THE CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) AND GRASSLAND BIRD CONSERVATION IN MICHIGAN

By

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ABSTRACT

THE CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) AND GRASSLAND BIRD CONSERVATION IN MICHIGAN

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Grassland birds are one of the most imperiled guilds of North American birds, and their populations continue to decline in Michigan. Habitat loss and fragmentation due to agricultural conversion have been the most important causes of decline, with losses of native temperate grasslands >83% in the Midwest. Farmland set-aside programs such as the Conservation Reserve Enhancement Program (CREP) increase the area of native and introduced grasslands in the U.S., contributing to the conservation of grassland birds. Local evaluations (e.g., within major watersheds or ecoregions) of the impacts of farmland set-aside programs on grassland bird populations are important because grassland bird habitat requirements vary among geographic regions and there is geographic variation in the effect of farmland set-aside lands on wildlife. I examined avian populations of CREP grasslands in the Saginaw Bay watershed of Michigan from 2005-06 to address the following questions: 1) Are grassland birds present? 2) Do densities of grassland species differ in native and introduced CREP grasslands? 3) Is grassland bird occupancy correlated with grassland size, type, and structural variables and/or the extent or proximity of woody vegetation near grasslands (≤100 m)? Results of this study show that CREP grasslands provide habitat for grassland bird species in Michigan. Of 15 grassland species documented to occur in the state in the most recent Michigan Breeding Bird Atlas, 11 species were observed in study grasslands, including 1 Michigan endangered species

(Henslow's Sparrow), 3 Michigan special concern species (Dickcissel, Grasshopper Sparrow, and Northern Harrier), and 4 species with significantly declining population trends ($p < 0.10$) in Michigan according to recent analyses of Breeding Bird Survey data (Bobolink, Eastern Meadowlark, Ring-necked Pheasant, and Vesper Sparrow). Detection probability estimates were calculated for observed species. Species with low detection probability estimates (i.e., < 0.10) were not included in comparisons of density or in occupancy modeling. Densities of 5 of 8 grassland bird species with detection probability estimates ≥ 0.10 were higher in native grasslands than introduced grasslands during at least one year of the study (i.e., Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow). One species, Bobolink, was more abundant in the introduced vegetation during one year of the study. Grassland type was positively associated with the occupancy of Ring-necked Pheasant, Sedge Wren, and Vesper Sparrow, with higher detection-corrected occupancy estimates in native grasslands for all species. Grassland size was positively associated with Sedge Wren occupancy and negatively associated with Dickcissel occupancy. Although grassland type and grassland size were not significant correlates of occupancy for any species, results suggest that these covariates are more than random effects of grassland bird occupancy and hence should be included in future studies and considered in grassland bird conservation strategies. Occupancy modeling results also suggest a diverse response of grassland bird species to habitat features, indicating that species-specific information should be used when managing grassland bird species. Study results demonstrate that CREP grasslands, particularly native grasslands, provide grassland bird habitat, suggesting that maintaining and increasing the area of CREP grasslands should be a grassland bird conservation strategy in Michigan.

This work is dedicated those who protect and care for little brown birds and other grassland species.

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INTRODUCTION

Native temperate grassland, also called prairie or steppe, is one of the most endangered ecosystems in the world (Sampson and Knopf 1996). Vast areas have been converted to agriculture, and temperate grassland currently has the lowest protected area relative to converted area of any major biome (Hoekstra et al. 2005). Remaining grasslands tend to be fragmented and isolated, which disrupts natural ecological processes and reduces the viability of the grassland patches that persist (Leach and Givnish 1996). Other significant and interacting threats to grassland persistence, quality, and extent include changes to fire intensity and frequency, incompatible grazing regimes, woody plant expansion, and climate change (Briggs et al. 2005). In North America, as much as 99.9% of native prairies have vanished since European settlement (Sampson and Knopf 1994), and in the American Midwest region, 83-99% of native grasslands have been lost (Noss et al. 1995).

The decline of native North American grasslands has had significant negative impacts on associated animal species (Arenz and Joern 1996, Benedict et al. 1996, Corn and Peterson 1996, Rabeni 1996). In particular, a decrease in grassland bird populations has been documented (Knopf 1994, Herkert 1995). Grassland birds include approximately 30 species that evolved in the pre-European settlement grassland landscape of the North American Great Plains (Sauer et al. 1995, Knopf 1996). Breeding Bird Survey (BBS) data indicate that grassland birds, particularly endemic species, have declined more rapidly, more consistently, and over a wider geographic area than any other guild of North American birds (Sauer et al. 1995, Knopf 1996).

Farmland set-aside programs such as the Conservation Reserve Enhancement Program (CREP) increase the number of native and non-native grasslands in the U.S. and regulate management practices on enrolled private lands (e.g., mowing occurrence and timing), which may help diminish or reverse the decline of grassland bird species (Herkert 2009). CREP, a federal farmland set-aside program authorized by the 1996 Farm Bill and based upon another farmland set-aside program, the Conservation Reserve Program (CRP), has goals of protecting environmentally sensitive land, controlling soil erosion, safeguarding ground and surface water quality, and restoring wildlife habitat (USDA 2009, FSA 2010a). In the U. S., 10,805,106 ha are currently enrolled in CRP, and 481,576 ha are enrolled in CREP (FSA 2010b). In Michigan, 94,156 ha are currently enrolled in CRP, and 28,152 ha are enrolled in CREP (USDA 2010).

Both CREP and CRP are administered by the U.S. Department of Agriculture's (USDA) Farm Service Agency (FSA) and provide cost-share assistance and annual rental payments to farmers that remove highly erodible, environmentally sensitive cropland from agricultural production and plant these set aside fields with grasses, trees, or other approved vegetation under 10- to 15-year contracts (USDA 2009, FSA 2010a). However, while CRP is strictly a federal program, CREP is a partnership between federal and state governments, with additional partners in some cases, including tribal governments and private groups (FSA 2010a). CREP also differs from CRP in that, to address high-priority conservation areas, enrollment is permitted only in select geographic areas within participating states (FSA 2010a). Also, CREP offers greater financial incentives to eligible private landowners than CRP by leveraging CRP funding against state, tribal and other non-federal funds (FSA 2010a).

In Michigan, CREP is administered by the Michigan Department of Agriculture (MDA), with support from local Soil and Water Conservation Districts. The focus of the program is restricted to the River Raisin, Lake Macatawa, and Saginaw Bay watersheds (MDA 2009). The CREP conservation practice that is the focus of this study is grass plantings, non-native grasses and legumes or native grasses and wildflowers planted in fields to reduce erosion of cropland and provide wildlife habitat (MDA 2009). Other CREP conservation practices used in Michigan include: 1) filter strips, which are grass and legume plantings in strips adjacent to waterways or at the lower edges of fields to trap sediments and pollution in rain or snowmelt runoff and to provide wildlife habitat; 2) riparian buffers, or plantings of trees, shrubs, and grasses at the edges of streams or drains to filter runoff and provide wildlife habitat; 3) field windbreaks, or plantings of trees or shrubs to reduce wind erosion, protect crops, and provide wildlife habitat; and 4) wetland restoration in lands with hydric soils to improve water quality, recharge aquifers, provide flood control, and provide wildlife habitat (MDA 2009).

Geographic variation in the effects of set-aside lands on wildlife exists due to local and regional climatic differences and variation in program implementation, including management practices (Heard et al. 2000, Riffell et al. 2008). As a result, it is important to assess the impacts of farmland set-aside programs on wildlife at sub-national scales, such as state or major watershed levels.

In Michigan, at least 39 tallgrass prairies covered approximately 1 million ha prior to European settlement (Sargent and Carter 1999). Fewer than 810 ha of these prairies, or less than 0.1%, remain (Sargent and Carter 1999). Six grassland bird species are among those that have been designated as Michigan priority land birds by the bird conservation group Partners in Flight (Knutson et al. 2001, Matteson et al. 2009) and have been given management priority by the Michigan Department of Natural Resources (MDNR). These species are Bobolink, Dickcissel, Grasshopper Sparrow, Henslow's Sparrow, Northern Harrier, and Sedge Wren (Knutson et al. 2001, Matteson et al. 2009). Dickcissel, Grasshopper Sparrow, and Northern Harrier are also state species of special concern, and Henslow's Sparrow is a state endangered species. Several studies have evaluated the impacts of CRP on bird communities in Michigan (e.g., Pearks 1995, Millenbah et al. 1996, Minnis 1996, Best et al. 1997, Best et al. 1998). Currently, no studies have evaluated the vegetation characteristics of CREP grasslands in Michigan or their suitability as habitat for grassland bird species.

RESEARCH OBJECTIVES

The goal of this study was to evaluate grassland bird responses to the administration of CREP in Michigan. Objectives included:

1. describe and compare the structure and composition of native and introduced plant communities in CREP grasslands (Chapter 2);
2. model detection probability for avian species observed in CREP grasslands to inform future survey designs and identify those species that could reliably be included in more extensive data analyses (Chapters 2, 3, and 4);
3. compare avian density for species with adequate detection probability estimates (i.e., $p \geq 0.10$) in:

a. CREP Conservation Practice 23 (CP23) and Conservation Practice 1 (CP1) whole fields (administrative-level comparison) (Chapter 2) and

b. native grassland plant communities of CP23 fields and introduced grassland plant communities of CP1 grassland plant communities (ecological-level comparison) (Chapter 2);

4. model occupancy for grassland bird species with adequate detection probability estimates in:

    a. CP23 and CP1 CREP whole fields (administrative-level comparison) (Chapter 3); and

    b. native and introduced portions of CREP grassland plant communities (ecological-level comparison) (Chapter 4) and

5. for grassland bird species with adequate detection probability estimates, evaluate the correlation of occupancy with structural variables of native and introduced plant communities and/or proximate environmental cues (e.g., area of woody vegetation ≤100 m from study areas) (Chapter 4).

STUDY AREA

This study occurred within the 8,219 km^2 Sandusky Lake Plain sub-subsection of Michigan's southern Lower Peninsula (Albert 1995). This sub-subsection consists of flat clay lake plain, bordered by shoreline dunes and sand plain. In the center of the flat clay lake plain, long and narrow till plains and ridges of end moraines parallel either Saginaw Bay or Lake Huron (Albert 1995). Prior to European settlement, extensive wet and wet-mesic prairies occurred upland from the coastal Great Lakes marshes of Saginaw Bay, but now occur only as small remnants, generally on State-owned lands (Albert 1995).

Other presettlement vegetation types of the Sandusky Lake Plain sub-subsection differed between the clay lake plain, end moraine, and sand lake plain (Albert 1995). The vegetation of the clay lake plain was dominated by hemlock (*Tsuga canadensis*) upland conifer forests, and included lowland hardwoods in the wettest areas and beech (*Fagus grandifolia*)- sugar maple (*Acer saccharum*) forest in more well-drained areas (Albert 1995). Beech - sugar maple forests were also typical of the sloping end moraines, with conifer swamps occupying the broad moraine footslope wetlands (Albert 1995). The flat sand plain and low dunes supported hemlock - white pine (*Pinus strobus*) conifer forests and small areas of white oak (*Quercus alba*) - black oak (*Quercus velutina*) forest (Albert 1995). Along the shores of Saginaw Bay, extensive dune - swale wetland complexes and emergent marshes occurred (Albert 1995). Intensive agricultural development has occurred throughout the sub-subsection due to productive loamy soils and a lake-moderated climate (Albert 1995). Most of the clay lake plain, parts of the sand plain, the moraines, and large areas of wet prairie and marsh have been converted to farmland, leaving few presettlement cover types in the Sandusky Lake Plain sub-subsection (Albert 1995).

In Michigan, 22,983 ha of privately-owned land in the Saginaw Bay, Lake Macatawa, and River Raisin watersheds were enrolled in CREP at the time of this study (USDA 2007). From CREP lands in the Saginaw Bay watershed, 14 CREP grassland fields in Tuscola County, ranging in size from 6.8 to 23.6 ha, were selected for study (Figure 1.1). All grassland study fields were planted in 2002 and were surveyed for birds in 2005 and 2006. Seven study fields were CREP conservation practice (CP) type 23 (hereafter CP23) that had been planted with a seed mixture of native grasses and native and introduced wildflowers over 100% of the field area, and 7 study fields were CP type

1 (hereafter CP1) fields that had been planted with introduced grasses and legumes over 70% of the field area, with the remaining 30% of the field area planted with the same native seed mixture used in CP23 fields (Figure 1.1, Table 1.1). Native sections of CP1 fields were established to provide winter shelter for wildlife (e.g., Ring-necked Pheasant, *Phasianus colchicus*) because native grasses tend to remain upright during winter whereas introduced grasses often collapse (USDA 2000a, USDA 2000b). In two CP1 study fields, the native seed mixture did not successfully establish and the field area was composed only of introduced vegetation (Figure 1.1). In 2005, the native seed mixture cost $16.00/pound (i.e., $104.00/acre or $256.99/hectare) and the introduced seed mixture cost $1.90/pound (i.e., $20.90/acre or $51.65/hectare) plus 6% sales tax through the Tuscola County Conservation District.

Within CREP, CP23 fields are wetland restorations and all CP23 fields had hydric soils. A small wetland (i.e., < 10% of upland area) was created in each CP23 field by cutting drain tiles, and if necessary, by creating a push-out, or shallow depression in the soil. The CREP program offers a whole-field native grassland conservation practice without a wetland component, CP2, which was rare in the Saginaw Bay area. Due to the scarcity of CP2 fields, upland areas of CP23 fields were used for comparison to 1) CP1 whole fields and 2 introduced portions of CP1 fields). Wetlands and buffers of 25 m surrounding each wetland were not included in CP23 avian or vegetation surveys. The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha), and the introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha).

APPENDIX 1.1: CHAPTER 1 TABLES AND FIGURES

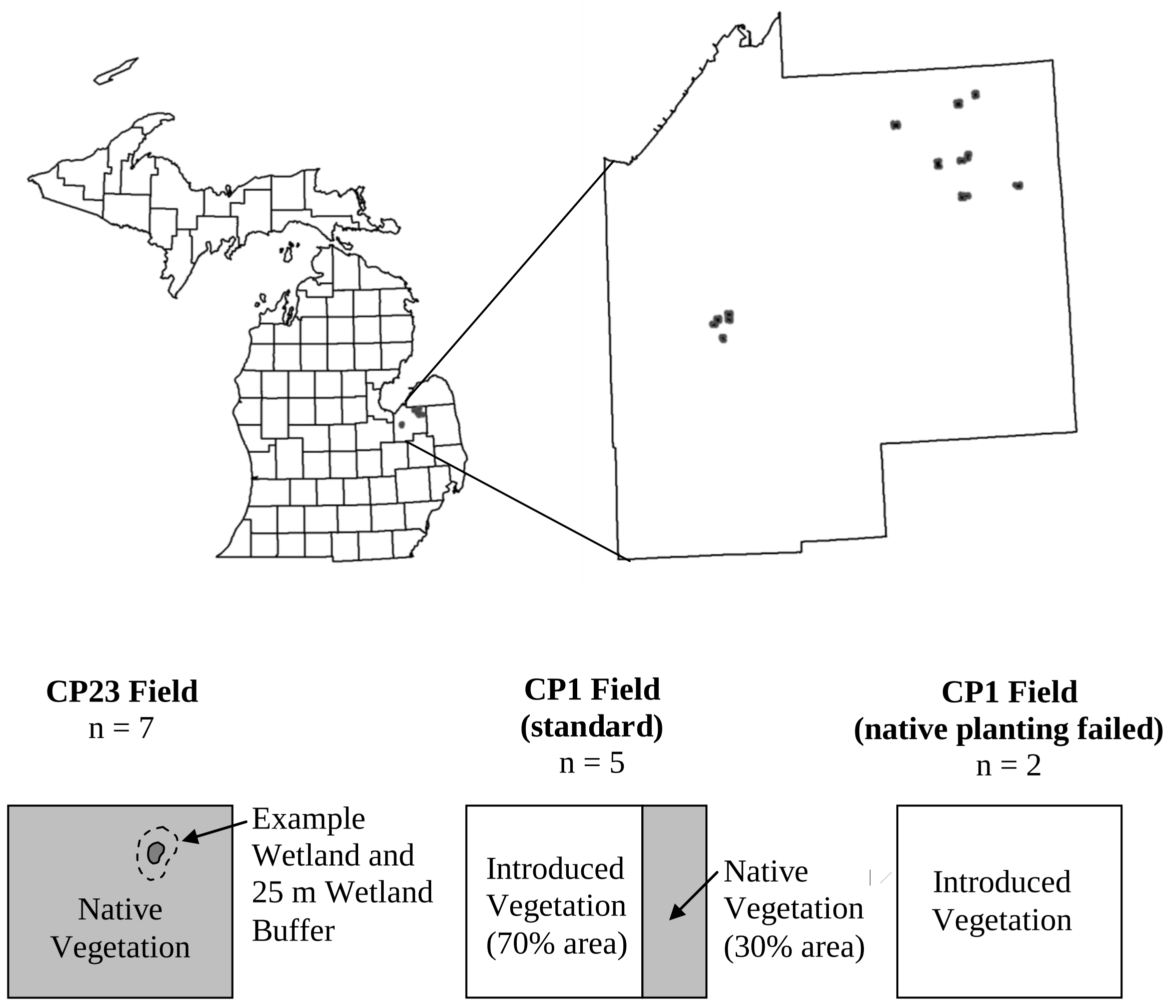

Fig. 1.1. Location, type, and number of Conservation Reserve Enhancement Program (CREP) study fields within Tuscola County, Saginaw Bay watershed, Michigan, 2005 and 2006. Wetlands and wetland buffers were excluded from avian surveys and vegetation sampling. The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). The introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha), and CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha).

Table 1.1. Plant species and proportions of species included in the seed mixtures used to establish the native and introduced vegetation communities of Conservation Reserve Enhancement Program (CREP) grasslands of Michigan's Saginaw Bay watershed, 2002.

Common Name	Scientific Name	% Total Seed Mixture	% Grass, Wildflower, or Legume Mixture
***CP23 Fields and Native Section of CP1 Fields** - planted 6.5 pounds/acre (16.1 pounds/ha)*			
Native Grasses		92.3	
Big Bluestem*	*Andropogon gerardii*		33.3
Indiangrass*	*Sorghastrum nutans*		33.3
Little Bluestem*	*Andropogon scoparius*		33.3
Plateau® Tolerant Wildflower Mixture		7.7	
Black Eyed Susan*	*Rudbeckia hirta*		10.0
Blanket Flower, Perennial	*Gaillardia aristata*		2.0
Catchfly	*Silene armeria*		2.0
Clover, Crimson	*Trifolim incarnatum*		10.0
Coneflower, Prairie, 'Mexican Hat'	*Ratibida columnifera*		3.0
Coneflower, Purple*	*Echinacea purpurea*		2.0
Coneflower, Yellow*	*Ratibida columnifera*		5.0
Coreopsis, Lance Leaf*	*Coreopsis lanceolata*		10.0
Coreopsis, Plains	*Coreopsis tinctoria*		3.0
Cosmos, 'Sensation' Mix	*Cosmos bipinnatus*		5.0
Cosmos, Sulphur	*Cosmos sulphureus*		5.0
Daisy, Shasta, 'Alaska'	*Chrysanthemum x superbum*		4.0
Dame's Rocket	*Hesperis matronalis*		5.0
Flax, Perennial	*Linum perenne lewisii*		10.0
Lupine, Perennial*	*Lupinus perennis*		4.0
Poppy, California	*Eschscholzia californica*		5.0
Poppy, Corn, 'Shirley Single' Mix	*Papaver rhoeas*		5.0
Wallflower, Siberian	*Erysiumum x marshallii*		10.0
***Introduced Section of CP1 Fields** - planted 11 pounds/acre (27.2 pounds/ha)*			
Introduced Grasses		45.5	
Orchardgrass	*Dactlyis glomerata*		50.0
Timothy	*Phleum pratense*		50.0
Introduced Legumes		54.5	
Alfalfa	*Medicago sativa*		50.0
Red Clover	*Trifolium pratense*		50.0

**Michigan native species (Voss 1985, Voss 1996, USDA NRCS 2011)*

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CHAPTER 2: GRASSLAND BIRD DETECTION PROBABILITY ESTIMATION AND DENSITY COMPARISON IN MICHIGAN CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) GRASSLANDS

INTRODUCTION

Temperate grassland is one of the most endangered ecosystems in the world, having the lowest protected area relative to converted area of any major biome (Sampson and Knopf 1996, Hoekstra et al. 2005). Vast areas of temperate grassland have been converted to agriculture, and other significant and interacting threats to grassland persistence, quality, and extent include changes to historical fire frequency and intensity, incompatible grazing regimes, woody plant expansion, and climate change (Briggs et al. 2005). Remaining North American native grasslands (1–17% in the Midwest) tend to be fragmented, which disrupts natural ecological processes and reduces the viability of grassland patches that persist (Noss et al. 1995, Leach and Givnish 1996).

The decline of native North American grasslands has negatively impacted associated animal species (Arenz and Joern 1996, Benedict et al. 1996, Corn and Peterson 1996, Rabeni 1996), and in particular, decreases in grassland bird populations have been quantified (Knopf 1994, Herkert 1995). Breeding Bird Survey (BBS) data indicate that grassland bird populations have declined more rapidly, more consistently, and over a wider geographic area than any other guild of North American birds (Sauer et al. 1995, Knopf 1996), and populations of grassland bird species continue to decline in Michigan (Sauer et al. 2008). The most recent population trend analysis of Breeding Bird Survey (BBS) data from 1980 – 2007 shows significantly decreasing population trends in

Michigan ($p < 0.10$) for 4 out of 10 grassland bird species included in the BBS analysis (i.e., Vesper Sparrow, Eastern Meadowlark, Bobolink, and Ring-necked Pheasant; species included in the analysis were those encountered on ≥ 15 survey routes), and no species showed significant population increases (Sauer et al. 2008).

Farmland set-aside programs such as the Conservation Reserve Enhancement Program (CREP) increase the number of native and non-native grasslands in the U.S., which may diminish or reverse the decline of grassland bird populations (Herkert 2009). Geographic variation in the effects of farmland set-aside lands on wildlife exists due to local and regional climatic differences and variation in program implementation, including management practices (Heard et al. 2000, Riffell et al. 2008). As a result, it is important to understand the impacts of farmland set-aside programs on wildlife at sub-national scales, such as state or watershed levels. Currently, no studies have evaluated the vegetation characteristics of CREP grasslands in Michigan or their suitability as habitat for grassland bird species.

The goal of this study was to quantify the relationships between CREP grasslands and grassland bird density in Michigan. In this study, CREP grasslands were identified as CREP conservation practice (CP) 23 plantings (hereafter CP23 fields or native fields), which were composed of native grasses and wildflowers, and CREP conservation practice 1 plantings (hereafter CP1 fields or introduced fields), in which 70% of field area consisted of introduced grasses and legumes and 30% of the field area was native grasses and wildflowers. Study objectives were to: 1) describe and compare the vegetation composition and structure of CP23 and CP1 fields and 2) describe and compare the density of grassland bird species in a) CP23 and CP1 fields, corresponding to the scale at which CREP is administered (hereafter referred to as the administrative level of inquiry) and b) native and introduced CREP grassland plant communities (hereafter referred to as the ecological level of inquiry).

METHODS

This study occurred within the 8,219 km^2 Sandusky Lake Plain sub-subsection of Michigan's southern Lower Peninsula (Albert 1995). This sub-subsection consists of flat clay lake plain, bordered by shoreline dunes and sand plain. In the center of the flat clay lake plain, long and narrow till plains and ridges of end moraines parallel either Saginaw Bay or Lake Huron. Prior to European settlement, extensive wet and wet-mesic prairies occurred upland from the coastal Great Lakes marshes of Saginaw Bay, but largely due to agricultural conversion, these prairies now occur only as small remnants, generally on State-owned lands.

In Michigan, 22,983 ha of land in the Saginaw Bay, Lake Macatawa, and River Raisin watersheds were enrolled in CREP at the time of this study (USDA 2007). From CREP lands in the Saginaw Bay watershed, 14 CREP grassland fields in Tuscola County, ranging in size from 6.8 to 23.6 ha, were selected for study (Figure 2.1). All grassland study fields were planted in 2002 and were surveyed for birds in 2005 and 2006. Seven study fields were CP23 fields that had been planted with a seed mixture of wildflowers and native grasses over 100% of the field area, and 7 study fields were CP1 fields that had been planted with introduced grasses and legumes over 70% of the field area, with the remaining 30% of the field area planted with the same native seed mixture used in CP23 fields (Figure 2.1, Table 2.1). Native sections of CP1 fields were established to provide winter shelter for wildlife (e.g., Ring-necked Pheasant, *Phasianus colchicus*) because native grasses tend to remain upright during winter conditions whereas introduced grasses generally collapse (USDA 2000a, 2000b). In two CP1 fields, the native seed mixture did not successfully establish and the field area was composed only of introduced vegetation (Figure 2.1). In 2005, the native seed mixture cost $16.00/pound (i.e., $104.00/acre or $256.99/hectare) and the introduced seed mixture cost $1.90/pound (i.e., $20.90/acre or $51.65/hectare) plus 6% sales tax through the Tuscola County Conservation District.

Within CREP, CP23 fields are wetland restorations and all CP23 fields had hydric soils. A small wetland was created in each CP23 field by cutting drain tiles, and, if necessary, by creating a push-out, or shallow depression in the soil. Wetlands of the CP23 fields were generally less than 3% of the upland field area, but in one field the wetland was approximately 10% of the upland field area. The CREP program offers a whole-field native grassland conservation practice without a wetland component, CP2, which was rare in the Saginaw Bay area. Due to the scarcity of CP2 fields, the upland area of CP23 fields was the native grassland used for comparison to the introduced portion of CP1 fields and to CP1 whole fields. Wetlands and a buffer of 25 m surrounding the wetlands were not included in CP23 avian surveys (Figure 2.1). The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha), and the introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha).

Vegetation composition and structure were investigated using one randomly located sampling point per acre (i.e., 1 sampling point per 0.4 ha) in each study field (n

=17 to 59 sampling points per field). Vegetation data were collected once in 2005 during mid-summer (15 June – 14 July) and three times in 2006 during early summer (15 May – 14 June), mid-summer (15 June – 14 July), and late summer (15 July – 14 August). Canopy height and litter depth were measured at each sampling point using a yardstick. A modified 50 cm x 25 cm Daubenmire frame was centered on each sampling point (Daubenmire 1959). All plant species within the Daubenmire frame were identified. The percentage of Daubenmire frame area composed of bare ground, litter cover, and canopy cover were visually estimated. Percent standing dead and living vegetation were also visually estimated, and within the living canopy vegetation, the percent cover of grasses, forbs, and woody vegetation were also determined. The total area of the Daubenmire frame was accounted for as:

$$100\% \text{ plot area} = \% \text{ bare ground} + \% \text{ litter cover} + \% \text{ total canopy cover}$$

where:

$$\% \text{ total canopy cover} = \% \text{ dead canopy vegetation} + \% \text{ live canopy vegetation}$$

and:

$$\% \text{ live canopy vegetation} = \% \text{ grasses} + \% \text{ forbs} + \% \text{ woody plants}.$$

Boxplots were used to compare vegetation variables of CP23 native and CP1 introduced plant communities. Native plant communities were present in portions of the CP1 fields in addition to occurring throughout the CP23 fields (Figure 2.1). CP1 native grassland plantings ranged in size from 0.3–6.9 ha (mean [SE] = 4.0 [1.0], median = 4.2) To investigate whether these two native plant communities provided similar potential grassland bird habitat, the structure of CP23 native vegetation and the native portions of

CP1 fields were also compared using boxplots. Boxplot whiskers extended to the minimum and maximum observed values unless outliers were present, in which case whiskers extend 1.5 times the interquartile range and outliers are denoted outside the range of the whiskers.

Avian data were collected using line-transect sampling (Edwards et al. 1981). The total area of each study field (i.e., except CP23 wetlands and associated wetland buffers) was surveyed every 2 weeks from 1 June – 14 August in 2005 (n=5 surveys in all fields) and from 15 May – 14 August in 2006 (n=6 surveys per field). Surveys took place along transects running the length of each field. The first transect was established 25 m from a randomly selected field corner and subsequent transects were established at 50 m intervals across the width of the field. An investigator recorded the species and location of each individual bird seen or heard. If an individual bird was flushed by the observer, the new location of the bird was noted in order to prevent duplicate counting of individuals. If a bird was observed but could not be identified, the location of the bird was recorded and the species designated as unknown. Unidentified birds were not included in data analyses. Avian surveys were conducted after sunrise and before 12:00 p.m., and surveys were not performed in the rain or if wind speeds were over 16 kph.

Comparisons of avian density were completed at administrative (i.e., between CP23 and CP1 fields) and ecological (i.e., between CP23 fields and the introduced portion of CP1 fields) levels using boxplots with whisker lengths extending to the minimum and maximum observed values unless outliers were present, in which case whiskers extend 1.5 times the interquartile range and outliers are denoted outside the range of the whiskers. Data from the introduced portion of all 7 CP1 fields were used during ecological-level comparisons, but for administrative-level comparisons, data were used from 5 of the 7 CP1 study fields because the native plant community portion of 2 CP1 fields did not establish successfully. For each grassland bird species, mean avian density values in CP23 fields, CP1 fields, and the introduced portion of CP1 fields were determined by averaging the density values of individual study areas of each field/cover type. Avian species were categorized as reliably detected (i.e., detection probability ≥ 0.10) or unreliably detected (i.e., detection probability could not be estimated or detection probability <0.10) in each CREP field and grassland type. Differences in bird densities were evaluated only for those species reliably detected. Density was selected as the measure of abundance for comparison in order to control for variation in study field size and help account for passive sampling, which is that large habitat patches are more likely to be occupied by a species than small patches through chance alone (Ribic et al. 2009).

Studies of species occurrence and abundance that assume perfect detection are often biased by imperfect detection (i.e., a species was actually present but was not observed) (MacKenzie et al. 2009). To account for imperfect detection, I estimated the detection probability, p, of each observed species, and omitted species with low detection probability estimates (i.e., $p \leq 0.10$) from further analyses. Avian detection modeling was conducted for species that were observed in at least 10% of the total number of surveys (MacKenzie et al. 2002). Species that were observed in > 10% of surveys were not included in density comparisons.

Detection probability of each avian species was estimated using single-season models (MacKenzie et al.2002) in the program PRESENCE (Hines 2006). The use of single season models was based on the results of likelihood ratio testing (MacKenzie et al. 2006, Nichols et al. 2008), which indicated that the more highly parameterized multi-season models did not fit the data significantly better than single season models ($p > 0.05$).

For studies with small sample sizes, model ranking using Akaike's information criterion corrected for small sample sizes (AIC_c) is recommended (Burnham and Anderson 2002). The top-ranking detection model for each avian species was identified as having the lowest AIC_c value among all candidate models with constant occupancy, $\psi(\cdot)$ (MacKenzie et al. 2002). Candidate (i.e., successful) detection models were those that achieved 1) numerical convergence of parameter estimates to >6 significant figures, 2) a balanced variance-covariance matrix, and 3) small- or moderately-sized standard errors of betas, the untransformed estimates of coefficients for covariates (i.e., standard error was no greater than 5 times the corresponding beta) (D. MacKenzie, Program PRESENCE – FAQ at http://www.phidot.org/forum, accessed October 2009).

Detection probability estimates were obtained from the top-ranking detection model for each species. For CREP administrative-scale analyses, potential detection covariates were ordinal date and field type (i.e., CP23 or CP1). For ecological-level analyses, potential detection covariates were ordinal date and plant community type (i.e., native or introduced). Ordinal date was standardized to reduce biases in parameter estimation (MacKenzie et al. 2002). When plant community or field type and/or ordinal date were covariates in the top-ranking detection model, multiple estimates of detection resulted, and the lowest estimate was reported. For species where no detection models were successfully generated using the full data set, detection probability estimates for individual study sites were examined, and if detection probability estimates for a subset of the data appeared adequate (e.g., detection estimates for all fields in 2006 were ≥ 0.10), then detection modeling was performed on this data subset. For species where the naïve estimate of occupancy, ψ, equaled 1, the fields were fully occupied and detection modeling was conducted with ψ fixed to 1.

RESULTS

Vegetation results

Sixty-nine plant species were observed among all fields in this study. Sixty species were found in CP23 native plant communities, and 37 species occurred in CP1 introduced plant communities (Table 2.2). Thirty-two plant species were unique to native communities, and nine species occurred only in introduced communities (Table 2.2). The most commonly observed grass species of native fields (i.e., observed in $\geq$ 20% of vegetation sample plots) were Big Bluestem (*Andropogon gerardii*), Indiangrass (*Sorghastrum nutans*), and Little Bluestem (*Andropogon scoparius*), while the most frequently observed grass species in introduced plant communities were Orchardgrass (*Dactlyis glomerata*), Timothy (*Phleum pratense*), and Annual Ryegrass (*Lolium perenne*). The most commonly observed forbs in native fields included Dandelion species (*Taraxacum spp.*), Thistle species (*Cirsium spp.*), Black Medick (*Medicago lupulina*), Goldenrod species (*Solidago spp.*), Queen Anne's Lace (*Daucus carota*), and Lettuce species (*Lactuca spp.*). The most frequently observed forb species in introduced plant communities were Dandelion and Thistle species, along with Alfalfa (*Medicago sativa*), an intentionally planted species.

Structural variables of the CP23 native plant communities and the introduced portion of CP1 fields were summarized and compared for 1) 2005 mid-summer, 2) 2006 early summer, 3) 2006 mid-summer, and 4) 2006 late summer (Figures 2.2-2.5).

Bare ground tended to compose the smallest proportion of total sample plot area (i.e., total plot area consists of bare ground, litter cover, and canopy cover) in native and introduced plant communities; in native vegetation, bare ground made up the smallest percentage of plot area throughout the avian breeding season in 2006, and in introduced vegetation, bare ground made up the smallest proportion of plot area in 2005 and throughout 2006 (Figures 2.2-2.5). In CP23 fields in 2006, bare ground ranged from 1.2 – 12.2% in early summer (mean = 7.5, SE = 1.4), from 0.8 – 14.3% in mid-summer (mean = 6.5, SE = 1.6), and from 2.4 – 16.5% in late summer (mean = 7.5, SE = 1.7). In introduced vegetation of CP1 fields in 2005, bare ground ranged from 0 – 6.3% in mid-summer (mean = 2.5, SE = 1.0), and in 2006, bare ground ranged from 0 – 10.6% in early summer (mean = 3.2, SE = 1.4), from 0 – 3.8% in mid-summer (mean = 1.1, SE = 0.6), and from 0 – 3.8% in late summer (mean = 1.1, SE = 0.5).

The total plant canopy (i.e., living and standing dead vegetation) tended to compose the largest proportion of sample plot area in native and introduced plant communities; in native vegetation, the total plant canopy made up the largest proportion of sample plot area in mid-summer 2005 and in mid- and late summer 2006, and in introduced vegetation, the total plant canopy composed the largest proportion of plot area in mid-summer 2005 and throughout 2006 (Figures 2.2-2.5). In CP23 fields in 2005, total canopy cover ranged from 54.8 – 72.1% in mid-summer (mean = 66.1, SE = 3.2), and in 2006, total canopy cover ranged from 36.3 – 71.1% in mid-summer (mean = 58.3, SE = 5.6) and from 49.1 – 73.7% in late summer (mean = 59.4, SE = 3.3). In introduced vegetation of CP1 fields in 2005, total canopy cover ranged from 71.3 – 95.6% in mid-summer (mean = 87.9, SE = 3.8), and in 2006, total canopy cover ranged from 28.7 – 85.3% in early summer (mean = 60.3, SE = 8.1), from 45.4 – 84.4% in mid-summer (mean = 63.8, SE = 5.4), and from 47.0 – 91.1% in late summer (mean = 69.4, SE = 6.5).

Standing dead vegetation made up a small portion of the total canopy within native and introduced plant communities throughout the study (Figures 2.2-2.5). In CP23 fields in 2005, dead vegetation in the canopy ranged from 0.7 – 4.6% (mean = 2.4, SE = 0.6) in mid-summer, and in 2006, standing dead vegetation ranged from 1.7 – 6.9% in early summer (mean = 42.5, SE = 7.6), from 0.1 – 6.6% in mid-summer (mean = 2.5, SE = 0.9), and from 0.1 – 3.4% in late summer (mean = 1.5, SE = 0.5). In introduced vegetation of CP1 fields in 2005, standing dead vegetation ranged from 0 – 11.8% in mid-summer (mean = 2.3, SE = 1.6), and in 2006, standing dead vegetation ranged from 0 – 2.6% in early summer (mean = 1.0, SE = 0.4), from 0 – 0.8% in mid-summer (mean = 0.3, SE = 0.1), and from 0 – 5.1% in late summer (mean = 0.9, SE = 0.7).

There were virtually no woody plants in the live canopy of both native and introduced vegetation, in both years and all time periods of the study (Figures 2.2-2.5). With the absence of woody vegetation, the live canopy was composed only of forbs and grasses, thus the relationship of grasses to forbs in the living canopy was inversely proportional, i.e., if the percentage of grasses in the living canopy increased, the percentage of forbs decreased. As expected for the warm-season grasses of native plantings, grasses tended to be less abundant than forbs in the live canopy of CP23 fields in early summer 2006 (Figure 2.2; forbs: range 8.3 – 41.1%, mean = 27.5, SE = 5.9;

grasses: range 6.4 – 33.3%, mean = 15.0, SE = 3.3), but by mid-summer forbs and grasses were equally abundant in native vegetation (Figure 2.4; forbs: range 9.0 – 52.1%, mean = 29.0, SE = 5.8; grasses: 21.0 – 37.6%, mean = 26.5, SE = 2.1), and by late summer grasses were more abundant than forbs (Figure 2.5; forbs: range 9.6 – 41.0%, mean = 22.2, SE = 4.6; grasses: 24.3 – 54.6%, mean = 35.3, SE = 4.1). Within introduced vegetation, forbs and the cool-season grasses were equally abundant in the live canopy throughout the summer in 2006 (Figures 2.3-2.5). In early summer 2006, forbs in the living canopy of introduced vegetation ranged from 8.7 – 49.8% (mean = 29.0, SE = 5.7) and grasses ranged from 19.3 – 42.4% (mean = 30.3, SE = 3.4); in mid-summer forbs ranged from 14.9 – 41.5% (mean = 36.4, SE = 6.1) and grasses ranged from 22.3 – 34.1% (mean = 27.0, SE = 1.5); and in late summer forbs ranged from 20.8 – 61.4% (mean = 37.7, SE = 6.0) and grasses ranged from 23.3 – 37.2% (mean = 30.8, SE = 2.4).

In addition to different patterns of change in the ratio of grasses and forbs within native and introduced plant canopies during the course of the avian breeding season, other differences in the structure of native and introduced vegetation existed. In particular, native plant communities tended to have more bare ground and less total plant canopy cover than introduced plant communities (Figures 2.2-2.5). Less bare ground was present in introduced vegetation than in native fields throughout the study, and more canopy cover existed in introduced than native vegetation in mid-summer 2005 and early and mid-summer 2006 (Figures 2.2-2.5). Bare ground in CP23 vegetation in 2005 ranged from 9.5 – 24.5% (mean = 15.9, SE = 2.4); total canopy cover in native vegetation in early summer 2006 ranged from 22.9 – 64.5% (mean = 46.2, SE = 7.1); see bare ground and total canopy cover results above for other range and mean information. Differences between native and introduced plant communities may be most important for grassland birds during early summer when breeding territories are selected. In early summer 2006, native vegetation had a larger proportion of litter cover than introduced vegetation (Figure 2.3). Litter cover in early summer 2006 ranged from 25.6 – 76.0% (mean = 46.3, SE = 7.9) in native vegetation and from 14.7 – 70.1% (mean = 36.6, SE = 7.3) in introduced vegetation. Also, since native vegetation tends to remain upright during winter whereas introduced grasses generally collapse (USDA 2000a, 2000b), it was not surprising that native plant communities had more standing dead vegetation in the canopy than introduced vegetation in early summer 2006 (Figure 2.3; see standing dead results above for range and mean information).

Native plant communities were present in portions of the CP1 fields as well as throughout CP23 fields. To examine whether these two types of native plant communities provided similar potential grassland bird habitat, the composition and structure of CP23 native vegetation and the native portions of CP1 fields were compared.

The composition of CP23 and CP1 native vegetation was similar, with few differences in the most commonly observed plant species (i.e., observed in ≥ 20% of vegetation sample plots). In CP1 native vegetation the most common plant species were Big Bluestem, Indiangrass, Kentucky Bluegrass (*Poa pratensis*), Dandelion species, Thistle species, Goldenrod species, Queen Anne's Lace, and Shasta Daisy (*Chrysanthemum x superbum 'Alaska'*), and the most common plant species in CP23 fields were Big Bluestem, Indiangrass, Little Bluestem, Dandelion species, Thistle species, Goldenrod species, Queen Anne's Lace, Black Medick, and Lettuce species.

Structural variables of CP23 native vegetation and the native portions of CP1 fields were summarized and compared for 1) 2005 mid-summer; 3) 2006 early summer; 4) 2006 mid-summer; and 5) 2006 late summer (Figures 2.6-2.9).

Structural characteristics of both types of native plantings were similar (Figures 2.6-2.9); only a few differences in the vegetation of CP23 and CP1 native plant communities were observed. In late summer 2006, CP23 native vegetation had less total canopy cover and slightly more bare ground than CP1 native vegetation (Figure 2.9). In late summer 2006, bare ground ranged from 2.4 – 16.5% (mean = 7.5, SE = 1.7) in CP23 fields and from 1.1 – 7.9% (mean = 4.1, SE = 1.4) in CP1 native vegetation; total canopy cover ranged from 49.1 – 73.7% (mean = 59.4, SE = 3.3) in CP23 fields and from 70.9 – 77.3% (mean 71.1, SE= 3.1) in CP1 native plant communities. Also, in mid-summer 2005, more standing dead vegetation was present in CP23 fields than in CP1 native plant communities (Figure 2.6), but the opposite was true the following year, when in mid-summer 2006, CP23 fields contained slightly less standing dead vegetation than CP1 native plant communities (Figure 2.8). In mid-summer 2005, standing dead vegetation ranged from 0.7 – 4.6% (mean = 2.4, SE = 0.6) in CP23 fields and from 0.2 – 0.9% (mean = 0.5, SE = 0.1) in CP1 native vegetation, and in mid-summer 2006, standing dead vegetation ranged from 0.1 – 6.6% (mean = 2.5, SE = 0.9) in CP23 fields and from 1.8 – 8.4% (mean = 4.4, SE = 1.4) in CP1 native plant communities.

Avian results

A total of 34 avian species were observed in CREP study fields during the 2 years of this study (Table 2.3). In 2005, 23% of individual birds observed during surveys were unidentified, and in 2006 surveyors were more experienced and only 8% of observed individuals were unidentified. Using the grassland bird list developed for the BBS by the U.S. Geographic Survey (Sauer et al. 1995), 11 of the observed species were identified as grassland specialists: Bobolink (*Dolichonyx oryzivorus*), Dickcissel (*Spiza americana*), Eastern Meadowlark (*Sturnella magna*), Grasshopper Sparrow (*Ammodramus savannarum*), Henslow's Sparrow (*Ammodramus henslowii*), Horned Lark (*Eremophila alpestris*), Northern Harrier (*Circus cyaneus*), Ring-necked Pheasant, Savannah Sparrow (*Passerculus sandwichensis*), Sedge Wren (*Cistothorus platensis*), and Vesper Sparrow (*Pooecetes gramineus*). Six of these grassland specialist species are of special conservation priority in Michigan (Table 2.3). Bobolink, Dickcissel, Grasshopper Sparrow, Henslow's Sparrow, Northern Harrier, and Sedge Wren are designated as Michigan priority landbirds by the bird conservation group Partners in Flight (Knutson et al. 2001, Matteson et al. 2009), while Dickcissel, Grasshopper Sparrow, and Northern Harrier are also state species of special concern and Henslow's Sparrow is a state endangered species (Table 2.3).

