Forest Ecology and Management 261 (2011) 10–18

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Effects of fuel reduction on birds in pitch pine–scrub oak barrens of the United States

David I. King[a,*], Scott Schlossberg[b], Robert T. Brooks[a], Michael E. Akresh[b]

[a] Northern Research Station, USDA Forest Service, 201 Holdsworth Hall, University of Massachusetts, Amherst, MA 01003, USA
[b] Department of Environmental Conservation, University of Massachusetts, 201 Holdsworth Hall, Amherst, MA 01003, USA

ARTICLE INFO

Article history:
Received 28 May 2010
Received in revised form 19 August 2010
Accepted 20 August 2010

Keywords:
Avian
Fire
Forest
Risk
Shrub
Thinning

ABSTRACT

Fire-dependent ecosystems include some of the most threatened ecosystems in the world, and where fuels are allowed to accumulate, they can present significant threats to human life and property. Fuel reduction activities can be effective in reducing the risk of wildfire, but these practices need to be evaluated relative to their effect on biodiversity. We surveyed birds in an inland pitch pine–scrub oak barren, a fire-dependent plant community, in which fuel reduction had been carried out via thinning of canopy trees to reduce the risk of running crown fires. We hypothesized that thinning pitch pine forest would negatively affect the abundance of mature forest birds and positively affect the abundance of scrub–shrub birds. Our results confirmed these expectations: several mature forest bird species were less abundant in thinned pitch pine than unthinned pitch pine, although most of these species were also present in mixed deciduous forest, and therefore regionally well represented. In contrast, another group of bird species was scarce or absent from unthinned pitch pine and mixed deciduous forest, but present in thinned sites and scrub oak stands. These were scrub–shrub species that do not nest in mixed deciduous or pitch pine forest but depend on shrubland or savannah habitats that cover ~3% of the region. We conclude that fuel reduction by thinning canopy trees at this site provides habitat for high-priority scrub–shrub bird species at the cost of modest reductions in numbers of forest birds whose regional aggregate population is large.

Published by Elsevier B.V.

1. Introduction

Fire-dependent plant communities have unique structural and floristic characteristics that evolved in the presence of periodic fire, which encourages the regeneration of fire adapted species and discourages competition from fire-intolerant species (Nowacki and Abrams, 2008). Suppression of wildfires has decreased the representation of fire-dependent ecosystems worldwide, including in Mediterranean pine forests (Moreira et al., 2003), South African fynbos (Manders and Richardson, 1992), Australian eucalypt forest (Penman et al., 2007; Burrows, 2008) and temperate pine forests of North America (Kalies et al., 2010). In addition to its negative effects on ecosystems, fire exclusion, when not accompanied by mitigating fuel treatments, permits the accumulation of fuels to levels that increase the risk of catastrophic wildfire (USDA, 2000; Duveneck and Patterson, 2007; Penman et al., 2007; Burrows, 2008; Kalies et al., 2010).

Pitch pine–scrub oak (*Pinus rigida-Quercus ilicifolia*) barrens occur within the Atlantic Coastal Barrens ecoregion that covers nearly 9000 km^2 in the northeastern U.S. (Ricketts et al., 1999). Pitch pine–scrub oak barrens are considered to be of high conservation concern in the Northeast (Swain and Kearsley, 2001), and barrens systems in general are among the most threatened ecosystems in North America (Noss and Peters, 1995). Pitch pine–scrub oak barrens are dominated by flammable species with adaptations to survive and regenerate after wildfire (Motzkin et al., 1999), and which comprise some of the most dangerous fuels in the region (Duveneck and Patterson, 2007).

In the absence of an active fire-management program, fuels in fire prone systems accumulate to hazardous levels that support extreme fire behavior and pose significant risks to human health and property (Duveneck and Patterson, 2007; Moreira et al., 2003; Penman et al., 2007; Burrows, 2008). Pitch pine forests are susceptible to crown fires that can spread with such rapidity as to be uncontrollable until they run out of fuel (Clark and Patterson, 2003). One such fire in Massachusetts consumed nearly 6000 ha of forest in 24 h in 1957 (Clark and Patterson, 2003). Pitch pine–scrub oak barrens encompass substantial portions of some of the most densely populated regions on the continent (U.S. Census Bureau, 2009). As a result, residential developments in this region are often adjacent to or embedded within pyrogenic plant communities such as pitch pine–scrub oak barrens, which increases the risk of human-caused

* Corresponding author. Tel.: +1 413 545 6795; fax: +1 413 545 1860.
E-mail address: dking@fs.fed.us (D.I. King).

0378-1127/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.foreco.2010.08.039

ignitions, as well as the potential consequences of uncontrollable wildfire.

Fuel reduction in pitch pine consists of reducing canopy coverage by thinning overstory trees to reduce the potential for running crown fires (Duveneck and Patterson, 2007). The alteration of habitat structure and floristic composition in the course of fuels management can be anticipated to affect birds (Moreira et al., 2003; Greenberg et al., 2007; Seavy et al., 2008), including high-priority disturbance-dependent species (Gifford et al., 2009). Managers and conservationists are obliged to consider the effects of management activities on native species, so an understanding of how management affects birds is essential. Here we characterize the avifauna of an inland pitch pine–scrub oak barren, describe the effects of fuels management on bird abundance and species composition, and discuss our findings relative to regional bird conservation. We hypothesized that thinning pitch pine forest would negatively affect the abundance of mature forest birds and positively affect the abundance of scrub–shrub birds.

2. Methods

2.1. Study area

The study took place on a 595 ha pitch pine–scrub oak barren within the Montague Plains Wildlife Management Area in western Massachusetts, U.S.A. This site is located on a sand delta formed more than 10,000 years ago when meltwater streams from retreating glaciers emptied into a large lake that has since drained. The site is characterized by well drained soils and dominant plant species that are highly flammable and adapted to regenerating after fire (Motzkin et al., 1999). Habitats included pitch pine forest (∼178 ha) and scrub oak barrens (∼36 ha) (Clark and Patterson, 2003). Mixed pitch pine and tree oaks cover most of the remaining area. Starting in 2004, the Massachusetts Division of Fisheries and Wildlife initiated a program to reduce fuel loads and fire risk to nearby towns. Fuel reduction consisted of mechanical thinning of dominant trees in pitch pine forest to a target canopy coverage of <50%, as well as thinning understory trees and saplings with either mechanical means or prescribed fire (Clark and Patterson, 2003). Overstory trees were removed from the site, and slash and understory trees were either chipped and left on site or stacked for later burning. By the conclusion of the study ∼50% of the total area of pitch pine forest had been thinned.

