Response of grassland birds to management in national battlefield parks

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**ABSTRACT** Grassland birds are in steep decline, with population declines reported in 74% of North American grassland species in the past 50 years, and declines are particularly severe in the eastern United States. Habitat loss and agricultural intensification are major drivers of this decline. The U.S. National Park Service (NPS) maintains civil war battlefields in the eastern U.S. as historical parks that may also provide habitat refuge for grassland birds within an increasingly urbanized matrix. To assess the conservation importance of four National Battlefield Parks, we collected point count data from 2014-2019 and in 2021 of two grassland-breeding species, Eastern Meadowlark (*Sturnella magna*) and Grasshopper Sparrow (*Ammodramus savannarum*). We modeled the impact of habitat, landscape, and management covariates such as prescribed fire and agricultural use on the occupancy of these species. We found that habitat, landscape, and management covariates were all included in top occupancy models. Agricultural lease had a positive impact on occupancy of Eastern Meadowlark, but both species responded positively to harvest timing restrictions intended for grassland bird conservation. Occupancy was also consistently higher in hayfields than in row crops. Prescribed fire within the past 2 years had a positive impact on occupancy of Grasshopper Sparrow. These parks are valuable habitat for grassland birds, compatible with management activities that maintain the parks’ cultural goals.

**KEYWORDS** Ammodramus savannarum, Eastern Meadowlark, Grasshopper Sparrow, grassland birds, habitat management, national parks, public lands, Sturnella magna

# Introduction

With the loss of as much as 99% of shortgrass and tallgrass prairie area in North America since 1800 ([Samson and Knopf 1994](#ref-samson1994)) it is not surprising that grassland birds are in steep decline, with population losses reported in 74% of North American grassland-breeding species in the past 50 years ([Rosenberg et al. 2019](#ref-rosenberg2019)). These precipitous losses are attributed to conversion of grassland habitats to urbanized or forested areas, as well as changing agricultural practices that have favored intensive row crops over pasturelands ([Bollinger et al. 1990](#ref-bollinger1990), [Masse et al. 2008](#ref-masse2008), [Hill et al. 2014](#ref-hill2014)). In addition to changing crop species, agricultural intensification can take the form of earlier and more frequent harvests, which leave grassland-breeding birds without habitat during crucial reproductive periods or in ecological traps where they cannot fledge young due to harvest ([Bollinger et al. 1990](#ref-bollinger1990), [Rodenhouse et al. 1995](#ref-rodenhouse1995), [Masse et al. 2008](#ref-masse2008), [Allen et al. 2019](#ref-allen2019)). Increased use of non-native cool-season grasses in pasture and hayfields, which are less favorable for native bird species, has further compounded habitat degradation ([Walk and Warner 2000](#ref-walk2000)). Management decisions made for agricultural profit can drive grassland bird populations at a regional scale ([Allen et al. 2021](#ref-allen2021)).

Declines of grassland birds in the past half-century have been particularly rapid in eastern North America ([Sauer et al. 2017](#ref-sauer2017)). However, the majority of studies of habitat and management have taken place in the Midwest and central U.S. ([Dechant et al. 2002](#ref-dechant2002), [Hull 2002](#ref-hull2002)). Eastern grasslands exist in a different context. Historical accounts and the existence of distinct eastern subpopulations show that grassland bird species have always existed in isolated habitat patches in the east, maintained prior to European colonization by Native American land management alongside natural disturbance patterns ([Askins 1999](#ref-askins1999)). The initial deforestation of eastern North America by Europeans may have then created and expanded grassland habitats, leading to population increases or range expansions of grassland species that were harmed by the conversion of Midwest prairies to agriculture ([Brennan and Kuvlesky 2005](#ref-brennan2005), [McCracken 2005](#ref-mccracken2005)). Subsequent regrowth of eastern forests following historical deforestation, while positive for some forest species, has been associated with reduced habitat and population declines for grassland species. Although there has been research on these species in the east e.g. ([Warren and Anderson 2005](#ref-warren2005), [Irvin et al. 2013](#ref-irvin2013)), particularly on reclaimed strip mines ([Wray et al. 1982](#ref-wray1982), [Hill and Diefenbach 2013](#ref-hill2013)), there is still a need to identify the drivers of occurrence in the east and the relationships to specific management activities in this region.

Conservation and management of eastern grasslands for declining bird species is complex because the majority of this habitat is privately owned. The average percent private ownership by area for states on the east coast is 85% ([Rasker 2019](#ref-rasker2019)). Privately owned lands present both challenges and opportunities for conservation, but in the case of agricultural practices, financial incentives to plant and harvest certain crops are difficult to overcome. Efforts to encourage suitable habitat for grassland birds have mostly seen success on private lands growing hay and pasturelands rather than row-cropped land ([West et al. 2016](#ref-west2016), [Johnson 2017](#ref-johnson2017), [Gruntorad et al. 2021](#ref-gruntorad2021)), although some grassland species do use row crop habitats for nesting, foraging, and overwintering ([Best et al. 1997](#ref-best1997), [Johnson 2017](#ref-johnson2017)). Nutrient quality is higher in earlier-harvested hay ([Brown and Nocera 2017](#ref-brown2017)), which encourages early and more frequent harvests, but fields with delayed and reduced harvests produce more grassland bird fledglings ([Perlut et al. 2006](#ref-perlut2006)). Public lands, conversely, are subject to different motivations for management. For example, management recommendations from the Massachusetts Audubon Society regarding grassland bird conservation have been more widely adopted on lands held in public trust than on privately-owned grasslands in New England ([Atwood et al. 2017](#ref-atwood2017)). Several studies have quantified the value of protected areas such as national parks for wildlife conservation ([Palomo et al. 2014](#ref-palomo2014), [Dettling et al. 2021](#ref-dettling2021)) particularly because these areas differ in quality from the surrounding privately-owned landscape. Privately-owned grasslands in a fragmented landscape also vary in their ability to support grassland species with differing patch size needs ([Weidman and Litvaitis 2011](#ref-weidman2011)). Public lands therefore present a valuable opportunity for grassland bird conservation.

Public lands managed by the U.S. National Park Service (NPS) host a diverse array of bird species, including several species that are in decline and of conservation concern ([Ladin and Shriver 2013](#ref-ladin2013), [Dettling et al. 2021](#ref-dettling2021)). In the NPS National Capital Region, which encompasses Maryland, the District of Columbia, and portions of Virginia and West Virginia, there are several civil war battlefield parks that may provide habitat for grassland birds within the increasingly urbanized matrix in this region. These parks are maintained as open grasslands to replicate their historical appearance for use as cultural landmarks and in historical interpretation ([National Park Service 2014](#ref-nationalparkservice2014)). However, the effectiveness of these parks as habitat for grassland species in an urbanizing landscape is not known. Bird populations are often seen as indicators of the health of natural resources in the parks ([National Park Service 2005](#ref-nationalparkservice2005)), with changes in bird populations reflecting both ecological change and potential ramifications for public experience in parks. Further, park managers require long-term scientific data and sound analyses to identify the best management practices of their resources. The NPS Inventory and Monitoring Program collects ecological data to assess the condition and changes in NPS natural resources over time to support scientific resource management decision-making ([Fancy 2009](#ref-fancy2009)). The NPS Inventory and Monitoring Program has collected forest bird occurrence and abundance data in the National Capital Region since 2005 ([National Park Service 2005](#ref-nationalparkservice2005)) and more recently in 2014 began long-term population monitoring for grassland birds. While previous analyses have focused on interior forest birds in National Parks ([Ladin and Shriver 2013](#ref-ladin2013)), little insight exists into the status of grassland birds in parks of this region. Further development and testing of management strategies such as those in the conceptual ecological model of Peterjohn ([2006](#ref-peterjohn2006)) will be made possible by high-quality long-term data on grassland birds.

Determine which habitat, landscape, and management factors influence focal species occupancy in order to inform grassland bird management in National Battlefield parks. We predicted that management activities, particularly burning and harvest timing restriction, would increase occupancy for grassland species relative to the absence of management. We predicted that harvest timing restrictions would increase occupancy, and that later first harvest dates under such restrictions would further increase occupancy.

Investigate potential temporal trends in occupancy. Although grassland birds have not been monitored as long as forest birds in this region, early indicators of temporal patterns in occupancy can further inform management priorities. We predicted that, in concert with their regional trends, there would be a decline in occupancy during the study period.

# Study area

The parks included in this study were historical battlefields located within the NPS National Capital Region, which surrounds Washington, D.C. (Figure 1). These four parks, Antietam National Battlefield (hereafter Antietam), Harpers Ferry National Historical Park (Harpers Ferry), Manassas National Battlefield Park (Manassas), and Monocacy National Battlefield (Monocacy), consist primarily of open areas maintained to replicate their historical appearance for use as cultural landmarks and in historical interpretation ([National Park Service 2014](#ref-nationalparkservice2014)). Antietam has an area of 13.8 km2 (48.5% grassland and 19.1% forest). Harpers Ferry has an area of 15.7 km2 (13.7% grassland and 30.3% forest). Manassas has an area of 21.5 km2 (39.1% grassland and 37.2% forest). Monocacy has an area of 6.9 km2 (25.7% grassland and 22.8% forest). Most parks are in close proximity to an urbanized area (Figure 1), most notably Frederick, MD (Monocacy) and the greater Washington, D.C. area (Manassas).

A variety of approaches are employed to manage the land in these parks. Prescribed fire is used to manage grassland habitat in all parks except Harpers Ferry, though the extent of burned area is limited to certain sub-units within the parks and only occasionally overlapped the sites in this study. Non-forested area within these parks are managed by both the parks and private entities through agricultural lease programs. These programs maintain the agricultural history of the parks as working lands ([National Park Service 2020](#ref-nationalparkservice2020)). Lease terms are set by the parks, but the leased fields are managed by the private entities awarded the lease and are typically farmed for hay or row crops, primarily corn.