Avian detection probability modeling

Three grassland bird species were too rarely observed to estimate their detection probabilities (i.e., observed in less than 10% of surveys): Henslow's Sparrow, Horned Lark, and Northern Harrier. Detection probability modeling was performed for the 8 other grassland specialist species observed during this study: Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow. Likelihood ratio testing indicated that more highly parameterized multi-season models did not fit the data significantly better than simpler single-season models ($p > 0.05$), so single season models were used in all detection and occupancy modeling efforts (MacKenzie et al. 2006, Nichols et al. 2008).

At the administrative level of analysis (i.e., CP23 and CP1 fields), all 8 grassland species were reliably detected. Adequate detection probability estimates (i.e., $p \geq 0.10$) were achieved for all 8 species, with detection probability estimates ranging from 0.12 to 0.81 (Table 2.4). However, an adequate detection probability estimate was obtained only when a subset of the data was used in the case of one species, Sedge Wren (2006 data), most likely because Sedge Wren was rare in 2005 (mean density = 0.01, SE = 0.01 in CP23 fields; mean density = 0.0, SE = 0.0 in introduced vegetation of CP1 fields). CREP field type and/or standardized ordinal date were included in the best detection models for Bobolink, Eastern Meadowlark, Ring-necked Pheasant, Savannah Sparrow, and Sedge Wren (Table 2.4). The detection probability of Bobolink varied between 2005 and 2006, but detection probability estimates were >0.63 regardless of year, indicating that Bobolink was readily detected throughout each breeding season (Figure 2.10). Eastern Meadowlark was reliably detected in CP23 and CP1 fields, but the species tended to be more likely to be detected in CP23 fields, although the 95% confidence intervals for detection estimates of the two field types were overlapping (i.e., 95% confidence interval was 0.10 – 0.16 in CP23 fields and 0.08 – 0.15 in CP1 fields; Figure 2.11). Ring-necked Pheasant detection probability varied between 2005 and 2006, but detection probability estimates were >0.17 in both years, indicating that the species was reliably detected throughout both breeding seasons (Figure 2.12). Ring-necked Pheasant was adequately detected in CP23 and CP1 fields, but the species tended to be more likely to be detected in CP23 fields, although the 95% confidence intervals for detection estimates of the two field types were overlapping (i.e., 95% confidence interval was 0.21 – 0.46 in CP23 fields and 0.05 – 0.32 in CP1 fields; Figure 3.4). Detection probability estimates for Savannah Sparrow were >0.81 regardless of year, indicating that the species was reliably detected throughout each breeding season (Figure 2.13). The detection probability of Sedge Wren was >0.15 throughout 2006, indicating that the species was reliably detected throughout the breeding season, but Sedge Wren detection probability was positively associated with standardized ordinal date, suggesting that optimal detection of this species occurred toward the end of the breeding season (Figure 2.14).

At the ecological level of analysis (i.e., native and introduced grasslands), all 8 grassland species were reliably detected. Adequate detection probability estimates (i.e., $p \geq 0.10$) were obtained for all 8 grassland species, with detection probability estimates ranging from 0.12 to 0.84 (Table 2.4). Standardized ordinal date was included in the best detection model for Dickcissel (Table 2.4). The detection probability for Dickcissel varied between 2005 and 2006, but detection probability estimates were > 0.12 during both years, indicating that Dickcissel was reliably detected throughout both breeding seasons (Figure 2.15) The best detection model for Sedge Wren included grassland type, and the species tended to be more likely to be detected in native grasslands ($p = 0.83$ [SE = 0.16]) than in introduced grasslands ($p = 0.14$ [SE = 0.16]), but was reliably detected in both cover types (Figure 2.16). For the remaining 6 grassland bird species the best detection model included only the intercept (Table 2.4).

Avian density comparisons

Avian species were categorized as reliably detected (i.e., $p \geq 0.10$) or unreliably detected (i.e., p could not be estimated or $p < 0.10$) in each CREP field/grassland type.

Avian densities were reported (Table 2.3) and compared (Figures 2.17-2.24) only for those species that were reliably detected. Species that were not reliably detected were designated as present if observed in study fields/grasslands and as undetected if not observed (i.e., species that were not observed in fields/grasslands could not be assumed to be absent if detection probability estimates were <0.10; Table 2.3).

Ten grassland bird species were observed in 2005; 9 grassland species were found in native grasslands and 7 species occurred in the introduced portion of CP1 fields and in CP1 whole fields (Table 2.3). Of the 8 grassland bird species reliably detected (i.e., $p \geq 0.10$) in 2005, Grasshopper Sparrow and Sedge Wren were unique to native vegetation and no species were unique to introduced plant communities. Note that in 2005, Northern Harrier, was detected in introduced vegetation of CP1 fields but is recorded as undetected for CP1 whole fields because data were used from only 5 of the 7 CP1 study fields (i.e., data from fields where the native plant community portion did not establish successfully were omitted (Table 2.3)). In 2006, 11 grassland bird species were observed (Table 2.3); all 11 species were found in native plant communities and 8 species were seen in the introduced portion of CP1 fields and in CP1 whole fields (Table 2.3). All 8 grassland species that were reliably detected (i.e., $p \geq 0.10$) in 2006 occurred in all field/grassland types (Table 2.3).

Among adequately detected species (i.e., $p \geq 0.10$), the most abundant grassland birds were Savannah Sparrow and Bobolink (Table 2.3, Figures 2.17-2.24). In 2005, Savannah Sparrow density ranged from 0.14 – 3.14 individuals/ha in CP23 fields (median = 0.58, mean = 0.98, SE = 0.39), from 0.03 – 0.79 individuals/ha in introduced vegetation (median = 0.29, mean = 0.37, SE = 0.12), and from 0.18 – 0.95 individuals/ha in CP1 fields (median = 0.51, mean = 0.55, SE = 0.17; Table 2.3, Figure 2.17). In 2006, Savannah Sparrow density ranged from 0.55 – 5.16 individuals/ha in CP23 fields (median = 1.64, mean = 2.14, SE = 0.62), from 0.12 – 1.15 individuals/ha in introduced vegetation (median = 0.83, mean 0.94, SE = 0.32), and from 0.06 – 2.81 individuals/ha in CP1 fields (median = 1.13, mean = 1.22, SE = 0.46; Table 2.3, Figure 2.17). In 2005, Bobolink density ranged from 0 – 0.94 individuals/ha in CP23 fields (median = 0.34, mean = 0.38, SE =0.15), from 0.09 – 2.55 individuals/ha in introduced vegetation (median =1.44, mean = 1.33, SE = 0.37), and from 1.01 – 5.54 individuals/ha in CP1 fields (median = 1.46, mean = 2.28, SE = 0.93; Table 2.3, Figure 2.18). In 2006, Bobolink density ranged from 0 – 1.44 individuals/ha in CP23 fields (median = 0.18, mean = 0.60, SE = 0.26), from 0 – 1.71 individuals/ha in introduced vegetation (median = 0.44, mean 0.68, SE = 0.27), and from 0 – 0.80 individuals/ha in CP1 fields (median = 0.30, mean = 0.35, SE = 0.15; Table 2.3, Figure 2.18).

One influence on grassland bird density in each study field was the breeding territory size of each species, which is affected by a variety of factors, including habitat suitability (e.g., Lanyon 1995) and local population density (e.g., Harmeson 1974 cited from Temple 2002). For each grassland bird species, the territory size range in Michigan was estimated using information from species accounts in the Cornell Lab of Ornithology’s Birds of North America Online (BNA Online) (Lanyon 1995, Martin et al. 1995, Vickery 1996, Giudice and Ratti 2001, Herkert et al. 2001, Jones and Cornely 2002, Temple 2002, Wheelwright and Rising 2008; Table 2.5). Based on the size ranges of 1) estimated breeding territories for each grassland bird species in Michigan and 2) CREP study fields, CP23 grasslands could support from 1 – 180 grassland bird breeding territories (e.g., CP23 native study fields provided from 5 – 23 breeding territories for Grasshopper Sparrow), while CP1 whole fields could provide from 1 – 218 grassland bird breeding territories, and CP1 introduced grasslands could support from 1 – 155 grassland bird breeding territories (Table 2.5). Overall, CP23 fields provided 0- 22% fewer breeding territories than CP1 whole fields (Table 2.5). However, CP23 fields supported 0 – 25% more breeding territories than CP1 introduced grasslands (e.g., CP23 fields provided 0 – 14% more breeding territories for Eastern Meadowlark than did CP1 introduced grasslands) (Table 2.5).

Avian density comparisons were conducted for species that were reliably detected (i.e., $p \geq 0.10$): Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow. Boxplots were used to compare avian densities in 1) CP23 and CP1 fields and in 2) CP23 fields and the introduced vegetation of CP1 fields (Figures 2.17-2.24).

In 2005, Bobolink density was lower in CP23 fields than in CP1 introduced vegetation or in CP1 fields, and no differences in Bobolink density were observed in 2006 (see above for range, median, and mean information; Figure 2.18). No differences between Dickcissel density in CP23 native fields and CP1 introduced vegetation or in CP23 and CP1 fields were observed in 2005 or 2006 (Figure 2.19). Eastern Meadowlark density also did not differ among field or grassland types during both years of the study (Figure 2.20). In contrast, Grasshopper Sparrow density was higher in CP23 fields than in introduced vegetation or in CP1 fields in 2005 and 2006 (Figure 2.21). In 2005, Grasshopper Sparrow density ranged from 0.01 – 0.14 individuals/ha in CP23 fields (median = 0.04, mean = 0.05, SE =0.02, and was 0 in CP1 native vegetation and CP1 whole fields (Table 2.3, Figure 2.21). In 2006, Grasshopper Sparrow density ranged from 0 – 0.16 individuals/ha in CP23 fields (median = 0.08, mean = 0.08, SE = 0.03), from 0 – 0.03 individuals/ha in introduced vegetation (median = 0, mean 0.01, SE = 0.01), and from 0 – 0.02 individuals/ha in CP1 fields (median = 0, mean = 0.01, SE = 0.01; Table 2.3, Figure 2.21). Ring-necked Pheasant density was higher in CP23 fields than in introduced vegetation in 2005, and no differences in Ring-necked Pheasant density were observed in 2006 (Figure 2.22). In 2005, Ring-necked Pheasant density ranged from 0.02 – 0.73 individuals/ha in CP23 fields (median = 0.06, mean = 0.15, SE =0.10) and from 0 – 0.12 individuals/ha in introduced vegetation (median =0.01, mean = 0.03, SE = 0.02; Table 2.3, Figure 2.22). In 2005 and 2006, Savannah Sparrow density was higher in CP23 fields than in introduced vegetation, and in 2006, Savannah Sparrow density was also higher in CP23 fields than in CP1 fields (see above for range, median, and mean information; Figure 2.17). Sedge Wren density was slightly higher in CP23 fields than CP1 fields in 2006 (Figure 2.23). In 2006, Sedge Wren density ranged from 0 – 0.14 individuals/ha in CP23 fields (median = 0.06, mean = 0.07, SE = 0.02) and from 0 – 0.18 individuals/ha in CP1 fields (median = 0, mean = 0.03, SE = 0.02; Table 2.3, Figure 2.23). In 2005, Vesper Sparrow density was higher in CP23 fields than introduced vegetation or in CP1 fields, and no differences in Vesper Sparrow density were observed in 2006 (Figure 2.24). In 2005, Vesper Sparrow density ranged from 0 – 0.18 individuals/ha in CP23 fields (median = 0.06, mean = 0.07, SE =0.03), from 0 – 0.04 individuals/ha in introduced vegetation (median = 0, mean 0.01, SE = 0.01), and from 0 – 0.04 individuals/ha in CP1 fields (median = 0.01, mean = 0.02, SE = 0.01; Table 2.3, Figure 2.24).

DISCUSSION

Vegetation sampling of CP23 fields and the introduced portion of CP1 fields resulted in the observation of 73 plant species, 32 of which were unique to CP23 plant communities, including all native grass species and a majority of native forbs. Plant species richness was greatest in CP23 fields, which contained over 1.5 times more plant species than the introduced vegetation of CP1 fields (i.e., 60 spp. vs. 37 spp.). During the avian breeding season, native plant communities tended to have more bare ground and less total plant canopy cover than introduced vegetation. Native and introduced plant communities also exhibited different patterns of change in the ratio of grasses and forbs within the plant canopy throughout the avian breeding season. Warm-season native grasses were less abundant than forbs in CP23 fields in early summer, but by mid-summer grasses and forbs were equally abundant, and by late summer grasses were more abundant than forbs. Cool-season introduced grasses were equally abundant in CP1 introduced plant communities throughout the summer. Differences between native and introduced plant communities may be most important for grassland birds during breeding territory selection, and early summer differences in the vegetation structure of CP23 native and CP1 introduced vegetation included a larger proportion of litter cover and more standing dead vegetation in CP23 native grasslands. The larger amount of standing dead vegetation in CP23 fields in early summer indicates that the native plant communities in Michigan CREP grasslands are providing more desirable winter habitat for overwintering wildlife such as Ring-necked Pheasant than are the introduced portions of CP1 fields (USDA 2000a, 2000b).

Avian surveys of CP23 and CP1 CREP grasslands in Michigan's Saginaw Bay watershed resulted in the observation of 34 species, including 11 grassland associates (Sauer et al. 1995). Survey results indicate that CREP grasslands provide habitat for a majority of grassland bird species found in Michigan (i.e., 11 of 15 species documented to currently occur in Michigan; BBAE 2010). Grassland bird species observed in CREP study fields included all 4 species found to have significantly declining population trends ($p < 0.10$) in Michigan during the most recent analysis of Breeding Bird Survey (BBS) data (i.e., Bobolink, Eastern Meadowlark, Ring-necked Pheasant; Sauer et al. 2008), as well as 1 state endangered species (i.e., Henslow's Sparrow), and 2 species of state special concern (i.e., Dickcissel and Grasshopper Sparrow), suggesting that CREP grasslands provide habitat for the most imperiled grassland species in Michigan.

Eight grassland bird species were detected frequently enough (i.e., in ≥10% of surveys) to allow detection probability modeling: Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow. Detection estimates were adequate for all 8 grassland bird species and were generally high (i.e., $p > 0.25$ for 6 species in CP1 and CP23 fields and/or CP23 fields and CP1 introduced vegetation), indicating that the survey technique used was an effective methodology for documenting grassland bird occurrence in the study system, and providing a sound basis for reliable density comparisons.

For 5 out of 8 grassland bird species included in density comparisons, higher densities were associated with CP23 fields than with CP1 whole fields and/or the introduced portion of CP1 fields during at least one year of the study (i.e., Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow).

The densities of these 5 grassland bird species were higher in CP23 fields than in CP1 whole fields even though CP1 fields could support 0 - 18% more grassland bird breeding territories for these species than CP23 fields, suggesting that CP1 fields containing introduced and native grasslands are not as attractive to several breeding grassland bird species as similar-sized or smaller CP23 fields containing only native grasslands. The densities of 2 species, Grasshopper Sparrow and Savannah Sparrow, showed particularly consistent trends, with higher densities in CP23 fields than in the introduced portion of CP1 fields and/or CP1whole fields during both years of the study. However, one species of management priority, Bobolink, was more abundant in the introduced vegetation of CP1 fields during the first year of the study, demonstrating the diverse response of grassland birds to habitat features. This observation provides support for using species-specific habitat preference information when managing grassland bird species (e.g., Murray et al. 2008).

I attempted to quantify the productivity of grassland bird species found in the native and introduced CREP grasslands of this study (Appendix 2.3). However, the intensive nest survey techniques used were unsuccessful in my study system (i.e., a total of 3 active grassland bird nests were located in all fourteen 16.8 ha survey areas), largely due to the highly cryptic nature of grassland bird nests (Appendix 2.3). For example, even when a nest was assumed to be in a particular area due to adult bird behaviors such as food transport, nests were generally impossible to locate below the grass canopy without severely degrading the vegetation structure and litter layer of the nest vicinity. There is little doubt that density information is most informative about habitat quality when coupled with information about reproduction and survival of the species of interest.

Van Horne (1983) raised consciousness that habitat quality as a measure of the importance of a habitat in maintaining the population of a species "should be defined in terms of the survival and production characteristics, as well as the density, of the species occupying that habitat." However, recently other researchers have provided evidence that density alone can be a reliable indicator of habitat quality (Bock and Jones 2004, Perot and Villard 2009). Bock and Jones (2004) surveyed 109 published studies of 67 European or North American bird species and found that habitats with higher densities displayed greater avian reproduction rates (i.e., greater recruitment per capita and per unit of land area) in the majority of cases (i.e., ≥72%. Bock and Jones (2004) concluded that "in most cases, density will be a reliable indicator of habitat quality, and bird-count data will be an appropriate basis from which to make land-management and conservation decisions."

Other researchers have also found differences in grassland bird abundances between native and introduced grasslands. In a comparison of avian abundance, richness, and productivity on privately owned warm- and cool-season grass farm fields in Pennsylvania, Giuliano and Daves (2002) found that warm-season grasslands supported a greater abundance and richness of birds, including Grasshopper Sparrow, Vesper Sparrow, and other species of conservation priority.

In contrast, other studies in similar systems found no differences between grassland bird abundances in native and introduced vegetation or found greater grassland bird abundances in introduced vegetation (e.g., McCoy et al. 2001, Wentworth 2010). In these studies, the structure and composition of native grasslands differed, often widely, from the native fields I examined, tending to be less diverse, have a denser canopy, and/or to be poorly established or switchgrass (*Panicum virgatum)* dominated (i.e., dense stands of switchgrass have been found to be poor grassland bird habitat; Norment et al. 1999). Wentworth et al. (2010) demonstrated that in Pennsylvania CREP fields, grassland obligate species were rare but were found in greater densities on fields of cool-season grasses than in fields of CP2 or mixed grasses. However, CP2 fields in the study were not yet completely established and some were actually dominated by cool-season grasses or other vegetation types (Wentworth et al. 2010). In a study of Conservation Reserve Program (CRP) grasslands in Missouri, McCoy et al. (2001) found that more grassland bird species occurred at higher abundances in introduced/CP1 fields than in native/CP2 fields, which were monocultures of switchgrass and were less diverse and denser than the CP1 fields in their study. The opposite was true of CP23 CREP fields in my study; CP23 fields were more diverse and less dense (i.e., with sparser canopy cover and more visible bare ground) than introduced vegetation of CP1 fields.

Differences in the findings of this study and other investigations underscore the importance of evaluating the effects of CREP and similar programs on grassland birds at local scales (e.g., ecoregions, or major watersheds). It is important to note that the consistency of results from studies in different regions may be affected by differences in grassland composition, age or successional stage, and structure.

Management recommendations

The results of this study indicate that Michigan CREP grasslands, especially native CP23 grasslands, provide grassland bird habitat, a critical resource for an avian guild whose continued decline has been identified as an impending conservation crisis (Knopf 1996, Brennan and Kuvlesky 2005). Density comparisons indicate that CP23 grassland plantings supported greater densities of more grassland bird species than the introduced vegetation of CP1 fields or CP1 whole fields. Despite the greater cost of native seed mixes (i.e., $16.00/pound v. $1.90/pound for introduced vegetation in 2005 in Tuscola County, Michigan) and the specialized planting techniques required for successful native vegetation establishment (e.g., use of native seed drills for planting the fluffy seed of some native grass species), I recommend that CP23 whole-field native plantings be utilized in favor of CP1 plantings within CREP in the Saginaw Bay area of Michigan. One species of management priority, Bobolink, was more abundant in the introduced vegetation of CP1 fields during the first year of this study. Because these results demonstrate a diverse response of grassland birds to CREP grasslands, I further recommend that the effect of similar grasslands be evaluated for individual species within the grassland bird guild rather than for grassland birds as a group and that species-specific information be used in developing grassland bird conservation strategies.

Maintaining and/or increasing the area of CREP grasslands, especially native grasslands, should be a grassland bird conservation strategy in Michigan. Threats to the persistence and/or increase of farmland set-aside grasslands exist, and include reconversion to corn or other crop. Recently, increased demand for biofuels has resulted in a trend toward reconversion of agricultural set-aside lands into crop production (Secchi and Babcock 2007, Searchinger et al. 2008, cited from Fargione et al. 2009), a trend that is likely to continue due to large, federally mandated increases in biofuel production (i.e., >700% over 2006 production by 2022) (Fargione et al. 2009). Fargione et al. (2009) suggested minimizing the negative effects of increased biofuel demand on wildlife by utilizing wildlife-compatible biomass sources that do not require use of additional production lands (e.g., wastes, cover crops, algae), and by maximizing the area of perennial grasslands, including diverse native prairie plantings, to produce biofuel in ways that are compatible with wildlife (e.g., harvesting biomass only after completion of the avian breeding season). Landowner participation is required for the maintenance of Michigan CREP lands. Other research has examined the response of landowners to land retirement incentives and found a positive response to CREP farmland set-aside incentives and that an increase in the incentive payment made at the time of enrollment generates a greater increase in enrollment than a small raise in the annual payments made over the term of the set-aside contract (Suter et al. 2008). I recommend that program administrators consider using this method of promoting the retention and increase of CREP lands in order to ensure the continued presence of grassland bird habitat the program provides.

APPENDIX 2.1: CHAPTER 2 TABLES AND FIGURES

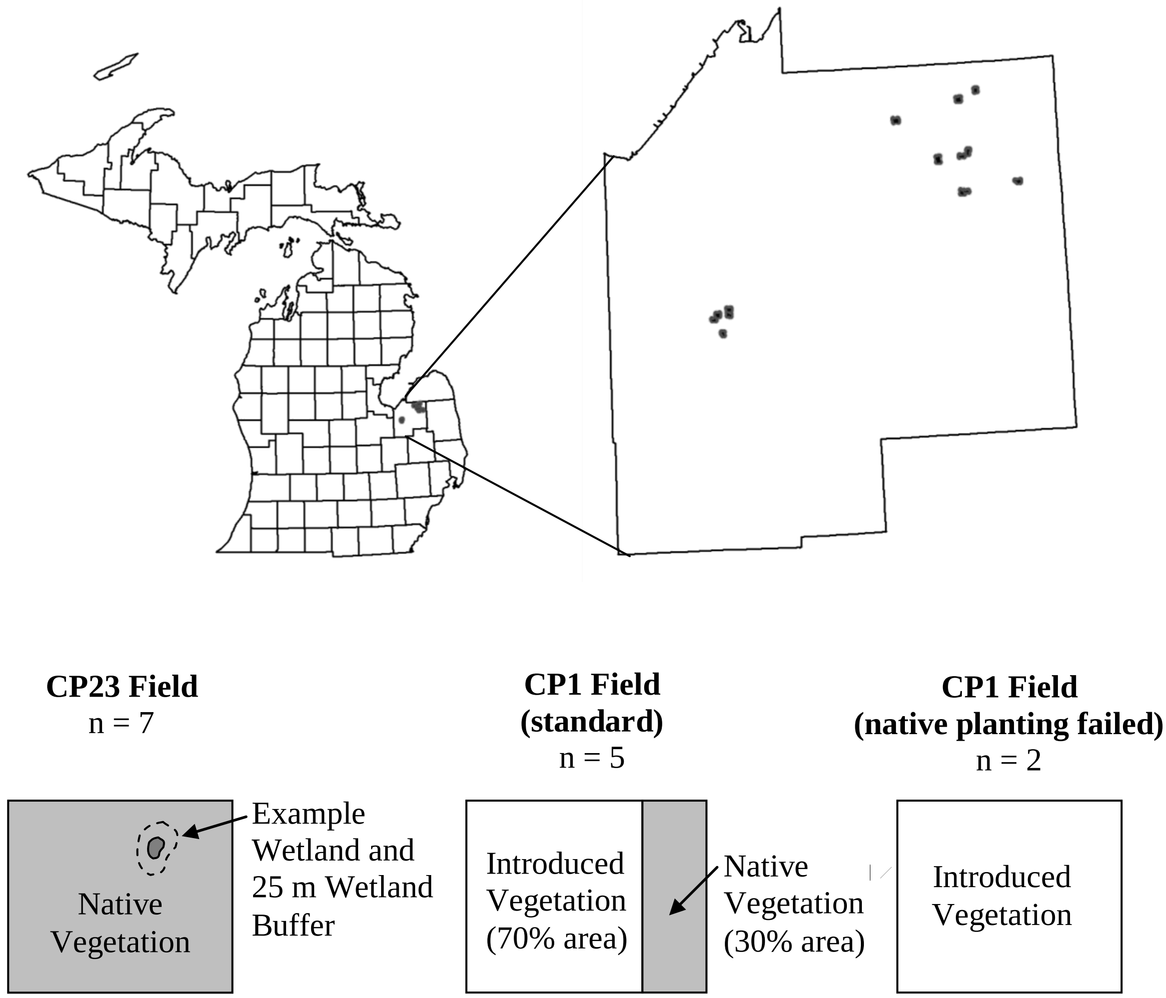

Fig. 2.1. Location, type, and number of Conservation Reserve Enhancement Program (CREP) study fields within Tuscola County, Saginaw Bay watershed, Michigan, 2005 and 2006. Wetlands and wetland buffers were excluded from avian surveys and vegetation sampling. The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). The introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha), and CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha).

Table 2.1. Plant species and proportions of species included in the seed mixtures used to establish the native and introduced vegetation communities of Conservation Reserve Enhancement Program (CREP) grasslands of Michigan's Saginaw Bay watershed, 2002.

Common Name	Scientific Name	% Total Seed Mixture	% Grass, Wildflower, or Legume Mixture
***CP23 Fields and Native Section of CP1 Fields** - planted at 6.5 pounds/acre (16.1 #/ha)*			
Native Grasses		92.3	
Big Bluestem*	*Andropogon gerardii*		33.3
Indiangrass*	*Sorghastrum nutans*		33.3
Little Bluestem*	*Andropogon scoparius*		33.3
Plateau® Tolerant Wildflower Mixture		7.7	
Black Eyed Susan*	*Rudbeckia hirta*		10.0
Blanket Flower, Perennial	*Gaillardia aristata*		2.0
Catchfly	*Silene armeria*		2.0
Clover, Crimson	*Trifolim incarnatum*		10.0
Coneflower, Prairie, 'Mexican Hat'	*Ratibida columnifera*		3.0
Coneflower, Purple*	*Echinacea purpurea*		2.0
Coneflower, Yellow*	*Ratibida columnifera*		5.0
Coreopsis, Lance Leaf*	*Coreopsis lanceolata*		10.0
Coreopsis, Plains	*Coreopsis tinctoria*		3.0
Cosmos, 'Sensation' Mix	*Cosmos bipinnatus*		5.0
Cosmos, Sulphur	*Cosmos sulphureus*		5.0
Daisy, Shasta, 'Alaska'	*Chrysanthemum x superbum*		4.0
Dame's Rocket	*Hesperis matronalis*		5.0
Flax, Perennial	*Linum perenne lewisii*		10.0
Lupine, Perennial*	*Lupinus perennis*		4.0
Poppy, California	*Eschscholzia californica*		5.0
Poppy, Corn, 'Shirley Single' Mix	*Papaver rhoeas*		5.0
Wallflower, Siberian	*Erysiumum x marshallii*		10.0
***Introduced Section of CP1 Fields** - planted at 11 pounds per acre (27.2 #/ha)*			
Introduced Grasses		45.5	
Orchardgrass	*Dactlyis glomerata*		50.0
Timothy	*Phleum pratense*		50.0
Introduced Legumes		54.5	
Alfalfa	*Medicago sativa*		50.0
Red Clover	*Trifolium pratense*		50.0

**Michigan native species (Voss 1985, Voss 1996, USDA NRCS 2011)*

Table 2.2. Plant species observed in native and introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006.

		Plant Community Type	
Common Name	Scientific Name	Native	Introduced
Alfalfa	*Medicago sativa*	n	i
Alsike Clover	*Trifolium hybridum*	n	i
Annual Ryegrass	*Lolium perenne*	n	i
Big Bluestem	*Andropogon gerardii*	n	
Black Medick	*Medicago lupulina*	n	i
Black-eyed Susan	*Rudbeckia hirta*	n	i
Blanket Flower	*Gaillardia aristata*	n	
Burdock	*Arctium spp.*		i
Canada Thistle	*Cirsium arvense*	n	i
Catchfly	*Silene spp.*	n	
Cattail	*Typha spp.*	n	
Chickweed	*Stellaria spp.*	n	
Chicory	*Cichorium intybus*		i
Cinquefoil	*Potentilla spp.*	n	
Common Mullein	*Verbascum thapsus*	n	
Common Sow Thistle	*Sonchus oleraceus*	n	i
Corn Poppy	*Papaver rhoeas*	n	
Cottonwood	*Populus spp.*	n	
Curly Dock	*Rumex crispus*	n	
Daisy Fleabane	*Erigeron annuus*	n	i
Daisy spp.	*Asteraceae spp.*	n	
Dame's Rocket	*Herperis matronalis*		i
Dandelion	*Taraxacum spp.*	n	i
Downy Brome (Cheat Grass)	*Bromus tectorum*	n	i
Fescue	*Festuca spp.*	n	i
Field Bindweed	*Convolvulus arvensis*	n	i
Fleabane	*Erigeron spp.*	n	
Forked Catchfly	*Silene dichotoma*	n	
Goat's-beard	*Tragopogon spp.*	n	
Goldenrod	*Solidago spp.*	n	i
Hawkweed	*Hieracium spp.*	n	
Horseweed	*Conyza canadensis*	n	
Indiangrass	*Sorghastrum nutans*	n	
Kentucky Bluegrass	*Poa pratensis*	n	i
Lance-leaved Coreopsis	*Coreopsis lanceolata*	n	

Table 2.2. Continued

		Plant Community Type	
Common Name	Scientific Name	Native	Introduced
Least Hop Clover	*Trifolium dubium*	n	i
Lettuce	*Lactuca spp.*	n	i
Lichen	Lichen *spp.*	n	
Little Bluestem	*Andropogon scoparius*	n	
Meadowgrass	*Poa spp.*		i
Milkweed	*Asclepias spp.*	n	i
Morning Glory	*Ipomoea spp.*		i
Moss	Bryophyta *spp.*	n	i
Musk Thistle	*Carduus nutans*	n	
Mustard	*Cruciferae spp.*		i
Orchard Grass	*Dactlyis glomerata*	n	i
Perennial Blue Flax	*Linum perenne lewisii*	n	
Perennial Ryegrass	*Lolium perenne*	n	
Pigweed (Lamb's-quarters)	*Chenopodium album*	n	
Plantain	*Plantago spp.*	n	i
Prickly Lettuce	*Lactuca serriola*	n	i
Queen Anne's Lace	*Daucus carota*	n	i
Ragweed	*Ambrosia spp.*		i
Red Clover	*Trifolium pratense*	n	i
Redtop	*Agrostis gigantea*	n	i
Rush	*Juncus spp.*	n	
Shasta Daisy 'Alaska'	*Chrysanthemum x superbum*	n	
Smooth Brome	*Bromus inermis*	n	i
Spiny-leaf Sow Thistle	*Sonchus asper*		i
Switchgrass	*Panicum virgatum*	n	
Thistle	*Cirsium spp.*	n	i
Timothy	*Phleum pratense*	n	i
Wheat	*Triticum aestivum*		i
White Clover	*Trifolium repens*	n	
White Sweet Clover	*Melilotus alba*	n	i
Willow	*Salix spp.*	n	
Wood Sorrel	*Oxalis spp.*	n	
Yellow Prairie Coneflower	*Ratibida columnifera*	n	
Yellow Sweet Clover	*Melilotus officinalis*	n	
Number of Species Present		60	37

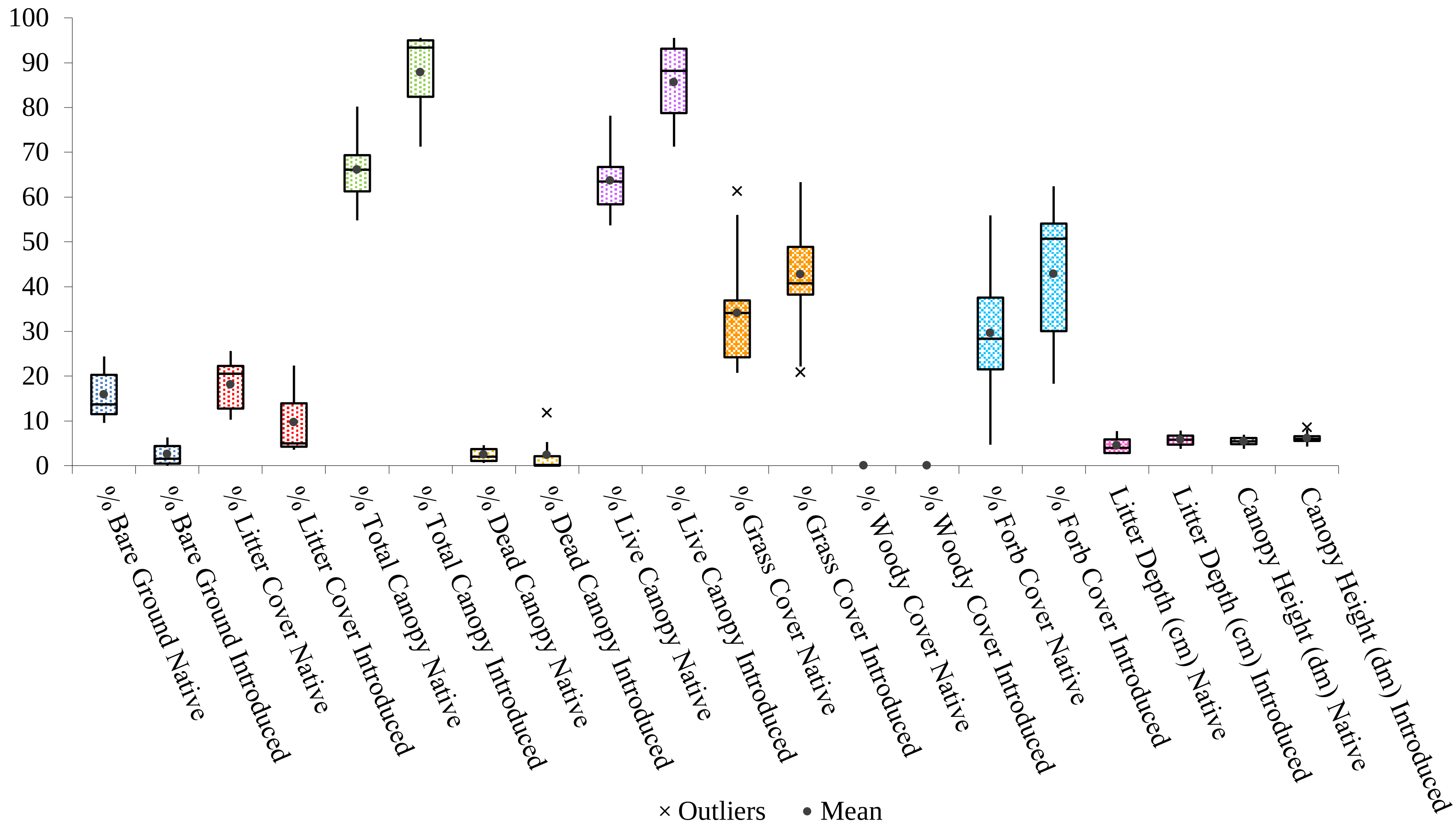

Fig. 2.2. 2005 mid-summer (15 June – 14 July) vegetation variables of native and introduced vegetation of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan. For interpretation of the references to color in this and all other figures, the reader is referred to the electronic version of this dissertation.

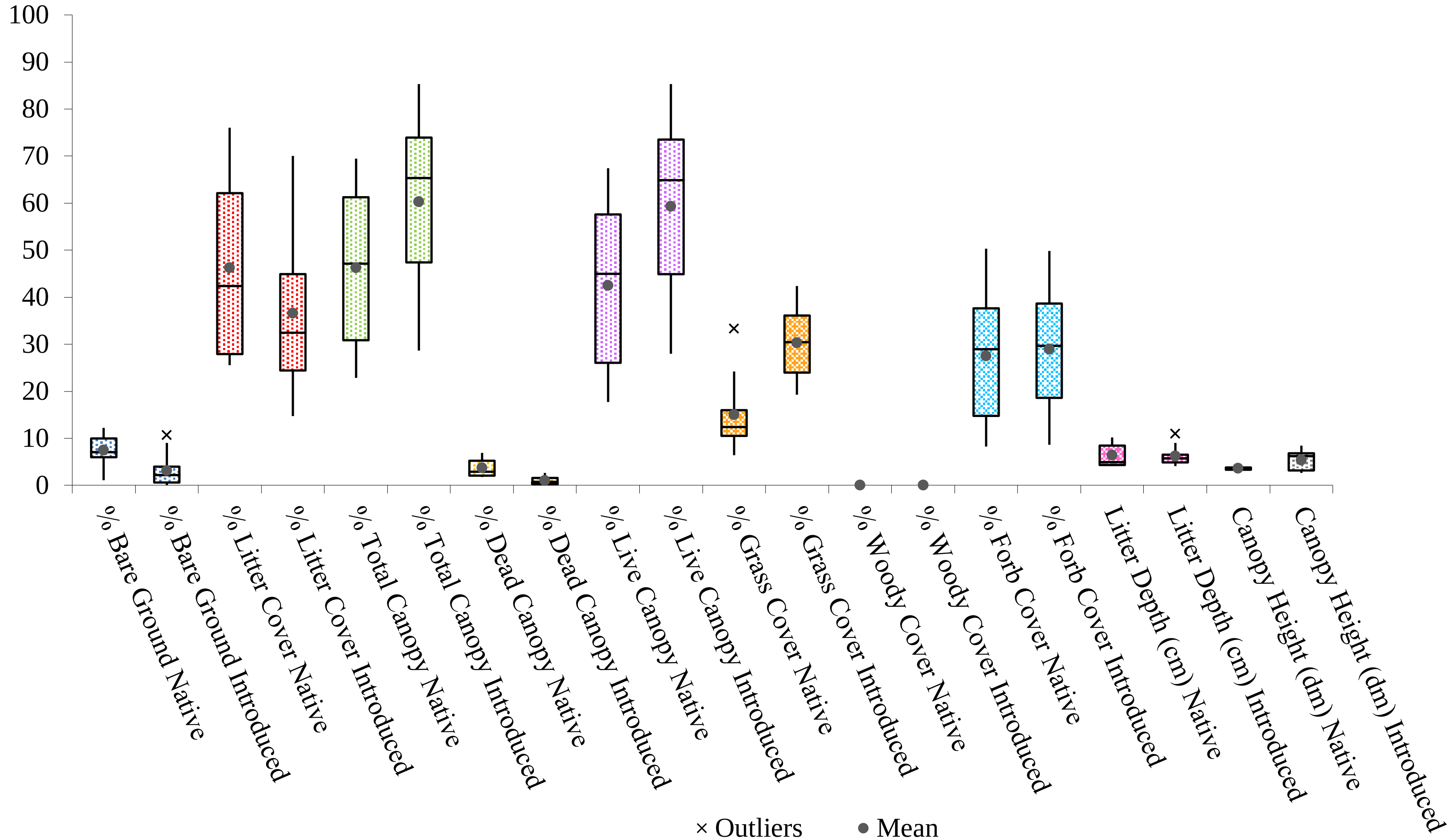

Fig. 2.3. 2006 early summer (15 May - 14 June) vegetation variables of native and introduced vegetation of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

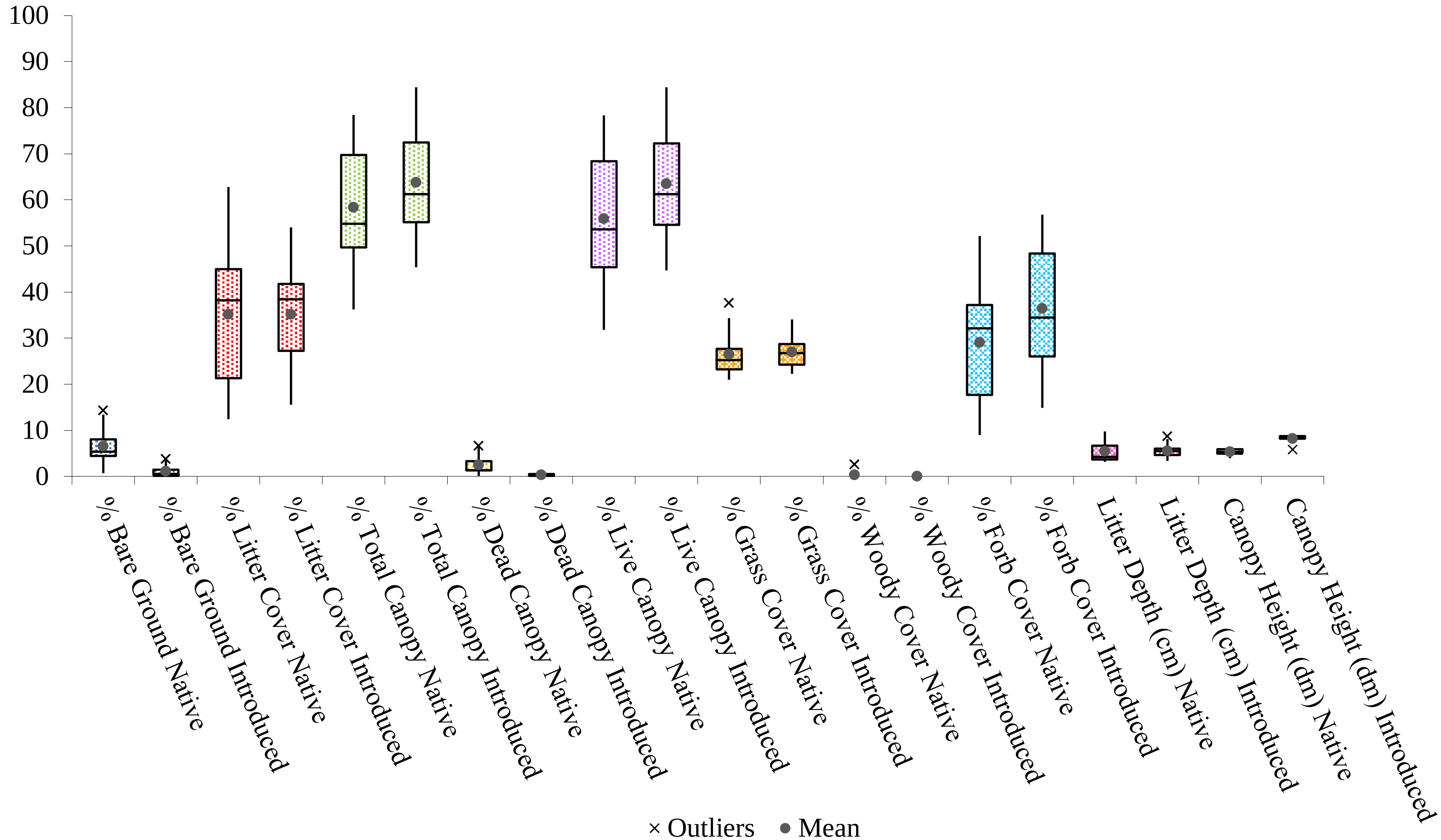

Fig. 2.4. 2006 mid- summer (15 June - 14 July) vegetation variables of native and introduced vegetation of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

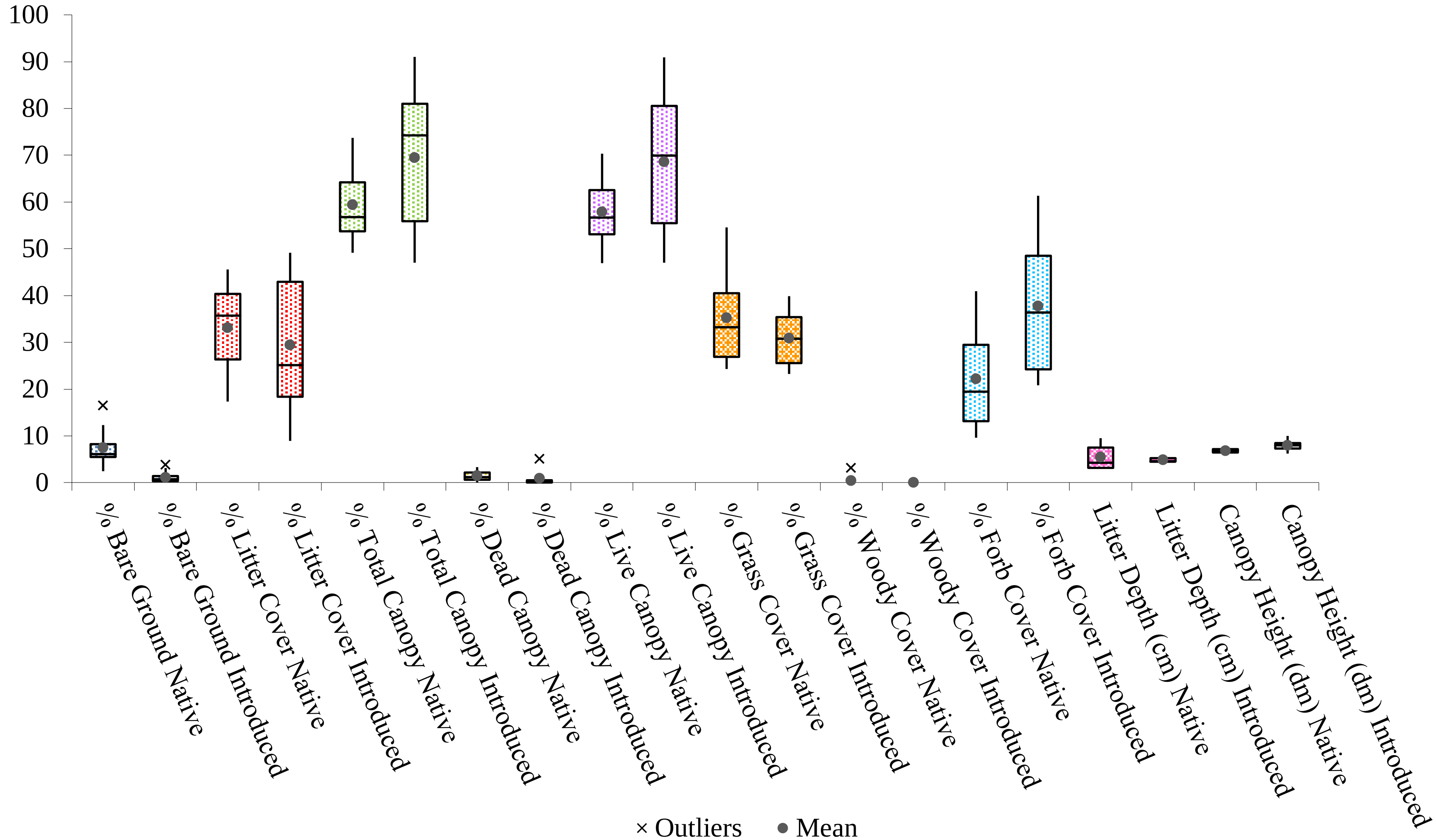

Fig. 2.5. 2006 late summer (15 July - 14 August) vegetation variables of native and introduced vegetation of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

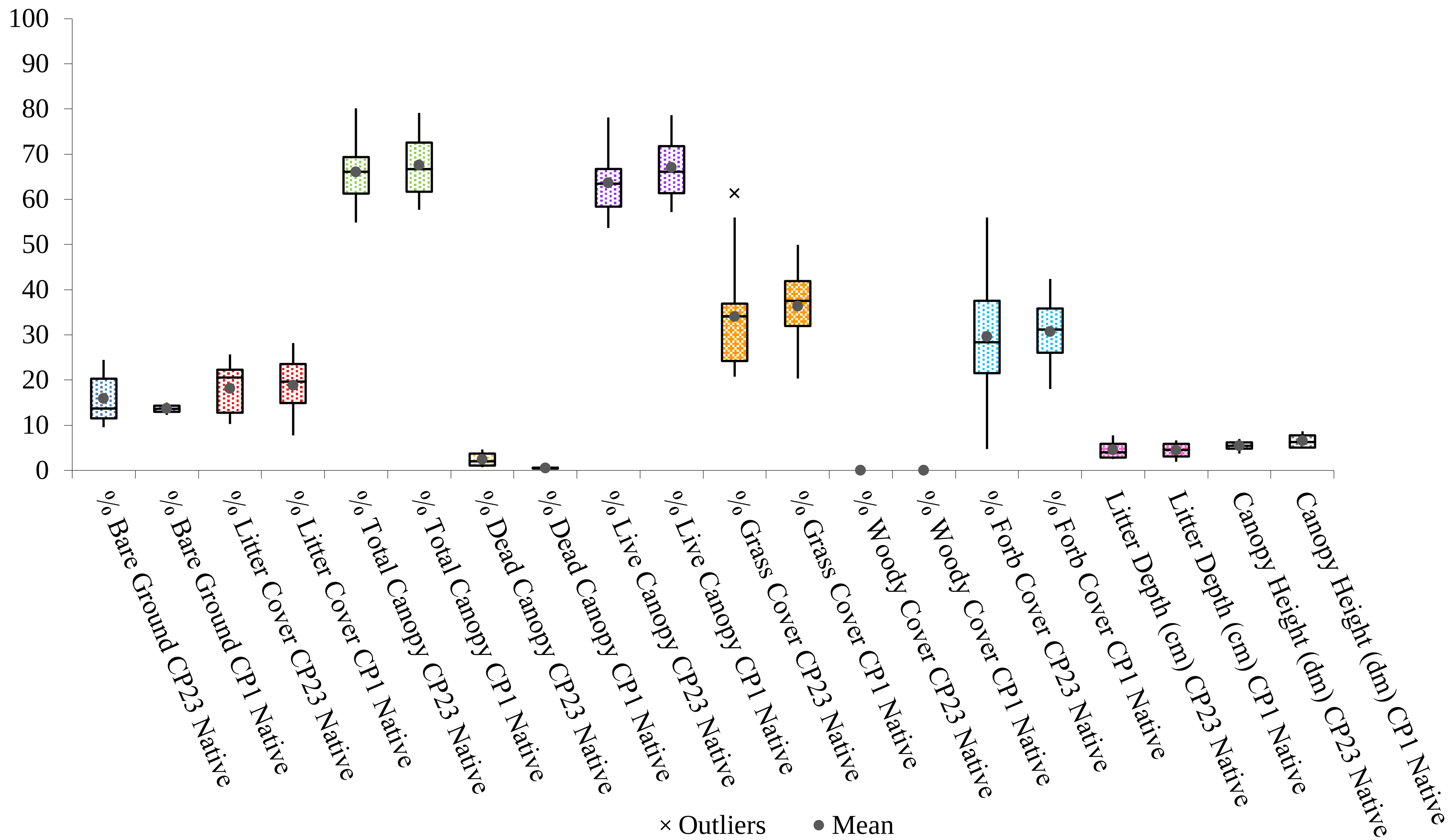

Fig. 2.6. Native vegetation variables of CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, mid-summer (15 June – 14 July) 2005.