2.2. Study design

Bird survey points were located at a subset of points originally established by Motzkin et al. (1999) to enable the linkage between our study and this extensive long-term database. We selected points located ≥250 m apart to allow statistical independence among samples (Ralph et al., 1995). Sample sizes represented the maximum number of points possible given the area of each habitat and the between-point distance constraint. Although thinned areas were chosen based on fire risk to neighboring communities and not randomly, thinned sites for which we had pre-treatment data were similar in tree density, canopy height and understory density to unthinned sites ($P > 0.11$), suggesting that thinned sites were generally representative of this habitat type.

Survey points were allocated among the following habitats, which represent the most common plant communities at the Montague Plains; pitch pine forest ($n = 40$), thinned pitch pine ($n = 11$), scrub oak barrens ($n = 16$), and mixed deciduous forest ($n = 13$). Unthinned pitch pine forest served as a reference by which to gauge the effects of thinning activities (Fig. 1a) and varied in composition from stands entirely dominated by pitch pine to stands with varying amounts of hardwoods such as tree oaks (*Quercus* spp.), red maple (*Acer rubrum*) and gray birch (*Betula populifolia*) as well as white pine (*Pinus strobus*). The shrub layer consisted of scrub oaks (*Q. ilicifolia* and *Q. prinoides*), pin cherry (*Prunus pensylvanica*), huckleberry (*Gaylussacia baccata*), blueberry (*Vaccinium* spp.), and oak and pine saplings.

Thinned pitch pine (Fig. 1b) consisted of pitch pine forest which had been thinned as described above. The remaining canopy consisted of mature pitch pines with scattered tree oaks. The understory consisted of scrub oaks, blueberry, and gray birch. Thinning occurred during the non-growing seasons of 2004, 2006 and 2007, so each of these years, several points were subtracted from the pitch pine sample and added to the treated pitch pine sample (3, 6 and 11 in 2004, 2006 and 2007, respectively). Three of these points had been treated prior to the bird surveys, but we had both pre- and post-treatment data for the remaining 8 points in thinned pitch pine.

In addition, we surveyed scrub oak barrens because shrublands represent a habitat of high conservation priority in eastern North America (Askins, 2000) and we wanted to use this habitat as a reference to evaluate the effectiveness of pitch pine treatments for accommodating the birds typical of this habitat type. Scrub oak barrens consisted of dense stands of scrub oaks generally <2 m tall with scattered pitch pine and tree oaks, as well as pin cherry, shadbush (*Amelanchier* spp.), huckleberry and blueberry (Fig. 1c).

Finally, we included mixed deciduous forest, because it represents the late-successional type which can develop on barrens in the absence of disturbance (Kerlinger and Doremus, 1981), and also to determine whether the bird communities of the various sandplain habitats were distinct from this widely distributed forest type. Mixed deciduous forest included both the deciduous and mixed deciduous types of Clark and Patterson (2003), and was dominated by tree oaks, red maple, and hickories (*Carya* spp.), as well as conifers including pitch pine, white pine and eastern hemlock (*Tsuga canadensis*) (Fig. 1d). The shrub layer was generally sparse, and consisted of regenerating trees of the canopy species as well as shadbush, black cherry (*Prunus serotina*), huckleberry, and mountain laurel (*Kalmia angustifolia*).

2.3. Vegetation measurements

At each bird survey location, we measured vegetation characteristics at 20 randomly selected points located within each 50-m point count radius. At each of the 20 sampling points we measured the presence and height of the overstory canopy, the height of the understory vegetation (i.e. vegetation ≤ 3 m tall), the plant species, or in the absence of vegetation, the type of ground cover (e.g. litter, slash, bare ground, etc.), and the number of contacts of vegetation with a 3-m pole held vertically divided into 1-m height intervals. We also counted all trees by species within a variable-circular plot using 10-factor metric cruising prism. These values were averaged for each sample point, and log-transformed to improve normality and equality of variances.

2.4. Bird surveys

We evaluated bird abundance during 2004–2008 with 10-min, 50-m radius point counts (Ralph et al., 1995). Each point was visited 3 times between 0530 and 1100 h between June 1 and July 15, and observers recorded the number of individuals of each species that were detected.

2.5. Statistical analyses

Because raw point counts may not provide a reliable index of bird abundance (Thompson, 2002) we estimated bird density

Fig. 1. Habitats sampled at the Montague Plains Wildlife Management Area, Franklin Co., Massachusetts: unthinned pitch pine (a.), thinned pitch pine (b.), scrub oak (c.), mixed deciduous (d.).

from point-count data while correcting for bias due to heterogeneity in detection probabilities using *N*-mixture models (Royle, 2004). We modeled abundance in relation to habitat type, and detectibility in relation to time of day, date and observer because these factors may bias count data (Johnson, 2008). We chose a negative binomial or Poisson distribution for the abundance of each species based on Akaike's information criterion (AIC). Year was included in both the abundance and detectability components of the model (Kery, 2008). We also included a quadratic term for date because detectability can rise and fall across the breeding season. Initially, we screened each variable alone to see if it improved the AIC over the null model, and variables that did were retained. We then ran models with all possible combinations of the retained variables. Models with ΔAIC values $\leq$2 were considered supported. We calculated 95% confidence intervals of parameters using the model-averaged coefficients and compared avian abundances among habitats based on these intervals.

Because we had pre-treatment survey data from points within pitch pine forest we were able to strengthen our inference that the changes in bird abundance we observed in treated areas were in fact due to the treatments. We did not have sufficient data to model abundance with *N*-mixture models, so we plotted mean point count values of three focal scrub scrub–shrub bird species for which we a sufficient sample size as a function of time since treatment in both thinned and unthinned areas. We calculated 95% confidence intervals and compared avian abundances among habitats based on these intervals. Because we did not adjust these point count values for detectibility, we concede that the results of these analyses could be influenced by observer effects. Since survey points in thinned pitch pine were added over several years, the thinned pitch pine sample included points ranging in age from 0–5 growing seasons; however, since our chief interest was the comparison of thinned pitch pine with other habitats, the inclusion of different aged stands should not affect our conclusions.