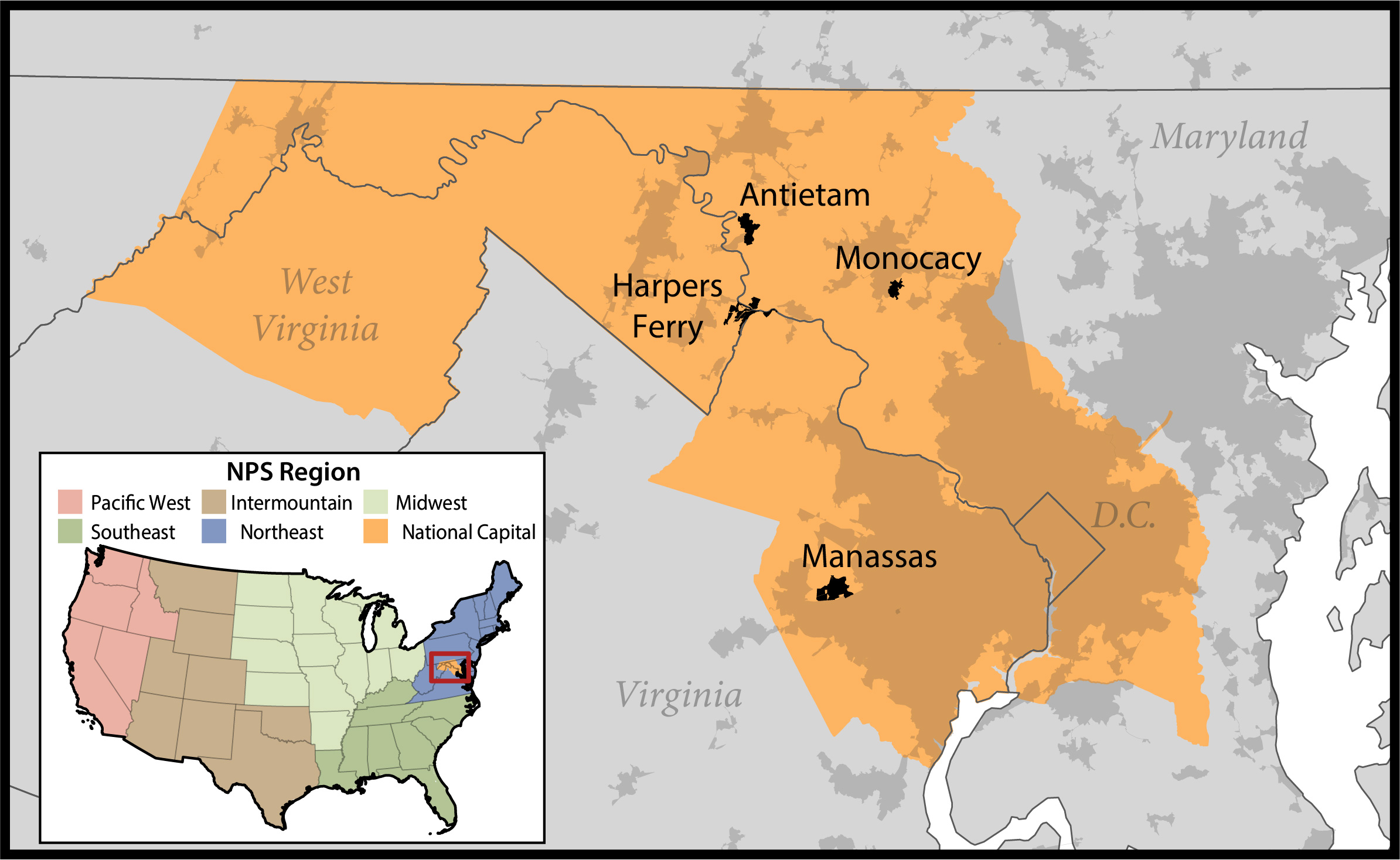


Figure 1: Map of battlefield parks in the National Park Service (NPS) National Capital Region where grassland bird populations were monitored from 2014-2021. Urbanized areas are shown in darker gray. Other NPS regions are shown in the inset map.

# Methods

## Bird surveys

The NPS National Capital Region Inventory and Monitoring program conducted point count surveys from 2014-2019 and in 2021 at Manassas, from 2015-2019 and in 2021 at Antietam and Monocacy, and from 2016-2019 and in 2021 at Harpers Ferry. A total of 242 sites were surveyed across all four parks, distributed in a spatially-balanced Generalized Random Tessellation-Stratified scheme ([Stevens and Olsen 2004](#ref-stevens2004)) in accordance with the National Capital Region avian monitoring protocol (*NCRN monitoring protocol does not seem to be published*). Sites were separated by at least 250 meters. Surveys were conducted two times per year during the breeding season between the first week of May and the last week of July. Each point count consisted of a single-observer survey divided into four 2.5-minute intervals for a total length of 10 minutes. Birds were recorded in three distance bands (0-25m, 26-50m, and 50-100m), excluding any individuals that were detected outside the maximum 100m radius.

Covariates with the potential to impact detection were recorded during each survey including date, disturbance, wind, temperature, humidity, time, and observer (Table 1). Disturbance was a subjective assessment by the observer that accounted for traffic noise consisting of four categories: 1) no disturbance, 2) disturbance with slight effect on count, 3) moderate effect on count, and 4) extreme effect on count. Wind was recorded in five categories using the Beaufort scale: 0 (0 mph, smoke rises vertically), 1 (1-3 mph, smoke drifts), 2 (4-7 mph, wind felt on face), 3 (8-12 mph, leaves in constant motion), 4 (13-18 mph, small branches sway), and 5 (19-24 mph, small trees in leaf sway). Temperature and humidity were recorded using a hygrometer. The time of each survey in minutes after local sunrise was calculated using the R package “suncalc” ([Thiermel and Elmarhraoui 2019](#ref-thiermel2019)).

Table 1: Range, mean and standard error for potential detection covariates included in candidate models for grassland bird occupancy in National Capital Region battlefield parks from 2014-2021. Sample size is reported for categorical covariates.

| **Detection covariates** | **Range** | **Mean (SE)** |
| --- | --- | --- |
| Day of year (1 Jan = Day 1) | 127 – 208 (7 May – 26 Jul) | 157 (6 Jun) |
| Disturbancea | None: 1214 |  |
|  | Slight: 1036 |  |
| Moderate: 450 |  |
| Extreme: 137 |  |
| Wind (Beaufort scale) | 0: 811 |  |
|  | 1: 944 |  |
| 2: 785 |  |
| 3: 250 |  |
| 4: 45 |  |
| 5: 3 |  |
| Temperature (C) | 4.6 – 43.5 | 21.6 (0.106) |
| Percent humidity | 7.3 – 99 | 70.7 (0.261) |
| Minutes since sunrise | -39 – 303 | 120 (1.56) |
| Observer | 24 unique |  |
| aDisturbance refers to the observer-assessed noise conditions during the survey | | |

## Focal species

To study the factors impacting grassland birds in this region, we selected two focal species for occupancy analysis, Eastern Meadowlark (*Sturnella magna*) and Grasshopper Sparrow (*Ammodramus savannarum*). Both are associated with grassland habitat during the breeding season ([North American Bird Conservation Initiative 2016](#ref-nabci2016)) and have shown significant declines in the Eastern Breeding Bird Survey region in the past 50 years ([Sauer et al. 2017](#ref-sauer2017)) and have been labeled “common birds in steep decline” by Partners in Flight ([2021](#ref-partnersinflight2021)). While broader definitions of grassland species range from grassland obligates to shrub- and scrub-breeding birds, these two species were the most abundant and widespread in the study area and provide the opportunity to compare species needs between two grassland obligates. The density and occurrence of Grasshopper Sparrow is a strong enough predictor of other upland eastern tallgrass prairie species that it has been proposed as an indicator species for the ecosystem ([Elliott and Johnson 2018](#ref-elliott2018)), an assertion we sought to examine for this ecosystem by comparing the two species’ response to management.

Grasshopper Sparrow has declined by by 5.47 percent per year in the Eastern Breeding Bird Survey region ([Sauer et al. 2017](#ref-sauer2017)). In studies across its range, the species has shown a negative response to woody shrub cover ([Johnston and Odum 1956](#ref-johnston1956), [Dechant et al. 2002](#ref-dechant2002), [Chapman et al. 2004](#ref-chapman2004), [Grant et al. 2004](#ref-grant2004)), thus, there is a need for management through mowing, hay harvest, or burning. The species is compatible with agriculture; it responds positively to managed grasslands, including those planted with non-native grasses ([Dechant et al. 2002](#ref-dechant2002)). Although Grasshopper Sparrows are known to use row crops, they do so in much lower densities than either native or non-native grasses ([Best et al. 1997](#ref-best1997)). At the landscape scale, occupancy of Grasshopper Sparrows in Delaware responded to forest, grassland, and low-intensity development land cover ([Irvin et al. 2013](#ref-irvin2013)). Timing of management matters. In New York, fields that were mowed early in the season had lower Grasshopper Sparrow densities due to nest destruction, an effect which persisted into the following year ([Bollinger 1995](#ref-bollinger1995)). Population growth rates decreased in years following high hay yields and in years with later harvest timing ([Allen et al. 2021](#ref-allen2021)). Grasshopper Sparrows move late from the breeding grounds ([Hill and Renfrew 2018](#ref-hill2018)), indicating that late-season management can affect them too. Grasshopper Sparrows show a mixed response to fire, which can increase nest productivity (Johnson and Temple 1986) and abundance in the ensuing two to three years (Forde 1984, Volkert 1992). However, after three ([Powell 2006](#ref-powell2006)) or four (Madden 1999) years, abundance decreases. Some research (Bollinger 1988) suggests that haying or mowing may be more beneficial for the species than burning.

Eastern Meadowlark has declined by 3.83 percent per year in the Eastern Breeding Bird Survey region ([Sauer et al. 2017](#ref-sauer2017)). Studies have found little change in abundance from burning (Zimmerman 1992, Heckert 1994, Zimmerman 1997, Powell 2006), though burning can reduce nest success ([Rohrbaugh et al. 1999](#ref-rohrbaugh1999)). Harvest and mowing contribute to low nest success ([Hull 2002](#ref-hull2002)). The species is somewhat more flexible in habitat needs than Grasshopper Sparrow, tolerating some woody vegetation for use as song perches (Kahl et al. 1985, Sample 1989). The species occasionally breeds in cropland ([Best et al. 1997](#ref-best1997)).

## Covariates

Site-specific habitat covariates were collected in 2021 (Table 2). The maximum angle to the horizon was collected using a clinometer. This measurement describes the visual enclosure of each site which has been shown to impact occupancy of other grassland species ([Keyel et al. 2012](#ref-keyel2012), [Marshall et al. 2020](#ref-marshall2020)). The percent area of woody shrub cover was estimated in four quadrants of the 100m-radius survey area and then averaged. Each site was classified as either hayfield (n = 76; cool-season grasses for hay production), row crop (n = 87; any non-hay crop), or meadow (n = 73; non-agricultural) by observers during the 2021 field season. All habitat information was assumed to be static during the 2014-2021 study period for lack of complete agricultural or vegetation monitoring history at each site.

