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Grassland bird abundance and habitat quality, and Sedge Wren (Cistothorus platensis) ecology on Fort Drum, New York

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By

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May 15, 2013

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Chapter 1:

Abstract:

Grassland bird abundance and habitat quality, and Sedge Wren (*Cistothorus platensis*) ecology on Fort Drum, New York

This research examines the breeding habitat preferences of grassland birds at Fort Drum, Jefferson County, New York during the 2011 and 2012 breeding seasons. In the past, Fort Drum and surrounding areas in Jefferson County have supported large numbers of obligate grassland breeding birds (OGBB). However, results from this study, when combined with data from past studies of grassland birds at Fort Drum, suggest that habitat specialists such as the sedge wren (*Cistothorus platensis)* and Henslow's sparrow (*Ammodramus henslowii)* have been continually declining. Reasons for the decline are most likely related to a decrease in agriculture resulting in habitat loss due to succession and a shift in agricultural practices, both in Jefferson County, New York and throughout the Northeast. Habitat models suggest that most OGBBs at Fort Drum, including savannah sparrows and bobolinks, prefer increased graminoid cover and shorter, less dense vegetation. Differences in the models between years suggest that the predictive power of modeling is limited and that models should be used only as management guidelines, with a concentrated effort made to manage for large, contiguous mosaic grassland habitat.

Grassland bird abundance and habitat quality, and Sedge Wren (*Cistothorus platensis*) ecology on Fort Drum, New York

General Introduction:

Background:

Grasslands are one of the most altered ecosystems in North America. An estimated 80% of all grassland habitat in the United States has been lost since the 1800s (Knopf 1994, Noss et al. 1995). In the Midwest, tallgrass prairies have declined by 96% and shortgrass prairies have declined by nearly 66%. Similarly, native grasslands in the Northeast have declined ≥90%, as has occurred in the Hempstead Plains on Long Island (Samson and Knopf 1994, Noss et al. 1995, Askins 2001).

Grasslands are easily adapted to human needs, particularly farming and land development (Askins 1999, Henwood 2010). In the Northeast, defined here as New York and the six New England states, the anthropogenic activity that has altered and created the most grassland is agriculture. However, agriculture has been declining since its peak in the mid-nineteenth century, and succession is rapidly transforming the landscape; much of what was once open land is now forested (Foster et al. 2002). This has resulted in the replacement of grassland breeding birds by shrubland and forest breeding birds. Accentuating the effects of the decline in agriculture is a shift in agricultural practices to more frequent and earlier haycropping (Bollinger et al. 1990). This makes an already limited and potentially poor quality habitat even less for wildlife. One of the consequences of declining grassland habitat is the well-documented decline in grassland breeding birds. For example, in New York bobolink (*Dolichonyx oryzivorus)* nest success decreased by >90% when haycropping occurred during the breeding season (Bollinger et al. 1990). Furthermore, agricultural fields tend to be a monoculture of a single crop, whereas grassland bird species diversity is more likely to increase in heterogeneous landscapes (Madden et al. 2000, Fuhlendorf and Engle 2001, Winter et al. 2005, Rahmig et al. 2009, Jacobs et al. 2012).

As land development, reforestation, and afforestation continue to affect available open space in the Northeast, habitat fragmentation has increased – resulting in a greater perimeter-to-area ratio of remnant habitat patches. Many grassland birds are area-sensitive, and their species diversity, abundance, and breeding success tend to increase with patch size (Herkert 1994, Vickery et al. 1994, Balent and Norment 2003, Bollinger and Gavin 2004). For example, in Maine, grassland birds such as grasshopper sparrows (*Ammodramus savannarum*) do not reach 50% occurrence until habitat area exceeds 100 ha, while upland sandpipers (*Bartramia longicauda*) do not reach 50% occurrence until 200 ha (Vickery et al. 1994). Although grassland birds sometimes may nest successfully in small habitat patches when the land is severely fragmented, their abundance and species diversity will often be lower in this situation (Weidman and Litvaitis 2011).

Given the state of North American grasslands, it is not surprising that obligate grassland breeding birds (*sensu* Vickery et al. 1999) (OGBB) in North America have had “steeper, more consistent, and more geographically widespread declines than any other behavioral or ecological guild (Knopf 1994).” The Breeding Bird Survey (BBS) reports that among 28 grassland bird species evaluated in North America, 15 declined significantly between 1966 and 2011 (Sauer et al. 2012). This trend is similar in the Northeast, where grassland breeding birds are the ecological guild with the most species listed as endangered, threatened, or of special concern (Vickery 1992). In New York, from 1966 – 2011, nine of 11 species declined significantly; the remaining two had statistically non-significant declines (Sauer et al. 2012). Also, in the past 20 y, grassland birds were the only breeding bird habitat group in New York to show a consistent, significant decline (McGowan and Corwin 2008).

Grassland history in New York State:

Natural grasslands occurred historically in New York, but they were not widespread. One of the largest historical grasslands was the Hempstead Plains on Long Island, of which only two small patches persist, representing <1% of the original 24,000 ha (Askins 2001). Typical grasslands were sparse clearings, often created by beaver (*Castor canadensis*), floods, or occasional fires (Askins 1999, Hunter et al. 2001). Native American agricultural practices probably increased the amount of open land, but on a small scale (Foster 1995). Since European colonization, habitat in the Northeast has undergone substantial changes (Foster et al. 2002, Norment 2002). The Northeast was predominantly forested until colonists began to clear land for agriculture, homesteading, and logging (Askins 1999). As agriculture has declined, however, grasslands also have declined. From 1936 to 1991, there was a 60% decline in grassland habitat in the Northeast (U.S. Department of Agriculture 1936-1991, Vickery et al. 1994).

As the composition of the habitat fluctuated, so did species abundance and diversity (Foster et al 2002). Although some grassland birds such as grasshopper sparrows, savannah sparrows (*Passerculus sandwichensis*), and bobolinks undoubtedly were present prior to European settlement, OGBB diversity and abundance must have increased as grassland habitat area increased, until the trend was reversed during the twentieth century (Askins 1997, Foster et al. 2002). Recent declines in OGBBs in the Northeast are not merely a return to pre-settlement abundances because declines have resulted in local extirpation and extinction (Hunter et al. 2001). The Henslow's sparrow (*Ammodramus henslowii*), for example, was once common along northeastern coasts, but has since declined (Stone 1937, Herkert et al. 2002). In New Jersey, where a possible subspecies commonly bred, only one singing male could be found in 1994 (Walsh et al. 1999, Herkert et al. 2002) Another example, the heath hen (*Tympanuchus cupido cupido*), was once abundant in the Northeast, including in the Hempstead Plains of New York, but is now extinct (Askins 1997). Considering these extinctions and the regional, continental, and global decline of grassland birds, protecting the remaining high-quality grasslands in the Northeast is an important management goal.

Management of grassland birds and their habitats:

Grasslands are disturbance-mediated ecosystems. Historically, fires, grazing, and flooding helped maintain grasslands (Askins 1999, Hunter et al. 2001, Foster et al. 2002). Without regular disturbance, grasslands and their disturbance-dependent species, such as grassland birds, will decline. Consequently, regular disturbance through management is necessary in the Northeast to maintain grassland bird habitat because of changes in land-use practices and an absence of an effective natural disturbance regime. Agriculture once provided a means of incidental management because the area farmed was greater, and haycropping was less frequent and occurred later in the season. Now, however, haycropping begins before the breeding season has ended. This greatly reduces nest success, with the effects of additive mortality reaching 94% in New York (Bollinger et al. 1990). Fires also were part of the historical disturbance regime, but fires have been suppressed historically since people are wary of prescribed fires (Askins 1999, Hunter et al. 2001). Prescribed fires in the Midwest benefit most grassland birds, and patch burning, where only certain portions are burned, increases habitat and species diversity (Fuhlendorf and Engle 2001, Grant et al. 2010). Unfortunately, dormant season burns in the Northeast, where cool season grasses dominate, often increase shrub and goldenrod abundance while decreasing grass abundance (Mitchell 2000). Grazing is another management option but can result in destroyed nests; however, after 15 July, grazing does not substantially reduce nest success (Bollinger and Gavin 1992, Norment et al. 1999, Perlut and Strong 2011).

One principal focus of grassland management is related to area sensitivity of grassland birds: as the probability of occurrence and abundance of most OGBB increases with area (Robbins et al. 1989, Ribic et al. 2009). For example, increasing area has a four times greater effect than habitat heterogeneity on bird species richness and abundance (Bollinger 1995, Ribic et al. 2009, Cerezo et al. 2011).

Current management practices designed to benefit grassland birds seek to increase habitat area, decrease fragmentation and edge effects, promote mosaic landscapes, promote grasses and other appropriate herbaceous species while controlling woody shrubs, and avoid disturbances during the breeding season (Vickery et al. 1994, Balent and Norment 2003, Ribic et al. 2009, Jacobs 2012 et al. 2011). Understanding the complex relationships among these different factors and how they interact to affect grassland birds, is difficult, although multivariate statistical modeling may help elucidate important ecological relationships. Multivariate statistical modeling uses a large number of habitat variables, ranging from the local to landscape scale, to reveal important bird-habitat relationships. In turn, these models may then be used to inform management decisions (Coppedge et al. 2008, Cerezo et al. 2011, Jacobs et al. 2012), although Rotenberry and Wiens (2009) have questioned the utility of bird-habitat models, suggesting that they are more effective when understood as guidelines that explain only a portion of the habitat system.

In the Northeast, one of the most important regions of remaining grassland habitat is the St. Lawrence Plain ecozone in northern New York (Shriver et al. 2005). The St. Lawrence Plain extends from northern New York, into Canada and includes regions of Vermont near the Lake Champlain Valley. The Partners in Flight (PIF) Bird Conservation Plan (BCP) states that the St. Lawrence Plain is "the largest and most important area of grassland in the Northeast (Rosenberg 2000)." Historically, the New York State threatened Henslow's sparrow and sedge wren (*Cistothorus platensis*) have been more abundant in the St. Lawrence Plain, especially in Jefferson County, than in any other area in the Northeast (Vickery 1992, Shriver et al. 2005, McGowan and Corwin 2008).

The Henslow's sparrow, sedge wren, and other listed grassland birds historically have been found on Fort Drum, a United States Army installation in Jefferson and Lewis Counties. It contains 5,577 ha of grassland, making it one of the largest grasslands in New York (Dobony and Rainbolt 2008). Fort Drum is listed by the National Audubon Society as an Important Bird Area and has been the site of multiple research projects focused on grassland birds (Wells 1998, Krebs 2002). The breeding population of Henslow's sparrows on the Fort Drum base has been one of the larger populations in the state; however, the population continues to decline not only on the base, but in nearby areas (Krebs 2002, Kirk pers. comm.). At Fort Drum, the Environmental Division is responsible for managing wildlife on the 43,442 ha base. Since Fort Drum must maintain open grasslands for military training and is one of the largest grasslands in the Northeast (>5,500 ha), it is both logical and necessary to manage the land cooperatively for military training and to promote OGBBs.

In light of the condition of New York State grasslands and their constituent breeding bird species, the two chapters of my thesis focus on the ecology of grassland breeding birds on Fort Drum. The first chapter reports on the abundance of grassland breeding birds on the base, describes grassland bird habitat models, and provides management recommendations to help improve the quality of the grassland habitat at Fort Drum. The second chapter focuses on one grassland bird species, the sedge wren (*Cistothorus platensis*), commenting on the habitat preferences of this habitat specialist, reasons for its fluctuating abundance but consistent presence, and provides suggestions for management and future research.

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Grassland bird abundance and habitat quality on Fort Drum

Introduction:

North American grasslands have experienced more habitat alteration than any other ecosystem in North America with declines for some grasslands reaching 96% (Knopf 1994, Samson and Knopf 1994, Noss et al. 1995). In the Northeast (New York and the six New England states) native grasslands have been nearly entirely absorbed by human land use practices – mainly agriculture and land development. As a result obligate grassland bird species have drastically declined. These declines are due to the effects of a decrease in agriculture, increased frequency of haycropping, land development, and succession, all of which negatively affect the amount of grassland habitat available and the quality of that habitat. According to the Breeding Bird Survey (BBS), nine of 11 grassland bird species in New York have declined significantly since the survey's inception in 1966 to 2011 (Sauer et al. 2012).

In New York, the St. Lawrence Plain ecozone is one of the last remaining strongholds for grassland bird habitat in New York and the Northeast (Shriver et al. 2005). Many species that are rare or absent in the rest of the Northeast are found consistently in the St. Lawrence Plain. Jefferson County, New York is in the St. Lawrence Plain ecozone and has had greater abundance of sedge wrens (*Cistothorus platensis*) and Henslow's sparrows (*Ammodramus henslowii*) than anywhere else in the state, both of which are rare grassland species in the Northeast (McGowan and Corwin 2008). It is also home to Fort Drum military installation which contains one of the largest contiguous grasslands in New York State, at over 5,500 ha. On Fort Drum multiple research projects have focused on grassland bird habitat and management; these generally have found that grassland bird abundance is declining as the grassland habitat gives way to succession and sporadic management practices (Wells 1998, Bollsinger et al. 1999, Krebs 2002, Kirk pers. comm.). The Environmental Division of the Department of Public Works is responsible for wildlife management; however, bureaucratic divisions make implementing good management practices difficult. Fort Drum's primary objective is to train military, but this does not need to conflict with grassland bird habitat management. Cooperative management will allow both military training and grassland bird conservation to occur on one of the last remaining large tracts of grassland in the Northeast.

