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Plants and Breeding Bird Response on a Managed Conservation Reserve Program Grassland in Maryland

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Abstract

*Currently over 14.6 million ha of land at an annual cost of US$1.76 billion are enrolled in the Conservation Reserve Program (CRP). The habitat benefits of CRP frequently are lauded, but documentation that wildlife is responding as hoped is urgently needed. We evaluated plant and breeding bird responses to 92.4 ha of CRP grasslands at Chino Farms in northeastern Maryland, USA. In 1999 we seeded 12 contiguous CRP fields with 5 mixtures of warm-season grasses representing various growth-form heights in a replicated experimental design, and used mowing and topical herbicide applications to control noxious weeds and facilitate stand establishment. In 6 years cumulative plant species richness increased to 261, 105 of which were species exotic to the region. During the third growing season, we initiated a schedule of prescribed burning on a 3-year rotation to remove accumulated litter and to retard woody succession, and in 2003 we added additional management to control aggressive plant species. Several at-risk bird species colonized the restored grasslands in the first year and established sustainable breeding populations. We implemented a comprehensive observation and banding program, which included mapping male territories for selected bird species and recording nest locations. We marked 1,985 grasshopper sparrows (*Ammodramus savannarum*; GRSPs) in 7 years. Breeding GRSP populations ranged annually from 70 to 90 socially monogamous pairs with an additional 40 non-territorial males. Annual return rates in the last 5 years were 57% for adult males, 41% for adult females, and 12% for hatch-year individuals. Adults and young birds exhibited high site fidelity, but overgrown fields left unburned for 2–3 years were unpopulated by GRSPs but attracted several shrub-land bird species. Habitat preference for territories was influenced more by vegetation structure than by plant species composition. We recommend the management of grasslands restored for birds include spatial and temporal rotation of prescribed fire and herbicide applications to sustain vegetation physical structure rather than species composition.* (WILDLIFE SOCIETY BULLETIN 34(4):944–956; 2006)

Key words

Ammodramus savannarum, *Conservation Reserve Program, grasshopper sparrow, grassland restoration, habitat, invasive species, management, prescribed burning, species richness, vegetation structure, warm-season grasses.*

Native grasslands are among the rarest natural ecosystems in North America (Samson and Knopf 1994, Noss et al. 1995), and wildlife populations associated with these habitats, particularly grassland-nesting birds, have declined dramatically in recent decades (Knick et al. 2003, Sauer et al. 2005). Some historical records indicate that grasslands, pineland savannah, and "heathland" were extensive in the precolonial landscape in coastal New England (Mehrhoff 1997, Foster et al. 2002) and the mid-Atlantic region of the United States (Kulikoff 1986, Tyndall 1992, Askins 1997, Cronon 2003). Several authors (Gleason 1913, Patterson and Sassaman 1988, Delcourt and Delcourt 1997, Krech 1999) argue that frequent and widespread fires, ignited mainly by indigenous peoples and some lightning strikes, were major factors that maintained precolonial open habitats. Due to agricultural conversion, fire suppression, and increasing development of rural areas, none of the original eastern native grassland remains today (Johnson and Temple 1986, Herkert 1994*a*,*b*, Askins 1997). However, the assertion that fire-maintained grassland–savannah was extensive along coastal regions of the mid-Atlantic and northeastern United States is disputed (Foster et al. 2004).

The nationwide loss of grassland habitat has seriously affected grassland-obligate bird species, most of which appear on federal and state sensitive, threatened, and endangered species lists (Dinsmore 1994, Peterjohn and Sauer 1999). The heath hen (*Tympanuchus cupido cupido*), an endemic specialist of coastal, fire-maintained, open savannah, was abundant in the 17th–19th centuries; its extinction

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in 1932 on remote Martha's Vineyard serves as vivid testimony to the tragic loss of this once common habitat.

For 20 years the United States Department of Agriculture's (USDA) Conservation Reserve Program (CRP) and USDA–state agency partnerships through the Conservation Reserve Enhancement Program (CREP) have provided financial and technical assistance to landowners and managers to remove highly erodible lands from agricultural production and establish permanent cover to achieve natural resource conservation objectives, including improved wildlife habitat (McGuire 2003, Brennan and Kuvlesky 2005, Haufler 2005). Currently over 14.6 million ha of farmland are enrolled in CRP nationwide, of which 11.5 million (79%) are in grasslands–upland wildlife habitat (CP1, CP2, CP4, CP8, CP10, CP15, CP29, CP33), 2.79 million (19.2%) are in native block grasslands (CP2 alone), and $1.76 billion is obligated to CRP annually (<http://www.fsa.usda.gov/dafp/cepd/stats/apr2006.pdf>). In view of this level of public resources committed to grassland conservation, evaluation of effectiveness is urgently needed.

Although some information exists regarding the wildlife benefits of CRP (Johnson and Schwartz 1993*a*,*b*, Reynolds et al. 1994, USDA Farm Service Agency 2006), few studies have been conducted that document the success of restoration and management of eastern native grasslands (Mitchell et al. 2000). It is essential to determine if the extensive grassland habitats established through CRP and CREP are restoring prairie plant diversity and generating new source populations for birds and other wildlife, or whether they are creating local population sinks in the metapopulations generated abundantly by the current fragmented landscape (Wiens 1969, McCoy et al. 1999).

In this paper we report the results of the first 6 years of a study on a large, experimental CRP grassland on the mid-Atlantic coast that was designed to 1) test the capacity of alternative vegetation assemblages (restoration seed mixes) to rebuild local populations of grassland-obligate plants, birds, and other wildlife, 2) assess how prairie plants and ground-nesting birds respond to grassland management, and 3) develop protocols for the restoration and management of native grassland for practical implementation in the eastern United States.

Study Area

This paper reports some results of an ongoing study begun in 1999 on CRP portions of the Chester River Field Research Center (CRFRC) at Chino Farms, Inc., located in Queen Anne's County in eastern Maryland, USA, within the Atlantic Coastal Plain Physiographic Province of the Delmarva Peninsula. Chino Farms had intensive grazing on exotic pasture grasses for over a century prior to its conversion to intensive row-crop production (corn, wheat, barley, and soybeans) in the early 1950s. Beginning in 1985 hundreds of hectares of crop fields and buffers on Chino Farms were sequentially enrolled in the CRP and also since 1997 in Maryland CREP. Twelve contiguous fields constituting 92.4 ha (39.23°N, 79.00°W) were enrolled in Conservation Practice 2 (CP2; native grasslands) in 1998 and designed as a replicated experiment to evaluate the effects of grassland establishment and management treatments on vegetation and grassland-obligate species of birds (Fig. 1). Our goal was to reestablish stands of mid-Atlantic coastal grasslands similar to dominant types on the eastern Atlantic seaboard in precolonial times.

From 1989 to 2005 the annual temperature at Chestertown, Maryland, has averaged 55.7°F, and the annual total precipitation has averaged 44.8 inches (Fig. 2; data from National Oceanic and Atmospheric Administration website, <http://hurricane.ncdc.noaa.gov/ancsum/ACS>). These averages include the decade prior to the study period reported here to serve as baseline. Departures from the annual temperature norm show rather striking cyclic behavior including notable paired highs in 1990–1991, 1997–1998, 2001–2002, and 2004, and significant lows in 1992, 1996, 2000, and 2003. Record-breaking wet years occurred in 1989, 1996, and 2003, while the droughts of 1991–1994, 1997–1998, 2001–2002, and 2004 ruined crop production in the region. For the 17 months of June the mean temperature was 73.3°F and the mean precipitation was 4.36 inches (Fig. 2). Notable high temperatures in June occurred in 1994 and 2001 and exceptional lows occurred in 1992, 1997–1998, and 2003. June precipitation was excessive in 1989, 1996, and 2003, but drought conditions were prolonged for 5 years in the first half of the 1990s and were severe in 2002, 2004, and 2005. Exceptional during the study

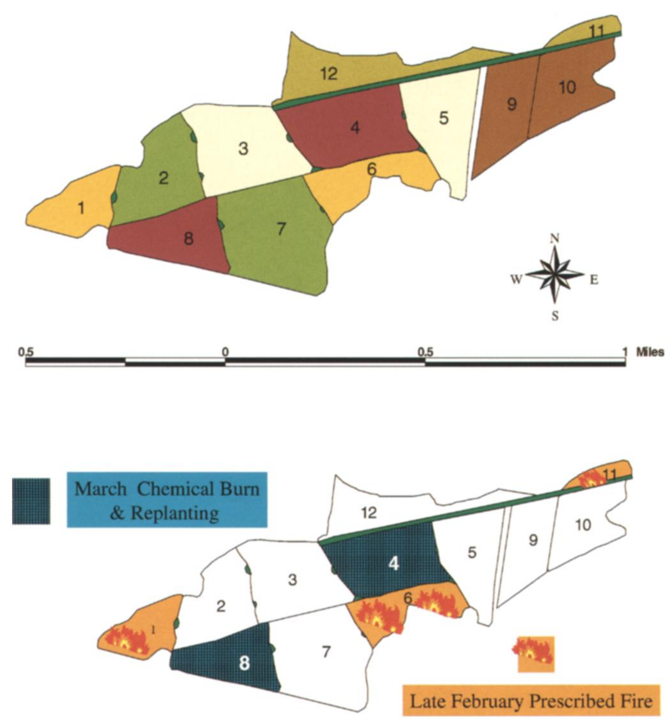

Figure 1. Upper: Plan view of the replicated experiment in native grassland restoration at Chino Farms, Maryland, USA. Fields in the same colors received identical installations in 1999 and management treatments every year (see Tables 1,2). Fields are approximately 3.2–12.9 ha. Lower: Example of prescribed fire management protocol, replicate fields 1 and 6, and 4 and 8, in 2004. See Table 2 for details of other years.

