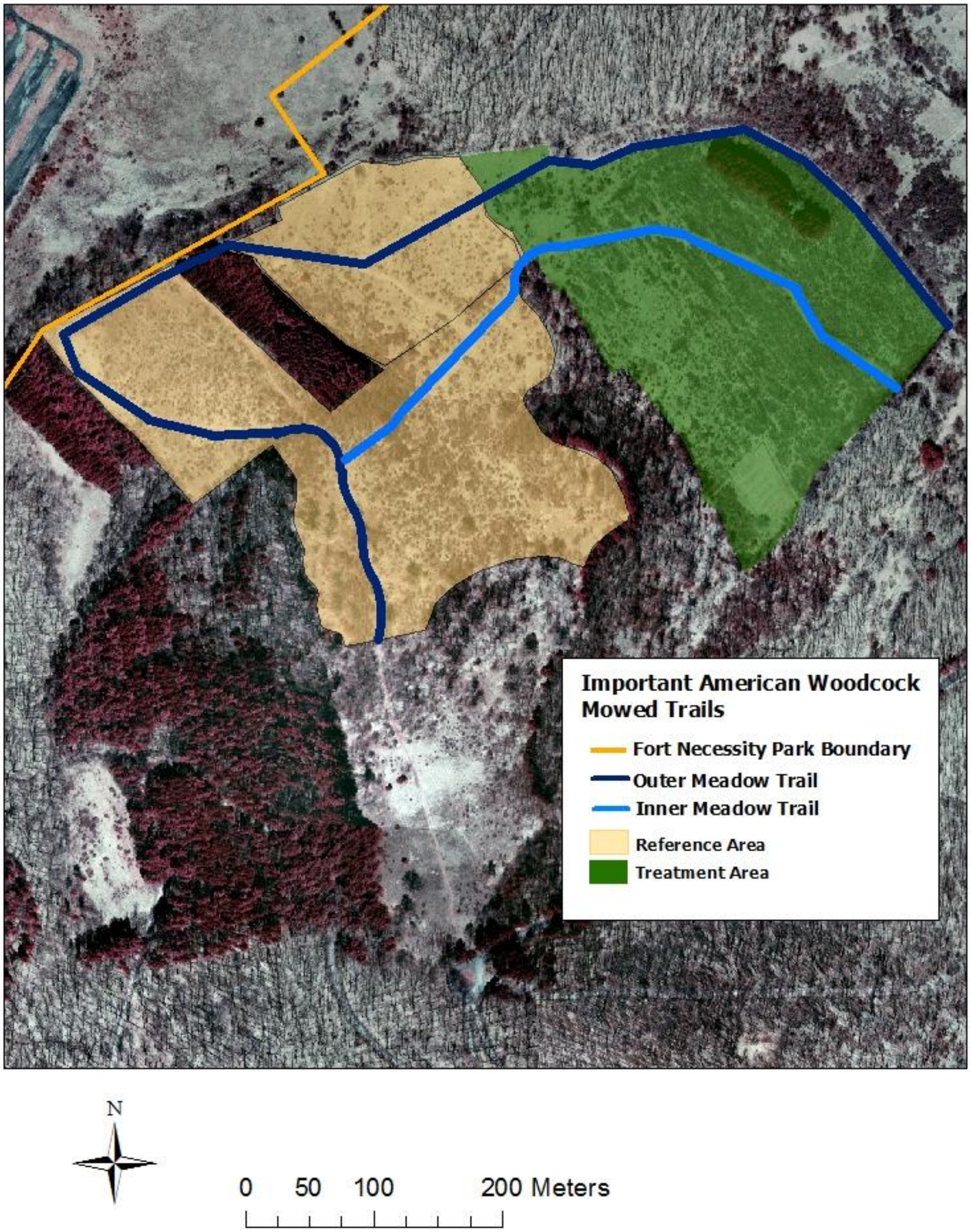
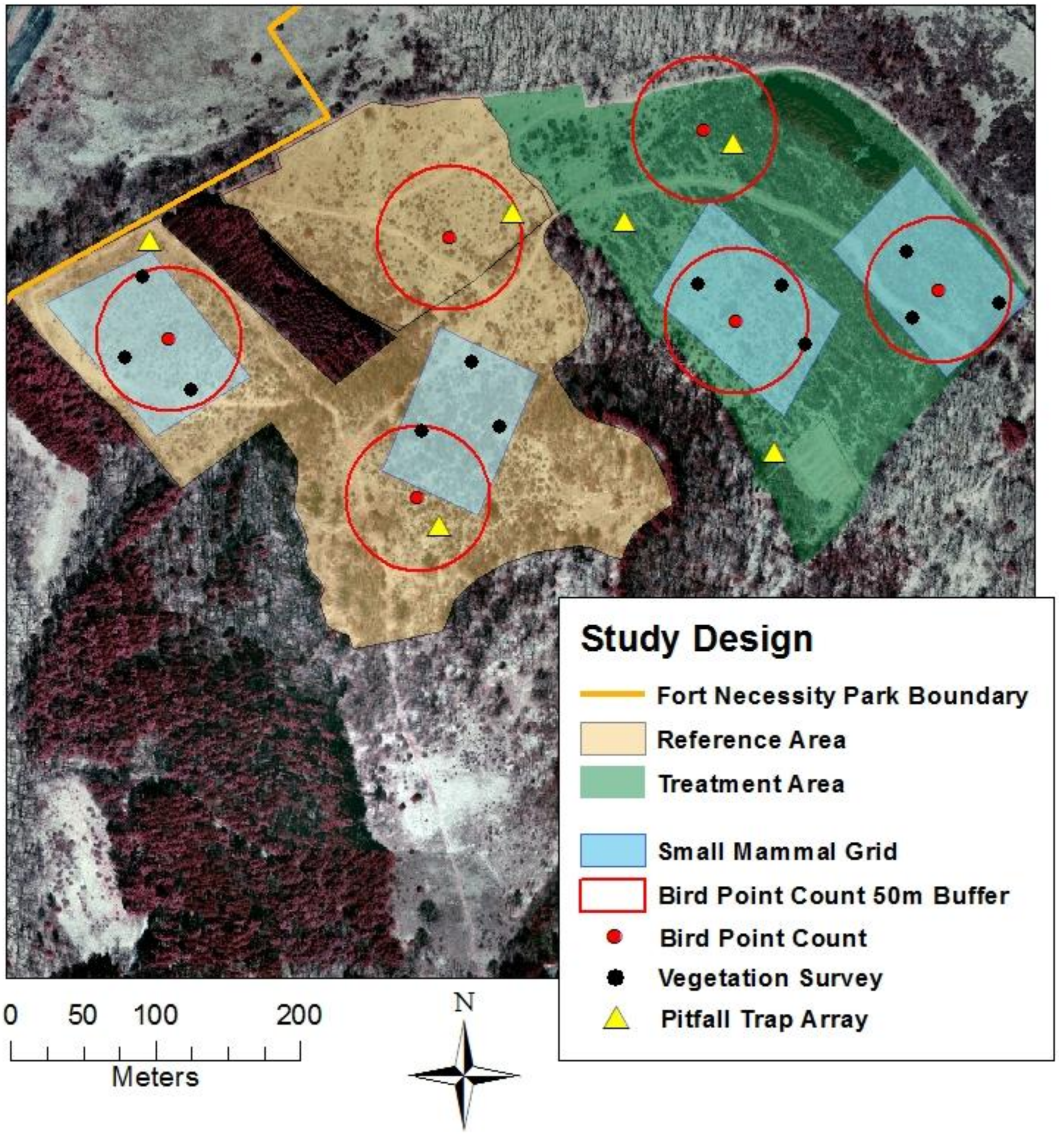
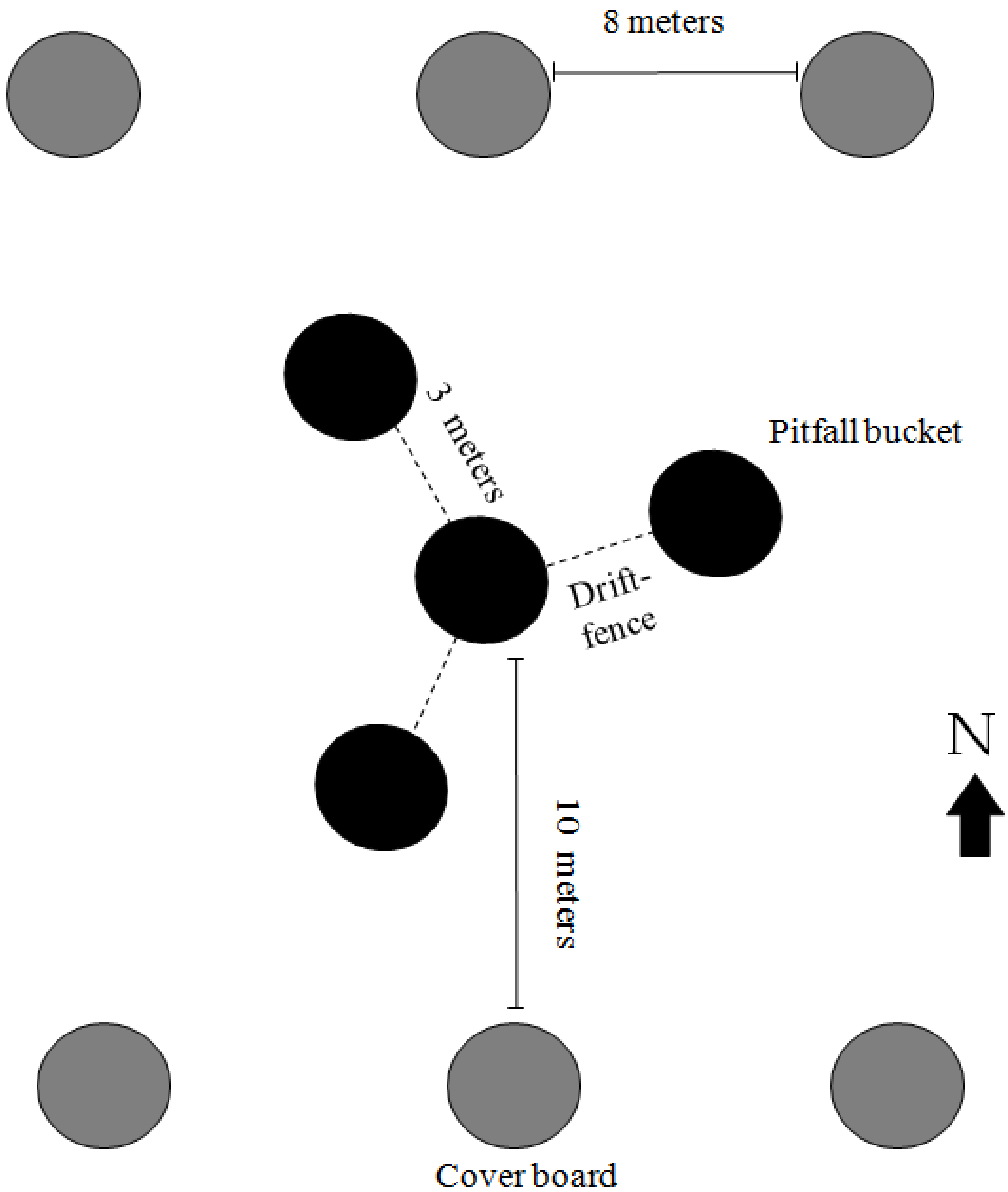
Appendix Ia: Location of the Outer Meadow and Inner Meadow trails in the treatment and reference plots at Fort Necessity National Battlefield, Pennsylvania, USA. American woodcock were often observed using these specific mowed paths in the park from 2004-2010.



Appendix IIa. Location of reference and treatment areas including corresponding small mammal Sherman trapping grids, vegetation survey points, bird point count locations, and pitfall trapping arrays at Fort Necessity National Battlefield, Pennsylvania, USA, 2004-2010.



Appendix IIIa. Each pitfall array consisted of four 20 litter buckets placed in a triad and connected with a 3-m long, 50-cm high silt fence. Six pitfall trap arrays, and 36 cover boards were placed throughout the study area at Fort Necessity National Battlefield, Pennsylvania, USA, 2004-2010.



Appendix IVa. Biotic community variables with non-significant *F*-tests, based on data collected from Fort Necessity National Battlefield, Pennsylvania, USA, 2004-2010.

Pitfall Variables	Year (*df* = 6,24)		Plot Type (*df* = 1,4)		Interaction (*df* = 6,24)	
	F-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value
White-footed mouse	1.22	0.332	0.29	0.621	1.22	0.332
Masked shrew	1.72	0.160	2.95	0.161	0.42	0.858
Smoky shrew	2.12	0.009	1.09	0.356	0.26	0.948
N. short-tailed shrew	1.59	0.192	4.00	0.116	0.94	0.487
Meadow jumping mouse	1.20	0.338	0.67	0.458	1.97	0.111
Bog lemming	3.23	0.018	0.00	1.000	0.16	0.985
Species diversity (H’)	3.49	0.013	0.00	0.973	1.09	0.399
Species richness (S)	1.71	0.162	0.33	0.597	1.07	0.408
Species evenness (J)	2.23	0.075	0.11	0.760	1.17	0.357
Total relative abundance	2.39	0.059	1.58	0.278	0.42	0.860
Sherman Variables	Year (*df* = 6,12)		Plot Type (*df* = 1,2)		Interaction (*df* = 6,12)	
	F-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value
Masked shrew	6.38	0.004	2.68	0.243	1.69	0.206
N. short-tailed shrew	1.32	0.319	2.04	0.289	1.22	0.360
Woodland jumping mouse	3.52	0.030	0.20	0.698	0.42	0.850
Species diversity (H’)	4.76	0.011	161.71	0.006	0.92	0.514
Species richness (S)	4.85	0.010	2.69	0.243	0.65	0.690
Species evenness (J)	3.02	0.049	20.47	0.046	0.90	0.524
Point-count Variables	Year (*df* = 2,8)		Plot Type (*df* = 1,4)		Interaction (*df* = 2,8)	
	F-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value
Early successional species	1.61	0.258	0.47	0.531	0.24	0.795
Generalist species	1.67	0.248	0.47	0.531	0.12	0.887
Species evenness (J)	0.37	0.700	0.07	0.807	0.54	0.604
Eastern towhee	4.28	0.054	0.23	0.655	0.21	0.812
Field sparrow	2.47	0.147	1.45	0.295	1.96	0.203
Vegetation Variables	Year (*df* = 2,19)		Plot Type (*df* = 1,10)		Interaction (*df* = 2,19)	
	F-value	*p*-value	*F*-value	*p*-value	*F*-value	*p*-value
Species Diversity (H’)	4.47	0.026	9.50	0.012	2.26	0.131
Exotic species richness	5.79	0.011	8.20	0.017	1.50	0.248

Appendix Va. Plant species identified at 12 random sampling points in reference (R) and treatment (T) plots at Fort Necessity National Battlefield, Pennsylvania, USA, in 2005, 2008, and 2010. Species in bold type are exotic. Coefficient of conservatism (C) is given for each species.

