



## Habitat Relations

# Influence of Hiking Trails on Montane Birds

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**ABSTRACT** Montane forests contribute significantly to regional biodiversity. Long-term monitoring data, often located along hiking trails, suggests that several indicator species of this ecosystem have declined in recent decades. Declining montane bird populations have been attributed to anthropogenic stressors such as climate change and atmospheric deposition. Several studies from montane systems have also documented decreased estimates of abundance and reproductive success as well as altered singing rates near hiking trails. Therefore, recreational hiking may be contributing to montane bird population declines and potentially biased population estimates because of trail avoidance or altered detection probabilities near trails. We studied the effect of hiking trails on the abundance, seasonal movements, and detection probabilities of montane birds in the White Mountains, New Hampshire in 2006 and 2007. We used hierarchical, generalized *N*-mixture models that account for imperfect detection probabilities to examine the effects of recreational hiking on bird communities. We also examined the potential effects of hiking on the reproductive success of a boreal forest songbird, blackpoll warbler (*Setophaga striata*). We found little evidence to suggest hiking trails influence abundance, detection probabilities, or within and among seasonal movements of montane forest birds. We also found no evidence to suggest daily nest survival of blackpoll warbler nests vary with distance from trail. Our study suggests that recent increases in hiking traffic are unlikely to have caused declines in montane birds in this region. Furthermore, our results provide evidence that trail-based monitoring programs can provide accurate and efficient estimates of abundance for some montane forest bird species in the White Mountains of New Hampshire. © 2014 The Wildlife Society.

**KEY WORDS** abundance, blackpoll warbler, Dail–Madsen model, detection probability, montane forests, recruitment, White Mountains.

High elevation forests provide habitat to diverse and unique plant and animal communities. Current research suggests that montane ecosystems are susceptible to a host of stressors, including atmospheric deposition (Rimmer et al. 2005) and climate change (Rodenhouse et al. 2008). Montane bird communities of the northern Appalachian Mountains may be particularly vulnerable to these stressors because of their close proximity to dense human population centers, relatively small spatial extent, and the naturally fragmented distribution of montane habitats. King et al. (2008) found population declines between 1993 and 2003 in the White Mountain National Forest, USA for 3 montane bird species, 2 of which, yellow-bellied flycatcher (*Empidonax flaviventris*) and Bicknell's thrush (*Catharus bicknelli*), are considered ecological indicator species for montane spruce-fir forests (King et al. 2008). These findings emphasize the need to identify the causes of declines in montane bird populations as well as the importance of accurate monitoring to detect and track population trends.

Evidence exists that suggests the abundance of some species is lower near trails compared to locations farther from trails.

Heil et al. (2007) found that transects located on recreational trails had lower bird species richness and diversity compared to transects away from trails. Furthermore, human intrusions have also been shown to reduce the abundance of some montane species in western North America by 46–57% (Gutzwiller and Anderson 1999). Human intrusions can alter the timing of singing by montane birds (Gutzwiller et al. 1997), as well as song rates (Gutzwiller et al. 1994). A male's primary song is used to attract females (Eriksson and Wallin 1986) and to establish and defend territories (Krebs et al. 1978). Human intrusions have also been found to influence directly the reproductive success of some birds. Miller et al. (1998) found that some species avoid nesting near trails and that nest survival increased with increasing distance from trails in grassland and forest habitats. Based on these studies documenting the negative impacts of hiking on birds, recreational hiking may be considered a potential stressor to montane bird abundance, diversity, behavior, and reproductive success in northeastern North America, particularly because the White Mountain National Forest is within a day's drive of 70 million people, and hiking visitation increased approximately 33% between 1998 and 2010 when over 700,000 hikers visited the forest (DuRocher 2011).

In addition to the potential impact of hiking trails on bird populations, the effects of human intrusions could also affect

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the validity of the trail-based surveys typically used to monitor these populations (King et al. 2008, Lambert et al. 2008) by changing the singing behavior of birds near trails, and hence their detectability. Low detection rates of birds near trails could affect how representative estimates of abundance derived from trailside surveys are of the regional population. Currently, the Breeding Bird Survey (BBS) is the primary program for monitoring populations of North American birds (Sauer and Link 2011); however, this program notoriously underrepresents habitats that are relatively rare across the landscape (Bart and Peterjohn 1995, Thogmartin et al. 2006). Montane forests encompass approximately 2% of the land area in New Hampshire (Lambert et al. 2005), most of which is inaccessible by roads and is, therefore, unavailable for sampling by the BBS. Because the vulnerable montane bird community of the Northeast is poorly monitored, 2 long-term monitoring efforts have been implemented in the region, both of which have surveys located on hiking trails (King et al. 2008, Lambert et al. 2008). Although these survey programs focus on montane ecosystems, they are potentially hampered by their restriction to hiking trails, warranting similar scrutiny as the BBS for being limited to roads (Thogmartin et al. 2006). The same potential shortcomings attributed to the BBS roadside survey protocol may apply to trail-based surveys, although disturbances along trails are considerably less than those along roads. There is little understanding of how accurately trail-based monitoring programs represent the montane bird community as a whole (Lambert et al. 2008).

Despite evidence that trail-based recreation negatively affects abundance and breeding ecology of montane birds elsewhere in North and South America, no detailed studies have investigated the effect of recreational trails on montane birds in eastern North America. We surveyed birds to determine whether proximity to hiking trails affects 1) abundance, 2) detection probabilities, 3) within- and among-season recruitment and abundance stability, and 4) nest success of a montane spruce-fir obligate, blackpoll warbler (*Setophaga striata*).