Table 2: Range, mean, and standard error by park for potential occupancy covariates included in candidate models for grassland bird occupancy in National Capital Region battlefield parks from 2014-2021. Sample size is reported for categorical covariates as the number of site-years unless otherwise specified.

| **Occupancy covariates** | **Range** | **Antietam** | **Harpers Ferry** | **Manassas** | **Monocacy** |
| --- | --- | --- | --- | --- | --- |
| **Habitat** |  |  |  |  |  |
| Field type | Hayfield: 532 | 33 sites | 19 sites | 22 sites | 2 sites |
|  | Meadow: 511 | 25 sites | 8 sites | 21 sites | 19 sites |
|  | Row crop: 609 | 40 sites | 0 sites | 0 sites | 47 sites |
| Percent woody shrub cover (100m) | 0 – 80 | 4.35 (0.68) | 5.43 (1.45) | 9.47 (2.46) | 6.03 (1.06) |
| Max. angle to horizon (degrees) | 4.33 – 71.7 | 37.5 (2.31) | 36.2 (4.44) | 23 (2.66) | 35.2 (2.87) |
| **Landscape, percent cover** |  |  |  |  |  |
| Grassland (500m) | 1.38 – 96.8 | 51.3 (1.09) | 49.5 (2.35) | 51.9 (2.98) | 25 (1.48) |
| Developed (500m) | 0 – 55.6 | 3.67 (0.28) | 11.2 (1.73) | 6.51 (1.63) | 12 (1.24) |
| Forest (500m) | 0 – 71.6 | 12.9 (1.23) | 15.2 (2.56) | 24.3 (2.28) | 16.3 (1.29) |
| Wetland (500m) | 0 – 49.4 | 0.005 (0.003) | 0.69 (0.31) | 12.3 (1.97) | 4.12 (0.44) |
| Grassland (1km) | 7.39 – 68.1 | 50.3 (0.51) | 38 (1.38) | 37.7 (1.98) | 22.6 (0.79) |
| Developed (1km) | 0.34 – 59.5 | 4.33 (0.28) | 13.7 (1.29) | 10.5 (1.89) | 17.4 (1.54) |
| Forest (1km) | 1.17 – 53.4 | 14.5 (0.87) | 23.7 (2.02) | 30.9 (1.17) | 18.6 (0.88) |
| Wetland (1km) | 0 – 30.5 | 0.06 (0.02) | 0.97 (0.22) | 14.5 (1.2) | 3.96 (0.19) |
| Grassland (5km) | 14.9 – 37.3 | 35.7 (0.08) | 21.3 (0.09) | 20.7 (0.44) | 21.3 (0.19) |
| Developed (5km) | 2.79 – 36.2 | 3.69 (0.03) | 8.62 (0.20) | 27.2 (0.98) | 26.8 (0.54) |
| Forest (5km) | 18.8 – 46.0 | 34.5 (0.31) | 38.8 (0.53) | 28.9 (0.65) | 21.7 (0.19) |
| Wetland (5km) | 0.2 – 10.6 | 0.37 (0.005) | 1.06 (0.01) | 9.54 (0.12) | 1.51 (0.009) |
| **Management** |  |  |  |  |  |
| Year | 2014 – 2021 | 6 years | 5 years | 7 years | 6 years |
| Years since last burn | 0-2: 58 | 11 sites | 0 sites | 4 sites | 2 sites |
|  | 3+: 43 |  |  |  |  |
| Agricultural lease | Present: 1519 | 88 sites | 28 sites | 35 sites | 66 sites |
|  | Absent: 1296 |  |  |  |  |
| Harvest restriction | Present: 343 | 2 sites | 26 sites | 27 sites | 0 sites |
|  | Absent: 1296 |  |  |  |  |
| Day of first harvest (1 Jan = Day 1) | 182 – 227 (1 Jul – 15 Aug) | 227 (15 Aug) | 197 (16 Jul) | 182 (1 Jul) |  |
| Park |  | 100 sites | 29 sites | 44 sites | 69 sites |

We calculated landscape-level covariates using the National Land Cover Database (NLCD) ([Dewitz and U.S. Geological Survey 2021](#ref-nlcd2019)), combining cover types into four broad categories: developed (developed low intensity, developed medium intensity, developed high intensity), forest (deciduous forest, evergreen forest, and mixed forest), grassland (grassland/herbaceous, pasture/hay, shrub/scrub), and wetland (woody wetlands, emergent herbaceous wetlands) cover within 500m, 1km, and 5km buffers of each survey site (Figure 2). These distance bands were chosen to cover a gradient of spatial scales over which landscape covariates can have different impacts on grassland birds ([Guttery et al. 2017](#ref-guttery2017)), without being specific to each species. These scales also represent a gradient of control by park managers, with the land within 500m being under park management, but not at the 5km scale. Burn history within the parks was obtained from the NPS Wildland Fire feature server. A site was counted as burned in a given year if any part of the annual fire perimeter polygon overlapped the 100m survey radius; only prescribed burns and not wildfire intersected with survey sites. We calculated years since burn for each site-year and grouped burn history into 0-2 years and 3 or more years since last burn because of sample size limitations and because 3 years is an approximate threshold for Grasshopper Sparrow ([Powell 2006](#ref-powell2006)). We obtained information from park managers on the agricultural lease status of all sites, along with information about the timing and year of implementation of harvest timing restrictions. There were no harvest timing restrictions set at Monocacy during the survey period, but they were instituted or maintained at other parks.

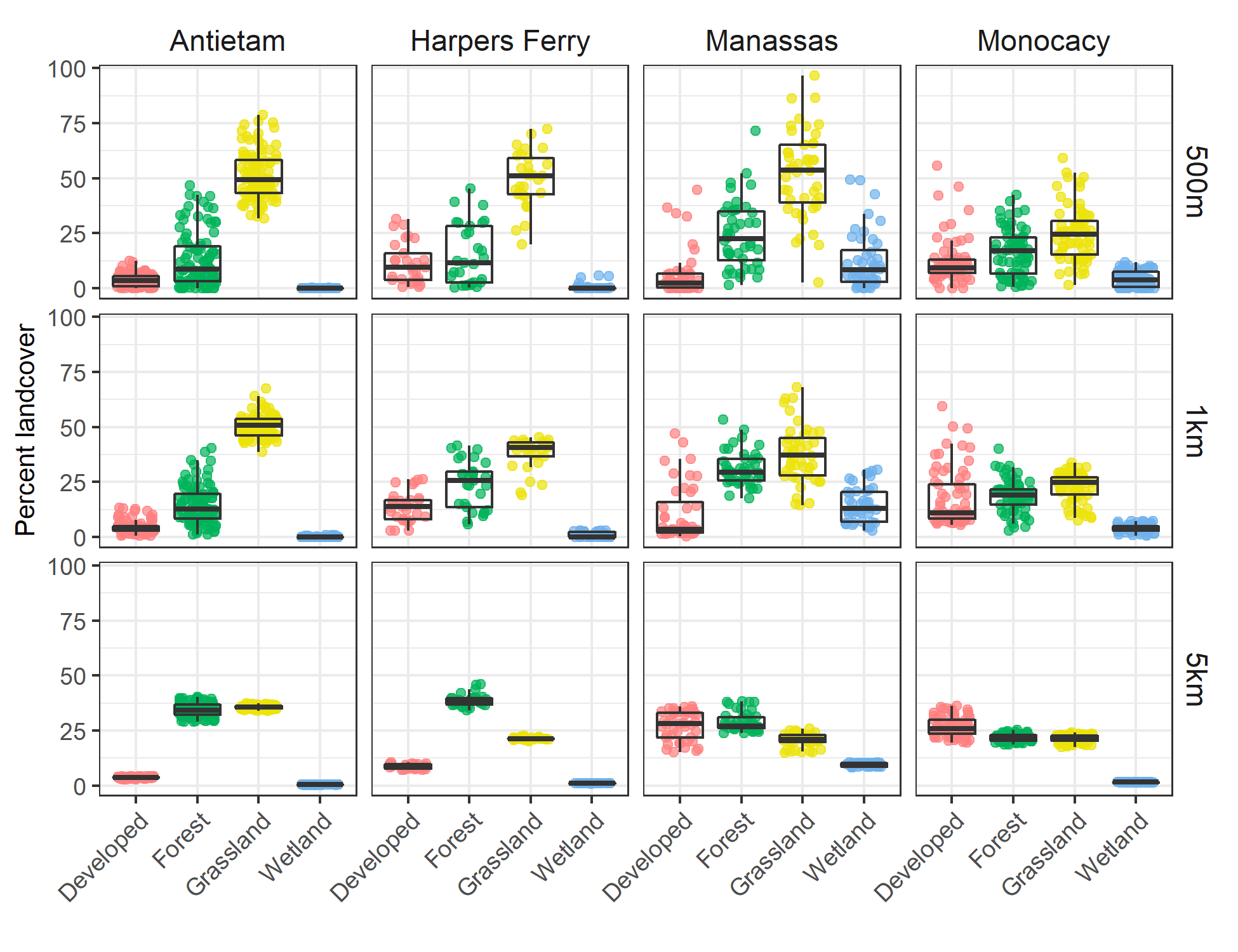


Figure 2: Mean percent cover of four landcover types surrounding surveyed sites in National Capital Region battlefield parks at three spatial scales.

## Analysis

Analysis was performed using program R version 4.1.2 ([R Core Team 2021](#ref-rcoreteam2021)). We scaled all continuous covariates by centering on the mean and dividing by their standard deviations. We modeled the impact of habitat, landscape, and management on grassland species using the “occu” function from the R package “unmarked” ([Fiske and Chandler 2011](#ref-fiske2011)). We used a single-season occupancy model ([MacKenzie et al. 2002](#ref-mackenzie2002)) with a stacked approach, treating each site-year combination as a site and including year as a site covariate. This approach is commonly used when data is sparse because it increases the effective sample size at the cost of some pseudoreplication that leads to underestimates of model error ([McClure and Hill 2012](#ref-mcclure2012), [Fogg et al. 2014](#ref-fogg2014)). Using a stacked model, the total number of sites became n = 1694 site-year combinations, rather than a maximum of 242 sites per year.