			Plot		
Family	**Species**	**Common Name**	**R**	**T**	**C**
Aceraceae	*Acer rubrum* L.	Red Maple	*	*	3
Anacardiaceae	*Toxicodendron radicans* (L.) Kuntze	Poison Ivy	*		3
Apiaceae	***Daucus carota* L.**	**Queen Anne's Lace**	*****	*****	**0**
Apocynaceae	*Apocynum cannabinum* L.	Indian Hemp	*	*	3
Asteraceae	***Achillea millefolium* L. var. occidentalis DC.**	**Yarrow**	*****	*****	**0**
Asteraceae	***Cirsium foliosum* (Hook.) DC.**	**Bull Elk Thistle**		*****	**0**
Asteraceae	*Coreopsis major* Walt.	Greater Tickseed	*		5
Asteraceae	*Doellingeria umbellata* (P. Mill.) Nees var. umbellata	Parasol Whitetop	*	*	5
Asteraceae	*Erigeron strigosus* Muhl. ex Willd. var. strigosus	Prairie Fleabane	*	*	2
Asteraceae	*Euthamia graminifolia* (L.) Nutt. var. graminifolia	Flat-top Goldenrod	*	*	4
Asteraceae	***Hieracium caespitosum* Dumort.**	**Meadow Hawkweed**	*****	*****	**0**
Asteraceae	*Lactuca canadensis* L.	Canada Lettuce	*		3
Asteraceae	***Leucanthemum vulgare* Lam.**	**Ox-eye Daisy**	*****	*****	**0**
Asteraceae	*Packera aurea* (L.) A.& D. Löve	Golden Ragwort		*	4
Asteraceae	*Rudbeckia hirta* L.	Black-eyed Susan		*	4
Asteraceae	*Solidago caesia* L.	Wreath Goldenrod	*	*	6
Asteraceae	*Solidago canadensis* L.	Canada Goldenrod	*	*	3
Asteraceae	*Solidago juncea* Ait.	Early Goldenrod	*	*	5
Asteraceae	*Solidago nemoralis* Ait. var. nemoralis	Gray Goldenrod	*	*	5
Asteraceae	*Solidago patula* Muhl. ex Willd. var. patula	Rough Goldenrod	*	*	8
Asteraceae	*Solidago rugosa* P. Mill.	Wrinkleleaf Goldenrod	*	*	3
Asteraceae	*Solidago uliginosa* Nutt.	Bog Goldenrod	*		6
Asteraceae	*Symphyotrichum lanceolatum* (Willd.) Nesom ssp. lanceolatum var. lanceolatum	Panicled Aster	*		4
Asteraceae	*Symphyotrichum lateriflorum* (L.) A.& D. Löve	Calico Aster	*	*	4
Asteraceae	*Symphyotrichum pilosum* (Willd.) Nesom	Hairy White Oldfield Aster		*	4
Asteraceae	*Symphyotrichum prenanthoides* (Muhl. ex Willd.) Nesom	Crooked-stem Aster	*	*	5
Asteraceae	*Symphyotrichum puniceum* (L.) A.& D. Löve var. puniceum	Purplestem Aster	*	*	6

Appendix Va. Continued

Asteraceae	***Taraxacum officinale*** **G.H. Weber ex Wiggers ssp. officinale**	**Common Dandelion**		*****	**0**
Asteraceae	*Vernonia gigantea* (Walt.) Trel. ssp. gigantea	Giant Ironweed	*	*	3
Campanulaceae	*Lobelia inflata* L.	Indian Tobacco		*	3
Caprifoliaceae	***Lonicera morrowii*** **Gray**	**Morrow's Honeysuckle**	*****	*****	**0**
Caprifoliaceae	*Viburnum recognitum* Fern.	Southern Arrowwood	*	*	6
Caryophyllaceae	***Cerastium fontanum*** **Baumg. ssp. vulgare (Hartman) Greuter & Burdet**	**Common Mouse-ear Chickweed**	*****	*****	**0**
Caryophyllaceae	*Cerastium nutans* Raf. var. nutans	Powderhorn Chickweed		*	4
Caryophyllaceae	***Dianthus armeria*** **L.**	**Deptford Pink**		*****	**0**
Caryophyllaceae	*Stellaria longifolia* Muhl. ex Willd. var. longifolia	Longleaf Starwort	*	*	6
Clusiaceae	*Hypericum perforatum* L.	Common St. John's Wort		*	0
Clusiaceae	*Hypericum punctatum* Lam.	Spotted St. John's Wort	*		4
Convolvulaceae	***Calystegia sepium*** **(L.) R. Br. ssp. sepium**	**Hedge False Bindweed**		*****	**0**
Cornaceae	*Cornus racemosa* Lam.	Gray Dogwood		*	6
Cyperaceae	*Carex digitalis* Willd.	Slender Woodland Sedge		*	4
Cyperaceae	*Carex hirsutella* Mackenzie	Fuzzy Wuzzy Sedge	*	*	4
Cyperaceae	*Carex virescens* Muhl. ex Willd.	Ribbed Sedge	*		6
Dryopteridaceae	*Dryopteris carthusiana* (Vill.) H.P. Fuchs	Spinulose Woodfern		*	6
Eleagnaceae	***Eleagnus umbellata*** **Thunb. Var.** ***parvifolia*** **(Royle) Schneid.**	**Autumn olive**		*****	**0**
Euphorbiaceae	*Euphorbia corollata* L.	Flowering Spurge	*		5
Fabaceae	*Amphicarpaea bracteata* (L.) Fern.	Hog Peanut		*	4
Fabaceae	***Coronilla varia*** **L.**	**Crown Vetch**	*****		**0**
Fabaceae	***Melilotus officinalis*** **(L.) Lam.**	**Yellow Sweet Clover**		*****	**0**
Fabaceae	*Robinia pseudoacacia* L.	Black Locust		*	2
Fabaceae	***Trifolium aureum*** **Pollich**	**Golden Clover**	*****	*****	**0**
Fabaceae	***Trifolium pratense*** **L.**	**Red Clover**		*****	**0**
Fabaceae	***Trifolium repens*** **L.**	**Sweet White Clover**	*****	*****	**0**
Fabaceae	*Trifolium* spp.	Clover spp.		*	0
Fagaceae	*Quercus rubra* L.	Red Oak	*	*	5
Fagaceae	*Quercus* spp.	Oak spp.	*		0
Iridaceae	*Sisyrinchium angustifolium* P. Mill.	Narrowleaf Blue-eyed Grass	*		4

Appendix Va. Continued

Lamiaceae	*Clinopodium vulgare* L.	Wild Basil	*	*	2
Lamiaceae	*Lycopus virginicus* L.	Bugleweed	*	*	4
Lamiaceae	*Prunella vulgaris* L.	Common Self-heal	*	*	1
Lamiaceae	*Pycnanthemum incanum* (L.) Michx.	Hoary Mountain Mint	*	*	6
Liliaceae	*Medeola virginiana* L.	Indian Cucumber	*		6
Lycopodiaceae	*Lycopodium digitatum* Dill. ex A. Braun	Fan Clubmoss	*	*	4
Magnoliaceae	*Liriodendron tulipifera* L.	Tulip Poplar	*		5
Magnoliaceae	*Magnolia acuminata* (L.) L.	Cucumber Magnolia	*		6
Oleaceae	*Fraxinus americana* L.	White Ash	*	*	6
Onagraceae	*Oenothera perennis* L.	Little Evening Primrose		*	5
Ophioglossaceae	*Botrychium dissectum* Spreng.	Cutleaf Grapefern	*	*	4
Orchidaceae	*Liparis liliifolia* (L.) Rich. ex Ker Gawl.	Brown Widelip Orchid		*	5
Oxalidaceae	*Oxalis stricta* L.	Yellow Woodsorrel	*	*	2
Pinaceae	*Pinus strobus* L.	White Pine		*	5
Plantaginaceae	***Plantago lanceolata* L.**	**Narrowleaf Plantain**	*	*	**0**
Plantaginaceae	***Plantago major* L.**	**Wide Leaf Plantain**		*	**0**
Poaceae	*Agrostis gigantea* Roth	Redtop Grass	*	*	0
Poaceae	*Agrostis perennan*s (Walt.) Tuckerman	Upland Bentgrass	*	*	4
Poaceae	*Andropogon virginicus* L. var. virginicus	Broomsedge Bluestem	*	*	3
Poaceae	***Anthoxanthum odoratum* L. ssp. odoratum**	**Sweet Vernal Grass**	*	*	**0**
Poaceae	***Arrhenatherum elatius* (L.) Beauv. Ex J.& K. Presl**	**Tall Oat Grass**	*	*	**0**
Poaceae	***Bromus inermis* Leyss. ssp. inermis var. inermis**	**Smooth Brome**		*	**0**
Poaceae	***Dactylis glomerata* L. ssp. glomerata**	**Orchard Grass**	*	*	**0**
Poaceae	*Danthonia compressa* Austin ex Peck	Flattened Oatgrass	*	*	6
Poaceae	*Danthonia spicata* (L.) Beauv. ex Roemer & J.A. Schultes	Poverty Grass	*	*	5
Poaceae	*Dichanthelium clandestinum* (L.) Gould	Deer-tongue Grass	*	*	3
Poaceae	*Dichanthelium scabriusculum* (Elliot) Gould & C.A. Clark	Wooly Rosette Grass	*	*	9
Poaceae	*Dichanthelium sphaerocarpon* (Ell.) Gould	Roundseed Panicgrass		*	4
Poaceae	*Dichanthelium* spp.	Panic Grass	*	*	0
Poaceae	*Elyleymus* spp.	Wild Rye	*		0
Poaceae	***Holcus lanatus* L.**	**Velvet Grass**	*	*	**0**
Poaceae	***Lolium arundinaceum* (Schreb.) S.J. Darbyshire**	**Tall Fescue**		*	**0**
Poaceae	***Lolium perenne* L. ssp. perenne**	**Perennial Ryegrass**		*	**0**

Appendix Va. Continued

Poaceae	*Danthonia* spp.	Oat Grass	*	*	0
Poaceae	***Phleum pratense* L.**	**Timothy Grass**		*****	**0**
Poaceae	*Poa trivialis* L.	Rough Bluegrass	*	*	0
Polygonaceae	***Rumex acetosella* L.**	**Sheep Sorrel**	*****	*****	**0**
Primulaceae	*Lysimachia lanceolata* Walt.	Lance-leafed Loostrife	*	*	6
Ranunculaceae	*Ranunculus* spp.	Ranunculus spp.	*	*	0
Rosaceae	*Amelanchier arborea* (Michx. f.) Fern. var. arborea	Common Serviceberry	*	*	5
Rosaceae	*Crataegus pruinosa* (Wendl. f.) K. Koch	Waxyfruit Hawthorne	*	*	5
Rosaceae	*Fragaria virginiana* Duchesne ssp. virginiana	Wild Strawberry	*	*	3
Rosaceae	*Malus coronaria* (L.) P. Mill. var. coronaria	Sweet Crabapple	*	*	3
Rosaceae	*Potentilla simplex* Michx.	Common Cinquefoil	*	*	4
Rosaceae	*Prunus serotina* Ehrh. var. serotina	Black Cherry	*	*	4
Rosaceae	***Rosa multiflora* Thunb. ex Murr.**	**Multiflora Rose**		*****	**0**
Rosaceae	*Rubus flagellaris* Willd.	Northern Dewberry	*	*	5
Rosaceae	*Rubus hispidus* L.	Bristly Dewberry	*	*	5
Rosaceae	*Rubus* spp.	Rubus	*	*	0
Rubiaceae	*Galium circaezans* Michx.	Licorice Bedstraw	*		6
Scrophulariaceae	***Veronica officinalis* L.**	**Common Gypsyweed**	*****	*****	**0**
Scrophulariaceae	***Veronica serpyllifolia* L. ssp. serpyllifolia**	**Thymeleaf Speedwell**	*****	*****	**0**
Smilacaceae	*Smilax glauca* Walt.	Cat Greenbrier	*		5
Smilacaceae	*Smilax rotundifolia* L.	Common Greenbriar		*	4
Solanaceae	*Solanum carolinense* L. var. carolinense	Carolina Horsenettle	*	*	3
Violaceae	*Viola blanda* Willd.	Sweet White Violet	*		5
Violaceae	*Viola sororia* Willd.	Common Blue Violet	*	*	4
Violaceae	*Viola* spp	Viola spp.	*	*	0
Vitaceae	*Parthenocissus quinquefolia* (L.) Planch.	Virginia Creeper	*		4
Vitaceae	*Vitis aestivalis* Michx.	Summer Grape	*		5

Appendix VIa. The most common shrub species identified, based on percent plot coverage, across Fort Necessity National Battlefield, Pennsylvania, USA, from 2005, 2008, and 2010 averaged. Any species with a total cover of 5% and greater were included in the table. Species in bold are exotic.

Family	Scientific name	Common name	Total cover	Reference	Treatment
Caprifoliaceae	***Lonicera morrowii* Gray**	**Morrow's honeysuckle**	**56.46**	**74.53**	**38.38**
Rosaceae	*Malus coronaria* (L.) P. Mill. var. coronaria	Sweet crabapple	11.41	12.50	10.31
Rosaceae	*Prunus serotina* Ehrh. var. serotina	Black cherry	8.33	14.17	2.50
Magnoliaceae	*Magnolia acuminata* (L.) L.	Cucumber magnolia	7.50	15.00	0.00
Fabaceae	*Robinia pseudoacacia* L.	Black locust	7.08	0.00	14.17
Caprifoliaceae	*Viburnum dentatum* L. var. dentatum	Southern arrowwood	5.83	2.50	9.17

Appendix VIIa. Mean ($\bar{X}$) and SE of floristic metrics measured at reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, during 2005, 2008, and 2010. *Post-removal surveys took place following removal procedures.

			***Lonicera morrowii* Percent cover**		**Native Shrub Percent Cover**		**Shrub Species Richness**		**Exotic Species Richness**		**Shannon-Wiener Diversity Index (H') - Herbs & Shrubs**	
		n	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Reference												
	2005	6	63.75	13.05	0.42	0.42	1.17	0.17	4.50	0.76	2.25	0.21
	2008	5	68.50	18.28	17.50	5.97	4.80	0.37	4.40	0.40	2.41	0.15
	2010	6	65.83	11.12	12.92	6.72	3.00	0.58	3.33	0.33	1.94	0.09
	Overall	17	65.88	7.56	9.85	3.20	2.88	0.43	4.06	0.33	2.18	0.10
Treatment												
	2005	6	69.58	7.84	19.17	7.60	2.50	0.43	6.50	0.61	2.65	0.09
	*2008	6	30.83	10.01	9.59	4.63	3.33	0.88	4.67	0.33	2.68	0.08
	*2010	6	8.33	3.00	10.83	5.73	2.67	0.99	4.33	0.21	2.60	0.11
	Overall	17	36.25	7.37	13.19	3.47	2.83	0.44	5.17	0.33	2.64	0.05

Appendix VIIa. Continued

	n	**Species Richness (S) Herbs & Shrubs**		**Species Evenness (J) - Herbs & Shrubs**		**Mean Coefficient of Conservatism (C)**		**Mean Floristic Quality Index (FQI)**	
		$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Reference									
2005	6	23.33	1.69	0.71	0.05	3.84	0.09	10.59	0.57
2008	5	24.40	2.32	0.76	0.04	2.74	0.19	6.92	0.63
2010	6	19.50	1.02	0.65	0.02	4.36	0.14	9.85	0.46
Overall	17	22.29	1.05	0.70	0.02	3.70	0.18	9.25	0.48
Treatment									
2005	6	32.67	1.94	0.76	0.02	3.80	0.14	11.30	0.86
*2008	6	24.33	1.74	0.84	0.02	2.44	0.14	5.46	0.51
*2010	6	19.33	1.87	0.88	0.02	3.40	0.30	6.99	1.05
Overall	17	25.44	1.67	0.83	0.02	3.21	0.18	7.90	0.75

Appendix VIIIa. The most common herbaceous species identified, based on percent plot coverage, across Fort Necessity National Battlefield, Pennsylvania, USA, from 2005, 2008, and 2010 averaged. Any species with a total cover of 5% and greater were included in the table. Species in bold are exotic.