Objectives:

It is imperative to continue grassland bird conservation efforts at Fort Drum as part of a state-wide management plan for this ecological guild (New York State Department of Environmental Conservation 2012). Thus, in 2011, I began a study on grassland bird populations and habitat relationships at Fort Drum, New York. The primary objectives of my study were to determine grassland bird species richness and abundance on the base and analyze grassland bird habitat, primarily at the local (vegetation) level. I used results from this study to evaluate the status of grassland birds, especially listed species, at Fort Drum, with particular attention given to sedge wren ecology. In addition, I created grassland bird habitat models using a Generalized Linear Model (GLM) approach, to help develop appropriate recommendations for managing grassland birds at Fort Drum and contribute to our understanding of grassland bird ecology and management in the Northeast.

Study site:

The study site was located on Fort Drum Military Installation, Jefferson and Lewis Counties, New York (Figure 1). Fort Drum is a 43,442 ha military training base with 5,577 ha of grassland habitat; the largest contiguous grassland patches occur in training areas 12 and 13 (Dobony and Rainbolt 2008; Figures 1 and 2). There are also patches of grassland throughout the base, including a region of sandier soil and scrub-oak habitat near the Wheeler-Sack Army Airfield, although I did not conduct research in these areas. Instead, I focused my research on training areas 12B, 12C, 12D, 13A, and 13B (hereafter, training areas 12 and 13) and searched for sedge wrens in nearby training areas and with roadside surveys in Jefferson County (Figure 2). Fort Drum is home to four state endangered and seven state threatened bird species (U.S. Army Garrison Fort Drum 2012), including six state-listed OGBB species (Appendix 1). Historically, the base was open farmland. Since farming ceased, succession has played a major role in the conversion of grassland to forest. Military training in grassland areas, especially with tracked vehicles, may have helped to slow succession but has not recently been intense enough to maintain open grassland spaces (J. Bolsinger pers. comm.). Complicating management directives is a difference in departmental primary objectives between Integrated Training Area Management (ITAM), the department responsible for maintaining open land for military training, and the Environmental Division, which is responsible for managing for wildlife.

Grasslands in Training Areas 12 and 13 are dominated by cool season grasses and are often fragmented by patches of shrubland and early successional forests. Common grasses and sedges are timothy (*Phleum pratense*), red top (*Agrostis gigantea*), reed canary grass (*Phalaris arundinacea*), and awlfruit sedge (*Carex stipata*). Forbs, such as goldenrod (*Solidago* spp.), wild strawberry *(Fragaria virginiana*), dwarf cinquefoil (*Potentilla canadensis*), moneywort (*Lysimachia nummularia*), and cow vetch (*Vicia cracca*) occur in most fields. Common shrubs include willow (*Salix* spp.), dogwood (*Cornus* spp.), and spirea (*Spiraea* spp.). There are no major bodies of water near the study site, although Hunter Creek, a small stream, runs east to west through Training Area 13A and 12C (Figure 3). The fields tend to be wet, with deep ruts made by military vehicles filling with water during the early part of the breeding season.

Methods:

Grassland birds:

My first field season began on 24 May and ended on 5 Aug 2011. During the following field season, I conducted research from 15 May to 7 Jun and 2 July to 5 July 2012. Obligate grassland breeding bird abundance was measured at 41, 100 m radius point count locations, with centers separated by >200m to avoid double counting (Norment et al. 1999) (Figure 3). Point count plot locations previously had been established for OGBB research that began in 1995. Of these 127 pre-existing plots, I selected 41 for my study, based on the criterion that plot centers were >250m from the forest edge (Figure 3). Plots that did not meet this criterion but were historically known to have state threatened species (sedge wren or Henslow's sparrow) were also used for the surveys. I used a 3-min wait period before recording data during the 5-min point counts. Both visualizations and vocalizations were used to identify bird species. I collected data on species, sex, and distance. I used a rangefinder to measure each individual's distance from the center of the plot. Point counts began at sunrise and ended by 1000 EDT; they were not conducted when winds were >12 km/h or when raining (Norment et al. 1999). I conducted four rounds of point counts in 2011 and three rounds in 2012. I used the first round of point counts in each year for my statistical analyses because they had the greatest abundance of OGBB; later rounds of point counts were used primarily for detecting rare grassland species such as sedge wrens and Henslow's sparrows. In 2011, the first round was from 7 Jun to 13 Jun, with subsequent rounds from 20 Jun to 22 Jun, 19 Jul to 21 Jul, and 1 Aug to 3 Aug. The following year point count rounds were completed earlier (21 May – 24 May) due to anticipated conflicts with military training in the study area, with later rounds completed from 5 Jun to 6 Jun and 2 Jul to 4 Jul. In late fall 2011, Integrated Training Area Management (ITAM) mowed or partially mowed 13 of the 41 point count plots and their surrounding areas.

Vegetation analysis – point count plots:

Vegetation was surveyed at 24 points within each 100-m radius point count plot along transects running in the four cardinal directions; sample points were separated by 16.7 m. At each point, I used a 1 m^2 sampling frame to estimate percent cover of the following groups: grass stems, forb stems, goldenrod (*Solidago* spp.) stems, standing dead stems, shrub stems, and bare ground. The ranges for each cover class, using a modified Daubenmire cover class system, were 0%, 1-5%, 6-15%, 15-25%, 26-50%, 51-75%, 76-95%, and 96-100% (Daubenmire 1959). Vegetation height and density were measured using a Robel pole (Robel et al. 1970). Litter depth was measured in the center of each 1m^2 sampling quadrat to the nearest tenth of a cm. The number of plant taxa (species or genus) within the sampling frame was also recorded. I used a Geographic Information System (ArcGIS ESRI, ArcMap 10.1), along with 2009 satellite imagery, to measure distance to the nearest forest edge at each point count plot.

Statistical Analyses:

An analysis of variance (ANOVA) was used to compare vegetation and OGBB variables at three levels of habitat manipulation for 2012 data: completely mown (CM), partially mown (PM), and unmown (UM). OGBB abundance, bobolink abundance, savannah sparrow abundance, and all vegetation variables, except distance to nearest forest edge, were compared among the three treatment categories. All data were transformed for normality as needed and had equal error variances (Levene's test, $p > 0.05$). I used the Kruskal-Wallis nonparametric alternative for variables that did not meet normality requirements (OGGB, savannah sparrow, and bobolink abundances, and forb cover). I compared vegetation variables of unmown point count plots for 2011 and 2012 (n=28) using a paired t-test and a Wilcoxon rank-sum test, for non-parametric variables, to determine if there was any significant differences between years.

I used Generalized Linear Models (GLM) to predict OGBB abundance based on habitat variables. Individual OGBB species abundance and total OGBB abundance at each point count plot were the response variables; predictor variables were Robel pole score, percent cover (grass, forb, shrub, standing dead, goldenrod, and bare ground), litter depth, plant taxa abundance, and distance to nearest forest edge for each point count plot (Winter et al. 2005, Peggie et al. 2011). If necessary, predictor variables were transformed to obtain normal distributions. Prior to modeling, I standardized predictor variables (Z-score) to remove unit effects (Shaw 2003). If two predictor variables were highly correlated (>0.80) one was eliminated (Shaw 2003). The response variables were fitted to a Poisson distribution because it is best suited for rare events with zero-inflated, skewed distributions (Hoffman 2004). Backwards elimination was used to remove non-significant ($p > 0.05$) predictor variables from the models (Gjerdrum et al. 2005). Akaike Information Criterion with a correction for small sample size (AIC_c) was used to determine the most parsimonious models by selecting models with a $\Delta AIC_c < 2.0$ (Burnham and Anderson 1998). AIC considers both the goodness of fit and the complexity of the model, allowing multiple models to be compared simultaneously (Johnson and Omland 2004, Cerezo et al. 2011). Best models were those with the lowest AIC_c

values because AIC_c reflects the amount of information lost (Burnham and Anderson 1998). I did not use the program DISTANCE (version 6.0) to adjust for detectability, since model-based predicted bird abundances and distance based density-estimates have been shown to have similar results (Jacobs et al 2012). Also, for most species, the number of detections did not meet sample size requirements or the assumption that detections decline with distance (Buckland et al. 2001).

Results:

In 2011 and 2012, I observed five OGBB species during point counts in the study area: savannah sparrows, bobolinks, sedge wrens, Henslow's sparrows, and Northern harriers (*Circus cyaneus*). I did not observe Eastern meadowlarks (*Sturnella magna*) or grasshopper sparrows (*Ammodramus savannarum*). Bobolinks (*Dolichonyx oryzivorus*) were the most common species (55% of all OGBB observations), followed by savannah sparrows (*Passerculus sandwichensis*) (41%) and the three less abundant species: sedge wrens (2%), Henslow's sparrows (1%), and Northern harriers (1%) (Figures 4 and 5). Sedge wrens were found in low areas with mesic soils, few shrubs, and tall graminoid vegetation. Henslow's sparrow territories were often near shrub patches, and males often sang from the tops of goldenrod. Both bobolinks and savannah sparrows were found in places where graminoid vegetation was dominant.

There were several notable differences in bird and vegetation data collected in 2011 and 2012. In 2012, the mean abundance/plot of OGBB decreased significantly by an average of 1.0 from 2011 (Wilcoxon rank-sum test, $W={}^{-}2.996$, $p=0.003$), while shrubland breeding birds (SBB) increased by 1.05 (Table 1). There also were several important habitat and weather differences between the 2011 and 2012 seasons. Fields were wetter in 2011, when precipitation between May and July totaled 31.14 cm, than in 2012, when May – July precipitation totaled 20.80 cm (National Oceanic and Atmospheric Administration 2013). Also, in 2012 13 point count plots were at least partially mowed before 10 May. This resulted in overall shorter vegetation in 2012, although for unmown plots sampled in 2011 and 2012, none of the vegetation variables differed significantly (Table 2). In 2012, four vegetation variables differed significantly among the three mowing treatments (mown, partially mown, and unmown): graminoid cover, liter depth, standing dead cover, and Robel pole score (Table 3, Appendices 2 and 3). Graminoid cover, litter depth, and standing dead cover each differed between two categories – completely mown (CM) and unmown (UM) (Table 3, Appendix 3). Robel pole score had significant differences between both CM and UM and CM and partially mown (PM) (Table 3, Appendix 2). All of the variables had greater values on the UM plots, except for graminoid cover, which had significantly greater cover on CM plots (Table 3, Appendix 3). Although there were significant differences in several vegetation variables among the mowing treatments, there was no statistically significant difference in the abundance of OGBB/plot among the three mowing treatments ($p = 0.069$; Table 3). There was a tendency toward higher numbers of OGBB in partially mown and completely mown plots (Figure 6).

I observed nine SBB species in the study area. The most abundant SBB species for 2011-2012 was the common yellowthroat (*Geothlypis trichas*), which comprised 52% of all SBB observations during the study (Figures 4 and 5). They were most commonly observed singing from within shrub patches of varying size. Other common SBB species included yellow warblers (*Setophaga petechia*), song sparrows (*Melospiza melodia*), and both alder and willow flycatchers (*Empidonax alnorum* and *Empidonax traillii*) (Table 5). I also observed thirteen species that I did not place in either the grassland or shrubland ecological guild because they occurred in more than one habitat type (e.g., clay-colored sparrow [*Spizella pallida*], red-winged blackbird [*Agelaius phoeniceus*], and swamp sparrow [*Melospiza georgiana*]) (Table 5).

For 2011 data, four OGBB abundance-habitat models had ΔAIC_c values < 2.0 (Tables 6, 7, and 8). The best habitat model retained three predictor variables: standing dead cover, plant taxa richness, and Robel pole score, each of which was negatively associated with 2011 OGBB abundance, indicating that OGBB abundance increased in habitats with lower and less dense vegetation, less standing dead cover, and fewer plant taxa. Habitat models for OGBB abundance in 2012 included three "best fit" models, with graminoid cover, live cover, and forb cover as predictor variables. The most parsimonious model had one variable, graminoid cover, which had a positive correlation with 2012 OGBB abundance (Table 6). There were three models with ΔAIC_c values < 2.0 for bobolink abundance in 2011 and four models that met that criterion for bobolink abundance in 2012. For 2011, bobolink abundance models had three predictor variables: standing dead, plant taxa richness, and Robel pole. Each had a negative association to the response variable. Robel pole score was the sole predictor variable for the model with the lowest AIC value. For 2012, bobolink abundance models had four predictor variables, including graminoid cover, Robel pole score, litter depth, and distance to nearest forest edge. In the model with the lowest AIC value, graminoid cover and Robel pole score were the two predictor variables retained; graminoid cover had a strong positive correlation with bobolink abundance in 2012, while Robel pole score had a negative association, as it did in 2011 (Table 7). Savannah sparrow abundance in 2011 had three models with $\Delta AIC_c < 2.0$. The best model had Robel pole score (negative correlation) and forb cover (positive correlation) as the only two predictor variables (Table 8). For savannah sparrows in 2012, there were two models; the best model had graminoid cover (positive correlation) as the only important covariate. Henslow's sparrow and sedge wren abundances were too low to construct models.