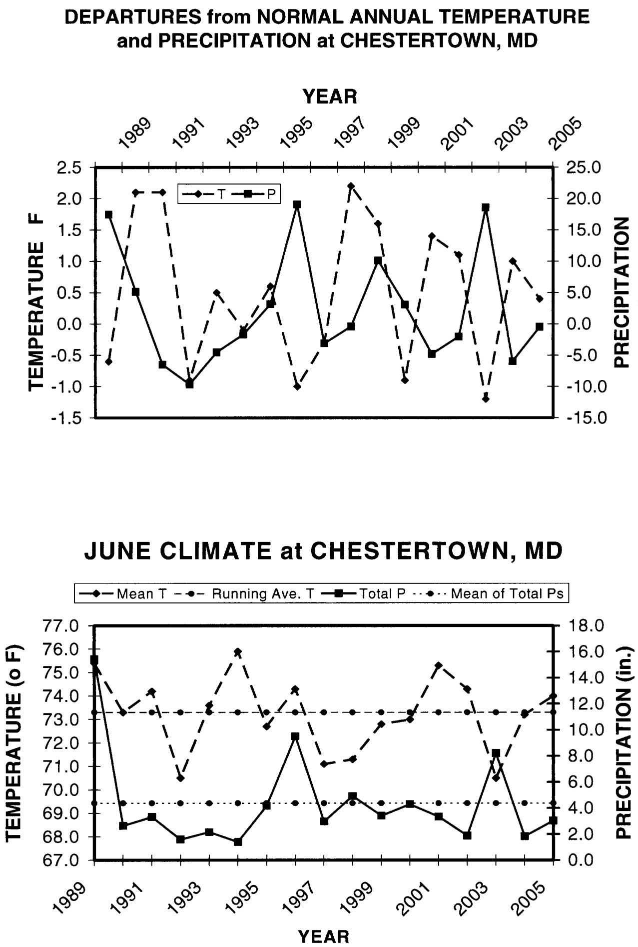

Figure 2. Temperature and precipitation at Chestertown, Maryland, USA, 1988–2005. Running mean annual temperature since 1970 to 2005 was 55.7°F, and average annual total precipitation from 1989 to 2005 was 44.8 inches. Upper: Departures of annual temperatures and precipitation from the annual norms above. Lower: Average temperatures and total precipitation for the months of June.

period, summer 2002 was oppressively hot and dry, and summer 2003 was record-breaking cold and flooding wet.

Methods

Grassland Establishment and Management

In April 1999 we used no-till drill methods to plant 8 native warm-season and 2 cool-season grass species in combinations of 3 species per treatment field. Randomly assigned to 2 replicate treatment fields ranging from 8 to 14 ha in area (Table 1), each mix represented a particular vegetation growth stature; the seed quantities were constrained by availability. One objective was to test experimentally the habitat value of 5 popular native grassland seeding mixes used for USDA grassland restoration in the area. Small bags of assorted prairie flowers were added to the bulk grasses to augment vegetation diversity in several fields. We implemented integrative management protocols, including mowing during establishment, prescribed fire on a 3-year cycle, and application of herbicides for noxious weed control as needed, to attain the vegetation stature desired for each seed

Table 1. Species and seeding rates used to establish grassland vegetation on treatment fields in 1999 at Chino Farms, Maryland, USA.

Field	Intended structure	Grass species	Seeding rate (kg/ha)
1, 6	Short	Little bluestem	4.5
		Sideoats grama	2.2
		Deertongue	2.2
		Blue grama	unknown
2, 7	Moderately short	Little bluestem	4.5
		Big bluestem	2.2
		Eastern gama grass	2.2
3, 5	Tall	Little bluestem	2.2
		Big bluestem	2.2
		Indian grass	4.5
4, 8	Moderately tall	Eastern gama grass	4.5
(1999–2003)	(original mix)	Switchgrass	2.2
		Red fescue[a]	2.2
		Tall meadow fescue[a,b]	unknown
4, 8 (2004)	Short (new mix)	Little bluestem	4.5
		Broomsedge	2.2
		Sideoats grama	2.2
9, 10	Mid-height	Coastal panic grass	4.5
		Little bluestem	3.4
		Indian grass	1.1

[a] Cool-season grass.
[b] Nonnative to North America. In the second growing season we discovered rows and patches of tall meadow fescue (*Festuca elatior* var. *arundinacea*) in fields 4 and 8, and surmise that some unknown number of seed bags labeled "Red Fescue" (*Festuca rubra*) actually contained seeds of tall meadow fescue. Similar discovery of small stands of blue grama (*Bouteloua gracilis*) in fields 1 and 6 indicated an unknown level of impurity in the bags labeled "Sideoats Grama".

mixture (Table 2; Fig. 1). By 2003 the proliferation of aggressive plant species required altering the original management schedules (see Table 2 for details).

Vegetation Studies

We monitored the establishment of grass stands and measured the rates of colonization of new plant species (species richness), changes in the relative abundance by percent cover, and rates of vegetative growth and reproductive performance of the principal prairie grasses (Schwartzman et al. 2002). Using randomizing procedures in ArcView 3.3 (Environmental Systems Research Institute [ESRI], Redlands, California), we located 8 random 1-m^2 vegetation plots in each field anew every month during the field season (Jun–Sep). These random samples provided independent sampling means and variances at several temporal and spatial scales; our time-series data refer to changes in sample means rather than repeated measures from the same individual genets. In each vegetation plot, we recorded the plant species present, percent cover of each species, heights of tallest vegetation, and number of vegetative and reproductive culms and lengths of tallest culms of the planted prairie grasses. In addition, by walking informal parallel transects, we searched each field for additional plant species not encountered in the random sample plots. Together these surveys provided minimum estimates of plant species richness.

Bird Studies

The University of Maryland Animal Care and Use Committee (IACUC R-03–36) approved the protocols used to conduct our bird studies. Beginning in 1999 we documented the bird use of restored grassland fields during the breeding season by daily observation and an intensive mist-netting and banding program (see details in Blank et al. 2003). We focused on grasshopper sparrows (*Ammodramus savannarum*; GRSPs) because of their abundance in our fields and because they are in sharp population decline throughout the eastern United States, including Maryland (Smith 1968*a,b*, Holmes 1996, Sauer et al. 2005). We monitored the grasslands from dawn to late morning (when territorial males often cease singing) most every day from mid-April to mid-September each year to locate banded and unbanded individuals. We caught the latter by strategically placing Japanese mist nets (4-shelf, 12-m-long, 2.6-m-high, 30-mm-mesh nylon nets, strung on 2.4-m aluminum conduit poles) on 1-m rebar posts temporarily installed next to activity sites (e.g., territorial perches and nest sites). We banded all individuals caught with an appropriate federal leg-band (Gustafson et al. 1997) and leg color-banded adult GRSPs and dickcissels (*Spiza americana*) with individually distinctive combinations of 11 colors. Thereafter, visual identification of birds every day with spotting telescopes, and precise recording using Global Positioning Systems (GPS) of perch sites, capture sites, and nests allowed us to document population sizes, behavior, and spatial and temporal activity of grassland songbirds in the study area. By mid-June we had marked virtually all breeding males, most female GRSPs, and most adults of other grassland-obligate species. We documented the breeding territories of singing male GRSPs, and continuous surveillance thereafter confirmed new breeding attempts by known resident GRSPs and ensured that new (unbanded or color-coded) breeding adults, though rare, were detected and targeted for capture and banding.

In July and August we used linear arrays of mist nets to capture birds that were not dependable by location (e.g., non-territorial males and hatch-year [HY] fledglings). In 1999–2002 when the vegetation was relatively short, we "swept" the fields by marching lines of nets across each field, 1 or 2 settings per morning, completing each field in 3–4 days. In 2003–2005, when vegetation in many of the fields was impenetrably tall (2–3 m) and dense, we placed long lines (up to 8 tandem nets) along firebreaks separating the study fields because both adult (off territory) and HY GRSPs fed actively in these firebreaks; these lines were moved to new locations every few days.

We documented territories of GRSPs by recording the coordinates of perches used by singing males with Garmin 12× handheld GPS units (GARMIN Corporation, Garmin International, Inc., Olathe, Kansas). Perches were predictably the tallest firm support above the surrounding vegetation, such as stalks of Canadian horseweed (*Conyza canadensis*), pokeberry (*Phytolacca americana*), and other emergent plants. Preliminary analysis from 2001 and 2002 indicated asymptotic estimates of territory area could be obtained from 8 or more GPS waypoints; in 2004 and 2005 as many as 25 waypoints were obtained for many males. Because males tend to advertise at the periphery of their territories, we delineated territories by constructing polygons around exterior waypoints in ArcView 3.3 (ESRI), using the "Polygon" rather than the "Kernel" procedure. Perch locations of non-singing birds were not included in construction of territory polygons.