Family	Scientific name	Common name	Total cover	Reference	Treatment
Poaceae	***Agrostis gigantea* Roth**	**Redtop grass**	**11.53**	**12.68**	**10.39**
Poaceae	*Danthonia compressa* Austin ex Peck	Flattened oatgrass	11.46	2.50	20.42
Asteraceae	*Solidago rugosa* P. Mill.	Wrinkle-leaf goldenrod	11.45	13.87	9.04
Asteraceae	*Solidago juncea* Ait.	Early goldenrod	9.64	12.10	7.19
Asteraceae	*Vernonia gigantea* (Walt.) Trel. ssp. gigantea	Giant ironweed	8.75	2.50	15.00
Asteraceae	*Solidago patula* Muhl. ex Willd. var. patula	Rough-leaved goldenrod	8.44	7.71	9.17
Convolvulaceae	***Calystegia sepium* (L.) R. Br. ssp. sepium**	**Hedge false bindweed**	**7.50**	**0.00**	**15.00**
Lycopodiaceae	*Lycopodium digitatum* Dill. ex A. Braun	Fan clubmoss	7.25	5.00	9.50
Rosaceae	*Rubus flagellaris* Willd.	Northern dewberry	7.03	7.40	6.67
Poaceae	***Dactylis glomerata* L. ssp. glomerata**	**Orchard grass**	**6.48**	**4.29**	**8.68**
Asteraceae	*Solidago canadensis* L.	Canada goldenrod	6.43	7.86	5.00
Poaceae	***Anthoxanthum odoratum* L. ssp. odoratum**	**Sweet vernal grass**	**6.06**	**4.84**	**7.28**
Asteraceae	***Achillea millefolium* L. var. occidentalis DC.**	**Yarrow**	**5.71**	**4.20**	**7.22**
Poaceae	***Holcus lanatus* L.**	**Common velvet grass**	**5.51**	**3.04**	**7.98**
Asteraceae	***Leucanthemum vulgare* Lam.**	**Ox-eye daisy**	**5.27**	**2.50**	**8.03**
Lamiaceae	*Clinopodium vulgare* L.	Wild basil	5.21	4.35	6.07

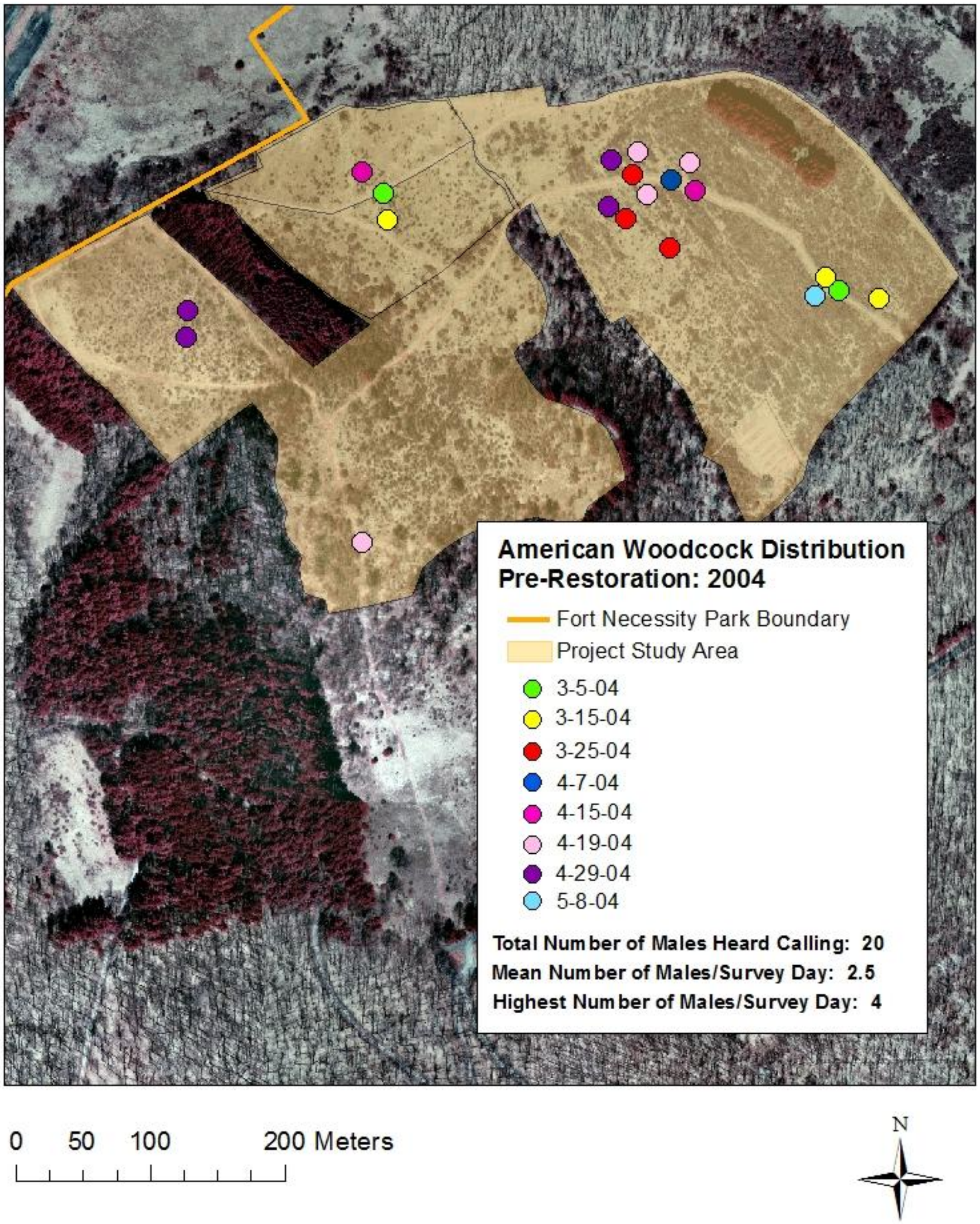
Appendix IXa: Overall mean ($\overline{X}$) and SE of male American woodcock for the overall reference and treatment areas at Fort Necessity National Battlefield, Pennsylvania, USA. We performed singing ground surveys from 2004 throughout 2010 during the winter and spring breeding months (February – May). *Post-removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		Total No. Males	**Highest No. of Males**	**Overall No. Males Heard Calling**	
	n	**Heard Calling/Year**	**Heard Calling/Survey**	$\overline{X}$	SE
Reference					
2004	8	6	2	0.75	0.25
2005	8	10	3	1.26	0.45
2006	6	1	1	0.18	0.17
2007	9	14	6	1.56	0.78
2008	8	7	3	0.87	0.40
2009	7	9	4	1.29	0.61
2010	15	30	5	2.01	0.40
Overall	61	77	6	1.27	0.15
Treatment					
2004	8	14	3	1.76	0.31
2005	8	14	3	1.76	0.41
2006	6	16	7	2.66	1.02
*2007	9	30	7	3.32	0.94
*2008	8	18	5	2.26	0.49
*2009	7	5	2	0.72	0.29
*2010	15	45	5	3.00	0.26
Overall	61	142	7	2.33	0.19

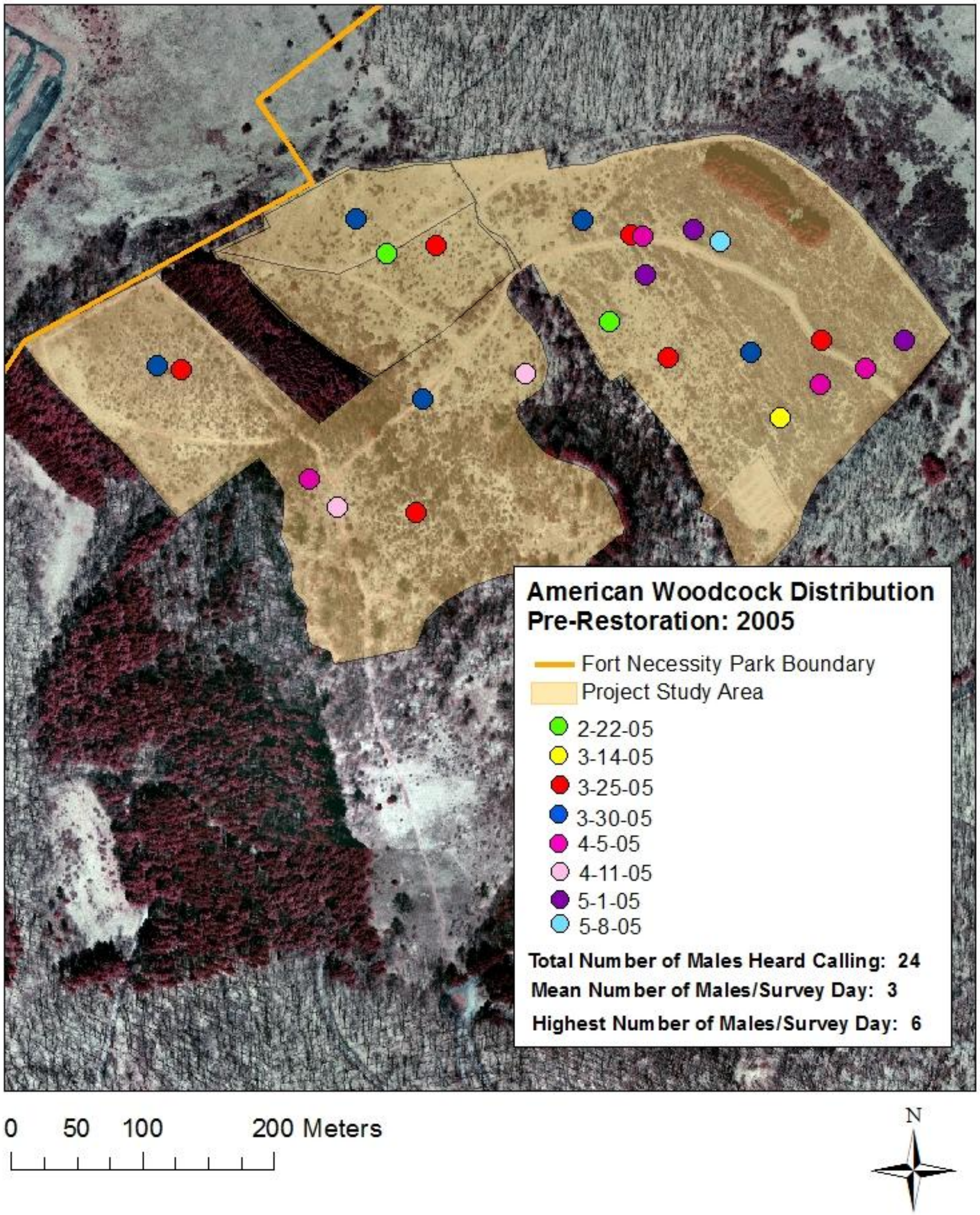
Appendix Xa: Mean ($\overline{X}$) and SE of male American woodcock for individual reference and treatment plots at Fort Necessity National Battlefield. We performed singing ground surveys from 2004 throughout 2010 during the winter and spring breeding months (February – May). *Post-removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		No. Males Heard Calling/Sample Plot	
	n	$\overline{X}$	SE
Reference			
2004	24	0.25	0.11
2005	24	0.42	0.12
2006	18	0.06	0.06
2007	27	0.52	0.17
2008	24	0.29	0.13
2009	21	0.43	0.15
2010	45	0.67	0.13
Overall	183	0.42	0.05
Treatment			
2004	16	0.88	0.18
2005	16	0.88	0.18
2006	12	1.33	0.40
*2007	18	1.66	0.36
*2008	16	1.13	0.20
*2009	14	0.36	0.13
*2010	30	1.50	0.14
Overall	122	1.17	0.09

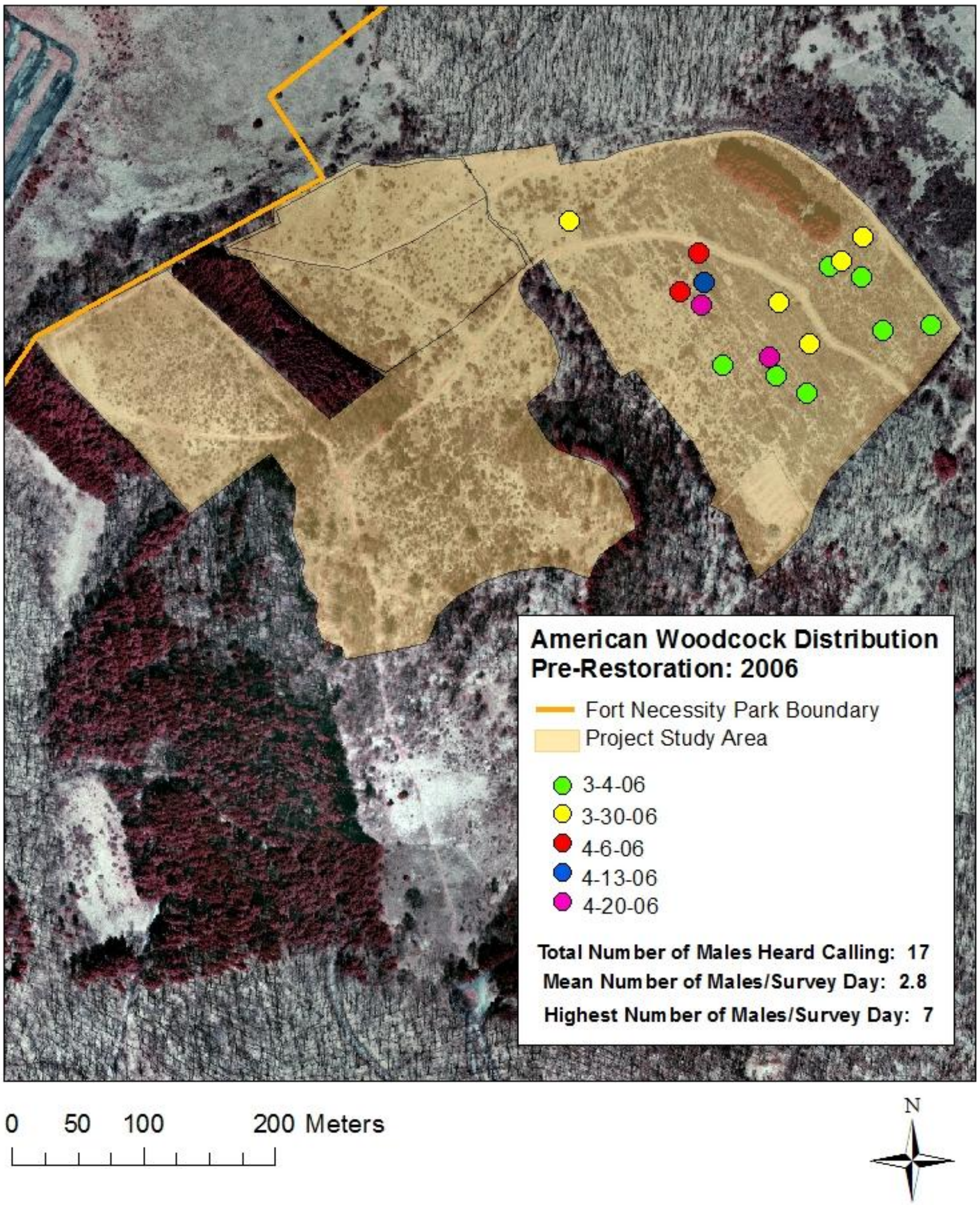
Appendix XIa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2004. Year 2004 represents a pre-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