In summary, for 2011 data, Robel pole score, an indicator of vegetation height and density, and plant taxa richness were two of the most important predictor variables and were retained in many of the models for abundance of OGBBs, bobolinks, and savannah sparrows (Tables 6, 7, and 8). Overall, in 2011, shorter, less dense vegetation with fewer plant species were favored by OGBB. Interestingly, in 2012, five novel predictor variables, which were not retained in any of the 2011 models, were retained in models with $\Delta AIC_c < 2.0$: graminoid cover, live cover, litter depth, distance to nearest forest edge, and goldenrod cover. For each response variable in 2012, the models with the lowest AIC_c values all had graminoid cover as an important covariate. In general, OGBB in 2012 predominately favored increased graminoid cover, as well as shorter, less dense vegetation.

Discussion:

My study examined OGBB abundances and habitat preferences for grassland birds at Fort Drum with the intention to use the results to inform management decisions locally and, hopefully, in grasslands throughout the Northeast. During the two-year study, I observed five OGBB. The most abundant were bobolinks and savannah sparrows (96% of all OGBB observations). Abundances of Henslow's sparrows and sedge wrens, both state-threatened species, were low (3% of all OGBB observations). Furthermore, in both years combined, the abundance of SBB exceeded OGBBs by 62.6% (Figures 4 and 5). Between years, unmanipulated plots had similar vegetation (Table 3). However, mowing in 2012 caused a significant decrease in standing dead cover, litter depth, and Robel pole score and an increase in graminoid cover on plots sampled in both years (Tables 1 and 2). Disturbance generated by the mowing regime apparently did not have a significant impact on the mean abundance of OGBBs, although there was some tendency toward greater abundances in mown and partially mown plots (Figure 6). Lastly, Robel pole score and plant taxa richness were the two most common predictor variables for habitat models in 2011, both of which decreased with increasing OGBB abundance. The most prevalent predictor variable in 2012 models was graminoid cover, which had a positive relationship with

OGBB abundance. Notably, there were five novel predictor variables (graminoid cover, live cover, litter depth, distance to nearest forest edge, and goldenrod cover) in 2012 models that were not seen in 2011 models (Tables 6, 7, and 8).

Historically, the abundance of grassland birds at Fort Drum, including rare species such as the sedge wren and Henslow's sparrow, has generally been higher than in most grassland sites in the Northeast (Shriver et al. 2005). The extremely low abundances of Henslow's sparrows and sedge wrens found during 2011 and 2012 point counts is concordant with the declining trend of grassland birds in New York and throughout the Northeast, as is the relatively low species richness at Fort Drum (Figures 4 and 5) (Shriver et al. 2005, Sauer et al. 2012). Fourteen years prior to this study, point counts from the same training areas on Fort Drum found much greater abundances of Eastern meadowlarks (*Sturnella magna*) and Henslow's sparrows, species that were absent or nearly absent during my study (Bolsinger et al. 1999). The decline of OGBBs, particularly rare species, on Fort Drum has been accompanied by concurrent decreases in detection of Henslow's sparrows during roadside surveys in Jefferson County, New York (adjacent to Fort Drum) between 1997 and 2012 (Krebs 2002, Lazazero and Norment 2006, A. Kirk unpublished data). Declines in the abundance of Eastern meadowlarks, Henslow's sparrows, and most other OGBBs also have been seen throughout New York State since the BBS began in 1966 (Sauer et al. 2012).

The decline of grassland bird species on Fort Drum could be due to diminishing habitat quality. Dwindling abundances of habitat specialists, such as

Henslow's sparrows and sedge wrens, often occur when habitat quality declines (Confer and Knapp 1981, McKinney and Lockwood 1999, Devictor et al. 2008). However, degradation of OGBB habitat at Fort Drum does not completely explain this historical collapse for two reasons: 1) the grasslands are large (>5,500 ha), habitat diversity generally increases with area, and as habitat diversity increases so should species diversity (Ribic et al. 2009); as evidence of this 2) OGBBs have been diverse and abundant at this location in the past (Bolsinger et al. 1999, Krebs 2002). Therefore, the decline of the habitat specialists at Fort Drum could be due to the precipitous decline of grassland habitat in the rest of the Northeast region, making it difficult to attain the ideal of locally large breeding populations of OGBB, even with suitable habitat, because there would be a smaller regional source pool of grassland birds (Askins 2001, Foster et al. 2002).

As an ecological guild, grassland birds have some general habitat preferences. In the Northeast, they prefer generally lower height and density of vegetation, increased graminoid cover, lower shrub cover, and large habitat areas away from edges (Vickery et al. 1994, Norment et al. 1999, Bollinger and Gavin 2004, Renfrew et al. 2005), as was also seen in this study. Individual grassland species, however, have unique habitat preferences. For example, Henslow's sparrows prefer greater vegetation height and density, some shrub cover, increased litter depth, wet meadows, and grassy swamps (Rising 1996, Winter 1999, Herkert 2007, Jacobs et al. 2012). Likewise, bobolinks in the Northeast prefer hay fields ≥ 8 y old (Bollinger and Gavin 1992). My study suggests that bobolinks and savannah sparrows at Fort Drum both prefer shorter, less dense vegetation with increased graminoid cover, although bobolinks preferred thicker litter while savannah sparrows preferred greater forb and goldenrod cover. Additionally, the savannah sparrow breeds in a variety of habitats across its range; although it prefers relatively dense ground vegetation, it may be found in any habitat type from alfalfa fields to sedge bogs and roadsides (Vickery et al. 1999, Wheelwright and Rising 2008). Although each species has unique habitat preferences, these can vary across their range and between years, as shown by Rotenberry and Wiens (2009) for a community of shrub-steppe birds in western North America. The above observations suggest that no one grassland habitat type will support all OGBBs, which may make it difficult to manage for the entire ecological guild in one area.

Furthermore, the lack of difference in OGBB abundance that I observed between mown, partially mown, and unmown areas suggests that common grassland species (bobolinks and savannah sparrows) are able to use a variety of habitat types with substantial differences in vegetation height and density, litter depth, standing dead cover, and graminoid cover (Figure 6). This supports the idea that these species are habitat generalists within their ecological guild and use a variety of grassland habitat types (Mengel 1970, Sample and Mossman 1997, Warren and Anderson 2005, Sliwinsky and Koper 2012). Another habitat generalist, the Eastern meadowlark (Sample and Mossman 1997), was absent from the study site. This could be because there is not enough of a regional “draw” due either to a lack of suitable habitat in the Northeast or to declining populations, even though Fort Drum apparently has suitable

Eastern meadowlark habitat available (Bolsinger et al. 1999). Another possibility for Eastern Meadowlark decline and for the decline of grassland birds in general is the lethal effect of insecticides on grassland birds during the breeding season (Mineau and Whiteside 2013).

This trend in OGBBs using a variety of habitat types is seen again in my results, which showed differences between the best grassland bird-habitat models for 2011 and 2012 (Tables 6, 7, and 8). For example, Robel pole score, a measure of vegetation height and density and a predictor variable in all 2011 bird-habitat models, was not nearly as prominent in 2012 models. Also, five predictor variables were selected in 2012 models that were not in any of the 2011 models. Because I conducted the study during two consecutive years, at the same locations, and with similar proportions of the OGBB species present, I expected similar results for the bird-habitat models. However, the habitat varied among years and so did the species' response. Some of the variation in the models is reflected in values of measured variables (i.e. Robel pole score, cover classes, etc.), while other possible sources of variation were not represented in any models because they were not measured and so are unknown. This is suggested by the similar vegetation in unmown plots between years (Table 2), indicating that the measured variables in unmanipulated parts of my study area were similar between years and that differences between the models were in part due to a change in unmeasured habitat variables between years or a change in species response to available habitat.

One possible source of between-year differences in habitat involved 2012 mowing operations, which resulted in some point counts plots having significantly shorter, less dense vegetation and more graminoid cover (Figures 3 and 4). However, 2011 bird-habitat models indicated that OGBBs preferred shorter, less dense vegetation, but OGBB abundance was not significantly affected by mowing treatment (Table 3, Figure 6). In an effort to remove any possible mowing treatment effect, I also ran 2012 bird-habitat models using only unmown plots. This still resulted in model discrepancies between years, with Robel pole not appearing in any of the models in 2012 and graminoid the most common predictor variable in 2012 (Appendix 4). Therefore, the OGBBs present may have responded to the habitat in ways that the models could not clearly represent. The OGBBs remain as they are – obligate grassland breeders, but habitat variation may affect their site selection. In turn, the response of grassland birds to among-year habitat differences affects bird-habitat modeling; in one year vegetation height and density may be prominent predictors in grassland bird-habitat models, while in the next it could be percent cover of graminoid vegetation, as in this study (Tables 6, 7, and 8). The ability of models to predict habitat preferences is especially limited for species that, within their ecological guild, are generalists (Rotenberry and Wiens 2009).

Similarly, bird-habitat models generated in the 1970s by Rotenberry and Wiens performed poorly when used to predict abundances for the same shrub-steppe region 20 years later (2009). Fuller et al. (1997) had similar results for a study of farmland birds that was repeated six times over a 20-yr period; each time the bird-habitat models had substantial variation in selected predictor variables. The poor ability of models to predict future bird abundance based on habitat selection indicates that annual habitat variation and species' responses may not be well-accounted for in bird-habitat models. Also, there always remains the reality that not all of the habitat variables important to grassland birds can be measured and that these unmeasured variables may be important for habitat selection (Wiens 1989). The results of this study demonstrate that models only explain a portion of OGBB habitat selection and that as habitat varies among years, so will species responses.

Taken as a whole, my results support what is commonly accepted about grassland bird habitat preferences in the Northeast, in reference to general bird-habitat relationships; however, my results more closely align with a growing body of evidence suggesting that management should not attempt to create extremely specific habitat features but, rather, preserve large areas that allow for increased grassland habitat diversity (e.g., Fuhlendorf and Engle 2001, Rahmig et al. 2009, Rotenberry and Wiens 2009, Jacobs et al. 2012).

Grassland Bird and Grassland Habitat Management at Fort Drum:

At Fort Drum, the effects of fragmentation, small habitat patches, and increased edge on OGBB are greatly reduced by the large and mostly contiguous grasslands in Training Areas 12 and 13. Although grassland birds sometimes may nest successfully in small (~10 ha) habitat patches (Wiedman and Litvatis 2011), these small patches do not have the same habitat diversity that large patches do (Ribic et al 2009). Also, small habitat patches may act as ecological traps (Schlaepfer et al.

2002) and many studies have shown that OGBB nest success tends to increase with patch size (e.g., Balent and Norment 2003, Bollinger and Gavin 2004). Habitat diversity and area are important when managing for a variety of species within the same ecological guild when: 1) each species has its own niche, slightly different habitat preferences, and 2) when habitat variation among years affects habitat site selection (Rotenberry and Wiens 2009, Jacobs et al. 2012). Therefore, I propose that management, especially on Fort Drum, should focus on maintaining large contiguous patches that promote a mosaic grassland landscape that contains smaller patches varying in vegetation height, density, species composition, and age.

In the Northeast, multiple studies have identified management methods that use land-use practices to promote OGBB habitat and populations, such as suggested dates for haycropping and how long to rest paddocks between grazing (Bollinger et al 1990, Perlut and Strong 2011). Disturbance in grasslands generated by military training, including use by tracked and other vehicles, could serve as a surrogate to a natural disturbance regime, as vegetative cover and biomass are significantly although temporarily reduced by vehicle use (Jones 2003, Althoff and Thien 2005, Foster et al. 2006). Perhaps a moderate amount of tracked vehicle use could be useful in maintaining open spaces and slowing succession. However, the long-term effects of tracked military vehicles on OGBBs needs further study, and care needs to be taken to ensure that the rate of vehicle disturbance is not greater than natural regeneration or does not occur more often than a natural disturbance regime (Severinghaus and Severinghaus 1982, Hirst et al. 2000, Foster et al. 2006).

Given that grasslands are disturbance-mediated systems that benefit from regular disturbance, forms of incidental management (e.g., hay cropping, grazing, and airfields) may be the best option for regional grassland management because they require no additional resources. Without a disturbance regime, northeastern grasslands quickly move through succession to become forested (Hunter et al. 2001, Foster et al. 2002). The means to maintain large grasslands continually solely for OGBBs are not always feasible at the landscape or regional scale because of large area requirements for OGBB, the persistent management required to maintain the grasslands and prevent succession, and limited resources. Ideally, grassland management should work alongside farmers, airports, and military bases to accomplish the goals of the organization while preserving grassland bird habitat in the Northeast.

At Fort Drum, this opportunity to cooperatively manage for military training and grassland birds exists at the local scale. The objectives for grassland bird management on Fort Drum should be to work cooperatively across multiple departments on the base to best achieve the following goals: 1) promote grassland bird species diversity by creating variety in the vegetation, and 2) maintain large open spaces. Because the primary purpose of the base is to train soldiers, grasslands at Fort Drum already need to be maintained as open space for military training. As the base works to maintain open spaces for training, it should simultaneously be able to accomplish the subsidiary task of managing for grassland birds. Several other military bases have effectively accomplished the primary objective of a military base while managing for wildlife (e.g., Conkle and Schwartz 2011, Natoli 2011). I suggest that 1) mowing be done on a rotational basis – creating varied field structure and ages for OGBB (Bollinger and Gavin 1992, Bollinger 1995), and 2) that at least one eighth (697 ha) of the grasslands not be used for training during the breeding season, from 1 May to 15 August, to allow for late-breeders and double broods (Herkert et al 2001, Herkert et al. 2002).