In 2003, 2004, and 2005 we placed artificial perches in recently burned treatment fields to determine whether the addition of singing perches increased the attractiveness of recently burned areas for male songbirds to establish breeding territories. Detailed results of this experiment will be reported elsewhere (D. E. Gill et al., in preparation), but mention is made here because it was a habitat manipulation that affected the number and sizes of the territories recorded.

Locating nests in rapidly growing grasslands proved difficult, as others have reported (Winter et al. 2003). Carefully observing the activities of attentive adults was useful, but our nest-searching methods in the first 5 years (1999–2003) detected disappointingly few (<12) nests each year compared to the numerous territories and known breeding pairs documented. In 2004–2005 we adopted the labor-intensive technique of marching all available personnel shoulder-to-shoulder across those fields with short vegetation in order to flush brooding females off nests. This technique increased 5-fold the number of nests found in previous years.

Table 2. Management treatments applied to Conservation Reserve Program fields at Chino Farms, Maryland, USA.

	Replicate treatment fields and seeding mix growth form				
Date	**Fields 1, 6; short**	**Fields 2, 7; moderately short**	**Fields 9, 10; mid-height**	**Fields 4, 8; moderately tall**	**Fields 3, 5; tall**
Autumn 1998	Crop harvest	Crop harvest	Crop harvest	Crop harvest	Crop harvest
March 1999	Grass planted	Grass planted	Grass planted	Grass planted	Grass planted
Summer 1999	Mow	Mow	Mow	Mow	Mow
2000					
Apr 2001	Prescribed burn			Prescribed burn	
Apr 2002			Prescribed burn		Prescribed burn
2003		Apr burn		Autumn burn	
2004	Apr burn	Autumn burn	Autumn herbicide	Spring herbicide, replanted	Autumn burn
2005			Spring herbicide, disc, replant	Summer mowing	

Table 3. Percent cover ($\bar{x}$ ± 2 SE of 8 random 1-m^2 plots/field) of 7 warm-season grass species and red fescue planted in Chester River Field Research Center fields 1–8 during establishment (Jul 1999–2002), Maryland, USA.

Grass species	1999	2000	2001	2002
Big bluestem	0.09 ± 0.02	1.38 ± 0.86	3.77 ± 1.78	8.7 ± 3.94
Sideoats grama	0.03 ± 0.02	0.27 ± 0.18	0.21 ± 0.14	0.08 ± 0.08
Deertongue	0.03 ± 0.02	0.05 ± 0.06	0.11 ± 0.08	0.46 ± 0.42
Switchgrass	0.02 ± 0.02	2.07 ± 1.38	4.38 ± 2.36	8.44 ± 5.64
Little bluestem	0.09 ± 0.04	0.57 ± 0.30	3.50 ± 1.24	0.91 ± 0.54
Indian grass	0.03 ± 0.01	0.66 ± 0.42	3.40 ± 1.82	2.65 ± 1.72
Eastern gama grass	0.00 ± 0.00	0.02 ± 0.02	1.74 ± 0.74	0.88 ± 0.80
Red fescue	0.02 ± 0.02	0.55 ± 0.56	1.27 ± 1.18	1.02 ± 1.02
All warm-season grasses	0.28 ± 0.08	5.00 ± 3.18	17.07 ± 8.10	22.1 ± 13.06

Results

Grassland Establishment and Habitat Development

All 10 planted grass species germinated within 2 months of planting, established dense vegetation stands, and flowered and set seed by the end of the second growing season (Tables 3, 4). Vegetation height in tall-stature fields (fields 3 and 5) reached 2–3 m by late summer of the third year (Table 4). In addition to the prairie grasses and forbs we planted, many other new plant species appeared de novo every year, often in great abundance. By the end of 2004, we documented a cumulative list of 261 plant species on the study area, 105 of which were exotic to the region; native species recolonized the fields at a greater rate than nonnative species (Fig. 3). As measured by average percent cover pooled across all random plots on the study area each year, the dominant plant species changed in each of the first 6 years (Fig. 4). After 3 years of vigorous growth, the severe drought of 2002 retarded vegetation response. In contrast,

Table 4. Mean culm density (culms/genet/m^2 ± 2 SE) and mean height of tallest culms (cm ± 2 SE) of 7 warm-season grasses from Jul sampling in Chester River Field Research Center fields 1–8 over the first 4 establishment years, Maryland, USA.

		Year			
Species	Culm	1999	2000	2001	2002
Big bluestem					
Nonflowering	Density	4.7 ± 0.8	8.6 ± 2.6	9.2 ± 6.4	32.5 ± 6.6
	Height	29.6 ± 4.0	68.0 ± 11.2	61.3 ± 14.2	79.8 ± 5.2
Flowering	Density		21.5 ± 4.8	37.0 ± 9.4	39.5 ± 10.4
	Height		86.8 ± 5.2	180.8 ± 10.4	119.0 ± 14.8
Indian grass					
Nonflowering	Density	2.4 ± 0.4	6.0 ± 2.0	9.9 ± 3.2	23.7 ± 5.2
	Height	22.1 ± 4.0	47.1 ± 5.0	56.8 ± 5.8	88.3 ± 5.0
Flowering	Density		22.9 ± 5.8	49.4 ± 11.4	42.0 ± 31.9
	Height		85.1 ± 6.0	166.9 ± 11.4	121.8 ± 19.8
Little bluestem					
Nonflowering	Density	10.4 ± 1.6	13.0 ± 4.2	17.5 ± 7.4	33.9 ± 7.2
	Height	18.9 ± 2.0	43.6 ± 6.6	65.4 ± 7.8	45.0 ± 4.0
Flowering	Density	33.0 ± 14.5	31.6 ± 5.4	65.5 ± 11.6	73.3 ± 14.8
	Height	37.2 ± 22.2	54.0 ± 3.0	113.5 ± 4.8	75.9 ± 6.0
Sideoats grama					
Nonflowering	Density	11.0 ± 2.6	6.6 ± 2.0	9.0 ± 9.2	12.1 ± 3.6
	Height	15.2 ± 2.2	22.6 ± 3.0	52.0 ± 19.6	39.7 ± 9.6
Flowering	Density	32.6 ± 5.8	28.2 ± 16.2	39.3 ± 13.0	16.7 ± 13.4
	Height	42.5 ± 4.6	32.2 ± 4.6	67.4 ± 9.4	49.7 ± 13.4
Deertongue					
Nonflowering	Density	3.5 ± 2.0	2.1 ± 1.0	7.7 ± 2.8	10.1 ± 4.6
	Height	12.8 ± 7.4	11.4 ± 3.8	42.6 ± 4.4	58.9 ± 6.4
Flowering	Density	0 ± 0	0 ± 0	0 ± 0	0 ± 0
	Height	0 ± 0	0 ± 0	0 ± 0	0 ± 0
Switchgrass					
Nonflowering	Density	7.4 ± 2.6	15.2 ± 8.0	8.0 ± 3.6	9.0 ± 4.0
	Height	53.8 ± 7.6	96.6 ± 11.6	89.8 ± 10.0	162.0 ± 54.0
Flowering	Density	14.1 ± 3.8	41.6 ± 9.8	54.1 ± 13.0	34.4 ± 13.2
	Height	84.7 ± 7.6	118.9 ± 7.6	179.3 ± 9.4	156.3 ± 12.0
Eastern gama grass					
Nonflowering	Density		14.2 ± 7.0	19.3 ± 6.6	65.0 ± 55.5
	Height		70.2 ± 9.8	79.0 ± 19.4	130.0 ± 41.8
Flowering	Density				30.0 ± 0.0
	Height				96.0 ± 0

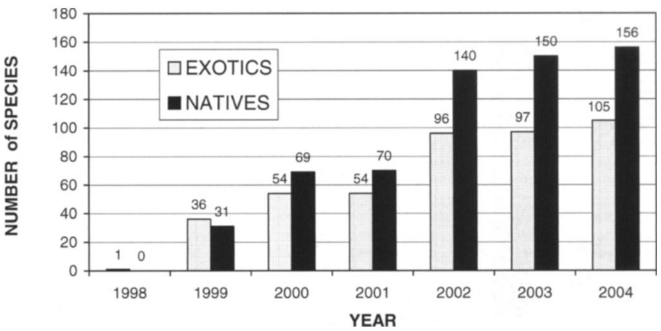

Figure 3. Cumulative plant species richness in restored grassland fields at Chino Farms, Maryland, USA.