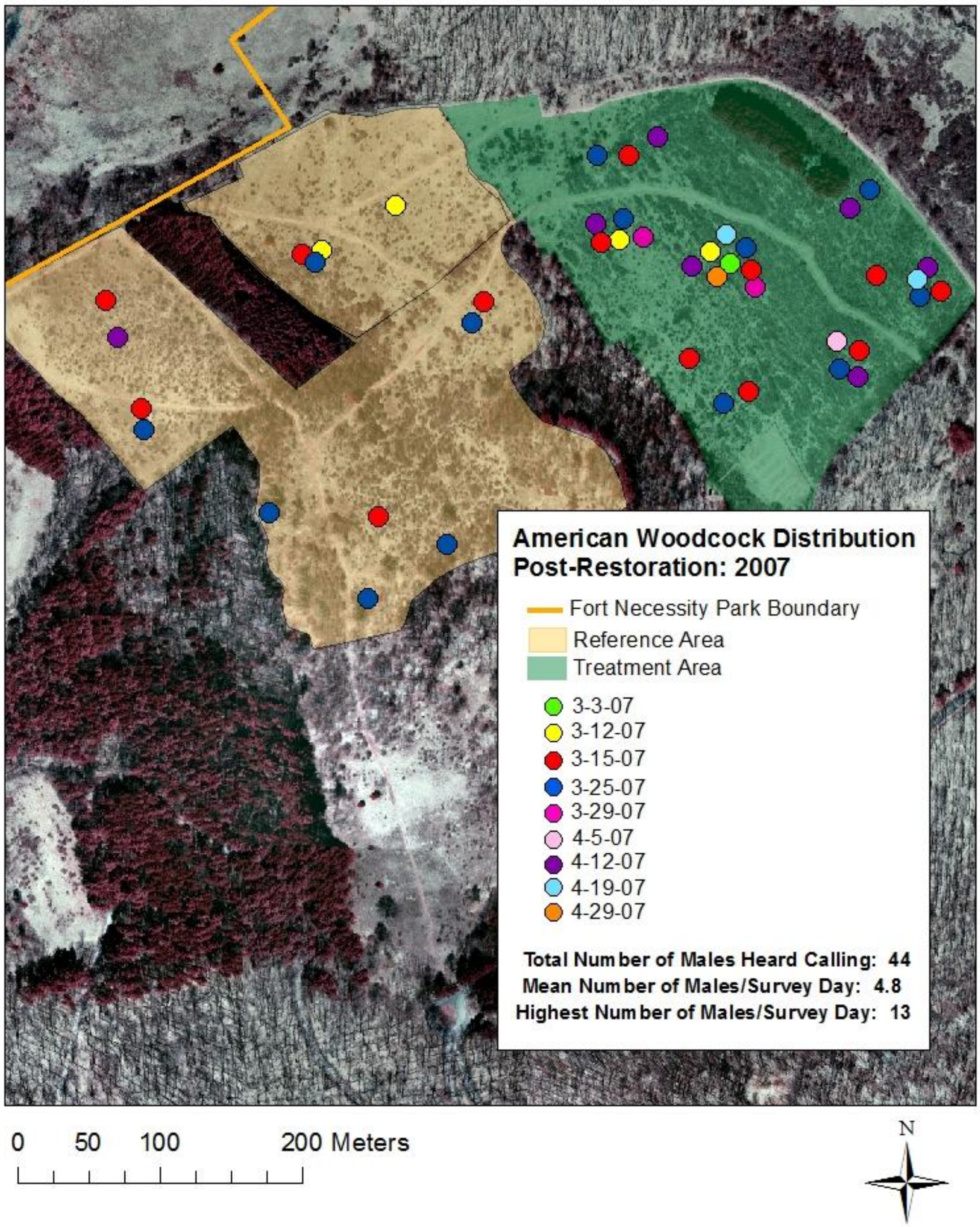
Appendix XIIa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2005. Year 2005 represents a pre-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



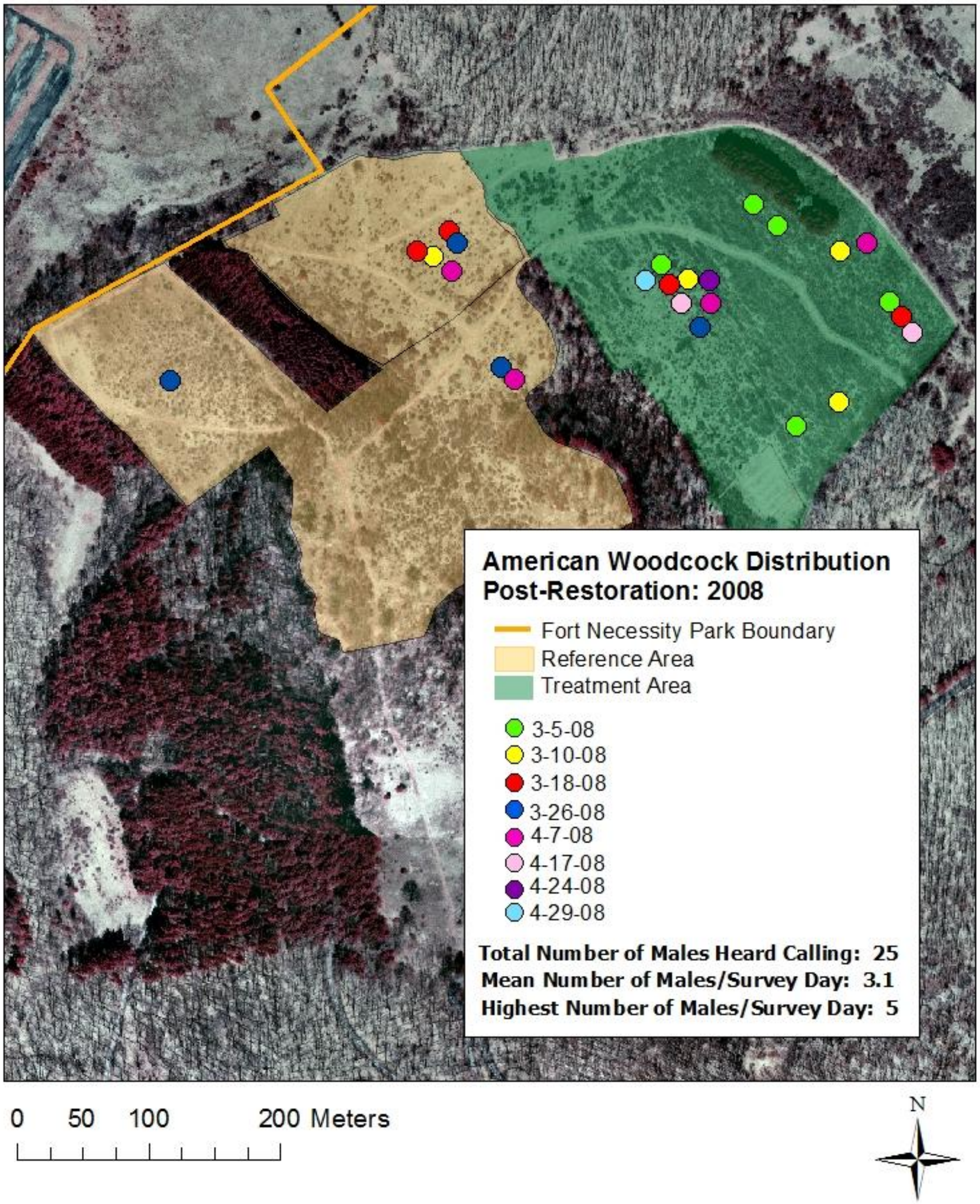
Appendix XIIIa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2006. Year 2006 represents a pre-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



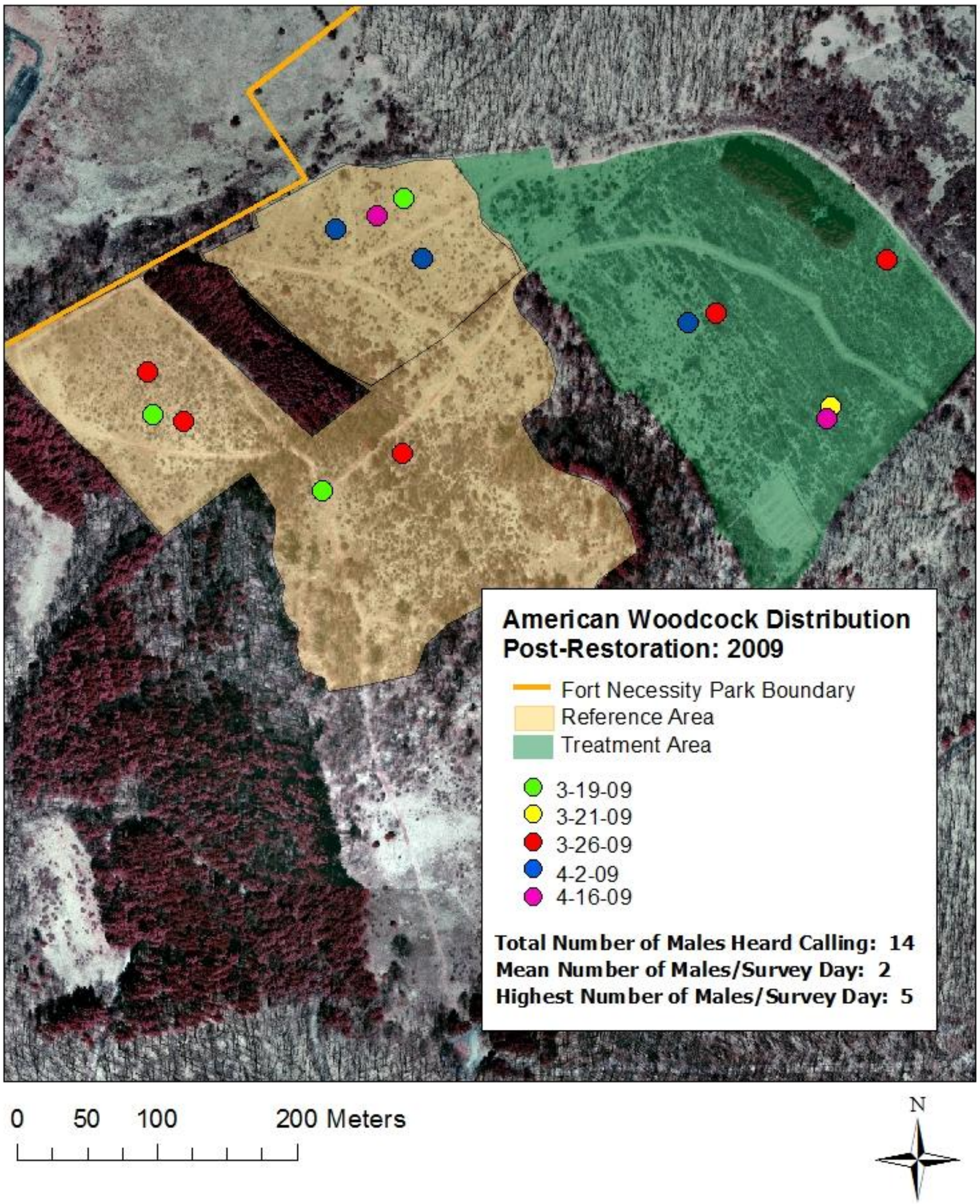
Appendix XIVa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2007. Year 2007 represents a post-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



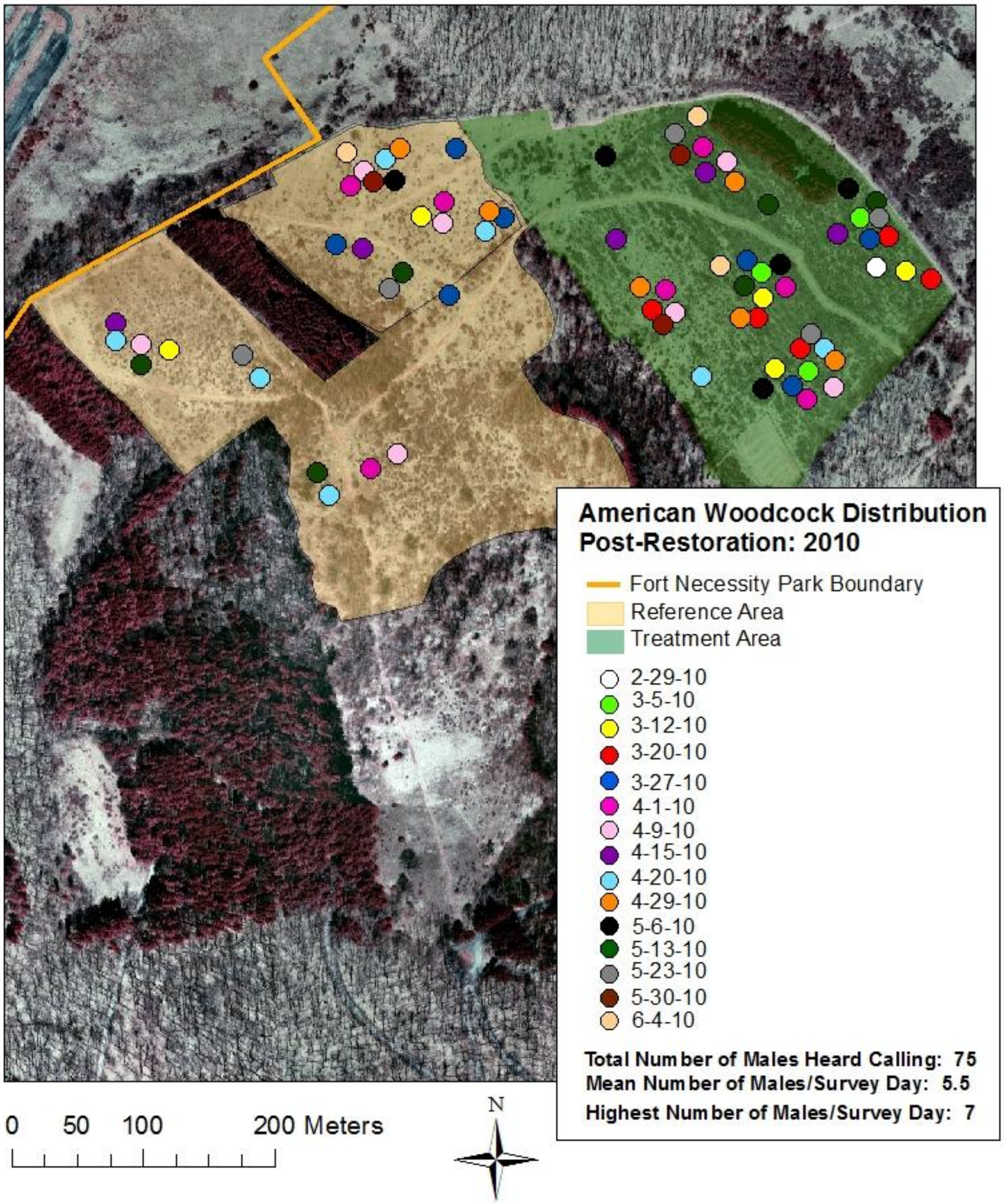
Appendix XVa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2008. Year 2008 represents a post-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



Appendix XVIa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2009. Year 2009 represents a post-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



Appendix XVIIa: Mapped locations of male American woodcock heard calling in reference and treatment plots at Fort Necessity National Battlefield, Pennsylvania, USA, by survey day, in year 2010. Year 2010 represents a post-removal year. Corresponding descriptive statistics are given: total number of males heard calling/year, mean number of males/survey day, and highest number of males/survey day.