We used a hierarchical approach for model selection for each species. First, we compared detection and occupancy models using Akaike’s Information Criterion for small samples (AICC), where models were ranked on ΔAICC with the lowest value being the best model ([Hurvich and Tsai 1989](#ref-aicc1989)). Null models were also included in each model comparison, including a detection-only model and an intercept-only model. In cases of multiple models with ΔAICC < 2, we only used the top-ranked model. The top-ranked detection model for each species was used in all of its subsequent occupancy models. To investigate the effect of different types of site covariates, we started by running separate models for the different covariate categories of habitat, landscape, and management. We did not include covariates with a Spearman’s rank correlation of r > 0.6 in the same model, ensuring no strong correlation between covariates (Figure 3). In cases where two related potential covariates were both supported but were highly correlated, such as in the case of mean and maximum angle to the horizon, or number of trees and angle to the horizon, we used only the covariate with the lower AIC value when tested singularly. The top-performing covariates in each category were then used to generate models that included or excluded each category (e.g. landscape and management, habitat and landscape, or all three) to assess their combined ability to predict occupancy and to determine the relative impacts of these covariate categories. We modeled several additional management covariates separately because they could only be applied to a subset of sites. These management models included the number of years since the last burn, presence of a harvest timing restriction (a subset of sites under agricultural lease), and, if present, the first allowed harvest date under the harvest timing restriction. These models were compared only to null models as opposed to the fully specified models for each species. We assessed potential trends in occupancy by including year as a potential covariate as either a numerical variable to test for overall trends, or as a categorical variable to test for interannual variation in occupancy. We assessed model fit using a parametric bootstrap approach ([Kéry and Royle 2016](#ref-kery2016)). Model predictions were generated using the top model for each species with all non-target covariates set to the median value. We report mean (± SE) throughout for all model predictions unless otherwise stated. *When I run my goodness-of-fit tests, which are still a work in progress, I will instead report mean and boostrapped confidence interval for beta estimates.*

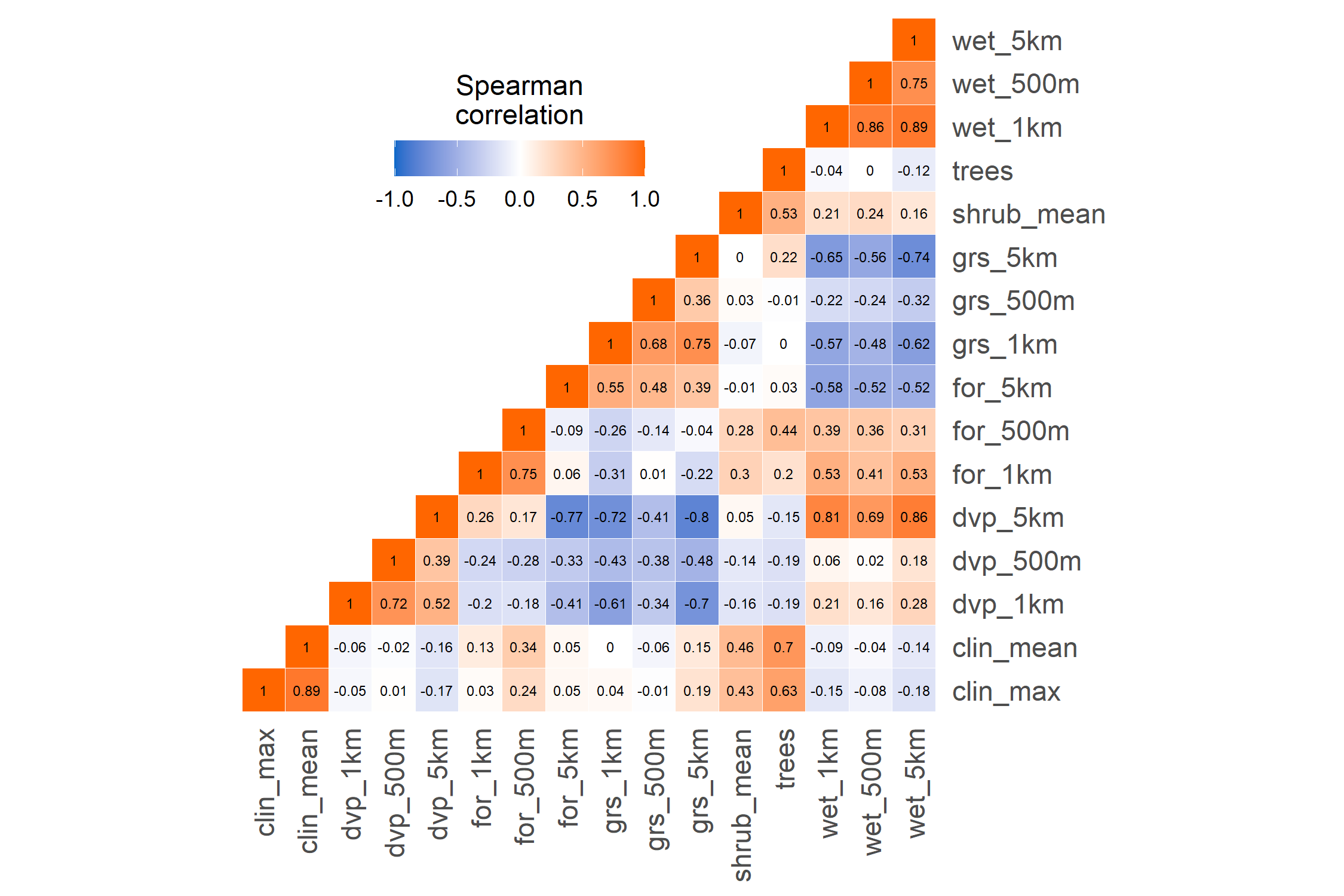


Figure 3: Correlation matrix of candidate variables for grassland bird occupancy modeling in National Capital Region battlefield parks in 2014-2021. When two variables were highly correlated (r > 0.6), they were not used in the same model. This is destined to become a supplemental figure.

# Results

A total of 3988 surveys across 7 years detected 128 total bird species, including 11 species designated as Common Birds in Steep Decline by Partners in Flight (PIF) and 7 species facing steep declines and major threats (designated PIF Yellow Watch List “D”, Table 3). Several of these vulnerable species are associated with grassland and agricultural habitat but were not included in analysis due to a low number of detections across parks and years. For example, though Grasshopper Sparrow was detected on 43.2% of all surveys and Eastern Meadowlark on 23.3%, Northern Bobwhite was only detected on 0.30% of surveys, Horned Lark on 4.14%, and Bobolink on 0.32%. The total number of grassland and agricultural breeding species detected was 17 at Antietam, 15 at Harpers Ferry, 18 at Manassas, and 18 at Monocacy.

Table 3: Detection by park of grassland- and agricultural-breeding species (20 out of 128 total species) detected in National Capital Region battlefield parks surveyed from 2014-2021.

| **Common name** | **Scientific name** | **Antietam** | **Harpers Ferry** | **Manassas** | **Monocacy** |
| --- | --- | --- | --- | --- | --- |
| Northern Bobwhitea | *Colinus virginianus* |  |  | X |  |
| Mourning Dove | *Zenaida macroura* | X | X | X | X |
| Killdeer | *Charadrius vociferus* | X | X | X | X |
| Northern Harrier | *Circus hudsonius* |  |  |  | X |
| American Kestrel | *Falco sparverius* | X | X | X | X |
| Eastern Kingbird | *Tyrannus tyrannus* | X | X | X | X |
| American Crow | *Corvus brachyrhynchos* | X | X | X | X |
| Common Raven | *Corvus corax* | X | X | X | X |
| Horned Larka | *Eremophila alpestris* | X | X |  | X |
| Barn Swallow | *Hirundo rustica* | X | X | X | X |
| Grasshopper Sparrowa | *Ammodramus savannarum* | X | X | X | X |
| Field Sparrowa | *Spizella pusilla* | X | X | X | X |
| Vesper Sparrow | *Pooecetes gramineus* | X | X | X | X |
| Savannah Sparrow | *Passerculus sandwichensis* | X |  | X | X |
| Bobolinkb | *Dolichonyx oryzivorus* | X |  | X | X |
| Eastern Meadowlarka | *Sturnella magna* | X | X | X | X |
| Red-winged Blackbird | *Agelaius phoeniceus* | X | X | X | X |
| Brown-headed Cowbird | *Molothrus ater* | X | X | X | X |
| Common Gracklea | *Quiscalus quiscula* | X | X | X | X |
| Dickcissel | *Spiza americana* |  |  | X |  |
| aPartners in Flight (PIF) Common Birds in Steep Decline | | | | | |
| bPIF Yellow Watch List “D” (Steep declines and major threats) | | | | | |

Both focal species were detected annually in each park. Occupancy varied strongly by park for Eastern Meadowlark (Figure 4), and while it was included in the top model for Grasshopper Sparrow, it was not strongly variable.