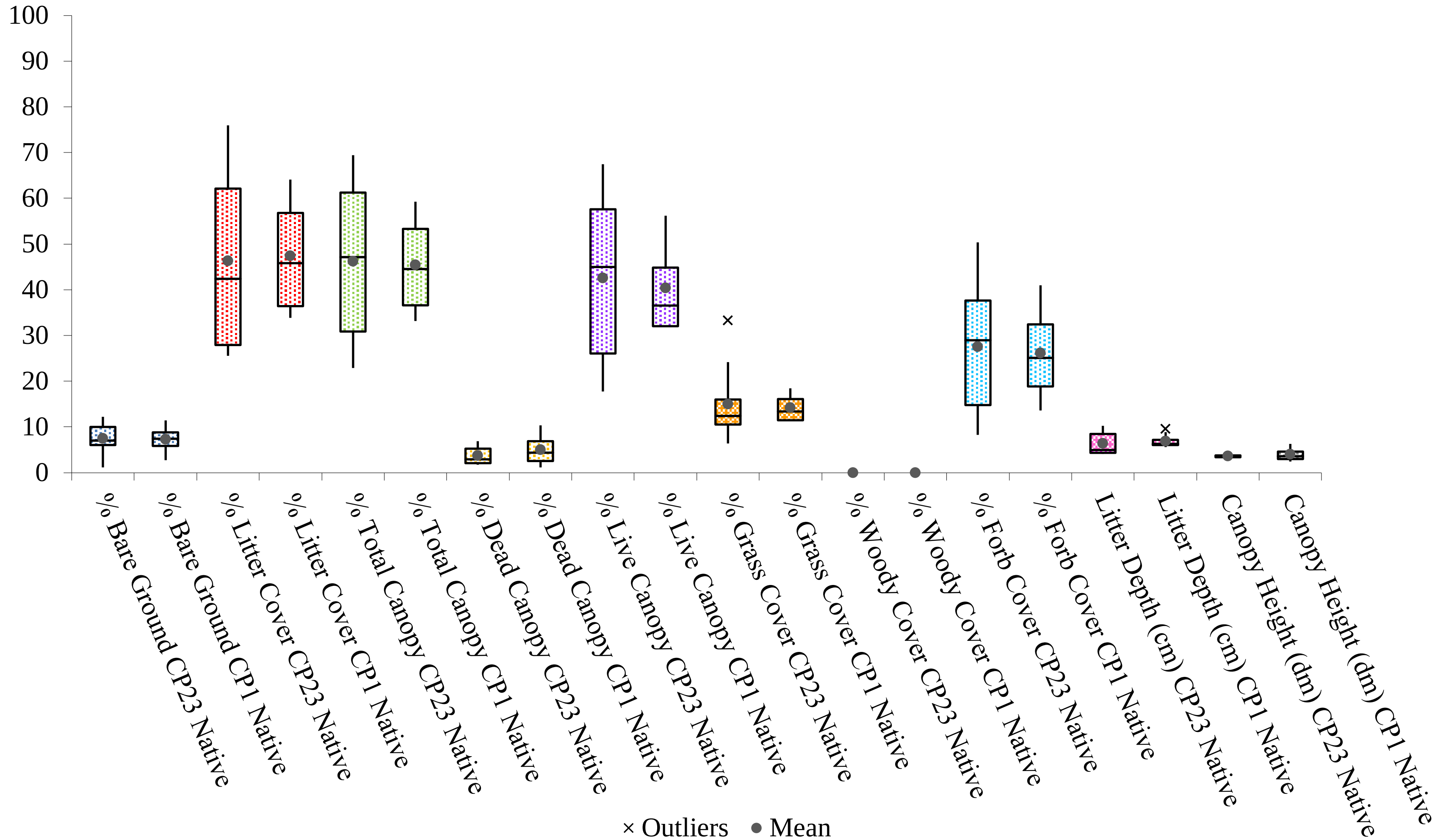

Fig. 2.7. Native vegetation variables of CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, early summer (15 May – 14 June) 2006.

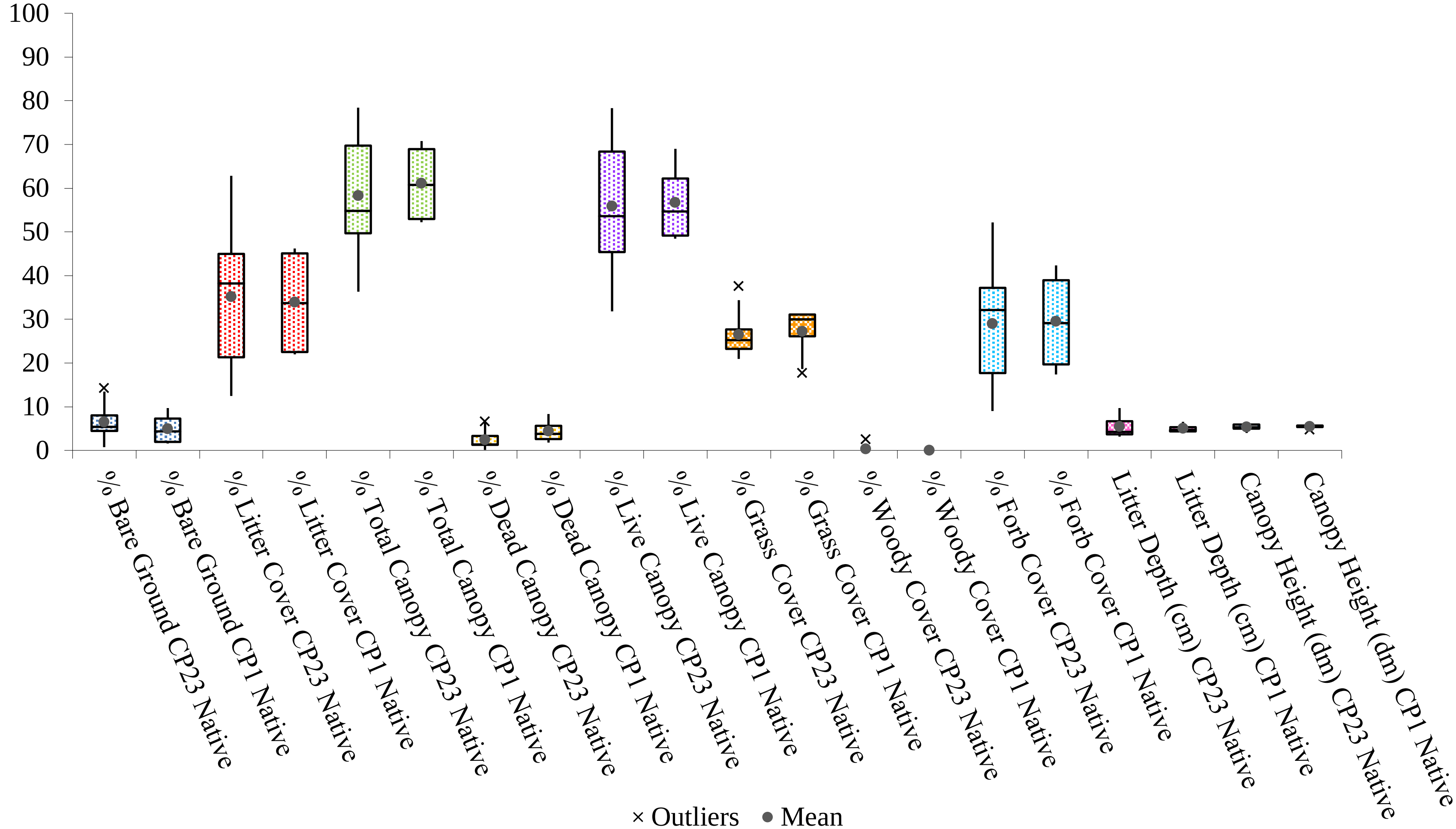

Fig. 2.8. Native vegetation variables of CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, mid-summer (15 June – 14 July) 2006.

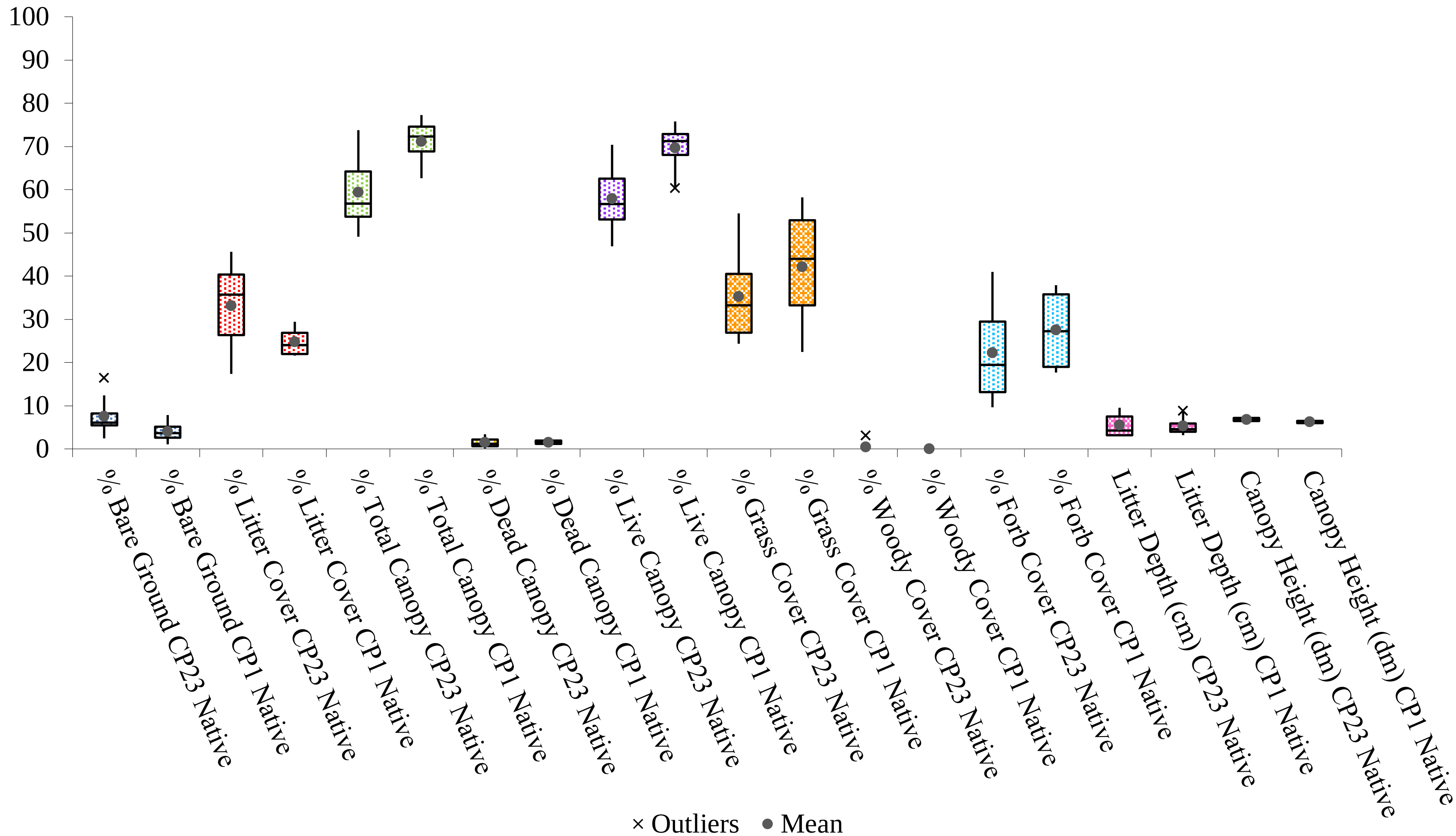

Fig. 2.9. Native vegetation variables of CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, late summer (15 July – 14 August) 2006.

Table 2.3. Avian species observed in CP23 native fields, the introduced portion of CP1 fields, and CP1 whole fields, Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Avian densities (# individuals /ha [SE]) are given for species with detection probability estimates ≥0.10. Species observed too rarely to support detection probability modeling (i.e., observed in < 10% of surveys) or with detection probability estimates <0.10 are noted as present when observed in at least one survey and as undetected when unobserved.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native/ CP23	Introduced	CP1
Grassland Specialist Species							
Bobolink [A]	*Dolichonyx oryzivorus*	0.38 [0.15]	1.33 [0.37]	2.28 [0.93]	0.60 [0.26]	0.68 [0.27]	0.35 [0.15]
Dickcissel [AB]	*Spiza americana*	0.04 [0.03]	0.08 [0.07]	0.05 [0.06]	0.03 [0.01]	0.05 [0.03]	0.05 [0.03]
Eastern Meadowlark	*Sturnella magna*	0.12 [0.09]	0.004 [0.004]	0.003 [0.003]	0.06 [0.04]	0.01 [0.01]	0.03 [0.02]
Grasshopper Sparrow [AB]	*Ammodramus savannarum*	0.05 [0.02]	0.00 [0.00]	0.00 [0.00]	0.08 [0.03]	0.01 [0.01]	0.01 [0.01]
Henslow's Sparrow [AC]	*Ammodramus henslowii*	present	undetected	undetected	present	undetected	undetected
Horned Lark	*Eremophila alpestris*	undetected	undetected	undetected	present	undetected	undetected
Northern Harrier [A]	*Circus cyaneus*	undetected	present	undetected	present	undetected	undetected
Ring-necked Pheasant	*Phasianus colchicus*	0.15 [0.10]	0.03 [0.02]	0.07 [0.04]	0.12 [0.07]	0.13 [0.08]	0.07 [0.06]
Savannah Sparrow	*Passerculus sandwichensis*	0.98 [0.39]	0.37 [0.12]	0.55 [0.17]	2.14 [0.62]	0.94 [0.32]	1.22 [0.46]
Sedge Wren [A]	*Cistothorus platensis*	0.01 [0.01]	0.00 [0.00]	present	0.07 [0.02]	0.06 [0.03]	0.03 [0.02]
Vesper Sparrow	*Pooecetes gramineus*	0.07 [0.03]	0.01 [0.01]	0.02 [0.01]	0.06 [0.01]	0.02 [0.01]	0.06 [0.04]

Table 2.3 Continued.

Common Name	Scientific Name	2005 Native/ CP23	2005 Introduced	2005 CP1	2006 Native	2006 Introduced	2006 CP1
Other Species							
American Goldfinch	*Carduelis tristis*	0.33 [0.13]	0.08 [0.03]	0.41 [0.32]	0.24 [0.08]	0.26 [0.07]	0.29 [0.08]
American Robin	*Turdus migratorius*	undetected	present	undetected	undetected	present	present
American Woodcock	*Scolopax minor*	undetected	undetected	undetected	present	undetected	undetected
Barn Swallow	*Hirundo rustica*	NA*	NA*	NA*	present	present	present
Brown Thrasher	*Toxostoma rufum*	undetected	undetected	undetected	undetected	present	present
Brown-headed Cowbird	*Molothrus ater*	present	undetected	undetected	present	present	present
Chipping Sparrow	*Spizella passerina*	undetected	undetected	undetected	undetected	present	present
Clay-colored Sparrow	*Spizella pallida*	present	present	present	0.05 [0.03]	present	present
Common Yellowthroat	*Geothlypis trichas*	0.03 [0.02]	0.05 [0.04]	0.09 [0.06]	0.12 [0.04]	0.10 [0.06]	0.16 [0.12]
Eastern Kingbird	*Tyrannus tyrannus*	present	present	present	0.03 [0.01]	0.004 [0.003]	present
House Wren	*Troglodytes aedon*	present	present	undetected	undetected	undetected	undetected
Indigo Bunting	*Passerina cyanea*	present	present	present	undetected	present	present

Table 2.3 Continued.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native/ CP23	Introduced	CP1
Killdeer	*Charadrius vociferus*	present	undetected	undetected	present	undetected	undetected
Mallard	*Anas platyrhynchos*	present	undetected	undetected	present	undetected	undetected
Marsh Wren [B]	*Cistothorus palustris*	undetected	undetected	undetected	present	undetected	undetected
Mourning Dove	*Zenaida macroura*	present	present	present	present	present	present
Northern Bobwhite Quail	*Colinus virginianus*	undetected	present	present	undetected	present	undetected
Northern Flicker	*Colaptes auratus*	present	undetected	undetected	present	undetected	present
Red-winged Blackbird	*Agelaius phoeniceus*	0.64 [0.26]	2.30 [0.29]	2.18 [0.62]	0.73 [0.19]	2.97 [0.36]	2.11 [0.26]
Song Sparrow	*Melospiza melodia*	1.54 [0.35]	0.46 [0.08]	1.47 [0.47]	1.89 [0.37]	0.76 [0.14]	1.12 [0.12]
Tree Swallow	*Tachycineta bicolor*	NA*	NA*	NA*	0.01 [0.01]	present	0.06 [0.05]
White-crowned Sparrow	*Zonotrichia leucophrys*	undetected	undetected	undetected	undetected	undetected	present
Wild Turkey	*Meleagris gallopavo*	present	undetected	undetected	present	undetected	present

[A] *Michigan Priority Landbird (Knutson et al. 2001, Matteson et al. 2009)*
[B] *Michigan Species of Special Concern*
[C] *Michigan Endangered Species*
** Barn Swallow and Tree Swallow were not included in the 2005 survey.*

Table 2.4. Grassland bird species, with corresponding best-fitting detection models [with constant occupancy, $\psi(\cdot)$] and detection probability estimates, observed in 1) CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grassland fields and 2) CP23 native and CP1 introduced grassland plant communities, Saginaw Bay watershed, Michigan, 2005-2006. The lowest detection probability estimate was reported when multiple detection probability estimates were generated.

	CP23 and CP1 Whole CREP Fields		Native and Introduced Plant Communities	
Common Name	Detection Model	$p(\cdot)$ [SE]	Detection Model	$p(\cdot)$ [SE]
Bobolink	*p(Ordinal Date)*	0.63 [0.06]	*p(.)*	0.75 [0.08]
Dickcissel	*p(.)*	0.35 [0.06]	*p(Ordinal Date)*	0.12 [0.07]
Eastern Meadowlark	*p(CREP Field Type)*	0.13 [0.07]	*p(.)*	0.34 [0.18]
Grasshopper Sparrow	*p(.)*	0.31 [0.05]	*p(.)*	0.47 [0.16]
Ring-necked Pheasant	*p(CREP Field Type, Ordinal Date)*	0.17 [0.05]	*p(.)*	0.28 [0.09]
Savannah Sparrow	*p(Ordinal Date)*	0.81 [0.05]	*p(.)*	0.84 [0.04]
Sedge Wren	*p(Ordinal Date)**	0.12 [0.07]	*p(Plant Community Type)*	0.14 [0.16]
Vesper Sparrow	*p(.)*	0.21 [0.04]	*p(.)*	0.18 [0.09]

**Sedge Wren data from 2006 only*

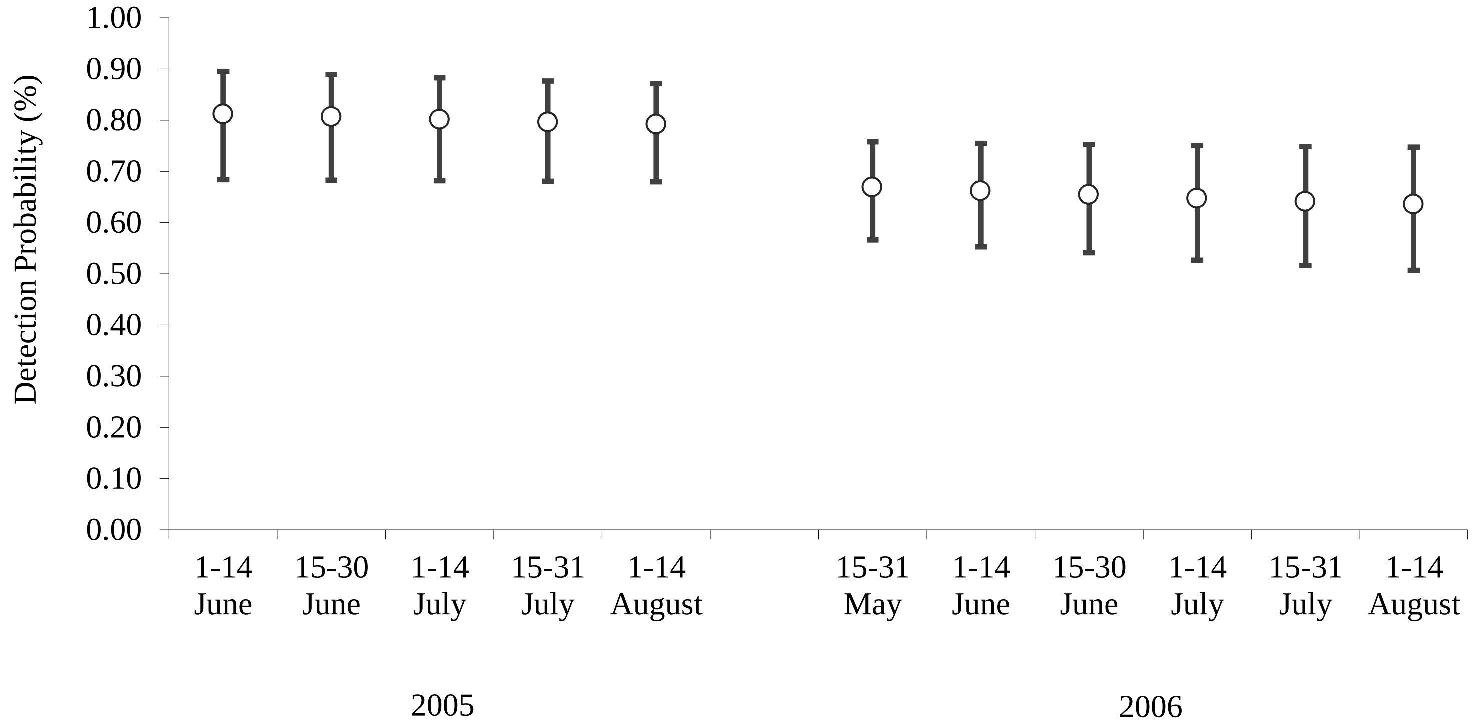

Fig. 2.10. Relationship between Bobolink detection probability and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Error bars indicate 95% confidence intervals.

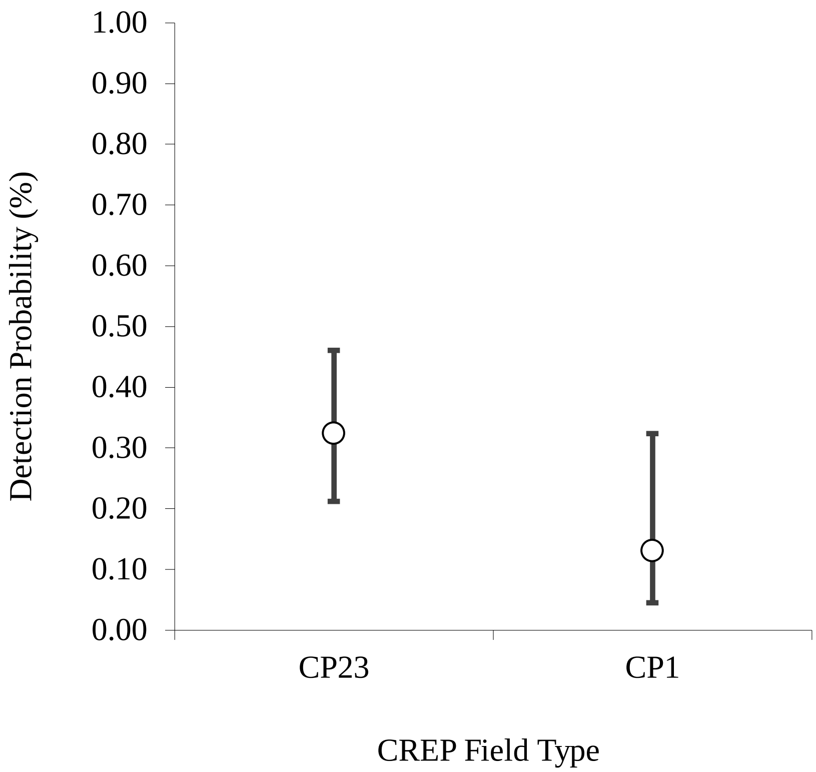

Fig. 2.11. Relationship between Eastern Meadowlark detection probability and Conservation Reserve Enhancement Program (CREP) administrative field type in CP23 and CP1 CREP grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006. Error bars indicate 95% confidence intervals.

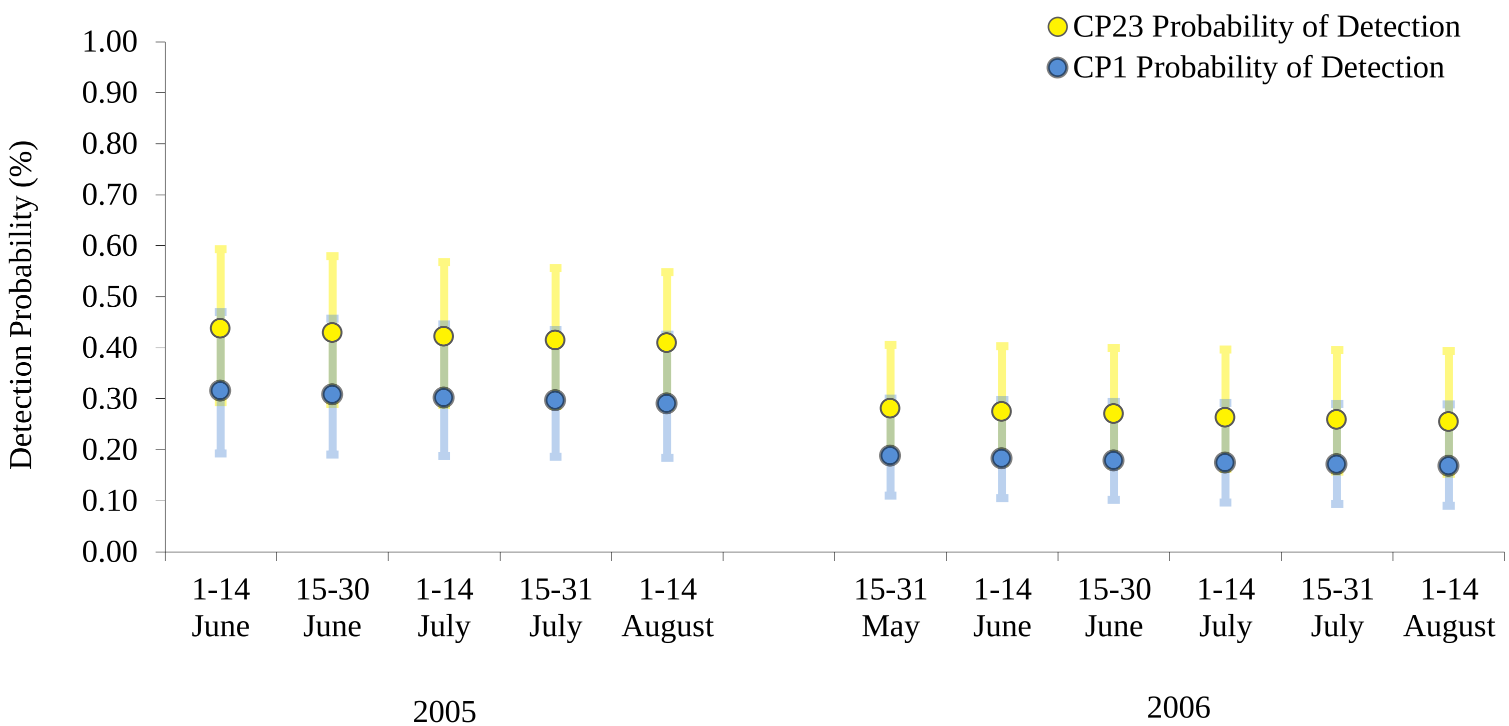

Fig. 2.12. Relationship between Ring-necked Pheasant detection probability, field type, and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006. Error bars indicate 95% confidence intervals.

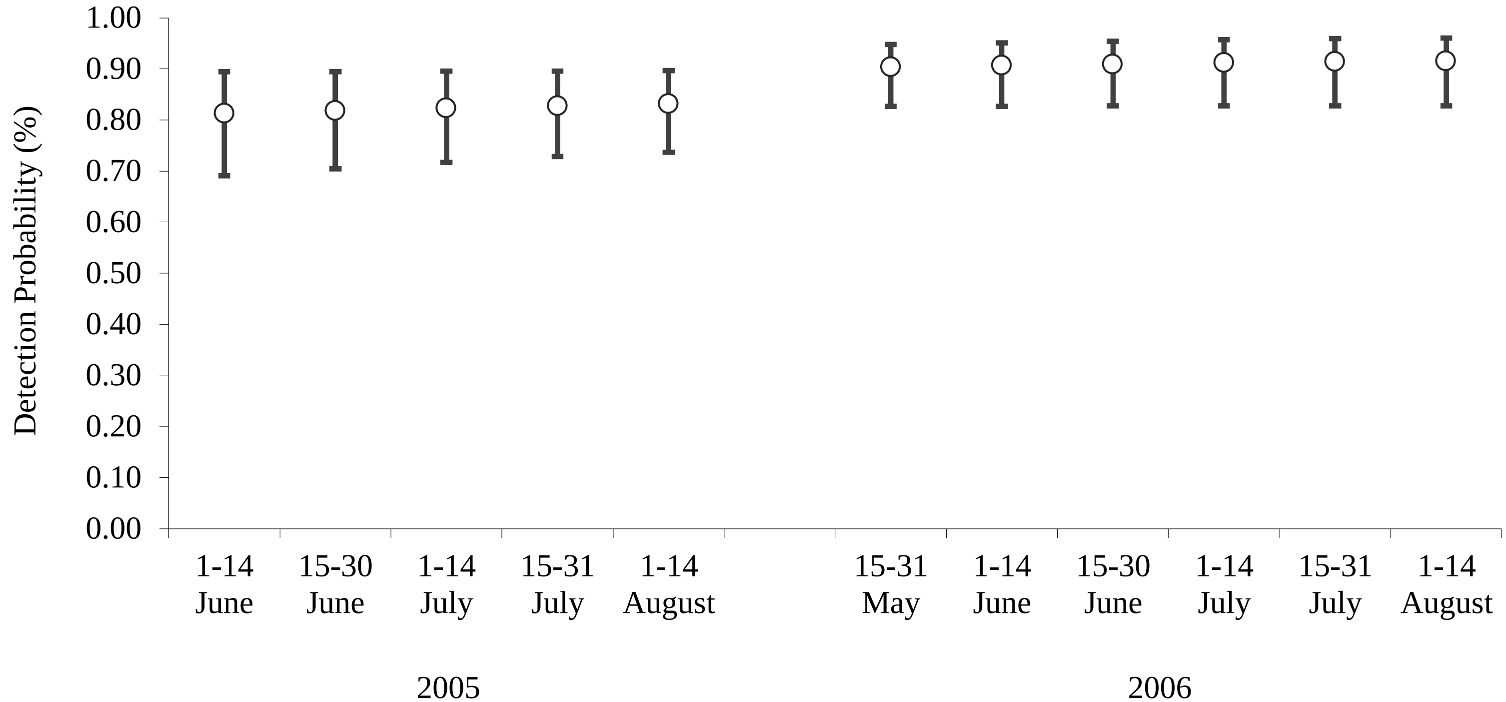

Fig. 2.13. Relationship between Savannah Sparrow detection probability and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Error bars indicate 95% confidence intervals.

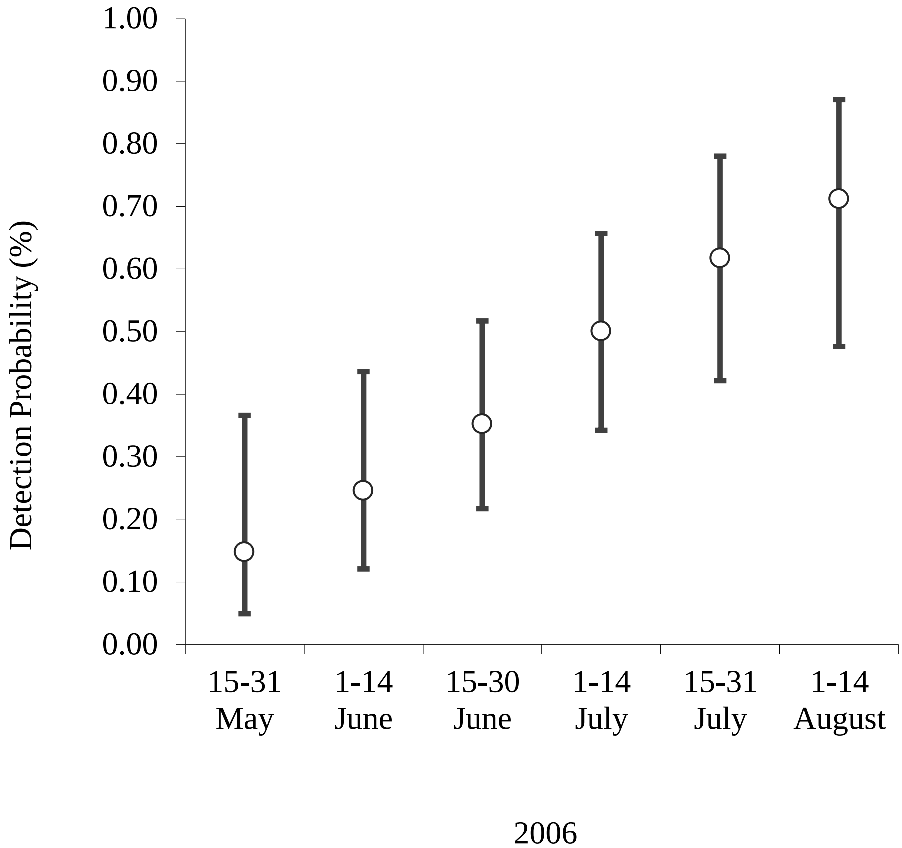

Fig. 2.14. Relationship between Sedge Wren detection probability and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2006. Error bars indicate 95% confidence intervals.

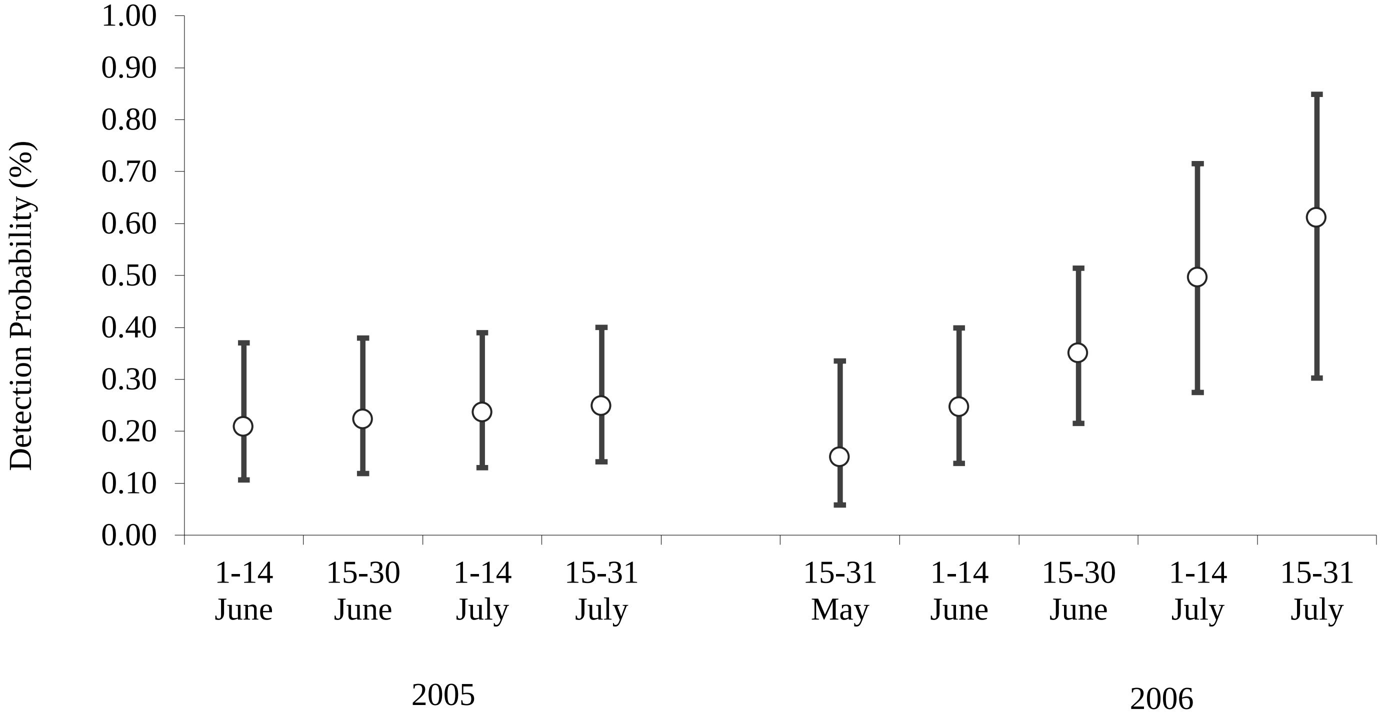

Fig. 2.15. Relationship between Dickcissel detection probability and ordinal date in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Error bars indicate 95% confidence intervals.