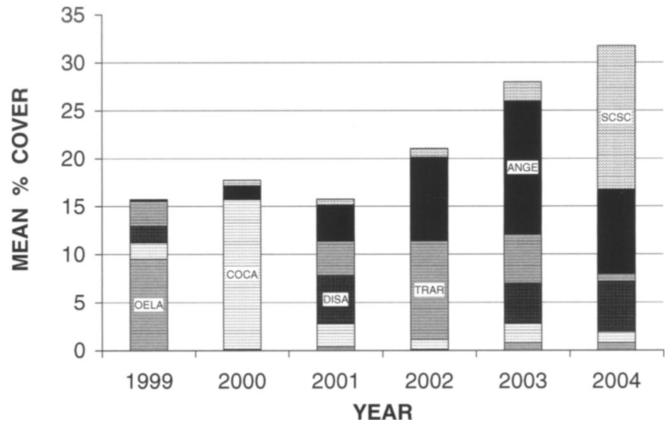

Figure 4. The dominant species of plants, as measured by percent cover, found in Chester River Field Research Center grasslands, Maryland, USA. OELA = *Oenothera lacniata*, COCA = *Conyza canadensis*, DISA = *Digitaria sanguinaria*, TRAR = *Trifolium arvense*, ANGE = *Andropogon gerardii*, SCSC = *Schizachyrium scoparium*.

in 2003 record-breaking cold temperatures and rainfall (which flooded many of the fields) stimulated the emergence of numerous new species of wetland plants (e.g., sedges and rushes) in dense stands (thousands of stems per m^2) in areas where they had not been evident in the previous 4 years. As expected, informal vegetation transects across the fields yielded 3–4 times as many plant species than found in 1-m^2 random plots (Table 5).

Management treatments strongly influenced grassland vegetation structure and composition. Prescribed burning stimulated vigorous growth of several grasses, especially big bluestem (*Andropogon gerardii*) and Indian grass (*Sorghastrum nutans*; Fig. 5). Emergence of exotic plant species, however, was faster in burned than unburned plots (Fig. 5).

Nine aggressive plant species, including 3 exotics and 6 natives, became dominant ground covers in several fields during the experiment and management concerns. By state statute landowners in Maryland are required to control certain noxious weeds. Topical application of herbicides (brand name, active ingredient, manufacturer: Roundup, glyphosate, Monsanto; Stinger, clopyralid, DowElanco; Plateau, Imazapic, O-BASF; 2,4-D, 2,4-D, Dow AgroSciences) and physical excavation by hand successfully controlled Johnson grass (*Sorgum halepense*; Fig. 6) and Canada thistle (*Cirsium arvense*). From relative obscurity in the first 3 establishment years, the exotic rabbitfoot clover (*Trifolium arvense*) proliferated in 2002, then unexpectedly declined in 2003 and further still in 2004 without any specific control measures (Fig. 7). Thus, 2 of these exotic invasive plants were easily controlled with management, and the third simply required patience.

Table 5. Species richness (no. of species) detected within 8 random sampling plots and 4 informal walking transects on Chester River Field Research Center fields sampled in early Jul 2002, Maryland, USA.

Field	Random 1-m^2 plots	Informal transect samples
1	20	68
2	17	59
3	22	69
4	14	56
5	27	83
6	27	85
7	21	93
8	15	71
9	30	61
10	30	58
12	32	74

Aggressive native woody and herbaceous species presented more enduring challenges than the exotics. Black locust (*Robinia pseudo-acacia*) rapidly spread by root-sucker sprouts from border hedge-rows into fields 7 and 8 in 2 years. Prescribed fires in the spring of 2002, application of 2,4-D herbicide in the autumn of 2003, and cutting and brush-hogging in 2004 and April 2006 temporarily curtailed its spread. In 2004 dwarf–shining sumac (*Rhus copallina*) unexpectedly invaded all of fields 6 and 7 in large numbers and reached heights of 2–3 m in 2 years. By 2004 fields 9 and 10 became carpeted by the native vine trumpet creeper (*Campsis radicans*). Prescribed fires in early December 2004 and chemical burn-down and replanting in the new short-stature seed mix in spring 2005 produced favorable reductions in this aggressive vine. Although switchgrass (*Panicum virgatum*) is a warm-season grass native to North America and was included in the initial USDA-recommended seed mixes used in 1998, its aggressive rhizatomous growth elevated it to a dominant monoculture in fields 4 and 8 within 3 years and its density caused a decline in the plant species richness in those experimental fields ($P < 0.01$; Fig. 8). We activated vigorous measures to remove switchgrass from those 2 fields (autumn 2003 prescribed fire, spring 2004 herbicide burn-down and replanting to a new mixture of low-stature warm-season grasses including little bluestem [*Schizachyrium scoparium*], broomsedge [*Andropogon virginianus*], and sideoats grama [*Bouteloua curtipendula*]; Fig. 1). In 2005 deertongue (*Dicanthelium clandestinum*), which had been sparse and minimally reproductive for 6 years, suddenly formed dense stands in field 6. Despite its popularity in grassland restoration seeding mixtures, sideoats grama steadily declined in abundance in fields 1 and 6, indicating the unsuitability of this species for eastern Maryland soil conditions (Fig. 9).

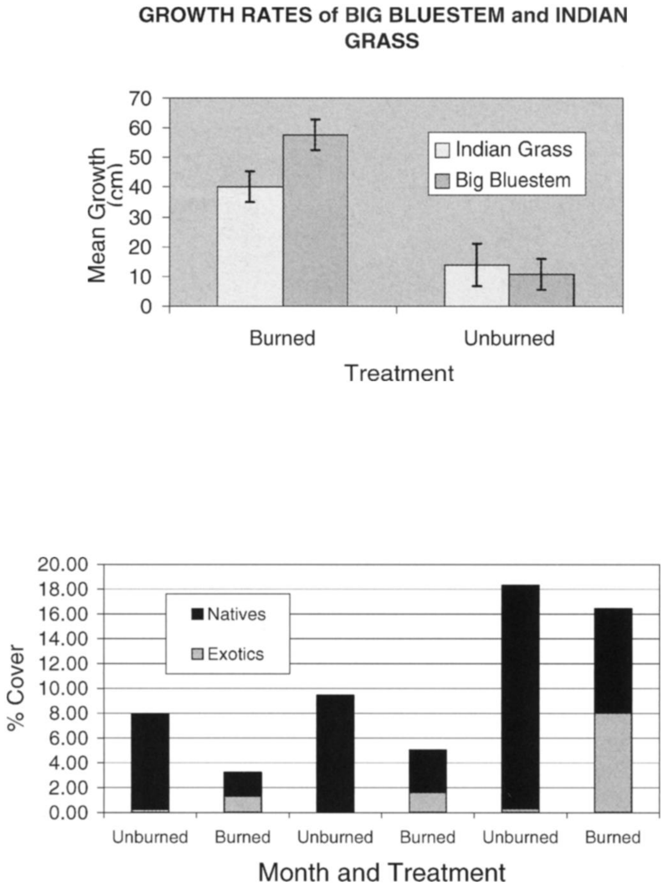

Figure 5. Upper: Mean growth rates of Indian grass and big bluestem in plots burned in Apr 2002 and unburned plots, Chester River Field Research Center (CRFRC) grasslands, Maryland, USA. Lower: Percent cover of native and exotic vegetation in plots burned in early Apr 2002 and unburned plots, CRFRC grasslands, Maryland, USA.

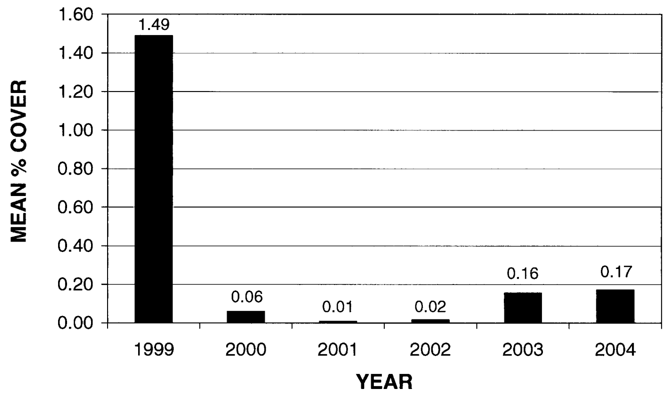

Figure 6. Changes in percent cover in Johnson grass in Chester River Field Research Center, Maryland, USA, grasslands over the first 6 years of restoration, 1999–2004, following topical application of herbicides and manual up-rooting.

Bird Response

Colonization of the restored grasslands by obligate grassland birds started within 1 month of planting. Horned larks (*Eremophila alpestris*), killdeer (*Charadrius vociferus*), and GRSPs appeared in April 1999 and immediately established territories, engaged in courtship, and reared young. Other grassland bird species of special concern, including vesper sparrow (*Pooecetes gramineus*), northern bobwhite (*Colinus virginianus*), and dickcissel, also colonized the grasslands within 2 years of establishment, albeit in smaller numbers. In 2004 and 2005, respectively, 1 and 2 sedge wrens (*Cistothorus platensis*) appeared and established late-summer territories.

The rapid colonization of the CRP fields by numerous GRSPs prompted our interest in studying them as bio-indicators for comparing grassland-nesting bird habitat value among the various treatments on the study area. Our mist-netting and color-banding techniques were satisfactory for determining the breeding population size and structure—most adults often were captured multiple times each breeding season, and by July each year new (unbanded) adult individuals were rarely seen or caught.

