

2010 Report to the Vermont Monitoring Cooperative



Bicknell's Thrush with a backpack geolocator attached.

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Part I. Demographic Monitoring of Montane Forest Birds on Mt. Mansfield

In 2010, we continued demographic monitoring of Bicknell's Thrush (*Catharus bicknelli*), Swainson's Thrush (*C. usutulatus*), Blackpoll Warbler (*Dendroica striata*), Yellow-rumped (Myrtle) Warbler (*D. coronata coronata*), and White-throated Sparrow (*Zonotrichia albicollis*), completing our 19th consecutive field season on Mt. Mansfield.

Study Areas and Methods

We used mist-netting and banding to sample breeding populations of the five target species on an established study plot on the Mt. Mansfield ridgeline between c. 1155-1190 m (3800-3900 ft) elevation. We conducted banding sessions from the end of May into July during most years, using 4-30 nylon mist nets (12 x 2.5-m and 6 x 2.5-m, 36-mm mesh) placed at sites that have been used annually since 1992, primarily on the Amherst, Lakeview, and Long trails. Nets were generally opened from late afternoon until dark and from pre-dawn until noon on the following morning. Bicknell's Thrushes were captured both passively and through the use of vocal lures (recorded conspecific vocalizations), while other species were passively captured. Each individual was fitted with a uniquely-numbered U.S. Fish and Wildlife Service (USFWS) leg band and in some cases a unique combination of 3 plastic colored leg bands. We recorded data on age, sex, breeding condition, subcutaneous fat class, ectoparasites, flight feather wear, and net site of capture. Standard morphometrics recorded include wing chord, tail length, weight, tarsal length, culmen length, bill length from mid-nares, bill width, and bill depth. Several non-destructive tissue samples were collected from mist-netted Bicknell's Thrushes for studies of isotope markers and mercury burdens. During some years, we took a small blood sample (c. 50 μ l) from the brachial vein. Each sample was stored in a heparinized capillary tube, refrigerated in a vacutainer in the field, and frozen within 12-48 hours. A tail feather (rectrix #5) on both sides was clipped and stored in envelopes. Approximately 2 mm of claw tip from the middle toe of both feet was collected using sharp dissection scissors and deposited in a small paper envelope.

We estimated and compared adult survivorship according to methods described in Lebreton et al. (1992), Pradel et al. (1997) and Sillett and Holmes (2002), using the program MARK (White and Burnham 1999). We defined a candidate model set that included a fully parameterized global model and all reduced parameter models derived from the global model. We used mark-capture-recapture data from three mountains along a north-south latitudinal gradient: Gaspé in Quebec, Mt. Mansfield and Stratton Mountain in Vermont. We compared two habitats of interest at each Vermont site: those with abrupt linear edges from ski trails and work roads, and natural areas that contain only hiking trails. Because evidence suggested that individuals aged as second-year

often emigrate after their first breeding season, while older birds tend to have very high site fidelity, we used transient in our modeling which compares Φ and t in year +1 as one group (transients) and year 2+ as a second group. The model with the lowest Akaike Information Criterium (AIC) was accepted as the most parsimonious model for our data. Model comparisons within the candidate were made by deriving an index of plausibility using normalized Akaike weights (Burnham and Anderson 2002). The ratio between weights of any two models indicated the relative degree to which a particular model was better supported by the data.

Results and Discussion

In 2010 on Mt. Mansfield, we operated mist nets on 9 days between 3 June and 2 July, accumulating 576 net-hours, with an average of 64 net-hours per day. We had 123 bird captures and banded 70 new individuals of the five target species.

For our analysis of Bicknell's Thrush survivorship for data from 1993-2008, no models with year effects for recapture probability (p) were supported. As expected, p was almost always higher for males and on ski areas (Figure 1). The model with the lowest AICc in that candidate set was $\{\Phi(\text{trans}*\text{site}) p(\text{site}*\text{sex}*\text{hab})\}$, in which survivorship was lower for transients than for residents, lowest at Gaspé (the northernmost site), and very similar between Stratton and Mansfield (Table 1 and 2). There was little statistical support for ski areas affecting Bicknell's Thrush survivorship (Table 1). There was some support for annual variation in survivorship among sites (Table 1). Little congruence among study sites in annual survivorship (Figure 2) may have been caused either by local stochastic events or relatively high variance in annual estimates. These estimates represent survivorship from one breeding season to the next; because the species' annual cycle involves migration to and from the Caribbean, as well as a 6-month overwintering period, a detailed understanding of population connectivity will be necessary to fully elucidate demographic patterns and develop targeted conservation actions.

Acknowledgements

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Table 1. Ten highest-ranked models for determining survivorship of Bicknell's Thrush adults. There was little support for an effect of ski areas on survivorship, and only modest support for a year effect.