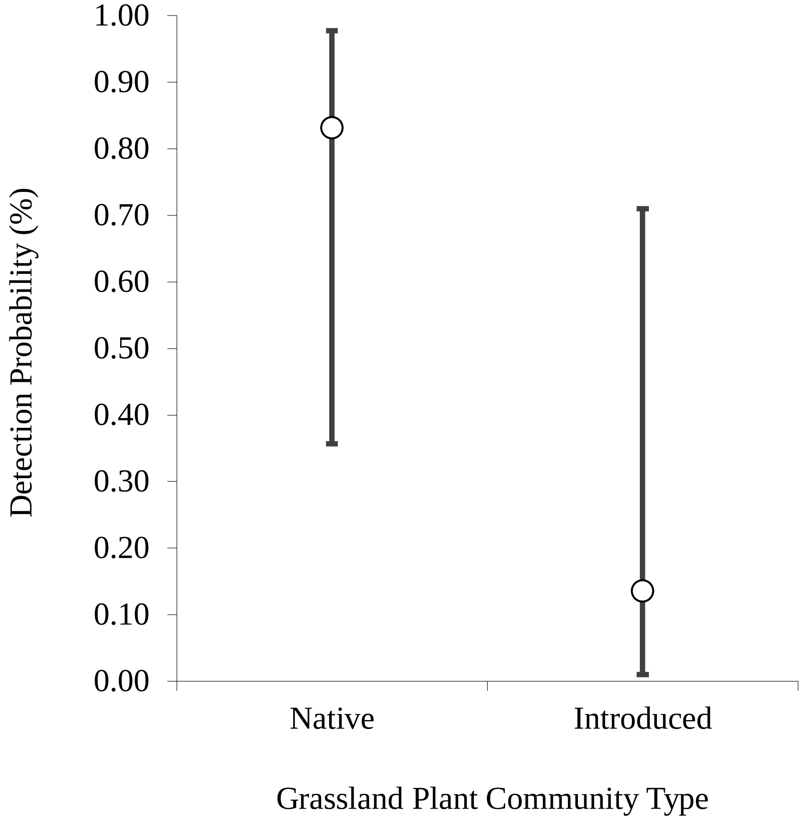

Fig. 2.16. Relationship between Sedge Wren detection probability and grassland type in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006. Error bars indicate 95% confidence intervals.

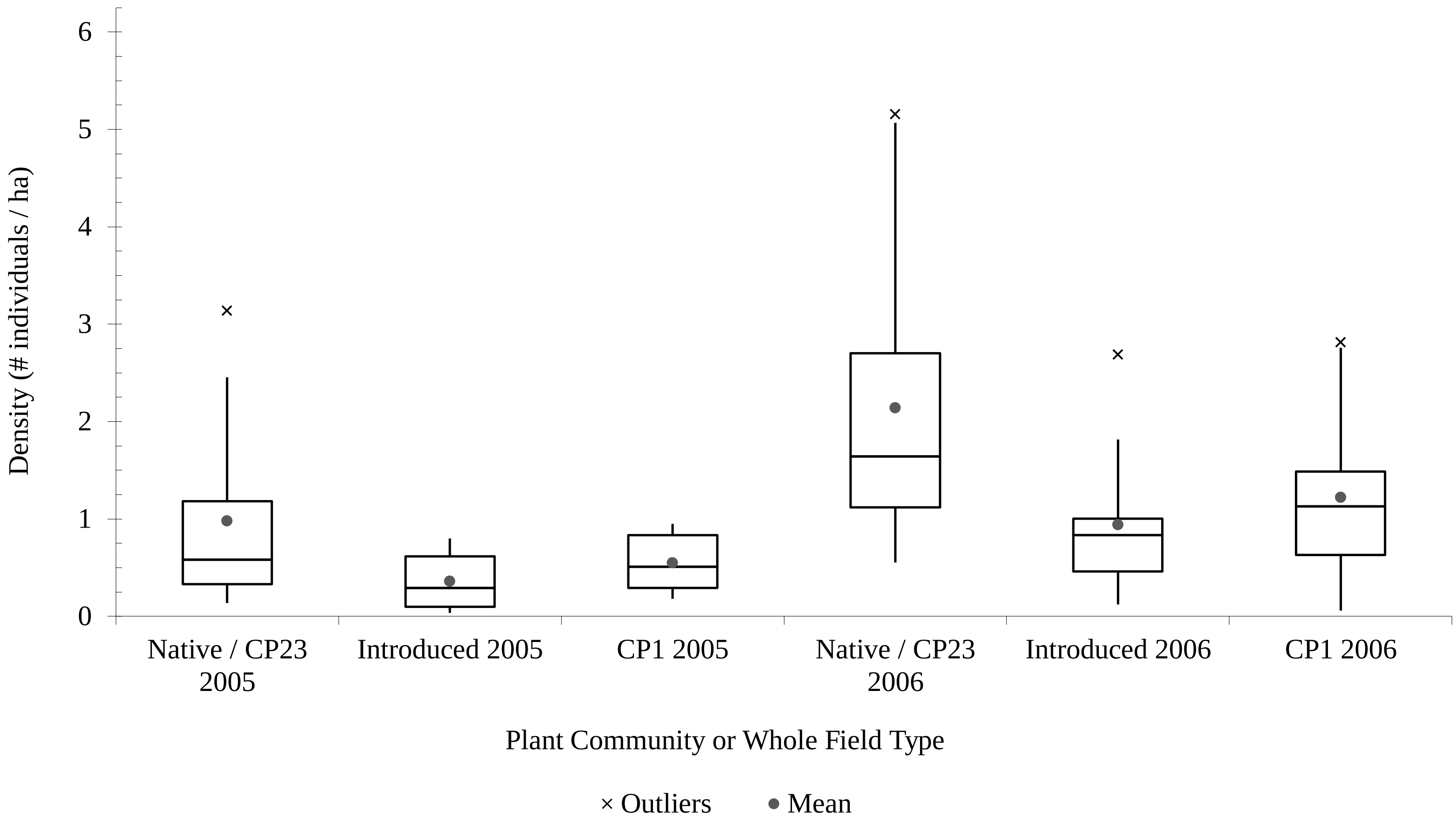

Fig. 2.17. Savannah Sparrow density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

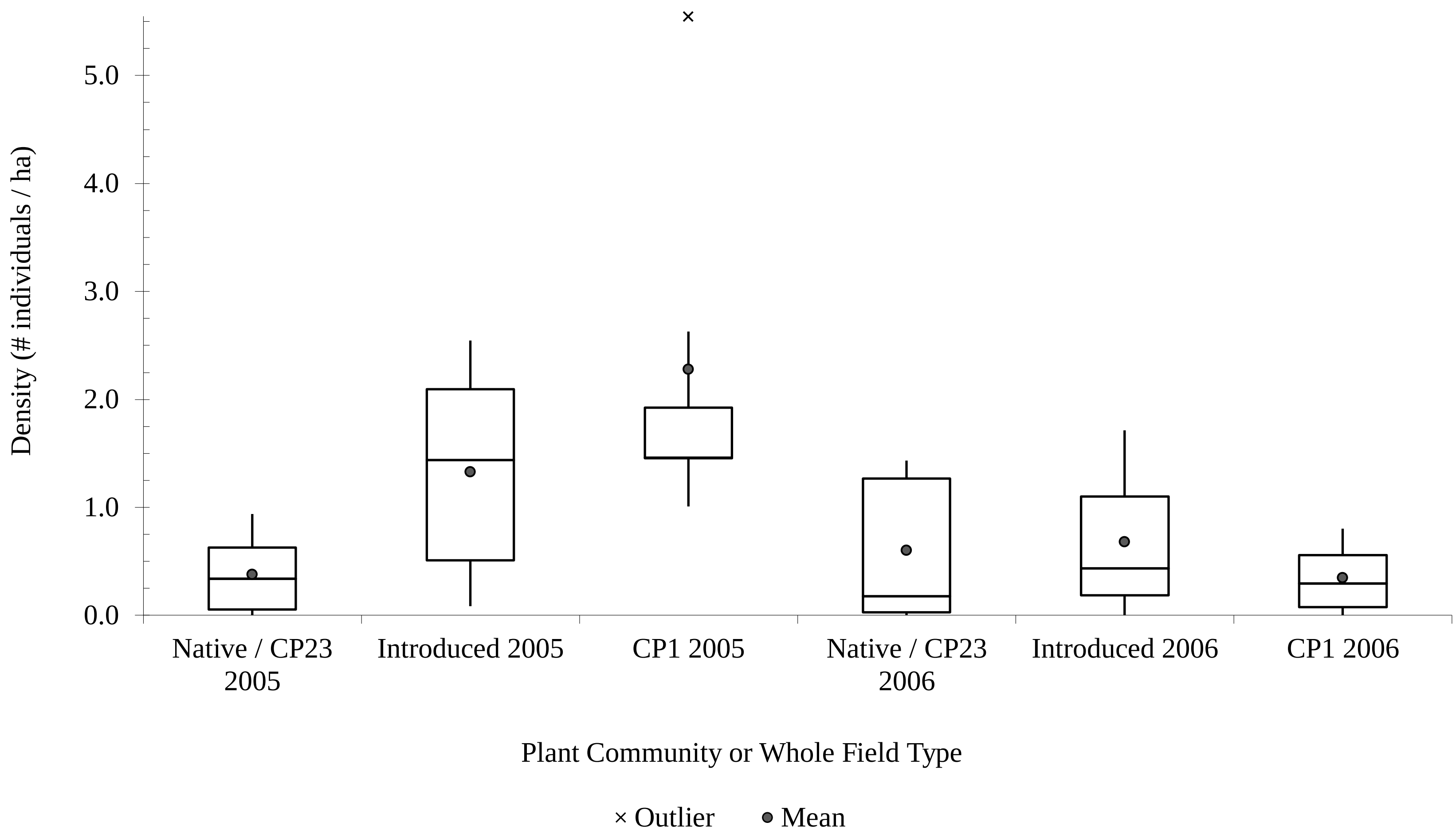

Fig. 2.18. Bobolink density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

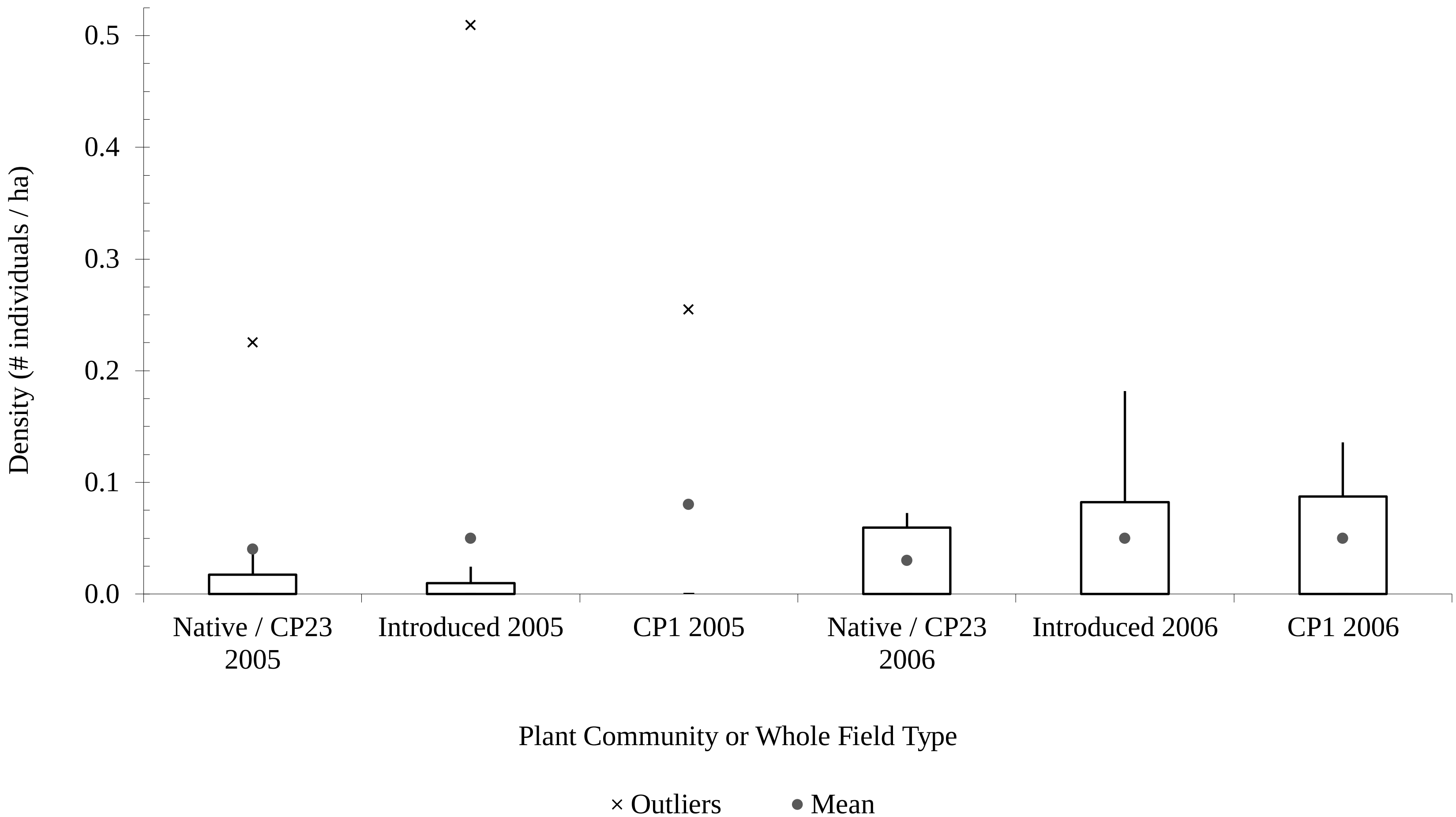

Fig. 2.19. Dickcissel density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

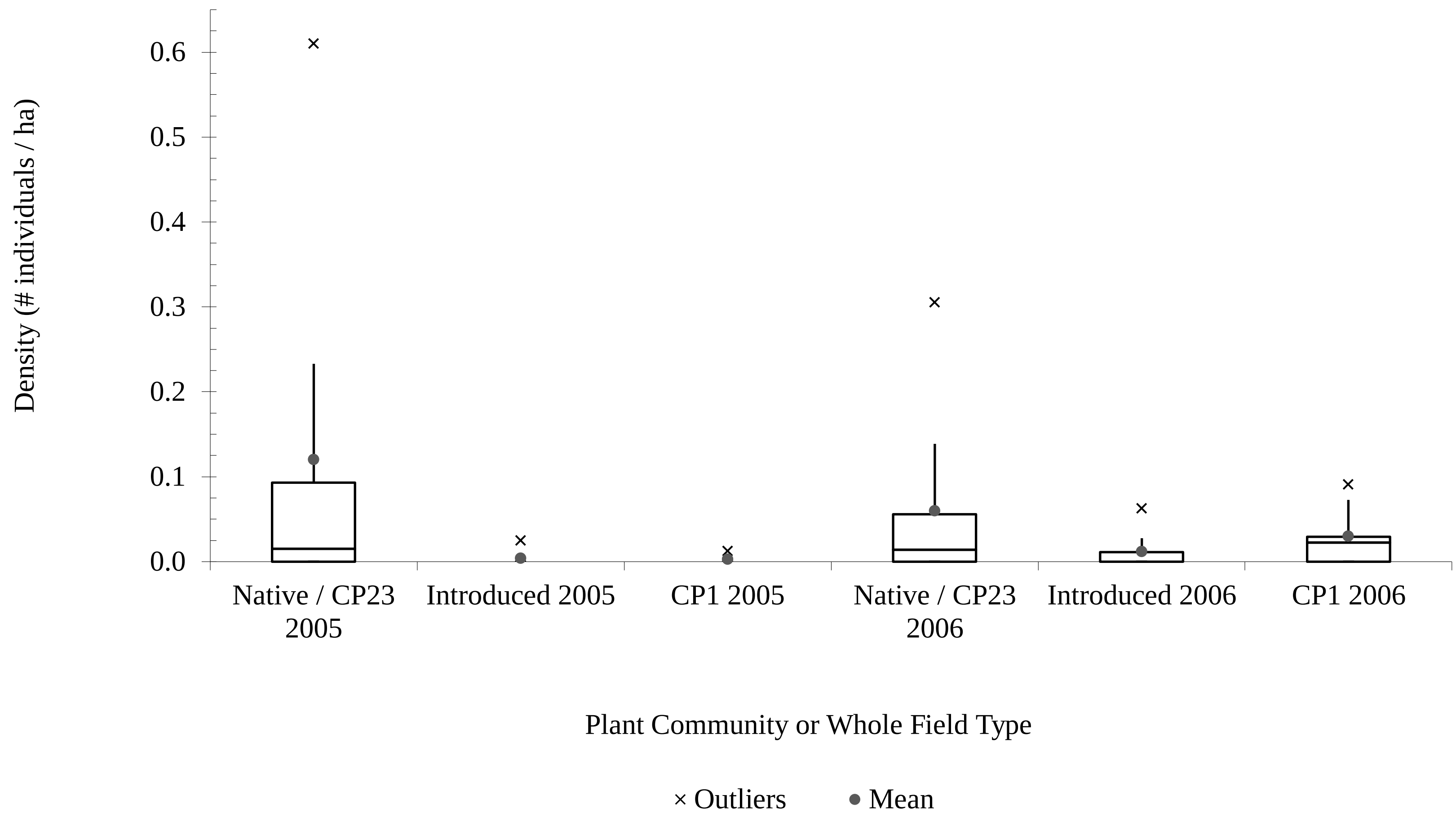

Fig. 2.20. Eastern Meadowlark density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

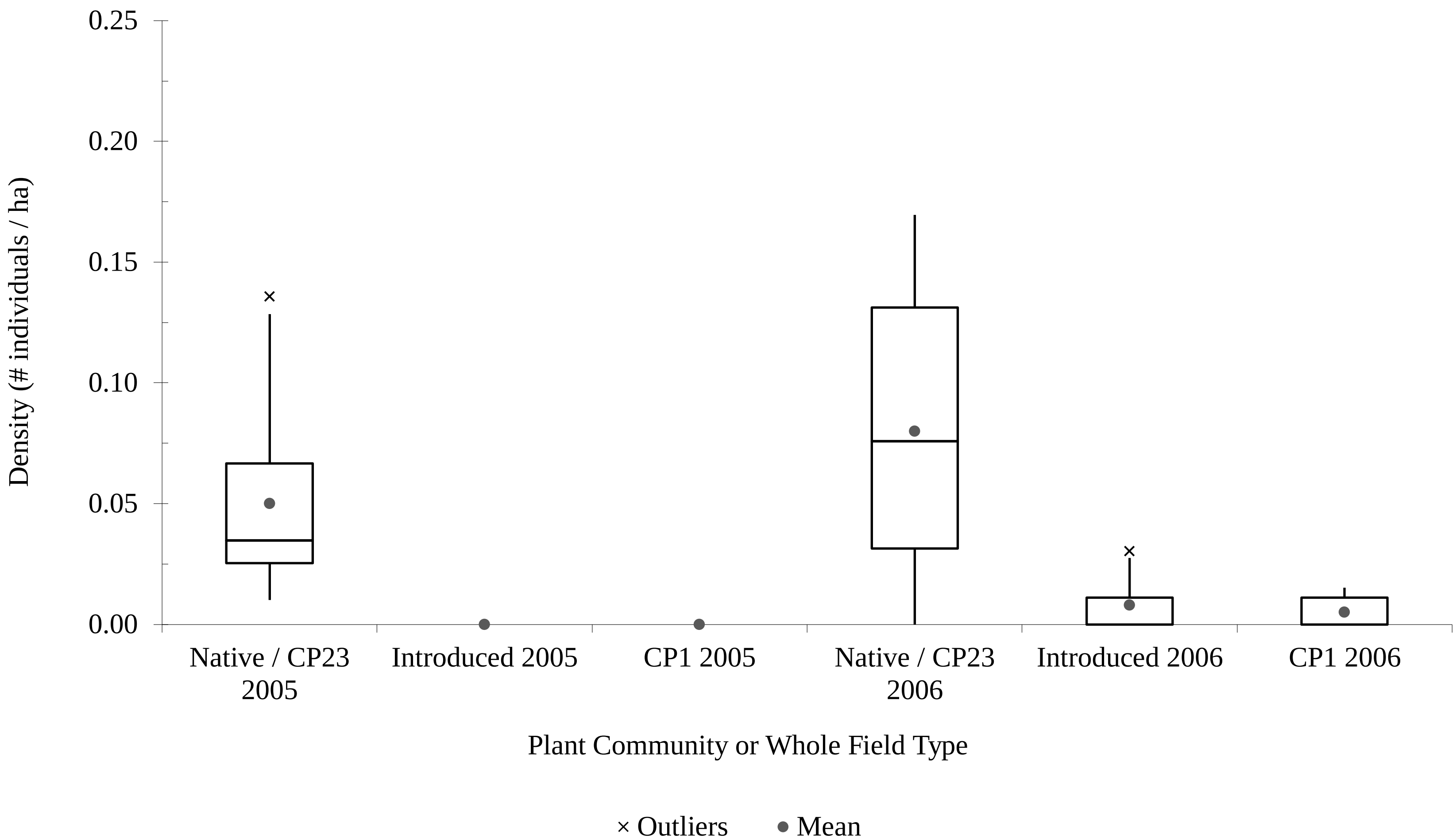

Fig. 2.21. Grasshopper Sparrow density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

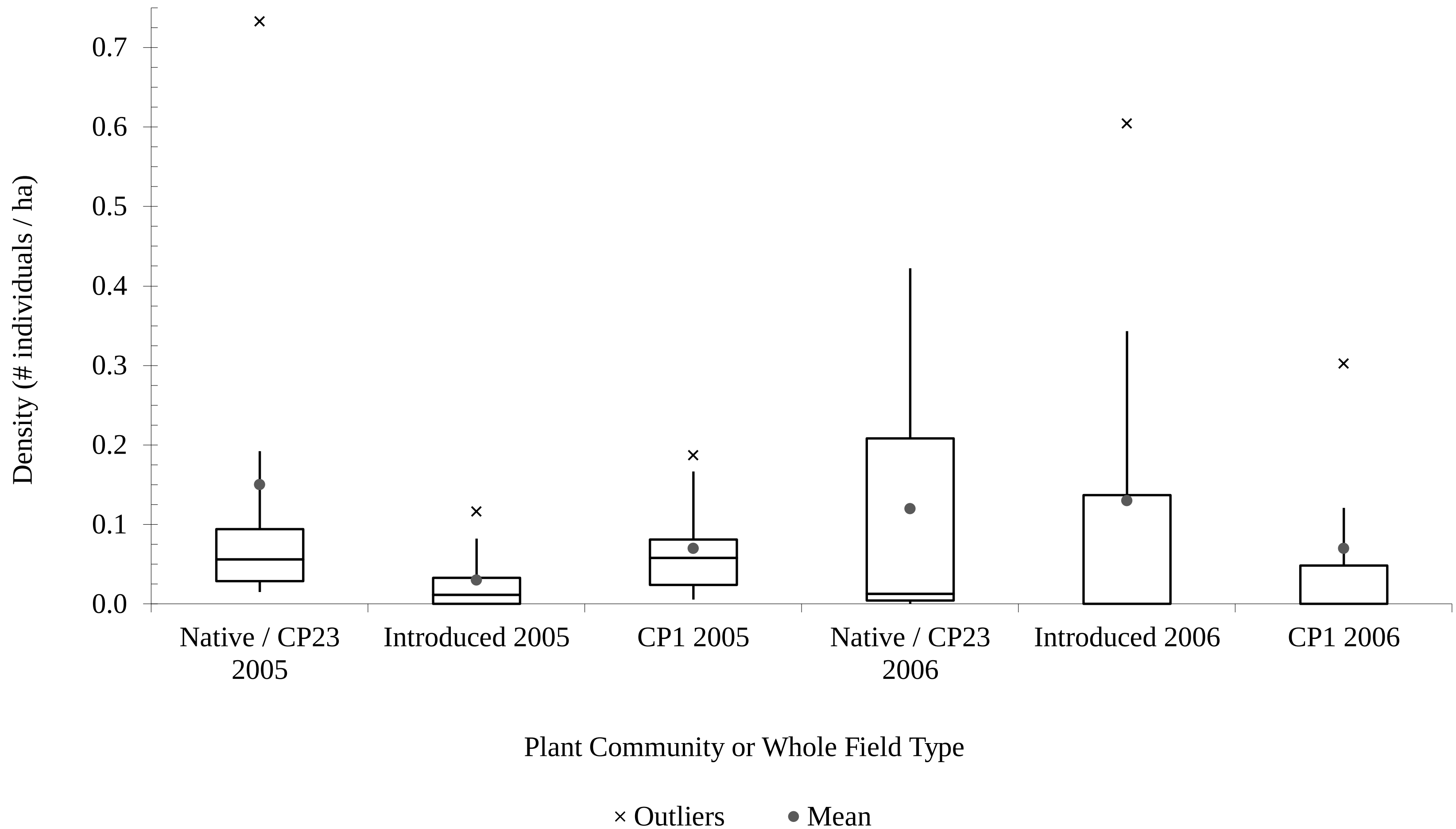

Fig. 2.22. Ring-necked Pheasant density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

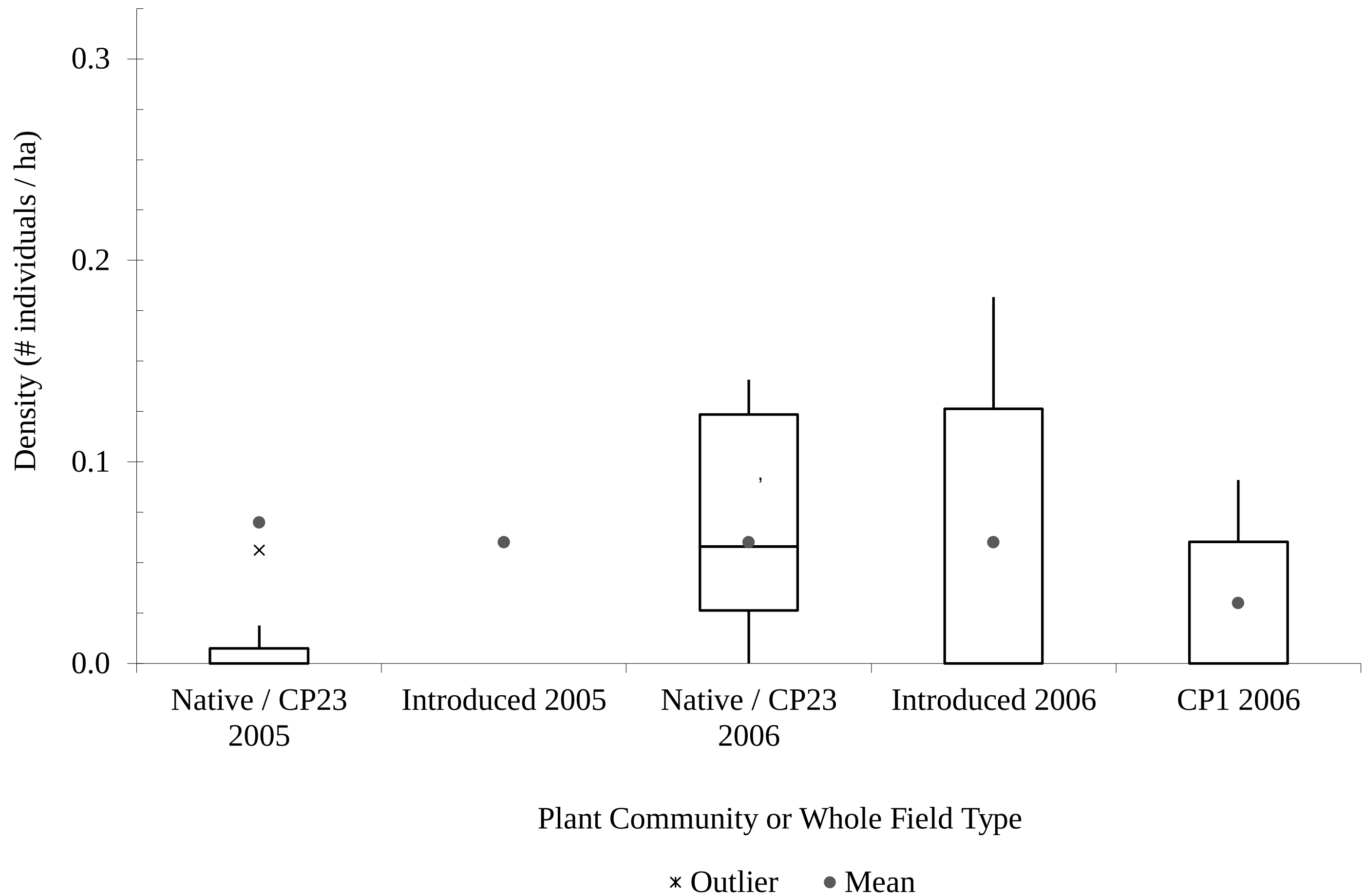

Fig. 2.23. Sedge Wren density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

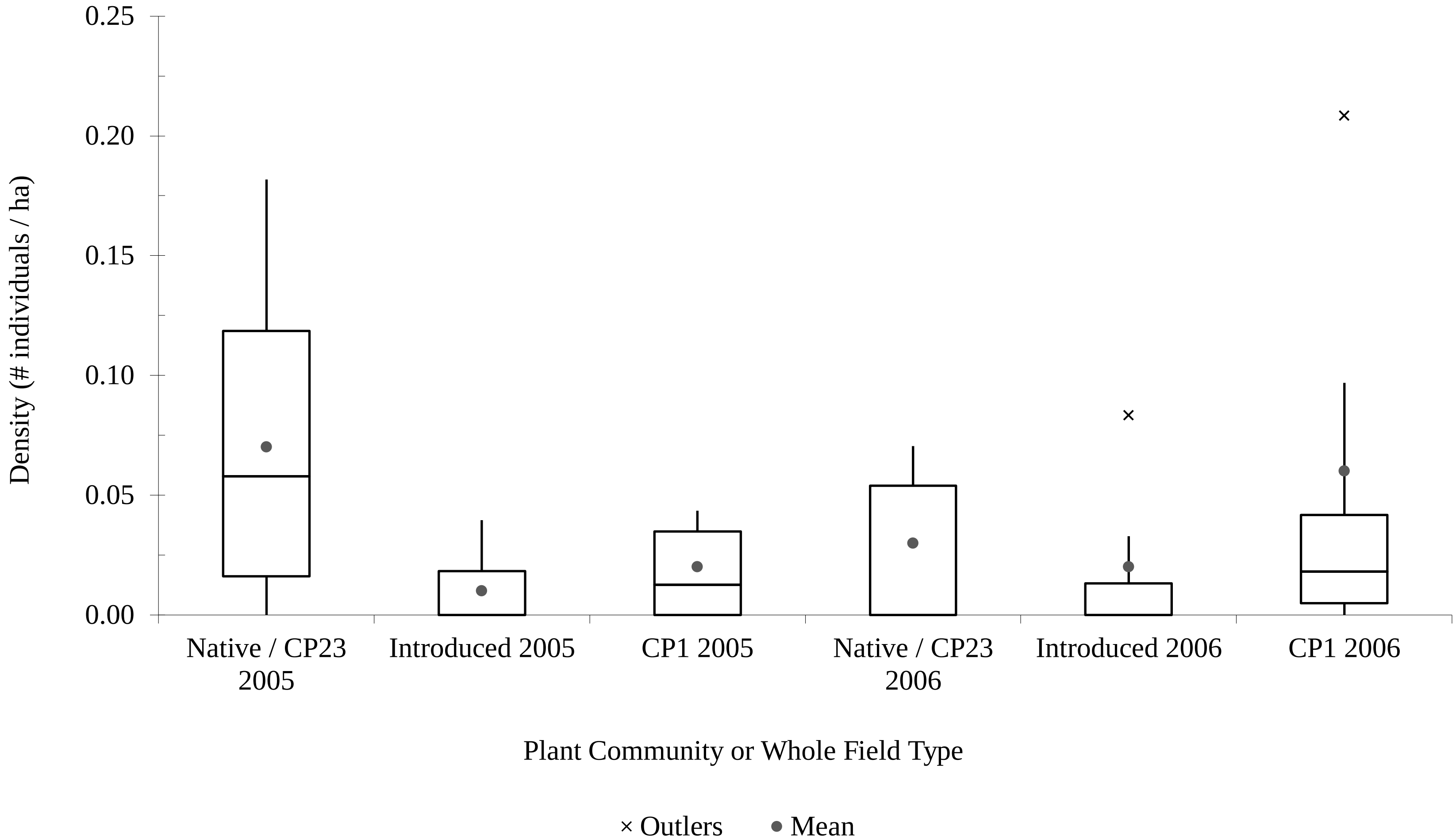

Fig. 2.24. Vesper Sparrow density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

Table 2.5. Grassland bird species, with estimated Michigan breeding territory size range (ha) and estimated number of breeding territories in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) fields and CP1 introduced grassland plant communities, Saginaw Bay watershed, Michigan, 2005-2006.

			Number of Territories (Range) In:		
Grassland Bird Species	Estimated Breeding Territory Size Range (ha)	Citation	CP23 Native Grasslands[A]	CP1 Whole Fields[B]	CP1 Introduced Grasslands[C]
Bobolink	0.49 - 2.0	Martin and Gavin 1995	4 - 40	4 - 49	3 - 35
Dickcissel	0.3 - 1.1	Temple 2002	6 - 66	7 - 80	5 - 57
Eastern Meadowlark	2.8 - 3.2	Lanyon 1995	2 - 7	2 - 9	2 - 6
Grasshopper Sparrow	0.85 - 1.4	Vickery 1996	5 - 23	5 - 28	4 - 20
Ring-necked Pheasant	4 – 8	Sargent and Carter 1999	1 - 5	1 - 6	1 - 4
Savannah Sparrow	0.11 - 0.86	Wheelwright and Rising 2008	8 - 180	9 - 218	6 - 155
Sedge Wren	0.13 - 0.36	Herkert et al. 2001	19 - 152	20 - 184	15 - 131
Vesper Sparrow	2.59 - 8.19	Jones and Cornely 2002	1 - 8	1 - 9	1 - 7

[A] *CP23 native grasslands range from 6.9 -19.8 ha in size (mean [SE] = 12.6 [1.9] ha, median = 11.8 ha).*

[B] *CP1 whole fields range from 7.3 – 23.9 ha in size (mean [SE] = 12.3 [2.1] ha, median =10.9).*

[C] *CP1 introduced grasslands range from 5.5 - 17.0 ha in size (mean [SE] = 9.4 [1.5] ha, median = 8.0).*

APPENDIX 2.2: DETECTION PROBABILITY MODELING AND DENSITY COMPARISON RESULTS FOR AVIAN SPECIES OTHER THAN GRASSLAND SPECIALISTS

RESULTS

A total of 34 avian species were observed in CREP study fields during the 2 years of this study (Table 2.3). Based upon the grassland bird list developed for the BBS by the U.S. Geographic Survey (Sauer et al. 1995), 23 of the observed species were identified as species other than grassland specialists (Table 2.3).

Avian detection probability modeling

Detection probability estimates were generated for non-grassland avian species in native and introduced CREP grassland plant communities. Fifteen non-grassland species were observed too rarely (i.e., observed in less than 10% of surveys) to estimate their detection probability: American Robin (*Turdus migratorius*), American Woodcock (*Scolopax minor*), Brown Thrasher (*Toxostoma rufum*), Brown-headed Cowbird (*Molothrus ater*), Chipping Sparrow (*Spizella passerina*), House Wren (*Troglodytes aedon*), Indigo Bunting (*Passerina cyanea*), Killdeer (*Charadrius vociferus*), Mallard (*Anas platyrhynchos*), Marsh Wren (*Cistothorus palustris*), Mourning Dove (*Zenaida macroura*), Northern Bobwhite Quail (*Colinus virginianus*), Northern Flicker (*Colaptes auratus*), White-crowned Sparrow (*Zonotrichia leucophrys*), and Wild Turkey (*Meleagris gallopavo*). Barn Swallow (*Hirundo rustica*), was observed frequently enough to perform detection probability modeling, but no successful detection models were generated. Successful detection models were obtained for Eastern Kingbird (*Tyrannus tyrannus*) only for CP23 fields and the introduced vegetation of CP1 fields.

To enable reliable comparisons of non-grassland bird density in 1) CP23 and CP1 fields (i.e., the program administrative-level comparison) and in 2) CP23 fields and CP1 introduced plant communities (i.e., the ecological-level comparison), the detection probability, p, of each avian species was estimated at both scales. Successful detection probability modeling occurred for 7 avian species: American Goldfinch (*Carduelis tristis*), Clay-colored Sparrow (*Spizella pallida*), Common Yellowthroat (*Geothlypis trichas*), Eastern Kingbird, Red-winged Blackbird (*Agelaius phoeniceus*), Song Sparrow (*Melospiza melodia*), and Tree Swallow (*Tachycineta bicolor*) (Table 2.6). Adequate detection probability estimates (i.e., $p \geq 0.10$) were achieved for all species, and ranged from 0.19 to 0.90 in CP23 and CP1 fields and from 0.17 to 0.84 in CP23 fields and introduced vegetation (Table 2.6). Adequate detection probability estimates were obtained only when a subset of the data was used for Clay-colored Sparrow (native 2006 data) and Eastern Kingbird (2006 data) (Table 2.6). The linear effects of the cover type (native or introduced vegetation) and/or standardized ordinal date were included in the best detection models for American Goldfinch, Clay-colored Sparrow, Common Yellowthroat, Red-winged Blackbird, and Song Sparrow (Table 2.6).

Avian density comparisons

Avian species were categorized as reliably detected (i.e., $p \geq 0.10$) or unreliably detected (i.e., p could not be estimated or $p < 0.10$) in each CREP field/grassland type. Avian densities were reported (Table 2.3) and compared (Figures 2.25-2.30) only for those species that were reliably detected. Species that were unreliably detected were designated as present if observed in a field/grassland type and as undetected if not observed (i.e., species that were unreliably detected actually may have been present when unobserved; Table 2.3).

Sixteen non-grassland avian species were observed in 2005; 14 species were recorded in CP23 fields, 11 species were found in introduced vegetation, and 9 species were detected in CP1 fields (Table 2.3). None of the 4 reliably detected species were unique to any field/vegetation type. In 2006, a total of 21 non-grassland species were observed, 16 species in CP23 fields, 15 species in introduced habitat, and 17 species in CP1 fields (Table 2.3). None of the 5 reliably detected species were unique to any field/vegetation type. Three species were detected in the native portions of CP1 fields that were unobserved in the introduced portions of CP1 fields: Northern Flicker, White-crowned Sparrow, and Wild Turkey in 2006 (Table 2.3). Two species, American Goldfinch (2005) and Northern Bobwhite Quail (2006) were detected in the introduced vegetation of CP1 fields but were recorded as undetected for CP1 whole fields because data were used from only 5 of the 7 CP1 whole fields (i.e., data from fields where the native plant community portion did not establish successfully were omitted (Table 2.3)). Among adequately detected species (i.e., $p \geq 0.10$), the most abundant non-grassland birds during both years of the study were Song Sparrow and Red-winged Blackbird, with the highest mean (Table 2.3) and median densities (Figures 2.25-2.30) in both native and introduced vegetation and in CP23 and CP1 whole fields (see range, median, and mean information below).

Avian density comparisons were conducted for species that were adequately detected (i.e., $p \geq 0.10$): American Goldfinch, Clay-colored Sparrow, Common Yellowthroat, Eastern Kingbird, Red-winged Blackbird, Song Sparrow, and Tree Swallow. Boxplots were used to compare avian densities in 1) CP23 and CP1 fields and in 2) CP23 fields and the introduced vegetation of CP1 fields (Figures 2.25-2.30).

American Goldfinch density was larger in CP23 fields than in introduced vegetation in 2005, and no differences in density were observed in 2006 (Figure 2.25). In

2005, American Goldfinch density ranged from 0.05 – 0.89 individuals/ha in CP23 fields (median = 0.20, mean = 0.33, SE = 0.13) and from 0 – 0.20 in CP1 fields (median = 0.08, mean = 0.08, SE = 0.03; Figure 2.25, Table 2.3). Clay-colored Sparrow was reliably detected only in 2006 in CP23 fields, so density comparisons were not possible. Clay Colored density ranged from 0 – 0.22 individuals/ha (median = 0.02, mean = 0.05, SE = 0.03) in CP23 fields in 2006. Common Yellowthroat density in CP1 fields was more variable but not clearly larger than in CP23 fields in 2005; in 2006, Common Yellowthroat density was larger in CP23 fields than in the introduced vegetation of CP1 fields or in CP1 whole fields (Figure 2.26). In 2006, Common Yellowthroat density ranged from 0.01 – 0.28 individuals/ha in CP23 fields (median = 0.08, mean = 0.12, SE = 0.04), from 0 – 0.40 individuals/ha in introduced vegetation (median = 0.02, mean = 0.10, SE = 0.06), and from 0 – 0.62 individuals/ha in CP1 fields (median = 0.03, mean = 0.16, SE = 0.12; Figure 2.26, Table 2.3). In 2006, Eastern Kingbird density was larger in CP23 fields than in introduced vegetation (Figure 2.27). In 2006, Eastern Kingbird density ranged from 0 – 0.05 individuals/ha in CP23 fields (median = 0.02, mean = 0.03, SE = 0.01) and from 0 – 0.02 individuals/ha in introduced vegetation (median = 0, mean = 0.004, SE = 0.003; Figure 2.27, Table 2.3). Red-winged Blackbird density was smaller in CP23 fields than in CP1 introduced vegetation or in CP1 whole fields during both years of the study (Figure 2.28). In 2005, Red-winged Blackbird density ranged from 0.02 – 1.48 individuals/ha in CP23 fields (median = 0.22, mean = 0.64, SE = 0.26), from 0.95 – 3.37 individuals/ha in introduced vegetation (median = 2.6, mean = 2.30, SE = 0.29), and from 0.93 – 4.06 individuals/ha in CP1 fields (median = 1.6, mean = 2.18, SE = 0.62; Figure 2.28, Table 2.3). In 2006, Red-winged Blackbird density ranged from 0.02 – 1.43

individuals/ha in CP23 fields (median = 0.67, mean = 0.73, SE = 0.19), from 1.67 – 4.55 individuals/ha in introduced vegetation (median = 2.63, mean = 2.97, SE = 0.36), and from 1.17 – 2.70 individuals/ha in CP1 fields (median = 2.19, mean = 2.11, SE = 0.26; Figure 2.28, Table 2.3). In contrast, Song Sparrow density was larger in CP23 fields than in introduced vegetation during both years of the study (Figure 2.29). In 2005, Song Sparrow density ranged from 0.80 – 3.01 individuals/ha in CP23 fields (median = 1.12, mean = 1.54, SE = 0.35) and from 0.18 – 0.75 individuals/ha in introduced vegetation (median = 0.37, mean = 0.46, SE = 0.08; Figure 2.29, Table 2.3). In 2006, Song Sparrow density ranged from 0.80 – 3.36 individuals/ha in CP23 fields (median = 1.38, mean = 1.89, SE = 0.37) and from 0.14 – 1.22 individuals/ha in introduced vegetation (median = 0.72, mean = 0.76, SE = 0.14; Figure 2.29, Table 2.3). No differences in Tree Swallow density in CP23 or CP1 fields occurred in 2006, the only year reliable density information for the species was available (Figure 2.30).

Table 2.6. Non-grassland avian species, with corresponding best-fitting detection models [with constant occupancy, $\psi(\cdot)$] and detection probability estimates, observed in 1) CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grassland fields and 2) CP23 native and CP1 introduced grassland plant communities, Saginaw Bay watershed, Michigan, 2005-2006. The lowest detection probability estimate was reported when multiple detection probability estimates were generated.