We banded 1,985 GRSPs on the restored grassland study area from 1999 to 2005, and another 200 were banded on other nearby grasslands. The average number of individual adults detected in the study area during the last 5 years was 223.4 individuals, with a high of 292 in 2001 (Fig. 10). The number of breeding pairs (territories) ranged from 70 to 80 each summer, and territory density reached 3–4 nonoverlapping territories/ha in fields with favorable habitat. Thirty to 50 additional males in breeding condition but not holding territories (not singing on perches with any regularity or predictability) also were present in most years.

Breeding adult and HY GRSPs returned to the CRFRC grasslands at rates nearly double previous estimates for most passerines (Gill 1990, Wells 1997) during the study period (Fig. 10). With nearly all of the breeding population caught, marked, and studied every summer, we confirmed that an average of 57.3% of the breeders each year returned from previous years. Record highs include 1) 83% of the males breeding in 2000 returned in 2001, 2) 62% of 2003 females returned in 2004, and 3) a staggering 22% of the 2003 hatchlings returned to breed in 2004 (Fig. 10). We estimate we caught and banded half of the 360–400 fledglings

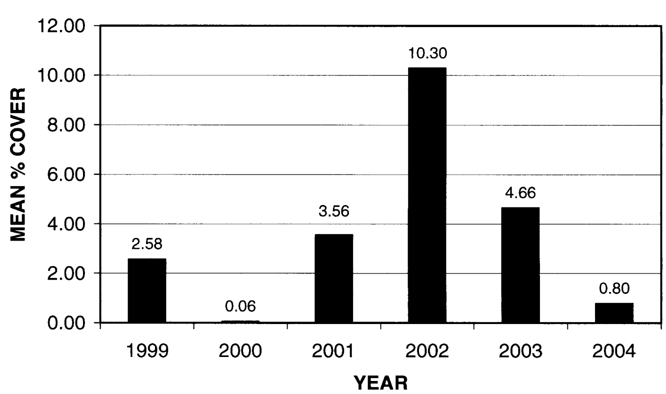

Figure 7. Changes in percent cover of rabbitfoot clover, a legume exotic to North America, in Chester River Field Research Center, Maryland, USA, grasslands in the first 6 years of the restoration project. Its decline after 2002 occurred without any specific remediation.

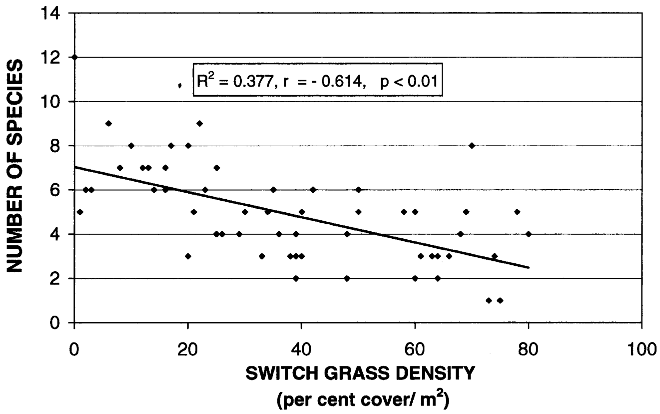

Figure 8. Plant species richness in 1-m² vegetation sample plots from fields 4 and 8 as a function of switchgrass density, Chester River Field Research Center, Maryland, USA.

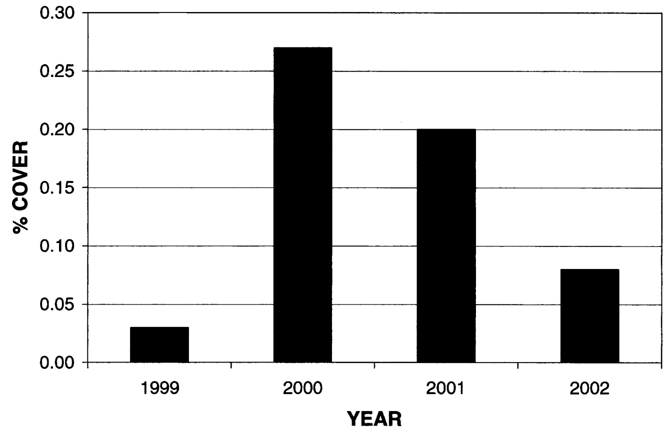

Figure 9. Changes in mean percent cover of sideoats grama in fields 1 and 6 in 16 random plots in Jul during the first 4 grassland establishment years at Chino Farms, Maryland, USA.

produced from the estimated 120 successful nests on the study area each summer.

Arriving from unknown wintering grounds in late April, grasshopper sparrows colonized and bred in all fields through late August (we had a record late nesting on 7 Sep 2000) in the first 3 years of the study, and most years thereafter, regardless of planting mix used. All fields during establishment had territorial males, nesting activity, and similar nesting success despite differences in plant composition. Four breeding cycles are possible each summer, each 1 month long if successful, but only 2 are typically used and we have a confirmed record of a female attempting 3 nests in 2005.

We observed intense territory site fidelity by males on the study area. For example, the oldest GRSP ever recorded (M. Gustafson, United States Geological Survey, Patuxent Wildlife Refuge, personal communication), male BT-BX, was at least 8 years old in 2005, and returned to the same territory site over 7 years. Nevertheless, slight shifts in location of the territories of some individual male GRSPs did occur between successive years when the vegetation was greatly altered. For example, field 7 had numerous GRSP territories in 2003 following a spring management burn but virtually no territories in 2004 when the field was covered by dense, 2- to 3-m tall prairie grasses (Fig. 11); those males that held territories in field 7 in 2003 and returned in 2004 shifted their 2004 territories to new areas in adjacent fields only a few meters away from their old turfs. As unburned fields deteriorated in attractiveness to GRSPs, they often became attractive to other species, such as dickcissels, red-winged blackbirds (*Agelaius phoeniceus*), indigo buntings (*Passerina cyanea*), blue grosbeaks (*Guiraca caerulea*), and migrant flocks of bobolinks (*Dolichonyx oryzivorus*) in August as the vegetation thickened.

Nests of GRSPs typically were on the ground, tucked into the base of a tussock of little bluestem, into a narrow windrow of thatch left behind by a mower, or in a scrape under a fallen cornstalk or stem. Fledging success per nest was estimated at 83% and 75% in 2004 and 2005, respectively. Although nest success in 1999–2003 was 50–60%, nest sample sizes during these years were too low to state values with confidence.

We found no brown-headed cowbird (*Molothrus ater*) eggs in any of the 213 GRSP nests we located, nor in any of the nests of other grassland birds, such as blue grosbeak, indigo bunting, the rare dickcissel (Smith 1996), common yellowthroat (*Geothlypis trichas*), song sparrow (*Melospiza melodia*), red-winged blackbird, and field sparrow (*Spizella pusilla*), observed in the grasslands. Adult brown-headed cowbirds rarely were observed at the CRFRC grasslands in spring and summer.

Absent from our study are quantitative evaluations of the bird and wildlife use of these grasslands in autumn, winter, and early spring.

Discussion

Our results demonstrate that complex ecosystems, such as eastern native grasslands, can be reassembled on heavily degraded lands through management within a relatively short time frame of 2–4 years. By "reassemble" we mean increasing species diversity by natural colonization so that the area becomes attractive to and is capable of supporting sustainable populations of species in other trophic levels within a few years. We are confident that the positive response of grassland specialists on the CRFRC grasslands primarily was due to the large blocks (CP2) of grassland habitat established on CRP fields (Johnson and Igl 2001) because, in contrast, narrow linear grass-buffer strips (such as CP21) in Maryland attract very few grassland-specialist bird species (Blank and Gill 2006).

Vegetation Response

Dozens of new plant species were discovered every year on the restored CRFRC grasslands, most notably in areas carefully searched in previous years. We surmise that the source of most new plant species that were discovered each year was seed germinating from soil seed banks rather than new propagules arriving by wind and/or animal transport. Preliminary laboratory experiments of germinating seeds