Model		AICc	Delta AICc	AICc Weight	Model Likelihood
{Phi(trans*site)	p(site*sex*hab)}	1691.0315	0	0.70549	1
{Phi(yearAD*trans*site)	p(site*sex*hab)}	1693.2752	2.2437	0.22976	0.3257
{Phi(trans*site*hab)	p(site*sex*hab)}	1697.4124	6.3809	0.02903	0.0411
{Phi(trans)	p(site*sex*hab)}	1698.1243	7.0928	0.02034	0.0288
{Phi(trans*hab)	p(site*sex*hab)}	1700.8521	9.8206	0.0052	0.0074
{Phi(trans*site*sex)	p(site*sex*hab)}	1701.1871	10.1556	0.0044	0.0062
{Phi(trans*sex)	p(site*sex*hab)}	1702.0659	11.0344	0.00283	0.004
{Phi(yearAD*trans*hab)	p(site*sex*hab)}	1703.2701	12.2386	0.00155	0.0022
{Phi(yearAD*trans)	p(site*sex*hab)}	1703.8779	12.8464	0.00115	0.0016
{Phi(trans*sex*hab)	p(site*sex*hab)}	1707.4001	16.3686	0.0002	0.0003

Table 2. Survivorship results derived from the best fitting model {Phi(trans*site) p(site*sex*hab)}.

Site	Stage	Survivorship Estimate	SE	Lower 95% CI	Upper 95% CI
Gaspe	Transient	0.38	0.07	0.25	0.53
Gaspe	Resident	0.48	0.07	0.35	0.61
Mt. Mansfield	Transient	0.51	0.05	0.42	0.60
Mt. Mansfield	Resident	0.61	0.03	0.55	0.68
Stratton Mountain	Transient	0.55	0.04	0.46	0.62
Stratton Mountain	Resident	0.71	0.03	0.64	0.77

Figure 1. Bicknell's Thrush recapture probabilities from the model $p(\text{site}*\text{sex}*\text{habitat})$.

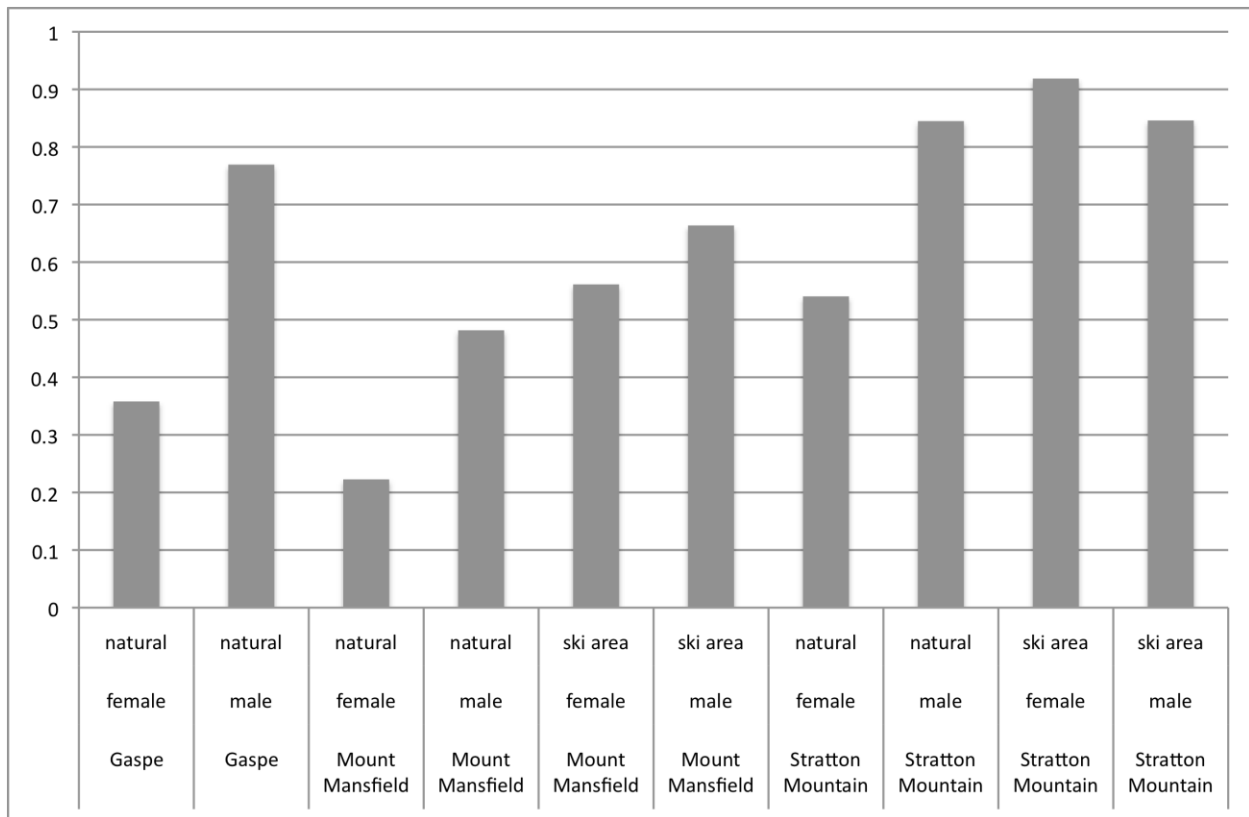
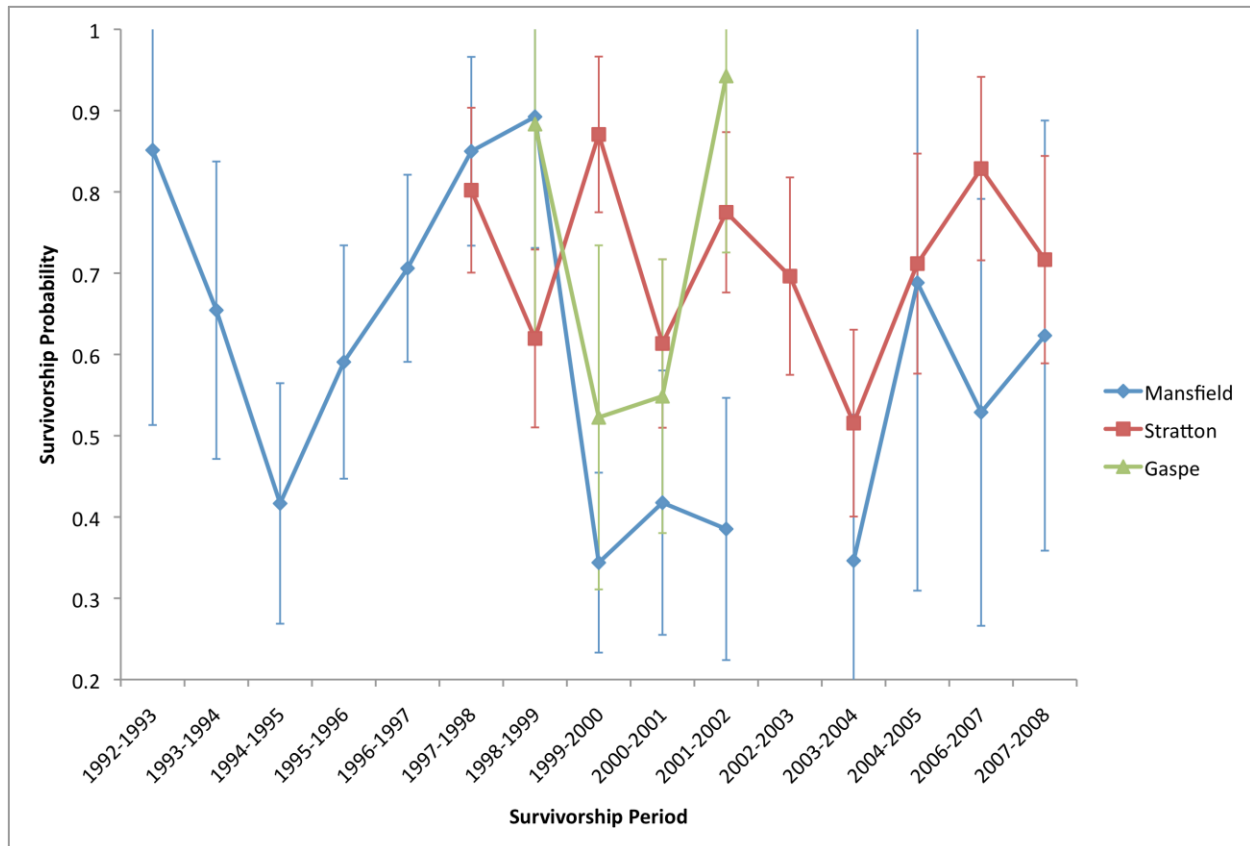