Although the opportunity exists to manage for both grassland birds and military training at Fort Drum simultaneously, there is an unfortunate bureaucratic division that exists between the Environmental Division of the Department of Public Works and Integrated Training Area Management (ITAM), the department responsible for maintaining the land for military training. The Environmental Division seeks to manage for conservation purposes, while ITAM's primary concern is managing habitat for military training purposes. In the past, the differences in each department's primary objectives have created some tension, although increased cooperation and communication could allow both departments to meet their goals. Some practices used at Fort Drum, such as dormant season burns, actually increase shrub cover and work against the objectives of creating open spaces for military training, and promoting grassland bird populations (Mitchell 2000). This is not beneficial for either the military or grassland bird management. It should be possible for both departments to work together to design a rotational mowing schedule for grasslands that creates and maintains a heterogeneous landscape, promotes the greatest amount of contiguous grassland habitat, and provides sufficient open space for military training exercises. Whenever possible, military training also could be scheduled to reduce disturbance to grassland birds during the breeding season.

Although the primary purpose of many military installations is training, this does not mean that wildlife management is impossible. The Department of Defense has done well managing the natural resources on its >11 million ha, with more listed endangered and threatened species than any other federal organization and >40 species that occur only on Department of Defense lands (Sabella 2011). Many military installations have successfully preserved their natural resources while still accomplishing the primary objectives of the base. A few examples of successful management are: the U.S. Army Garrison in Hawaii that has successfully managed for 75 pairs of ‘elepaio (*Chasiempis sandwichensis ibidis*),a rare bird species; Fort Bragg’s protection of an endangered species, the red-cockaded woodpecker (*Picoides borealis*); and the Navy’s use of prescribed burns to manage for several species in the Southeast, resulting in the first sighting of the endangered reticulated flatwoods salamander (*Ambystoma bishop*) in more than a decade (Conkle and Schwartz 2011, Natoli 2011) . As one of the largest remaining grasslands in the Northeast, and as stewards of land entrusted to it by its citizens, Fort Drum has the responsibility to promote diverse grassland bird habitat, as long as doing so does not conflict with its primary military mission. Documenting population trends and habitat preferences is only useful if the opportunity for change in management practices exists (Elzinga et al. 2001, Shriver et al. 2005). At Fort Drum this opportunity exists.

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Tables and Figures:

	OGBB	SBB
	Mean abundance/plot	Mean abundance/plot
2011	3.29 (±0.301)(0,9)	3.90 (±0.304)(1,8)
2012	2.29 (±0.267)(0,6)	4.95 (±0.418)(0,11)
2011 & 2012	2.79 (±0.207)(0,9)	4.43(±0.263)(0,11)

Table 1: Abundance (mean ± SE, and range: minimum and maximum values) of OGBB (obligate grassland breeding birds) and SBB (shrubland breeding birds) at Fort Drum, New York.

Predictor variable	Test statistic		df	p-value (2-tailed)	2011		2012	
					Mean	±SE	mean	SE
Live cover	t	-1.165	27	0.254	83.973	0.838	77.826	1.418
Graminoid	t	1.550	27	0.133	39.439	1.983	42.949	1.741
Goldenrod	t	-0.341	27	0.736	23.740	1.629	16.646	1.608
Woody veg	t	-0.850	27	0.403	12.111	1.526	16.990	2.499
Standing dead	t	-0.284	27	0.779	1.411	0.187	2.989	0.403
Plant taxa richness	t	0.098	27	0.922	8.264	0.230	5.395	0.197
Litter depth (cm)	t	-1.339	27	0.192	3.422	0.270	4.611	0.206
Robel pole score	t	-1.965	27	0.060	5.703	0.248	3.576	0.374
Forb	W	0.547	27	0.585	33.535	1.794	37.094	1.877

t: paired t-test   W: Wilcoxon rank-sum test

Table 2: Comparison of unmown vegetation plots in 2011 and 2012 at Fort Drum, New York.

Dependent Variable	df	Test statistic		P-value	UM (n=28)		PM (n=5)		CM (n=8)	
					Mean	±SE	Mean	±SE	Mean	±SE
Live cover	2	F	2.106	0.136	77.826	1.418	75.058	2.197	71.487	3.568
Graminoid	2	F	4.135	0.024*	42.949 [a]	1.741	49.938	4.555	52.327[a]	2.292
Goldenrod	2	F	1.242	0.300	16.646	1.608	11.613	3.336	12.500	3.253
Woody veg	2	F	1.624	0.211	16.990	2.499	16.646	3.020	8.182	2.197
Standing Dead	2	F	11.232	0.000*	2.989 [b]	0.403	1.500	0.254	0.599 [b]	0.214
Plant taxa richness	2	F	1.254	0.297	5.395	0.197	5.908	0.157	5.891	0.272
Litter depth	2	F	5.165	0.010*	4.611[c]	0.206	3.851	0.346	3.360 [c]	0.298
Robel	2	F	9.704	0.000*	3.576 [d]	0.374	2.880 [d]	0.640	1.314 [d]	0.207
Forb	2	H	1.138	0.566	37.094	1.877	40.354	0.840	39.534	2.136
OGBB†	2	H	5.337	0.069	1.893	0.314	2.800	0.860	3.375	0.460
BOBO†	2	H	4.595	0.101	1.357	0.263	2.200	0.800	2.625	0.532
SAVS†	2	H	0.870	0.647	0.464	0.131	0.600	0.245	0.750	0.366

* significant at the 0.05 level.

† mean abundance/plot

Table 3: ANOVA (F) and Kruskal-Wallis (H) results for vegetation and grassland bird predictor variables at Fort Drum, Jefferson and Lewis Counties, New York, for three habitat manipulation treatment groups in 2012: unmown (UM), partially mown (PM), and completely mown (CM). Letters indicated statistically significant (p<0.05) differences among mowing treatments using Tukey's post-hoc test.

Species list and abundance (2011-2012)			
Abundance	Common name	Binomial	Alpha code
117	Bobolink	*Dolichonyx oryzivorus*	BOBO
87	Savannah sparrow	*Passerculus sandwichensis*	SAVS
5	Sedge wren	*Cistothorus platensis*	SEWR
2	Northern harrier	*Circus cyaneus*	NOHA
1	Henslow's sparrow	*Ammodramus henslowii*	HESP
183	Common yellowthroat	*Geothlypis trichas*	COYE
55	Song sparrow	*Melospiza melodia*	SOSP
50	Yellow warbler	*Setophaga petechia*	YWAR
35	Alder flycatcher	*Empidonax alnorum*	ALFL
22	Willow flycatcher	*Empidonax traillii*	WIFL
5	Gray catbird	*Dumetella carolinensis*	GRCA
2	American goldfinch	*Spinus tristis*	AMGO
2	Field sparrow	*Spizella pusilla*	FISP
1	Northern cardinal	*Cardinalis cardinalis*	NOCA
55	Swamp sparrow	*Melospiza georgiana*	SWSP
21	Red-winged blackbird	*Agelaius phoeniceus*	RWBL
18	Clay-colored sparrow	*Spizella pallida*	CCSP
16	American robin	*Turdus migratorius*	AMRO
4	American crow	*Corvus brachyrhynchos*	AMCR
3	American bittern	*Botaurus lentiginosus*	AMBI
3	Veery	*Catharus fuscescens*	VEER
3	Cedar waxwing	*Bombycilla cedrorum*	CEWA
2	Eastern kingbird	*Tyrannus tyrannus*	EAKI
1	Tree swallow	*Tachycineta bicolor*	TRES
1	Wild turkey	*Meleagris gallopavo*	WITU
1	Northern flicker	*Colaptes auratus*	NOFL

Table 5: Species list for Fort Drum, New York in 2011 and 2012. Species are divided by ecological guild – the top section contains OGBB (obligate grassland breeding birds), the middle section contains SBB (shrubland breeding birds), and the bottom section contains species that belong to neither ecological guild.

Response Variable	Rank	AICc	ΔAICc	Wi	K	Predictor Variable	β
OGBB 2011	1	165.559	0.000	0.32499	3	Standing dead	-0.174
						Plant taxa richness	-0.195
						Robel	-0.288
	2	166.125	0.566	0.24983	1	Robel	-0.243
	3	166.142	0.583	0.24772	2	Plant taxa richness	-0.145
						Robel	-0.318
	4	166.809	1.250	0.17747	4	Forb	0.119
						Standing dead	-0.165
						Plant taxa richness	-0.236
						Robel	-0.259
OGBB 2012	1	135.585	0.000	0.45256	1	Graminoid	0.575
	2	136.090	0.505	0.35157	2	Live cover	-0.144
						Graminoid	0.579
	3	137.260	1.675	0.19586	3	Forb	0.142
						Live cover	-0.217
						Graminoid	0.552

Table 6: Bird-habitat models (ΔAIC_c values < 2.0) for obligate grassland breeding birds (OGBB) for 2011 and 2012 at Fort Drum, New York. The following predictor variables have different units: litter depth (cm), plant taxa richness (abundance), Robel (pole score), and nearest forest edge (m). The remaining predictor variables are vegetation cover classes. Beta (β) is the slope of the relationship between the predictor and response variables.

Response Variable	Rank	AICc	ΔAICc	Wi	K	Predictor Variable	β
BOBO 2011	1	131.443	0.000	0.4428	1	Robel	-0.124
	2	131.699	0.256	0.3896	2	Taxa diversity	-0.180
						Robel	-0.210
	3	133.385	1.942	0.1677	3	Standing dead	-0.113
						Plant taxa richness	-0.213
						Robel	-0.196
BOBO 2012	1	120.910	0.000	0.3330	2	Robel	-0.272
						Graminoid	0.557
	2	121.646	0.736	0.2305	3	Litter depth	0.167
						Robel	-0.339
						Graminoid	0.570
	3	121.684	0.774	0.2261	1	Graminoid	0.686
	4	121.828	0.918	0.2104	4	Nearest forest	0.202
						Litter depth	0.201
						Robel	-0.283
						Graminoid	0.611

Table 7: Bird-habitat models (ΔAIC_c values < 2.0) for bobolinks (BOBO) for 2011 and 2012 at Fort Drum, New York. The following predictor variables have different units: litter depth (cm), plant taxa richness (abundance), Robel (pole score), and nearest forest edge (m). The remaining predictor variables are vegetation cover classes. Beta (β) is the slope of the relationship between the predictor and response variables.

Response Variable	Rank	AICc	ΔAICc	Wi	K	Predictor Variable	β
SAVS 2011	1	118.343	0.000	0.4221	2	Robel	-0.287
						Forb	0.270
	2	118.859	0.516	0.3261	1	Forb	0.377
	3	119.377	1.034	0.2517	3	Standing dead	-0.187
						Robel	-0.235
						Forb	0.245
SAVS 2012	1	82.623	0.000	0.6895	1	Graminoid	0.201
	2	84.219	1.596	0.3105	2	Goldenrod	0.184
						Graminoid	0.251

Table 8: Bird-habitat models (ΔAIC_c values < 2.0) for savannah sparrows (SAVS) for 2011 and 2012 at Fort Drum, New York. The following predictor variables have different units: litter depth (cm), plant taxa richness (abundance), Robel (pole score), and nearest forest edge (m). The remaining predictor variables are vegetation cover classes. Beta (β) is the slope of the relationship between the predictor and response variables.

Figure 1: Location of Fort Drum, New York.

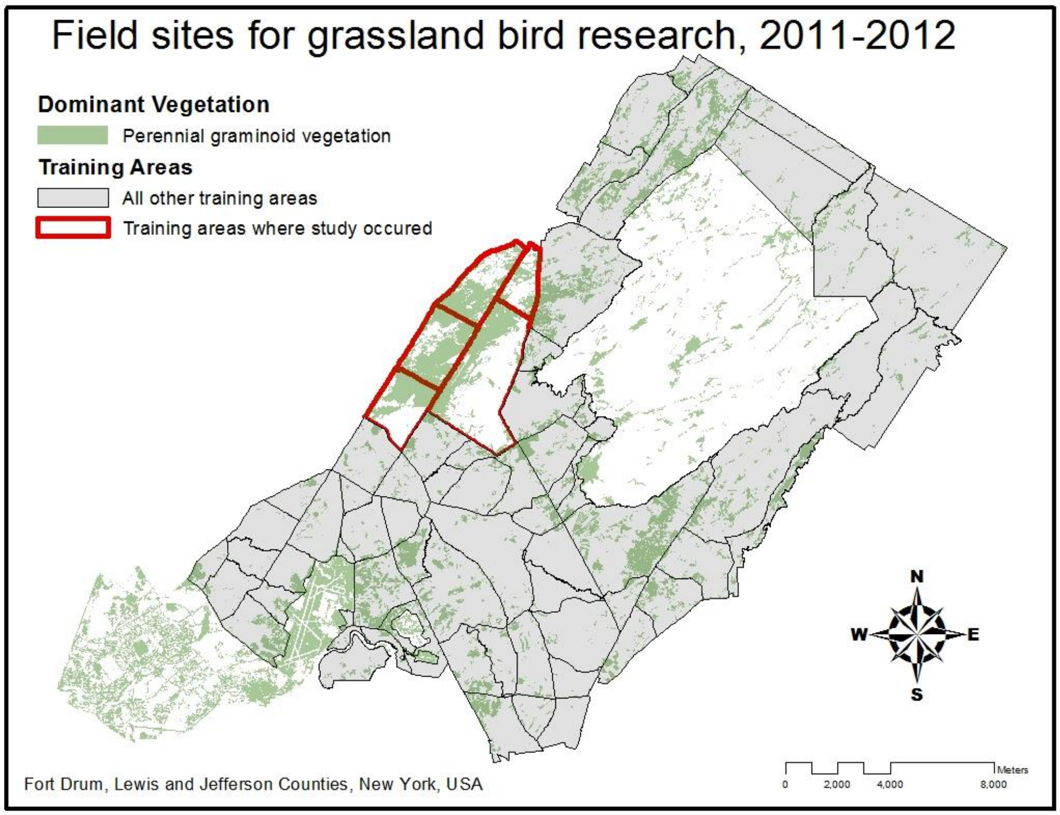


Figure 2: The five Training Areas (12B, 12C, 12D, 13A, and 13B) where most field research on grassland birds occurred during my study.