For species that were not abundant enough to analyze with *N*-mixture models, we used the maximum count for each species in each year, averaged them for each point over all years, and then compared abundance among treatments using generalized linear models with a log link. We used linear contrasts to examine differences in abundance among habitats. Because of the large number of statistical tests (one for each species) we used a Bonferroni-corrected *P*-value for these comparisons to hold the overall probability of Type I error at 5%. This analysis was restricted to species detected at >3 survey points.

We compared habitat characteristics between treatments using Kruskal–Wallis tests with Bonferroni-corrected *P*-values. We used canonical correspondence analysis (CCA) to assess multivariate relationships between bird species and habitat measurements (Ter Braak, 1986). Vegetation variables ordinated were counts of all trees combined, coniferous and deciduous trees separately, vegetation contacts 0–1 m and >1–2 m above ground (contacts >2–3 m were highly correlated with overstory height), bare ground/litter cover, and height of overstory and understory vegetation. For the bird data, we used the maximum of the three counts at each point. Species occurring at $\geq$ 10% of points were included. Point count data were standardized by maxima and site values were standardized by totals to emphasize relative differences due to habitat selection rather than differences due to abundance.

Table 1
Habitat characteristics compared among 4 habitats with Kruskal–Wallis tests at the Montague Plains WMA, 2004–2008. Common letters denote means that are not statistically different.

Habitat	MD[a]	PP	SO	TP	H[a]	P
Tree cover (%)	100 (0.00)[a]	83.4 (1.69)[a]	21.4 (5.20)[b]	37.5 (6.16)[c]	43.3	<0.001
Pitch pine cover (%)	1.67 (1.67)[a]	51.3 (2.78)[b]	3.93 (1.40)[a]	30.0 (5.48)[c]	39.9	<0.001
Canopy height	27.8 (2.43)[a]	23.3 (0.71)[a]	1.67 (0.19)[b]	22.4 (0.82)[a]	28.7	<0.001
Understory cover (%)	60.0 (10.0)[a]	81.2 (2.22)[b]	96.1 (3.20)[c]	65.8 (7.46)[a]	27.0	<0.001
Structure (0–1 m)	0.95 (0.38)[a]	2.13 (0.15)[b]	4.40 (0.57)[c]	2.80 (0.93)[b]	20.3	<0.001
Structure (>1–2 m)	0.28 (0.03)[a]	0.73 (0.09)[b]	2.55 (0.41)[c]	0.04 (0.03)[d]	25.3	<0.001
Structure (>2–3 m)	0.55 (0.23)[a]	0.98 (0.14)[b]	2.04 (0.46)[c]	0.09 (0.04)[d]	15.0	0.002
Ground cover (%)	40.0 (10.0)[a]	21.8 (2.50)[b]	1.43 (0.82)[c]	27.5 (8.04)[b]	30.9	<0.001

MD, mature deciduous forest; PP, unthinned pitch pine forest; SO, scrub oak barren; TP, thinned pitch pine.

3. Results

3.1. Vegetation characteristics among habitats

Thinned pitch pine forest had lower overall canopy cover, lower canopy cover of pitch pine, lower understory cover, and lower habitat structure between 1–2 m and 2–3 m than unthinned pitch pine (Table 1). Canopy height, structure between 0–1 m and ground cover did not differ between thinned and unthinned pitch pine. Overstory canopy cover was greatest at mixed deciduous sites, and canopy height was significantly lower, and understory cover and vertical structure significantly higher, in scrub oak barrens.

3.2. Bird abundance among habitats

We encountered 60 bird species during the point-count surveys, of which 13 were abundant enough for *N*-mixture models to converge. Best supported models for all species included habitat (Table 2). Ovenbirds (scientific names in Table 3) were more abundant in unthinned pitch pine than all other habitats (Fig. 2). Black-capped chickadee abundance was similar between

Table 2
Results from *N*-mixture models of bird abundance in 4 habitats at the Montague Plains WMA, 2004–2008. Models with ΔAIC ≤ 2 are presented. Scientific names are in Table 3.

Species	Abundance	Detectability	AIC	K	ΔAIC	w_i[a]	Model
Black-capped chickadee	Habitat		1003.8	5	0.00	1.00	Poisson
Hermit thrush	Habitat	Year + Date + Time	474.6	11	0.00	0.24	Neg. Binomial
	Habitat	Year + Date	475.3	10	0.71	0.17	Neg. Binomial
	Habitat	Year + Date^2 + Time	475.5	12	0.93	0.15	Neg. Binomial
	Habitat	Year + Date^2	476.1	11	1.47	0.11	Neg. Binomial
American robin	Habitat	Date^2	349.2	8	0.00	0.60	Neg. Binomial
	Habitat	Date	350.1	7	0.92	0.38	Neg. Binomial
Gray catbird	Habitat	Year	380.5	8	0.00	0.65	Poisson
Chestnut-sided Warbler	Habitat	Date	247.4	7	0.00	0.50	Neg. Binomial
	Habitat		247.4	6	0.03	0.50	Neg. Binomial
Pine warbler	Habitat	Year + Date^2	1073.7	11	0.00	0.46	Neg. Binomial
	Habitat	Year + Date	1075.4	10	1.73	0.19	Neg. Binomial
Prairie warbler	Habitat	Year	686.6	8	0.00	0.75	Poisson
Black-and-white warbler	Habitat	Date^2 + Time	695.5	9	0.00	0.23	Neg. Binomial
	Habitat	Date + Time	695.7	8	0.19	0.21	Neg. Binomial
	Habitat	Time	696.1	7	0.60	0.17	Neg. Binomial
	Habitat	Date^2	696.1	8	0.62	0.17	Neg. Binomial
	Habitat	Date	696.9	7	1.38	0.12	Neg. Binomial
	Habitat		697.5	6	1.99	0.09	Neg. Binomial
Ovenbird	Habitat	Year + Date	1024.9	9	0.00	0.28	Poisson
	Habitat	Year	1025.0	8	0.05	0.27	Poisson
	Habitat	Year + Date^2	1025.5	10	0.60	0.20	Poisson
Common yellowthroat	Habitat + Year	Date	423.7	10	0.00	0.64	Poisson
Eastern towhee	Habitat + Year	Date	972.3	10	0.00	0.25	Poisson
	Habitat	Year + Date	972.8	9	0.47	0.20	Poisson
	Habitat + Year	Date^2	973.2	11	0.84	0.17	Poisson
	Habitat	Year + Date^2	973.6	10	1.27	0.13	Poisson
Chipping sparrow	Habitat + Year	Date^2 + Time	959.0	13	0.00	0.17	Neg. Binomial
	Habitat + Year	Time	959.4	11	0.42	0.14	Neg. Binomial
	Habitat	Date^2 + Time	959.7	9	0.72	0.12	Neg. Binomial
	Habitat + Year	Date + Time	960.4	12	1.41	0.09	Neg. Binomial
	Habitat	Time	960.5	7	1.51	0.08	Neg. Binomial
	Habitat + Year	Date^2	960.6	12	1.62	0.08	Neg. Binomial
	Habitat + Year		960.7	10	1.70	0.07	Neg. Binomial
	Habitat	Date^2	961.0	8	1.94	0.07	Neg. Binomial
Field sparrow	Habitat + Year		398.4	10	0.00	0.51	Neg. Binomial
	Habitat	Year	398.5	9	0.13	0.48	Neg. Binomial