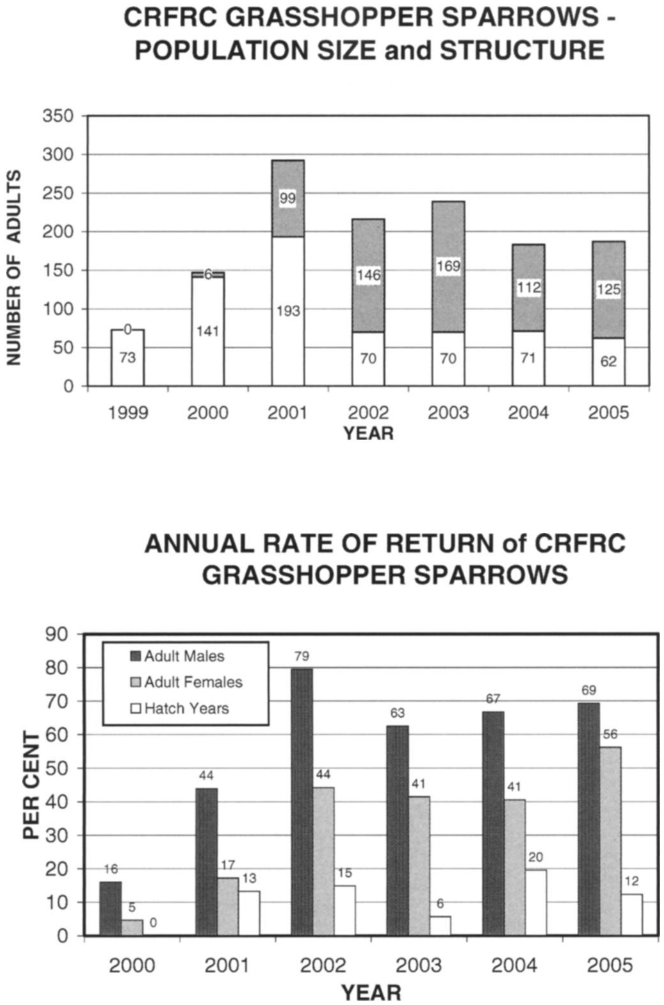

Figure 10. Upper: Population structure of grasshopper sparrows (GRSPs) detected on restored grasslands at Chino Farms, Maryland, USA. Shaded bars are returning adults, open bars are new, previously unmarked adults. Lower: Annual return rates of adult males (densely shaded bars), adult females (speckled bars) and nestling–fledgling (open bars) GRSPs at the Chester River Field Research Center grasslands, Maryland, USA.

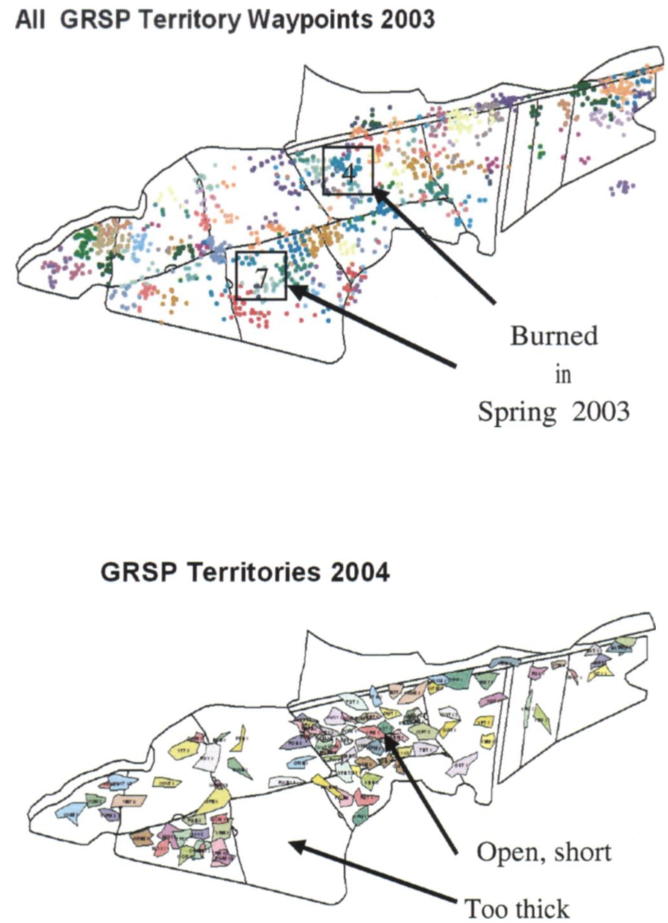

Figure 11. Spatial density of territorial male grasshopper sparrows (GRSPs) on restored grasslands, Chester River Field Research Center, Maryland, USA. Upper: Waypoints of perched singing males in 2003, color-coded for each individual male. Lower: Waypoints of perched, singing male GRSPs in 2004, with peripheral points connected into polygons by ArcView 3.3 to depict nonoverlapping territories. Note changes between years in density of GRSPs in fields 4 and 7 due to management protocols.

from CRFRC soils and soils from neighboring crop fields support this conclusion (D. E. Gill, University of Maryland, unpublished data), but definitive experiments are still needed. Cumulative plant species richness appears to be approaching a stable level of around 300 species on the study area; species turnover vis-à-vis island biogeographic theory is expected, but we do not yet have reliable estimates of annual turnover rate. Actual species richness in any given year is difficult to determine because the absence of species in samples is most easily explained as failure of detection (Longino et al. 2002); disappearance does not necessarily imply local extinction. A seemingly inexhaustible bank of dormant but viable seeds in the soil seems remarkable given the more than 200-year history of intensive pastoral and agricultural use of the area, the most recent of which consisted of row-crop monocultures. Resident farmers, however, express no surprise in the viability of seed banks in local soils (E. Miles and W. Kemp, Bluestem Farms, Chestertown, Maryland, personal communication).

To date only 9 undesirable plant species have invaded the grassland fields, but 6 aggressive native species required stronger actions of control than did 3 exotic plant species. Simple and inexpensive measures of topical herbicide application and mechanical uprooting easily controlled the exotic noxious weeds Johnson grass and Canada thistle. The proliferation and auto-decline of the exotic rabbitfoot clover was instructive. We surmise that the process of increasing density and maturation of the native grassland species caused the terrain to be inimical to rabbitfoot clover, either through interspecific competition or unknown and undetected factors.

Our brief experience with rabbitfoot clover illustrates the need for land managers to approach intervention measures with caution and the potential advantages of capitalizing on natural processes. Even this conclusion might be hasty, however, because new patterns are appearing as we write: those fields (4 and 8) that had been chemically burned and replanted in 2004 are suddenly carpeted in rabbitfoot clover and four vetch species (*Vicia* spp.) in May and June 2006. While 2 of the 3 exotic invasive plant species on the managed grasslands remain in check, the 100+ other nonnative plant species found in the study area appear to be contributing ecological services such as providing habitat structure, nectar resources for pollinators, and enriching the soil.

The aggressive behavior of 6 native plant species has been a much greater concern than the "invasiveness" of exotic species. Our vigorous action to curtail switchgrass appears to have been successful, but persistent management may be necessary to prevent reestablishment of switchgrass monocultures in the future. The invasion of the grasslands by black locust, winged sumac, persimmon, and trumpet creeper is, of course, simply the process of secondary succession. Our stated objective was the restoration of coastal grasslands and, therefore, we anticipated we would need management protocols to retard secondary succession by woody species. Control of aggressive woody species on a large scale remains the most costly challenge in maintaining grassland habitats at Chino Farms.

Bird Response

The 2 most striking results of the bird studies were the rapid colonization by grassland-obligate species and the high rates of known survival and annual return of GRSPs on the restored grasslands. The origin of the large numbers of colonizing GRSPs in April 1999 is unknown. We had guessed that they originated from distant pastures because they seemed not to be present (or sparsely so) in cultivated crop fields elsewhere in the vicinity. Our recent surveys contradict that earlier impression. We have observed some singing GRSPs in wheat, bearded barley, corn, and soybean fields in early, mid-, and late summer. We doubt that nesting is successful in these crop fields because of the persistent farming practices, machinery, harvesting, etc., but have no data at present to compare success in active croplands to our CRP grasslands. In contrast, GRSPs did promptly fill our study fields that had been recently burned or replanted, packing 2–3 nonoverlapping territories into each hectare of suitable habitat. Our results demonstrate that large-scale grassland restoration projects can provide habitat suitable for locally rare species to colonize rapidly.

The high rate of annual return by GRSPs, not only of breeding adults but also prior-year nestlings returning as new breeding adults, was extraordinary compared to published rates for most other birds and other GRSP populations (Delany et al. 1993). The actual number of returning GRSPs is likely much higher than our records show—many of the unbanded new breeders are likely HY individuals produced on the study area that were not caught in previous years. Adult males outnumbered females by 2:1 during the breeding season apparently because of significantly higher annual survivorship in males, as reflected by the higher rates of males over females (Fig. 10). The number of adult females matches the number of territories, suggesting that all females are breeding every year. This, in turn, generates a surplus of non-territorial males. Because most non-territorial males were unbanded when initially captured, we interpret these "floaters" as some combination of new arrivals, and behaviorally subordinate, second-year males that are unable to establish territories in the crowded grassland but roam the area, and persistently challenge territorial males. These results indicate that the restored mid-Atlantic grasslands on our study area provide habitat that consistently attracts GRSPs and supports sustainable reproduction from adults and young alike, year after year.

The site fidelity of both breeding adult and returning young birds at the restored CRFRC grasslands we observed is noteworthy. These birds are migratory to unknown winter sites; none has yet been recovered from wintering grounds nor seen on the study-area grasslands between late October and mid-April. Breeders and non-territorial adults return annually in late April at high rates, not only on the scale of the 94-ha grassland study area but also to previously occupied 0.3- to 0.4-ha territories. In so doing, the change in vegetation (growth within the summer and accumulation between years) that individual GRSPs tolerate in defending and reoccupying territory sites is remarkable.