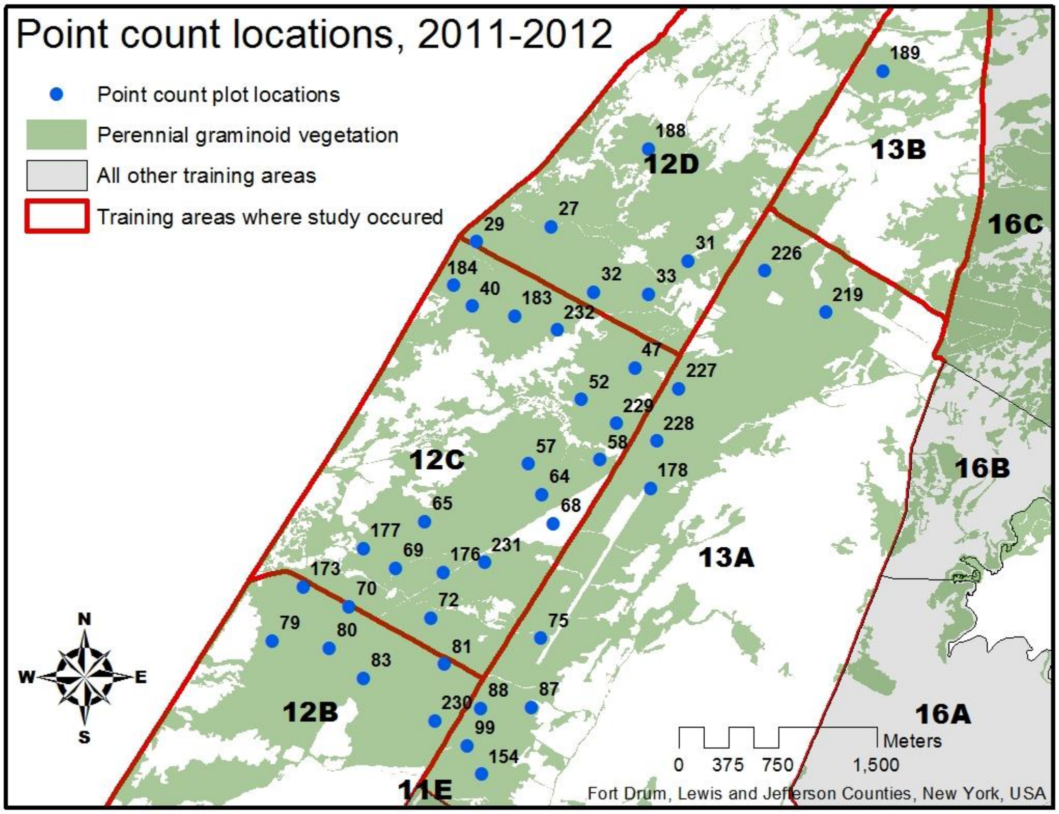


Figure 3: The location of the 41 point count plots used for grassland bird field research in 2011 and 2012.

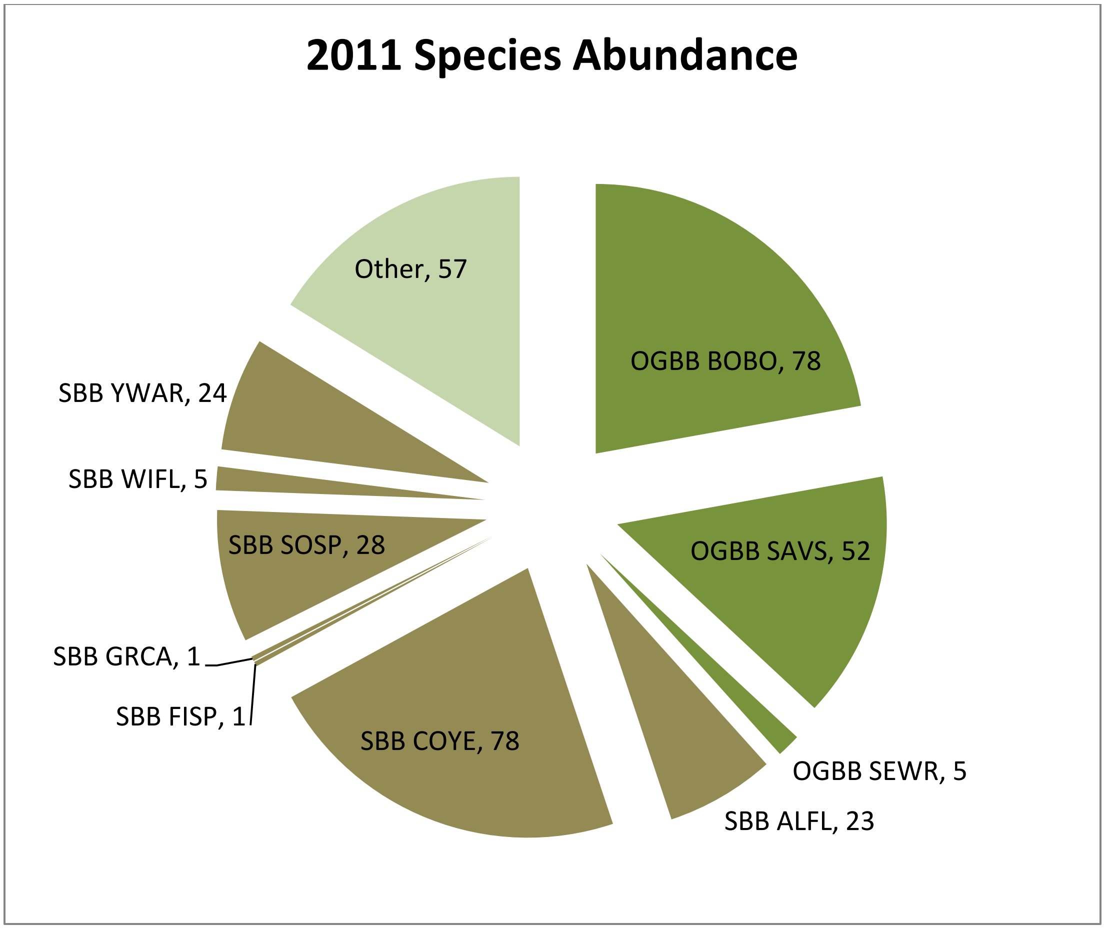


Figure 4: 2011 breeding bird species abundance at Fort Drum, Jefferson and Lewis Counties, New York for OGBB (obligate grassland breeding birds), SBB (shrubland breeding birds), and other (species that use various habitats). Abundances are from the first round of point counts, which had the greatest abundance of OGBB. Names of species represented by alpha codes are found in Table 5.

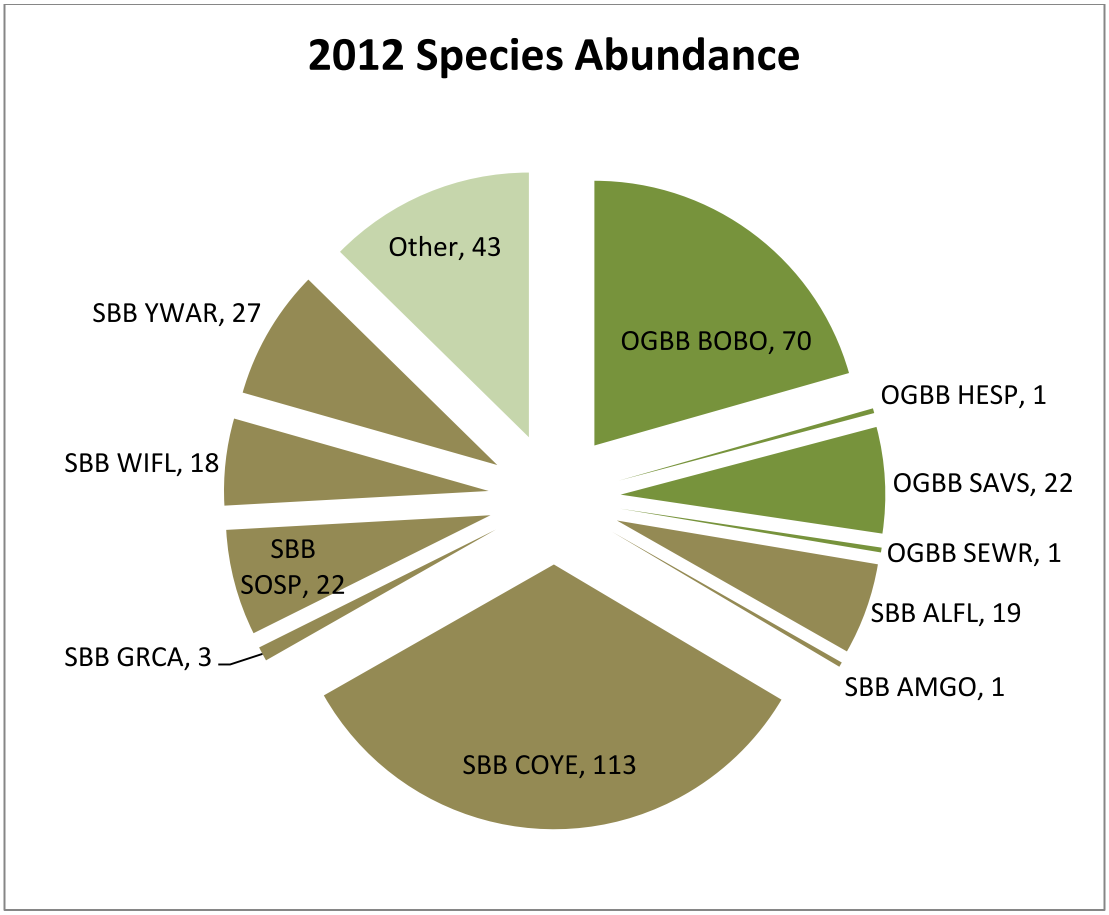


Figure 5: 2012 breeding bird species abundance at Fort Drum, Jefferson and Lewis Counties, New York for OGBB (obligate grassland breeding birds), SBB (shrubland breeding birds), and other (species that use various habitats). Abundances are from the first round of point counts that, which had the greatest abundance of OGBB. Names of species represented by alpha codes are found in Table 5.