## STUDY AREA

We surveyed montane birds in 2006 and 2007 at 10 sites within the Presidential Range and Crawford Notch region of the White Mountain National Forest (N44°7' to N44°21' W71°27' to W71°14'). Each site encompassed a segment of hiking trail classified by the White Mountain National Forest as having a relatively heavy volume of hiking traffic (L. S. Prout, White Mountain National Forest, personal communication) and was within a region of uniformly high trail densities (Smith and Dickerman 2012). Trail width typically ranged between 0.5 m and 1.5 m. We selected 1 of these trails, Caps Ridge Trail on Mount Jefferson of the Presidential Range, as a site to monitor blackpoll warbler daily nest survival from 2006 to 2009. We selected this site because it was the most accessible site that provided a good representation of montane forest habitat throughout the complete elevation gradient.

We established 4 survey locations on each trail with distances between survey locations no less than 250 m and no more than 1 km. We spaced survey locations evenly along hiking trails at a site within the montane spruce-fir forest which is typically between elevations of 750 m and treeline, which is approximately 1,500 m above sea level (ASL) in the region. Therefore, if only 750 m of trail occurred within the montane forest, survey locations were 250 m apart and if 1,000 m of trail occurred within the montane forest, then survey locations were approximately 333 m apart. Within this elevation zone, the vegetation is characterized by a transition with increasing elevation, from a high canopy, mixed coniferous-deciduous forest to a low canopy forest dominated by balsam fir (*Abies balsamea*) and red spruce (*Picea rubens*) with inclusions of paper birch (*Betula cordifolia*) and mountain ash (*Sorbus americana*; Sabo 1980). For each survey on the trail, we established 1 survey at 200 m and 1 at 400 m perpendicular to the trail. We surveyed 90 locations in 2006; 40 survey locations on trails, 40 at 200 m from a trail, and 10 at 400 m from a trail. In 2007, we surveyed the same 90 locations as well as an additional 23 locations that we did not survey in 2006 that were 400 m from the trail (total of 33 survey locations 400 m from the trail). Because of excessive trail densities or extreme topography, 7 trail surveys did not have accompanying surveys 400 m from the trail in 2007.

## METHODS

### Field Methods

We conducted point count surveys in both years, 3 times at each location during the height of the breeding season between 6 June and 17 July by 1 of 4 trained observers. Each survey consisted of a 10-minute point count during which we recorded singing males within 50 m of the survey location (Betts et al. 2008). We restricted analysis to the 5 species of high-elevation forest-specialist birds with >40 detections: blackpoll warbler, Bicknell's thrush, boreal chickadee (*Poecile hudsonicus*), Swainson's thrush (*Catharus ustulatus*), and yellow-bellied flycatcher. Surveys began at dawn and continued until 1100 hours. The first round of surveys within each year within each site started at lower elevation survey locations and proceeded upslope, the second round of surveys began at the highest elevations and proceeded downslope, and the last round of surveys of the year followed the order of the first. We resampled sites between 10 and 15 days after the previous survey. We assigned a weather score of 1 through 4 for each survey, where a score of 1 indicated clear conditions and a score of 4 indicated clouds with some drizzle. We also assigned a wind score of 1 through 4, where a 1 indicated calm conditions and a score of 4 indicated that wind was strong enough to move tree branches, but not the main trunk of the tree. We did not conduct surveys when weather or wind conditions warranted a score higher than 4. Weather and wind scores follow the protocol for Mountain Birdwatch, a regional montane bird monitoring program (Mountain Birdwatch, <http://www.vtecostudies.org/MBW/pdf/MBWdatasheetB.pdf>, accessed 15 Apr 2005).

Habitat characteristics varied substantially among survey locations because of large-scale influences such as climate variation along the elevation gradient (Seidel et al. 2009) as well as smaller scale disturbances such as wind, ice, and exposure (Sprugel and Bormann 1981). Therefore, we measured several habitat characteristics at each survey location. In general, habitat characteristics were static over the course of the breeding season; therefore, we collected measurements once at the conclusion of the third survey. We centered 25-m radius subplots on each bird survey location. Within each subplot, we quantified canopy height, canopy closure, mid-canopy height, and mid canopy closure. We estimated the percent cover for each species within the 25-m subplot for the canopy and mid-canopy. We also established a 10-m radius subplot at the center of the survey location where we counted all softwood (primarily balsam fir and red spruce) stems and identified them as being greater than or less than 10 cm diameter at breast height (dbh; Eliason 1986). We considered the measured habitat characteristics a parsimonious description of relevant habitat structure and composition based on previous studies (Eliason 1986, Frey et al. 2011).

At each trail, we measured daily trail use with an active infrared trail counter (TrailMaster TM1000, Lenexa, KS). We collected daily trail use for 5- to 8-day periods, 3 times in 2007. The first trail-use sampling period was between 7 June and 19 June, the second sampling period was between 23 June and 8 July, and the final sampling period was between 13 July and 24 July. We measured elevation at each survey location with a global positioning system.