[a] AIC weights.

Table 3
Average counts and standard errors for birds compared among 4 habitats with generalized linear models and post-hoc contrasts at the Montague Plains WMA, 2004–2008. Significant differences among habitats (Bonferroni-corrected) are in bold type. Common letter subscripts denote means that are not statistically different. Habitat codes as in Table 1.

Species	MD	PP	SO	TP	χ^2	*P*
Ruffed grouse (*Bonasa umbellus*)	0.07 (0.05)	0.01 (0.01)	0.00 (0.00)	0.00 (0.00)	13.4	0.004
Mourning dove (*Zenaida macroura*)	0.04 (0.04)	0.08 (0.02)	0.14 (0.04)	0.21 (0.09)	6.72	0.08
Downy woodpecker (*Picoides pubescens*)	0.22 (0.08)	0.01 (0.01)	0.08 (0.03)	0.03 (0.03)	7.67	0.005
Hairy woodpecker (*Picoides villosus*)	**0.30 (0.10)**^{a}	**0.04 (0.02)**^{b}	**0.02 (0.02)**^{b}	**0.07 (0.05)**^{ab}	**17.4**	**<0.001**
Eastern wood-Pewee (*Contopus virens*)	**0.48 (0.10)**^{a}	**0.01 (0.01)**^{b}	**0.02 (0.02)**^{b}	**0.21 (0.08)**^{a}	**41.0**	**<0.001**
Great-crested flycatcher (*Myiarchus crinitus*)	0.07 (0.05)	0.07 (0.02)	0.05 (0.03)	0.07 (0.05)	0.38	0.95
Blue-headed vireo (*Vireo solitarius*)	0.07 (0.07)	0.05 (0.02)	0.00 (0.00)	0.07 (0.05)	6.17	0.10
Red-eyed vireo (*Vireo olivaceus*)	**1.04 (0.10)**^{a}	**0.07 (0.02)**$^{\mathbf{b}}$	**0.16 (0.05)**^{c}	**0.00 (0.00)**^{b}	**67.8**	**<0.001**
Blue jay (*Cyanocitta cristata*)	0.30 (0.12)	0.28 (0.04)	0.10 (0.04)	0.03 (0.03)	11.8	0.008
American crow (*Corvus brachyrhynchos*)	0.00 (0.00)	0.03 (0.01)	0.00 (0.00)	0.03 (0.03)	3.59	0.31
Black-capped chickadee (*Poecile atricapillus*)	**0.30 (0.09)**^{a}	**0.82 (0.05)**^{b}	**0.60 (0.09)**^{bc}	**0.31 (0.09)**^{ac}	**16.3**	**<0.001**
Tufted titmouse (*Baeolophus bicolor*)	0.26 (0.09)	0.12 (0.03)	0.05 (0.03)	0.00 (0.00)	12.87	0.006
Red-breasted nuthatch (*Sitta canadensis*)	**0.00 (0.00)**^{a}	**0.47 (0.05)**^{b}	**0.08 (0.03)**^{a}	**0.34 (0.10)**$^{\mathbf{b}}$	**36.6**	**<0.001**
White-breasted nuthatch (*Sitta carolinensis*)	0.26 (0.10)	0.11 (0.03)	0.05 (0.03)	0.24 (0.09)	10.4	0.016
Brown Creeper (*Certhia americana*)	0.04 (0.04)	0.18 (0.03)	0.05 (0.03)	0.14 (0.07)	9.12	0.028
Veery (*Catharus fuscescens*)	0.07 (0.05)	0.09 (0.02)	0.14 (0.05)	0.00 (0.00)	5.86	0.12
Hermit thrush (*Catharus guttatus*)	**0.07 (0.05)**^{a}	**0.45 (0.05)**^{b}	**0.06 (0.03)**^{a}	**0.10 (0.06)**^{a}	**28.1**	**<0.001**
Wood thrush (*Hylocichla mustelina*)	0.07 (0.05)	0.01 (0.01)	0.02 (0.02)	0.00 (0.00)	4.76	0.19
American robin (*Turdus migratorius*)	**0.33 (0.09)**^{a}	**0.10 (0.03)**^{b}	**0.14 (0.04)**^{b}	**0.48 (0.12)**^{a}	**20.2**	**<0.001**
Gray catbird (*Dumetella carolinensis*)	**0.04 (0.04)**^{ac}	**0.05 (0.02)**^{ac}	**0.75 (0.08)**^{b}	**0.14 (0.07)**^{a}	**71.5**	**<0.001**
Brown thrasher (*Toxostoma rufum*)	**0.00 (0.00)**^{a}	**0.01 (0.01)**^{a}	**0.25 (0.06)**$^{\mathbf{b}}$	**0.10 (0.08)**^{b}	**30.2**	**<0.001**
Cedar waxwing (*Bombycilla cedrorum*)	**0.07 (0.07)**$^{\mathbf{a}}$	**0.14 (0.03)**$^{\mathbf{a}}$	**0.43 (0.10)**$^{\mathbf{b}}$	**0.31 (0.12)**$^{\mathbf{b}}$	**17.8**	**<0.001**
Blue-winged warbler (*Vermivora pinus*)	0.00 (0.00)	0.01 (0.01)	0.02 (0.02)	0.00 (0.00)	1.23	0.74
Chestnut-sided warbler (*Dendroica pensylvanica*)	**0.00 (0.00)**^{a}	**0.01 (0.01)**^{a}	**0.52 (0.08)**^{b}	**0.00 (0.00)**^{a}	**76.8**	**<0.001**
Magnolia warbler (*Dendroica magnolia*)	0.00 (0.00)	0.00 (0.00)	0.03 (0.02)	0.00 (0.00)	5.78	0.12
Yellow-rumped warbler (*Dendroica coronata*)	0.00 (0.00)	0.06 (0.02)	0.00 (0.00)	0.03 (0.03)	8.57	0.04
Pine warbler (*Dendroica pinus*)	**0.15 (0.07)**^{ac}	**1.19 (0.06)**^{b}	**0.29 (0.07)**^{c}	**0.97 (0.12)**^{b}	**68.5**	**<0.001**
Prairie warbler (*Dendroica discolor*)	**0.04 (0.04)**^{a}	**0.13 (0.03)**^{a}	**1.25 (0.09)**^{b}	**0.93 (0.17)**^{b}	**132.3**	**<0.001**