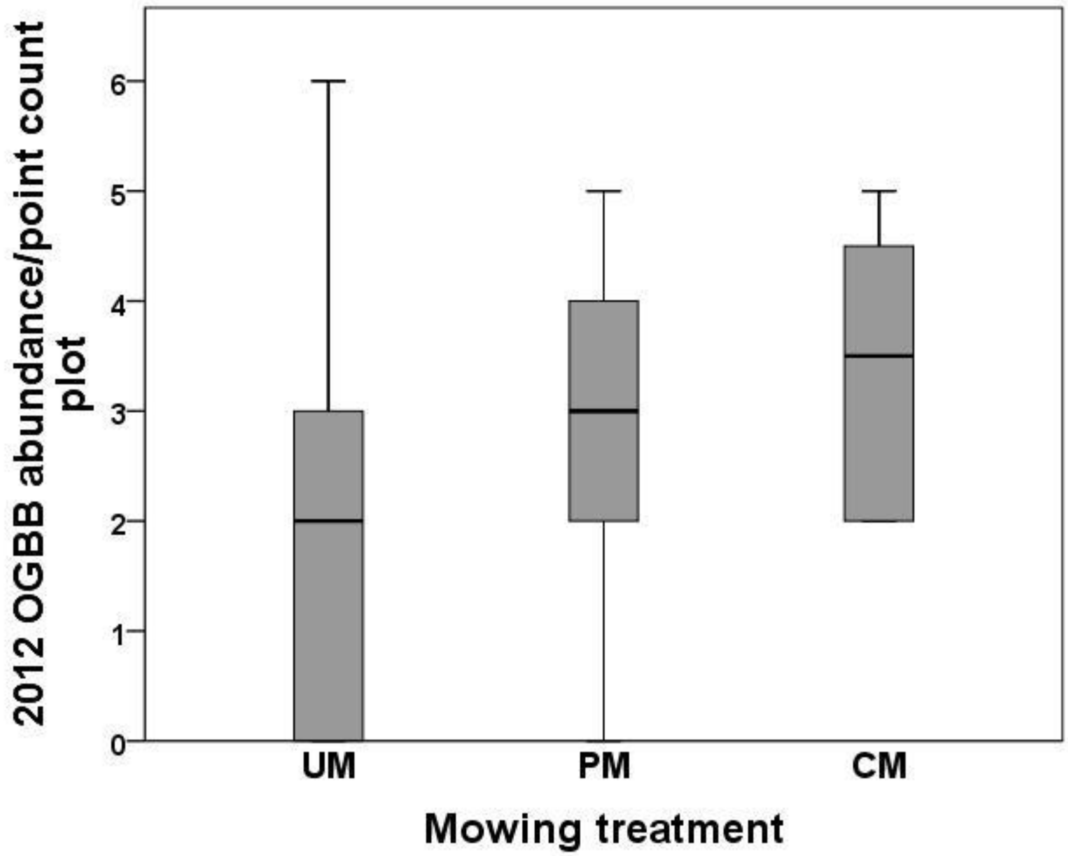
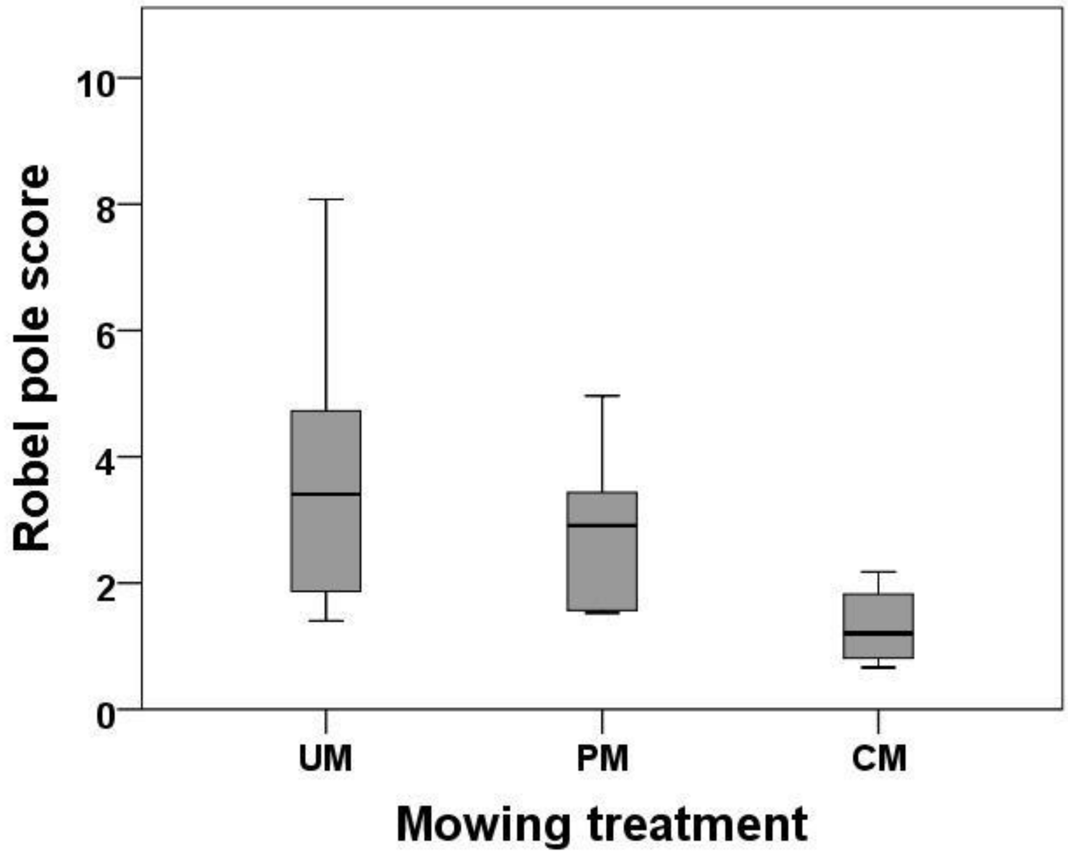


Figure 6: Comparison of 2012 obligate grassland breeding bird (OGBB) abundance/plot at Fort Drum, Jefferson and Lewis Counties, New York, among three habitat manipulations: unmown (UM, n=28), partially mown (PM, n=5), and completely mown (CM, n=8). The center line in each box represents the median, the boundaries of the box represent the first and third quartile, and the lines extending beyond the box represent the range of most of the values.

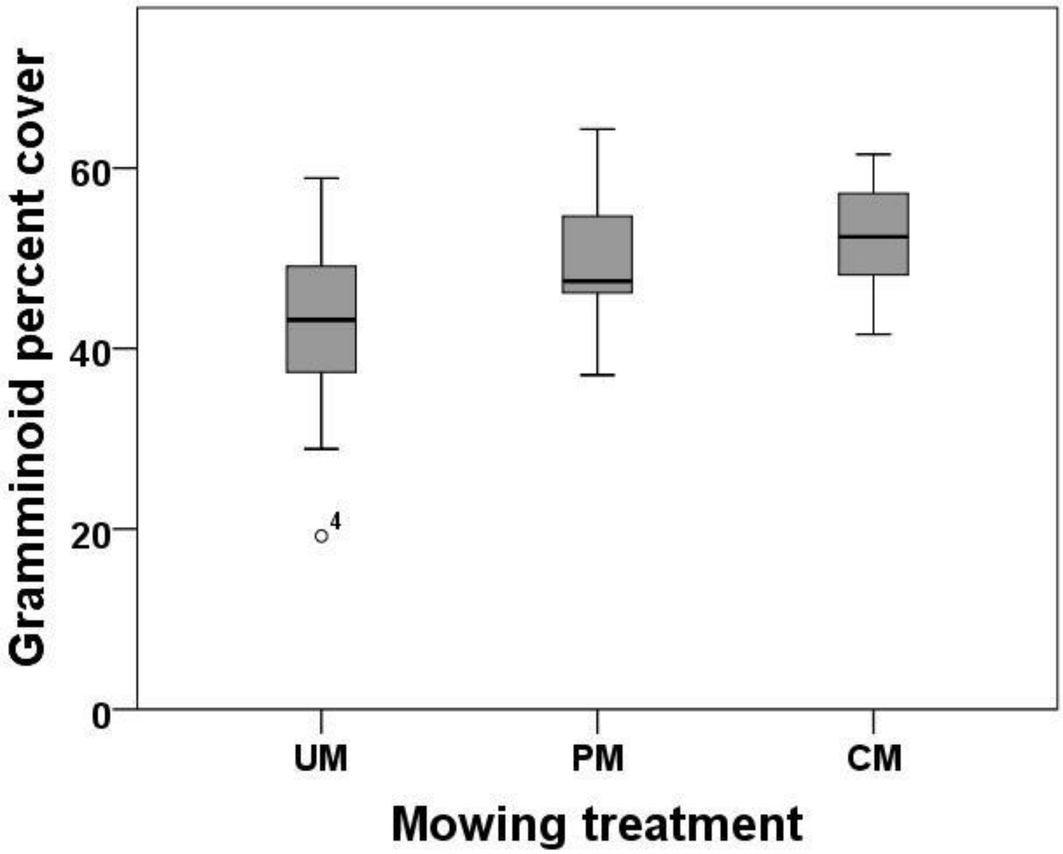
Appendices:

State-listed bird species: Fort Drum, NY		
Common name	**Binomial**	**Status**
Golden eagle	*Aquila chrysaetos*	endangered
Short-eared owl	*Asio flammeus*	endangered
Black tern	*Chlidonias niger*	endangered
Peregrine falcon	*Falco peregrinus*	endangered
Henslow's sparrow	*Ammodramus henslowii*	threatened
Upland sandpiper	*Bartramia longicauda*	threatened
Northern harrier	*Circus cyaneus*	threatened
Sedge wren	*Cistothorus platensis*	threatened
Bald eagle	*Haliaeetus leucocephalus*	threatened
Least bittern	*Ixobrychus exilis*	threatened
Pied-billed grebe	*Podilymbus podiceps*	threatened

Appendix 1: State endangered and threatened bird species present at Fort Drum, NY.



Appendix 2: Comparison of 2012 Robel pole score at Fort Drum, Jefferson and Lewis Counties, New York, within three habitat manipulations: unmown (UM, n=28), partially mown (PM, n=5), and completely mown (CM, n=8). The center line in each box represents the median, the boundaries of the box represent the first and third quartile, and the lines extending beyond the box represent the range of most of the values.



Appendix 3: Comparison of 2012 graminoid vegetation percent cover at Fort Drum, Jefferson and Lewis Counties, New York, within three habitat manipulations: unmown (UM, n=28), partially mown (PM, n=5), and completely mown (CM, n=8). The center line in each box represents the median, the boundaries of the box represent the first and third quartile, and the lines extending beyond the box represent the range of most of the values.

Response Variable	Rank	AIC_c	ΔAIC_c	W_i	K	Predictor Variable	β
OGBB 2012UM	1	86.897	0	1.000	2	Graminoid	0.690
						Litter depth	0.327
BOBO 2012UM	1	76.637	0	0.694	2	Graminoid	0.805
						Litter depth	0.404
	2	78.274	1.637	0.306	3	Graminoid	0.760
						Litter depth	0.373
						Live cover	-0.206
SAVS 2012UM	1	52.842	0	0.711	1	Standing dead	0.410
	2	54.642	1.800	0.289	2	Standing dead	0.373
						Graminoid	0.262

Appendix 4: Habitat models (ΔAIC_c values < 2.0) using only unmown (UM) plots for obligate grassland breeding birds (OGBB), bobolinks (BOBO), and savannah sparrow (SAVS) for 2012 grassland birds at Fort Drum, New York. The predictor variables litter depth (cm), plant taxa richness (abundance), Robel (pole score), and nearest forest edge (m) have unique units. The remaining predictor variables are vegetation cover classes. Beta (β) is the slope of the relationship between the predictor and response variables.

Ecology and management of the Sedge Wren (*Cistothorus platensis*) at Fort Drum, New York

Introduction:

As described in the first chapter, obligate grassland breeding bird (OGBB) (*sensu* Vickery et al. 1999) habitat and abundance is declining throughout the Northeast. One OGBB species of particular interest is the sedge wren (*Cistothorus platensis*). This little-known species is a persistent breeder in New York, despite its low abundance in the state (McGowan and Corwin 2008, Sauer et al. 2012). Although recent declines in New York State abundance, as indicated by Breeding Bird Survey data (Sauer et al. 2012) and Breeding Bird Atlas distributions (McGowan and Corwin 2008), have been moderate, the sedge wren is listed as threatened in New York and endangered, threatened, or a species of special concern in eight other mid-western and northeastern states (Vickery 1992). Historically, the most abundant populations of sedge wrens in New York State have been in Jefferson County (McGowan and Corwin 2008), part of the ecologically significant St. Lawrence plain ecozone and home of Fort Drum – a military installation listed as an Important Bird Area (IBA) by the Audubon Society (Wells 1998).

Sedge wren ecology:

Sedge wrens are a small (7-10g), conspicuous passerine (Cory 1909). Their primary color ranges from off-white to tawny brown and black (Cory 1909). Their wings have a checked pattern and their tail tends to be erect – typical of wrens. Their short, thin beak and striped crown and back distinguish them from the morphologically similar and closely related marsh wren (*Cistothorus palustrus*).

Sedge wrens are often found in low depressions, preferring tall, dense grasses with moist soil but no standing water (Walkinshaw 1935, Sample 1989, Herkert et al. 2001). In Minnesota, sedge wrens preferred habitat that on average had 303 sedge stems/m^2, 16 forb stems/m^2, 50 shrub stems/ m^2, and an average grass height of 1.1m (Niemi 1985). Also in Minnesota, territory size averaged 1,780 m^2 (Burns 1982).

During the breeding season, sedge wrens are found primarily in the Midwest, near the Great Lakes, and into Canada. New York is near the northeastern limit of its range (Herkert et al. 2001, McGowan and Corwin 2008). They are persistent but not abundant in the state and are found most commonly in Jefferson and St. Lawrence Counties (Levine 1998). Sedge wren presence in New York appears to be a consequence of agricultural land clearing that began with European colonization, but they may have occurred in beaver and sedge meadows or in grasses near bogs (Herkert et al. 2001). Sedge wren wintering grounds stretch from southern New Jersey to Florida and across to the Gulf Coast of Mexico (Herkert et al. 2001).

Sometimes double brooded and polygynous, sedge wrens can raise one brood in May or June and a second as late as September, similar to other grassland breeding birds (Walkinshaw 1935, Burns 1982, McGowan and Corwin 2008). Interestingly, among North American passerines, sedge wrens are suspected to be uniquely nomadic in that they may breed in one location, only to migrate farther and breed again during the same breeding season (Burns 1982, Herkert et al. 2001). The ability of sedge wrens to improvise and invent new songs (Kroodsma and Verner 1978, Kroodsma et al. 1999, Herkert et al. 2001) suggests that their highly adaptable song may in part reflect their nomadic lifestyle (Kroodsma et al. 1999).

Limited quantitative data suggest that during the breeding season sedge wrens primarily eat insects and spiders (Howell 1932, Walkinshaw 1935). However, little research has been done on sedge wren diets. Several other gaps in knowledge about sedge wrens present priorities for future research, such as the need for a quantitative study of winter habitat preferences, population demographics, and nomadic movement during the breeding season (Herkert et al. 2001).