We located blackpoll nests with systematic searches of suitable vegetation as well as by following individuals carrying nest material or food. We captured and individually marked most breeding individuals. We monitored nests every third day and determined nest success following the recommendations of Manolis et al. (2000). Upon nest completion, we attempted to confirm success by searching for fledglings near the nest or being fed by a known parent because nest fate cannot be reliably assigned by nest condition (King and DeGraaf 2006). After nestlings fledged or the nest failed, we recorded the elevation of the nest, height, and dbh of the nest tree, and the shortest linear distance from the nest to the hiking trail. We also recorded slope, canopy height, canopy closure, percent of canopy that was balsam fir, the number of balsam fir stems <10 cm dbh, and the number of balsam fir stems >10 cm dbh on a 10-m radius plot centered on the nest. We selected nest-site variables based on findings of Eliason (1986) and Dalley et al. (2009).

### Analysis

We created a parsimonious set of habitat characteristic variables from a total set of 15 habitat variables with a principal components analysis (PCA) that extracted the dominant underlying environmental gradients (McGarigal et al. 2000). To assess the significance of each principal component, we performed a Monte Carlo test with 1,000 permutations to test whether the observed eigenvalue was

significantly different from the distribution of eigenvalues under the null hypothesis. We used the factor loadings  $>0.3$  or  $<-0.3$  to interpret the ecological meaning of significant principal components (McGarigal et al. 2000). We then used the reduced set of principal components as independent variables in subsequent analyses. We used an analysis of variance (ANOVA) to test whether significant principal components varied by distance-to-trail categories and a Tukey's honest significant difference (HSD) test to determine which distance-from-trail categories differed. To determine whether hiker activity varied by sampling period, we used an ANOVA followed by a Tukey's HSD test to determine which sampling period categories differed from each other ( $\alpha = 0.05$ ).

We used the Dail-Madsen model (Dail and Madsen 2011) to derive estimates of abundance ( $\lambda$ ), recruitment ( $\gamma$ ), apparent survival ( $\omega$ ), and detection probability ( $p$ ) from point count data. The Dail-Madsen model is a generalization of the  $N$ -mixture model (Royle 2004) and is also hierarchical because it includes explicit models of the ecological process as well as the detection process (Royle and Dorazio 2008). The model is unique because it does not rely upon the assumption that survey locations are closed to population changes between locations and survey intervals and thus allows abundance to fluctuate over time. The Dail-Madsen model is also unique because it provides estimates of recruitment ( $\gamma$ ) and apparent survival ( $\omega$ ) within a breeding season and among breeding seasons while incorporating covariates for both estimates for unmarked populations (Dail and Madsen 2011).

Although  $\omega$  is typically interpreted as apparent survival, we considered it a measure of abundance maintenance over a given period of time and made no assumptions as to whether negative changes in abundance were due to relocations or mortality. Therefore, we refer to  $\omega$  as a measure of abundance stability, defined as changes in abundance due to the loss of individuals to permanent emigration out of the survey area and/or losses due to mortality (Chandler and King 2011), such that estimates close to 1.0 were interpreted as minimal changes in abundance and estimates close to 0.0 were interpreted as substantial decreases in abundance. Estimates of  $\gamma$  and  $\omega$  enabled us to test whether or not recruitment or abundance stability changed over the course of a breeding season, particularly if hiking activity also increased, or between breeding seasons in response to hiking trails.

We considered 8 potential covariates for  $p$  including wind, weather, temperature, observer, date, time, distance from trail, and elevation. We also attempted to maximize detection probability by limiting the analysis to singing males within 50 m of the survey location (Chandler et al. 2009). We treated distance from trail as a categorical variable and considered it as a covariate for  $\lambda$ ,  $\gamma$ ,  $\omega$ , and  $p$ . To control for the potentially confounding effects of habitat characteristics, we considered significant principal components as covariates on all parameter estimates. We included elevation as a covariate in candidate models to account for unmeasured abiotic or habitat conditions that exist along the elevation gradient. Because many species have a modal (or

Gaussian) response to elevation or habitat conditions (Able and Noon 1976), we also considered quadratic terms for elevation and habitat principal components. We estimated recruitment and abundance stability at monthly intervals and then extrapolated for within and among breeding season estimates. We included a season covariate for  $\gamma$  and  $\omega$  in all candidate models to acquire estimates within and among breeding seasons. The model uses a binomial distribution to model  $\omega$  and  $p$  and a Poisson distribution to model  $\lambda$  and  $\gamma$ . We fit Dail–Madsen models using the pcountOpen function from the unmarked library (R version 2.14.1, www.r-project.org, accessed 2 Dec 2011).

We compared single-covariate models for each covariate using corrected Akaike's Information Criterion ( $AIC_c$ ; Burnham and Anderson 2002). We then considered all possible combinations of covariates that performed better than the null model in the single-covariate comparisons. We first assessed the effect of distance to trail on  $\lambda$ ,  $\gamma$ ,  $\omega$ , and  $p$  by determining whether the distance from trail variable was included in a supported ( $\Delta AIC_c \leq 2$ ) model (Burnham and Anderson 2002). When supported models did not include the trail distance covariate, we used  $\Delta AIC_c$  to compare the null model to the model with the most support that included the distance from trail covariate anywhere in the model

(Table 1). If the best model with the trail distance covariate had less support than the null model, we interpreted it as assurance that the trail distance covariate was not an influential predictor variable. Finally, we generated parameter estimates for  $\lambda$ , within breeding season  $\gamma$ , among breeding seasons  $\gamma$ , within breeding season  $\omega$ , among breeding seasons  $\omega$ , and  $p$  for survey locations on the trail and 200 m and 400 m from the trail by including the distance-from-trail covariate in the model with most support for each species. We then compared the 95% confidence intervals of the parameter estimates for survey locations on the trail, at 200 m from the trail, and at 400 m from the trail for each species to determine whether parameter estimates varied as a function of the distance-from-trail categories.