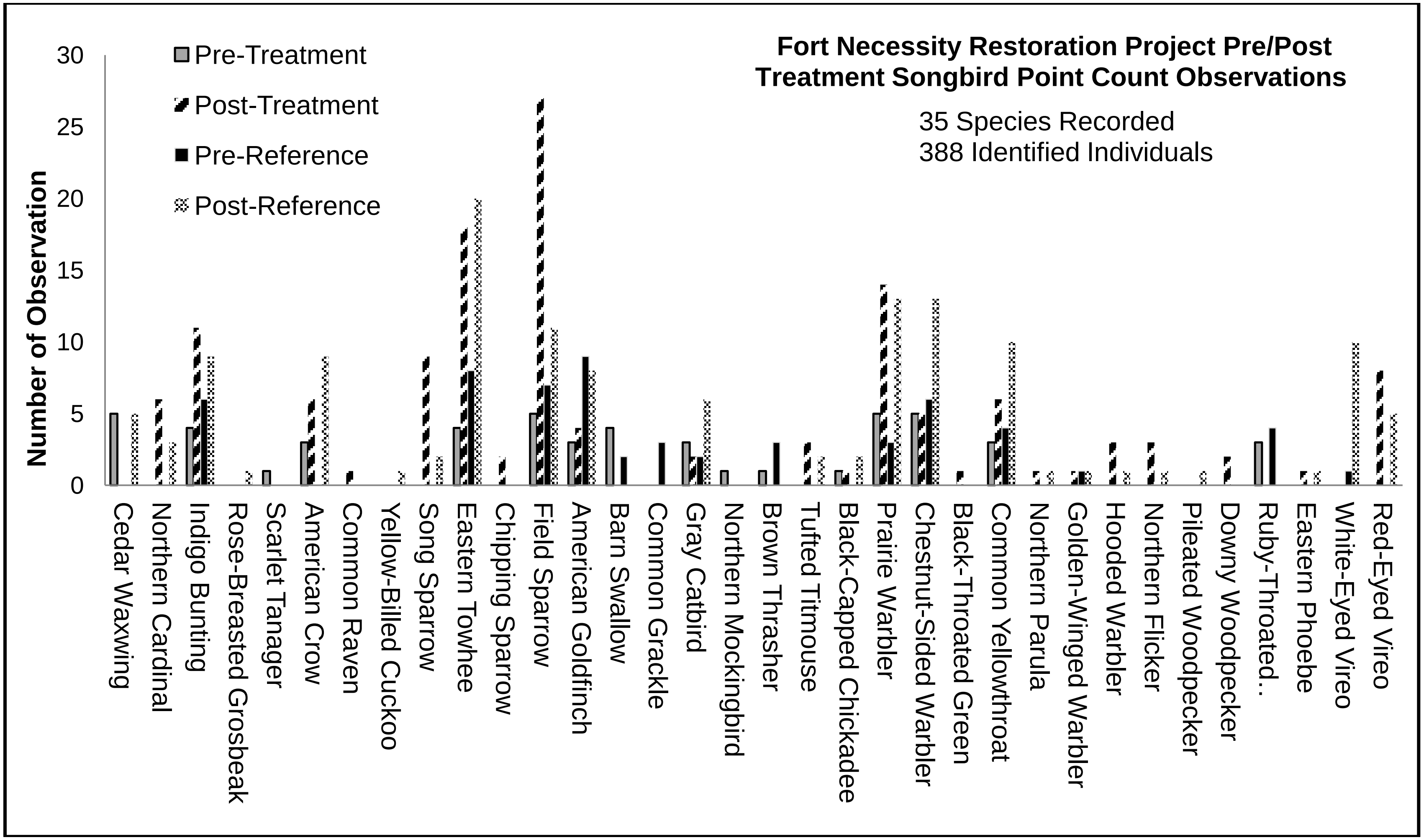
Appendix XVIIIa: Species of songbirds and their associated habitat guilds observed at Fort Necessity National Battlefield, Pennsylvania, USA, during 2004, 2008, and 2010 point counts surveys.

Family	**Scientific Name**	**Species Code**	**Species**
Early-Successional Habitat Guild			
Cardinalidae	*Cardinalis cardinalis*	Northern Cardinal	NOCA
Cardinalidae	*Passerina cyanea*	Indigo Bunting	INBU
Emberizidae	*Melospiza melodia*	Song Sparrow	SOSP
Emberizidae	*Pipilo erythrophthalmus*	Eastern Towhee	ETOW
Emberizidae	*Spizella pusilla*	Field Sparrow	FISP
Fringillidae	*Carduelis tristis*	American Goldfinch	AMGO
Hirundinidae	*Hirundo rustica*	Barn Swallow	BARS
Icteridae	*Quiscalus quiscula*	Common Grackle	COGR
Mimidae	*Dumetella carolinensis*	Grey Catbird	GRCA
Mimidae	*Mimus polyglottos*	Northern Mockingbird	NOMO
Mimidae	*Toxostoma rufum*	Brown Thrasher	BRTH
Parulidae	*Dendroica discolor*	Prairie Warbler	PRAW
Parulidae	*Dendroica pensylvanica*	Chestnut-sided Warbler	CSWA
Parulidae	*Geothlypis trichas*	Common Yellowthroat	COYE
Parulidae	*Vermivora chrysoptera*	Golden-winged Warbler	GWWA
Tyrannidae	*Sayornis phoebe*	Eastern Phoebe	EAPH
Vireonidae	*Vireo griseus*	White-eyed Vireo	WEVI
Generalist Habitat Guild			
Bombycillidae	*Bombycilla cedrorum*	Cedar Waxwing	CEDW
Cardinalidae	*Pheucticus ludovicianus*	Rose-breasted Grosbeak	RBGR
Corvidae	*Corvus brachyrhynchos*	American Crow	AMCR
Corvidae	*Corvus corax*	Common Raven	CORA
Cuculidae	*Coccyzus americanus*	Yellow-billed Cuckoo	YBCU
Emberizidae	*Spizella passerina*	Chipping Sparrow	CHSP

Appendix XVIIIa. Continued

Family	Scientific Name	Species Code	Species
Generalist Habitat Guild			
Paridae	*Baeolophus bicolor*	Tufted Titmouse	ETTI
Paridae	*Poecile atricapilla*	Black-capped Chickadee	BCCH
Parulidae	*Parula americana*	Northern Parula	NOPA
Parulidae	*Wilsonia citrina*	Hooded Warbler	HOWA
Picidae	*Colaptes auratus*	Northern Flicker	NOFL
Picidae	*Picoides pubescens*	Downy Woodpecker	DOWO
Trochilidae	*Archilochus colubris*	Ruby-throated Hummingbird	RTHU
Vireonidae	*Vireo olivaceus*	Red-eyed Vireo	REVI
Late-Successional Habitat Guild			
Cardinalidae	*Piranga olivacea*	Scarlet Tananger	SCTA
Parulidae	*Dendroica virens*	Black-throated Green Warbler	BTGW
Picidae	*Dryocopus pileatus*	Pileated Woodpecker	PIWO

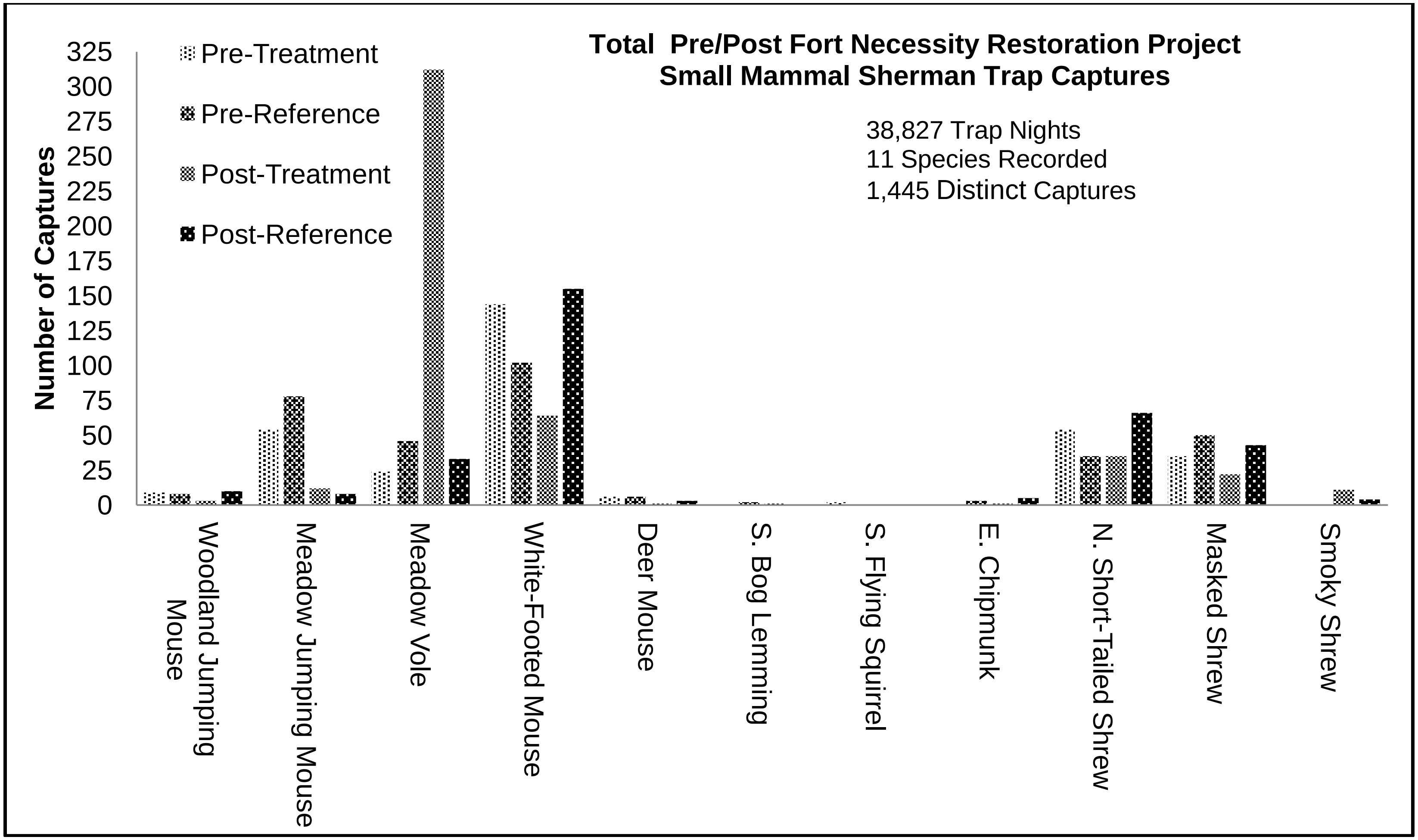
Appendix XIXa: Total songbird species observations during point count surveys at Fort Necessity National Battlefield, Pennsylvania, USA, during 2004, 2008, and 2010. Observations are categorized based on plot type and timing of restoration (Pre-T, Post-T, Pre-R, Post-R). Code is as follows: T=treatment, R=Reference, Pre = 2004, Post = 2008, 2010.



Appendix XXa: Mean ($\overline{X}$) and SE of songbird metrics measured at reference and treatment point count locations at Fort Necessity National Battlefield, Pennsylvania, USA. We performed point count surveys during the 2004, 2008, and 2010 breeding period. Numbers of observations of the 2 most common species captured are listed using the following species codes: Eastern towhee (EATO), and Field sparrow (FISP). *Post removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		Proportion of Early Successional species		**Proportion of Generalist Species**		**Shannon-Wiener Diversity Index (H')**		**Species Richness (S)**		**Species Evenness (J)**		**EATO Observations (per 5 minute count)**		**FISP Observations (per 5 minute count)**	
	n	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Reference															
2004	6	0.92	0.04	0.08	0.04	1.62	0.07	5.67	0.49	0.94	0.13	1.33	0.61	1.17	0.54
2008	3	0.80	0.10	0.20	0.20	2.11	0.07	9.33	0.88	0.95	0.01	2.33	0.88	1.00	0.00
2010	6	0.78	0.06	0.21	0.05	2.18	0.08	10.00	0.73	0.95	0.01	2.17	0.54	1.33	0.56
Overall	15	0.84	0.04	0.16	0.03	1.94	0.08	8.13	0.65	0.95	0.01	1.87	0.36	1.20	0.30
Treatment															
2004	6	0.83	0.09	0.15	0.10	1.65	0.14	6.17	0.87	0.94	0.02	0.67	0.33	0.83	0.31
*2008	3	0.77	0.02	0.23	0.02	2.08	0.09	8.67	0.88	0.97	0.01	2.33	0.33	2.33	0.33
*2010	6	0.77	0.04	0.22	0.03	2.07	0.14	9.50	0.85	0.92	0.04	1.83	0.48	3.33	1.17
Overall	15	0.79	0.04	0.20	0.04	1.90	0.10	8.00	0.63	0.94	0.02	1.47	0.29	2.13	0.55

Appendix XXIa: Total small mammal species captures during Sherman trapping at Fort Necessity National Battlefield, Pennsylvania, USA, from 2004 – 2010. Observations are categorized based on plot type and timing of restoration (Pre-T, Post-T, Pre-R, Post-R). Code is as follows: T=treatment, R=Reference, Pre = 2005, Post = 2008, 2010.



Appendix XXIIa: List of mammal species and their associated observation method at Fort Necessity National Battlefield, Pennsylvania, USA, 2004 – 2010.

Family	**Scientific Name**	**Species**	**Species Code**	**Observation Method**
Canidae	Red Fox	*Vulpes vulpes*	VUVU	Observation In Study Plot
Didelphidae	Virginia Opossum	*Didelphis virginianus*	DIVI	Pitfall/Tomahawk
Dipodidae	Meadow Jumping Mouse	*Zapus hudsonius*	ZAHU	Pitfall/Sherman
Dipodidae	Woodland Jumping Mouse	*Napaeozapus insignis*	NAIN	Pitfall/Sherman
Muridae	Deer Mouse	*Peromyscus maniculatus*	PEME	Pitfall/Sherman
Muridae	Meadow Vole	*Microtus pennsylvanicus*	MIPE	Pitfall/Sherman
Muridae	Pine Vole	*Microtus pinetorum*	MIPI	Sherman
Muridae	Southern Bog Lemming	*Synaptomys cooperi*	SYCO	Pitfall/Sherman
Muridae	White-footed Mouse	*Peromyscus leucopus*	PELE	Pitfall/Sherman
Procyonidae	Raccoon	*Procyon lotor*	PRLO	Tomahawk
Sciuridae	Eastern Chipmonk	*Tamias striatus*	TAST	Sherman
Sciuridae	Southern Flying Squirrel	*Glaucomys volans*	GLVO	Sherman
Soricidae	Masked Shrew	*Sorex cinereus*	SOCI	Pitfall/Sherman
Soricidae	Short-tailed Shrew	*Blarina brevicauda*	BLBR	Pitfall/Sherman
Soricidae	Smoky Shrew	*Sorex fumeus*	SOFU	Pitfall/Sherman
Talpidae	Hairy-tail Mole	*Parascalops breweri*	PABR	Observation In Study Plot
Talpidae	Star-nosed Mole	*Condylura cristata*	COCR	Observation In Study Plot

Appendix XXIIIa: Mean ($\overline{X}$) and SE of small mammal relative abundance (No. of captures/100 trap nights) and metrics measured at reference and treatment trapping grids at Fort Necessity National Battlefield, Pennsylvania, USA. We performed Sherman live trapping from 2004 through 2010 during the summer season from May-August. *Post-removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		White-footed Mouse *P. leucopus*		**Deer Mouse** *P. maniculatus*		**Meadow Vole** *M. pennsylvanicus*		**Masked Shrew** *S. cinereus*		**Smoky Shrew** *S. fumeus*	
	n	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Reference											
2004	5	0.54	0.08	0.00	0.00	0.34	0.11	0.24	0.11	0.00	0.00
2005	5	1.34	0.21	0.00	0.00	1.02	0.24	1.46	0.39	0.00	0.00
2006	6	1.83	0.30	0.23	0.14	0.30	0.15	0.08	0.08	0.00	0.00
2007	6	0.62	0.18	0.08	0.05	0.08	0.05	0.28	0.14	0.03	0.03
2008	6	0.32	0.11	0.00	0.00	0.05	0.05	0.82	0.36	0.10	0.06
2009	6	2.67	0.51	0.05	0.05	0.62	0.25	0.60	0.14	0.05	0.05
2010	6	2.25	0.34	0.00	0.00	0.48	0.24	0.07	0.04	0.00	0.00
Overall	40	1.39	0.17	0.06	0.03	0.40	0.09	0.49	0.10	0.03	0.01
Treatment											
2004	5	1.34	0.12	0.00	0.00	0.10	0.06	0.36	0.21	0.00	0.00
2005	5	2.16	0.65	0.00	0.00	0.38	0.24	0.74	0.19	0.00	0.00
2006	6	1.65	0.12	0.20	0.10	0.33	0.10	0.13	0.04	0.00	0.00
*2007	6	0.97	0.29	0.03	0.03	0.05	0.05	0.12	0.08	0.00	0.00
*2008	6	0.28	0.17	0.00	0.00	0.00	0.00	0.07	0.04	0.05	0.05
*2009	6	0.53	0.30	0.00	0.00	1.32	0.23	0.62	0.37	0.15	0.09
*2010	6	0.43	0.10	0.00	0.00	8.45	2.21	0.05	0.05	0.17	0.08
Overall	40	1.02	0.14	0.04	0.02	1.58	0.56	0.29	0.07	0.06	0.02