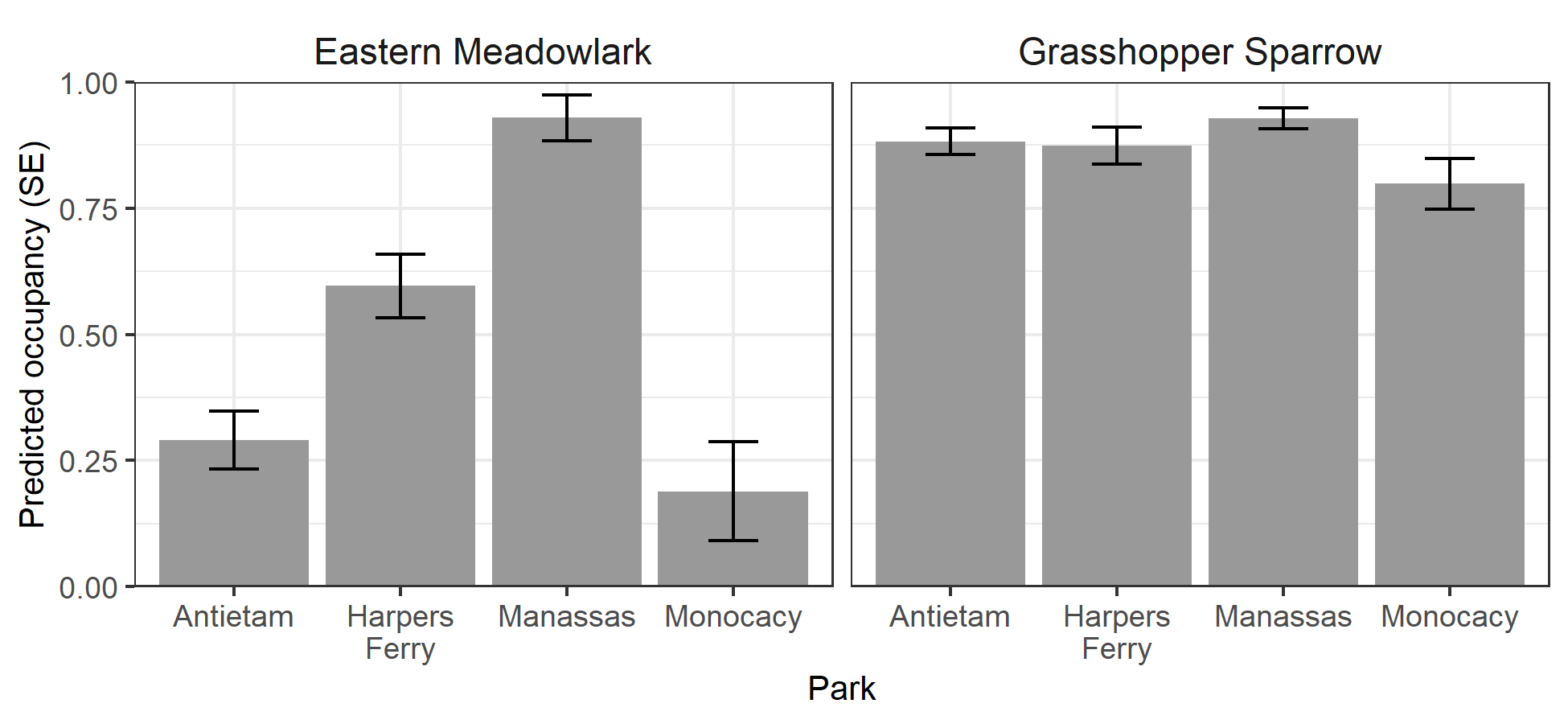


Figure 4: Predicted occupancy by park for focal bird species monitored in National Capital Region battlefield parks from 2014-2021.

The covariates that impacted detection probability (p) varied among species. Detection probability varied as a function of wind speed for Grasshopper Sparrow and disturbance due to traffic noise for Eastern Meadowlark (Table 4). Grasshopper Sparrow occupancy was also well-explained by other covariates. Mean probability of detection was similar for both species at approximately p = 0.7.

Table 4: Summary of top models affecting probability of detection for for four grassland bird species surveyed in National Capital Region battlefield parks from 2014-2021. We report the relative difference in Akaike’s Information Criterion compared to the top-ranked model for the species (Delta AICc), the number of parameters in the model (K), and AICc weights for all models within 2 AICc.

| **Species** | **Detection model** | **Delta AICc** | **Weight** | **K** |
| --- | --- | --- | --- | --- |
| Eastern Meadowlark | disturbance | 0.00 | 0.894 | 5 |
| Grasshopper Sparrow | wind | 0.00 | 0.176 | 7 |
| disturbance | 0.61 | 0.129 | 5 |
| wind + day of year | 1.44 | 0.086 | 8 |
| day of year | 1.59 | 0.079 | 3 |
| wind + temperature | 1.72 | 0.074 | 8 |
| wind + time | 1.73 | 0.074 | 8 |
| temperature | 1.97 | 0.066 | 3 |
| time | 2.00 | 0.065 | 3 |

Habitat, landscape, and management covariates (see Table 2) all affected occupancy, while their overall influence and specific covariates differed by species (Table 5).

Table 5: Summary of top occupancy models for four grassland bird species surveyed in National Capital Region battlefield parks from 2014-2021. We report the relative difference in Akaike’s Information Criterion compared to the top-ranked model for the species (Delta AICc), and the number of parameters in the model (K) for all models within 2 AICc.

| **Species** | **Occupancy model** | **Delta AICc** | **Weight** | **K** |
| --- | --- | --- | --- | --- |
| Eastern Meadowlark | field type + angle + grassland (500m) + developed (5km) + park + leased | 0.00 | 0.441 | 14 |
| field type + angle + grassland (500m) + developed (5km) + park + leased + year | 0.43 | 0.355 | 15 |
| field type + angle + grassland (500m) + developed (5km) + park + leased + year (factor) | 1.61 | 0.197 | 20 |
| Grasshopper Sparrow | field type + angle + shrub + forest (500m) + grassland (500m) + developed (500m) + park + leased + year | 0.00 | 0.741 | 19 |

### Habitat

Habitat covariates were included in top models for both species. Field type was included in the top model for all species, with hayfields having higher predicted occupancy than row crop for both species (Figure 5). For Eastern Meadowlark, predicted occupancy in hayfields was 0.29 (± 0.06) while row crop was only 0.24 (± 0.05). For Grasshopper Sparrow, predicted occupancy in hayfields was 0.88 (± 0.03) while row crop was only 0.73 (± 0.04). Occupancy in meadow, a catch-all group for non-agricultural habitat, was intermediate to other habitat types for Grasshopper Sparrow (0.79 ± 0.05) and equal to hayfield for Eastern Meadowlark.

The maximum angle to the horizon had a negative impact on occupancy of both species (Figure 5). Woody shrub cover within 100m had a strong negative impact on the occupancy of Grasshopper Sparrow but was not included in the model for Eastern Meadowlark (Figure 5).

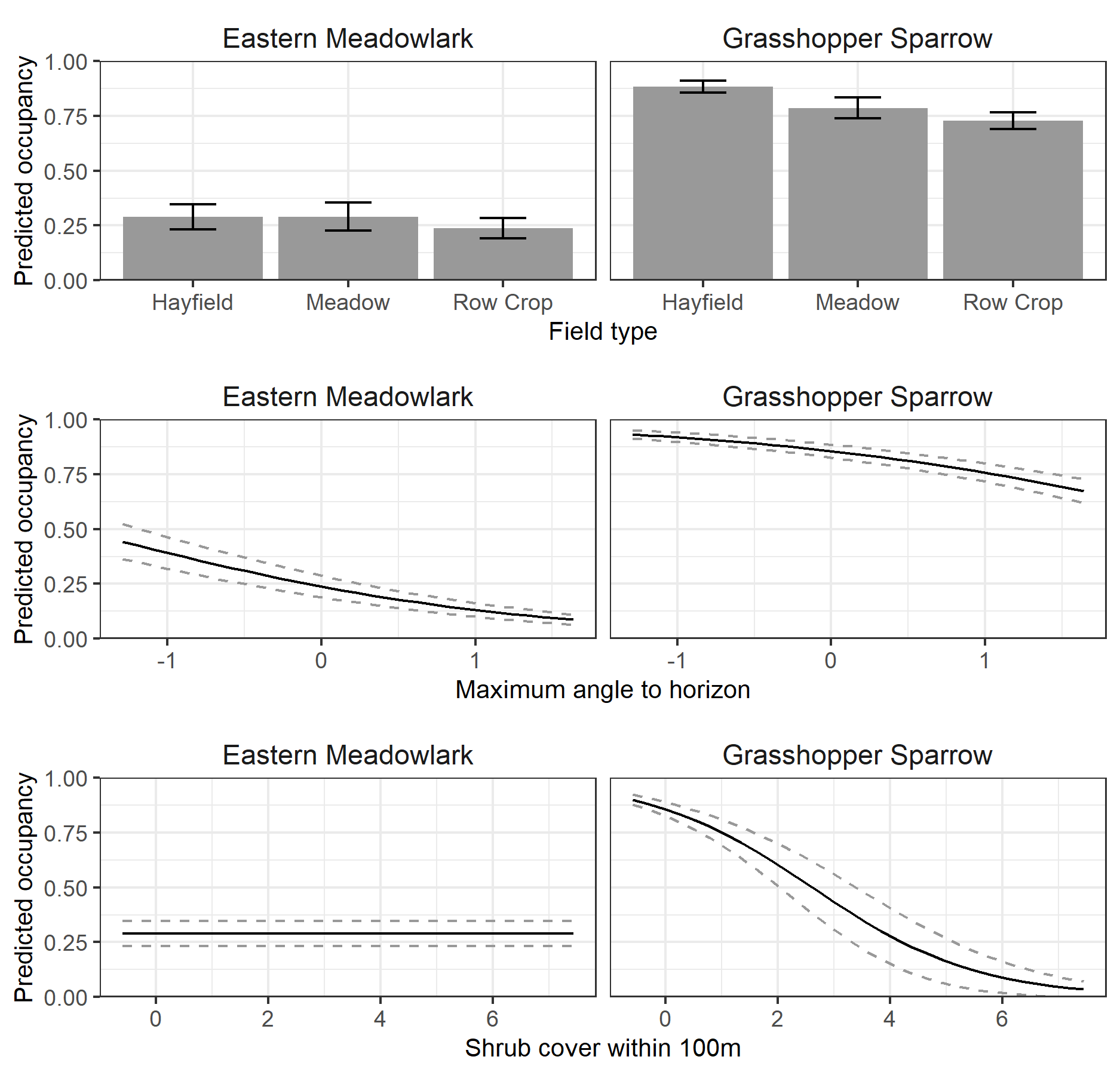


Figure 5: Predicted occupancy by habitat covariates of field type, angle to horizon, and shrub cover for focal bird species monitored in National Capital Region battlefield parks from 2014-2021.

### Landscape

Eastern Meadowlark and Grasshopper Sparrow responded to landscape variables at different spatial scales. Forest, grassland, and development were included in top-performing models but wetland was not. Eastern Meadowlark responded to landscape variables at mixed spatial scales. Increased grassland within 500m had a positive impact on meadowlark occupancy, while development at a 5km scale had a slight negative impact (Figure 6). Grasshopper Sparrow occupancy was positively impacted by grassland within 500m and negatively impacted by development and forest within 500m (Figure 6).