We used the logistic-exposure model described by Shaffer (2004) to determine whether distance to trail affected daily nest survival of blackpoll warblers. To calculate exposure days, we used the midpoint between the final 2 nest checks for unsuccessful nests (Manolis et al. 2000, Chandler et al. 2009). We compared candidate models with a forward selection process considering all nest-site habitat variables using  $AIC_c$  and assessed the effect of distance to trail, as a continuous variable, on daily nest survival by determining whether the distance from trail variable was included in a

**Table 1.** Dail–Madsen model selection results for models with  $\Delta$  Akaike's Information Criterion adjusted for small sample size ( $\Delta AIC_c$ )  $\leq 2.0$ . Models that include the distance-from-trail covariate with the lowest  $\Delta AIC_c$  are also included for comparison with the null model. We investigated covariates that influenced abundance ( $\lambda$ ), recruitment ( $\gamma$ ), abundance stability ( $\omega$ ), and detection probability ( $p$ ). Covariates are references as E: elevation; H1: habitat principal component 1; H2: habitat principal component 2; T: distance from trail; D: date; O: observer; K: weather; W: wind; C: temperature; and I: time. Species are reference as BLPW: blackpoll warbler (*Setophaga striata*); BITH: Bicknell's thrush (*Catharus bicknelli*); BOCH: boreal chickadee (*Poecile hudsonicus*); SWTH: Swainson's thrush (*Catharus ustulatus*), and YBFL: yellow-bellied flycatcher (*Empidonax flaviventris*). Squared terms indicate quadratic effects and a period indicates no covariate effects. We derived models from data collected at 113 survey locations in the White Mountain National Forest in 2006 and 2007.

Species	Model	$\Delta AIC_c^a$	$w_i^b$	$R^{2c}$	$K^d$
BLPW	$\lambda(H1^2) \gamma(.) \omega(.) p(O+W)$	0.00	0.87	0.35	11
	Null	36.87	<0.01	<0.01	6
BITH	$\lambda(.) \gamma(.) \omega(.) p(T)$	39.98	<0.01	<0.01	8
	$\lambda(H2) \gamma(H1) \omega(H1) p(E+I)$	0.00	0.14	0.34	11
	$\lambda(H2) \gamma(H1) \omega(H1) p(D+E+I)$	0.08	0.13	0.36	12
	$\lambda(H1+H2) \gamma(H1) \omega(H1) p(D)$	0.29	0.12	0.34	11
	$\lambda(H1+H2) \gamma(H1) \omega(H1) p(I)$	1.69	0.06	0.35	11
	Null	34.29	<0.01	<0.01	6
BOCH	$\lambda(T) \gamma(.) \omega(.) p(.)$	36.49	<0.01	<0.01	8
	$\lambda(H2) \gamma(H1^2) \omega(H1^2) p(O+K)$	0.00	0.22	0.33	15
	$\lambda(H2) \gamma(H1^2) \omega(H1^2) p(O+K+W)$	0.66	0.16	0.33	16
	$\lambda(H2) \gamma(H1^2) \omega(H1^2) p(K+W)$	0.75	0.15	0.30	13
	$\lambda(.) \gamma(H1^2) \omega(H1^2) p(O+K+W)$	1.14	0.13	0.32	15
	$\lambda(.) \gamma(H1^2) \omega(H1^2) p(O+K)$	1.18	0.12	0.31	14
	Null	23.36	<0.01	<0.01	6
	$\lambda(T) \gamma(E) \omega(E) p(K+W)$	23.72	<0.01	<0.01	12
SWTH	$\lambda(.) \gamma(H1^2) \omega(H1^2) p(C+W)$	0.00	0.47	0.52	12
	$\lambda(.) \gamma(H1^2) \omega(H1^2) p(E+C+W)$	1.02	0.28	0.53	13
	Null	71.26	<0.01	<0.01	6
	$\lambda(T) \gamma(.) \omega(.) p(.)$	72.23	<0.01	<0.01	8
YBFL	$\lambda(H2) \gamma(.) \omega(.) p(O+I+W)$	0.00	0.40	0.31	12
	$\lambda(.) \gamma(.) \omega(.) p(O+I+W)$	0.53	0.31	0.30	11
	$\lambda(H1+H2) \gamma(.) \omega(.) p(O+I+W)$	1.99	0.15	0.31	13
	Null	30.16	<0.01	<0.01	6
	$\lambda(.) \gamma(.) \omega(.) p(T)$	45.32	<0.01	<0.01	8

<sup>a</sup> Difference between model's Akaike's Information Criterion corrected for small sample size ( $AIC_c$ ) and the lowest  $AIC_c$  value.

<sup>b</sup> Relative weight attributed to the model.

<sup>c</sup> Measure of variation explained by the model.

<sup>d</sup> Number of parameters estimated by the model.

**Table 2.** Factor loadings from a principal components (PC) analysis describing variation in habitat characteristics at 113 point count locations in the White Mountains, New Hampshire sampled in 2006 and 2007. We interpreted only factor loadings  $>0.3$  or  $<-0.3$ .