Black-and-white warbler (*Mniotilta varia*)	**0.04 (0.04)**^{a}	**0.36 (0.04)**^{b}	**0.94 (0.06)**^{c}	**0.10 (0.06)**^{ab}	**49.4**	**<0.001**
American redstart (*Setophaga ruticilla*)	**0.41 (0.12)**^{a}	**0.01 (0.01)**^{b}	**0.10 (0.04)**^{c}	**0.00 (0.00)**^{b}	**39.3**	**<0.001**
Ovenbird (*Seiurus aurocapilla*)	**0.44 (0.12)**^{a}	**1.24 (0.06)**^{b}	**0.14 (0.04)**^{c}	**0.28 (0.08)**^{b}	**88.5**	**<0.001**
Common yellowthroat (*Geothlypis trichas*)	**0.00 (0.00)**^{a}	**0.06 (0.02)**^{a}	**0.78 (0.08)**^{b}	**0.28 (0.10)**^{c}	**85.5**	**<0.001**
Canada warbler (*Wilsonia Canadensis*)	0.00 (0.00)	0.01 (0.01)	0.00 (0.00)	0.00 (0.00)	1.16	0.76
Eastern towhee (*Pipilo erythrophthalmus*)	**0.07 (0.05)**a	**0.45 (0.05)**b	**1.59 (0.09)c**	**0.72 (0.14)**d	**88.1**	**<0.001**
Chipping sparrow (*Spizella passerina*)	**0.11 (0.06)**^{a}	**0.84 (0.05)**^{b}	**0.35 (0.07)**^{c}	**1.41 (0.11)**^{d}	**45.8**	**<0.001**
Field sparrow (*Spizella pusilla*)	**0.00 (0.00)**^{a}	**0.08 (0.02)**^{a}	**0.52 (0.07)**^{b}	**0.48 (0.12)**^{b}	**53.7**	**<0.001**
Scarlet tanager (*Piranga olivacea*)	**0.52 (0.10)**^{a}	**0.14 (0.03)**^{b}	**0.03 (0.02)**^{c}	**0.00 (0.00)**$^{\mathbf{c}}$	**33.4**	**<0.001**
Northern cardinal (*Cardinalis cardinalis*)	0.00 (0.00)	0.03 (0.01)	0.00 (0.00)	0.00 (0.00)	4.65	0.20
Rose-breasted grosbeak (*Pheucticus ludovicianus*)	0.15 (0.07)	0.01 (0.01)	0.06 (0.04)	0.00 (0.00)	13.4	0.004
Indigo bunting (*Passerina cyanea*)	0.04 (0.04)	0.01 (0.01)	0.00 (0.00)	0.07 (0.05)	6.55	0.09
Baltimore oriole (*Icterus galbula*)	**0.26 (0.09)**^{a}	**0.01 (0.01)**^{b}	**0.08 (0.03)**^{a}	**0.03 (0.03)**^{b}	**21.5**	**<0.001**
Purple finch (*Carpodacus purpureus*)	0.00 (0.00)	0.04 (0.02)	0.00 (0.00)	0.03 (0.03)	3.75	0.290
American goldfinch (*Spinus tristis*)	**0.04 (0.04)**^{a}	**0.06 (0.02)**^{a}	**0.25 (0.06)**^{b}	**0.07 (0.05)**^{a}	**17.4**	**<0.001**

unthinned and thinned pitch pine forest. Chipping sparrow abundance in thinned pitch pine was greater than in mixed deciduous forest; all other pairwise comparisons for chipping sparrows were non-significant. Chestnut-sided warblers and eastern towhees were most abundant in scrub oak. Abundance of common yellowthroats, prairie warblers, gray catbirds and field sparrows did not differ between scrub oak and treated pitch pine. Black-and-white warbler abundance was similar between scrub oak and pitch pine forest, but greater in scrub oak than mixed deciduous forest, and greater in mixed deciduous forest than thinned pitch pine.

Forty-three species were abundant enough to analyze with Poisson regression. The results of this analysis were similar to the results of the *N*-mixture model analyses for species for which both analyses were done (Table 3). Both analyses showed that black-capped chickadees, hermit thrushes and ovenbirds were most abundant in unthinned pitch pine forest, chipping sparrows and pine warblers were abundant in both unthinned and thinned pitch pine, black-and-white warblers, common yellowthroats, chestnut-sided warblers, eastern towhees and gray catbirds were most abundant in scrub oak, and prairie warblers and field sparrows were most abundant in scrub-oak and thinned pitch pine. In addition, the Poisson regression showed that red-breasted nuthatches were abundant in both pitch pine forest and thinned pitch pine, American gold finches were most abundant in scrub oak, and cedar waxwings and brown thrashers were abundant in both scrub oak and treated pitch pine. Finally, hairy woodpeckers, eastern wood-pewees, red-eyed vireos, American redstarts, Baltimore orioles and scarlet tanagers were all most abundant in mixed deciduous forest.