Rotational management practices applied to treatment fields created varying habitat conditions for GRSPs between successive years (Dechant et al. 2003). These shifts in habitat conditions forced some individual males and females to move their territories each year, but the movement was in the order of 50–100 m to suitable places essentially adjacent to their previous territories. For example, field 7 had the greatest density of GRSP territories in 2000 and 2001, followed by a decline in use in 2002 because the 3-year-old vegetation had grown impenetrably thick and rank. This field had become very attractive again in 2003 following its first prescribed burn, but became unattractive the following year because of the vigorous grass growth (Fig. 11). These results indicate that changes in habitat quality, as illustrated by actual GRSP use, were primarily due to the change in structure of the habitat rather than plant species composition, which had changed little due to the dominance of perennial species. Our schedule of management protocols, especially continual control of encroaching woody vegetation, is the most important factor dictating the quality of GRSP habitat.

We were puzzled in 2001 and 2002 why returning male GRSPs did not instantly colonize the most open, restored grassland fields the way they initially did in 1999. Habitat quality and food availability in these sites appeared better, perhaps ideal ecologically for GRSPs, yet occupation was minimal in recently burned fields for the first month. The positive response to a controlled experiment in 2003 in which tomato stakes were added in replicate grids to newly burned fields (D. E. Gill, unpublished data) suggested the essential requirement of singing perches for territorial males (Madden et al. 1999; D. E. Gill et al., in preparation). Our results to date suggest strongly that habitat structure influences breeding birds far more than vegetation species composition (Rotenberry 1985, Fleishman et al. 2003), similar to conclusions drawn for Botteri's sparrow (*Aimophila botterii*) in southern Arizona, USA, grasslands (Jones and Bock 2005).

Systematic searches of fields with territories by lines of people shoulder-to-shoulder discovered 60+ nests in recent years, a 5-fold increase over the first several years when nest searchers were spaced more widely. Apparently brooding females tended to sit tight on nests and tolerated people walking by within several meters. The high GRSP nest success we observed (low predation rates) compares favorably with other accounts (Martin 1992, 1993, Martin et al. 2000). Although nest parasitism by brown-headed cowbirds has been shown to significantly affect grassland bird reproduction in other studies (Johnson and Temple 1990, Smith 1996), we did not detect any evidence of nest parasitism by cowbirds on our study area. Nesting success was tightly correlated with local population densities, so census counts of territorial adults served as an excellent indicator of habitat quality as measured by reproductive success (Van Horne 1983, Vickery et al. 1992).

While at least 20 species of rare grassland-obligate bird species might be attracted to breed in restored grasslands in the mid-Atlantic region, we have succeeded in sustaining 4 (horned lark, grasshopper sparrow, field sparrow, and northern bobwhite quail) in abundance and 2 others in low numbers (dickcissel and vesper sparrow). Most other grassland-obligate species, however, have not yet appeared in our restored CRP grasslands study area. Desired grassland bird species such as Henslow's sparrows (*Ammodramus henslowii*) and eastern meadowlarks (*Sturnella magna*) were expected but not detected on the study area, although local habitat conditions appear suitable for these species (Boone and Dowell 1996). We suspect the lack of Henslow's sparrows on the study area may be due to the absence of a nearby population source. Breeding pairs of meadowlarks have been detected within a few kilometers of the study area, and possible reasons for their absence at Chino Farms includes the lack of tall singing perches. Tall fence posts have been installed recently in several locations on the study area to test this hypothesis.

The stunning GRSP response to our CP2 grasslands illustrates several key points: 1) some rare and at-risk species are quickly attracted to newly established habitat, 2) these breeding birds and their young returned every year to the study area at rates nearly double previous estimates for most passerines (Gill 1990, Wells 1997), 3) high return rates imply annual survival rates at least double previous estimates (if unseen individuals that disperse to other breeding sites are included, estimates of annual survival are even greater), and 4) GRSPs are overwhelmingly breeding-site faithful on a micro-spatial scale, once they become established breeders. In our study area, abundance and density are positive indicators of population success (Whitmore 1981, Van Horne 1983, Vickery et al. 1992, 1994, 2000).

Management Implications

Our integrated management protocols (mowing during establishment, replanting, prescribed burning on a 3-year cycle, and application of herbicides for noxious weed control) were necessary and effective for establishing and maintaining the study area as a species-rich herbaceous habitat on the mid-Atlantic coast that is potentially valuable to targeted grassland-obligate rare species. Prescribed burning is the least expensive and most effective management tool for maintaining grasslands on a large scale (Blair 1997). A temporary challenge to this goal was caused by inclusion of switchgrass in the original planting mixture, which resulted in this species crowding out most other plant species. We successfully removed switchgrass at considerable effort and expense. Our experience with switchgrass during this study, along with the experience of others, has led area agronomists and natural resource specialists to exclude switchgrass from, or greatly reduce the seeding rate in, recommended native grassland seeding mixtures.

No one protocol works for all targeted species of concern, necessitating compromises in management strategies. The speed of encroachment by aggressive second-growth woody plant species was faster than we anticipated. Accordingly, a 2-year prescribed burning rotation may be more effective at maintaining grassland habitat than the 3-year cycle used in this study. Timing the fire to when sap flows in emerging woody stems can be very effective (E. Miles, personal communication). More frequent burning would likely increase management costs. In contrast to the low cost of controlling exotic invasive species, native successional woody species were responsible for the majority of our vegetation maintenance costs.

Determining desirable versus undesirable species in restoration projects is a complex issue. Vegetation management decisions depend strongly on defined stewardship goals. In our study area, we attempted to reconstruct native mid-Atlantic grasslands because of that habitat's extreme rarity. This objective required discouraging secondary succession by preventing invasion by woody species, whether native or exotic. Elsewhere on Chino Farms, studies on restoring other habitat types, including later successional stages, are underway. In our study area, mowing during the first 2 years of grassland establishment and the use of prescribed burning and selective herbicide treatments were successful management tools for our habitat goals. Periodic prescribed grazing by bison (*Bison bison*) or cattle as a simulated natural animal-caused disturbance is under consideration for future treatments.

Our experience with native grassland restoration and management at Chino Farms illustrates the capacity for grassland-nesting birds, particularly GRSPs, to respond to the presence of suitable grassland habitat in the largely agricultural landscape matrix of eastern Maryland. The high GRSP reproduction and survival rates we observed on the study area and the sighting of banded GRSPs off-site indicate that the 92-ha grasslands established are now serving as a population source for this priority species in the local area. The results of this study demonstrate that conservation efforts supported by CRP and CREP, combined with proper grassland establishment and management practices, have great potential to cumulate in diverse plant communities and benefit high-priority, grassland-obligate bird species.

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Literature Cited

Askins, R. A. 1997. History of grasslands in the Northeast. Pages 15–23 *in* P. D. Vickery and P. W. Dunwiddie, editors. Grasslands of northeastern North America. Massachusetts Audubon Society, Lincoln, USA.

Blair, J. M. 1997. Fire, N availability, and plant response in grasslands: a test of the transient maxima hypothesis. Ecology 78:2359–2368.

Blank, P. J., and D. E. Gill. 2006. Bird use of filter strip buffers on the Eastern Shore of Maryland. Report to the Maryland Department of Natural Resources, University of Maryland, College Park, USA.

Blank, P. J., B. Lohr, and D. E. Gill. 2003. Restoration of grassland birds at the Chester River Field Research Center. I. The grasshopper sparrow. Report to the Sears Foundation, Chestertown, Maryland, USA.

Boone, D. D., and B. A. Dowell. 1996. Henslow's sparrow (*Ammodramus henslowii*). Pages 400–401 *in* C. S. Robbins and E. A. T. Bloom, editors. Atlas of breeding birds of Maryland and the District of Columbia. University of Pittsburgh, Pittsburgh, Pennsylvania, USA.

Brennnan, L. A., and W. P. Kuvlesky Jr. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1–13.

Cronon, W. 2003. Changes in the land. Indians, colonists, and the ecology of New England. Hill and Wang, New York, New York, USA.

Dechant, J. A., M. L. Sondreal, D. H. Johnson, L. D. Igl, C. M. Goldada, M. P. Nenneman, and B. R. Euliss. 2003. Effects of management practices on grassland birds: grasshopper sparrow. Northern Prairie Wildlife Research Center, Jamestown, North Dakota, USA. <http://www.npwrc.usgs.gov/resource/literatr/grasbird/grasshop/grasshop.htm>. Accessed 2006 Jan 20.

Delany, M. F., C. T. Moore, and D. R. Progulske. 1993. Survival and longevity of adult male Florida grasshopper sparrows. Proceedings of the Annual Conference of Southeast Fish and Wildlife Agencies 47: 366–399.

Delcourt, H. R., and P. A. Delcourt. 1997. Pre-Columbian Native American use of fire on southern Appalachian landscapes. Conservation Biology 11:1010–1014.

Dinsmore, J. J. 1994. A country so full of game: the story of wildlife in Iowa. University of Iowa, Iowa City, USA.

Fleishman, E., N. Mcdonal, R. MacNally, D. D. Murphy, J. Walters, and T. Floyd. 2003. Effects of floristics, physiognomy and non-native vegetation on riparian bird communities in a Mojave Desert watershed. Journal of Animal Ecology 72:484–490.