Habitat characteristic	PC 1	PC 2	PC 3
Canopy height (m)	-0.41		
Canopy cover (%)			-0.51
Balsam fir ( <i>Abies balsamea</i> ) canopy cover (%)		-0.58	
Red spruce ( <i>Picea rubens</i> ) canopy cover (%)	-0.34		
Paper birch ( <i>Betula cordifolia</i> ) canopy cover (%)		0.54	
Standing dead trees (count)			0.43
Mid-canopy height (m)	-0.47		
Mid-canopy cover (%)	-0.37		0.44
Balsam fir mid-canopy cover (%)	-0.35		
Red spruce mid-canopy cover (%)		0.37	
Mountain ash ( <i>Sorbus Americana</i> ) mid canopy cover (%)			-0.33

supported ( $\Delta AIC_c \leq 2$ ) model (Burnham and Anderson 2002). We then compared the most supported model with the distance-from-trail covariate to the null model. We considered parameter estimates with 95% confidence intervals that did not include 0 as significant. We fit regression models using the glm package in R.

## RESULTS

The Monte Carlo test revealed that the observed eigenvalues of first 3 habitat principal components were different ( $P < 0.05$ ) from the distribution of eigenvalues under the null hypothesis. The 3 principal components explained 46% of variation in the habitat data. The first principal component described sites with increasing elevation primarily varying in structure, such that the forest decreases in canopy height, mid-canopy height, and mid-canopy cover (Table 2). The second principal component primarily described variation in forest composition from sites with canopies dominated by red spruce to those dominated by balsam fir (Table 2). The third principal component described variation associated with smaller scale disturbances resulting in localized regeneration. These sites were characterized by open canopies and abundant standing dead trees with relatively abundant regeneration (Table 2). Principal component 1 did not vary as a function of the trail distance categories ( $F_{2,110} = 15.57$ ,  $P = 0.06$ ). Principal component 2 was different among the distance-from-trail survey locations ( $F_{2,110} = 15.28$ ,  $P = 0.03$ ), however, it did not differ between trail survey locations and locations 200 m ( $P = 0.73$ ) or 400 m ( $P = 0.13$ ) from the trail. Principal component 3 also did not vary as a function of the trail distance categories ( $F_{2,110} = 11.48$ ,  $P = 0.13$ ).

Daily trail hiking traffic differed between the 3 sampling periods ( $F_{2,193} = 17.16$ ,  $P < 0.01$ ). Mean daily hiking traffic of the first sampling period (6.5,  $SD = 5.8$ ) was less than the second (57.9,  $SD = 55.4$ ) and third (53.7,  $SD = 63.3$ ) sampling periods ( $P < 0.01$  for both comparisons). Daily trail use between the second and third sampling periods was similar ( $P = 0.87$ ).

Dail-Madsen models explained 27–53% of the variation in high elevation bird abundance (Table 1). Distance from trail was not included as a covariate for  $\lambda$ ,  $\gamma$ ,  $\omega$ , or  $p$  in any of the supported models for blackpoll warbler, Bicknell's thrush, Swainson's thrush, yellow-bellied flycatcher, and boreal

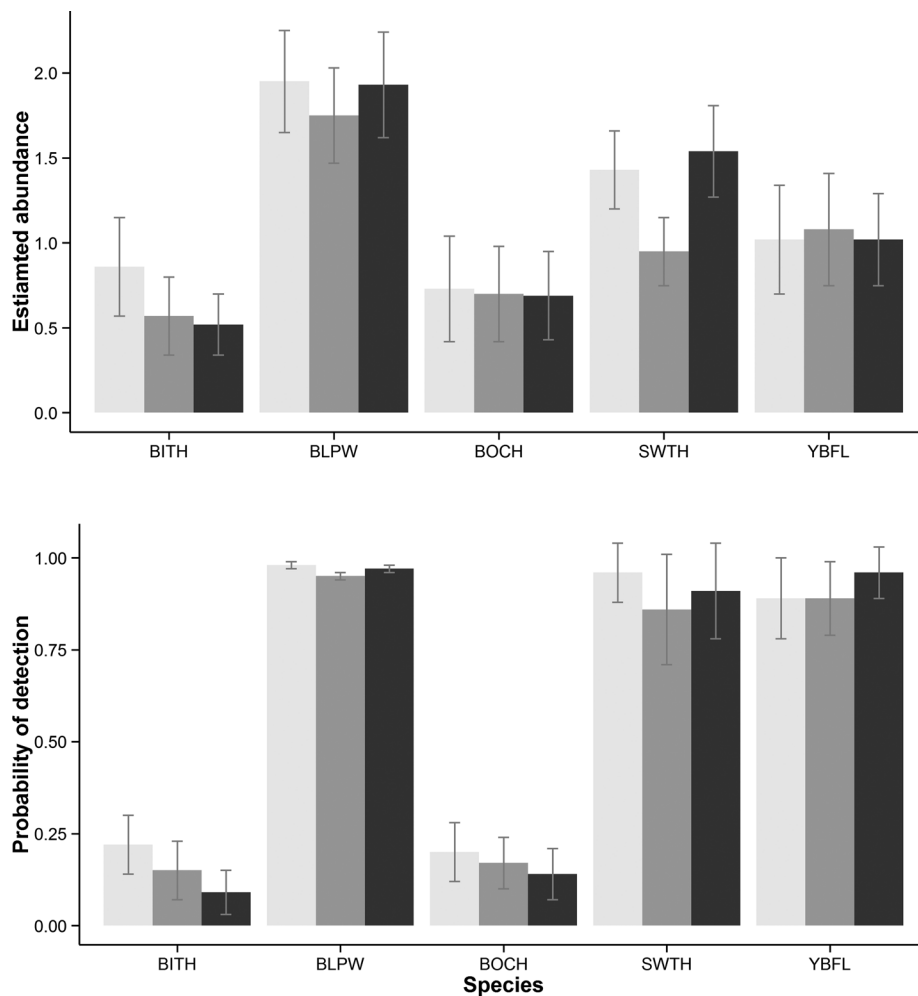
chickadee (Table 1). For all species, the best models with the trail-distance covariate had less support than the null model (Table 1). Parameter estimates for abundance ( $\lambda$ ) and detection probability ( $p$ ) did not differ between distance-from-trail categories for any species (Fig. 1). Within and among breeding season recruitment (Fig. 2) and within and among breeding season abundance stability (Fig. 3) parameter estimates also did not vary as a function of distance from trail for any species.