Prairie warblers, field sparrows and eastern towhees were absent or virtually absent from pitch pine forest prior to treatment, but occupied treated areas the year after thinning and increased thereafter, with the exception of prairie warblers, that suggested a slight decline in numbers 4 years post-treatment (Fig. 3).

3.3. Bird habitat relationships

CCA revealed that bird habitat associations could be separated into three groups. One group, including brown creeper, red-breasted nuthatch, pine warbler, black-capped chickadee, ovenbird, and hermit thrush, was associated with conifers and tree cover. A second group, including prairie warbler, field sparrow, American goldfinch, common yellowthroat, and chestnut-sided warbler, was associated with low and medium vegetation heights and lower overstory height. A final group, composed of tufted titmouse, red-eyed vireo, scarlet tanager and American robin, was associated with deciduous trees (Fig. 4).

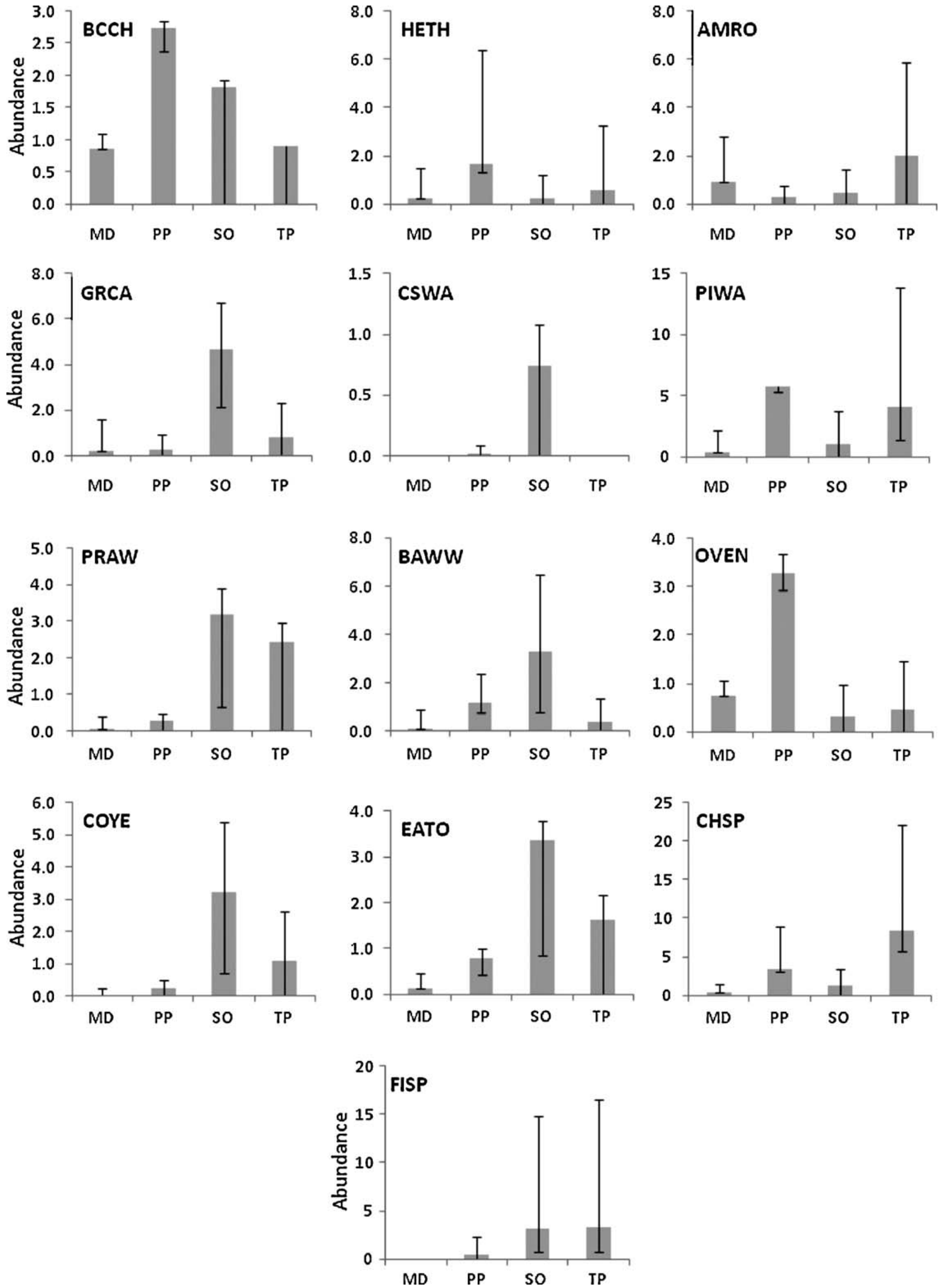

Fig. 2. Bird abundance estimates with 95% CIs from *N*-mixture model analyses of point count data from the Montague Plains WMA, 2004–2008. BCCH, black-capped chickadee; HETH, hermit thrush; AMRO, American robin; GRCA, gray catbird; CSWA, chestnut-sided warbler; PIWA, pine warbler; PRWA, prairie warbler; BAWW, black-and-white warbler; OVEN, ovenbird; COYE, common yellowthroat; EATO, eastern towhee; CHSP, chipping sparrow; FISP, field sparrow. CIs that extended beyond the axis have been truncated.