Objectives:

The state-listed status of the sedge wren in the Northeast makes conservation of this species a priority. Therefore, in 2011 and 2012, I conducted field research on sedge wrens at Fort Drum, NY. Sedge wren populations have been consistently present at Fort Drum but with variable abundance (Bolsinger pers. comm.). The relative abundance of the species at Fort Drum, with over 5,500 ha of grassland on the base, potentially make it an ideal site for studying sedge wrens. The main objective for my study was to gather information on sedge wren breeding ecology, habitat selection, and population size on For Drum. Additionally, I also sought to compare habitat vegetation variables within sedge wren territories to random locations outside of the territories to determine sedge wren breeding habitat preferences.

Methods

The methods and study site used for sedge wren territory research were similar to those for the more general study on OGBB habitat preferences at Fort Drum, NY (Chapter 1), with a few exceptions and additions: Although point counts were used as the primary method for identifying the location of sedge wrens, extensive searches for sedge wrens were made throughout training areas 12B, 12C, 12D, 13A, and 13B (hereafter, training areas 12 and 13) as well as some additional areas on and off the base. Once a territory was found, I then determined a perimeter and approximate center for the territory by observing male song perches and locating females. Since females are morphologically identical to males, the identification was based on behavior and the known presence of a singing male. The same vegetation variables within the territory were measured using the same cover classes as in the more general study of grassland breeding bird habitat selection at Fort Drum (Chapter 1). The vegetation was still sampled in the four cardinal directions but at shorter (5 m) intervals to still provide adequate sampling within the small territories. Lastly, in addition to measuring vegetation within territories, for comparison, I also measured vegetation at a random location outside of each territory. The random location was established with a Geographic Information System (ArcGIS ESRI, ArcMap 10.1) by using parameters to ensure that the random location was not within an adjacent territory or within a 30 m radius of the territory center (a 30 m radius was slightly larger than radius of any of the territories), or greater than 100 m from the territory center. Additionally, data from 2006-2010 on the location and abundance of sedge wren territories were provided by the Environmental Division at Fort Drum (Bolsinger pers. comm.). These data did not contain any information on vegetation characteristics associated with sedge wren territories.

Statistical analysis combined 2011 and 2012 territory data as the vegetation did not differ significantly between years (Chapter 1). Descriptive statistics were used for the majority of the analysis due to the small sample size; although, to compare the sedge wren territories to the random locations I used the non-parametric Wilcoxon rank-sum test.

Results

In 2011 and 2012, I observed 9 sedge wren territories the first year and 3 the next in training areas 12 and 13, despite extensive searches made throughout Fort Drum (Figures 1 and 2). Several territories were in low depressions, such as the site of an ephemeral streambed, although standing water was present only in one territory. Tall grasses (1.5-2 m), such as reed canary grass (*Phalaris arundinacea*), were common near many sedge wren territories, with two territories occurring within the tall grasses. Territories did not contain any trees, although several had small (2 – 5 m) trees within 50 m of their borders. Male sedge wrens often used several shrub species (willow [*Salix* spp.], dogwood [*Cornus* spp.], and spirea [*Spiraea* spp.]) as song perches within their territories. Arrival dates, signaled by the presence of the first singing males, were on 3 June 2011 and on 21 May in 2012.

I observed several known females (n=5) during both years. Twice, an adult (presumed female) was seen carrying a bright green grub about 2.5 cm long to a nest. Once, I also observed three fledglings. They spent most of their time hopping in a small shrub and did not attempt to find cover even when I approached to within 1 m. They were remarkably visible, in contrast to their well-camouflaged nests. I found one non-breeding nest. The globular nest was elevated 34.5 cm in 65.0 cm high spirea and grasses. The 9.2 cm diameter nest had a 1.9 cm diameter opening, facing 83° E (true bearing), away from the 1.5 m high spirea located 1.3 m to the north of the non-breeding nest.

At Fort Drum, sedge wren territories found between 2006 and 2012 were clustered in training areas 12 and 13 (Figure 1). During that 7-yr period, the number of territories per year ranged from 3 – 41 (Mean: 15.429±5.01 standard error) (Figures 1 and 2). Typically, sedge wren territory abundance has remained lower than the mean, with the exception of 41 and 24 territories in 2009 and 2010, respectively (Figure 2).

In a comparison of habitat variables from sedge wren territories and a random location, no statistically significant differences were found (Table 1). There were minor, not statistically significant ($p<.05$), difference between percent live cover and litter depth with more cover and greater litter depth within the territories (Wilcoxon rank-sum tests; p = 0.099 and 0.060; Table 1). The small sample size may also be obscuring any significant differences. Within sedge wren territories, both graminoid cover (Figure 3) and litter depth (Figure 4) tended to be greater, but differences were not statistically significant than at random locations outside the territory. Goldenrod (*Solidago*) cover and woody vegetation cover tended to be lower within sedge wren territories than outside them. The remaining variables, live cover, forb cover, standing dead cover, Robel pole score, and plant taxa richness, had similar or identical values for the territories and random locations, (Figures 3 and 4).

Discussion

This portion of the obligate grassland breeding bird study conducted at Fort Drum, New York focused on sedge wren ecology. Through intensive observation, I sought to gain an increased understanding of this New York State threatened species. At Fort Drum, sedge wrens appear to prefer mesic soils, shallow depressions, tall grasses such as reed canary grass, and to use shrubs as singing perches. There was only one non-breeding nest observed, and the few observations of foraging behavior indicated an insectivorous diet. Overall, only 12 territories were observed during the 2011 and 2012 study; however, data from 2006-2012 indicate that sedge wren populations are established at Fort Drum (Figures 1 and 2), even if abundance varies substantially among years.

The low abundance of sedge wrens observed during this study is typical of local and regional patterns. Using similar methods to this study, an average of 1.33

(range 0-3) sedge wrens per year were recorded from 1996 to 1998 during point counts in the same training areas on Fort Drum (Bolsinger et al. 1999). Abundance during the 2011 and 2012 point counts was similar, with an average of 3 per year (range 1-5). Although sedge wren abundance on Fort Drum has varied widely, from no territories in 1997 to 41 territories in 2009 (Figure 2), the persistence of sedge wrens on Fort Drum adds to its standing as an Important Bird Area (IBA), but also to the weight of responsibility to properly manage the land for this and other rare species (Wells 1998).

Sedge wren abundance, even dating back to the mid-nineteenth century, has never been high in New York (DeKay 1844, Giraud 1844, Eaton 1914); some southeastern portions of the state rarely, if ever, report observations (McGowan and Corwin 2008). However, among the regions where sedge wrens frequently occur, Jefferson County tends to have higher abundances (McGowan and Corwin 2008). Furthermore, both the BBS and the New York State Breeding Bird Atlas reveal no significant changes in sedge wren abundance from 1966 to 2011 and 1980 to 2005, respectively (McGowan and Corwin 2008, Sauer et al. 2012). This indicates that sedge wrens are uncommon in New York, but persistent. Regionally, the status of sedge wrens in the Northeast is difficult to determine. As a state endangered species in all of the New England states except Rhode Island, where it is considered a rare vagrant, it is obvious that species abundance is low, but how low remains in question. The BBS does not report any trends for sedge wrens for the New England states because observations are rare and sedge wrens are nearly absent from the literature for the entire Northeast (Sauer et al. 2012). This may be due in part to the difficulties of studying a species that has low site tenacity, historically low abundances, and a sometimes nomadic late-season arrival that may be missed by typical monitoring protocols (Herkert et al. 2001, McGowan and Corwin 2008).

Several of my observations of sedge wren behavior and habitat are similar to patterns seen elsewhere, and reveal areas of interest for further research. My observations of sedge wrens defending territories that included low depressions are similar to that described by Herkert et al. (2001). This suggests that topography and soil moisture content should be included in sedge wren habitat modeling as important variables that may influence site selection. Also, low site tenacity may be influenced by sedge wren preference for mesic soils, since mesic habitat types are prone to drying or flooding (Hoffman and Sample 1988, Herkert et al. 2001). A positive correlation between reed canary grass and sedge wren territories has also been observed in the Upper Midwest (Kirsch et al. 2007). The observations of female sedge wrens carrying insects support suggestions of a primarily insectivorous diet, which fits with the taxon that wrens belong to, but again points out the lack of knowledge of the basic biology of the sedge wren (Herkert et al. 2001). The one non-breeding nest observed fell within the range of measurements described by Peck and James in Ontario (1987). Lastly, adding to other observations, the late May arrival of sedge wrens in 2012 suggest that some individuals may spend the entirety of their breeding season in New York and are not late-arriving nomadic migrants that breed once in the Midwest and then again in the Northeast, as is common for the species (Herkert et al. 2001, McGowan and Corwin 2008).

Management suggestions:

Overall, management suggestions for sedge wrens are typically to minimize disturbance, promote tall grasses, focus on mesic areas, and manage for low shrub cover (Hanowski et al. 1999, Roth et al. 2005, Kirsch et al. 2007, Robert et al. 2009). Grazing, burning, and mowing all reduce sedge wren abundance (Frawley and Best 1991, Herkert et al. 1996), but grasslands are a disturbance-mediated ecosystem and are maintained by disturbance, so appropriate disturbances ultimately benefit sedge wrens (Hunter et al 2001). Shrub removal, a necessary disturbance for maintaining grasslands, has effectively resulted in more sedge wrens in managed areas, where shrubs were removed by both burning and shearing, than unmanaged areas (Hanowski et al. 1999). At Fort Drum, some point count plots (n=13 of 41) were mowed in 2012. Sedge wrens were not found in any of the mowed sites in 2012, perhaps supporting habitat preference for taller vegetation (Herkert et al. 2001, Roth et al. 2005). Management for sedge wrens on Fort Drum, NY should also focus on areas with low depressions and mesic soils. Shrub-removal needs to be an active part of the management plan, although dormant season burning should not be used, as it increases forb and shrub cover in the Northeast (Mitchell 2000). Furthermore, increased cooperation between the department responsible for managing for military training purposes, Integrated Training Area Management (ITAM), and the

Environmental Division would allow for more effective management that accomplishes both the needs of the military base and fulfills the habitat needs of grassland birds, especially the sedge wren. It is possible, based on recent years with relatively high sedge wren abundance (Figure 2), that there is a large amount of suitable habitat for the species at Fort Drum, NY. This habitat is capable of supporting relatively large breeding populations; therefore, a regional decline in available habitat, combined with the species' low site tenacity and nomadic breeding season behavior, may be affecting sedge wren abundance on Fort Drum. If any sedge wren-specific management occurs, monitoring its effectiveness on abundance and habitat quality is important (Robert et al. 2009), as is the understanding that sedge wren populations often have low site tenacity because of their preference for a habitat that needs consistent management and is prone to drying and flooding (Herkert et al. 2001).

Grassland habitat management would be greatly helped by increased cooperation between two parties: Integrated Training Area Management (ITAM) and the Environmental Division. The Environmental Division's goal of managing for wildlife is often confounded by the actions of ITAM, whose goal is to manage the land for military training. It is clear that both parties' objectives can be realized. Sedge wren habitat would benefit from planned disturbances that promote low shrub cover and tall, dense vegetation. This also would promote ITAM's directive of maintaining open spaces for military training. Furthermore, many grassland management best practices in the Northeast would only further benefit ITAM's objectives, such as not using dormant season burns that will only increase shrub cover, effectively creating less open space than before burning (Mitchell 2000).

The need to prioritize grassland bird-habitat conservation is accentuated by the low abundance of sedge wrens found in this study and throughout the Northeast. This research also demonstrates some of the difficulties of studying sedge wrens in the Northeast, where there may be some consistently breeding populations, but with low to variable abundance. The persistent presence of this locally rare bird at Fort Drum, New York adds support to Fort Drum's status as an Important Bird Area (IBA) and necessity to the conservation of grassland habitat at Fort Drum.

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Tables and Figures:

Habitat variable	Test statistic		N	p-value (2-tailed)	Territories 2011 & 2012		Random 2011 & 2012	
					Mean	±SE	Mean	±SE
Live cover	W	1.647	12	0.099	87.662	1.164	91.167	1.951
Graminoid	W	-1.490	12	0.136	45.263	4.674	32.500	7.965
Forb	W	-0.549	12	0.583	33.296	3.726	30.208	7.408
Goldenrod	W	1.020	12	0.308	14.324	1.697	22.083	6.635
Woody veg	W	0.235	12	0.814	9.303	2.843	21.250	9.131
Standing dead	W	-0.059	12	0.953	1.240	0.330	1.458	0.840
Plant taxa richness	W	0.178	12	0.859	8.072	0.573	8.083	0.917
Litter depth (cm)	W	-1.883	12	0.060	5.262	0.722	3.750	0.869
Robel	W	1.098	12	0.272	6.123	0.404	7.073	0.963

W: Wilcoxon rank-sum test

Table 1: Comparison of sedge wren (*Cistothorus platensis*) territories with random locations for the combined years of 2011 and 2012 at Fort Drum, NY.