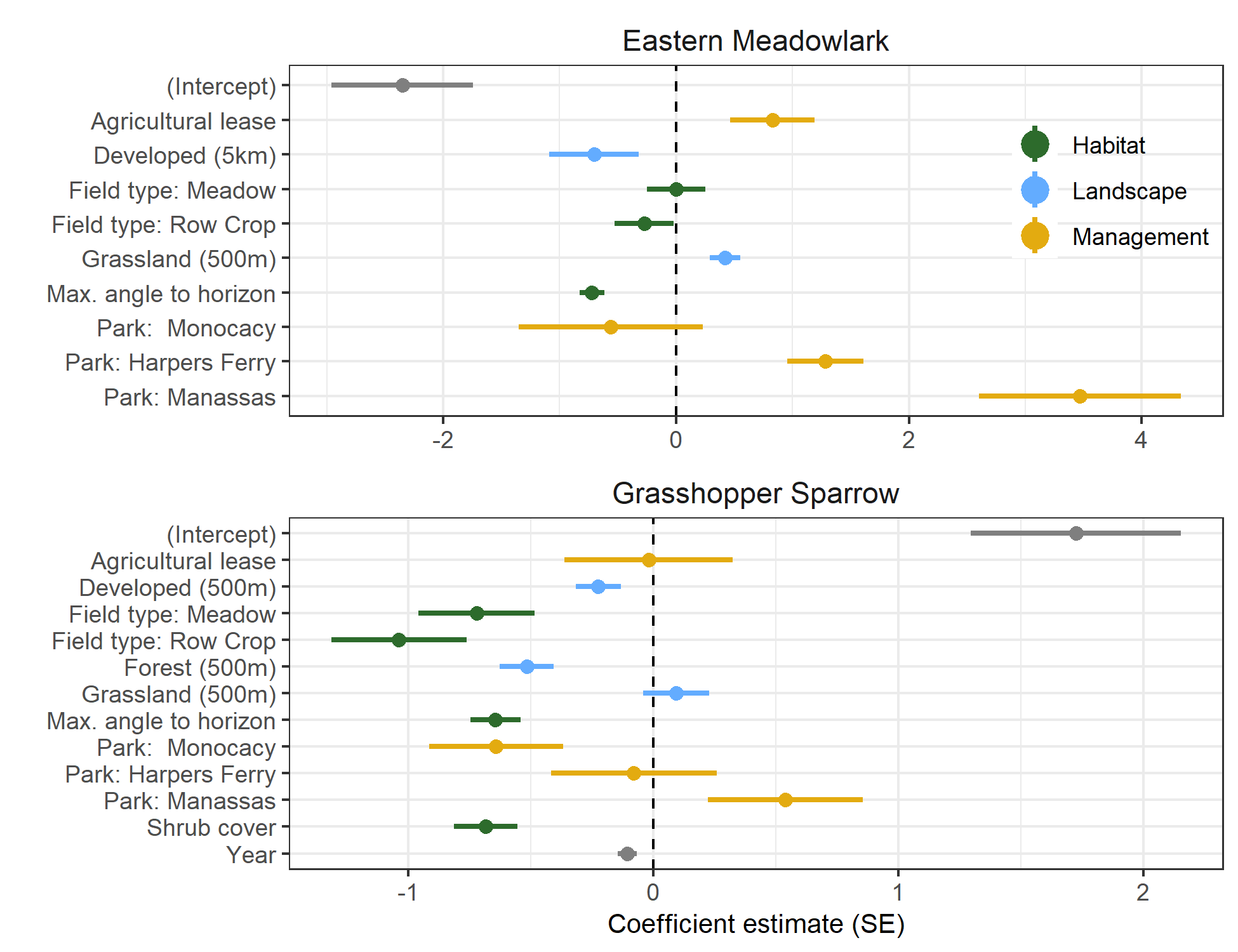


Figure 6: Coefficient estimates of habitat, landscape, and management covariates included in top occupancy models for grassland birds surveyed in National Capital Region battlefield parks from 2014-2019.

### Management

Aspects of prescribed burning and agricultural leasing increased occupancy. We modeled agricultural lease (217 total sites leased and 25 not leased), prescribed burns (17 sites burned, 225 never burned), and overall park-level impact on occupancy. Presence of an agricultural lease had a positive impact on occupancy of Eastern Meadowlark and no strong impact on Grasshopper Sparrow (Figure 7). A subset of 55 of the 217 leased sites were subject to harvest timing restrictions. In both species, a harvest timing restriction increased occupancy (Figure 7), with occupancy at sites with harvest timing restrictions being over twice as high as those without for Eastern Meadowlark (0.69 ± 0.03 vs 0.24 ± 0.02). The date of first allowed harvest among those restricted sites was a significant predictor of occupancy for Eastern Meadowlark; however, there was little variation in the dates since the timing restrictions were set at the park level, and the strong variation of Eastern Meadowlark among parks was a better single predictor of occupancy (Figure 4).

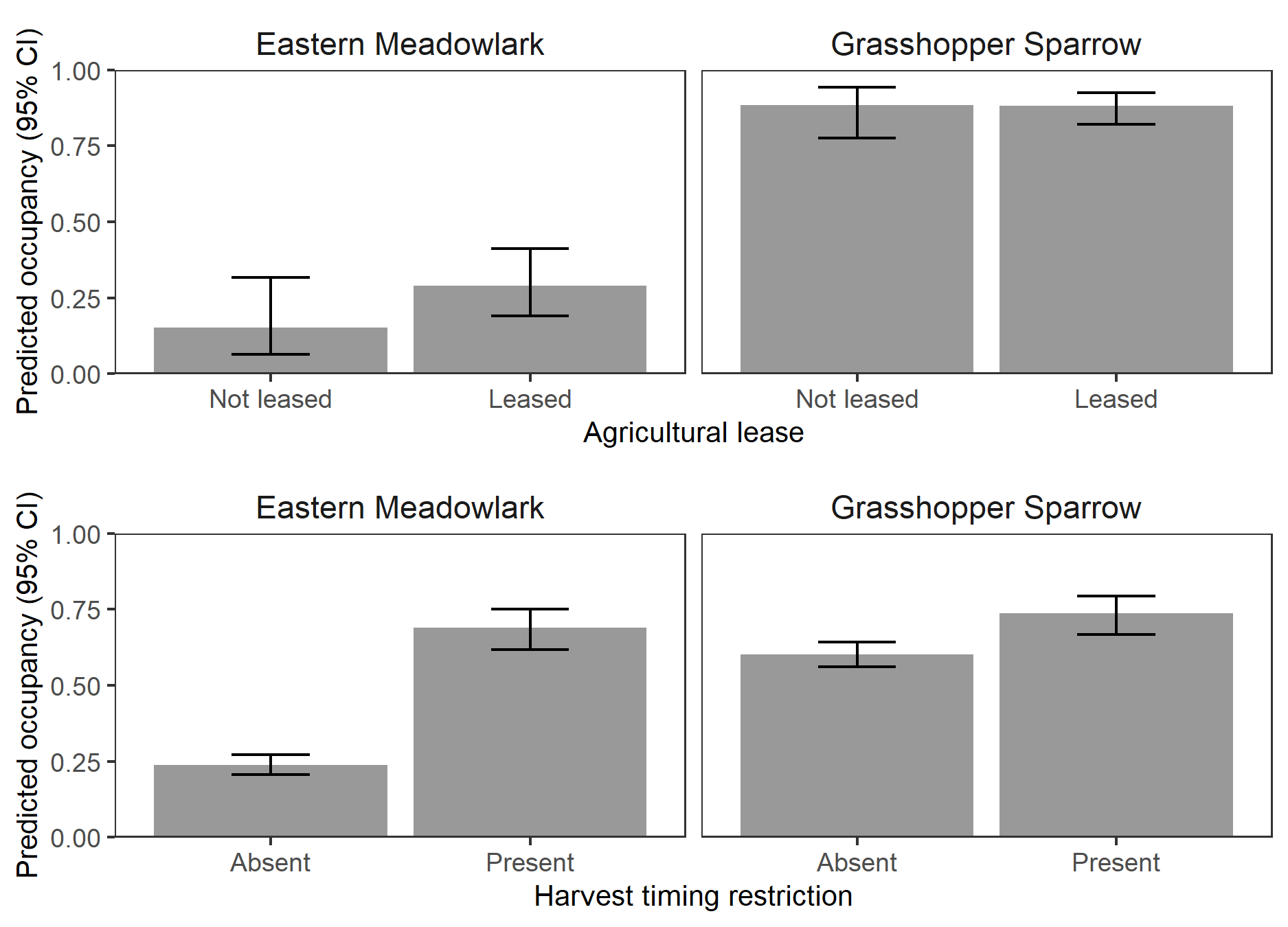


Figure 7: Predicted occupancy by agricultural lease status for focal bird species monitored in National Capital Region battlefield parks from 2014-2021.

A total of 17 sites were had a history of prescribed burns, making for a total of 101 site-years. Neither species had a significant response to the presence of prescribed burns at the site. However, within the subset of sites that were burned, there was a response to the burn interval. During the 8-year time span of the study, 58 site-years were surveyed within 0-2 years of last burn and 43 were surveyed when 3 or more years had elapsed since the last burn. Grasshopper Sparrow had a significant response to burn interval, with sites burned in the past 2 years having higher occupancy than those burned 3 or more years ago (Figure 8). Eastern Meadowlark occupancy was non-significantly lower when it had been 3 or more years since the last burn.