4. Discussion

4.1. Effects of treatments on fuels

Our results show that fuel reduction by thinning overstory trees caused significant changes in habitat structure. In addition to reducing the canopy coverage in pitch pine forest by half, fuel reduction also reduced the amount of "ladder fuels" in the form of shrubby growth >1 m in height that could transmit flame from the ground to the canopy. These ladder fuels can potentially transform a controllable ground fire into an uncontrollable conflagration (Duveneck and Patterson, 2007). Simulation models based on detailed mea-

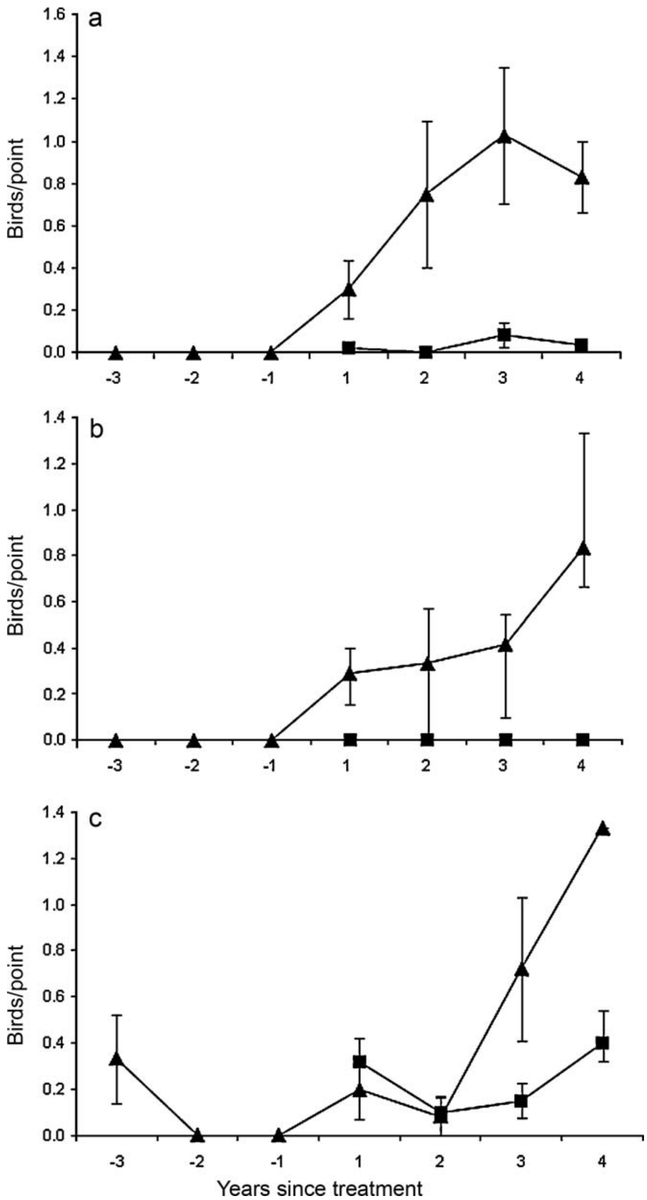

Fig. 3. Mean point count values with 95% confidence intervals for Prairie Warbler (a.) Field Sparrow (b.) and Eastern Towhee (c.) as a function of time since treatment in treated pitch pine forest (▲) and untreated pitch pine forest controls (■) from the Montague Plains WMA, 2004–2008.

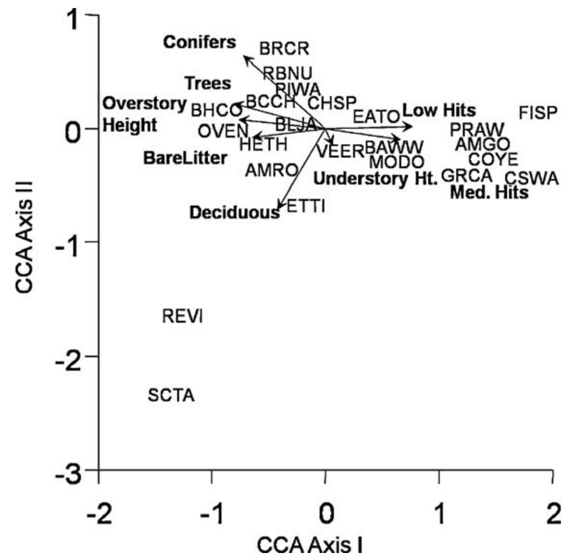

Fig. 4. Canonical correspondence analysis of 20 bird species occurring at ≥10% of 70 bird survey points the Montague Plains WMA 2004–2008. Species close together occur in similar environmental conditions. Environmental variables are represented by arrows. Their length is directly proportional to their importance in influencing bird community structure. The projected location of each species point along each arrow indicates how important the environmental variable is to the abundance and distribution of that species. Species codes as in Fig. 2, plus MODO, mourning dove; RBNU, red-breasted nuthatch; BRCR, brown creeper; BLJA, blue jay; VEER, veery; BHCO, brown-headed cowbird. "Trees" is the percent coverage of all trees; "Conifers" is the percent coverage of coniferous trees; "Deciduous" is the percent coverage of deciduous trees; "Overstory Height" is the height of the overstory (defined as >3 m) vegetation; "BareLitter" is the percent coverage of bare ground or leaf litter; "Low Hits" is the number of vegetation contacts with a 3-m pole between 0 and 1 m above ground; "Med Hits" is the number of contacts >1–2 m; "Understory Height" is the height of the understory (defined as <3 m) vegetation.

surements of fuel loadings and fire behavior indicate that these fuel treatments have significantly reduced the likelihood of crown fires (Duveneck, 2005), which represent the most immediate threat to human life and property in the area surrounding the Montague Plains WMA (Clark and Patterson, 2003).

4.2. Effects of treatments on birds

The changes in habitat structure caused by fuels management at our study site were accompanied by significant changes in bird species abundance. For the purposes of evaluating the effects of management on birds, we grouped them into those species that were less abundant in areas that had been thinned versus those that were more abundant in treated areas. This latter category would include species that were present in unthinned pitch pine that were more abundant in thinned pitch pine, as well as those species that were not present in unthinned pitch pine and were present in thinned pitch pine. Here we discuss the patterns we observed in bird abundance associated with fuels control treatments in our study, the factors likely responsible for these patterns, and finally, the significance of these patterns to regional bird conservation.