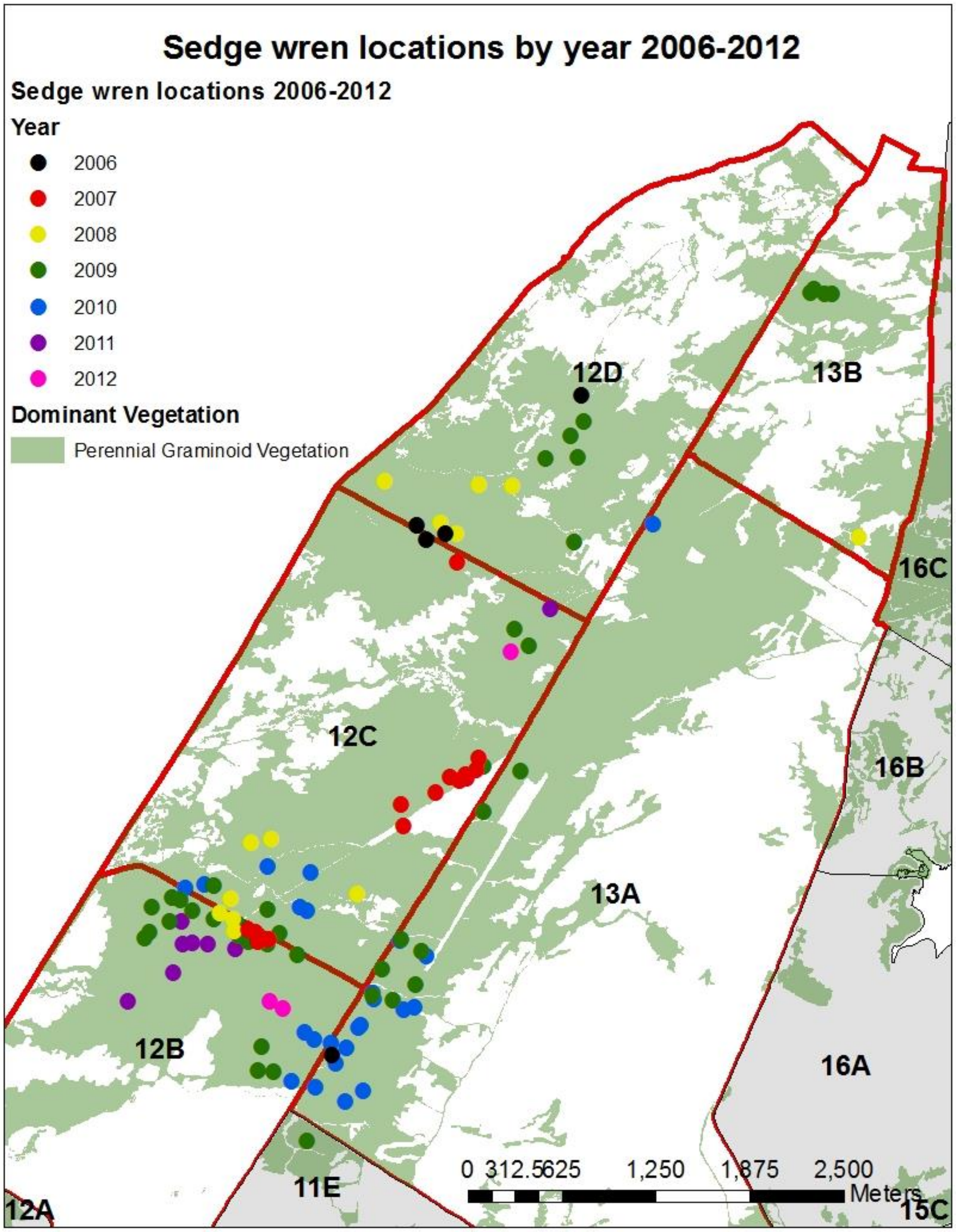

Figure 1: Sedge wren (*Cistothorus platensis*) abundance and location for 2006-2012 at Fort Drum, New York.

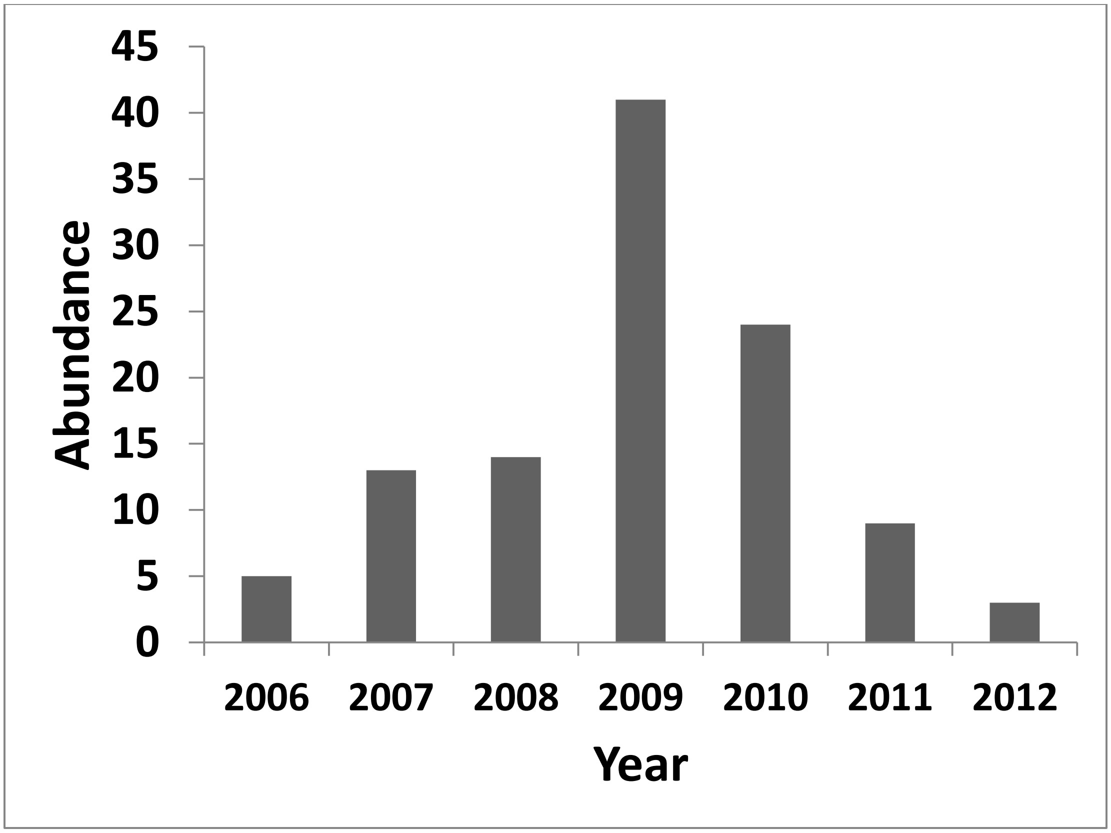

Figure 2: Number of sedge wren (*Cistothorus platensis*) territories at Fort Drum, New York, 2006-2012.

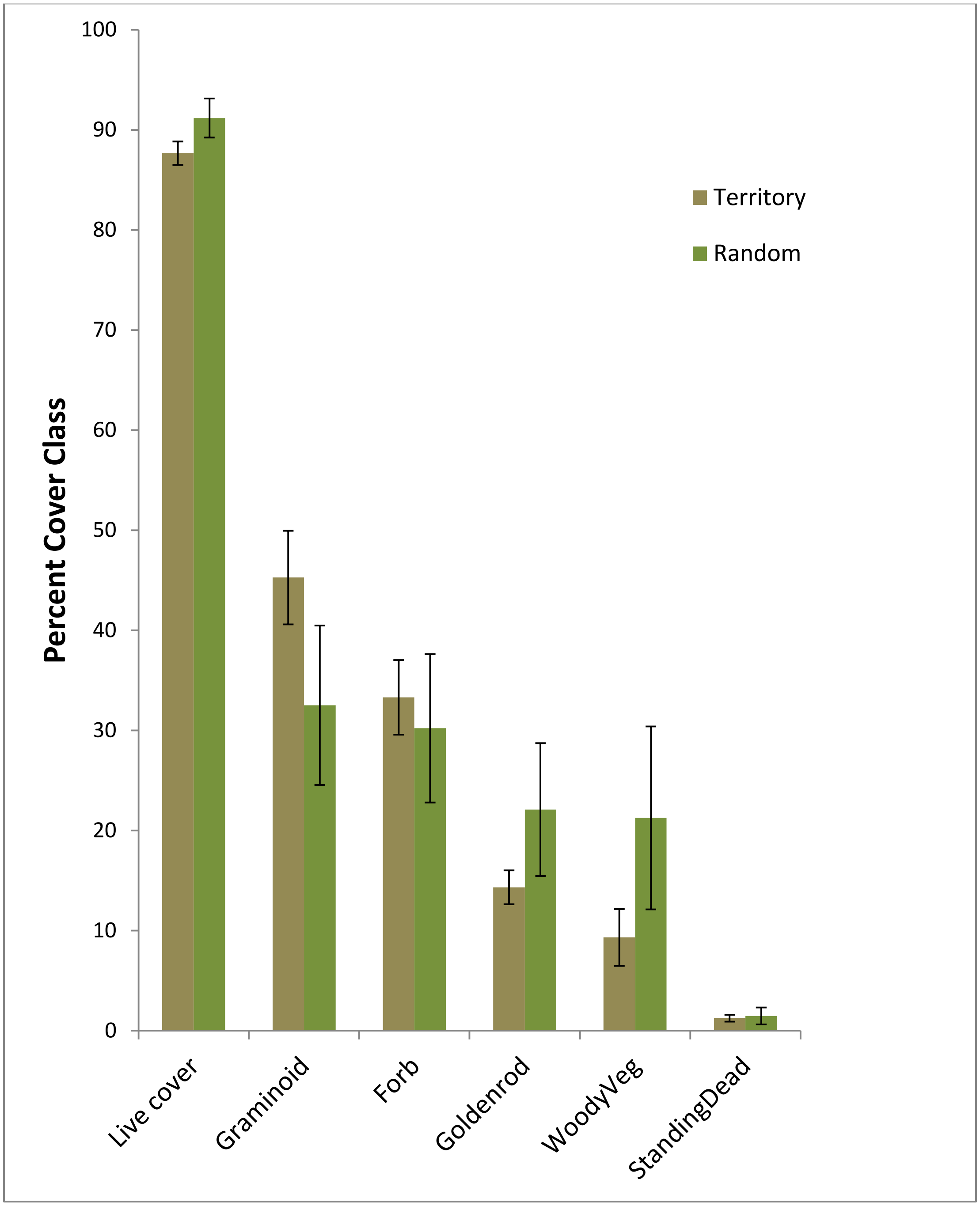

Figure 3: Vegetation cover class means at 12 sedge wren (*Cistothorus platensis*) territories and random locations outside of the territories at Fort Drum, NY.

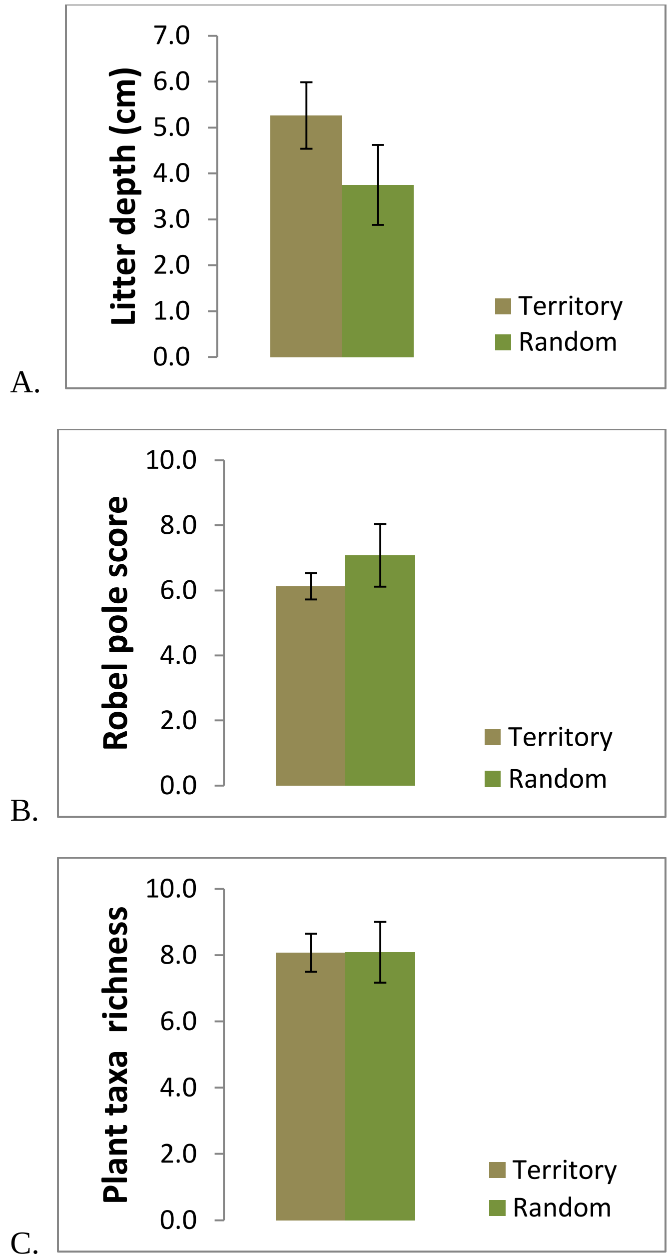

Figure 4: A) Litter depth (cm), B) Robel pole score, C) plant taxa richness means at 12 sedge wren (*Cistothorus platensis*) territories and random locations outside of the territories at Fort Drum, NY.