Foster, D., G. Motzkin, J. O'Keefe, E. Boose, D. Orwig, J. Fuller, and B. Hall. 2004. The environmental and human history of New England. Pages 43–100 *in* D. R. Foster and J. D. Aber, editors. Forests in time. Yale University, New Haven, Connecticut, USA.

Foster, D. R., B. Hall, S. Barry, S. Clayden, and T. Parshall. 2002. Cultural, environmental, and historical controls of vegetation patterns and the modern conservation setting on the island of Martha's Vineyard, USA. Journal of Biogeography 29:1381–1400.

Gill, F. B. 1990. Ornithology. Second edition. Freeman, New York, New York, USA.

Gleason, H. A. 1913. The relationship of forest distribution and prairie fires in the Middle West. Torreya 13:173–181.

Gustafson, M. E., J. Hildenbrand, and L. Metras. 1997. The North American bird banding manual (electronic version). Version 1.0. Patuxent Wildlife Research Center, Laurel, Maryland, USA. <http://www.pwrc.usgs.gov/BBL/manual/manual.htm>. Accessed 2006 May 1.

Haufler, J. B., editor. 2005. Fish and wildlife benefits of Farm Bill conservation programs: 2000–2005 update. The Wildlife Society Technical Review 05-2, Bethesda, Maryland, USA.

Herkert, J. R. 1994*a*. Breeding bird communities of midwestern prairie fragments—the effects of prescribed burning and habitat-area. Natural Areas Journal 14:128–135.

Herkert, J. R. 1994*b*. The effects of habitat fragmentation on midwestern grassland bird communities. Ecological Applications 4: 461–471.

Holmes, D. W. 1996. Grasshopper sparrow (*Ammodramus savannarum*). Pages 398–399 *in* C. S. Robbins and E. A. T. Bloom, editors. Atlas of breeding birds of Maryland and the District of Columbia. University of Pittsburgh, Pittsburgh, Pennsylvania, USA.

Johnson, D. H., and L. D. Igl. 2001. Area requirements of grassland birds: a regional perspective. Auk 118:24–34.

Johnson, D. H., and M. D. Schwartz. 1993*a*. The Conservation Reserve Program and grassland birds. Conservation Biology 7:934–937.

Johnson, D. H., and M. D. Schwartz. 1993*b*. The Conservation Reserve Program: habitat for grassland birds. Great Plains Research 3:273–295.

Johnson, R. G., and S. A. Temple. 1986. Assessing habitat quality for birds nesting in fragmented tallgrass prairies. Pages 245–249 *in* J. Verner, M. L. Morrison, and C. J. Ralph, editors. Wildlife 2000: modeling habitat relationships of terrestrial vertebrates. University of Wisconsin, Madison, USA.

Johnson, R. G., and S. A. Temple. 1990. Nest predation and brood parasitism of tallgrass prairie birds. Journal of Wildlife Management 54:106–111.

Jones, Z. F., and C. E. Bock. 2005. The Botteri's sparrow and exotic Arizona grasslands: an ecological trap or habitat regained? Condor 107:731–742.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schoeder, W. M. Vander Haegen, and C. Van Riper III. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611–634.

Krech, S., III 1999. The ecological Indian. W. W. Norton, New York, New York, USA.

Kulikoff, A. 1986. Tobacco and slaves: the development of southern cultures in the Chesapeake, 1680–1800. University of North Carolina, Chapel Hill, USA.

Longino, J. T., J. Coddington, and R. K. Colwell. 2002. The ant fauna of a tropical rainforest: estimating species richness three different ways. Ecology 83:689–702.

Madden, E. M., A. J. Hansen, and R. K. Murphy. 1999. Influence of prescribed fire history on habitat and abundance of passerine birds in northern mixed-grass prairie. Canadian Field-Naturalist 113:627–640.

Martin, T. E. 1992. Breeding productivity considerations: what are the appropriate habitat features for management? Pages 455–473 *in* J. M. Hagan and D. W. Johnston, editors. Ecology and conservation of Neotropical migrants. Smithsonian Institution, Washington, D.C., USA.

Martin, T. E. 1993. Nest predation and nest sites: new perspectives on old patterns. BioScience 43:523–532.

Martin, T. E., J. Scott, and C. Menge. 2000. Nest predation increases with parental activity: separating nest site and parental activity effects.

Proceedings of the Royal Society of London, Series B 267:2287–2293.
McCoy, T. D., M. R. Ryan, E. W. Kurzejeski, and L. W. Burger. 1999. Conservation Reserve Program: source or sink habitat for grassland birds in Missouri. Journal of Wildlife Management 63:530–537.
McGuire, B. 2003. Conservation and the new Farm Bill. Missouri Conservationist 64:12–16.
Mehrhoff, L. J. 1997. Thoughts on the biogeography of grassland plants in New England. Pages 15–23 *in* P. D. Vickery and P. W. Dunwiddie, editors. Grasslands of northeastern North America. Massachusetts Audubon Society, Lincoln, USA.
Mitchell, L. R., C. R. Smith, and R. A. Maleki. 2000. Ecology of grassland breeding birds in the northeastern United States—a literature review with recommendations for management. U.S. Geological Survey, Biological Resources Division, New York Cooperative Fish and Wildlife Research Unit, Department of Natural Resources, Cornell University, Ithaca, New York, USA.
Noss, R. F., E. T. La Roe, and J. M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. U.S. Department of Interior, Washington, D.C., USA.
Patterson, W. A. I., and K. E. Sassaman. 1988. Indian fires in the prehistory of New England. Pages 107–135 *in* G. P. Nichols, editor. Holocene human ecology in northeastern North America. Plenum, New York, New York, USA.
Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American breeding bird survey, 1966–1996. Studies in Avian Biology 19:27–44.
Reynolds, R. E., T. L. Shaffer, J. R. Sauer, and B. G. Peterjohn. 1994. Conservation Reserve Program: benefit for grassland birds in the northern plains. Transactions of the North American Wildlife and Natural Resources Conference 59:328–336.
Rotenberry, J. T. 1985. The role of habitat in avian community composition: physiognomy or floristics? Oecologia 67:213–217.
Samson, F., and F. Knopf. 1994. Prairie conservation in North America. BioScience 44:418–421.
Sauer, J. R., J. E. Hines, and J. Fallon. 2005. The North American Breeding Bird Survey, results and analysis 1966–2004. Version 2005.2. U.S. Geological Survey, Patuxent Wildlife Research Center, Laurel, Maryland, USA.
Schwartzman, E., L. A. Zamora, P. J. Blank, and D. E. Gill. 2002. Grassland restoration on the eastern shore of Maryland: warm-season grass establishment and plant community development at the Chester River Field Research Center, MD 1999–2002. Report to the Sears Foundation, Chestertown, Maryland, USA.
Smith, R. L. 1968*a*. Some ecological notes on the grasshopper sparrow. Wilson Bulletin 75:159–165.
Smith, R. L. 1968*b*. Grasshopper sparrow. Pages 725–745 *in* O. L. Austin Jr., editor. Life histories of North American cardinals, grosbeaks, buntings, towhees, finches, sparrows, and allies. U.S. National Museum Bulletin 237, Washington, D.C., USA.
Smith, S. A. 1996. Dickcissel (*Spiza americana*). Pages 386–387 *in* C. S. Robbins and E. A. T. Bloom, editors. Atlas of breeding birds of Maryland and the District of Columbia. University of Pittsburgh, Pittsburgh, Pennsylvania, USA.
Tyndall, R. W. 1992. Historical considerations of conifer expansion in Maryland serpentine "barrens". Castanea 57:123–131.
USDA Farm Service Agency. 2006. Farm Service Agency home page. <http://www.fsa.usda.gov/dafp/cepd/crep.htm>. Accessed 2006 Jan 20.
Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47:893–901.
Vickery, P. D., J. R. Herkert, F. L. Knopf, J. Ruth, and C. E. Keller. 2000. Grassland birds: an overview of threats and recommended management strategies. Version 06MAR2000. <http://www.birds.cornell.edu/pifcapaemay/vickery.htm>. Accessed 2006 Jan 20.
Vickery, P. D., M. L. Hunter, and S. M. Melvin. 1994. Effects of habitat area on the distribution of grassland birds in Maine. Conservation Biology 8:1087–1097.
Vickery, P. D., M. L. Hunter Jr., and J. V. Wells. 1992. Is density an indicator of breeding success? Auk 109:706–710.
Wells, J. V. 1997. Population viability analysis for Maine grasshopper sparrows. Pages 15–23 *in* P. D. Vickery and P. W. Dunwiddie, editors. Grasslands of northeastern North America. Massachusetts Audubon Society, Lincoln, USA.
Whitmore, R. C. 1981. Structural characteristics of grasshopper sparrow habitat. Journal of Wildlife Management 45:811–814.
Wiens, J. A. 1969. An approach to the study of ecological relationships among grassland birds. Ornithological Monographs 8.
Winter, M., S. E. Hawks, J. A. Shaffer, and D. H. Johnson. 2003. Guidelines for finding nests of passerine birds in tallgrass prairie. Prairie Naturalist 35:197–211.

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