Appendix XXIIIa. Continued

		Short-tailed Shrew *B. brevicauda*		Meadow Jumping Mouse *Z. hudsonius*		Woodland Jumping Mouse *N. insignis*		Eastern Chipmunk *T. striatus*	
	n	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Reference									
2004	5	0.62	0.28	1.64	0.83	0.00	0.00	0.00	0.00
2005	5	0.38	0.24	1.02	0.29	0.30	0.18	0.13	0.08
2006	6	0.28	0.10	0.08	0.05	0.00	0.00	0.00	0.00
2007	6	0.18	0.09	0.05	0.05	0.00	0.00	0.15	0.10
2008	6	0.93	0.33	0.05	0.05	0.00	0.00	0.03	0.03
2009	6	0.88	0.39	0.22	0.22	0.20	0.16	0.00	0.00
2010	6	0.53	0.14	0.03	0.03	0.17	0.11	0.03	0.03
Overall	40	0.55	0.10	0.40	0.14	0.09	0.04	0.05	0.02
Treatment									
2004	5	0.54	0.23	0.70	0.36	0.00	0.00	0.00	0.00
2005	5	0.78	0.30	1.12	0.20	0.34	0.21	0.00	0.00
2006	6	0.60	0.28	0.08	0.05	0.00	0.00	0.00	0.00
*2007	6	0.07	0.04	0.20	0.09	0.03	0.03	0.00	0.00
*2008	6	0.37	0.22	0.03	0.03	0.00	0.00	0.04	0.04
*2009	6	0.43	0.16	0.03	0.03	0.08	0.08	0.00	0.00
*2010	6	0.33	0.16	0.15	0.07	0.00	0.00	0.00	0.00
Overall	40	0.44	0.08	0.30	0.08	0.06	0.03	0.01	0.01

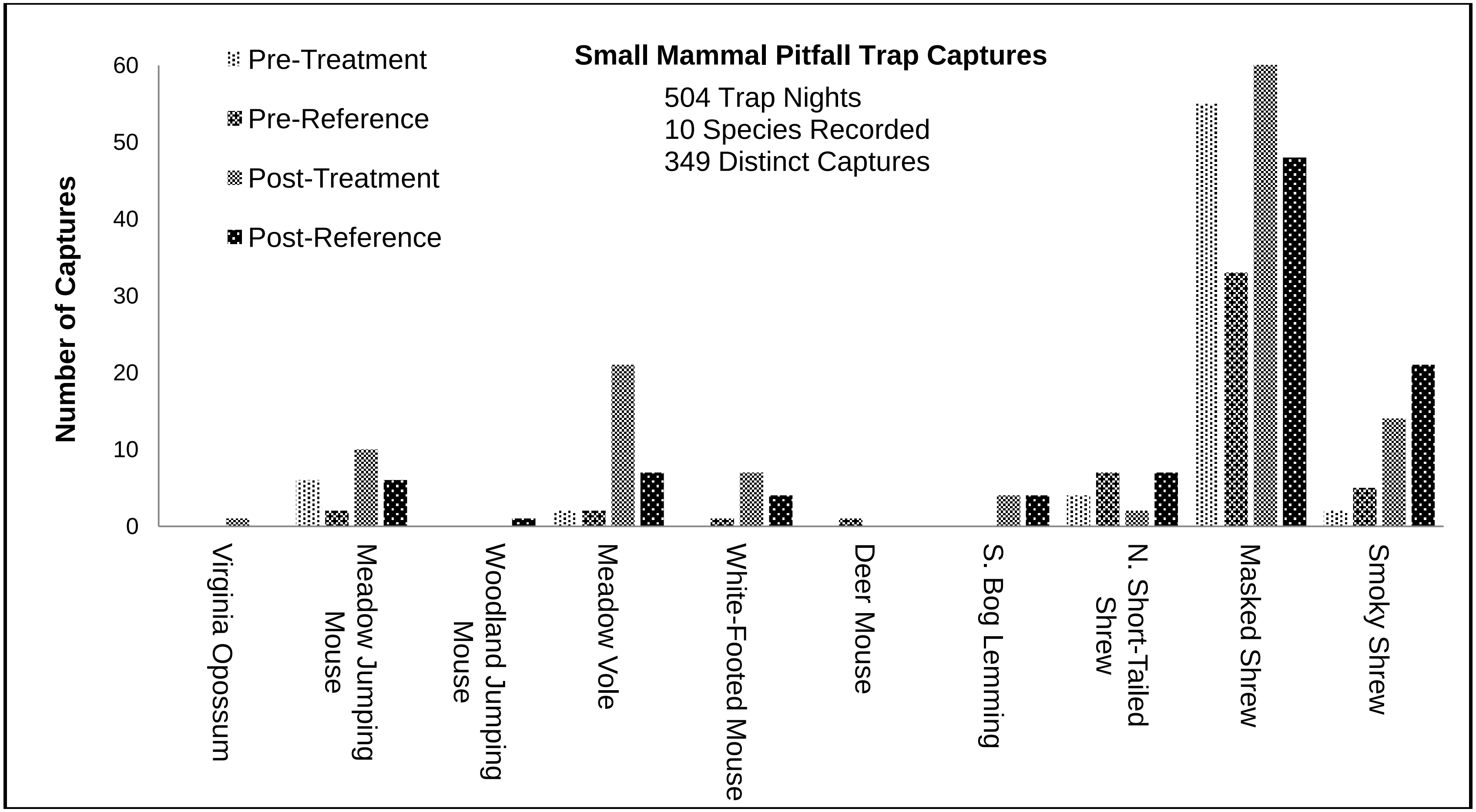
Appendix XXIIIa. Continued

		Southern Flying Squirrel *G. volans*		Southern Bog Lemming *S. cooperi*		Shannon-Wiener Index *H'*		Species Richness *S*	
	n	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Reference									
2004	5	0.00	0.00	0.00	0.00	1.05	0.12	3.50	0.65
2005	5	0.00	0.00	0.04	0.04	1.52	0.12	5.25	0.75
2006	6	0.00	0.00	0.00	0.00	0.86	0.07	3.20	0.37
2007	6	0.00	0.00	0.00	0.00	0.98	0.10	3.20	0.31
2008	6	0.00	0.00	0.00	0.00	0.84	0.20	0.20	0.63
2009	6	0.00	0.00	0.00	0.00	1.08	0.17	4.33	0.84
2010	6	0.00	0.00	0.00	0.00	0.91	0.17	3.67	0.42
Overall	40	0.00	0.00	0.01	0.01	1.02	0.06	3.75	0.23
Treatment									
2004	5	0.00	0.00	0.00	0.00	1.00	0.23	2.67	0.67
2005	5	0.00	0.00	0.00	0.00	1.35	0.08	5.33	0.33
2006	6	0.10	0.10	0.00	0.00	1.08	0.12	4.00	0.71
*2007	6	0.00	0.00	0.00	0.00	0.75	0.22	3.00	0.58
*2008	6	0.00	0.00	0.00	0.00	0.47	0.17	2.20	0.37
*2009	6	0.00	0.00	0.03	0.03	1.07	0.16	4.00	0.68
*2010	6	0.00	0.00	0.00	0.00	0.54	0.13	3.40	0.60
Overall	40	0.01	0.01	0.01	0.01	0.88	0.07	3.58	0.23

Appendix XXIIIa. Continued

	n	Species Evenness *J*		Total Relative Abundance	
		$\overline{X}$	SE	$\overline{X}$	SE
Reference					
2004	5	0.85	0.07	3.44	0.99
2005	5	0.91	0.02	5.76	0.93
2006	6	0.78	0.04	2.80	0.41
2007	6	0.87	0.03	1.47	0.22
2008	6	0.74	0.15	2.20	0.71
2009	6	0.78	0.03	5.27	1.05
2010	6	0.69	0.07	3.57	0.18
Overall	40	0.80	0.03	3.45	0.33
Treatment					
2004	5	0.80	0.08	3.06	0.84
2005	5	0.84	0.02	5.64	0.88
2006	6	0.77	0.04	3.12	0.39
*2007	6	0.66	0.16	1.43	0.27
*2008	6	0.52	0.18	0.85	0.27
*2009	6	0.81	0.03	3.20	0.92
*2010	6	0.41	0.07	9.65	2.12
Overall	40	0.68	0.04	3.83	0.57

Appendix XXIVa: Total small mammal species captures during pitfall arrays at Fort Necessity National Battlefield, Pennsylvania, USA, from 2004 – 2010. Observations are categorized based on plot type and timing of restoration (Pre-T, Post-T, Pre-R, Post-R). Code is as follows: T=treatment, R=Reference, Pre = 2005, Post = 2008, 2010.



Appendix XXVa: Mean ($\overline{X}$) and SE of small mammal relative abundance (No. of captures/100 trap nights) and metrics measured at reference and treatment pitfall arrays at Fort Necessity National Battlefield, Pennsylvania, USA. We trapped using the pitfall arrays from 2004 through 2010 during the summer season from May-August. *Post-removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		White-footed Mouse *P. leucopus*		**Deer Mouse** *P. maniculatus*		**Meadow Vole** *M. pennsylvanicus*		**Masked Shrew** *S. cinereus*		**Smoky Shrew** *S. fumeus*	
	n	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Reference											
2004	9	0.00	0.00	0.00	0.00	2.78	2.78	50.00	12.50	2.78	2.78
2005	9	0.00	0.00	0.00	0.00	2.78	2.78	27.78	10.58	11.11	6.05
2006	9	2.78	2.78	2.78	2.78	0.00	0.00	16.67	7.22	0.00	0.00
2007	9	0.00	0.00	0.00	0.00	0.00	0.00	25.00	11.78	11.11	6.05
2008	9	5.56	5.56	0.00	0.00	2.78	2.78	36.11	14.50	11.11	11.11
2009	9	2.78	2.78	0.00	0.00	11.11	8.45	63.89	26.72	27.78	12.11
2010	9	2.78	2.78	0.00	0.00	5.56	3.67	8.33	5.89	8.33	8.33
Overall	63	1.98	1.03	0.40	0.40	3.57	1.48	32.54	5.60	10.32	2.97
Treatment											
2004	9	0.00	0.00	0.00	0.00	0.00	0.00	47.22	15.28	2.78	2.78
2005	9	0.00	0.00	0.00	0.00	2.78	2.78	41.67	21.25	2.78	2.78
2006	9	0.00	0.00	0.00	0.00	2.78	2.78	63.89	13.89	0.00	0.00
*2007	9	8.33	4.17	0.00	0.00	5.56	3.67	38.89	11.87	2.78	2.78
*2008	9	2.78	2.78	0.00	0.00	0.00	0.00	61.11	27.67	11.11	8.45
*2009	9	0.00	0.00	0.00	0.00	25.00	9.32	72.22	26.82	16.67	11.02
*2010	9	8.33	5.89	0.00	0.00	27.78	10.58	27.78	13.47	8.33	5.89
Overall	63	2.78	1.15	0.00	0.00	9.13	2.49	50.40	7.28	6.35	2.26

Appendix XXVa. Continued

	n	**Short-tailed Shrew** *B. brevicauda*		**Meadow Jumping Mouse** *Z. hudsonius*		**Woodland Jumping Mouse** *N. insignis*		**Southern Bog Lemming** *S.cooperi*	
		$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Reference									
2004	9	2.78	2.78	2.78	2.78	0.00	0.00	0.00	0.00
2005	9	13.89	7.35	2.78	2.78	0.00	0.00	0.00	0.00
2006	9	2.78	2.78	0.00	0.00	0.00	0.00	0.00	0.00
2007	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2008	9	8.33	5.89	5.56	3.67	0.00	0.00	2.78	2.78
2009	9	0.00	0.00	11.11	7.35	2.78	2.78	8.33	5.89
2010	9	11.11	8.45	0.00	0.00	0.00	0.00	0.00	0.00
Overall	63	5.56	1.91	3.17	1.33	0.40	0.40	1.59	0.96
Treatment									
2004	9	2.78	2.78	13.89	11.11	0.00	0.00	0.00	0.00
2005	9	8.33	4.17	2.78	2.78	0.00	0.00	0.00	0.00
2006	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
*2007	9	2.78	2.78	0.00	0.00	0.00	0.00	0.00	0.00
*2008	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
*2009	9	2.78	2.78	0.00	0.00	0.00	0.00	11.11	8.45
*2010	9	0.00	0.00	27.78	19.30	0.00	0.00	0.00	0.00
Overall	63	2.38	0.93	6.35	3.30	0.00	0.00	1.59	1.24