We monitored 35 blackpoll nests on Mount Jefferson, 11 of which failed. Overall, daily nest survival was 0.969 and nest success was 51.6%. The dbh of the nest tree and elevation were supported predictors of daily nest survival (Table 3). We did not find strong support for logistic-exposure models that included distance from trail (Table 3). Logistic exposure models with the forced trail-distance covariate were not supported over the originally supported models ( $\Delta AIC_c \geq 5$  for all comparisons).

## DISCUSSION

We found no evidence to suggest that recreational hiking trials have adverse effects on the abundance, detection probabilities, abundance stability, or recruitment of the species considered in this study in the White Mountains, New Hampshire. We also found no evidence that recreational hiking trails influence the daily nest survival of a montane forest indicator bird species for the White Mountain National Forest. Our results suggest that recreational hiking in the montane areas of New Hampshire is not likely an influential stressor to the 5 species considered in this study, and that trail-based surveys may represent a valid methodology for sampling montane birds in the region.

Trail avoidance by birds found in other studies could have been caused directly by human intrusions or indirectly by birds responding to differences in habitat characteristics near trails. Some studies have controlled for effects of habitat characteristics by either confirming no habitat differences (Gutzwiller and Anderson 1999) or by including remotely sensed habitat characteristics as covariates in their models (Heil et al. 2007, Kangas et al. 2010). Although we did not find that any of the habitat principal components differed between trail and off-trail locations, we did control for the effects of habitat by including the habitat principal components in our candidate models.

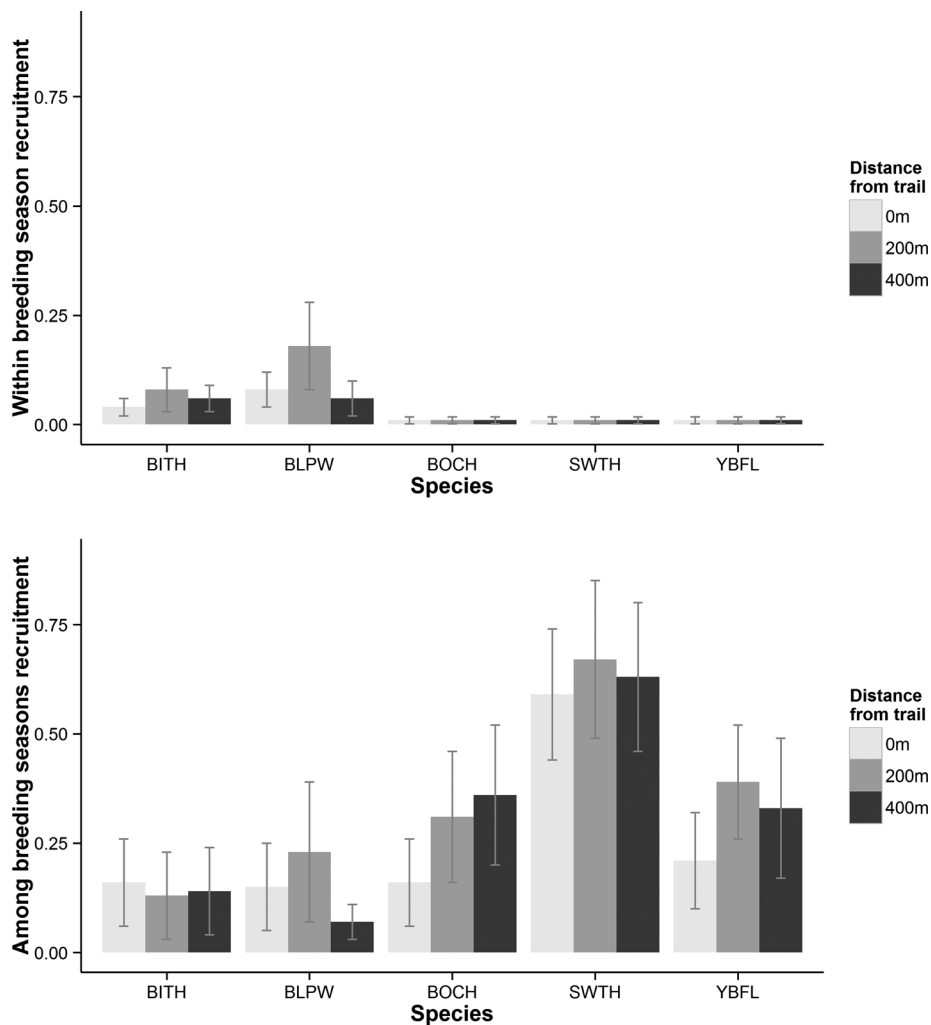


**Figure 1.** Abundance and detection probability parameter estimates ( $\pm 1$  SE) for 5 songbird species from the Dail–Madsen model with the most support for surveys located on the trail (light gray), 200 m from the trail (gray), or 400 m from the trail (dark gray). The 95% confidence intervals for each estimate overlapped between the 3 distance-from-trail categories for all species. We derived models from data collected at 113 survey locations in the White Mountain National Forest in 2006 and 2007. Species are referenced as BITH: Bicknell’s thrush (*Catharus bicknelli*); BLPW: blackpoll warbler (*Setophaga striata*); BOCH: boreal chickadee (*Poecile hudsonicus*); SWTH: Swainson’s thrush (*Catharus ustulatus*), and YBFL: yellow-bellied flycatcher (*Empidonax flaviventris*).