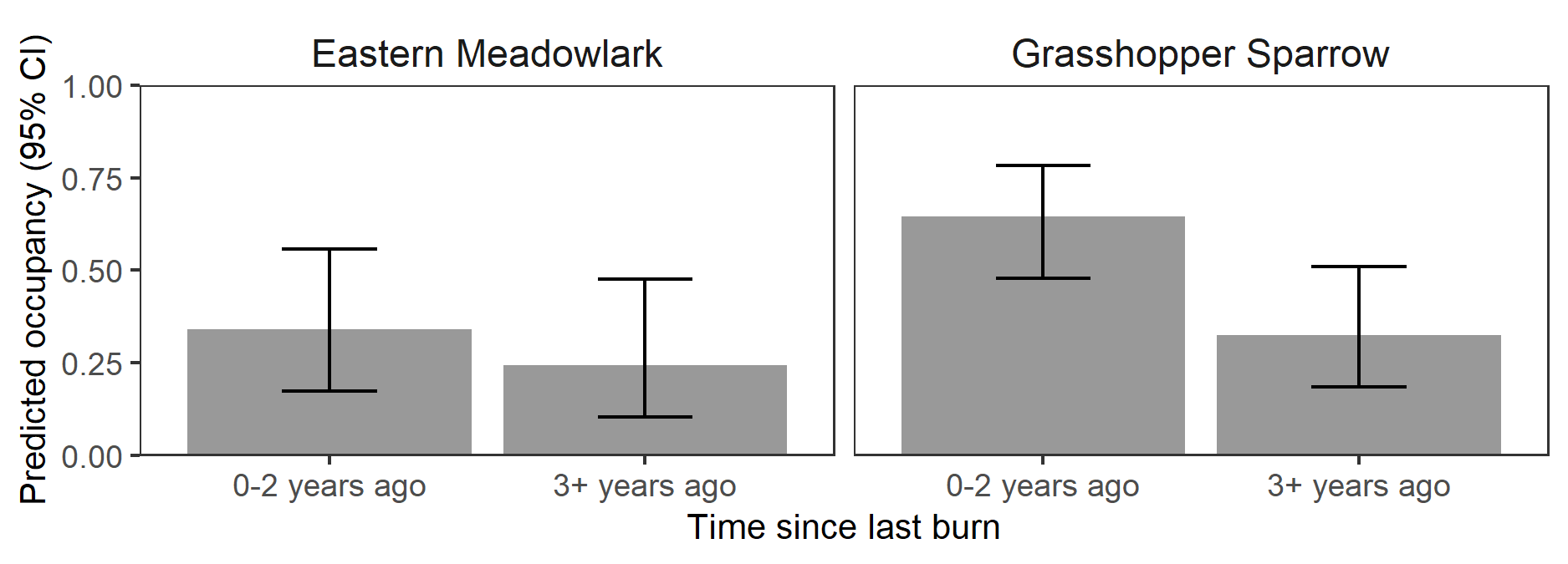


Figure 8: Predicted occupancy by time since the site was last burned (burn inverval) for focal bird species monitored in National Capital Region battlefield parks from 2014-2021.

### Trends

We investigated long-term patterns in occupancy by including additional models with year as a numerical covariate to account for temporal trends or year as a factor to account for interannual variation. The only species that showed a significant annual trend was Grasshopper Sparrow, with year having a coefficient of -0.107 (± 0.04) when included (Figure 6). Eastern Meadowlark had neither an annual or interannual variation in its best model, but each was included in competitive models (Table 5).

# Discussion

The National Battlefield Parks were host to numerous grassland and agricultural-breeding species of concern, indicating that these parks contain priority targets for conservation. Grasshopper Sparrow had high occupancy across parks but showed a slight but significant decline in occupancy even over a relatively short (<10 year) study period. Examination of trends in abundance may or may not reflect the same declines.

Several covariates were significant predictors of occupancy when modeled alone, and their inclusion in the top model decreased AICc, but they became non-significant as predictors in that top model.

Habitat features were important predictors of occupancy and were largely in line with previous research. Grassland bird occupancy decreased as angle to the horizon increased, indicating that they respond to visual openness. Woody shrub cover had a negative impact on Grasshopper Sparrow but not Eastern Meadowlark, which aligns with previous research. Eastern Meadowlarks use of song perches and known tolerance for more woody vegetation aligns with higher occupancy, but their nesting success, which can be impacted by shrub content ([Hull 2002](#ref-hull2002)), was not monitored. Occupancy of both species was negatively impacted by row crops, which are concentrated in Monocacy and Antietam. Habitat is the result of management activities, and these results suggest that changing what types of fields are planted could increase or decrease occupancy of these species. Hayfields were suitable for both species, however, it is worth noting that the classification of field types was done by observers in 2021 and may not reflect the true state of the site ever year; in particular, agricultural leases have historically been single-year ([National Park Service 2020](#ref-nationalparkservice2020)) and documentation of the precise management and planting done by lessees is lacking. Other covariates such as woody shrub cover were also measured in a single year but could have varied during the study period.

The response of both species to some features of habitat at the 500m scale is promising in that it suggests park management can impact these species. However, Eastern Meadowlark responded at two different spatial scales, and the negative impact of urbanization at the 5km scale is not within the parks’ control. Even in an urbanized landscape, if species respond to highly local conditions then isolated parks can still be valuable habitat. Grassland birds excel at colonizing patches in the east, possibly related to the fact these patches have always been ephemeral in a historically forested landscape ([Askins 1999](#ref-askins1999)). Management at the park level is always important: each park has a different baseline for what can be expected. They also control management such as harvest timing limits, which have expanded over the course of the study. Local management activities can have benefits for grassland birds using parks even as habitat in wider landscape is lost.

Whether or not a point was ever managed with prescribed fire did not impact occupancy, but the time since last burn was significant for Grasshopper Sparrow. This contrast in results is likely because there were only a few locations that were ever burned, and other forms of disturbance such as mowing and hay harvest were used elsewhere.

Agricultural leases are generally good for Eastern Meadowlark, but kind of inconclusive for GRSP. As long as they aren’t growing row crops. Even though leases consist of private entities farming on public lands, which might seem contradictory to the advantages of public lands not being used for the goal of agricultural production, the parks have the power to set the lease terms to be friendly to grassland birds and can institute harvest timing restrictions. Harvest timing restriction was important for both species, and it appears that this effort is working where it is present. Although harvest timing restrictions typically mean fewer hay cuttings per year, lessees understand that maximizing yield is not the primary goal when operating on the cultural landscape of a National Park (A. S. Lee, National Park Service, personal communication). Future years may see these restrictions expanded to parks where they are absent.

Grasshopper Sparrow did not have the same reactions to all covariates as Eastern Meadowlark, and Eastern Meadowlark tended to have more of a response to agricultural leasing. There are other species of concern we did not model, but we have shown that the two most abundant grassland bird species have similar but not identical controls on their occupancy.

To support management, there is a need for additional data comparable to the long-term vegetation monitoring associated with NPS forest bird monitoring. More local, and annually-measured, habitat variables such as coverage of bare ground, litter depth, and vegetation height ([Fisher and Davis 2010](#ref-fisher2010)) are important for these species and could not be measured as part of this study. Additionally, collecting and centralizing information about management activities across parks could help target management for grassland birds across the region. In the densely-populated and developed region surrounding the nation’s capital, public lands serve as valuable and permanent habitat for grassland birds when they are managed to the species’ benefit.

# Management implications

*“The management implications section should be short (usually one paragraph) and direct but explain issues important to management, conservation, or advancing wildlife science that are derived directly from or addressed in your results. Do not offer recommendations that are beyond the scope of your study. Address specific research, conservation, or management opportunities or problems in this section.”* – JWM style guide

Leasing public lands to private entities in agricultural leases can benefit Eastern Meadowlark and does not harm Grasshopper Sparrow, particularly when these leases grow hay under harvest timing guidelines meant to conserve grassland birds. Since the maximum angle to the horizon negatively impacted occupancy of grassland species, managers can increase grassland bird occupancy by removing large trees and tree rows overhanging fields. This management activity is compatible with park goals of restoring historic viewsheds as well ([Peterjohn 2006](#ref-peterjohn2006), [National Park Service 2020](#ref-nationalparkservice2020)), although there are potential tradeoffs with any forest or edge species that also may be of management concern.

# Acknowledgments

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# Ethics statement

*“The ETHICS STATEMENT section appears below acknowledgments and should explicitly state that the study adhered to relevant regulations and guidelines regarding the ethics of animal welfare and include protocol numbers parenthetically.”* – JWM style guide; new requirement

# References

Allen, M. C., J. L. Lockwood, and J. Burger. 2021. [Finding clarity in ecological outcomes using empirical integrated socialecological systems: A case study of agriculture-dependent grassland birds](https://doi.org/10.1111/1365-2664.13776). Journal of Applied Ecology 58:528–538.

Allen, M., J. Burger, and J. Lockwood. 2019. [Evaluation of unharvested refugia for grassland bird conservation within active hayfields](https://doi.org/10.5751/ACE-01457-140215). Avian Conservation and Ecology 14.

Askins, R. A. 1999. History of grassland birds in eastern North America. Studies in Avian Biology 19:60–71.

Atwood, J., J. Collins, L. Kidd, M. Servison, and J. Walsh. 2017. Best management practices for nesting grassland birds. Mass Audubon, Lincoln, MA.

Best, L. B., H. Campa, K. E. Kemp, R. J. Robel, M. R. Ryan, J. A. Savidge, H. P. Weeks, and S. R. Winterstein. 1997. Bird abundance and nesting in CRP fields and cropland in the Midwest: A regional approach. Wildlife Society Bulletin 25:864–877.

Bollinger, E. K. 1995. [Successional Changes and Habitat Selection in Hayfield Bird Communities](https://doi.org/10.1093/auk/112.3.720). The Auk 112:720–730.

Bollinger, E. K., P. B. Bollinger, and T. A. Gavin. 1990. Effects of hay-cropping on eastern populations of the bobolink. Wildlife Society Bulletin 18:142–150.

Brennan, L. A., and W. P. Kuvlesky. 2005. [North American grassland birds: An unfolding conservation crisis?](https://doi.org/10.2193/0022-541X) Journal of Wildlife Management 69:1–13.

Brown, L. J., and J. J. Nocera. 2017. [Conservation of breeding grassland birds requires local management strategies when hay maturation and nutritional quality differ among regions](https://doi.org/10.1016/j.agee.2016.11.004). Agriculture, Ecosystems & Environment 237:242–249.

Chapman, R. N., D. M. Engle, R. E. Masters, and D. M. Leslie. 2004. [Tree invasion constrains the influence of herbaceous structure in grassland bird habitats](https://doi.org/10.1080/11956860.2004.11682809). Ecoscience 11:55–63.

Dechant, J. A., M. L. Sondreal, D. H. Johnson, L. D. Igl, C. M. Goldade, M. P. Nenneman, and B. R. Euliss. 2002. [Effects of management practices on grassland birds: Grasshopper Sparrow](https://doi.org/10.3133/93877). USGS Northern Prairie Wildlife Research Center, Jamestown, ND, USA.

Dettling, M. D., K. E. Dybala, D. L. Humple, and T. Gardali. 2021. [Protected areas safeguard landbird populations in central coastal California: Evidence from long-term population trends](https://doi.org/10.1093/ORNITHAPP/DUAB035). Ornithological Applications 123:1–12.

Dewitz, J., and U.S. Geological Survey. 2021. [National Land Cover Database (NLCD) 2019 Products (ver. 2.0, June 2021)](https://doi.org/10.5066/P9KZCM54). U.S. Geological Survey data release.

Elliott, L. H., and D. H. Johnson. 2018. [The grasshopper sparrow as an indicator species in tallgrass prairies](https://doi.org/10.1002/JWMG.21447). The Journal of Wildlife Management 82:1074–1081.