	CP23 and CP1 Whole CREP Fields		Native and Introduced Plant Communities	
Common Name	Detection Model	$p(\cdot)$ [SE]	Detection Model	$p(\cdot)$ [SE]
American Goldfinch	*p(CREP Field Type)*	0.56 [0.06]	*p(Grassland Type)*	0.42 [0.06]
Clay-colored Sparrow, native 2006 data	*p(Ordinal Date)*	0.21 [0.14]	*p(Ordinal Date)*	0.21 [0.14]
Common Yellowthroat	*p(Ordinal Date)*	0.23 [0.06]	*p(Ordinal Date)*	0.20 [0.05]
Eastern Kingbird, 2006 data			*p(.)*	0.17 [0.07]
Red-winged Blackbird	*p(CREP Field Type, Ordinal Date)*	0.56 [0.09]	*p(Grassland Type, Ordinal Date)*	0.56 [0.09]
Song Sparrow	*p(CREP Field Type)*	0.90 [0.03]	*p(Grassland Type)*	0.84 [0.04]
Tree Swallow	*p(.)*	0.19 [0.09]		

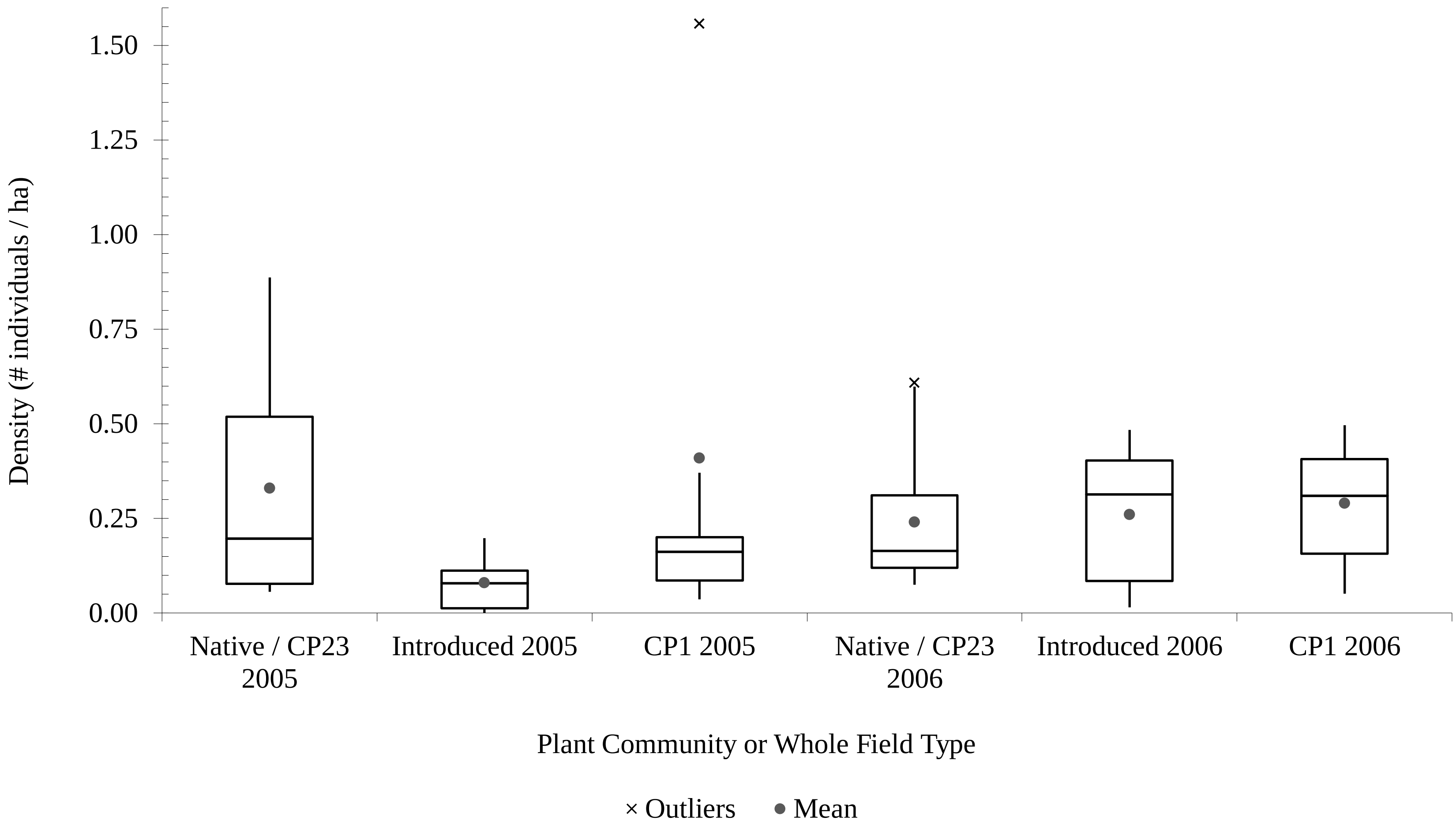

Fig. 2.25. American Goldfinch density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

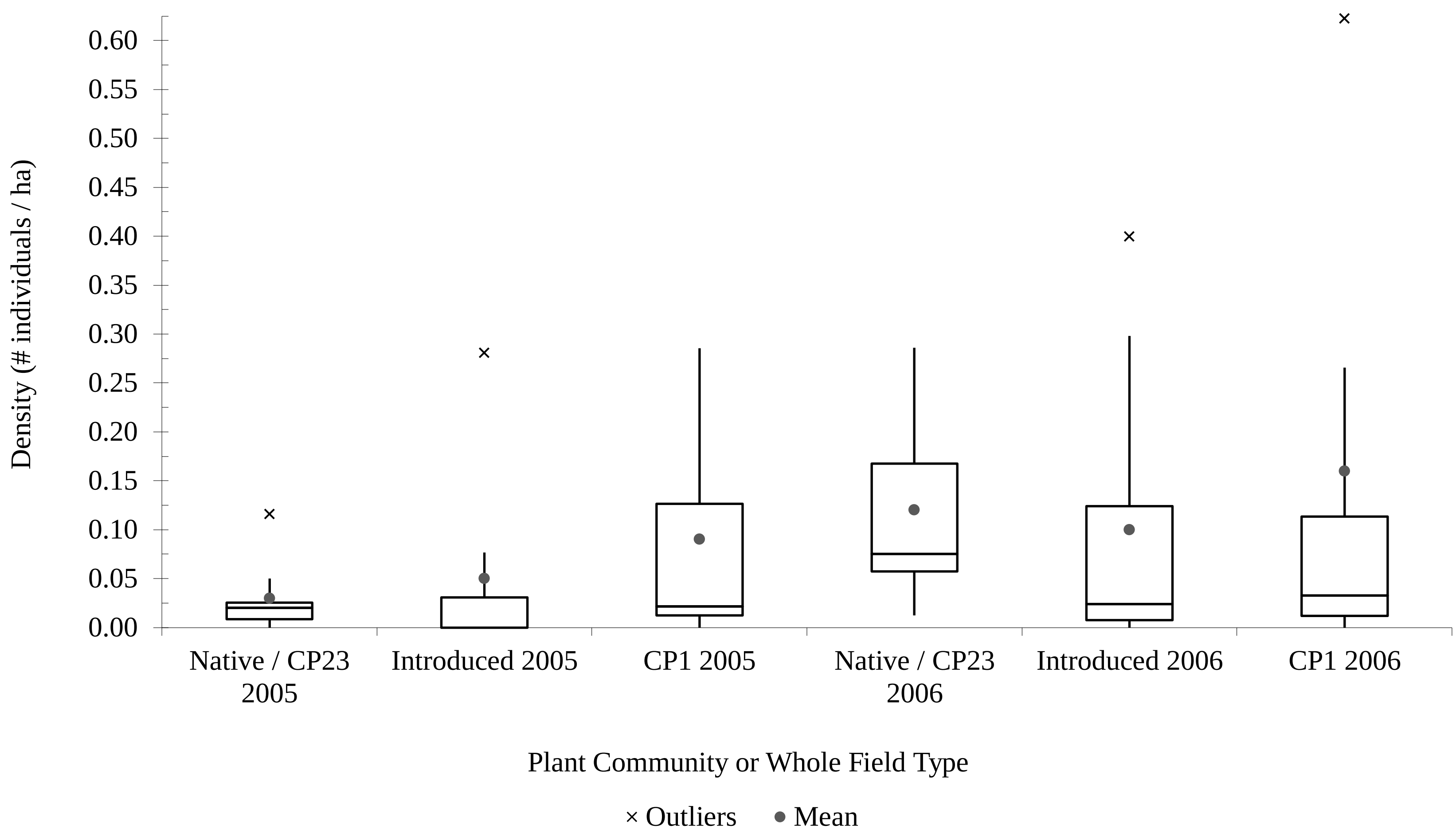

Fig. 2.26. Common Yellowthroat density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

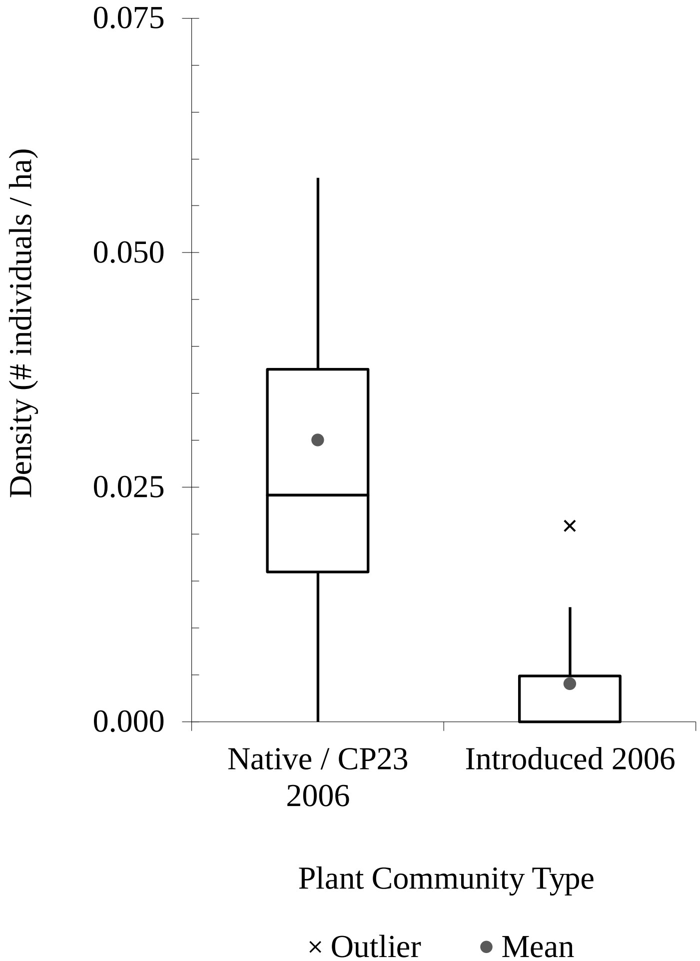

Fig. 2.27. Eastern Kingbird density (# individuals / ha) in native/CP23 whole fields and introduced vegetation of CP1 fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2006.

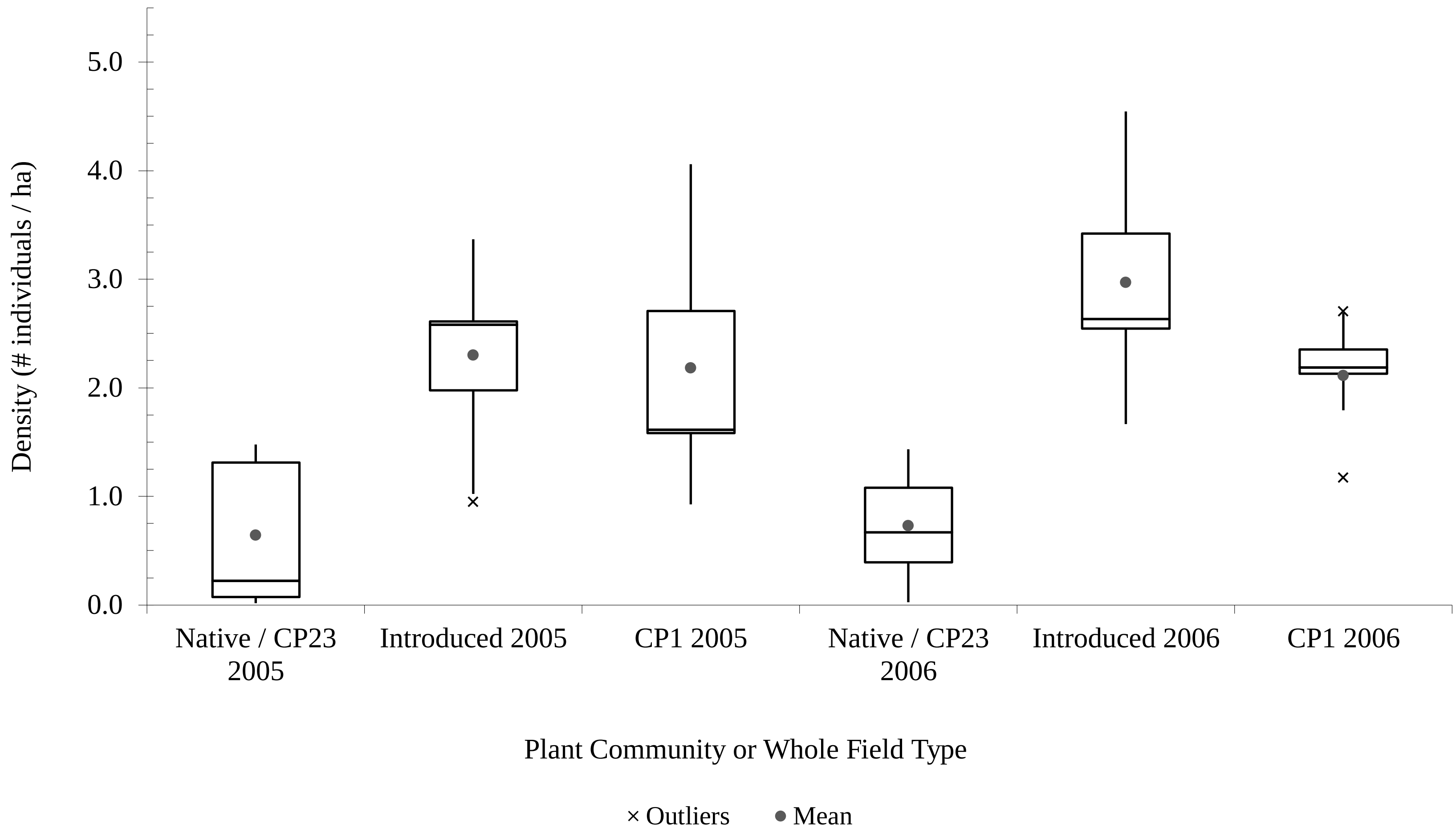

Fig. 2.28. Red-winged blackbird density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

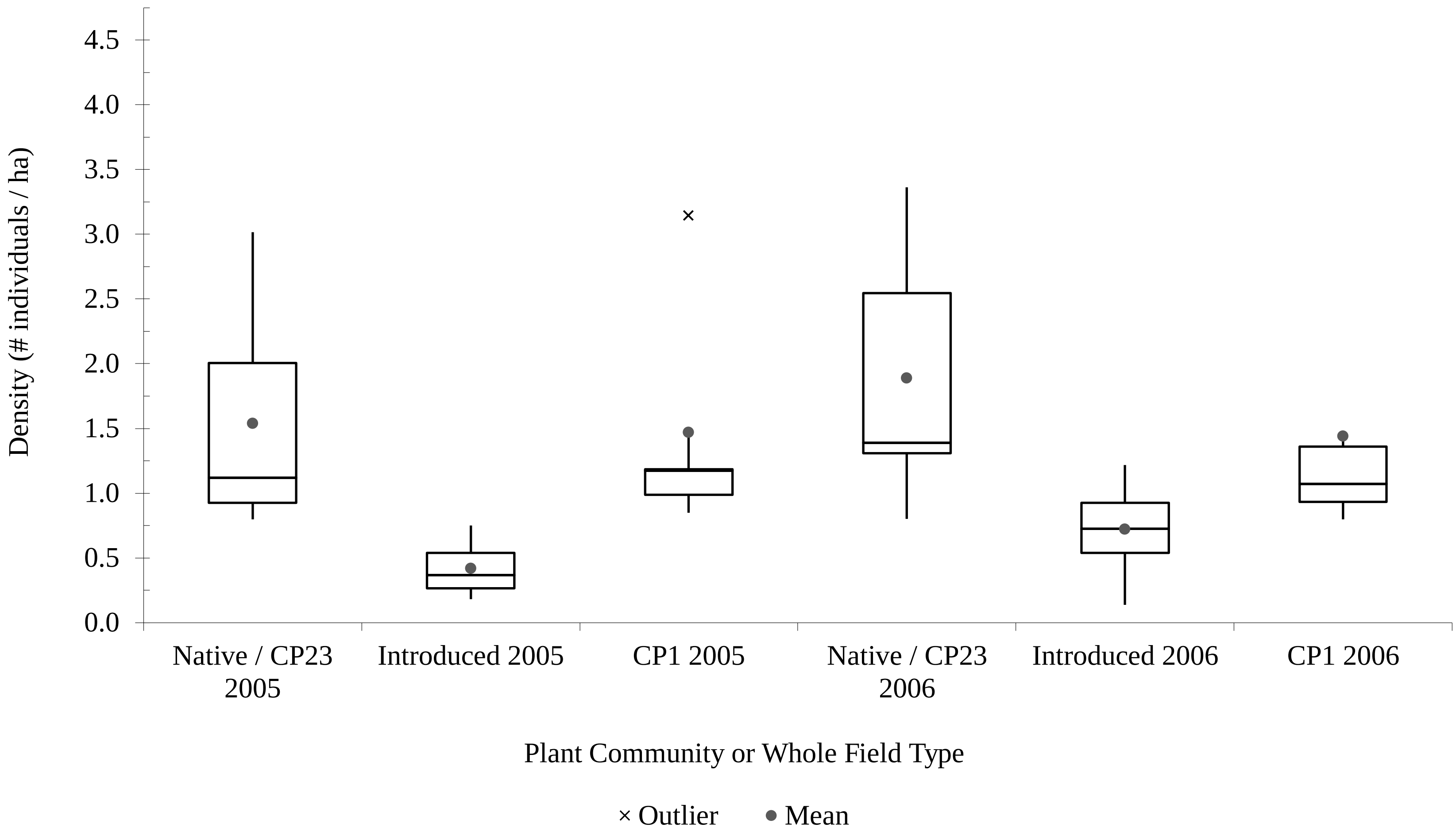

Fig. 2.29. Song Sparrow density (# individuals / ha) in native/CP23 whole fields, introduced vegetation of CP1 fields, and CP1 whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2005 and 2006.

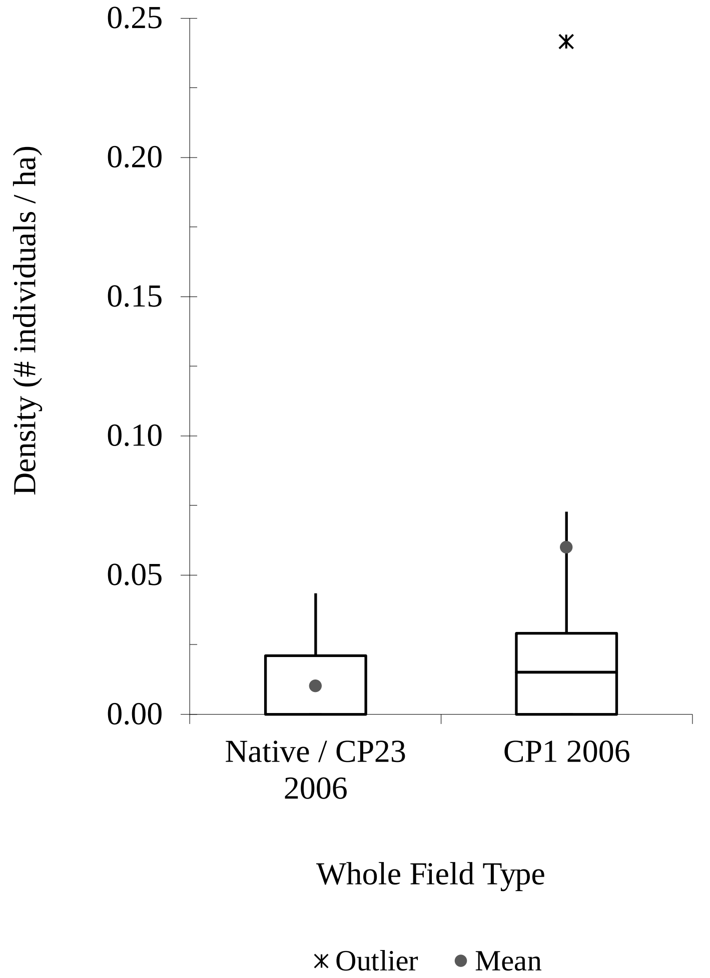

Fig. 2.30. Tree Swallow density (# individuals / ha) in native/CP23 and whole fields of Conservation Reserve Enhancement Program (CREP) grasslands in Michigan's Saginaw Bay watershed, 2006.

APPENDIX 2.3: COMPARISON OF AVIAN PRODUCTIVITY WITHIN CP23 NATIVE AND CP1 INTRODUCED PLANT COMMUNITIES OF MICHIGAN CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) GRASSLANDS

METHODS

To investigate the source/sink status of CREP study fields, avian productivity and nesting success in CREP lands were examined. All study fields were searched for nests one time in June and once in July, 2006. Due to the large size of several study fields (i.e., >20 ha), the nest search area in all 14 study fields was equal to the smallest study field size, or approximately 6.8 hectares. Nest searches were conducted by 3 – 6 observers walking 3 – 5 m abreast while traversing the nest search areas. As the observers traveled, they swept the top of the vegetation with yardsticks to encourage sitting birds to flush from their nests. Nests were revisited every 2 – 3 days until the young fledged or nests were abandoned or destroyed. To minimize the chance that predators would be attracted to nests due to researcher activity, nests were marked with flags placed 5 m north of actual nest location, and different paths were taken to nests during each visit. Nest location was mapped using a handheld Global Positioning System (GPS) unit which recorded the latitude and longitude coordinates of nest sites and facilitated navigation to nests during revisits. Nests discovered while performing bird surveys or vegetation sampling were also revisited and data were recorded as for any other nest. Avian species and nest outcome were recorded for each nest. Successful nests were defined as those that produced at least one fledgling. If no fledglings were noted in the area of an empty nest, fledglings were assumed to be produced and the nest outcome was recorded as successful if 1) nestlings were close to fledgling status at the time of the last nest visit and 2) there was no evidence of predation (i.e., highly disturbed or destroyed nest structure or the presence of nestling remains). Unsuccessful nests were those from which eggs or young disappeared and fledglings could not be assumed to have been produced. Active nests were classified as those that produced at least one egg or hatchling within the time period of the study. Percent nesting success for an avian species was calculated as the number of successful nests divided by the number of active nests after the number of nests with unknown outcome was subtracted.

$$\% \text{ Nesting Success} = \frac{\# \text{ Successful Nests}}{(\# \text{ Active Nests} - \# \text{ Nests with Unknown Outcome})}$$

RESULTS

Intensive grassland bird nest search efforts in CREP fields were unsuccessful, largely due to the extremely cryptic nature of the nests of most grassland bird species. Lack of visual cues for nests was often compounded by the tendency of adult birds to flee on foot beneath the plant canopy prior to or instead of flying when flushed from the nest, making it difficult to pinpoint nest locations. Unknown nest outcome was the result of unsuccessful efforts to relocate highly cryptic nests, despite the use of flagging and a GPS with nest location coordinates. Nest relocation efforts, as well as efforts to initially locate a nest that was assumed to be in a particular area due to adult bird behaviors such as food transport, were abandoned if nest searches could not be performed without endangering the nest by severely disturbing the nearby vegetation, which would make direct damage to the nest and/or attraction of predators to the nesting area likely.

During 2006, a total of 28 individual active nests of seven species were found and monitored in the 14 nest search areas (Table 2.7). The vast majority of nests observed (i.e., 21) were those of Red-winged Blackbirds. Two Song Sparrow nests were also discovered, along with 1 nest of each of the following species: Ring-necked Pheasant, Savannah Sparrow, Vesper Sparrow, American Goldfinch, and Mallard. Ring-necked

Pheasant, Vesper Sparrow, and Song Sparrow nests all occurred in native vegetation, while Savannah Sparrow, American Goldfinch, Mallard, and Red-wing Blackbird nests were observed in introduced vegetation (Table 2.7).

The nesting success of American Goldfinch could not be defined because the single active nest observed was not relocated and nest outcome was unknown. The percent nesting success of all other species could be calculated, but so few nests were discovered for most species that this statistic was only informative for Red-winged Blackbird. Sixty-nine percent of active Red-wing Blackbird nests with known outcome produced at least 1 fledgling (Table 2.7). The Ring-necked Pheasant and Mallard nests that were observed were unsuccessful; both nests were destroyed, apparently by predators (Table 2.7). The single active Savannah Sparrow nest was successful in producing at least one fledgling, as was the single active Vesper Sparrow nest observed and the 2 active Song Sparrow nests that were monitored (Table 2.7).

DISCUSSION

Given unsuccessful efforts to quantify avian productivity, it is impossible to evaluate the sink/source status of the CREP grasslands in this study. However, there is increasing evidence that farmland set-aside lands are benefiting grassland bird species at the population level (e.g., Veech 2006, Herkert 2009). In a study of the response of avian populations in the north-central region of the U.S. to CRP farmland set-aside programs, Herkert (2009) found that avian species with abundances that were much higher in CRP fields than in row crop fields tended to have declining populations prior to CRP establishment (1966-86) and tended to have an increasing population trend in the years after CRP establishment (1987-2007), a pattern that appeared to be strongest for the group of grassland obligate species in the study. Herkert (2009) also found that other avian species, including some grassland species, showed population trends that declined more steeply in the years after CRP establishment. However it is impossible to predict what the trend in population size of any species would have been without the effect of farmland set-aside programs; perhaps population trends would have been even more negative in the absence of set-aside fields. Veech (2006) compared the landscapes occupied by increasing and decreasing grassland-nesting avian species in 18 states in the Midwest and Great Plains regions and found that CRP grasslands were beneficial to grassland bird population growth. Even small (3-142 ha) grassland fragments have been found to have a positive effect on the population fecundity of some grassland birds, such as Dickcissel (estimated breeding territory size = 0.3 – 1.1 ha; Temple 2002) and Eastern Meadowlark (estimated breeding territory size = 2.8 – 3.2 ha; Lanyon 1995) (Walk et al. 2010), indicating that the loss of small habitat patches such as the CREP grasslands in this study could have negative, perhaps cumulative, impacts on the populations of grassland birds, and emphasizing the importance of maintaining farmland set-aside grasslands such as those in this study.

Table 2.7. Active avian nests observed, nest outcome, and vegetation type of the nest location in Conservation Reserve Enhancement Program (CREP) whole field grasslands, Saginaw Bay watershed, Michigan, 2006.

Avian Species	Total active nests	Successful nests	Unsuccessful nests	Unknown nest outcome
Grassland Species				
Ring-necked Pheasant	1[A]	0	1[A]	0
Savannah Sparrow	1[B]	1[B]	0	0
Vesper Sparrow	1[A]	1[A]	0	0
Non-grassland Species				
American Goldfinch	1[B]	0	0	1[B]
Mallard	1[B]	0	1[B]	0
Red-winged Blackbird	21[B]	11[B]	5[B]*	5[B]
Song Sparrow	2[A]	2[A]	0	0
Total	28	15	7*	6

[A] *Nest(s) occurred in native vegetation.*

[B] *Nest(s) occurred in introduced vegetation.*

**A total of two nests were destroyed when landowner/manager mowed within and between thistle patches.*

APPENDIX 2.4: COMPARISON OF AVIAN SPECIES PRESENCE IN MICHIGAN CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) AND CONSERVATION RESERVE PROGRAM (CRP) INTRODUCED GRASSLAND VEGETATION

METHODS

Results from this 2005-06 study of avian populations of Conservation Reserve Enhancement Program (CREP) grasslands were compared to those from a study of avian populations of Conservation Reserve Program (CRP) grasslands performed in 1991-92 (Millenbah 1993). CRP introduced grassland plantings in Gratiot County, Michigan, were studied in 1991 (n = 12, 7.3-19.8 ha) and 1992 (n = 19, including 10 fields also studied in 1991, 7.7-19.8 ha; Millenbah 1993). Vegetation sampling was performed at 20 m intervals along six 100 m long transects in each CRP field in April just after snow melt, in May, and in July (Millenbah 1993). Structural variables of the CRP vegetation were visually estimated using a 50 x 50 cm Daubenmire frame (Daubenmire 1959). Avian surveys in CRP fields were conducted using line transect sampling and the entire field area was observed as previously described for CREP fields (Chapter 2) (Millenbah 1993).

Detection probability modeling was not performed during the CRP study, so the accuracy of absence records of species observed at that time is uncertain, as is the accuracy of absence records for species without adequate detection probability estimates, p, (i.e., $p \geq 0.10$) from the CREP study. However, no such uncertainly is associated with the presence records of species from either study, so patterns of avian presence in the two studies were compared.

RESULTS

Means of mid-summer vegetation variables from same-aged CREP and CRP introduced grasslands were compared (Table 2.8). In CREP vegetation, the plant canopy composed the highest percentage of field area, followed by a smaller proportion of litter cover and even less bare ground, a pattern that also appeared to occur in CRP fields (Table

2.8). Within the canopy, live vegetation was more abundant than dead vegetation in both types of grasslands (Table 2.8). Grasses tended to be more abundant than forbs in the live canopy of CRP fields, while grasses and forbs in CREP vegetation grasses tended to be equally abundant (Table 2.8). Although similar in both grassland types, litter depth in CREP vegetation tended to be greater than in CRP fields, while horizontal cover tended to be greater in CRP fields, possibly due to the inclusion of sweet clover in CRP planting mixes (Table 2.8).

Presence records for all avian species observed in CRP and CREP introduced grasslands were summarized, along with avian density information for species that were reliably detected in the CREP study (Table 2.9). A total of 10 grassland specialist species were observed in the introduced vegetation of both the CRP and CREP studies: Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Horned Lark, Northern Harrier, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow. All 10 species were observed in introduced CRP introduced fields, while 9 species, all except Horned Lark, were observed in introduced CREP introduced vegetation (Table 2.9). A total of 31 non-grassland species were observed in the introduced vegetation of both the CRP and CREP studies; 14 species were observed in both studies, 3 species were observed only in the introduced CREP vegetation, and 14 species were observed only in the CRP fields studied in 1991-92 (Table 2.9).

Table 2.8. Mean [SE] vegetative characteristics of 1.) introduced Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, mid-summer (15 June – 14 July) 2005 and mid-summer 2006 (n = 7), and 2.) Conservation Reserve Program (CRP) grasslands, Gratiot County, Michigan, July 1991 and July 1992 (n = 3). CREP fields were planted in 2002 and CRP fields were planted in 1988.

	CREP Introduced Grasslands (5.5 – 17.1 ha) [A]		CRP Introduced Grasslands (8.1 – 19.8 ha) [BC]	
Characteristic	2005	2006	1991	1992
% Bare Ground	2.5 [1.0]	1.1 [0.6]	~0.5	~3.2
% Litter Cover	9.6 [2.9]	35.2 [5.1]	~0.5	4.3 [3.8]
% Total Canopy	87.9 [3.8]	63.8 [5.4]	99.0 [1.8]	92.5 [4.5]
% Dead Canopy	2.3 [1.6]	0.3 [0.1]	6.6 [0.6]	10.5 [0.3]
% Live Canopy	85.5 [3.7]	63.4 [5.5]	92.4 [2.9]	81.9 [4.8]
% Live Grass	42.7 [5.0]	27.0 [1.5]	~55.9	~51.1
% Live Woody	0.0 [0.0]	0.0 [0.0]	0.0 [0.0]	0.0 [0.0]
% Live Forb	42.8 [6.4]	36.4 [6.1]	~36.5	~30.8
Litter Depth (cm)	5.8 [0.6]	5.6 [0.6]	NA	3.4 [0.8]
Horizontal Cover (dm)	6.2 [0.5]	8.2 [0.4]	10.8 [0.1]	9.0 [0.4]

[A] *Planting mixture all fields: 23% Timothy, 23% Orchardgrass, 27% Alfalfa, 27% Red Clover; planted 12.4 kg/ha.*

[B] *Planting mixture 2 fields: 22% Timothy, 34% Orchardgrass, 22% Alfalfa, 22% White Sweet Clover; planted 10 kg/ha.*

[C] *Planting mixture 1 field: 22% Timothy, 45% Orchardgrass, 22% Alfalfa, 11% White Sweet Clover; planted 10 kg/ha.*

Table 2.9. Avian species observed in 1.) introduced vegetation of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006, and 2.) introduced vegetation of Conservation Reserve Program (CRP) grasslands, Gratiot County, Michigan, 1991-1992. Avian densities (# individuals/ha) [SE] are given for species with successful detection probability modeling ($p \geq 0.10$). In CREP vegetation, species observed too rarely to support detection probability modeling (i.e., observed in < 10% of surveys) or with detection probability estimates <0.10 are noted as present when observed in at least one survey and as undetected when unobserved. Detection probability modeling was not performed for avian species observed in CRP vegetation.

Common Name	Scientific Name	2005	2006	1991	1992
Grassland Specialist Species					
Bobolink [A]	*Dolichonyx oryzivorus*	1.33 [0.37]	0.68 [0.27]	present	present
Dickcissel [AB]	*Spiza americana*	0.08 [0.07]	0.05 [0.03]	undetected	present
Eastern Meadowlark	*Sturnella magna*	0.004 [0.004]	0.01 [0.01]	present	present
Grasshopper Sparrow [AB]	*Ammodramus savannarum*	0.00 [0.00]	0.01 [0.01]	present	present
Horned Lark	*Eremophila alpestris*	undetected	undetected	present	present
Northern Harrier [A]	*Circus cyaneus*	present	undetected	present	present
Ring-necked Pheasant	*Phasianus colchicus*	0.03 [0.02]	0.13 [0.08]	present	present
Savannah Sparrow	*Passerculus sandwichensis*	0.37 [0.12]	0.94 [0.32]	present	present
Sedge Wren [A]	*Cistothorus platensis*	0.00 [0.00]	0.06 [0.03]	present	present
Vesper Sparrow	*Pooecetes gramineus*	0.01 [0.01]	0.02 [0.01]	present	present

Table 2.9. Continued.

Common Name	Scientific Name	2005	2006	1991	1992
Other Species					
American Bittern	*Botaurus lentiginosus*	undetected	undetected	present	undetected
American Crow	*Corvus brachyrhynchos*	undetected	undetected	undetected	present
American Goldfinch	*Carduelis tristis*	0.08 [0.03]	0.26 [0.07]	present	present
American Robin	*Turdus migratorius*	present	present	present	undetected
Baltimore Oriole	*Icterus galbula*	undetected	undetected	undetected	present
Blue Jay	*Cyanocitta cristata*	undetected	undetected	undetected	present
Blue-winged Teal	*Anas discors*	undetected	undetected	undetected	present
Barn Swallow	*Hirundo rustica*	NA*	present	undetected	undetected
Brown Thrasher	*Toxostoma rufum*	undetected	present	undetected	present
Brown-headed Cowbird	*Molothrus ater*	undetected	present	undetected	undetected
Chipping Sparrow	*Spizella passerina*	undetected	present	undetected	present
Clay-colored Sparrow	*Spizella pallida*	present	present	undetected	undetected
Common Yellowthroat	*Geothlypis trichas*	0.05 [0.04]	0.10 [0.06]	present	present

Table 2.9. Continued.

Common Name	Scientific Name	2005	2006	1991	1992
Eastern Bluebird	*Sialia sialis*	undetected	undetected	undetected	present
Eastern Kingbird	*Tyrannus tyrannus*	present	0.004 [0.003]	present	present
Eastern Phoebe	*Sayornis phoebe*	undetected	undetected	undetected	present
European Starling	*Sturnus vulgaris*	undetected	undetected	present	undetected
Field Sparrow	*Spizella pusilla*	undetected	undetected	present	present
House Wren	*Troglodytes aedon*	present	undetected	undetected	present
Indigo Bunting	*Passerina cyanea*	present	present	present	undetected
Mallard	*Anas platyrhynchos*	undetected	undetected	present	present
Mourning Dove	*Zenaida macroura*	present	present	present	undetected
Northern Bobwhite Quail	*Colinus virginianus*	present	present	undetected	present
Northern Mockingbird	*Mimus polyglottos*	undetected	undetected	undetected	present
Red-tailed Hawk	*Buteo jamaicensis*	undetected	undetected	undetected	present
Red-winged Blackbird	*Agelaius phoeniceus*	2.30 [0.29]	2.97 [0.36]	present	present
Ruby-throated Hummingbird	*Archilochus colubris*	undetected	undetected	present	present
Song Sparrow	*Melospiza melodia*	0.46 [0.08]	0.76 [0.14]	present	present

Table 2.9. Continued.

Common Name	Scientific Name	2005	2006	1991	1992
Tree Swallow	*Tachycineta bicolor*	NA*	present	undetected	present
White-throated Sparrow	*Zonotrichia leucophrys*	undetected	undetected	undetected	present
Yellow Warbler	*Dendroica petechia*	undetected	undetected	undetected	present

[A] *Michigan Priority Landbird (Knutson et al. 2001, Matteson et al. 2009)*
[B] *Michigan Species of Special Concern*
* *Barn Swallow and Tree Swallow were not included in the 2005 survey*.

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CHAPTER 3: GRASSLAND BIRD DETECTION AND OCCUPANCY IN MICHIGAN CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) GRASSLANDS

INTRODUCTION

Breeding Bird Survey (BBS) data indicate that grassland birds have declined more rapidly, more consistently, and over a wider geographic area than any other guild of North American birds (Sauer et al. 1995, Knopf 1996). Six grassland bird species found in Michigan are among those that have been designated as Michigan priority land birds by the bird conservation group Partners in Flight: Bobolink, Dickcissel, Grasshopper Sparrow, Henslow's Sparrow, Northern Harrier, and Sedge Wren (Knutson et al. 2001, Matteson et al. 2009). Dickcissel, Grasshopper Sparrow, and Northern Harrier are also state species of special concern, and Henslow's Sparrow is a state endangered species.

Native temperate grasslands currently have the lowest protected area compared to converted area of any major biome (Hoekstra et al. 2005) and are one of the most endangered ecosystems in the world (Sampson and Knopf 1996). Farmland set-aside programs such as the Conservation Reserve Enhancement Program (CREP) may diminish or reverse the decline of grassland bird species by increasing the area of native and non-native grasslands in the U.S. (Herkert 2009). Geographic variation in the effects of set-aside lands on wildlife exists due to local and regional climatic differences and variation in program implementation, including differences in management practices (i.e., prescribed burning; Heard et al. 2000, Riffell et al. 2008). As a result, it is important to conduct assessments of the impacts of farmland set-aside programs on grassland birds and other wildlife at sub-national scales, such as state, ecoregional, or major watershed levels.

Grassland bird habitat requirements have also been found to vary among geographic regions (e.g., Johnson and Igl 2001, Winter et al. 2006). For example, Winter et al. (2006) found that the density of 3 grassland bird species varied across relatively small regional scales (i.e., separated by about 50 km) in the tallgrass prairie of southeastern North Dakota and northwestern Minnesota. Regional differences in grassland bird habitat selection highlight the importance of evaluating grassland bird habitat use at local scales, consistent with this assessment of grassland bird occupancy in CREP fields of the Saginaw Bay watershed in Michigan. This is the first study in Michigan to examine grassland bird occupancy of CREP grasslands. The goal of this analysis was to model grassland bird occupancy as a function of CREP field size and type for all observed grassland bird species that were adequately detected within two different types of CREP grassland plantings, CREP conservation practice (CP) 23 and CP1 fields.

METHODS

This study occurred within the 8,219 km^2 Sandusky Lake Plain sub-subsection of Michigan's Lower Peninsula (Albert 1995). This sub-subsection consists of flat clay lake plain, bordered by shoreline dunes and sand plain. In the center of the flat clay lake plain, long and narrow till plains and end moraine ridges parallel either Saginaw Bay or Lake Huron (Albert 1995). Prior to European settlement, extensive wet and wet-mesic prairies occurred inland from the coastal Great Lakes marshes of Saginaw Bay, but largely due to agricultural conversion, these prairies now occur only as small remnants, generally on State-owned lands (Albert 1995).

In Michigan, 22,983 ha of privately-owned land in the Saginaw Bay, Lake Macatawa, and River Raisin watersheds were enrolled in CREP at the time of this study (USDA 2007). From CREP lands in the Saginaw Bay watershed, 14 CREP grassland fields in Tuscola County, ranging in size from 6.8 to 23.6 ha, were selected for study (Figure 3.1). All grassland study fields were planted in 2002 and were surveyed for birds in 2005 and 2006. Seven study fields were CREP conservation practice (CP) type 23 (hereafter CP23) that had been planted with a seed mixture of native grasses and native and introduced wildflowers over 100% of the field area, and 7 study fields were CP type 1 (hereafter CP1) fields that had been planted with introduced grasses and legumes over 70% of the field area, with the remaining 30% of the field area planted with the same native seed mixture used in CP23 fields (Figure 3.1). Native sections of CP1 fields were established to provide winter shelter for wildlife (e.g., Ring-necked Pheasant, *Phasianus colchicus*) because native grasses tend to remain upright under winter conditions whereas introduced grasses generally collapse (USDA 2000a, USDA 2000b). In two CP1 study fields, the native seed mixture did not successfully establish and the field area was composed only of introduced vegetation (Figure 3.1). In 2005, the native seed mixture cost $16.00/pound (i.e., $104.00/acre or $256.99/hectare) and the introduced seed mixture cost $1.90/pound (i.e., $20.90/acre or $51.65/hectare) plus 6% sales tax through the Tuscola County Conservation District.

Within CREP, CP23 fields are wetland restorations and all CP23 fields had hydric soils. A small wetland (i.e., 3-10% of upland area) was created in each CP23 field by cutting drain tiles, and if necessary, by creating a push-out, or shallow depression in the soil. The CREP program offers a whole-field native grassland conservation practice without a wetland component, CP2, which was rare in the Saginaw Bay area. Due to the scarcity of CP2 fields, the upland portions of CP23 fields were used for comparison to the introduced portions of CP1 fields and to CP1 whole fields. Wetlands and a buffer of 25 m surrounding the wetlands were not included in CP23 avian surveys. The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha), and the introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha).

Avian data were collected using line-transect sampling (Edwards et al. 1981). The total area of each study field (i.e., except CP23 wetlands and associated wetland buffers) was surveyed every 2 weeks from 1 June – 14 August in 2005 (n=5 surveys in each field) and from 15 May – 14 August in 2006 (n=6 surveys per field). Surveys took place along transects running the length of each field. The first transect was established 25 m from a randomly selected field corner and subsequent transects were established at 50 m intervals across the width of the field. An investigator recorded the species and location of each individual bird observed. If an individual bird was flushed by the observer, the new location of the bird was noted in order to prevent counting the same bird more than once. If a bird was observed but could not be identified, the location of the bird was recorded and the species was designated as unknown. Unidentified individuals were not included in analyses. Avian surveys were conducted after sunrise and before 12:00 p.m., and surveys were not performed in the rain or if wind speeds were over 16 kph.

Avian species detection, *p*, and occupancy, ψ, probabilities in CP23 and CP1 study fields were estimated using single-season occupancy models (MacKenzie et al.2002) in the program PRESENCE (Hines 2006). The use of single season models was based on the results of likelihood ratio testing (MacKenzie et al. 2006, Nichols et al. 2008), which indicated that the more highly parameterized multi-season models did not fit the data significantly better than single season models ($p > 0.05$).

For studies with small sample sizes, model ranking using Akaike's information criterion corrected for small sample sizes (AIC_c) is recommended (Burnham and Anderson 2002). The top-ranking detection model for each avian species was identified as having the lowest AIC_c value among all candidate models with constant occupancy, $\psi(\cdot)$ (MacKenzie et al. 2002). Candidate (i.e., successful) detection models were those that achieved 1) numerical convergence of parameter estimates to >6 significant figures, 2) a balanced variance-covariance matrix, and 3) small- or moderately-sized standard errors of betas, the untransformed estimates of coefficients for covariates (i.e., standard error was no greater than 5 times the corresponding beta) (D. MacKenzie, Program PRESENCE – FAQ at http://www.phidot.org/forum, accessed October 2009).

Avian detection probability was calculated for each species observed in at least 10% of the total number of surveys (MacKenzie et al. 2002). Detection probability estimates were obtained from the top-ranking detection model for each species. Potential detection covariates were ordinal date and CREP field type. Ordinal date was standardized to reduce biases in parameter estimation (MacKenzie et al. 2002). When field type and/or ordinal date were covariates in the top-ranking detection model, multiple estimates of detection resulted, and the lowest estimate was reported. For species where no detection models were successfully generated using the full data set, detection probability estimates for individual study sites were examined, and if detection probability estimates for a subset of the data appeared adequate (e.g., detection estimates for all fields in 2006 were $\geq$0.10), then detection modeling was performed on this data subset. For species where the naïve estimate of ψ equaled 1, study fields were fully occupied, and detection modeling was conducted with ψ fixed to 1.