The bird species that were most abundant in unthinned pitch pine forest (black-capped chickadee, hermit thrush, ovenbird, pine warbler and red-breasted nuthatch) have also been reported from pine barrens in other parts of the eastern U.S. (Kerlinger and Doremus, 1981; Beachy and Robinson, 2008; Gifford et al., 2009). Although none of these studies compared the abundance of these species between untreated and treated habitats, Kerlinger and Doremus (1981) identified black-capped chickadees, ovenbirds and pine warblers as characteristic of mid- or late-successional pine barrens, which have a high degree of canopy closure relative to other pine barrens habitats. Gifford et al. (2009) also reported that black-capped chickadees and ovenbirds were most abundant in pitch pine–scrub oak forest relative to open canopy scrub oak barrens. King and DeGraaf (2000) reported that both ovenbirds and hermit thrushes were more abundant in mature northern hardwoods forest than in partially harvested sites, and Brawn (2006) reported that ovenbirds were less abundant in restored savannah than in closed canopy forest. In contrast, Greenberg et al. (2007) reported that ovenbirds were not affected by mechanical fuel treatments at their sites, however unlike the fuel treatments at our study area, the practices at their site were directed at understory fuels, and did not affect canopy closure.

No species were significantly more abundant in thinned pitch pine than other habitats according to the *N*-mixture analyses. Nevertheless, density estimates of American robins and chipping sparrows were ≥2 times higher in thinned pitch pine than the other habitats and chipping sparrows were most abundant in thinned pitch pine according to the Poisson regression. The association of these species with the open habitat provided by the thinned pitch pine sites is consistent with their classification as "birds of parklands, savannahs and open forests" by Schlossberg and King (2007).

The bird species that were scarce or absent in unthinned pitch pine, yet present in thinned pitch pine (black-and-white warbler, common yellowthroat, chestnut-sided warbler, eastern towhee, gray catbird and prairie warbler, brown thrasher and American goldfinch) were all most abundant in scrub oak barrens. These species are all closely associated with a low, dense woody understory and few overstory trees, and are thus classified as "scrub–shrub" species (Schlossberg and King, 2007). All have been reported from open habitats in other pitch pine–scrub oak barrens but are scarce or absent from forested habitats with dense canopy (Kerlinger and Doremus, 1981; Beachy and Robinson, 2008; Gifford et al., 2009). Historically, suitable habitat for these species was created by natural disturbances such as wind events and flooding (Askins, 2000; Chandler et al., 2009), or in the case of the Montague Plains, by occasional severe fires that opened up the canopy (Motzkin et al., 1999). The conditions created by periodic fire would likely have created a range of habitat conditions, including open savannah, similar to the conditions created by the fuel reduction applied at our sites (Wagner et al., 2003).

Mixed deciduous forest comprises a distinctive habitat component of the Montague Plains. Several species were strongly associated with deciduous canopy cover and present in mixed deciduous forest but scarce or absent from all other habitats. These included species that are typically associated with eastern hardwood forests, some of which avoid conifers (King and DeGraaf, 2000). One of the negative consequences of fire suppression is increased dominance of fire-intolerant species which can result in the reduced abundance of fire-adapted species and even changes in site conditions (Nowacki and Abrams, 2008). The effects of fire suppression are also reflected in the bird fauna. The bird species closely associated with mesophytic vegetation in our study are species that are uncharacteristic of pitch pine–scrub oak barrens, but which become more common as the result of fire exclusion and the subsequent invasion of fire-intolerant tree species (Kerlinger and Doremus, 1981; Brawn, 2006; Beachy and Robinson, 2008). Although many of the bird species that were associated with mixed deciduous forest in this study are sensitive to habitat fragmentation (e.g. Robinson et al., 1995) and threatened by increasing urbanization (Kluza et al., 2000) their populations are generally stable in the Northeast (Sauer et al., 2008), and they are well represented in the mixed deciduous forest that covers most of the region. Maintaining their populations by allowing the development of the mesophytic vegetation with which they are associated on pitch pine–scrub oak barrens such as the Montague Plains would make a minor contribution to their regional populations, yet would cause the loss of the distinctive ecological characteristics of this globally threatened community.

5. Conclusions

Understanding the ecological effects of fuels management on wildlife is a necessary part of designing and evaluating these treatments (Moreira et al., 2003; Penman et al., 2007; Burrows, 2008; Kalies et al., 2010). Our finding that pitch pine–scrub oak barrens are occupied by bird species that are both negatively and positively affected by fuels management presents managers with a potential tradeoff between the needs of these two species groups (Artman et al., 2005; Brawn, 2006). Although both groups merit conservation attention, key distinctions in their regional vulnerability can be drawn based on their habitat use and availability. The species that were most abundant in untreated pitch pine (black-capped chickadee, hermit thrush, pine warbler, red-breasted nuthatch and ovenbird) were also present in mixed deciduous forest, albeit at lower densities. Nevertheless, mixed deciduous forest covers >80% of the region (Schlossberg and King, 2007), so on a regional scale, it supports a large aggregate population of these species. In contrast, habitat for scrub–shrub species comprises only ~3% of the land area of Massachusetts and this amount continues to decrease (Schlossberg and King, 2007). This imbalance in availability of mature forest and scrub–shrub habitats is reflected in population trends. A much higher proportion of significant population trends for scrub–shrub birds in the eastern U.S. from the National Breeding Bird Survey are negative (77%) compared to mature forest species (44%). Based on similar evidence, Brawn (2006) argued in favor of the restoration of Midwestern savannahs that favored scrub–shrub and other disturbance dependent species despite its negative effects on mature forest birds, concluding that active management "favors a suite of species with comparatively pressing conservation needs" (p.467).

It is estimated that there are 83,180 ha of pitch pine–scrub oak woodlands in southern New England (Zuckerberg et al., 2004). Because pitch pine–scrub oak is naturally disturbance-prone, it is likely that 10–30% occurred in a scrub–shrub condition historically (Lorimer and White, 2003). Since there are ~173,322 ha of scrub–shrub habitat in New England (Schlossberg and King, 2007), the amount of scrub–shrub habitat that could be maintained in pitch pine–scrub oak barrens under historical disturbance regimes could comprise a substantial proportion of the regional total of scrub–shrub habitat in New England. In pitch pine–scrub oak barrens, scrub–shrub conditions also persist for a long period after disturbance (Lorimer and White, 2003) relative to other scrub–shrub habitats like regenerating clearcuts, which are unusable by scrub–shrub birds within a decade or so (Schlossberg and King, 2009). Thus, pitch pine–scrub oak barrens could provide an important contribution to the regional conservation of these species (Gifford et al., 2009).

Acknowledgements

We thank the Massachusetts Natural Heritage and Endangered Species Program and the National Fire Program for support of this project. R. Chandler, B. Mazzei and R. Packard assisted with field work.

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