For blackpoll warbler, Bicknell’s thrush, Swainson’s thrush, boreal chickadee, and yellow-bellied flycatcher, we found no evidence that individuals that establish territories near trails are likely to move from those locations either as the breeding season progresses or between breeding seasons, despite the fact that hiker activity dramatically increased over the course of the breeding season. We also found that for the same species, sites at 200 m and 400 m from the trail are no more likely to be settled than trail sites as the season progresses or between breeding seasons. In contrast to our findings, Kangas et al. (2010) reported that occurrence and composition of bird communities were altered by recreational hiking in a protected area in Finland; however, hiking pressure in their study contributed much less to model performance than measures of habitat characteristics. They questioned whether the relatively low contribution of hiking pressure when explaining variation in bird occurrence was because hiking visits are relatively low when birds are arriving and settling on territories in May and June compared to later in the season. By showing that abundance stability and

recruitment did not differ over the course of a breeding season under similar changes in hiking pressure as documented in Kangas et al. (2010), our study suggests that variations in hiking traffic over the course of a season do not affect abundance or occurrence at our sites.

We did not find evidence that detection probabilities differed between trail sites and sites 200 m or 400 m from trails, suggesting that changes in song consistency and/or song occurrence between trail and off-trail locations were similar. However, Gutzwiller et al. (1994) found that for some subalpine birds, song occurrence and consistency differed between sites experimentally exposed to human intrusions and control sites. Although we did not explicitly measure song consistency or occurrence, some differences may exist as a function of distance from trail but the signal may not have been strong enough to be identified by a surrogate measure such as detection probability. Furthermore, we conducted our surveys between dawn and 1100 hours, before daily hiking activity peaked. Detection probabilities could have varied as a function of distance from



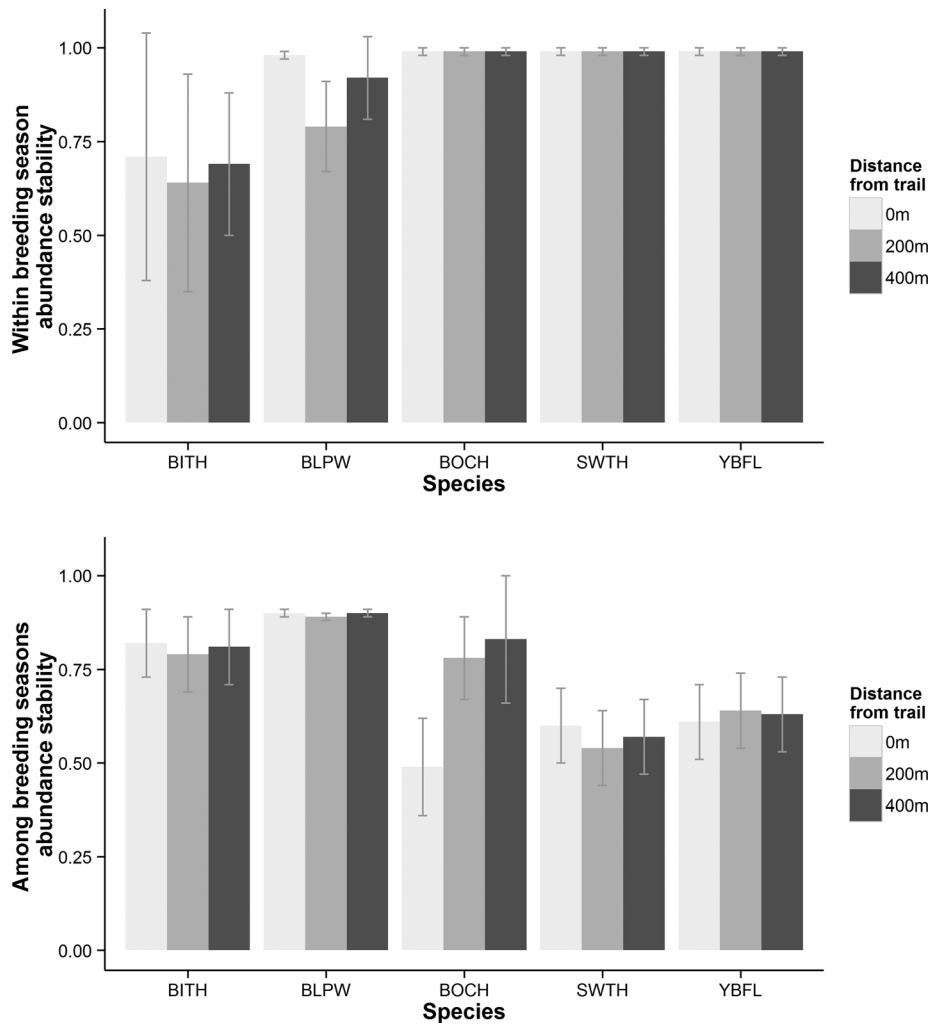
**Figure 2.** Recruitment ( $\gamma$ ) parameter estimates ( $\pm$ SE) for 5 songbird species from Dail–Madsen models with the most support including the distance-from-trail covariate. We back-transformed coefficient standard errors using the delta method (Powell 2007). Species are referenced as BITH: Bicknell’s thrush (*Catharus bicknelli*); BLPW: blackpoll warbler (*Setophaga striata*); BOCH: boreal chickadee (*Poecile hudsonicus*); SWTH: Swainson’s thrush (*Catharus ustulatus*), and YBFL: yellow-bellied flycatcher (*Empidonax flaviventris*). Within season estimates for BOCH, SWTH, and YBFL were not distinguishable from 0. We derived models from data collected at 113 survey locations in the White Mountain National Forest in 2006 and 2007.