Occupancy modeling was performed only for species with detection probability estimates $\geq$0.10. Potential occupancy covariates were ordinal date, CREP field type, and CREP field size. Ordinal date and field size were standardized to reduce biases in parameter estimation (MacKenzie et al. 2002).

For each avian species with a minimum detection probability estimate ≥ 0.10, all possible occupancy models were constructed using the best-fitting detection model as a function of the occupancy covariates (i.e., ψ[covariates], *p*[as in best fitting detection model]) (Roloff et al. *in press*). Candidate occupancy models met the convergence, variance-covariance matrix, and standard error size criteria as described above. The global model was identified as the most parameterized occupancy model. Global model fit was evaluated using the chi-square goodness-of-fit statistic to estimate an overdispersion parameter, $\hat{c}$, from 2,000 bootstrap iterations (MacKenzie and Bailey 2004, MacKenzie et al. 2006). Ecological models are commonly overdispersed (i.e., $\hat{c} \neq 1$), and adjustment of the model selection criteria to account for it is advised (MacKenzie and Bailey 2004, MacKenzie et al. 2006). Overdispersed occupancy models were ranked using $QAIC_c$, an adjusted AIC_c with standard errors inflated using $\hat{c}$, the overdispersion parameter (McCullagh and Nelder 1989, MacKenzie et al. 2006). High overdispersion parameter values of a global model indicate that any derived occupancy models would not have useful structure (Burnham and Anderson 1998), so for species with extremely overdispersed global models (i.e., $\hat{c} > 4$), no occupancy models were successful. For species with suitable global models (i.e., $\hat{c} \leq 4$), all occupancy models in the set of candidate models were ranked according to AIC_c or $QAIC_c$ (Burnham and Anderson 2002). When one best occupancy model did not emerge, competing models (i.e., models where ΔAIC_c or $\Delta QAIC_c \leq 2.00$) were model averaged to derive a composite model (Burnham and Anderson 2002).

RESULTS

A total of 34 avian species were observed in CREP study fields during the 2 years of this study (Appendix 3.2). In 2005, 23% of individual birds observed during surveys were unidentified, and in 2006 surveyors were more experienced and only 8% of observed individuals were unidentified. Using the grassland bird list developed for the BBS by the U.S. Geographic Survey (Sauer et al. 1995), 11 of the observed species were identified as grassland specialists: Bobolink (*Dolichonyx oryzivorus*), Dickcissel (*Spiza americana*), Eastern Meadowlark (*Sturnella magna*), Grasshopper Sparrow (*Ammodramus savannarum*), Henslow's Sparrow (*Ammodramus henslowii*), Horned Lark (*Eremophila alpestris*), Northern Harrier (*Circus cyaneus*), Ring-necked Pheasant, Savannah Sparrow (*Passerculus sandwichensis*), Sedge Wren (*Cistothorus platensis*), and Vesper Sparrow (*Pooecetes gramineus*). Six of these grassland specialist species are of special conservation priority in Michigan (Appendix 3.2). Bobolink, Dickcissel, Grasshopper Sparrow, Henslow's Sparrow, Northern Harrier, and Sedge Wren are designated as Michigan priority landbirds by the bird conservation group Partners in

Flight (Knutson et al. 2001, Matteson et al. 2009), while Dickcissel, Grasshopper Sparrow, and Northern Harrier are also state species of special concern and Henslow's Sparrow is a state endangered species (Appendix 3.2).

Bobolink and Savannah Sparrow were the most abundant grassland bird species in native and introduced grasslands (Appendix 3.2). Three grassland bird species were too rarely observed to estimate their detection probabilities (i.e., observed in less than 10% of surveys): Henslow's Sparrow, Horned Lark, and Northern Harrier. Detection probability modeling was performed for the 8 other grassland specialist species observed during this study. Likelihood ratio testing indicated that more highly parameterized multi-season models did not fit the data significantly better than simpler single-season models ($p > 0.05$), so single season models were used in all detection and occupancy modeling efforts (MacKenzie et al. 2006, Nichols et al. 2008).

Adequate detection probability estimates (i.e., $p \geq 0.10$) were achieved for all 8 species, with detection probability estimates ranging from 0.12 to 0.81 (Table 3.1). However, an adequate detection probability estimate was obtained only when a subset of the data was used in the case of one species, Sedge Wren (2006 data), most likely because Sedge Wren were rare in 2005 (mean density = 0.01, SE = 0.01 in CP23 fields; mean density = 0.0, SE = 0.0 in introduced vegetation of CP1 fields; 2005 Sedge Wren mean density information was reported for CP23 fields and CP1 introduced vegetation in Appendix 3.2 because detection probability estimates were adequate for Sedge Wren in native and introduced CREP grassland plant communities [Chapter 4]).

CREP field type and/or standardized ordinal date were included in the best detection models for Bobolink, Eastern Meadowlark, Ring-necked Pheasant, Savannah

Sparrow, and Sedge Wren (Table 3.1). The detection probability of Bobolink varied between 2005 and 2006, but detection probability estimates were >0.63 regardless of year, indicating that Bobolink was readily detected throughout each breeding season (Figure 3.2). Eastern Meadowlark was reliably detected in CP23 and CP1 fields, but the species tended to be more likely to be detected in CP23 fields, although the 95% confidence intervals for detection estimates of the two field types were overlapping (i.e., 95% confidence interval was 0.10 – 0.16 in CP23 fields and 0.08 – 0.15 in CP1 fields; Figure 3.3). Ring-necked Pheasant detection probability varied between 2005 and 2006, but detection probability estimates were >0.17 in both years, indicating the species was reliably detected throughout both breeding seasons (Figure 3.4). Ring-necked Pheasant was adequately detected in CP23 and CP1 fields, but the species tended to be more likely to be detected in CP23 fields, although the 95% confidence intervals for detection estimates of the two field types were overlapping (i.e., 95% confidence interval was 0.21 – 0.46 in CP23 fields and 0.05 – 0.32 in CP1 fields; Figure 3.4). Detection probability estimates for Savannah Sparrow were >0.81 regardless of year, indicating the species was readily detected throughout each breeding season (Figure 3.5). The detection probability of Sedge Wren was >0.15 throughout 2006, indicating the species was reliably detected throughout the breeding season, but Sedge Wren detection probability was positively associated with standardized ordinal date, suggesting that optimal detection of this species occurred toward the end of the breeding season (Figure 3.6).

Occupancy modeling was performed for all 8 grassland bird species with detection probabilities ≥0.10 (i.e., Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and

Vesper Sparrow; Table 3.1). The naïve estimate of occupancy was 1.0 for Ring-necked Pheasant and Savannah Sparrow, indicating that all surveyed fields were occupied. Global occupancy model fit was evaluated for the remaining 6 grassland bird species, and for 2 species, Bobolink and Eastern Meadowlark, the global occupancy models were extremely overdispersed (i.e., $\hat{c} > 4$; Table 3.2), indicating the occupancies of these 2 species could not be reliably estimated from this study. Global model fit indicated that informative occupancy models could be derived for the 4 remaining grassland bird species: Dickcissel, Grasshopper Sparrow, Sedge Wren (2006 data), and Vesper Sparrow (Table 3.2).

A single best-fitting occupancy model was identified for Dickcissel that included field size and was not overdispersed ($\hat{c} = 0.90$) (Table 3.3). This model accounted for 64% of the total AIC_c weight among four candidate models (Table 3.3). The naïve estimate of occupancy was 0.43, while the detection corrected estimates of occupancy by site ranged from 0.003 to 0.85. The beta estimate for the effect of standardized field size on occupancy was negative (beta = -2.21, SE = 1.44; Table 3.4), suggesting that larger CREP fields had lower Dickcissel occupancy. However, the 95% confidence interval of the beta estimate indicated that field size was not a significant determinant of Dickcissel occupancy (Table 3.4).

For Grasshopper Sparrow, the intercept-only model ranked highest ($\hat{c} = 2.79$; $w = 0.84$; Table 3.5), indicating that none of the covariates tested in this study were important correlates of Grasshopper Sparrow occupancy. The naïve estimate of occupancy was 0.64, and the detection corrected occupancy estimate was 0.65.

Occupancy modeling for Sedge Wren (2006 data) resulted in two competing models (i.e., $\Delta AIC_c \leq 2$) that together accounted for 80% of the total AIC_c weight (Table 3.6). The highest ranking occupancy model was the intercept-only model ($\hat{c}$ = 1.65), and the competing model included CREP field type ($\hat{c}$ = 1.61; Table 3.6). The naïve occupancy estimate for Sedge Wren was 0.64, and the detection corrected occupancy estimates by site ranged from 0.44 in CP1 fields to 0.88 in CP23 fields. The model averaged beta estimate of the effect of CREP field type on occupancy was negative (beta = -2.25, SE = 1.53; Table 3.4), indicating that Sedge Wren occupancy was greater in CP23 fields than in CP1 fields (i.e., CP23 fields were assigned a value of 0 and CP1 fields were categorized as 1 during modeling). However, the 95% confidence interval for the field type parameter included zero indicating that CREP field type was not a significant determinant of Sedge Wren occupancy (Table 3.4).

Two competing occupancy models (i.e., $\Delta AIC_c \leq 2$) were identified for Vesper Sparrow that together accounted for 87% of the total AIC_c weight among 3 candidate models (Table 3.7). Similar to the findings for Sedge Wren, the intercept-only model ($\hat{c}$ = 0.70) was the top-ranked occupancy model for Vesper Sparrow and the competing model included CREP field type ($\hat{c}$ = 0.67; Table 3.7). The naïve estimate of occupancy was 0.86, while the detection corrected estimates of occupancy by site ranged from 0.76 in CP1 fields to 1.0 in CP23 fields. The beta estimate for CREP field type on occupancy was negative (beta = -21.83, SE = 13.42; Table 3.4), indicating that Sedge Wren occupancy was greater in CP23 fields than in CP1 fields (i.e., CP23 fields were assigned a value of 0 and CP1 fields were categorized as 1 during modeling). However, the 95%

confidence interval of the field type beta estimate included zero indicating that CREP field type was not a significant determinant of Vesper Sparrow occupancy (Table 3.4).

DISCUSSION

Avian surveys of CP23 and CP1 CREP fields in Michigan's Saginaw Bay watershed resulted in the observation of 34 species, 11 of which were grassland specialist species (Sauer et al. 1995). Sufficient numbers of 8 grassland species were observed to allow detection probability and occupancy modeling (i.e., present in ≥10% of surveys). All 8 species had adequate detection probability estimates (i.e., ≥ 0.10), which were above 0.25 for 4 species, indicating that the survey technique used in this study was an appropriate methodology for documenting occupancy. Covariates included in the top-ranking occupancy models of grassland bird species included field type (i.e., CP23 or CP1) and CREP field size. Although the effects of these covariates were not significant, findings of this study are likely constrained by small sample sizes, and these covariates warrant evaluation in further research.

Grassland bird species that occurred in this study were similar to those documented in comparable research. Published studies on grassland bird species in CREP fields of the Lake States are rare; only one study in Pennsylvania was identified (Wentworth et al. 2010). Wentworth et al. (2010) found 8 grassland bird species that were also observed in my study: Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Henslow's Sparrow, Ring-necked Pheasant, Savannah Sparrow, and Vesper Sparrow (i.e., Horned Lark, Northern Harrier, and Sedge Wren were not observed in Pennsylvania but were present in Michigan CREP grasslands). The grassland bird species that occurred in my study were also similar to those that occurred in comparable work in grasslands of a similar farmland set-aside program, the Conservation Reserve Program (CRP). In an investigation of the effects of different age classes of Michigan CRP fields on avian diversity, density, and productivity, Millenbah (1993) found 10 grassland bird species, all of which were also observed in my study along with Henslow's Sparrow. Nine grassland bird species observed in my surveys also occurred in CRP fields in a large-scale analysis of Midwestern bird abundance and nesting in CRP fields and cropland (i.e., Horned Lark and Northern Harrier were absent in CRP fields) (Best et al. 1997). With the exception of Savannah Sparrow, the grassland species that were observed in my surveys were also observed in a study of CP1 and switchgrass (*Panicum virgatum)* dominated CP2 CRP fields in Missouri (McCoy et al. 2001). These comparisons indicate that grassland bird community composition is similar in CREP and CRP, however other research suggests that grassland bird abundances vary within and between CREP and CRP (Chapter 2, McCoy et al. 2001, Wentworth 2010), underscoring the importance of evaluating CREP and other such conservation programs at local scales (e.g., within states, ecoregions, or major watersheds). It is important to note that the consistency of results from studies in different regions may be affected by differences in grassland composition, age or successional stage, and structure.

The survey methodology of this study generated adequate detection probability estimates for all grassland species observed in sufficient numbers to perform detection probability modeling (i.e., observed in ≥10% of surveys). However, it is possible to recommend methodological improvements for future studies in this or similar study systems. Improvements to survey methodology to increase the probability of detection would include restricting avian survey efforts to the period of the breeding season prior to the latest truncation date of expected species or species of interest. Additionally, increasing the number of surveys prior to the truncation date would improve the probability of detecting rare species, and increasing the number of fields surveyed would ease or remove analytical constraints due to small sample sizes.

With one exception, global occupancy models included only one occupancy covariate, likely due to constraints caused by small sample size (i.e., n = 14). Despite the constraints of small sample size, field size or field type appeared in top-ranking occupancy models for three of the four grassland bird species for which occupancy modeling was successful. The 95% confidence intervals for field size and field type consistently included 0, but 0 occurred near the tails of the covariate distributions suggesting that non-zero central tendencies existed. Hence, future studies should strive to include more replicates to further evaluate the correlations between field size and field type on grassland bird occupancy.

I expected a relationship between CREP field type and grassland bird occupancy because of the different proportions of native plant community area in CP23 (100%) and CP1 (70%) CREP fields. Due to the coevolutionary history of native grasses and grassland birds (Sampson and Knopf 1996) and clear differences in native and introduced plant community composition and structure (Chapter 2), I anticipated that grassland bird occupancy would be higher in CP23 fields than in CP1 fields. For the species exhibiting a relationship to field type (i.e., Sedge Wren and Vesper Sparrow), higher occupancy estimates were consistently associated with CP23 fields. My results indicate that these 100% native fields are providing habitat conditions that support higher occupancy for some grassland bird specialists.

Because other research has shown that habitat patch size is an important positive influence on grassland bird occupancy (e.g., Ribic et al. 2009), I predicted that larger CREP field size would be associated with higher occupancies of grassland birds in my study. CREP field size emerged as a correlate to the occupancy Dickcissel, and occupancy estimates among sites exhibited a broad range (0.01 to 0.86) in response to field size. Unexpectedly, field size was negatively related to Dickcissel occupancy, with higher occupancy estimates for smaller-sized CREP fields. My finding contradicts other research that found a positive relationship of patch size to Dickcissel occurrence (Bakker et al. 2002).

Combinations of within-field and landscape-scale variables have been identified as important covariates in other studies of grassland birds (e.g., Ribic and Sample 2001, Bakker et al. 2002, Cunningham and Johnson 2006, Winter et al. 2006). Grassland birds likely use a hierarchical process that is affected by variables at multiple scales when choosing breeding territories (Ribic et al. 2009, Fisher and Davis 2010). For example, Winter et al. (2006) found that grassland bird density was affected by a combination of habitat patch size, landscape, region, and local vegetation structure more than by local vegetation structure alone. In further analyses of my data, I will model grassland bird occupancy in native plant communities of CP23 CREP fields and introduced plant communities of CP1 CREP fields and examine the effect of within-field and landscape-scale variables on grassland bird occupancy (Chapter 4).

Management recommendations

My occupancy modeling results indicate that CREP fields, especially CP23 native grasslands, provide habitat for grassland birds in Michigan, a critical resource for an avian guild whose decline has occurred more rapidly, more consistently, and over a wider geographic area than any other guild of North American birds (Sauer et al. 1995, Knopf 1996). Naïve occupancy estimates were particularly high for Ring-necked Pheasant and Savannah Sparrow in all study fields. Occupancy estimates of several other grassland bird species (i.e., Grasshopper Sparrow, Sedge Wren, and Vesper Sparrow) were higher in the native CP23 grasslands than in CP1 fields, thus I recommend that CREP administrators prioritize the establishment of CP23 fields within the Saginaw Bay area of Michigan.

Because occupancy covariates differed among grassland species, I recommend that in similar studies the effect of occupancy covariates be evaluated for individual species within the grassland bird guild rather than for grassland birds as a group. I also advocate local (e.g., ecoregion, major watershed) evaluation of grassland bird habitat selection, since grassland bird habitat requirements have been found to vary among geographic regions (e.g., Johnson and Igl 2001), even at small regional scales, such as those separated by as little as 50 km (Winter et al. 2006). Additionally, local evaluation of the impacts of CREP on wildlife conservation are important, since geographic variation in the effects of set-aside lands on wildlife exists due to local and regional climatic differences and variation in program implementation, including differences in management practices (Heard et al. 2000, Riffell et al. 2008).

Maintaining and increasing the area of CREP grasslands, especially native grasslands, should be a grassland bird conservation strategy in Michigan. A trend toward reconversion of lands enrolled in set-aside programs such as CREP to crop production has been identified (Fargione et al. 2009), and I recommend that program administrators provide incentives for landowners to join or continue in CREP through an increase in the one-time program sign-up bonus when possible, as this strategy that has been found to generate a greater increase in CREP enrollment than did smaller increases in the annual payments made over the term of the CREP contract (Suter et al. 2008).

APPENDIX 3.1: CHAPTER 3 TABLES AND FIGURES

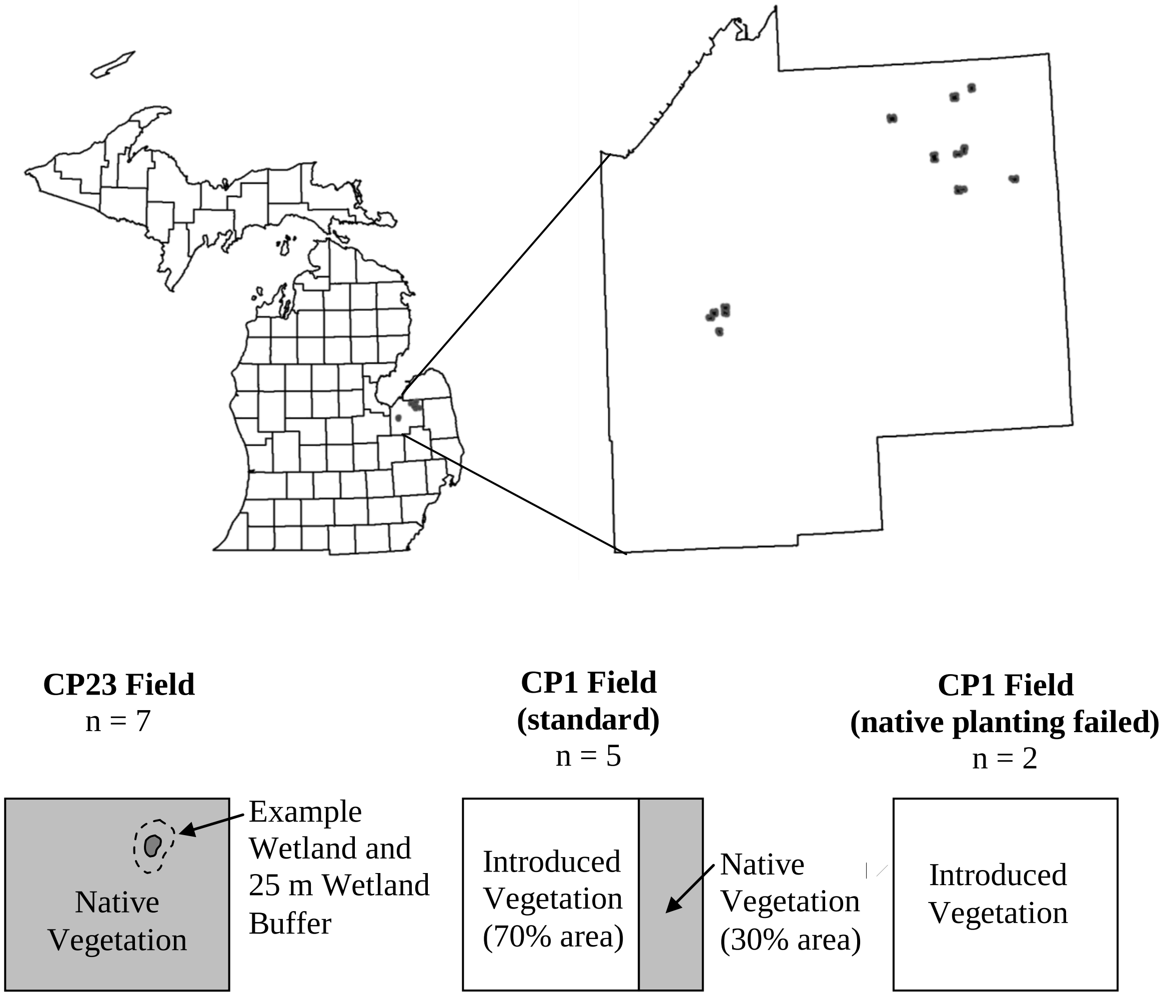

Fig. 3.1. Location, type, and number of Conservation Reserve Enhancement Program (CREP) study fields within Tuscola County, Saginaw Bay watershed, Michigan, 2005 and 2006. Wetlands and wetland buffers were excluded from avian surveys and vegetation sampling. The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). The introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha), and CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha).

Table 3.1. Grassland bird species, with corresponding best-fitting detection models (with constant occupancy, $\psi[\cdot]$) and detection probability estimates, observed in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. The lowest detection probability estimate was reported when multiple detection probability estimates were generated.

Common Name	Detection Model	$p(\cdot)$ [SE]
Bobolink	*p(Ordinal Date)*	0.63 [0.06]
Dickcissel	*p(.)*	0.35 [0.06]
Eastern Meadowlark	*p(CREP Field Type)*	0.13 [0.07]
Grasshopper Sparrow	*p(.)*	0.31 [0.05]
Ring-necked Pheasant	*p(CREP Field Type, Ordinal Date)*	0.17 [0.05]
Savannah Sparrow	*p(Ordinal Date)*	0.81 [0.05]
Sedge Wren, 2006 data	*p(Ordinal Date)*	0.12 [0.07]
Vesper Sparrow	*p(.)*	0.21 [0.04]

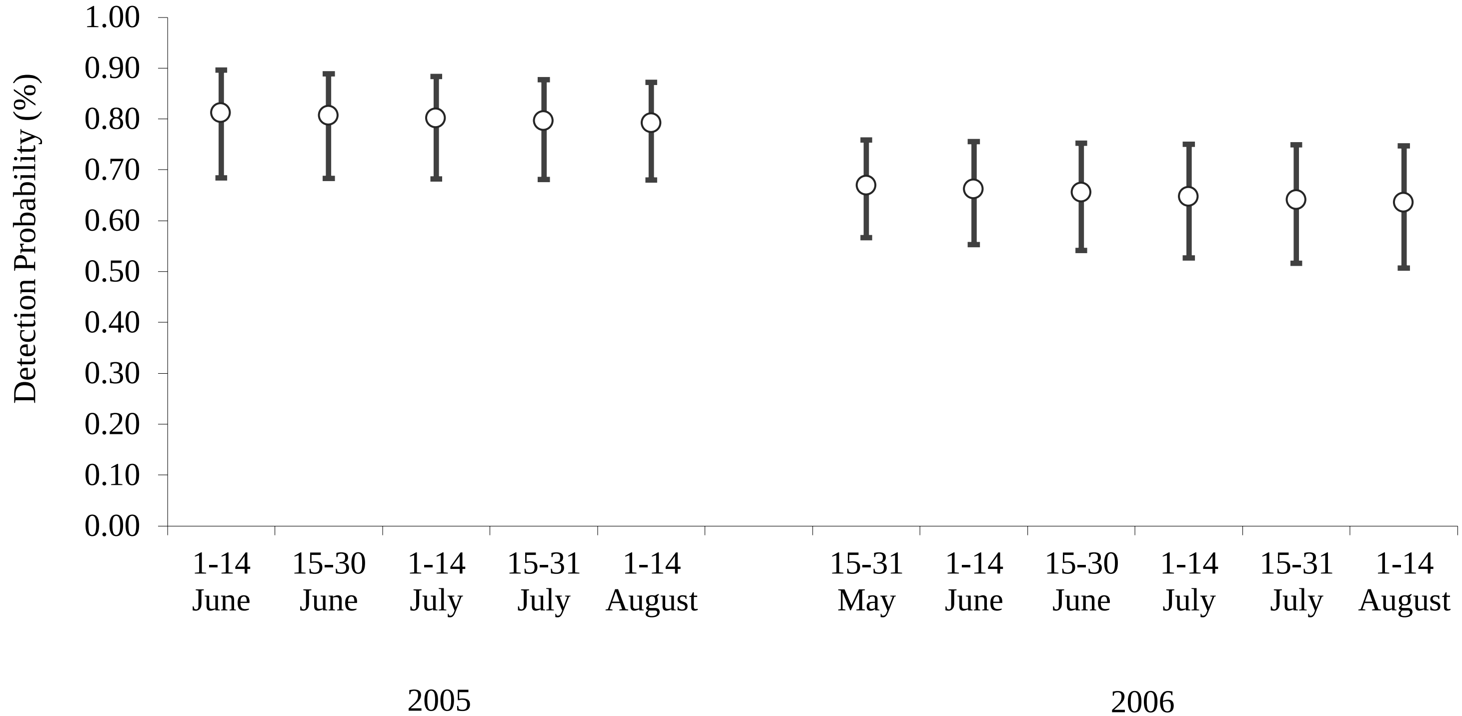

Fig. 3.2. Relationship between Bobolink detection probability and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Error bars indicate 95% confidence intervals.

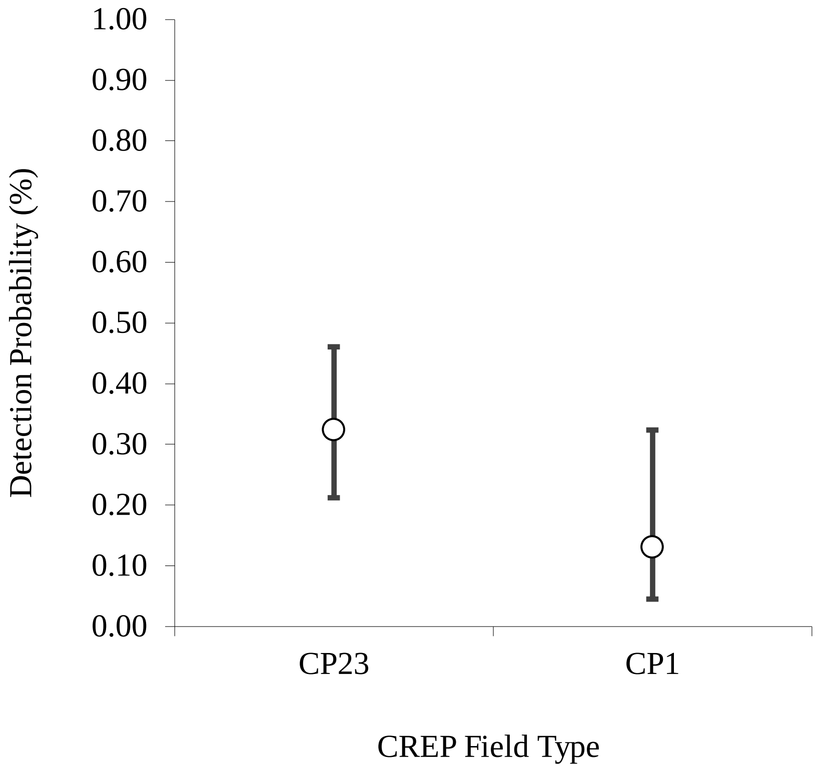

Fig. 3.3. Relationship between Eastern Meadowlark detection probability and Conservation Reserve Enhancement Program (CREP) administrative field type in CP23 and CP1 CREP grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006. Error bars indicate 95% confidence intervals.

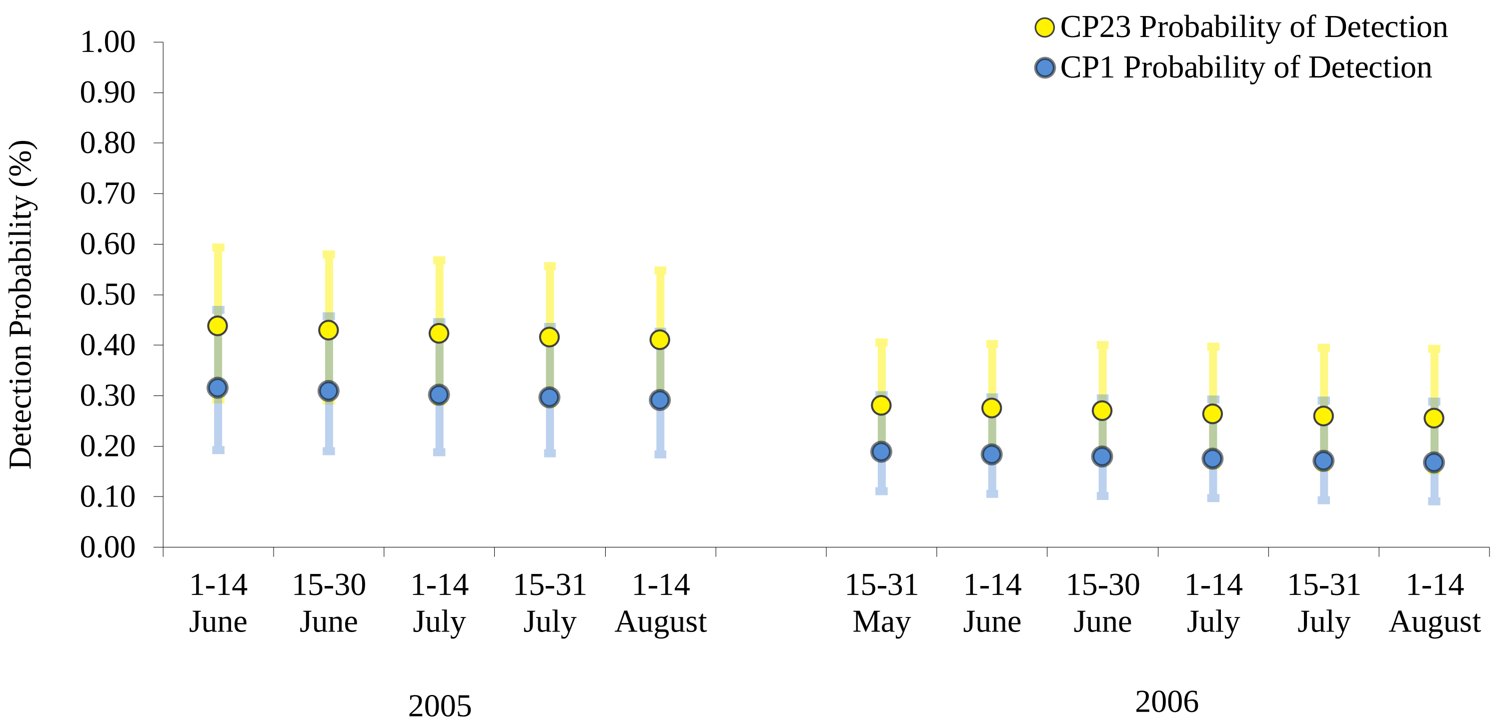

Fig. 3.4. Relationship between Ring-necked Pheasant detection probability, field type, and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006. Error bars indicate 95% confidence intervals.

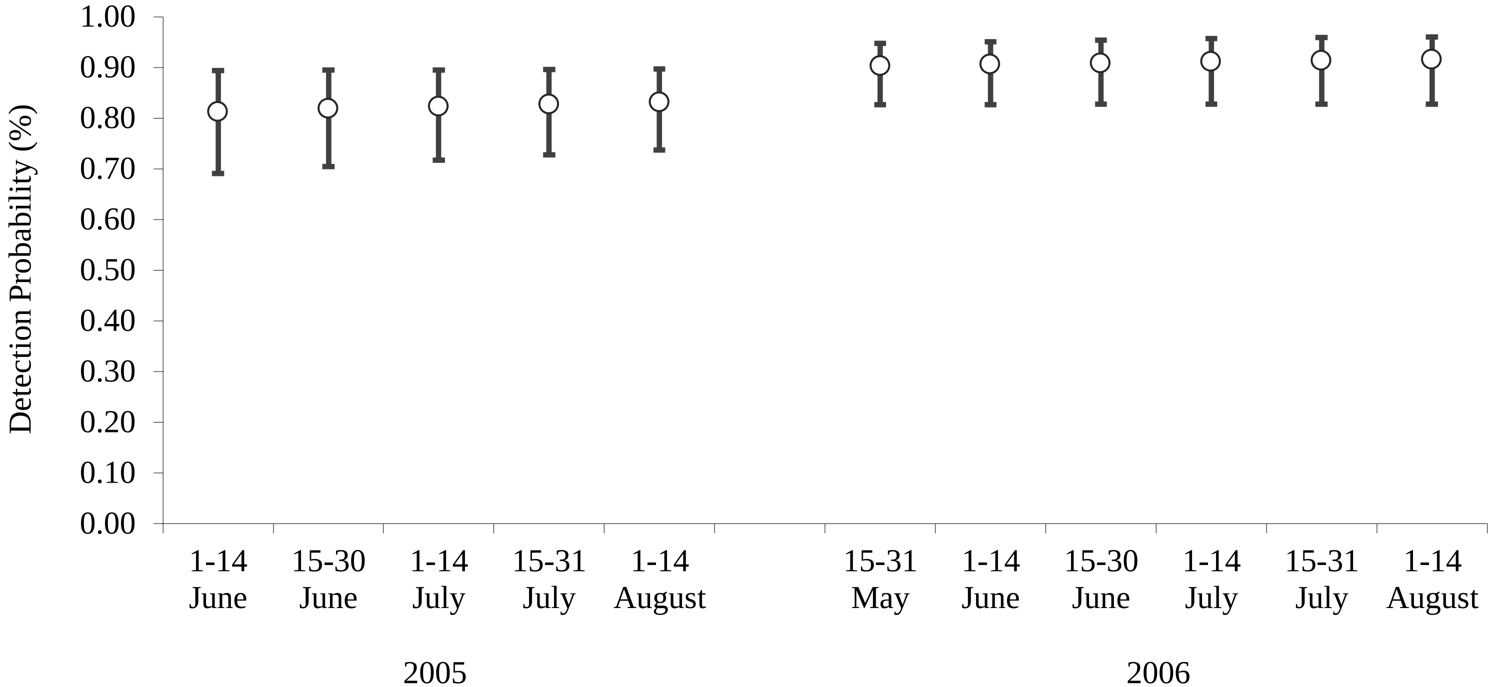

Fig. 3.5. Relationship between Savannah Sparrow detection probability and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Error bars indicate 95% confidence intervals.

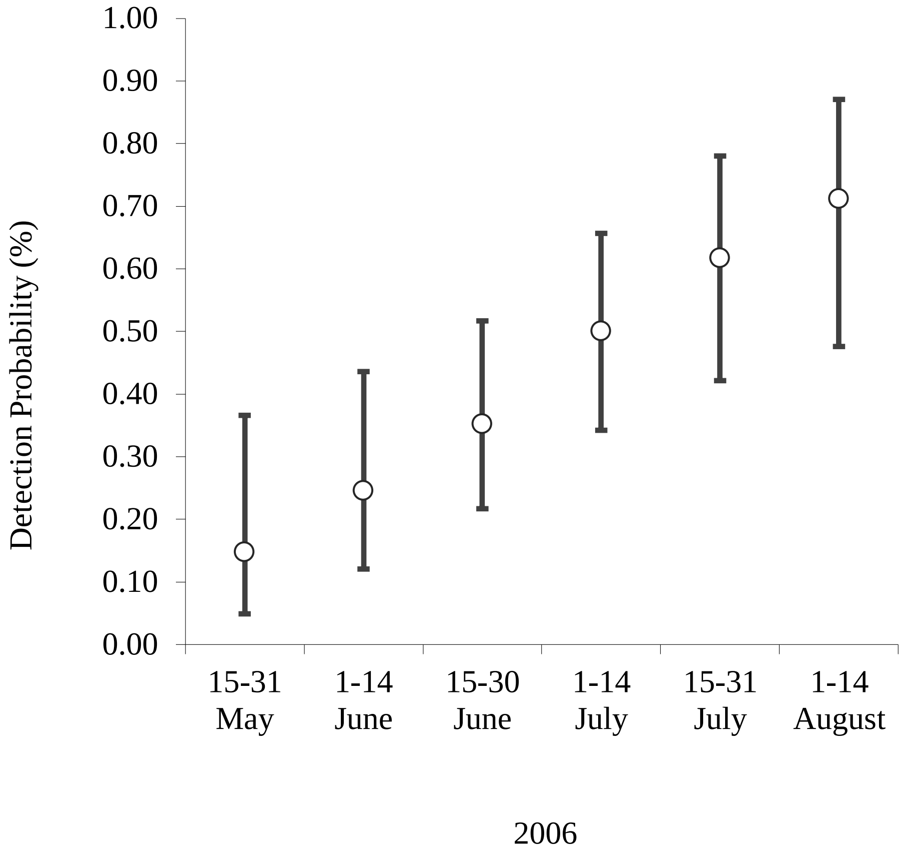

Fig. 3.6. Relationship between Sedge Wren detection probability and ordinal date in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2006. Error bars indicate 95% confidence intervals.

Table 3.2. Grassland bird species and global occupancy models with corresponding number of parameters, *K*, and overdispersion estimate, $\hat{c}$, observed in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Common Name	Global Model	*K*	$\hat{c}$
Bobolink	*Ψ(Field Size), p(Ordinal Date)*	4	6.03
Dickcissel	*Ψ(CREP Field Type, Field Size), p(.)*	4	0.92
Eastern Meadowlark	*Ψ(CREP Field Type), p(CREP Field Type)*	4	5.49
Grasshopper Sparrow	*Ψ(Field Size), p(.)*	3	2.78
Sedge Wren, 2006 only	*Ψ(CREP Field Type), p(Ordinal Date)*	4	1.66
Vesper Sparrow	*Ψ(CREP Field Type), p(.)*	3	0.67

Table 3.3. Summary of AIC occupancy model selection adjusted by the best detection model for Dickcissel in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(Field Size), p(.)	98.98	3	107.38	0.00	0.64
Ψ(.), p(.)	104.35	2	109.44	2.06	0.23
Ψ(CREP Field Type, Field Size), p(.)	98.90	4	111.34	3.96	0.09
Ψ(CREP Field Type), p(.)	104.35	3	112.75	5.37	0.04

Table 3.4. Parameter estimates for the best fitting occupancy models for Dickcissel and Grasshopper Sparrow and model averaged parameter estimates for Sedge Wren (2006 data only) and Vesper Sparrow in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

				95% Confidence Interval	
Species	Parameter	β Estimate	SE	Upper	Lower
Dickcissel	Standardized Field Size	-2.21	1.44	-5.02	0.61
Grasshopper Sparrow	Intercept	0.64	0.58	-0.49	1.77
Sedge Wren	CREP Whole-field Type	-2.25	1.53	-5.25	0.74
Vesper Sparrow	CREP Whole-field Type	-21.83	13.42	-48.13	4.47

Table 3.5. Summary of QAIC occupancy model selection adjusted by the best detection model for Grasshopper Sparrow in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	$QAIC_c$	$\Delta QAIC_c$	*w*
Ψ(.), p(.)	141.02	2	55.87	0.00	0.84
Ψ(Field Size), p(.)	140.95	3	59.15	3.28	0.16

Table 3.6. Summary of AIC occupancy model selection adjusted by the best detection model for Sedge Wren, 2006 data only, in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(Ordinal Date)	82.24	3	90.64	0.00	0.50
Ψ(CREP Field Type), p(Ordinal Date)	79.26	4	91.70	1.06	0.30
Ψ(Field Size), p(Ordinal Date)	80.83	4	93.27	2.63	0.13
Ψ(Ordinal Date), p(Ordinal Date)	82.23	4	94.67	4.03	0.07

Table 3.7. Summary of AIC occupancy model selection adjusted by the best detection model for Vesper Sparrow in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(.)	151.34	2	156.43	0.00	0.56
Ψ(CREP Field Type), p(.)	149.24	3	157.64	1.21	0.31
Ψ(Field Size), p(.)	150.95	3	159.35	2.92	0.13

APPENDIX 3.2: ALL AVIAN SPECIES OBSERVED

Appendix 3.2. Avian species observed in CP23 native fields, the introduced portion of CP1 fields, and CP1 whole fields, Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Avian densities (# individuals /ha [SE]) are given for species with detection probability estimates ≥0.10. Species observed too rarely to support detection probability modeling (i.e., observed in < 10% of surveys) or with detection probability estimates <0.10 are noted as present when observed in at least one survey and as undetected when unobserved.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native/ CP23	Introduced	CP1
Grassland Specialist Species							
Bobolink [A]	*Dolichonyx oryzivorus*	0.38 [0.15]	1.33 [0.37]	2.28 [0.93]	0.60 [0.26]	0.68 [0.27]	0.35 [0.15]
Dickcissel [AB]	*Spiza americana*	0.04 [0.03]	0.08 [0.07]	0.05 [0.06]	0.03 [0.01]	0.05 [0.03]	0.05 [0.03]
Eastern Meadowlark	*Sturnella magna*	0.12 [0.09]	0.004 [0.004]	0.003 [0.003]	0.06 [0.04]	0.01 [0.01]	0.03 [0.02]
Grasshopper Sparrow [AB]	*Ammodramus savannarum*	0.05 [0.02]	0.00 [0.00]	0.00 [0.00]	0.08 [0.03]	0.01 [0.01]	0.01 [0.01]
Henslow's Sparrow [AC]	*Ammodramus henslowii*	present	undetected	undetected	present	undetected	undetected
Horned Lark	*Eremophila alpestris*	undetected	undetected	undetected	present	undetected	undetected
Northern Harrier [A]	*Circus cyaneus*	undetected	present	undetected	present	undetected	undetected
Ring-necked Pheasant	*Phasianus colchicus*	0.15 [0.10]	0.03 [0.02]	0.07 [0.04]	0.12 [0.07]	0.13 [0.08]	0.07 [0.06]
Savannah Sparrow	*Passerculus sandwichensis*	0.98 [0.39]	0.37 [0.12]	0.55 [0.17]	2.14 [0.62]	0.94 [0.32]	1.22 [0.46]
Sedge Wren [A]	*Cistothorus platensis*	0.01 [0.01]	0.00 [0.00]	present	0.07 [0.02]	0.06 [0.03]	0.03 [0.02]
Vesper Sparrow	*Pooecetes gramineus*	0.07 [0.03]	0.01 [0.01]	0.02 [0.01]	0.06 [0.01]	0.02 [0.01]	0.06 [0.04]

Appendix 3.2 Continued.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native	Introduced	CP1
Other Species							
American Goldfinch	*Carduelis tristis*	0.33 [0.13]	0.08 [0.03]	0.41 [0.32]	0.24 [0.08]	0.26 [0.07]	0.29 [0.08]
American Robin	*Turdus migratorius*	undetected	present	undetected	undetected	present	present
American Woodcock	*Scolopax minor*	undetected	undetected	undetected	present	undetected	undetected
Barn Swallow	*Hirundo rustica*	NA*	NA*	NA*	present	present	present
Brown Thrasher	*Toxostoma rufum*	undetected	undetected	undetected	undetected	present	present
Brown-headed Cowbird	*Molothrus ater*	present	undetected	undetected	present	present	present
Chipping Sparrow	*Spizella passerina*	undetected	undetected	undetected	undetected	present	present
Clay-colored Sparrow	*Spizella pallida*	present	present	present	0.05 [0.03]	present	present
Common Yellowthroat	*Geothlypis trichas*	0.03 [0.02]	0.05 [0.04]	0.09 [0.06]	0.12 [0.04]	0.10 [0.06]	0.16 [0.12]
Eastern Kingbird	*Tyrannus tyrannus*	present	present	present	0.03 [0.01]	0.004 [0.003]	present
House Wren	*Troglodytes aedon*	present	present	undetected	undetected	undetected	undetected
Indigo Bunting	*Passerina cyanea*	present	present	present	undetected	present	present

Appendix 3.2 Continued.