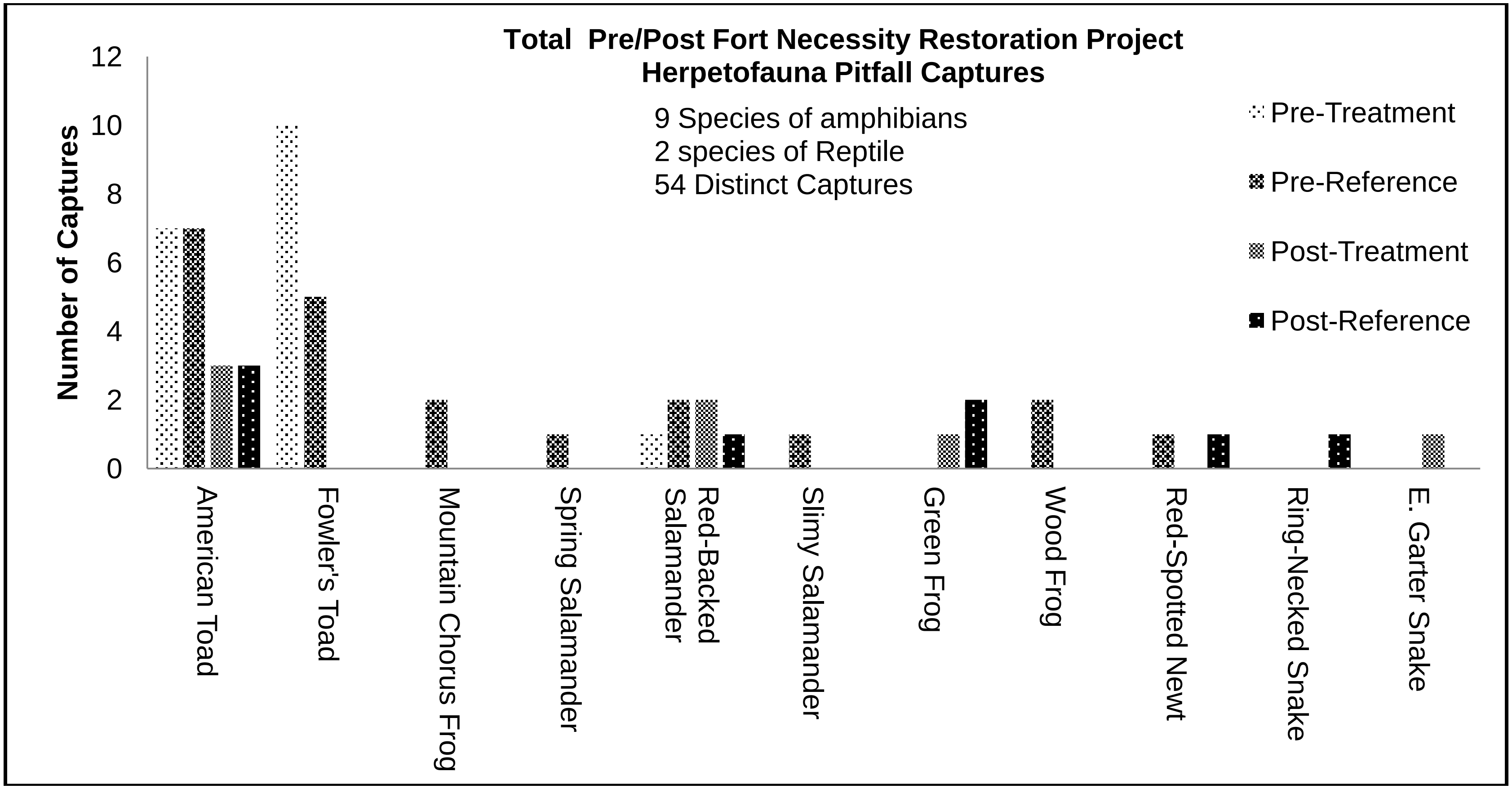
Appendix XXVa. Continued

		Virginia Opossum *D. virginiana*		Shannon-Wiener Index *H'*		Species Richness *S*		Species Evenness *J*		Total Relative Abundance	
	n	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Reference											
2004	9	0.00	0.00	0.20	0.10	1.22	0.22	0.29	0.15	58.33	13.18
2005	9	0.00	0.00	0.45	0.15	1.44	0.41	0.53	0.17	47.22	19.30
2006	9	0.00	0.00	0.08	0.08	0.78	0.22	0.11	0.11	25.00	7.22
2007	9	0.00	0.00	0.15	0.10	0.78	0.28	0.21	0.14	36.11	12.58
2008	9	0.00	0.00	0.28	0.15	1.44	0.38	0.31	0.16	72.22	25.83
2009	9	0.00	0.00	0.60	0.20	2.11	0.65	0.49	0.16	127.78	49.90
2010	9	0.00	0.00	0.22	0.14	0.89	0.42	0.20	0.13	36.11	19.59
Overall	63	0.00	0.00	0.28	0.05	1.24	0.15	0.31	0.06	57.54	9.79
Treatment											
2004	9	0.00	0.00	0.15	0.10	1.11	0.20	0.21	0.14	66.66	22.44
2005	9	0.00	0.00	0.24	0.12	1.22	0.43	0.28	0.14	58.33	28.87
2006	9	0.00	0.00	0.00	0.00	0.89	0.11	0.00	0.00	66.66	12.50
*2007	9	2.78	2.78	0.31	0.13	1.56	0.24	0.39	0.16	61.11	12.58
*2008	9	0.00	0.00	0.20	0.10	1.22	0.22	0.29	0.15	75.00	27.32
*2009	9	0.00	0.00	0.53	0.15	1.78	0.49	0.57	0.14	127.00	42.58
*2010	9	0.00	0.00	0.56	0.16	1.89	0.45	0.60	0.15	100.00	30.33
Overall	63	0.40	0.40	0.29	0.05	1.38	0.13	0.33	0.05	79.37	10.17

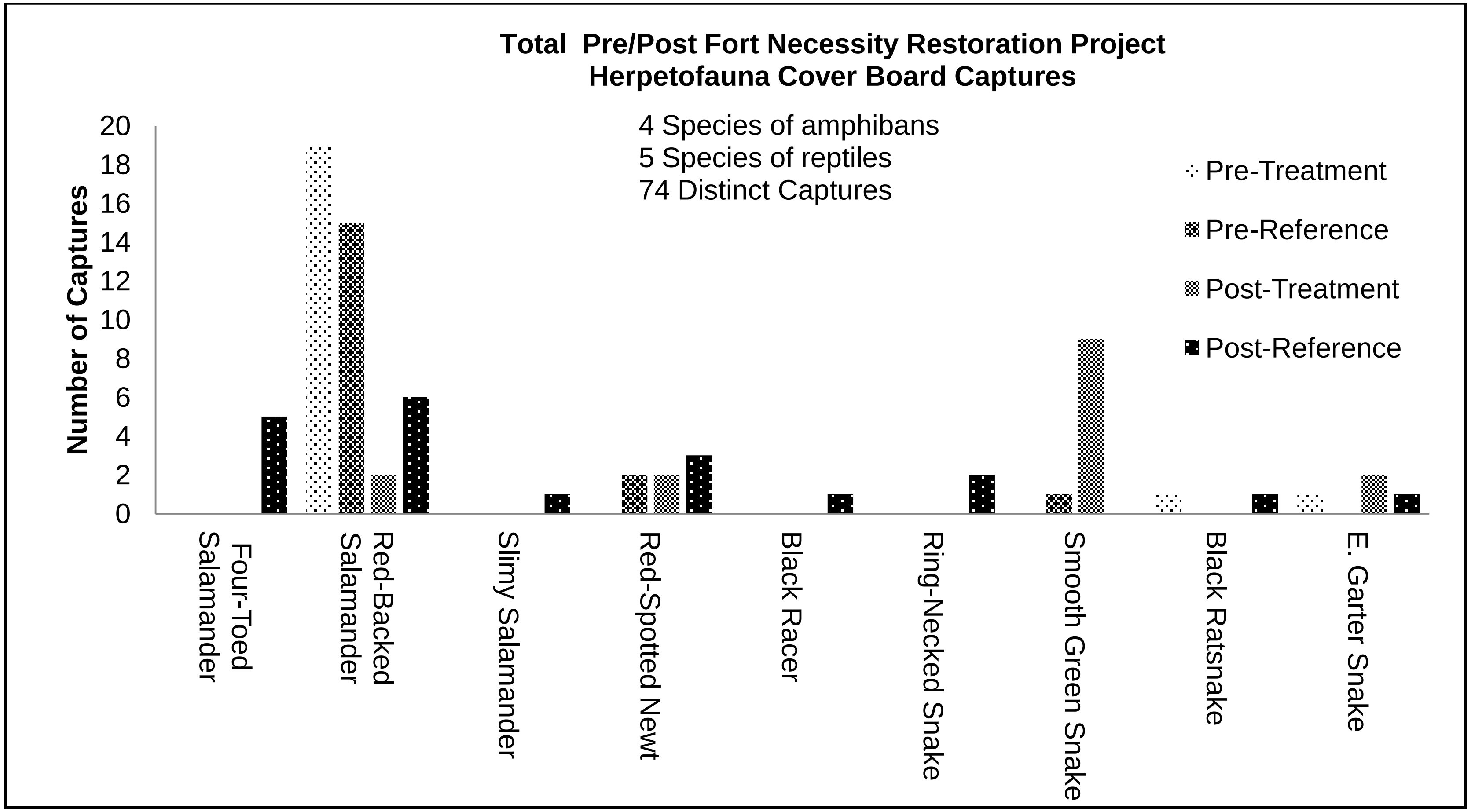
Appendix XXVIa: List of Amphibian and Reptile species and their associated observation method at Fort Necessity National Battlefield, Pennsylvania, USA, from 2004 – 2010.

Family	Scientific Name	Species	Species Code	Observation Method
Amphibians				
Ambystomatidae	*Ambystoma jeffersonianum*	Jefferson Salamander	AMJE	FONE Forest
Ambystomatidae	*Ambystoma opacum*	Marbled Salamander	AMOP	FONE Forest
Bufonidae	*Anaxyrus americanus*	American Toad	ANAM	Pitfall Array
Bufonidae	*Anaxyrus fowleri*	Fowler's Toad	ANFO	Pitfall Array
Hylidae	*Acris crepitans*	Northern Cricket Frog	ACCR	FONE Visitor's Center
Hylidae	*Pseudacris brachyphona*	Mountain Chorus Frog	PSBR	Pitfall Array
Plethodontidae	*Gyrinophilus p. porphyriticus*	Northern Spring Salamander	GYPO	Pitfall Array
Plethodontidae	*Hemidactylium scutatum*	Four-toed Salamander	HESC	Cover Board
Plethodontidae	*Plethodon cinereus*	Redback salamander	PLCI	Cover Board/Pitfall Array
Plethodontidae	*Plethodon glutinosus*	Northern Slimy Salamander	PLGL	Cover Board Array/Pitfall Array
Ranidae	*Lithobates clamitans melanotus*	Green Frog	LICL	Pitfall Array
Ranidae	*Lithobates sylvaticus*	Wood Frog	LISY	Pitfall Array
Salamandridae	*Notophthalmus v. viridescens*	Red-spotted Newt	NOVI	Cover Board/Pitfall Array
Reptiles				
Colubridae	*Coluber constrictor*	Northern Black Racer	COCO	Cover Board
Colubridae	*Diadophis punctatus edwardsii*	Northern Ringneck Snake	DIPU	Cover Board/Pitfall Array
Colubridae	*Opheodrys vernalis*	Smooth Greensnake	OPVE	Cover Board
Colubridae	*Scotophis alleghaniensis*	Black Ratsnake	SCAL	Cover Board
Colubridae	*Thamnophis s. sirtalis*	Eastern Gartersnake	THIS	Cover Board/Pitfall Array

Appendix XXVIIa: Total herpetofauna species captures during pitfall arrays at Fort Necessity National Battlefield from 2004 – 2010. Observations are categorized based on plot type and timing of restoration (Pre-T, Post-T, Pre-R, Post-R). Code is as follows: T=treatment, R=Reference, Pre = 2005, Post = 2008, 2010.



Appendix XXVIIIa: Total herpetofauna species captures during cover board flips at Fort Necessity National Battlefield from 2004 – 2010. Observations are categorized based on plot type and timing of restoration (Pre-T, Post-T, Pre-R, Post-R). T=treatment, R=Reference, Pre = 2005, Post = 2008, 2010.



Appendix XXIXa: Mean ($\overline{X}$) and SE of herpetofauna relative abundance (No. of captures/100 trap nights) at reference and treatment pitfall arrays at Fort Necessity National Battlefield, Pennsylvania, USA. We trapped using the pitfall arrays from 2004 through 2010 during the summer season from May-August. *Post-removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		American Toad *A.americanus*		**Fowler's Toad** *A. fowleri*		**Green Frog** *L. c. melanotus*		**Mountain Chorus Frog** *P. brachyphona*	
	n	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Pre-removal Reference	27	6.48	3.62	4.63	3.7	0.00	0.00	1.85	1.85
Pre-removal Treatment	27	6.48	4.56	9.26	6.72	0.00	0.00	0.00	0.00
Post-removal Reference	36	2.08	1.09	0.00	0.00	1.39	0.94	0.00	0.00
Post-removal Treatment	36	2.08	1.09	0.00	0.00	0.69	0.69	0.00	0.00
Overall Reference	63	1.07	0.53	0.58	0.45	0.17	0.17	0.23	0.23
Overall Treatment	63	1.07	0.50	1.16	0.82	0.87	0.87	0.00	0.00

		N. Slimy Salamander *P. glutinosus*		**N. Spring Salamander** *G. p. porphyriticus*		**Red eft** *N. v. viridescens*		**Redback salamander** *P. cinereus*	
	n	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Pre-removal Reference	27	0.93	0.93	0.93	0.93	0.93	0.93	1.85	1.22
Pre-removal Treatment	27	0.00	0.00	0.00	0.00	0.00	0.00	0.93	0.93
Post-removal Reference	36	0.00	0.00	0.00	0.00	0.69	0.69	0.69	0.69
Post-removal Treatment	36	0.00	0.00	0.00	0.00	0.00	0.00	1.39	0.94
Overall Reference	63	0.12	0.12	0.12	0.12	0.2	0.13	0.32	0.23
Overall Treatment	63	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.13

Appendix XXIXa. Continued

		Wood frog *L. sylvaticus*		Eastern Gartersnake *T. s. sirtalis*		N. Ringneck Snake *D. p. edwardsii*		Total Relative Abundance	
	n	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
Pre-removal Reference	27	1.85	1.85	0.00	0.00	0.00	0.00	19.44	6.80
Pre-removal Treatment	27	0.00	0.00	0.00	0.00	0.00	0.00	16.67	11.02
Post-removal Reference	36	0.00	0.00	0.00	0.00	0.69	0.69	5.56	2.78
Post-removal Treatment	36	0.00	0.00	0.69	0.69	0.00	0.00	4.86	1.24
Overall Reference	63	0.23	0.23	0.00	0.00	0.09	0.09	12.50	6.94
Overall Treatment	63	0.00	0.00	0.09	0.09	0.00	0.00	10.77	5.91

Appendix XXXa: Mean ($\bar{X}$) and SE of herpetofauna relative abundance (No. of captures/100 trap nights) at reference and treatment cover board arrays at Fort Necessity National Battlefield, Pennsylvania, USA. We trapped using the cover board arrays from 2004 through 2010 during the summer season from May-August. *Post-removal surveys took place following the implementation of management procedures designed to remove Morrow's honeysuckle.

		Four-toed Salamander *H. scutatum*		**Northern Slimy Salamander** *P. glutinosus*		**Red eft** *N. v. viridescens*		**Redback salamander** *P. cinereus*	
	n	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Pre-removal Reference	27	0.00	0.00	0.00	0.00	3.70	2.45	27.78	7.35
Pre-removal Treatment	27	0.00	0.00	0.00	0.00	0.00	0.00	35.19	6.48
Post-removal Reference	36	6.94	6.94	1.39	1.39	4.17	2.18	8.33	2.51
Post-removal Treatment	36	0.00	0.00	0.00	0.00	2.78	1.87	2.78	1.87
Overall Reference	63	3.47	3.47	0.70	0.70	3.94	1.36	18.06	5.87
Overall Treatment	63	0.00	0.00	0.00	0.00	1.39	0.88	18.98	8.12

		Eastern Gartersnake *T. s. sirtalis*		**Eastern Ratsnake** *S. alleghaniensis*		**Northern Black Racer** *C. c. constrictor*		**Northern Ringneck Snake** *D. p. edwardsii*	
	n	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Pre-removal Reference	27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pre-removal Treatment	27	1.85	1.85	1.85	1.85	0.00	0.00	0.00	0.00
Post-removal Reference	36	1.39	1.39	1.39	1.39	1.39	1.39	2.78	2.78
Post-removal Treatment	36	2.78	1.87	0.00	0.00	0.00	0.00	0.00	0.00
Overall Reference	63	0.69	0.69	0.69	0.69	0.69	0.69	1.39	1.39
Overall Treatment	63	2.31	1.06	0.93	0.93	0.00	0.00	0.00	0.00

Appendix XXXa. Continued

		Smooth Greensnake *O. vernalis*		**Total Relative Abundance**	
	n	$\overline{X}$	SE	$\overline{X}$	SE
Pre-removal Reference	27	1.85	1.85	33.33	6.21
Pre-removal Treatment	27	0.00	0.00	38.89	6.80
Post-removal Reference	36	0.00	0.00	27.78	9.48
Post-removal Treatment	36	12.5	11.00	20.83	11.26
Overall Reference	63	0.93	0.93	30.56	2.78
Overall Treatment	63	6.25	5.46	29.86	9.03

Appendix Ib. Fruit species located during three study arounds at Fort Necessity National Battlefield, Pennsylvania, USA, from October 2009 – August 2010. Study species were chosen using a random number generator. When species could not be located in sufficient quantities, another species was chosen at random to replace it.

Round 1	**Round 2**
Oct. 20 – Nov. 14, 2009	July 12 – Aug. 6, 2010
Common greenbrier *Smilax rotundifolia*	Staghorn sumac *Rhus typhina*
Winter grape *Vitis cinerea*	Northern dewberry *Rubus flagellaris*
Southern arrowwood *Viburnum dentatum*	Common Serviceberry* *Amelanchier arborea*
Black cherry *Prunus serotina*	
Flowering dogwood *Cornus florida*	
Waxy-fruit Hawthorne* *Crataegus pruinosa*	
Staghorn sumac *Rhus typhina*	
Poison Ivy* *Toxicodendron radicans*	
Black gum* *Nyssa sylvatica*	

*Species present without quantities sufficient for the study

Appendix Ic. Values for the environmental variables measured at each foraging station at Fort Necessity National Battlefield, Pennsylvania, USA, in October – November 2009 (fall) and July – August 2010 (summer). All values were averaged over 20 stations within each cover type (total N=60). Averaged values (Mean), standard errors (SE), minimum values (Min) and maximum values (Max) are reported.