trail if surveys were conducted during peak daily hiking activity. However, Gutzwiller et al. (1994) found an effect of song consistency and occurrence when experimental intrusions were applied between 1 and 6 days prior to surveys. Regardless, the fact that detectability at our sites did not differ between trailside areas and sites away from trails suggests any differences in singing behavior that might exist do not affect the validity of trailside counts for monitoring montane birds at our sites.

Daily nest survival was not influenced by the distance of the nest to the trail. Previous studies have found conflicting results. Miller et al. (1998) found that nest survival increased with increasing distance from trails, whereas Miller and Hobbs (2000) found that nest predation rates were higher farther from trails. The assumption is that mammalian and avian nest predators are more abundant and have greater access to nests near habitat edges or corridors because predators frequently travel along habitat edges or established routes (Rich et al. 1994). Red squirrel (*Tamiasciurus hudsonicus*) is a primary nest predator of birds in the

montane forests of the White Mountains (Holmes 2011). Within the relatively unfragmented landscape of the White Mountains, small corridors, such as recreational trails, may not be preferred by red squirrels because habitat near trails does not differ from habitat farther from trails in a way that facilitates travel.

Our finding that nest survival rates did not vary as a function of distance from trail is consistent with our observation that abundance, recruitment, and abundance stability also did not vary with distance from trail. This implies that not only was bird abundance near trails similar to sites farther from trails, but that breeding success near trails was similar to areas farther from trails. Because nest predation rates can vary with nest height (Sloan et al. 1998), these results might only be relevant to species that generally nest in similar vegetation strata to that of the blackpoll warbler. Furthermore, forest passerines are typically more likely to have greater annual return rates to territories where they were reproductively successful during past breeding attempts (Hoover 2003). Therefore, our findings that these



**Figure 3.** Abundance stability ( $\omega$ ) parameter estimates ( $\pm$ SE) for 5 songbird species from Dail–Madsen models with the most support including the distance-from-trail covariate. We back-transformed coefficient standard errors using the delta method (Powell 2007). Species are referenced as BITH: Bicknell’s thrush (*Catharus bicknelli*); BLPW: blackpoll warbler (*Setophaga striata*); BOCH: boreal chickadee (*Poecile hudsonicus*); SWTH: Swainson’s thrush (*Catharus ustulatus*), and YBFL: yellow-bellied flycatcher (*Empidonax flaviventris*). Within season estimates for BOCH, SWTH, and YBFL were not distinguishable from 1. We derived models from data collected at 113 survey locations in the White Mountain National Forest in 2006 and 2007.

species were not more likely to move away from trails between breeding seasons may be an additional indication that relative breeding success does not substantially vary between trail and off-trail sites for other species for which we did not collect nesting data.

## MANAGEMENT IMPLICATIONS

Management plans for montane birds of conservation concern in the northeastern United States that attempt to alter or reduce hiking activity along non-motorized

**Table 3.** Parameter estimates of logistic exposure models of blackpoll warbler daily nest survival with  $\Delta$  Akaike’s Information Criterion adjusted for small sample size ( $AIC_c \leq 2.0$ ). We include the null model as well as the most supported model with the distance-from-trail covariate (trail). Parameter estimates are indicated with an asterisk (\*) if 95% confidence intervals do not include 0. Covariates include % canopy closure (canopy), diameter at breast height of nest tree (dbh), elevation of nest (elev), and number of balsam fir stems <10 cm dbh (fir). We measured vegetation variables on a 10-m radius plot centered on the nest tree. Data are from 35 nests on Mount Jefferson in the White Mountain National Forest from 2006 to 2008.

$\beta$	0 <sup>a</sup>	Canopy	dbh	Elev	Fir	Trail	$\Delta AIC_c$	$w_i^b$	$R^2$
-12.37			0.38*	0.41*	0.01		0.00	0.37	0.21
-14.01			0.32*	0.35*			0.36	0.35	0.19
0.84			0.43*		0.01		1.86	0.17	0.17
-15.12		0.02	0.15*	0.35*	0.01		1.93	0.12	0.20
3.17							7.62	0.01	0.08
-21.21			0.53*	0.02	0.01	0.01	8.11	0.01	0.07

<sup>a</sup> Parameter estimate for the intercept.

<sup>b</sup> Relative weight attributed to the model.



recreational trails will not likely have any meaningful positive influence on the conservation of these species. Therefore, management resources may be more effective elsewhere. Our results found no evidence to suggest that current trail-based monitoring programs of montane birds in the northeastern United States are providing biased estimates of abundance. We recommend the continued use of trails to efficiently monitor montane birds.

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