Common Name	Scientific Name	2005 Native/ CP23	2005 Introduced	2005 CP1	2006 Native/ CP23	2006 Introduced	2006 CP1
Killdeer	*Charadrius vociferus*	present	undetected	undetected	present	undetected	undetected
Mallard	*Anas platyrhynchos*	present	undetected	undetected	present	undetected	undetected
Marsh Wren [B]	*Cistothorus palustris*	undetected	undetected	undetected	present	undetected	undetected
Mourning Dove	*Zenaida macroura*	present	present	present	present	present	present
Northern Bobwhite Quail	*Colinus virginianus*	undetected	present	present	undetected	present	undetected
Northern Flicker	*Colaptes auratus*	present	undetected	undetected	present	undetected	present
Red-winged Blackbird	*Agelaius phoeniceus*	0.64 [0.26]	2.30 [0.29]	2.18 [0.62]	0.73 [0.19]	2.97 [0.36]	2.11 [0.26]
Song Sparrow	*Melospiza melodia*	1.54 [0.35]	0.46 [0.08]	1.47 [0.47]	1.89 [0.37]	0.76 [0.14]	1.12 [0.12]
Tree Swallow	*Tachycineta bicolor*	NA*	NA*	NA*	0.01 [0.01]	present	0.06 [0.05]
White-crowned Sparrow	*Zonotrichia leucophrys*	undetected	undetected	undetected	undetected	undetected	present
Wild Turkey	*Meleagris gallopavo*	present	undetected	undetected	present	undetected	present

[A] *Michigan Priority Landbird (Knutson et al. 2001, Matteson et al. 2009)*
[B] *Michigan Species of Special Concern*
[C] *Michigan Endangered Species*
* *Barn Swallow and Tree Swallow were not included in the 2005 survey.*

APPENDIX 3.3: OCCUPANCY MODELING RESULTS FOR AVIAN SPECIES OTHER THAN GRASSLAND SPECIALISTS

RESULTS

Avian detection and occupancy for non-grassland species were modeled in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) fields. Adequate detection probability estimates, *p*, were achieved for 7 species (i.e., $p \geq 0.10$), with detection probability estimates ranging from 0.14 to 0.90 (Table 3.8). For Clay-colored Sparrow, an adequate detection probability estimate was obtained only when a subset of the data was used (CP23 2006 data). For Tree Swallow, only data from 2006 could be used to obtain a detection probability estimate because Tree Swallow data were not collected in 2005. Avian detection was also modeled in CP23 fields and the introduced portion of CP1 fields to obtain detection probability estimates (Table 3.8) and enable the reporting of density information for species in native and introduced vegetation (Appendix 3.2).

Occupancy modeling was performed for all 7 non-grassland avian species with adequate detection probability estimates (Table 3.8). The naïve estimate of occupancy was 1.0 for the detection-only models (i.e., $\psi[\cdot]\ p[\text{covariates}]$) of American Goldfinch, Red-winged Blackbird, and Song Sparrow, indicating that these species occupied all fields in this study. For Common Yellowthroat, the global occupancy model was extremely overdispersed (i.e., $\hat{c} > 4$; Table 3.2), so no further occupancy models were derived. Global model fit indicated that informative occupancy models could be derived for the 3 remaining bird species: Clay-colored Sparrow (CP23 2006 data only), Eastern Kingbird and Tree Swallow (Table 3.9). The best fitting occupancy models for all 3 species did not include occupancy covariates, indicating that none of the covariates tested in this study were important correlates of the occupancy of Clay-colored Sparrow (CP23 fields only), Eastern Kingbird, or Tree Swallow (2005 only) in CP23 and CP1 CREP fields.

From among three candidate occupancy models, a single best fitting model was identified for Clay-colored Sparrow (CP23 2006 data only) that included no occupancy covariates and accounted for 99.8% of the total AIC_c weight among successful models (Table 3.10). The naïve estimate of occupancy was 0.57, and the detection corrected occupancy estimate was 0.58. Beta estimates and 95% confidence intervals for the effect of the intercept on the best fitting occupancy models of all 3 species for which occupancy modeling was performed are presented in Table 3.11.

A single best fitting occupancy model was identified from two candidate models for the Eastern Kingbird data that accounted for 93% of the total AIC_c weight and did not include occupancy covariates (Table 3.12). The naïve estimate of occupancy was 0.64, and the detection corrected estimates of occupancy was 0.80.

For Tree Swallow in CP23 and CP1 whole CREP fields, a single best fitting occupancy model that accounted for 69% of the total AIC_c weight was identified from among 4 candidate models (Table 3.13). This model did not include occupancy covariates. The naïve estimate of occupancy was 0.36, and the detection corrected occupancy estimate was 0.49.

Table 3.8. Non-grassland avian species, with corresponding best-fitting detection models (with constant occupancy, $\psi[\cdot]$) and detection probability estimates, observed in 1) CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands and in 2) CP23 fields and the introduced portion of CP1 fields, Saginaw Bay watershed, Michigan, 2005-2006. Ordinal date was standardized to reduce biases in parameter estimation. The lowest detection probability estimate was reported when multiple detection probability estimates were generated.

Common Name	Detection Model	$p(\cdot)$ [SE]
CP23 and CP1 Whole CREP Fields		
American Goldfinch	*p(CREP Field Type)*	0.56 [0.06]
Clay-colored Sparrow, native/CP23 2006 data	*p(Ordinal Date)*	0.21 [0.14]
Common Yellowthroat	*p(Ordinal Date)*	0.23 [0.06]
Eastern Kingbird	*p(.)*	0.14 [0.04]
Red-winged Blackbird	*p(CREP Field Type, Ordinal Date)*	0.56 [0.09]
Song Sparrow	*p(CREP Field Type)*	0.90 [0.03]
Tree Swallow, 2005 data	*p(.)*	0.19 [0.09]
CP23 (Native) and Introduced CP1 Plant Communities		
American Goldfinch	*p(Grassland Type)*	0.42 [0.06]
Clay-colored Sparrow, native/CP23 2006 data	*p(Ordinal Date)*	0.21 [0.14]
Common Yellowthroat	*p(Ordinal Date)*	0.20 [0.05]
Eastern Kingbird, 2006 data	*p(.)*	0.17 [0.07]
Red-winged Blackbird	*p(Grassland Type, Ordinal Date)*	0.56 [0.09]
Song Sparrow	*p(Grassland Type)*	0.84 [0.04]

Table 3.9. Non-grassland avian species and global occupancy models with corresponding number of parameters, *K*, and overdispersion parameters, $\hat{c}$, observed in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Common Name	Global Model	*K*	$\hat{c}$
Clay-colored Sparrow			
Native 2006 data	*Ψ(Field Size), p(Ordinal Date)*	4	1.10
Common Yellowthroat	*Ψ(Field Size), p(Ordinal Date)*	4	4.56
Eastern Kingbird	*Ψ(Field Size), p(.)*	3	0.41
Tree Swallow	*Ψ(CREP Field Type, Field Size), p(.)*	4	1.06

Table 3.10. Summary of AIC occupancy model selection adjusted by the best detection model for Clay-colored Sparrow in native/CP23 vegetation of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(Ordinal Date)	39.45	3	53.45	0.00	0.998
Ψ(Native Patch Size), p(Ordinal Date)	38.96	4	66.96	13.50	0.001
Ψ(Ordinal Date), p(Ordinal Date)	39.44	4	67.44	13.99	0.001

Table 3.11. Best fitting single occupancy model for Clay-colored Sparrow (Native/CP23 2006 data only), Eastern Kingbird, and Tree Swallow in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006

				95% Confidence Interval	
Species	Parameter	β Estimate	SE	Upper	Lower
Clay-colored Sparrow Native 2006 data	Intercept	0.34	0.79	-1.21	1.89
Eastern Kingbird	Intercept	1.37	1.17	-0.92	3.66
Tree Swallow	Intercept	-0.03	0.97	-1.74	1.68

Table 3.12. Summary of AIC occupancy model selection adjusted by the best detection model for Eastern Kingbird in CP23 and CP1 CREP grasslands, Saginaw Bay watershed, Michigan, 2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(.)	106.03	2	113.03	0.00	0.93
Ψ(CREP Field Size), p(.)	104.28	3	118.28	5.25	0.07

Table 3.13. Summary of AIC occupancy model selection adjusted by the best detection model for Tree Swallow in CP23 and CP1 Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(.)	50.77	2	55.86	0.00	0.69
Ψ(CREP Field Type), p(.)	50.46	3	58.86	3.00	0.15
Ψ(Field Size), p(.)	50.71	3	59.11	3.25	0.14
Ψ(CREP Field Type, Field Size), p(.)	50.45	4	62.89	7.03	0.02

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CHAPTER 4: GRASSLAND BIRD DETECTION AND OCCUPANCY IN NATIVE AND INTRODUCED PLANT COMMUNITIES OF MICHIGAN CONSERVATION RESERVE ENHANCEMENT PROGRAM (CREP) GRASSLANDS

INTRODUCTION

Grassland bird populations have declined over much of their breeding habitat, and Breeding Bird Survey (BBS) data indicate that grassland birds have declined more rapidly, more consistently, and over a wider geographic area than any other guild of North American birds (Sauer et al. 1995, Knopf 1996). Habitat loss and fragmentation due to agricultural conversion has been the most important cause of declines (Briggs et al. 2005, Sauer et al. 2008, Fisher and Davis 2010), with losses of native temperate grasslands >83% in the Midwest. (Noss et al. 1995). Farmland set-aside programs such as the Conservation Reserve Enhancement Program (CREP) increase the area of native and non-native grasslands in the U.S., and may diminish or reverse the decline of grassland bird species (Herkert 2009).

Combinations of within-field and landscape-scale variables have been identified as important covariates in other studies of grassland birds (e.g., Ribic and Sample 2001, Bakker et al. 2002, Cunningham and Johnson 2006, Winter et al. 2006), and grassland birds likely use a hierarchical process that is affected by variables at multiple scales when choosing breeding territories (Ribic et al. 2009, Fisher and Davis 2010). For example, Winter et al. (2006) found that grassland bird density was affected by a combination of habitat patch size, landscape, region, and local vegetation structure more than by local vegetation structure alone. Other research has also demonstrated that the extent and proximity of woody vegetation influence the distribution and abundance of many grassland bird species, and larger bird abundances are often associated with less woody vegetation in the landscape (e.g., Ribic and Sample 2001, Cunningham and Johnson 2006). For example, Cunningham and Johnson (2006) found that tree cover negatively affected the occurrence of 13 out of 15 observed grassland bird species at proximate scales (i.e., ≤100 m from birding transects).

Due to geographic variation both in grassland bird habitat requirements (e.g., Johnson and Igl 2001, Winter et al. 2006) and in farmland set-aside program implementation (Heard et al. 2000, Riffell et al. 2008), it is important to evaluate the impacts of set aside programs on grassland birds at local scales (i.e., state, ecoregional, or major watershed levels). This study is the first to examine grassland bird occupancy of Michigan CREP grasslands. The goal of this analysis was to evaluate the correlation of grassland bird occupancy with 1) structural variables and other characteristics of CREP grasslands and/or 2) the extent and proximity of nearby (≤100 m) woody vegetation. Grassland bird occupancy was modeled as a function of these covariates for all observed grassland bird species that were adequately detected within two different types of CREP grasslands, native vegetation of CREP conservation practice (CP) 23 plantings and introduced vegetation of CP1 fields.

METHODS

This study occurred within the 8,219 km^2 Sandusky Lake Plain sub-subsection of Michigan's southern Lower Peninsula (Albert 1995). This sub-subsection consists of flat clay lake plain, bordered by shoreline dunes and sand plain. In the center of the flat clay lake plain, long and narrow till plains and ridges of end moraines parallel either Saginaw Bay or Lake Huron (Albert 1995). Prior to European settlement, extensive wet and wet-mesic prairies occurred upland from the coastal Great Lakes marshes of Saginaw Bay, but largely due to agricultural conversion, these prairies now occur only as small remnants, generally on State-owned lands (Albert 1995).

In Michigan, 22,983 ha of land in the Saginaw Bay, Lake Macatawa, and River Raisin watersheds were enrolled in CREP at the time of this study (USDA 2007). From CREP lands in the Saginaw Bay watershed, 14 CREP grassland fields in Tuscola County, ranging in size from 6.8 to 23.6 ha, were selected for study. All grassland study fields were planted in 2002 and were surveyed for birds in 2005 and 2006. Seven study fields were CREP conservation practice (CP) type 23 (hereafter CP23) that had been planted with a seed mixture of native grasses and native and introduced wildflowers over 100% of the field area, and 7 study fields were CP type 1 (hereafter CP1) fields that had been planted with introduced grasses and legumes over 70% of the field area, with the remaining 30% of the field area planted with the same native seed mixture used in CP23 fields (Figure 4.1). In 2005, the native seed mixture cost $16.00/pound (i.e., $104.00/acre or $256.99/hectare) and the introduced seed mixture cost $1.90/pound (i.e., $20.90/acre or $51.65/hectare) plus 6% sales tax through the Tuscola County Conservation District. Within CREP, CP23 fields are wetland restorations, and, as administered in the Saginaw Bay area, all CP23 fields had hydric soils. A small wetland was created in each CP23 field by cutting drain tiles, and, if necessary, by creating a push-out, or shallow depression in the soil. For this analysis, I evaluated information from the upland portion of CP23 fields (hereafter native grasslands) and the introduced portion of CP1 fields (hereafter introduced grasslands). Native grasslands ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha), and introduced grasslands ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha).

Avian data were collected using line-transect sampling (Edwards et al. 1981). The total upland area of each study grassland was surveyed every 2 weeks from 1 June – 14 August in 2005 (n=5 in all grasslands) and from 15 May – 14 August in 2006 (n=6 per grassland). Surveys took place along transects running the length of each grassland. The first transect was established 25 m from a randomly selected grassland corner and subsequent transects were established at 50 m intervals across the width of the grassland. An investigator recorded the species and location of each individual bird observed. If a bird was flushed by the observer, the new location of the bird was noted in order to prevent counting the same individual bird more than once. If a bird was observed but could not be identified, the location of the bird was recorded and the species was designated as unknown. Unidentified individuals were not included in analyses of this study. Avian surveys were conducted after sunrise and before 12:00 p.m., and surveys were not performed in the rain or if wind speeds were over 16 kph.

Avian species detection, p, and occupancy, ψ, probabilities in native and introduced CREP grasslands were estimated using single-season occupancy models (MacKenzie et al.2002) and the program PRESENCE (Hines 2006). The use of single season models was based on likelihood ratio testing (MacKenzie et al. 2006, Nichols et al. 2008), which indicated that the more highly parameterized multi-season models did not fit the data significantly better than single season models ($p > 0.05$).

Occupancy estimation that is adjusted for imperfect detection is recognized as an improvement to traditional approaches that assume species are perfectly detected (e.g., logistic regression) (Rota et al. 2009). The estimation of a species' detection probability requires multiple surveys at each study site and assumes that the occupancy of each site is closed to changes between surveys (MacKenzie et al. 2002, MacKenzie et al. 2006). Concerns about the application of occupancy modeling in ecological systems that violate this closure assumption have arisen, and violations of closure have been found to typically result in overestimation of occurrence probability (Rota et al. 2009).

To more closely conform to the closed population assumption, the occupancy data set for each grassland bird species was truncated based on the estimated date of initial flight by the first young of the year, or the date at which the population births would likely be observed during avian surveys. For each species, the date at which initial flight was likely to occur was estimated from life history information from species accounts in the Cornell Lab of Ornithology's Birds of North America Online (BNA Online) (Lanyon 1995, Martin et al. 1995, Vickery 1996, Giudice and Ratti 2001, Herkert et al. 2001, Jones and Cornely 2002, Temple 2002, Wheelwright and Rising 2008). Life history information included arrival time to the breeding grounds, time until nest initiation, time until incubation, duration of incubation, time until fledging, and, when present, duration of the post-fledging, pre-flight interval. Estimated dates at which initial flight by the first young of the year occurred for each grassland species were: Bobolink, June 17; Dickcissel, July 27; Eastern Meadowlark, June 21; Grasshopper Sparrow, June 17; Ring-necked Pheasant, June 27; Savannah Sparrow, July 1; Sedge Wren, June 18; and Vesper Sparrow, July 1. For each species, occupancy information collected after the estimated date of initial flight by the first young of the year was not included in detection probability and occupancy modeling.

For studies with small sample sizes, model ranking using Akaike's information criterion corrected for small sample size (AIC_c) is recommended (Burnham and Anderson 2002). The top-ranking detection model for each avian species was identified as having the lowest AIC_c value among all candidate models with constant occupancy, $\psi(\cdot)$ (MacKenzie et al. 2002). Candidate (i.e., successful) detection models were those that achieved 1) numerical convergence of parameter estimates to >6 significant figures, 2) a balanced variance-covariance matrix, and 3) small- or moderately-sized standard errors of betas, the untransformed estimates of coefficients for covariates (i.e., standard error was no greater than 5 times the corresponding beta) (D. MacKenzie, Program PRESENCE – FAQ at http://www. phidot.org/forum, accessed October 2009).

Avian detection probability was calculated for each species observed in at least 10% of the total number of surveys (MacKenzie et al. 2002). Detection probability estimates were calculated from the top-ranking detection model for each species. Potential detection covariates were ordinal date and grassland type (i.e., native or introduced plant community). Ordinal date was standardized to reduce biases in parameter estimation (MacKenzie et al. 2002). When grassland type and/or ordinal date were covariates in the top-ranking detection model, multiple estimates of detection resulted, and the lowest estimate was reported. For species where the naïve estimate of ψ equaled 1, study grasslands were fully occupied, and detection modeling was conducted with ψ fixed to 1.

Occupancy modeling was performed only for species with detection probability estimates ≥0.10. Potential occupancy covariates included grassland type (i.e., native or introduced), grassland size, and early summer (i.e., 15 May – 15 June) vegetation variables such as percent bare ground, canopy height, percent total canopy cover, percent standing dead vegetation, percent grass in the live canopy, percent litter cover, and litter depth (Table 4.1). Potential occupancy covariates also included two landscape-scale variables: 1) the total area of woody vegetation within 100 m of study grasslands and 2) the average proximity of each grassland bird species to woody vegetation (Table 4.1). Continuous variables were standardized to reduce biases in parameter estimation (MacKenzie et al. 2002).

Vegetation structure was investigated using one randomly located sampling point per acre in each CP23 and CP1 field (i.e., 1 sampling point per 0.4 ha; n = 17 to 59 sampling points per field). Vegetation data from CP23 fields and the introduced portion of CP1 fields were included in occupancy modeling (i.e., data from the native portion of CP1 fields were omitted). Vegetation data were collected once in 2005 during mid-summer (15 June – 14 July) and three times in 2006 during early summer (15 May – 14 June), mid-summer (15 June – 14 July), and late summer (15 July – 14 August). Canopy height and litter depth were measured at each sampling point using a yardstick. A modified 50 cm x 25 cm Daubenmire frame was centered on each sampling point (Daubenmire 1959). All plant species within the Daubenmire frame were identified. The percentage of Daubenmire frame area composed of bare ground, litter cover, and canopy cover were visually estimated. Percent standing dead and living vegetation were also visually estimated, and within the living canopy vegetation, the percent cover of grasses, forbs, and woody vegetation were also determined. The total area of the Daubenmire frame was accounted for as:

100% plot area = % bare ground + % litter cover + % total canopy cover

where:

% total canopy cover = % dead canopy vegetation + % live canopy vegetation

and:

% live canopy vegetation = % grasses + % forbs + % woody plants.

The percent of woody vegetation within 100 m of each study site was determined by digitizing land cover images from the National Agricultural Imagery Program (NAIP) (i.e., 1 m x 1m resolution digital orthophoto quadrangles collected June-August 2005, http://www.mcgi.state.mi.us/mgdl/DOQQ2005.asp?cnty=Tuscola/) and using ArcGIS (version 8.2; ESRI 2002) to compute the area of woody vegetation within 100 m buffers around each grassland. Then the percentage of woody vegetation within each 100m buffer was calculated as the area of woody vegetation within the buffer divided by the total area of the buffer. Additionally, the average proximity of each bird species to woody vegetation was determined for each study using ArcGIS (version 8.2; ESRI 2002), the digitized woody vegetation data layer, and the location of each individual bird observed (i.e., the latitude and longitude of each bird). The distance from each observed bird to the nearest woody vegetation was calculated and these distances were averaged to obtain the mean proximity of each species to woody vegetation in each study field.

Selection of the set of potential occupancy covariates for each species was based on variables found to influence abundance or occupancy of the species in other research, including research summarized in the species accounts of BNA Online. Occupancy covariate selection for each grassland bird species was also informed by results from CREP administrative-level occupancy modeling (i.e., CP23 and CP1 whole fields; Chapter 3). Occupancy covariates for Bobolink and Eastern Meadowlark were grassland type, grassland size, percent litter cover, percent total canopy cover, percent grass in the live canopy, area of woody vegetation within 100m, and proximity to woody vegetation (Lanyon 1995, Martin et al. 1995, Ribic and Sample 2001; Table 4.2). For Dickcissel, occupancy covariates were grassland type, grassland size, percent total canopy cover, litter depth, area of woody vegetation within 100m, and proximity to woody vegetation (Temple 2002; Table 4.2). Occupancy covariates for Grasshopper Sparrow and Savannah Sparrow were grassland type, grassland size, percent bare ground cover, percent litter cover, percent total canopy cover, litter depth, area of woody vegetation within 100m, and proximity to woody vegetation (Vickery 1996, Ribic and Sample 2001, Bakker et al. 2002, Wheelwright and Rising 2008; Table 4.2). For Ring-necked Pheasant, occupancy covariates were grassland type, grassland size, and percent standing dead vegetation in the canopy (Giudice and Ratti 2001; Table 4.2). Occupancy covariates for Sedge Wren and Vesper Sparrow were grassland type, grassland size, percent bare ground, percent total canopy cover, canopy height, area of woody vegetation within 100m, and proximity to woody vegetation (Herkert et al. 2001, Jones and Cornely 2002; Table 4.2)

For each avian species with a minimum detection probability estimate ≥ 0.10, all possible occupancy models were constructed using the best-fitting detection model as a function of the occupancy covariates (i.e., ψ[covariates], *p*[as in best fitting detection model]) (Roloff et al. *in press*). Candidate occupancy models met the convergence, variance-covariance matrix, and standard error size criteria as described above. The global model was identified as the most parameterized occupancy model. Global model fit (MacKenzie and Bailey 2004, MacKenzie et al. 2006) and top-ranking occupancy model fit were evaluated using the chi-square goodness-of-fit statistic to estimate an overdispersion parameter, $\hat{c}$, from 2,000 bootstrap iterations. Ecological models are commonly overdispersed (i.e., $\hat{c} \neq 1$), and adjustment of the model selection criteria to account for it is advised (MacKenzie and Bailey 2004, MacKenzie et al. 2006). For species with suitable global models (i.e., $\hat{c} \leq 4$), all occupancy models in the set of candidate models were ranked according to AIC_c (Burnham and Anderson 2002). For species with overdispersed global models, the fit of the top-ranking occupancy model was evaluated using ĉ, and if the model was not overdispersed, results from the top-ranking model were reported. When one best occupancy model did not emerge, competing models (i.e., models where $\Delta AIC_c \leq 2.00$) were model averaged to derive a composite occupancy model (Burnham and Anderson 2002).

RESULTS

Thirty-three avian species were observed in CP23 fields and the introduced portion of CP1 fields during 2005 and 2006 (Appendix 4.2). In 2005, 23% of individual birds observed during surveys were unidentified, and in 2006 surveyors were more experienced and only 8% of observed individuals were unidentified. Using the grassland bird list developed for the Breeding Bird Survey by the U.S. Geographic Survey (Sauer et al. 1995), 11 species were identified as grassland specialists: Bobolink (*Dolichonyx oryzivorus*), Dickcissel (*Spiza americana*), Eastern Meadowlark (*Sturnella magna*), Grasshopper Sparrow (*Ammodramus savannarum*), Henslow's Sparrow (*Ammodramus henslowii*), Horned Lark (*Eremophila alpestris*), Northern Harrier (*Circus cyaneus*), Ring-necked Pheasant, Savannah Sparrow (*Passerculus sandwichensis*), Sedge Wren (*Cistothorus platensis*), and Vesper Sparrow (*Pooecetes gramineus*). Six of these grassland specialist species are of special conservation priority in Michigan (Appendix 4.2). Bobolink, Dickcissel, Grasshopper Sparrow, Henslow's Sparrow, Northern Harrier, and Sedge Wren are designated as Michigan priority landbirds by the bird conservation group Partners in Flight (Knutson et al. 2001, Matteson et al. 2009), while Dickcissel, Grasshopper Sparrow, and Northern Harrier are also state species of special concern and Henslow's Sparrow is a state endangered species (Appendix 4.2).

Bobolink and Savannah Sparrow were the most abundant grassland bird species in native and introduced grasslands (Appendix 4.2). Three grassland bird species were too rarely observed to estimate their detection probabilities (i.e., observed in less than 10% of surveys): Henslow's Sparrow, Horned Lark, and Northern Harrier. Detection probability modeling was performed for the 8 other grassland specialist species observed during this study: Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow. Likelihood ratio testing indicated that more highly parameterized multi-season models did not fit the data significantly better than simpler single-season models ($p > 0.05$), so single season models were used in all detection and occupancy modeling efforts (MacKenzie et al. 2006, Nichols et al. 2008). In the case of one species, Savannah Sparrow, the naïve estimate of occupancy equaled 1, so detection modeling was conducted with ψ fixed equal to 1.

Adequate detection probability estimates (i.e., $p \geq 0.10$) were obtained for all 8 grassland species, with detection probability estimates ranging from 0.12 to 0.84 (Table 4.3). Ordinal date was included in the best detection model for Dickcissel (Table 4.3). The detection probability of the species varied between 2005 and 2006, and detection probability estimates were > 0.12 during both years, indicating that Dickcissel was adequately detected throughout both breeding seasons (Figure 4.2) The best detection model for Sedge Wren included grassland type, and the species was adequately detected in both cover types but tended to be more likely to be detected in native grasslands (p = 0.83 [SE = 0.16]) than in introduced grasslands (p = 0.14 [SE = 0.16]; Figure 4.3). For the remaining 6 grassland bird species the best detection model included only the intercept (Table 4.3).

The naïve estimate of occupancy was 1.0 for Savannah Sparrow, indicating that all surveyed fields were occupied. Global occupancy model fit was evaluated for the 7 remaining grassland bird species. For Bobolink, global model fit was highly overdispersed (i.e., $\hat{c} > 4$) (Table 4.4), as was the top-ranking occupancy model ($\hat{c}$ = 8.91), indicating that Bobolink occupancy could not be reliably estimated from this study. Global model fit indicated that derived occupancy models for the 6 remaining grassland bird species (i.e., Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Sedge Wren (2006 data), and Vesper Sparrow) were not overdispersed and hence would have informative structure (Table 4.4).

Three competing occupancy models (i.e., $\Delta AIC_c \leq 2$) were identified for Dickcissel that together accounted for 64% of the total AIC_c weight among 27 candidate models (Table 4.5). The top-ranking and second-ranking models were not overdispersed (top-ranked AIC_c = 87.27, $\hat{c}$ = 0.87; second-ranked AIC_c = 87.81, $\hat{c}$ = 0.86) and included grassland size (Table 4.5). The second-ranking model also included area of woody vegetation within 100 m of CREP grasslands (Table 4.5). The third-ranking model was the intercept-only model ($\hat{c}$ = 0.87) (Table 4.5). The naïve occupancy estimate was 0.43, while the detection-corrected occupancy estimates by site ranged from 0.00 to 1.0. The model averaged beta estimate of the effect of grassland size on Dickcissel occupancy was negative (-1.94, SE = 1.43; Table 4.6), and the model averaged beta estimate of the effect of the area of woody vegetation within 100 m of CREP grasslands was positive (beta = 0.18, SE = 0.19; Table 4.6), indicating that Dickcissel occupancy was higher in smaller grasslands and higher in grasslands with more woody vegetation within 100 m. However, the 95% confidence interval of both beta estimates included zero, indicating that grassland type and area of woody vegetation within 100 m were not significant determinants of Dickcissel occupancy (Table 4.6).

For Eastern Meadowlark, the intercept-only model ranked highest (w = 0.37; AIC_c = 29.43; $\hat{c}$ = 1.52; Table 4.7), indicating that none of the covariates tested in this study were important correlates of Eastern Meadowlark occupancy. The naïve estimate of occupancy was 0.14, and the detection corrected occupancy estimate was 0.17.

Two competing (i.e., $\Delta AIC_c \leq 2$) occupancy models were identified for Grasshopper Sparrow that together accounted for 51% of the total AIC_c weight among 19 candidate models (Table 4.8). The top-ranking model (AIC_c = 41.49; $\hat{c}$ = 0.23) included area of woody vegetation within 100 m of CREP grasslands, and the second-ranking model (AIC_c = 43.11; $\hat{c}$ = 0.36) was the intercept-only model (Table 4.8). The naïve estimate of occupancy was 0.36, while detection-corrected occupancy estimates by site ranged from 0.001 to 1.0. The model averaged beta estimate for the effect of the area of woody vegetation within 100 m of CREP fields was positive (beta = 0.80, SE = 1.07; Table 4.6) indicating that Grasshopper Sparrow occupancy was higher in grasslands with more woody vegetation within 100 m. However, the 95% confidence interval of the beta estimate included zero, indicating that area of woody vegetation within 100 m was not a significant determinant of Grasshopper Sparrow occupancy (Table 4.6).

For Ring-necked Pheasant, two competing occupancy models (i.e., $\Delta AIC_c \leq 2$) were identified that together accounted for 74% of the total AIC_c weight among 5 candidate models (Table 4.9). The intercept-only model (AIC_c = 67.29; $\hat{c}$ = 0.25) was the top-ranking model, and the second-ranking model (AIC_c = 67.85; $\hat{c}$ = 0.25) included grassland type (Table 4.9). The naïve estimate of occupancy was 0.50, and the detection-corrected occupancy estimate ranged from 0.35 in introduced vegetation to 0.89 in native vegetation. The model-averaged beta estimate for the effect of grassland type on Ring-necked Pheasant was negative (beta = -2.74, SE = 2.74; Table 4.6), suggesting that native grasslands had greater occupancy rates than introduced grasslands (i.e., native grasslands were assigned a value of 0 and introduced vegetation was categorized as 1 during modeling). The 95% confidence interval of the beta estimate included zero, indicating that grassland type was not a significant determinant of Ring-necked Pheasant occupancy (Table 4.6).

Two competing occupancy models (i.e., $\Delta AIC_c \leq 2$) were identified for Sedge Wren that together accounted for 64% of the total AIC_c weight among 16 candidate models (Table 4.10). The top-ranking model (AIC_c = 30.75; $\hat{c}$ = 0.16) did not include occupancy covariates and the second-ranking model (AIC_c = 32.13; $\hat{c}$ = 0.11) included grassland size (Table 4.10). The naïve occupancy estimate was 0.21, and the detection-corrected occupancy estimate by site ranged from 0.06 to 0.77. The model averaged beta estimate for the effect of grassland size on Sedge Wren occupancy was positive (beta = 0.34, SE = 0.35; Table 4.6), indicating that Sedge Wren occupancy was greater in larger grasslands. However, zero was included in the 95% confidence interval of the beta estimate, indicating that grassland type was not a significant determinant of Sedge Wren occupancy (Table 4.6).

For Vesper Sparrow, the intercept-only model was the top-ranking model among three candidate models (w = 0.63; AIC_c = 63.06; $\hat{c}$ = 0.53; Table 4.11), indicating that no covariates included in this study were important correlates of Vesper Sparrow occupancy. The naïve estimate of occupancy was 0.50, and the detection-corrected estimate of occupancy was 0.78.

DISCUSSION

Avian surveys of CP23 and CP1-introduced CREP grasslands in Michigan's Saginaw Bay watershed resulted in the observation of 34 species, 11 of which were grassland associates (Sauer et al. 1995). Eight grassland bird species were detected frequently enough to allow detection probability and occupancy modeling. Detection estimates were adequate (i.e., $\geq$ 0.10) for all 8 grassland bird species, and were above 0.25 for 5 species, indicating that the survey technique used in this study was an appropriate methodology for documenting occupancy. Covariates included in the top-ranking occupancy models of grassland bird species included grassland type, grassland size, and area of woody vegetation within 100 m of study areas. Although the effects of these covariates were not significant, findings of this study are likely constrained by small sample sizes, and these covariates warrant evaluation in further research.

Grassland bird species that occurred in this study were similar to those documented in comparable research. Published studies on grassland bird species in CREP fields of the Lake States are rare; only one study in Pennsylvania was identified (Wentworth et al. 2010). Wentworth et al. (2010) found 8 grassland bird species that were also observed in my study: Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Henslow's Sparrow, Ring-necked Pheasant, Savannah Sparrow, and Vesper Sparrow (i.e., Horned Lark, Northern Harrier, and Sedge Wren were not observed in Pennsylvania but were present in Michigan CREP grasslands). The grassland bird species that occurred in my study were also similar to those that occurred in comparable work in grasslands of a similar farmland set-aside program, the Conservation Reserve Program (CRP). In an investigation of the effects of different age classes of Michigan CRP fields on avian diversity, density, and productivity, Millenbah (1993) found 10 grassland bird species, all of which were also observed in my study along with Henslow's Sparrow. Nine grassland bird species observed in my surveys also occurred in CRP fields in a large-scale analysis of Midwestern bird abundance and nesting in CRP fields and cropland (i.e., Horned Lark and Northern Harrier were absent in CRP fields) (Best et al. 1997). With the exception of Savannah Sparrow, the grassland species that were observed in my surveys were also observed in a study of CP1 and switchgrass (*Panicum virgatum)* dominated CP2 CRP fields in Missouri (McCoy et al. 2001). These comparisons indicate that grassland bird community composition is similar in CREP and CRP, however other research suggests that grassland bird abundances vary within and between CREP and CRP (Chapter 2, McCoy et al. 2001, Wentworth 2010), underscoring the importance of evaluating CREP and other such conservation programs at local scales (e.g., within states, ecoregions, or major watersheds). It is important to note that the consistency of results from studies in different regions may be affected by differences in grassland composition, age or successional stage, and structure.

The survey methodology of this study generated adequate detection probability estimates for all grassland species observed in ≥10% of surveys. The technique of truncating the survey data for each species based upon expected initial flight of the first young of the year presumably conformed more closely with the closed population assumption of detection probability and occupancy modeling, and detection probability estimates for most species were higher for truncated data than for the full set of data (i.e., for Bobolink, Eastern Meadowlark, Grasshopper Sparrow, Savannah Sparrow, and Sedge Wren) (Table 4.3; Chapter 3). In addition to timing surveys to avoid including population births, other methodological improvements for future avian studies in similar systems include increasing the number of surveys to improve the probability of detecting rare species, and increasing the number of fields surveyed to ease or remove analytical constraints due to small sample sizes.

Despite the constraints of small sample size (i.e., n= 14), the top-ranking or competing (i.e., $\Delta AIC_C \leq 2$) occupancy models of 4 of the 6 grassland bird species with successful occupancy modeling included covariates. Occupancy modeling results indicate that grassland type (i.e., native or introduced plant communities) and grassland size may be associated with the occupancy of some grassland bird species in Michigan CREP grasslands. Grassland type was associated with the occupancy of Ring-necked Pheasant, with larger occupancy estimates in native grasslands. Grassland size was associated with both Dickcissel and Sedge Wren occupancy. Higher Sedge Wren occupancy was associated with larger grasslands, but unexpectedly, higher Dickcissel occupancy was associated with smaller grasslands, which contrasts with findings of a study documenting a positive relationship between habitat patch size and Dickcissel occurrence (Bakker et al. 2002), as well as other research that has shown habitat patch size to be an important positive influence on the occupancy of Dickcissel and many other species of grassland birds (summarized in Ribic et al. 2009). Although grassland type and grassland size were not significant correlates of the occupancy of any species in my study (i.e., the 95% confidence intervals of the beta estimates for the effect of field/grassland type and grassland size on occupancy included 0), non-zero tendencies existed (i.e., 0 occurred near the tails of the 95% confidence intervals of the covariates), suggesting that these covariates should be included in future work that evaluates grassland bird occupancy in this or similar ecosystems.

The only other potential occupancy covariate that occurred within top-ranking or competing occupancy models was area of woody vegetation within 100 m of study grasslands, which was positively associated with the occupancy of Grasshopper Sparrow and Dickcissel. However, for both species, zero was present centrally within the 95% confidence interval of the beta estimate (i.e., within the central 50% of the confidence interval), indicating that the area of woody vegetation within 100 m of grasslands was not an important grassland bird occupancy correlate in this study.

Other studies have shown that area of woody vegetation around grasslands is an influence on the occupancy or abundance of grassland birds, often with larger bird occupancy or abundance estimates associated with less woody vegetation in the landscape (e.g., Ribic and Sample 2001, Cunningham and Johnson 2006). For example, in research with similar woody cover digitization methods to those of my study (i.e., digitized from 1 m x 1m resolution digital orthophoto quadrangles), Cunningham and Johnson (2006) demonstrated that tree cover ≤100 m from birding transects negatively affected the occurrence of Grasshopper Sparrow, Savannah Sparrow, Sedge Wren, and Bobolink. Results from studies with different methods of quantifying woody vegetation (e.g., woody vegetation that is mapped at various scales/resolutions) can indicate broad-scale trends in the influence of woody vegetation on grassland bird species. In a summary of over 80 studies pertaining to the effect of woody vegetation on grassland birds, Bakker (2003) found that investigations that included a metric of the proportion or increase of woody vegetation in the landscape surrounding grasslands and/or a metric of the distance to woody vegetation showed negative associations with many grassland species, including Bobolink, Dickcissel, Eastern Meadowlark, Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, and Sedge Wren. Both Bakker (2003) and Cunningham and Johnson (2006) found a positive association of Vesper Sparrow occurrence and woody vegetation, indicating that there is variation in the response of grassland bird species to woody vegetation in the landscape, and suggesting that species-specific information should be considered in conservation planning for grassland birds. Although area of woody vegetation and proximity to woody vegetation were not important covariates of grassland bird occupancy in this study, the results of other research indicate that these variables should be included in future grassland bird studies.

In a review of 57 studies of grassland bird habitat selection, Fisher and Davis (2010) identified 9 vegetation features that were important predictors of grassland bird habitat use (i.e., features correlated with grassland bird occupancy, nest and territory selection, density, or abundance), which included several vegetation variables that were potential occupancy covariates in this study: coverage of bare ground, grass, dead vegetation, and litter, as well as litter depth, and vegetation height. Surprisingly, none of the top-ranking or competing (i.e., $\Delta AIC_c \leq 2$) occupancy models of grassland birds in my analysis included these variables. In many cases, occupancy models that included such covariates were unsuccessful (e.g., did not converge), and when such models were successful, they did not rank as high as null occupancy models or occupancy models that included other covariates. Other researchers have suggested that in fragmented landscapes, grassland bird habitat requirements may change in response to habitat abundance and composition (Andren 1994, Bakker et al. 2002), and it is possible that in the agriculturally-dominated study system of this project, grasslands were so much more preferred by grassland birds than rowcrops and other available breeding habitats that vegetation structural variables were much less important than grassland type and size in determining grassland bird occupancy.

Management recommendations

Results of this analysis show that CREP grasslands of the Saginaw Bay watershed provide habitat for a majority of the grassland bird species found in Michigan. Of the 15 grassland species documented in Michigan during development of the most recent

Michigan Breeding Bird Atlas (BBAE 2010), 11 species were observed in study areas of this research project, including all 4 species found to have significantly declining population trends ($p < 0.05$) in Michigan in the most recent analysis of grassland bird Breeding Bird Survey (BBS) data (i.e., Vesper Sparrow, Eastern Meadowlark, Bobolink, and Ring-necked Pheasant; Sauer et al. 2008). The 4 grassland bird species found in Michigan during atlas development but not observed in this study were Le Conte's Sparrow, Sharp-tailed Grouse, Short-eared Owl, and Upland Sandpiper.

Although grassland type and grassland size were not significant correlates of the occupancy of any species, non-zero tendencies existed, suggesting that these covariates should be included in future studies and considered in grassland bird conservation strategies. Future studies would benefit from including more study site replicates to increase sample size and analytical power. Because occupancy covariates differed among grassland species, I support the recommendations of other researchers that species-specific habitat preference information should be used when managing grasslands for grassland bird species (e.g., Murray et al. 2008). For the same reason, I also recommend that in similar studies the effect of potential occupancy covariates should be evaluated for individual species within the grassland bird guild rather than for grassland birds as a group.

Threats to the persistence and increase of farmland set-aside grasslands include reconversion to corn or other crop. Recently, increased demand for biofuels has resulted in reconversion of agricultural set-aside lands into crop production (Secchi and Babcock 2007, Searchinger et al. 2008, cited from Fargione et al. 2009), a trend that is likely to continue due to large, federally mandated increases in biofuel production (i.e., >700%

over 2006 production by 2022; Fargione et al. 2009). Fargione et al. (2009) suggest minimizing the negative effects of increased biofuel demand on wildlife by utilizing wildlife-compatible biomass sources that do not require use of additional production lands (e.g., wastes, cover crops, algae), and maximizing the area of perennial grasslands, including diverse native prairie plantings, to produce biofuel in ways that are compatible with wildlife. Additionally, I recommend that program administrators provide incentives for landowners to join or continue in CREP through an increase in the one-time program sign-up bonus when possible, as this strategy that has been found to generate a greater increase in CREP enrollment than did smaller increases in the annual payments made over the term of the CREP contract (Suter et al. 2008). Maintaining and increasing the area of CREP grasslands should be a grassland bird conservation strategy in Michigan.

APPENDIX 4.1: CHAPTER 4 TABLES AND FIGURES

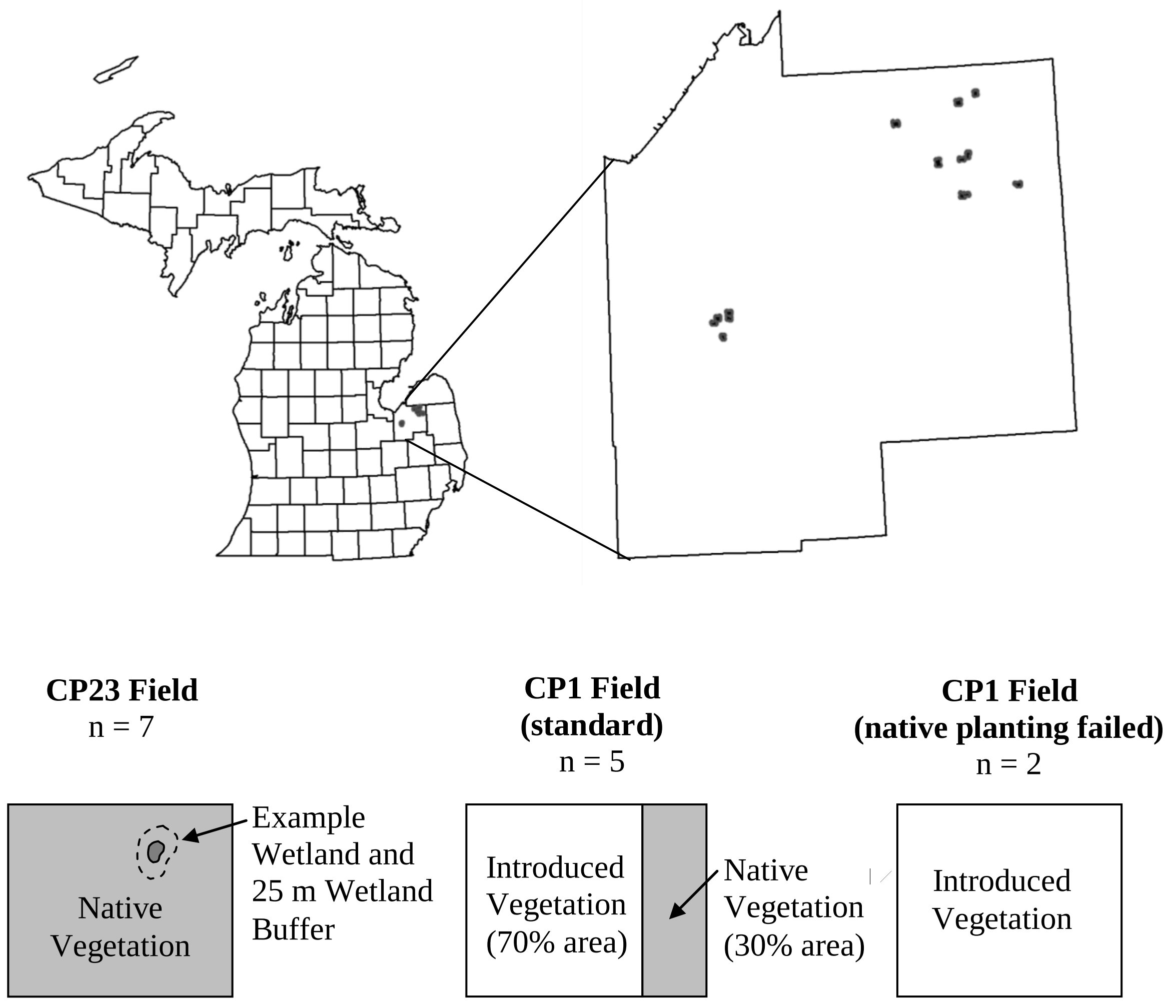

Fig. 4.1. Location, type, and number of Conservation Reserve Enhancement Program (CREP) study fields within Tuscola County, Saginaw Bay watershed, Michigan, 2005 and 2006. Wetlands and wetland buffers were excluded from avian surveys and vegetation sampling. The surveyed (i.e., upland) area of CP23 fields ranged in size from 6.9 – 19.8 ha (median [SE] = 12.6 [1.9] ha, median = 11.8 ha). The introduced portion of CP1 fields ranged in size from 5.5 – 17.0 ha (mean [SE] = 9.4 [1.5], median = 8.0 ha), and CP1 whole fields ranged in size from 7.3 – 23.9 ha (mean [SE] = 12.3 [2.1] ha, median = 10.9 ha).