	Edge 100 m^2 - Summer				**Edge 400 m^2 - Summer**			
Variable	$\bar{X}$	S.E.	Min	Max	$\bar{X}$	S.E.	Min	Max
Brush cover (%)	4.25	0.63	1.20	15.00	3.97	0.81	0.00	18.75
Canopy cover (%)	79.93	3.22	46.25	97.50	79.11	3.29	46.25	98.35
Fern cover (%)	7.84	2.76	0.00	45.00	9.15	2.73	0.00	38.75
First sight (cm)	5.99	0.48	5.00	12.50	6.90	1.03	3.00	21.25
Forb cover (%)	17.41	4.00	2.00	63.75	15.70	3.28	5.00	52.50
Grass cover (%)	19.32	3.53	10.00	68.75	18.73	3.27	10.00	62.50
Green cover (%)	47.04	6.42	6.25	97.50	43.38	5.71	8.00	91.25
Leaf cover (%)	28.93	4.15	3.51	67.50	29.11	4.11	4.00	67.50
Log cover (%)	2.27	0.37	1.00	6.25	1.88	0.28	1.00	0.40
Moss cover (%)	3.25	0.33	0.00	7.75	3.10	0.39	0.00	9.50
Road cover (%)	2.38	0.89	1.00	15.00	9.52	1.69	1.00	25.00
Rock cover (%)	1.36	0.18	0.75	5.00	1.40	0.19	0.75	4.25
Shrub cover (%)	9.29	2.48	0.11	46.25	8.01	1.88	5.00	33.75
Shrub distance (m)	4.14	1.61	0.35	25.00	4.14	1.61	0.35	25.00
Shrub stem number	2.05	0.49	1.00	10.00	2.05	0.49	1.00	10.00
Shrub volume (m^3)	0.43	0.16	0.02	2.89	0.43	0.16	0.01	2.89
Slope (°)	8.99	1.07	3.25	19.25	8.99	1.07	3.25	19.25
Soil moisture	1.31	0.24	0.13	3.50	1.31	0.24	0.13	3.50
Tallest sight (cm)	22.12	3.00	5.00	47.50	25.69	3.83	5.00	57.50
Tree cover (%)	3.48	0.20	1.51	5.05	3.37	0.26	1.50	6.25
Tree crown (m)	7.96	0.64	2.30	13.20	7.96	0.64	2.30	13.20
Tree DBH (cm)	41.96	6.13	11.15	133.76	41.96	6.13	11.15	133.76
Tree distance (m)	0.94	0.21	0.05	3.30	0.94	0.21	0.05	3.30
Tree height (m)	11.59	0.51	7.00	14.75	11.59	0.51	7.00	14.75
Visitation rate (%)	72.50	6.27	14.29	100.00	72.50	6.27	14.29	100.00
Water (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix Ic. Continued

Variable	Edge 100 m^2 - Fall				Edge 400 m^2 - Fall			
	$\bar{X}$	S.E.	Min	Max	$\bar{X}$	S.E.	Min	Max
Brush cover (%)	10.46	1.00	5.00	22.00	10.21	1.00	3.00	19.30
Canopy cover (%)	31.73	2.91	0.00	55.00	32.04	2.97	1.00	55.00
Fern cover (%)	4.35	2.17	0.00	42.50	6.44	2.55	0.00	43.75
First sight (cm)	6.31	0.45	5.00	11.25	6.56	0.50	3.75	12.56
Forb cover (%)	8.55	4.36	0.00	63.75	9.21	4.28	0.00	63.75
Grass cover (%)	10.57	5.66	0.00	85.00	10.29	5.37	0.00	83.75
Green cover (%)	19.79	3.72	0.00	52.50	21.54	3.58	0.00	55.00
Leaf cover (%)	72.00	7.04	3.75	100.00	70.31	6.83	3.75	100.00
Log cover (%)	4.43	1.56	0.00	28.75	4.51	1.39	1.00	22.50
Moss cover (%)	4.08	0.89	0.00	17.50	5.28	0.91	0.00	12.50
Road cover (%)	2.39	0.89	0.00	15.00	7.78	1.54	0.75	20.00
Rock cover (%)	1.28	0.18	0.75	4.50	1.26	0.18	0.75	4.50
Shrub cover (%)	11.77	2.26	0.00	33.75	15.20	2.99	0.00	48.75
Shrub distance (m)	3.71	1.64	0.30	25.00	3.71	1.64	0.30	25.00
Shrub stem number	3.50	0.72	1.00	10.00	3.50	0.72	1.00	10.00
Shrub volume (m^3)	0.53	0.16	0.01	2.83	0.53	0.16	0.01	2.83
Slope (°)	8.99	1.07	3.25	19.25	8.99	1.07	3.25	19.25
Soil moisture	1.36	0.27	0.13	4.50	1.36	0.27	0.13	4.50
Tallest sight (cm)	16.81	4.59	5.00	76.25	19.31	6.24	3.75	125.00
Tree cover (%)	3.72	1.17	0.00	25.00	3.94	1.12	0.00	23.75
Tree crown (m)	6.67	0.58	1.50	12.20	6.67	0.58	1.50	12.20
Tree DBH (cm)	32.88	3.13	7.93	54.14	32.88	3.13	7.93	54.14
Tree distance (m)	1.49	0.61	0.10	12.50	1.49	0.61	0.10	12.50
Tree height (m)	10.72	0.75	6.00	19.00	10.72	0.75	6.00	19.00
Visitation rate (%)	41.79	6.58	0.00	100.00	41.79	6.58	0.00	100.00
Water (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix Ic. Continued

Variable	Field 100 m^2 - Summer				Field 400 m^2 - Summer			
	$\bar{X}$	S.E.	Min	Max	$\bar{X}$	S.E.	Min	Max
Brush cover (%)	1.20	0.34	0.00	5.00	2.40	0.63	0.00	10.00
Canopy cover (%)	0.25	0.25	0.00	5.00	1.06	0.68	0.00	12.50
Fern cover (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
First sight (cm)	23.88	5.14	6.25	50.00	31.19	7.13	6.25	73.75
Forb cover (%)	70.81	6.35	11.25	100.00	69.77	6.26	11.25	100.00
Grass cover (%)	61.08	7.52	30.00	100.00	60.50	7.37	31.75	100.00
Green cover (%)	94.19	0.91	85.00	100.00	93.25	1.33	75.00	100.00
Leaf cover (%)	3.88	1.35	0.00	27.50	3.56	1.26	0.00	25.00
Log cover (%)	0.04	0.04	0.00	0.75	0.00	0.00	0.00	0.00
Moss cover (%)	0.40	0.28	0.00	5.00	0.40	0.28	0.00	5.00
Road cover (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rock cover (%)	3.00	0.62	0.00	10.00	3.24	0.60	0.00	9.50
Shrub cover (%)	17.34	6.09	0.00	90.00	20.30	5.61	0.00	78.75
Shrub distance (m)	3.97	0.84	0.20	13.40	3.97	0.84	0.20	13.40
Shrub stem number	8.05	1.10	2.00	20.00	8.05	1.10	2.00	20.00
Shrub volume (m^3)	3.27	1.22	0.16	25.07	3.27	1.22	0.16	25.07
Slope (°)	2.56	0.58	0.25	8.85	2.56	0.57	0.25	8.85
Soil moisture	1.90	0.21	0.25	3.88	1.90	0.21	0.25	3.88
Tallest sight (cm)	76.25	2.90	18.75	121.25	95.31	4.43	40.00	145.00
Tree cover (%)	0.15	0.08	0.00	1.25	0.54	0.21	0.00	2.75
Tree crown (m)	7.16	0.73	2.50	14.00	7.16	0.73	2.50	14.00
Tree DBH (cm)	23.57	2.95	4.78	57.32	23.57	2.95	4.78	57.32
Tree distance (m)	15.19	2.79	0.40	45.00	15.19	2.79	0.40	45.00
Tree height (m)	8.07	0.73	2.50	13.60	8.07	0.73	2.50	13.60
Visitation rate (%)	37.60	6.06	0.00	92.86	37.60	6.06	0.00	92.86
Water (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix Ic. Continued

Variable	Field 100 m^2 - Fall				Field 400 m^2 - Fall			
	$\bar{X}$	S.E.	Min	Max	$\bar{X}$	S.E.	Min	Max
Brush cover (%)	55.19	5.16	0.00	92.50	56.88	5.26	0.00	88.75
Canopy cover (%)	1.00	1.00	0.00	20.00	2.35	1.62	0.00	25.00
Fern cover (%)	0.11	0.08	0.00	1.50	0.20	0.16	0.00	3.25
First sight (cm)	7.69	5.74	5.00	15.00	10.74	9.11	5.00	41.25
Forb cover (%)	46.02	5.61	3.00	80.00	44.85	5.26	3.75	80.00
Grass cover (%)	47.63	5.71	5.00	95.00	47.54	5.58	7.00	90.00
Green cover (%)	28.61	3.22	0.00	47.50	27.51	3.22	0.00	51.25
Leaf cover (%)	3.96	1.44	0.00	17.25	5.00	1.32	0.00	15.00
Log cover (%)	0.00	0.00	0.00	0.00	0.09	0.09	0.00	1.75
Moss cover (%)	0.54	0.26	0.00	4.00	0.88	0.34	0.00	4.50
Road cover (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rock cover (%)	5.39	0.87	0.00	11.00	5.95	0.87	0.00	11.50
Shrub cover (%)	21.76	6.78	0.00	95.00	28.98	7.00	5.00	90.00
Shrub distance (m)	3.05	0.57	0.20	11.00	3.05	0.57	0.20	11.00
Shrub stem number	10.45	2.15	1.00	43.00	10.45	2.15	1.00	43.00
Shrub volume (m^3)	5.99	2.22	0.02	35.78	5.99	2.22	0.02	35.78
Slope (°)	2.53	0.58	0.00	8.75	2.53	0.58	0.00	8.75
Soil moisture	1.90	0.21	0.25	3.88	1.90	0.21	0.25	3.88
Tallest sight (cm)	42.81	0.68	7.50	93.75	70.00	1.92	15.00	155.00
Tree cover (%)	0.23	0.16	0.00	3.00	0.65	0.30	0.00	5.50
Tree crown (m)	7.43	0.70	2.50	14.00	7.43	0.70	2.50	14.00
Tree DBH (cm)	32.22	4.15	4.78	73.25	32.22	4.15	4.78	73.25
Tree distance (m)	15.19	2.83	0.40	45.00	15.19	2.83	0.40	45.00
Tree height (m)	10.14	0.95	5.00	21.00	10.14	0.95	5.00	21.00
Visitation rate (%)	61.43	5.94	14.29	100.00	61.43	5.94	14.29	100.00
Water (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix Ic. Continued

Variable	Forest 100 m^2 - Summer $\bar{X}$	S.E.	Min	Max	Forest 400 m^2 - Summer $\bar{X}$	S.E.	Min	Max
Brush cover (%)	5.04	0.73	3.00	17.50	4.78	0.57	3.00	13.75
Canopy cover (%)	81.31	2.96	37.50	93.75	81.25	1.80	60.00	92.50
Fern cover (%)	3.57	0.79	0.00	11.25	3.59	0.70	0.00	9.50
First sight (cm)	5.00	0.00	5.00	5.00	5.00	0.00	5.00	5.00
Forb cover (%)	2.67	0.57	0.00	11.25	2.61	0.53	0.00	10.00
Grass cover (%)	11.45	0.34	10.00	13.75	11.63	0.38	10.00	14.75
Green cover (%)	37.87	5.36	12.50	85.00	39.06	5.27	12.50	85.00
Leaf cover (%)	51.00	3.17	30.00	80.00	50.88	3.13	30.00	80.00
Log cover (%)	4.52	0.76	0.00	10.00	4.28	0.68	0.00	9.50
Moss cover (%)	5.58	1.94	0.00	40.00	5.68	1.93	0.00	40.00
Road cover (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rock cover (%)	3.86	0.72	0.00	13.25	3.67	0.53	0.00	10.25
Shrub cover (%)	26.21	4.50	2.25	80.00	26.56	4.43	3.00	80.00
Shrub distance (m)	0.89	0.13	0.21	2.90	0.89	0.13	0.21	2.90
Shrub stem number	1.95	0.33	1.00	7.00	1.95	0.33	1.00	7.00
Shrub volume (m^3)	0.06	0.02	0.01	0.40	0.06	0.02	0.01	0.40
Slope (°)	9.33	1.27	0.75	19.00	9.33	1.27	0.75	19.00
Soil moisture	0.29	0.11	0.00	2.00	0.29	0.11	0.00	2.00
Tallest sight (cm)	13.43	2.45	3.75	50.00	30.62	4.21	7.50	77.50
Tree cover (%)	2.49	0.29	0.00	5.75	2.81	0.19	0.10	5.00
Tree crown (m)	10.29	1.10	5.60	28.00	10.29	1.10	5.60	28.00
Tree DBH (cm)	39.44	2.79	20.70	57.32	39.44	2.79	20.70	57.32
Tree distance (m)	1.71	0.35	0.09	5.80	1.71	0.35	0.09	5.80
Tree height (m)	14.53	0.58	10.07	21.00	14.53	0.58	10.07	21.00
Visitation rate (%)	66.07	5.00	28.57	92.86	66.07	5.00	28.57	92.86
Water (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix Ic. Continued

Variable	Forest 100 m^2 - Fall $\bar{X}$	S.E.	Min	Max	Forest 400 m^2 - Fall $\bar{X}$	S.E.	Min	Max
Brush cover (%)	53.71	8.56	3.50	95.00	53.90	8.58	3.50	95.00
Canopy cover (%)	31.87	4.53	6.25	85.00	29.43	2.83	16.25	63.75
Fern cover (%)	1.79	0.46	0.00	6.50	2.10	0.49	7.00	0.00
First sight (cm)	6.25	0.36	5.00	10.00	6.25	0.36	5.00	10.00
Forb cover (%)	0.67	0.24	0.00	3.00	0.67	0.24	3.00	0.00
Grass cover (%)	1.05	0.31	0.00	4.00	1.04	0.30	0.00	3.25
Green cover (%)	21.41	4.70	3.00	87.50	22.61	5.02	3.00	90.00
Leaf cover (%)	84.86	3.09	36.25	100.00	84.36	3.13	36.25	95.00
Log cover (%)	4.62	0.71	0.00	11.25	4.89	0.78	0.00	11.25
Moss cover (%)	9.14	4.45	1.50	87.50	9.73	4.59	1.50	90.00
Road cover (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rock cover (%)	1.98	0.25	0.00	5.00	1.99	0.24	1.00	5.00
Shrub cover (%)	16.54	3.56	0.00	65.00	16.62	3.63	0.00	58.75
Shrub distance (m)	2.48	1.23	0.10	25.00	2.48	1.23	0.10	25.00
Shrub stem number	2.00	0.42	1.00	9.00	2.00	0.42	1.00	9.00
Shrub volume (m^3)	0.56	0.32	0.01	6.08	0.56	0.32	0.01	6.08
Slope (°)	9.33	1.27	0.75	19.00	9.33	1.27	0.75	19.00
Soil moisture	23.25	0.06	0.00	1.13	23.25	0.06	0.00	1.13
Tallest sight (cm)	17.43	3.39	5.00	55.00	20.00	2.87	5.00	43.75
Tree cover (%)	4.66	0.16	3.50	5.75	4.61	0.15	3.50	5.75
Tree crown (m)	13.38	2.20	3.20	36.30	13.38	2.20	3.20	36.30
Tree DBH (cm)	57.10	9.76	14.01	187.90	57.10	9.76	14.01	187.90
Tree distance (m)	2.61	0.58	0.30	9.10	2.61	0.58	0.30	9.10
Tree height (m)	18.75	1.15	7.00	28.00	18.75	1.15	7.00	28.00
Visitation rate (%)	59.65	5.91	0.00	85.71	59.65	5.91	0.00	85.71
Water (%)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00