Fancy, S. 2009. Program Brief: Inventory and Monitoring Program. National Park Service Natural Resource Program Center, Inventory and Monitoring Division.

Fisher, R. J., and S. K. Davis. 2010. [From Wiens to Robel: A review of grassland-bird habitat selection](https://doi.org/10.2193/2009-020). The Journal of Wildlife Management 74:265–273.

Fiske, I., and R. Chandler. 2011. Unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of Statistical Software 43:1–23.

Fogg, A. M., L. J. Roberts, R. D. Burnett, A. M. Fogg, L. J. Roberts, and R. D. Burnett. 2014. [Occurrence patterns of black-backed woodpeckers in green forest of the Sierra Nevada Mountains, California, USA.](https://doi.org/10.5751/ACE-00671-090203) Avian Conservation and Ecology 9.

Grant, T. A., E. Madden, and G. B. Berkey. 2004. [Tree and shrub invasion in northern mixed-grass prairie: Implications for breeding grassland birds](https://doi.org/10.2193/0091-7648(2004)032%5b0807:TASIIN%5d2.0.CO;2). Wildlife Society Bulletin 32:807–818.

Gruntorad, M., K. Graham, N. Arcilla, and C. Chizinski. 2021. Is hay for the birds? Investigating landowner willingness to time hay harvests for grassland bird conservation. Animals 11.

Guttery, M. R., C. A. Ribic, D. W. Sample, A. Paulios, C. Trosen, J. Dadisman, D. Schneider, and J. A. Horton. 2017. [Scale-specific habitat relationships influence patch occupancy: Defining neighborhoods to optimize the effectiveness of landscape-scale grassland bird conservation](https://doi.org/10.1007/S10980-016-0462-Y/FIGURES/5). Landscape Ecology 32:515–529.

Hill, J. M., and D. R. Diefenbach. 2013. [Experimental Removal of Woody Vegetation Does not Increase Nesting Success or Fledgling Production in Two Grassland Sparrows (Ammodramus) in Pennsylvania](https://doi.org/10.1525/auk.2013.12240). The Auk 130:764–773.

Hill, J. M., J. F. Egan, G. E. Stauffer, and D. R. Diefenbach. 2014. [Habitat availability is a more plausible explanation than insecticide acute toxicity for U.S. Grassland bird species declines](https://doi.org/10.1371/journal.pone.0098064). R. M. Brigham, editor. PLoS ONE 9:e98064.

Hill, J. M., and R. B. Renfrew. 2018. [Migratory patterns and connectivity of two North American grassland bird species](https://doi.org/10.1002/ECE3.4795). Ecology and Evolution 9:680–692.

Hull, S. 2002. Effects of management practices on grassland birds: Eastern meadowlark. USGS Northern Prairie Wildlife Research Center, Jamestown, ND, USA.

Hurvich, C. M., and C.-L. Tsai. 1989. [Regression and time series model selection in small samples](https://doi.org/10.1093/biomet/76.2.297). Biometrika 76:297–307.

Irvin, E., K. R. Duren, J. J. Buler, W. Jones, A. T. Gonzon, and C. K. Williams. 2013. [A multi-scale occupancy model for the grasshopper sparrow in the Mid-Atlantic](https://doi.org/10.1002/JWMG.609). The Journal of Wildlife Management 77:1564–1571.

Johnson, A. E. M. 2017. Conservation and land management practices and their impact on sustaining breeding and non-breeding grassland bird populations in the southeast. PhD thesis, George Mason University.

Johnston, D. W., and E. P. Odum. 1956. [Breeding Bird Populations in Relation to Plant Succession on the Piedmont of Georgia](https://doi.org/10.2307/1929668). Ecology 37:50–62.

Kéry, M., and A. Royle. 2016. Applied hierarchical modeling in ecology. Elsevier.

Keyel, A. C., C. M. Bauer, C. R. Lattin, L. M. Romero, and J. M. Reed. 2012. [Testing the role of patch openness as a causal mechanism for apparent area sensitivity in a grassland specialist](https://doi.org/10.1007/S00442-011-2213-8/FIGURES/4). Oecologia 169:407–418.

Ladin, Z. S., and G. W. Shriver. 2013. Forest bird monitoring in the National Capital Region Network: Summary report 2007-2011. Natural {{Resource Technical Report}}, National Park Service, Fort Collins, CO.

MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, A. J. Royle, and C. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248–2255.

Marshall, H., E. J. Blomberg, V. Watson, M. Conway, J. B. Cohen, M. D. Correll, C. S. Elphick, T. P. Hodgman, A. R. Kocek, A. I. Kovach, W. G. Shriver, W. A. Wiest, and B. J. Olsen. 2020. [Habitat openness and edge avoidance predict saltmarsh sparrow abundance better than habitat area](https://doi.org/10.1093/CONDOR/DUAA019). The Condor 122:1–13.

Masse, R. J., A. M. Strong, and N. G. Perlut. 2008. The potential of uncut patches to increase the nesting success of grassland songbirds in intensively managed hayfields: A preliminary study from the Champlain Valley of Vermont. Northeastern Naturalist 15:445–452.

McClure, C. J. W., and G. E. Hill. 2012. [Dynamic versus static occupancy: How stable are habitat associations through a breeding season?](https://doi.org/10.1890/ES12-00034.1) Ecosphere 3:1–13.

McCracken, J. D. 2005. [Where the bobolinks roam: The plight of North America’s grassland birds](https://doi.org/10.1080/14888386.2005.9712771). Biodiversity 6:20–29.

National Park Service. 2005. Long-term monitoring plan for natural resources in the National Capital Region Network.

National Park Service. 2014. Manassas National Battlefield Park foundation document. U.S. Department of the Interior.

National Park Service. 2020. National Park Service seeks applicants for long-term agricultural land leases at Manassas National Battlefield Park. Manassas National Battlefield Park. https://www.nps.gov/mana/learn/news/national-park-service-seeks-applicants-for-long-term-agricultural-land-leases-at-manassas-national-battlefield-park.htm.

North American Bird Conservation Initiative. 2016. The state of North America’s birds 2016. Environment and Climate Change Canada, Ottawa, Ontario.

Palomo, I., C. Montes, B. Martín-López, J. A. González, M. García-Llorente, P. Alcorlo, and M. R. G. Mora. 2014. [Incorporating the social-ecological approach in protected areas in the Anthropocene](https://doi.org/10.1093/BIOSCI/BIT033). BioScience 64:181–191.

Partners in Flight. 2021. Avian Conservation Assessment Database, version 2021.

Perlut, N. G., A. M. Strong, T. M. Donovan, and N. J. Buckley. 2006. [Grassland Songbirds in a Dynamic Management Landscape: Behavioral Responses and Management Strategies](https://doi.org/10.1890/1051-0761(2006)016%5b2235:GSIADM%5d2.0.CO;2). Ecological Applications 16:2235–2247.

Peterjohn, B. G. 2006. Conceptual ecological model for management of breeding grassland birds in the Mid-Atlantic Region. Natural Resource Report. National Park Service, Philadelphia, PA.

Powell, A. F. L. A. 2006. [Effects of prescribed burns and bison (Bos bison) grazing on breeding bird abundances in tallgrass prairie](https://doi.org/10.1642/0004-8038(2006)123%5b0183:EOPBAB%5d2.0.CO;2). The Auk 123:183–197.

R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rasker, R. 2019. Public Land Ownership in the United States. Headwaters Economics. https://headwaterseconomics.org/public-lands/protected-lands/public-land-ownership-in-the-us/.

Rodenhouse, N. L., L. B. Best, R. J. O’Connor, and E. K. Bollinger. 1995. Effects of agricultural practices and farmland structures. Pages 269–293 *in* T. E. Martin and D. M. Finch, editors. Ecology and management of neotropical migratory birds. Oxford University Press, New York, NY, USA.

Rohrbaugh, R. W., D. L. Reinking, D. H. Wolfe, S. K. Sherrod, and M. A. Jenkins. 1999. Effects of prescribed burning and grazing on nesting and reproductive success of three grassland passerine species in tallgrass prairie. Studies in Avian Biology 19:165–170.

Rosenberg, K. V., A. M. Dokter, P. J. Blancher, J. R. Sauer, A. C. Smith, P. A. Smith, J. C. Stanton, A. Panjabi, L. Helft, M. Parr, and P. P. Marra. 2019. [Decline of the North American avifauna](https://doi.org/10.1126/science.aaw1313). Science 366:120–124.

Samson, F., and F. Knopf. 1994. [Prairie conservation in North America](https://doi.org/10.2307/1312365). BioScience 44:418–421.

Sauer, J. R., D. K. Niven, J. E. Hines, D. J. Ziolkowski, K. L. Pardieck, J. E. Fallon, and W. A. Link. 2017. The North American breeding bird survey, results and analysis 1966 - 2015. USGS Patuxent Wildlife Research Center, Laurel, MD.

Stevens, D. L., and A. R. Olsen. 2004. [Spatially balanced sampling of natural resources](https://doi.org/10.1198/016214504000000250). Journal of the American Statistical Association 99:262–278.

Thiermel, B., and A. Elmarhraoui. 2019. Suncalc: Compute sun position, sunlight phases, moon position and lunar phase.

Walk, J. W., and R. E. Warner. 2000. [Grassland management for the conservation of songbirds in the Midwestern USA](https://doi.org/10.1016/S0006-3207(99)00182-2). Biological Conservation 94:165–172.

Warren, K. A., and J. T. Anderson. 2005. [Grassland songbird nest-site selection and response to mowing in West Virginia](https://doi.org/10.2193/0091-7648(2005)33%5b285:GSNSAR%5d2.0.CO;2). Wildlife Society Bulletin 33:285–292.

Weidman, T., and J. A. Litvaitis. 2011. [Are small habitat patches useful for grassland bird conservation?](https://doi.org/10.1656/045.018.0207) Northeastern Naturalist 18:207–216.

West, A. S., P. D. Keyser, C. M. Lituma, D. A. Buehler, R. D. Applegate, and J. Morgan. 2016. [Grasslands bird occupancy of native warm-season grass](https://doi.org/10.1002/jwmg.21103). The Journal of Wildlife Management 80:1081–1090.

Wray, T., II, K. A. Strait, and R. C. Whitmore. 1982. [Reproductive success of grassland sparrows on a reclaimed surface mine in West Virginia](https://doi.org/10.2307/4086032). The Auk 99:157–164.