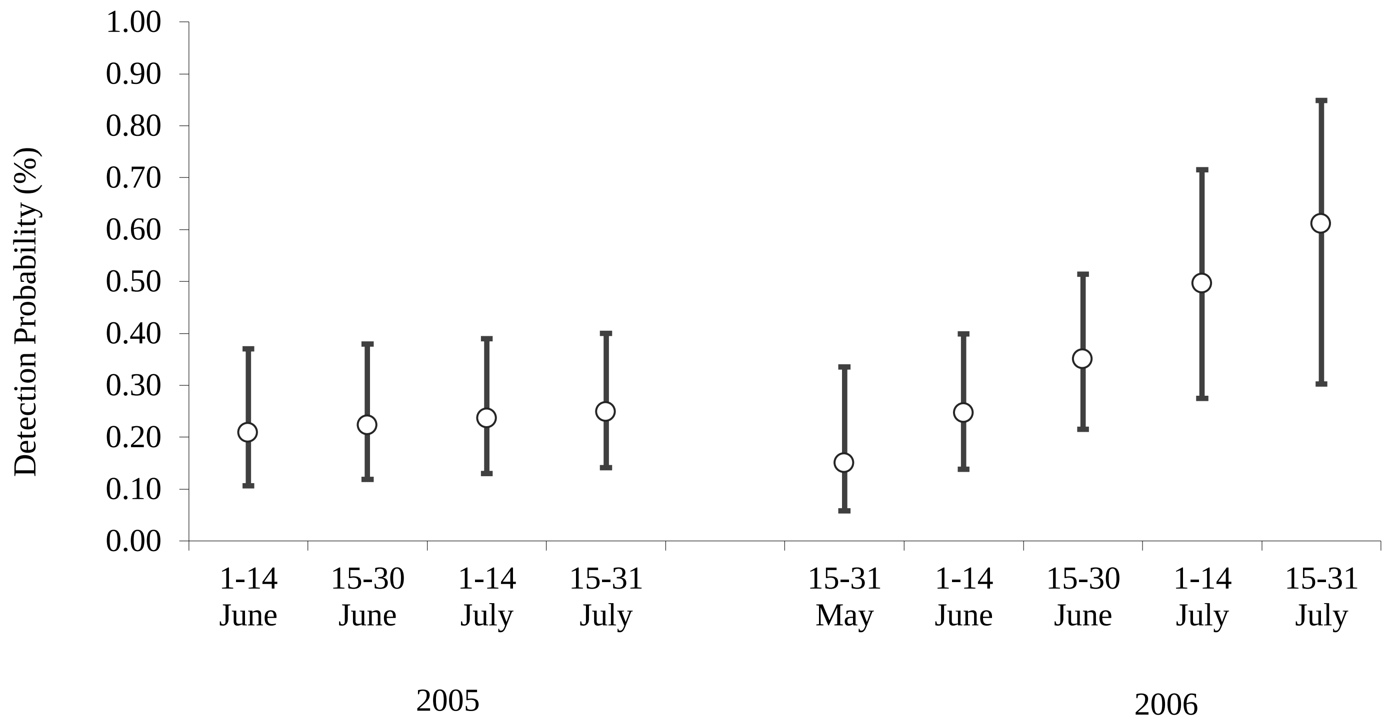

Fig. 4.2. Relationship between Dickcissel detection probability and ordinal date in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Error bars indicate 95% confidence intervals.

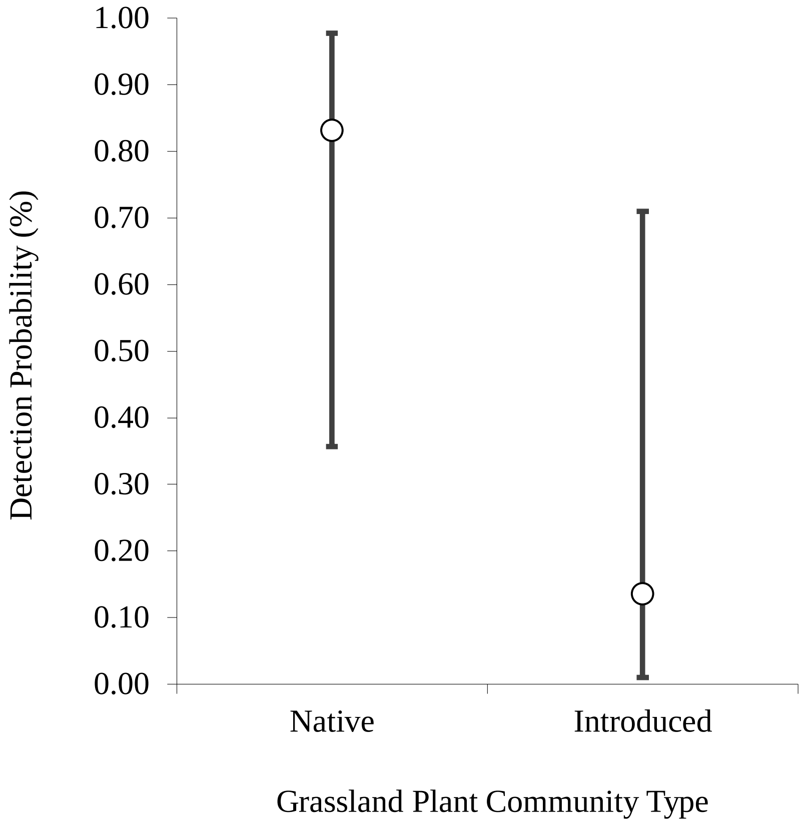

Fig. 4.3. Relationship between Sedge Wren detection probability and grassland type in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005 and 2006. Error bars indicate 95% confidence intervals.

Table 4.1. All potential grassland bird occupancy covariates for CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

			Early Summer Vegetation Variables						
Field	Plant Community Type*	Plant Community Size (ha)	% Bare Ground	% Litter Cover	% Total Canopy	% Dead Canopy	% Live Grass Canopy	Litter Depth (cm)	Canopy Height (dm)
CP23-1	0	17.78	6.98	68.36	24.66	6.89	6.43	10.20	3.20
CP23-2	0	11.78	9.48	26.03	64.48	1.76	12.40	4.14	3.79
CP23-3	0	6.94	7.06	55.88	37.06	2.90	15.95	7.56	4.29
CP23-4	0	13.32	1.12	76.00	22.88	4.93	9.66	9.35	2.74
CP23-5	0	19.81	12.19	29.79	58.02	5.54	11.39	4.32	3.71
CP23-6	0	7.08	5.00	25.56	69.44	1.98	33.28	4.33	3.78
CP23-7	0	11.47	10.46	42.39	47.14	2.16	16.02	4.95	3.46
CP1-1	1	10.09	3.28	30.32	66.40	0.05	41.33	10.92	7.10
CP1-2	1	6.93	4.69	36.56	58.75	2.64	26.43	5.84	3.50
CP1-3	1	7.60	0.00	18.53	81.47	0.79	30.89	4.76	8.41
CP1-4	1	5.53	0.00	14.71	85.29	0.00	42.42	5.00	6.24
CP1-5	1	7.96	10.68	53.26	36.05	2.36	21.51	4.05	2.63
CP1-6	1	10.90	1.22	70.07	28.70	0.76	19.26	7.20	2.85
CP1-7	1	17.08	2.20	32.45	65.34	0.46	30.45	5.75	6.55

**(0=Native, 1=Introduced)*

Table 4.1 Continued.

Field	% Woody Vegetation within 100m	Mean Proximity to Woody Vegetation (m)*						
		Bobolink	Dickcissel	Eastern Meadowlark	Grasshopper Sparrow	Savannah Sparrow	Sedge Wren	Vesper Sparrow
CP23-1	2.34	45.66	.	.	.	82.04	90.16	87.45
CP23-2	4.98	113.57	.	.	118.48	98.69	.	79.77
CP23-3	11.10	.	24.59	.	.	55.87	.	67.49
CP23-4	7.63	.	.	90.50	118.15	134.24	.	.
CP23-5	8.92	303.70	.	.	344.73	202.84	65.37	195.54
CP23-6	9.46	76.67	68.94	89.48	50.80	76.10	.	110.50
CP23-7	10.92	111.64	88.35	.	101.26	108.27	.	.
CP1-1	5.61	181.41	190.79	.	.	153.09	.	80.40
CP1-2	4.78	.	.	.	.	75.23	88.50	.
CP1-3	2.75	118.75	129.82	.	.	86.98	.	.
CP1-4	4.14	158.35	119.20	.	.	137.78	.	.
CP1-5	1.20	132.66	.	.	.	126.68	.	61.84
CP1-6	2.95	148.11	.	.	.	132.11	.	.
CP1-7	10.77	179.53	.	.	.	158.21	.	.

**A period indicates the species was absent in the corresponding field.*

Table 4.2. Potential occupancy covariates for grassland bird species observed in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

	Bobolink	Dickcissel	Eastern Meadowlark	Grasshopper Sparrow	Ring-necked Pheasant	Sedge Wren	Vesper Sparrow
Plant Community Type	x	x	x	x	x	x	x
Plant Community Size (ha)	x	x	x	x	x	x	x
% Bare Ground				x		x	x
% Litter Cover	x		x	x			
% Total Canopy	x	x	x	x		x	x
% Dead Canopy					x		
% Live Grass Canopy	x		x				
Litter Depth (cm)		x		x			
Canopy Height (dm)						x	x
% Woody Vegetation within 100m	x	x	x	x		x	x
Proximity to Woody Vegetation (m)	x	x	x	x		x	x

Table 4.3. Grassland bird species, with corresponding best-fitting detection models (with constant occupancy, $\psi[\cdot]$) and detection probability estimates, observed in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. The lowest detection probability estimate was reported when multiple detection probability estimates were generated.

Common Name	Detection Model	$p(\cdot)$ [SE]
Bobolink	*p(.)*	0.75 [0.08]
Dickcissel	*p(Ordinal Date)*	0.12 [0.07]
Eastern Meadowlark	*p(.)*	0.34 [0.18]
Grasshopper Sparrow	*p(.)*	0.47 [0.16]
Ring-necked Pheasant	*p(.)*	0.28 [0.09]
Savannah Sparrow	*p(.)*	0.84 [0.04]
Sedge Wren	*p(Grassland Type)*	0.14 [0.16]
Vesper Sparrow	*p(.)*	0.18 [0.09]

Table 4.4. Grassland bird species and global occupancy models with corresponding number of parameters, *K*, and overdispersion parameters, $\hat{c}$, observed in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Common Name	Global Model	*K*	$\hat{c}$
Bobolink	*Ψ(Grassland Type, Grassland Size), p(.)*	4	8.28
Dickcissel	*Ψ(Grassland Type, Grassland Size, Area Woody Vegetation, Litter Depth), p(Ordinal Date)*	7	0.88
Eastern Meadowlark	*Ψ(Litter Cover, Total Canopy Cover, Live Grass Canopy Cover, Grassland Size), p(.)*	6	1.36
Grasshopper Sparrow	*Ψ(Bare Ground, Grassland Size, Area Woody Vegetation), p(.)*	5	0.33
Ring-necked Pheasant	*Ψ(Standing Dead Vegetation, Grassland Size), p(.)*	4	0.25
Sedge Wren	*Ψ(Total Canopy Cover, Bare Ground, Canopy Height, Grassland Size), p(Grassland Type)*	7	0.23
Vesper Sparrow	*Ψ(Grassland Size), p(.)*	3	0.57

Table 4.5. Summary of AIC occupancy model selection adjusted by the best detection model for Dickcissel in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ (Grassland Size, p(Ordinal Date)	74.83	4	87.27	0.00	0.29
Ψ (Grassland Size, Area Woody Vegetation), p(Ordinal Date)	70.31	5	87.81	0.54	0.23
Ψ (.), p(Ordinal Date)	80.72	3	89.12	1.85	0.12
Ψ (Total Canopy Cover), p(Ordinal Date)	77.11	4	89.55	2.28	0.09
Ψ (Total Canopy Cover, Grassland Size), p(Ordinal Date)	73.09	5	90.59	3.32	0.06
Ψ (Grassland Type, Grassland Size), p(Ordinal Date)	73.58	5	91.08	3.81	0.04
Ψ (Litter Depth, Grassland Size), p(Ordinal Date)	74.30	5	91.80	4.53	0.03
Ψ (Area Woody Vegetation), p(Ordinal Date)	79.63	4	92.07	4.80	0.03
Ψ (Total Canopy Cover, Area Woody Vegetation), p(Ordinal Date)	75.56	5	93.06	5.79	0.02
Ψ (Litter Depth), p(Ordinal Date)	80.71	4	93.15	5.88	0.02
Ψ (Grassland Type), p(Ordinal Date)	80.72	4	93.16	5.89	0.02
Ψ (Litter Depth, Grassland Size, Area Woody Vegetation), p(Ordinal Date)	69.73	6	93.73	6.46	0.01
Ψ (Litter Depth, Total Canopy Cover), p(Ordinal Date)	76.25	5	93.75	6.48	0.01
Ψ (Grassland Type, Total Canopy Cover), p(Ordinal Date)	76.45	5	93.95	6.68	0.01
Ψ (Grassland Type, Grassland Size, Area Woody Vegetation), p(Ordinal Date)	70.15	6	94.15	6.88	0.01
Ψ (Grassland Type, Total Canopy Cover, Grassland Size), p(Ordinal Date)	71.18	6	95.18	7.91	0.01

Table 4.5 Continued.

Occupancy Model	-2 log-likelihood	*K*	AIC$_c$	ΔAIC$_c$	*w*
Ψ (Grassland Type, Litter Depth, Grassland Size), p(Ordinal Date)	71.82	6	95.82	8.55	0.00
Ψ (Litter Depth, Total Canopy Cover, Grassland Size), p(Ordinal Date)	72.11	6	96.11	8.84	0.00
Ψ (Grassland Type, Area Woody Vegetation), p(Ordinal Date)	79.14	5	96.64	9.37	0.00
Ψ (Litter Depth, Area Woody Vegetation), p(Ordinal Date)	79.63	5	97.13	9.86	0.00
Ψ (Grassland Type, Litter Depth), p(Ordinal Date)	80.71	5	98.21	10.94	0.00
Ψ (Grassland Type, Litter Depth, Total Canopy Cover), p(Ordinal Date)	74.37	6	98.37	11.10	0.00
Ψ (Grassland Type, Total Canopy Cover, Area Woody Vegetation), p(Ordinal Date)	75.54	6	99.54	12.27	0.00
Ψ (Grassland Type, Litter Depth, Total Canopy Cover, Grassland Size), p(Ordinal Date)	68.23	7	100.90	13.62	0.00
Ψ (Grassland Size, Area Woody Vegetation, Grassland Type, Litter Depth), p(Ordinal Date)	69.70	7	102.37	15.09	0.00
Ψ (Grassland Type, Litter Depth, Area Woody Vegetation), p(Ordinal Date)	79.14	6	103.14	15.87	0.00
Ψ (Grassland Type, Litter Depth, Total Canopy Cover, Area Woody Vegetation), p(Ordinal Date)	72.34	7	105.01	17.73	0.00

Table 4.6. Best fitting single occupancy models for Dickcissel and Grasshopper Sparrow and composite (i.e., model averaged) occupancy models for Sedge Wren (2006 data only) and Vesper Sparrow in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

				95% Confidence Interval	
Species	Parameter	β Estimate	SE	Lower	Upper
Dickcissel	Grassland Size	-1.93	1.43	-4.73	0.86
	Area Woody Vegetation in 100m Buffer	0.18	0.19	-0.19	0.55
Eastern Meadowlark	Intercept	-1.57	0.84	-3.21	0.08
Grasshopper Sparrow	Area Woody Vegetation in 100m Buffer	0.80	1.07	-1.30	2.90
Ring-necked Pheasant	Grassland Type	-2.74	2.74	-8.11	2.64
Sedge Wren	Grassland Size	0.34	0.35	-0.34	1.03
Vesper Sparrow	Intercept	1.25	1.80	-2.28	4.78

Table 4.7. Summary of AIC occupancy model selection adjusted by the best detection model for Eastern Meadowlark in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(.)	24.34	2	29.43	0.00	0.37
Ψ(Area Woody Vegetation), p(.)	23.23	3	31.63	2.20	0.12
Ψ(Litter Cover), p(.)	23.80	3	32.20	2.77	0.09
Ψ(Total Canopy Cover), p(.)	24.03	3	32.43	3.00	0.08
Ψ(Grassland Size), p(.)	24.25	3	32.65	3.22	0.07
Ψ(Live Grass Cover), p(.)	24.31	3	32.71	3.28	0.07
Ψ(Litter Cover, Area Woody Vegetation), p(.)	22.41	4	34.85	5.42	0.02
Ψ(Total Canopy Cover, Area Woody Vegetation), p(.)	22.78	4	35.22	5.79	0.02
Ψ(Grassland Size, Area Woody Vegetation), p(.)	22.94	4	35.38	5.95	0.02
Ψ(Litter Cover, Total Canopy Cover), p(.)	23.04	4	35.48	6.05	0.02
Ψ(Live Grass Cover, Area Woody Vegetation), p(.)	23.22	4	35.66	6.23	0.02
Ψ(Litter Cover, Grassland Size), p(.)	23.45	4	35.89	6.46	0.01
Ψ(Litter Cover, Live Grass Cover), p(.)	23.54	4	35.98	6.55	0.01
Ψ(Total Canopy Cover, Grassland Size), p(.)	23.73	4	36.17	6.74	0.01
Ψ(Total Canopy Cover, Live Grass Cover), p(.)	23.85	4	36.29	6.86	0.01
Ψ(Live Grass Cover, Grassland Size), p(.)	24.06	4	36.50	7.07	0.01

Table 4.7 Continued.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(Litter Cover, Total Canopy Cover, Area Woody Vegetation), p(.)	20.87	5	38.37	8.94	<0.01
Ψ(Litter Cover, Live Grass Cover. Area Woody Vegetation), p(.)	21.56	5	39.06	9.63	<0.01
Ψ(Total Canopy Cover, Live Grass Cover, Area Woody Vegetation), p(.)	22.15	5	39.65	10.22	<0.01
Ψ(Litter Cover, Total Canopy Cover, Grassland Size), p(.)	22.78	5	40.28	10.85	<0.01
Ψ(Litter Cover, Total Canopy Cover, Live Grass Cover), p(.)	23.04	5	40.54	11.11	<0.01
Ψ(Litter Cover, Live Grass Cover, Grassland Size), p(.)	23.37	5	40.87	11.44	<0.01
Ψ(Total Canopy Cover, Live Grass Cover, Grassland Size), p(.)	23.70	5	41.20	11.77	<0.01
Ψ(Litter Cover, Total Canopy Cover, Live Grass Cover, Grassland Size), p(.)	22.75	6	46.75	17.32	<0.01

Table 4.8. Summary of AIC occupancy model selection adjusted by the best detection model for Grasshopper Sparrow in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(Area Woody Vegetation), p(.)	33.09	3	41.49	0.00	0.35
Ψ(.), p(.)	38.02	2	43.11	1.62	0.16
Ψ(Bare Ground), p(.)	35.26	3	43.66	2.17	0.12
Ψ(Grassland Size), p(.)	36.63	3	45.03	3.54	0.06
Ψ(Bare Ground, Area Woody Vegetation), p(.)	32.71	4	45.15	3.66	0.06
Ψ(Litter Cover, Area Woody Vegetation), p(.)	33.03	4	45.47	3.98	0.05
Ψ(Total Canopy Cover, Area Woody Vegetation), p(.)	33.05	4	45.49	4.00	0.05
Ψ(Total Canopy Cover), p(.)	37.97	3	46.37	4.88	0.03
Ψ(Litter Cover), p(.)	38.00	3	46.40	4.91	0.03
Ψ(Bare Ground, Grassland Size), p(.)	34.85	4	47.29	5.80	0.02
Ψ(Total Canopy Cover, Bare Ground), p(.)	35.25	4	47.69	6.20	0.02
Ψ(Litter Cover, Bare Ground), p(.)	35.25	4	47.69	6.20	0.02
Ψ(Litter Cover, Total Canopy Cover), p(.)	35.25	4	47.69	6.20	0.02
Ψ(Litter Cover, Grassland Size), p(.)	36.43	4	48.87	7.38	0.01
Ψ(Total Canopy Cover, Grassland Size), p(.)	36.60	4	49.04	7.55	0.01
Ψ(Bare Ground, Grassland Size, Area Woody Vegetation), p(.)	32.46	5	49.96	8.47	0.01

Table 4.8 Continued.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(Litter Cover, Bare Ground, Grassland Size), p(.)	34.78	5	52.28	10.79	<0.01
Ψ(Total Canopy Cover, Bare Ground, Grassland Size), p(.)	34.78	5	52.28	10.79	<0.01
Ψ(Litter Cover, Total Canopy Cover, Grassland Size), p(.)	34.78	5	52.28	10.79	<0.01

Table 4.9. Summary of AIC occupancy model selection adjusted by the best detection model for Ring-necked Pheasant in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(.)	60.61	2	67.29	0.00	0.42
Ψ(Grassland Type), p(.)	60.61	3	67.85	0.56	0.32
Ψ(Grassland Size), p(.)	60.61	3	69.47	2.18	0.14
Ψ(Standing Dead Vegetation), p(.)	60.61	3	70.22	2.93	0.10
Ψ(Standing Dead Vegetation, Grassland Size), p(.)	60.61	4	73.05	5.76	0.02

Table 4.10. Summary of AIC occupancy model selection adjusted by the best detection model for Sedge Wren in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(Grassland Type)	22.35	3	30.75	0.00	0.43
Ψ(Grassland Size), p(Grassland Type)	19.69	4	32.13	1.38	0.21
Ψ(Canopy Height), p(Grassland Type)	21.18	4	33.62	2.87	0.10
Ψ(Bare Ground), p(Grassland Type)	21.24	4	33.68	2.93	0.10
Ψ(Total Canopy Cover), p(Grassland Type)	22.14	4	34.58	3.83	0.06
Ψ(Canopy Height, Grassland Size), p(Grassland Type)	19.05	5	36.55	5.80	0.02
Ψ(Bare Ground, Grassland Size), p(Grassland Type)	19.50	5	37.00	6.25	0.02
Ψ(Total Canopy Cover, Grassland Size), p(Grassland Type)	19.68	5	37.18	6.43	0.02
Ψ(Bare Ground, Canopy Height), p(Grassland Type)	20.37	5	37.87	7.12	0.01
Ψ(Total Canopy Cover, Bare Ground), p(Grassland Type)	20.98	5	38.48	7.73	0.01
Ψ(Total Canopy Cover, Canopy Height), p(Grassland Type)	21.07	5	38.57	7.82	0.01
Ψ(Total Canopy Cover. Canopy Height, Grassland Size),p(Grassland Type)	18.21	6	42.21	11.46	<0.01
Ψ(Bare Ground, Canopy Height, Grassland Size), p(Grassland Type)	18.89	6	42.89	12.14	<0.01
Ψ(Total Canopy Cover, Bare Ground, Grassland Size), p(Grassland Type)	19.49	6	43.49	12.74	<0.01
Ψ(Total Canopy Cover, Bare Ground, Canopy Height), p(Grassland Type)	20.37	6	44.37	13.62	<0.01
Ψ(Total Canopy Cover, Bare Ground, Canopy Height, Grassland Size), p(Grassland Type)	18.14	7	50.81	20.06	<0.01

Table 4.11. Summary of AIC occupancy model selection adjusted by the best detection model for Vesper Sparrow in CP23 native and CP1 introduced plant communities of Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan.

Occupancy Model	-2 log-likelihood	*K*	AIC_c	ΔAIC_c	*w*
Ψ(.), p(.)	57.97	2	63.06	0.00	0.63
Ψ(Canopy Height), p(.)	56.67	3	65.07	2.01	0.23
Ψ(Grassland Size), p(.)	57.69	3	66.09	3.03	0.14

APPENDIX 4.2: ALL AVIAN SPECIES OBSERVED

Appendix 4.2. Avian species observed in CP23 native fields, the introduced portion of CP1 fields, and CP1 whole fields, Conservation Reserve Enhancement Program (CREP) grasslands, Saginaw Bay watershed, Michigan, 2005-2006. Avian densities (# individuals /ha [SE]) are given for species with detection probability estimates ≥0.10. Species observed too rarely to support detection probability modeling (i.e., observed in < 10% of surveys) or with detection probability estimates <0.10 are noted as present when observed in at least one survey and as undetected when unobserved.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native/ CP23	Introduced	CP1
Grassland Specialist Species							
Bobolink [A]	*Dolichonyx oryzivorus*	0.38 [0.15]	1.33 [0.37]	2.28 [0.93]	0.60 [0.26]	0.68 [0.27]	0.35 [0.15]
Dickcissel [AB]	*Spiza americana*	0.04 [0.03]	0.08 [0.07]	0.05 [0.06]	0.03 [0.01]	0.05 [0.03]	0.05 [0.03]
Eastern Meadowlark	*Sturnella magna*	0.12 [0.09]	0.004 [0.004]	0.003 [0.003]	0.06 [0.04]	0.01 [0.01]	0.03 [0.02]
Grasshopper Sparrow [AB]	*Ammodramus savannarum*	0.05 [0.02]	0.00 [0.00]	0.00 [0.00]	0.08 [0.03]	0.01 [0.01]	0.01 [0.01]
Henslow's Sparrow [AC]	*Ammodramus henslowii*	present	undetected	undetected	present	undetected	undetected
Horned Lark	*Eremophila alpestris*	undetected	undetected	undetected	present	undetected	undetected
Northern Harrier [A]	*Circus cyaneus*	undetected	present	undetected	present	undetected	undetected
Ring-necked Pheasant	*Phasianus colchicus*	0.15 [0.10]	0.03 [0.02]	0.07 [0.04]	0.12 [0.07]	0.13 [0.08]	0.07 [0.06]
Savannah Sparrow	*Passerculus sandwichensis*	0.98 [0.39]	0.37 [0.12]	0.55 [0.17]	2.14 [0.62]	0.94 [0.32]	1.22 [0.46]
Sedge Wren [A]	*Cistothorus platensis*	0.01 [0.01]	0.00 [0.00]	present	0.07 [0.02]	0.06 [0.03]	0.03 [0.02]
Vesper Sparrow	*Pooecetes gramineus*	0.07 [0.03]	0.01 [0.01]	0.02 [0.01]	0.06 [0.01]	0.02 [0.01]	0.06 [0.04]

Appendix 4.2 Continued.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native	Introduced	CP1
Other Species							
American Goldfinch	*Carduelis tristis*	0.33 [0.13]	0.08 [0.03]	0.41 [0.32]	0.24 [0.08]	0.26 [0.07]	0.29 [0.08]
American Robin	*Turdus migratorius*	undetected	present	undetected	undetected	present	present
American Woodcock	*Scolopax minor*	undetected	undetected	undetected	present	undetected	undetected
Barn Swallow	*Hirundo rustica*	NA*	NA*	NA*	present	present	present
Brown Thrasher	*Toxostoma rufum*	undetected	undetected	undetected	undetected	present	present
Brown-headed Cowbird	*Molothrus ater*	present	undetected	undetected	present	present	present
Chipping Sparrow	*Spizella passerina*	undetected	undetected	undetected	undetected	present	present
Clay-colored Sparrow	*Spizella pallida*	present	present	present	0.05 [0.03]	present	present
Common Yellowthroat	*Geothlypis trichas*	0.03 [0.02]	0.05 [0.04]	0.09 [0.06]	0.12 [0.04]	0.10 [0.06]	0.16 [0.12]
Eastern Kingbird	*Tyrannus tyrannus*	present	present	present	0.03 [0.01]	0.004 [0.003]	present
House Wren	*Troglodytes aedon*	present	present	undetected	undetected	undetected	undetected
Indigo Bunting	*Passerina cyanea*	present	present	present	undetected	present	present

Appendix 4.2 Continued.

		2005			2006		
Common Name	Scientific Name	Native/ CP23	Introduced	CP1	Native/ CP23	Introduced	CP1
Killdeer	*Charadrius vociferus*	present	undetected	undetected	present	undetected	undetected
Mallard	*Anas platyrhynchos*	present	undetected	undetected	present	undetected	undetected
Marsh Wren [B]	*Cistothorus palustris*	undetected	undetected	undetected	present	undetected	undetected
Mourning Dove	*Zenaida macroura*	present	present	present	present	present	present
Northern Bobwhite Quail	*Colinus virginianus*	undetected	present	present	undetected	present	undetected
Northern Flicker	*Colaptes auratus*	present	undetected	undetected	present	undetected	present
Red-winged Blackbird	*Agelaius phoeniceus*	0.64 [0.26]	2.30 [0.29]	2.18 [0.62]	0.73 [0.19]	2.97 [0.36]	2.11 [0.26]
Song Sparrow	*Melospiza melodia*	1.54 [0.35]	0.46 [0.08]	1.47 [0.47]	1.89 [0.37]	0.76 [0.14]	1.12 [0.12]
Tree Swallow	*Tachycineta bicolor*	NA*	NA*	NA*	0.01 [0.01]	present	0.06 [0.05]
White-crowned Sparrow	*Zonotrichia leucophrys*	undetected	undetected	undetected	undetected	undetected	present
Wild Turkey	*Meleagris gallopavo*	present	undetected	undetected	present	undetected	present

[A] *Michigan Priority Landbird (Knutson et al. 2001, Matteson et al. 2009)*
[B] *Michigan Species of Special Concern*
[C] *Michigan Endangered Species*
** Barn Swallow and Tree Swallow were not included in the 2005 survey.*

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CHAPTER 5: SUMMARY AND MANAGEMENT RECOMMENDATIONS

Results of this study show that CREP grasslands in the Saginaw Bay watershed provide habitat for grassland bird species in Michigan, including those species of conservation priority in the State. Grassland birds are one of the most imperiled guilds of North American birds (Knopf 1996), and populations of grassland bird species continue to decline in Michigan (Sauer et al. 2008). The most recent population trend analysis of Breeding Bird Survey (BBS) data from 1980 – 2007 shows significantly decreasing population trends in Michigan ($p < 0.10$) for 4 out of 10 grassland bird species included in the BBS analysis (i.e., Bobolink, Eastern Meadowlark, Ring-necked Pheasant, and Vesper Sparrow; species included in the analysis were those encountered on ≥ 15 survey routes), and no species showed significant population increases (Sauer et al. 2008). CP23 and CP1 CREP grasslands of the Saginaw Bay watershed provide habitat for a majority of grassland birds found in Michigan; of the 15 grassland species documented in Michigan during development of the most recent Michigan Breeding Bird Atlas (BBAE 2010), 11 species were observed in study areas of this research project, including all 4 species with declining population trends from the BBS analysis (the 4 species found in Michigan during atlas development but not observed in this study were Le Conte's Sparrow, Sharp-tailed Grouse, Short-eared Owl, and Upland Sandpiper). Six grassland bird species observed during this study are among those that have been designated as Michigan priority land birds by the bird conservation group Partners in Flight (Knutson et al. 2001, Matteson et al. 2009) and have been given management priority by the Michigan Department of Natural Resources (MDNR). These species are Bobolink, Dickcissel, Grasshopper Sparrow, Henslow's Sparrow, Northern Harrier, and Sedge Wren (Knutson et al. 2001, Matteson et al. 2009), and include 1 Michigan endangered species (i.e., Henslow's Sparrow) and 3 Michigan special concern species (i.e., Dickcissel, Grasshopper Sparrow, and Northern Harrier).

Although it was not possible to evaluate the source/sink status of the CREP grasslands in this study due to unsuccessful efforts to gather avian productivity data, there is increasing evidence from other studies that farmland set-aside lands are benefiting grassland bird species at the population level (e.g., Veech 2006, Herkert 2009). Even small (3-142 ha) grassland fragments have been found to benefit the populations of some grassland birds, such as Dickcissel and Eastern Meadowlark (Walk et al. 2010), indicating that the loss of small habitat patches such as the CREP grasslands in this study could have negative, and perhaps cumulative, impacts on the populations of grassland birds, and underscoring the importance of maintaining farmland set-aside grasslands such as those in this study.

There is little doubt that density information is most informative about habitat quality when coupled with information about reproduction and survival of the species of interest. Van Horne (1983) raised consciousness that habitat quality as a measure of the importance of a habitat in maintaining the population of a species "should be defined in terms of the survival and production characteristics, as well as the density, of the species occupying that habitat." However, recently other researchers have provided evidence that density alone can be a reliable indicator of habitat quality (Bock and Jones 2004, Perot and Villard 2009). Bock and Jones (2004) surveyed 109 published studies of 67 European or North American bird species and found that habitats with higher densities displayed greater avian reproduction rates (i.e., greater recruitment per capita and per unit of land area) in the majority of cases (i.e., ≥72%. Bock and Jones (2004) concluded that "in most cases, density will be a reliable indicator of habitat quality, and bird-count data will be an appropriate basis from which to make land-management and conservation decisions."

Despite the greater cost of native seed mixtures (i.e., \$16.00/pound v. \$1.90/pound for the introduced seed mixture in 2005 in Tuscola County, Michigan) and the specialized planting techniques required for successful native vegetation establishment (e.g., use of native seed drills for planting the fluffy seed of some native grass species), I recommend prioritizing the planting of CP23 whole-field native grasslands within CREP in the Saginaw Bay area of Michigan. In this study, the densities of 5 out of 8 grassland bird species that could be included in density comparisons (i.e., those found in ≥ 10% of surveys and with $p \geq 0.10$) were higher in CP23 fields than in CP1 whole fields and/or the introduced portion of CP1 fields during at least one year of the study (i.e., Grasshopper Sparrow, Ring-necked Pheasant, Savannah Sparrow, Sedge Wren, and Vesper Sparrow). The densities of these 5 grassland bird species were higher in CP23 fields than in CP1 whole fields even though CP1 fields were large enough to support 0 - 22% more grassland bird breeding territories than CP23 fields, suggesting that CP1 fields containing introduced and native grasslands are not as attractive to several breeding grassland bird species as similar-sized or smaller CP23 fields containing only native grasslands. Additionally, the larger amount of standing dead vegetation in CP23 fields in early summer indicates that the native plant communities in Michigan CREP grasslands are providing more desirable winter habitat for overwintering wildlife such as Ring-necked Pheasant than are the introduced portions of CP1 fields (USDA 2000a, 2000b). Although these results suggest that CP23 CREP grasslands provided preferred habitat for most grassland species, one species of management priority, Bobolink, was more abundant in the introduced vegetation of CP1 fields during one year of this study. These results demonstrate the diverse response of grassland birds to habitat features, supporting the recommendations of other researchers that species-specific habitat preference information should be used when managing grasslands for grassland bird species (e.g., Murray et al. 2008).

Density comparison results of this study differ from those of other similar studies, which found no differences between grassland bird abundances in native and introduced vegetation or greater grassland bird abundances in introduced vegetation (e.g., McCoy et al. 2001, Wentworth 2010). In these studies, the structure and composition of native grasslands differed, often widely, from the native fields I examined, tending to be less diverse, have a denser canopy, and/or to be switchgrass (*Panicum virgatum)* dominated (i.e., dense stands of switchgrass have been found to be poor grassland bird habitat; Norment et al. 1999). In this project, detection probability estimates were adequate (i.e., $p \geq 0.10$) for all species observed frequently enough to estimate detection (i.e., observed in ≥ 10% of surveys), indicating that the survey method used in this investigation was appropriate for detecting grassland bird species in the study system. It is important to note that the consistency of results from studies in different regions may be affected by differences in grassland composition, age or successional stage, and structure. Differences in the findings of this study and of researchers in different regions underscores the importance of evaluating CREP and other such conservation programs at local scales (e.g., within states, ecoregions, or major watersheds). Local evaluations of the impacts of farmland set-aside programs on grassland bird populations are also important because grassland bird habitat requirements vary among geographic regions (e.g., Johnson and Igl 2001, Winter et al. 2006) and geographic variation in farmland set-aside lands exists due to local and regional climatic differences and variation in program implementation, including management practices (Heard et al. 2000, Riffell et al. 2008).

Occupancy modeling indicated that the occupancy of some grassland bird species in CREP grasslands of the Saginaw Bay watershed may be associated with CREP field type (i.e., CP23 or CP1) or grassland type (i.e., native or introduced) as well as grassland/field size. In CP23 and CP1 CREP whole fields, field type was associated with the occupancy of Sedge Wren and Vesper Sparrow and field size was associated with Dickcissel occupancy. In CP23 native and CP1 introduced grasslands, grassland type was associated with Ring-necked Pheasant occupancy, and grassland size was associated with Sedge Wren and Dickcissel occupancy. As expected from the coevolutionary history of native grasses and grassland birds (Sampson and Knopf 1996) and clear differences in native and introduced plant community composition, higher occupancies of some grassland bird species (i.e., Sedge Wren, Vesper Sparrow, and Ring-necked Pheasant) were linked to CP23 fields. Higher Sedge Wren occupancy was associated with larger grasslands (i.e., CP23 native grasslands ranged from 6.94 – 19.81 ha and CP1 introduced grasslands ranged from 5.53 – 17.01 ha), but unexpectedly, higher Dickcissel occupancy was associated with smaller fields and grasslands (i.e., see size ranges for introduced grasslands and CP23 fields or native grasslands above; CP1 whole fields ranged from 7.32 – 23.9 ha), a finding in contrast with a study documenting a positive relationship between habitat patch size and Dickcissel occurrence (Bakker et al. 2002) as well as other research that has shown habitat patch size to be an important positive influence on the occupancy of many other species of grassland birds (summarized in Ribic et al. 2009). Although field/grassland type and field/grassland size were not significant correlates of the occupancy of any species in my study (i.e., the 95% confidence intervals of the beta estimates for the effect of field/grassland type and field/grassland size on occupancy included 0), non-zero tendencies existed (i.e., 0 occurred near the tails of the 95% confidence intervals of the covariates), suggesting that these covariates should be included in future work that evaluates grassland bird occupancy in this or similar ecosystems. Such future studies would benefit from including more study site replicates to increase sample size and analytical power. Because occupancy covariates differed among grassland species, I also recommend that in similar studies the effect of potential occupancy covariates should be evaluated for individual species within the grassland bird guild rather than for grassland birds as a group.

Many other potential occupancy covariates were evaluated in addition to field/grassland type and field/grassland size, but these variables either were not included in top-ranking/competing models or zero was present centrally within the 95% confidence interval of the beta estimate (i.e., within the central 50% of the confidence interval), indicating that they were not important correlates of grassland bird occupancy. These potential covariates included the average distance to nearest woody vegetation for each species in each study field, the area of woody vegetation within 100 m of each study field, and 10 structural variables of grassland vegetation. When choosing breeding territories, grassland birds likely use a hierarchical process that is affected by variables at multiple scales (Ribic et al. 2009, Fisher and Davis 2010), and combinations of within-field and landscape-scale variables have been identified as important covariates in other studies of grassland birds (e.g., Ribic and Sample 2001, Bakker et al. 2002, Cunningham and Johnson 2006, Winter et al. 2006, Renfrew and Ribic 2008). With one exception, top-ranking/competing occupancy models did not include more than one covariate, likely due to constraints caused by small sample size (i.e., n = 14). My findings may have differed from other research due to analytical restrictions from small sample size (i.e., n = 14) and/or because of differences in the structure or composition of grassland vegetation, landscape composition, or other study system differences. Based on the results of other research, information from multiple scales (i.e., within-field and landscape-level variables) should be included in future grassland bird occupancy or abundance studies and in conservation management decision making processes.

Other study design improvements are possible in addition to easing or removing analytical constraints due to small sample size by increasing the number of fields/grasslands surveyed. The estimation of a species' detection probability and occupancy assumes that the occupancy of each study site is closed to changes between surveys (MacKenzie et al. 2002, MacKenzie et al. 2006), and violations of the closure assumption tend to result in overestimates of the probability of occurrence of a species, which could cause negative consequences when included in conservation and management decisions for rare or declining species (Rota et al. 2009). To more closely conform to the closure assumption of occupancy modeling, I recommend using life history information to truncate the occupancy data set for each grassland species based on the estimated date of initial flight by the first young of the year, or the date by which the population births would be likely to be observed during avian surveys. Alternatively, field surveys in breeding habitats could be timed to avoid population births to better conform to the closed population assumption of detection and occupancy modeling. In this case, field surveys would need to be conducted only until the latest estimated initial flight date among species of interest. In my study, detection probability estimates for 5 out of the 8 grassland species that could be included in detection modeling efforts (i.e., present in ≥ 10% of surveys) were higher for the truncated data set than for the full set of data (i.e., 15 May – 15 Aug), providing evidence that this approach is a better methodological technique. To increase the probability of detection of rare species, I also recommend maximizing the number of surveys conducted in each study field.

Maintaining and increasing the area of CREP grasslands should be a grassland bird conservation strategy in Michigan. The results of this study indicate that Michigan CREP fields, especially CP23 native grasslands, provide grassland bird habitat, a critical resource for an avian guild whose population declines have occurred more rapidly, more consistently, and over a wider geographic area than any other guild of North American birds (Sauer et al. 1995, Knopf 1996), declines that continue to occur in Michigan (Sauer et al. 2008). When planting new native grasslands, use of Michigan genotypes of native grass and wildflower species would augment and help maintain local genetic diversity of grassland plant communities. Also, planting monocultures of switchgrass should be avoided in grassland bird conservation initiatives, as dense stands of switchgrass have been found to be poor grassland bird habitat (Norment et al. 1999). Threats to the maintenance and increase of farmland set-aside grasslands include reconversion of set-aside lands to corn or other crop. Recently, increased demand for biofuels has resulted in reconversion of agricultural set-aside lands into crop production (Secchi and Babcock 2007, Searchinger et al. 2008, cited from Fargione et al. 2009), a trend that is likely to continue due to large, federally mandated increases in biofuel production (i.e., >700% over 2006 production by 2022; Fargione et al. 2009). Fargione et al. (2009) suggest minimizing the negative effects of increased biofuel demand on wildlife by utilizing wildlife-compatible biomass sources that do not require use of additional production lands (e.g., wastes, cover crops, algae), and maximizing the area of perennial grasslands, including diverse native prairie plantings, to produce biofuel in ways that are compatible with wildlife (e.g., prohibiting the harvest of biomass during the avian breeding season). Additionally, I recommend that program administrators provide incentives for landowners to join or continue in CREP through an increase in the one-time program sign-up bonus when possible, as this strategy that has been found to generate a greater increase in CREP enrollment than did smaller increases in the annual payments made over the term of the CREP contract (Suter et al. 2008). Continued landowner participation is essential to ensure the lasting presence of grassland bird habitat provided by farmland set-aside programs such as CREP.

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