Figure 2. Annual survivorship estimates for Bicknell's Thrush adults from the model $\{\Phi(\text{year}*\text{trans}*\text{site}) p(\text{site}*\text{sex}*\text{hab})\}$. The data for period 2002-2003 were insufficient to allow an estimate for Mt. Mansfield.



Part II. Population Connectivity of Bicknell's Thrush: Insights from Geolocators.

Introduction

Natural selection acts on individual animals throughout the annual cycle, and events during each phase of the annual cycle likely influence subsequent events. For migratory animals, understanding these selection processes has been impossible because of our inability to follow individuals year-round and determine where breeding populations winter, where winter populations breed, as well as exact routes during migration. An understanding of these factors, which could operate during breeding and/or non-breeding periods to limit and ultimately determine bird abundance, is of urgent conservation concern. The most pressing need, and to date the most seemingly intractable problem, has been to determine the movement patterns and population connectivity of individuals between their breeding and wintering grounds. This is critical to understand how limiting factors (e.g. habitat destruction, climate change, etc.) operate in different parts of a species' annual cycle and to determine population size and local abundance.

Migratory connectivity is defined as the amount of population mixing between summer-breeding and winter-non-breeding, as well as the stop-over and migratory pathways between them (Webster et al. 2002). Although understanding space-use strategies and habitat preferences of Nearctic-Neotropical migratory songbirds has grown substantially over the last 20 years, knowledge of migratory connectivity remains poor (Remsen 2001, Marra and Webster 2005). Information from bird banding has been limited by the scattered and irregular nature of banding returns (Webster et al. 2002). Although stable isotope measures have provided latitudinal gradients of habitat occupancy and a useful tool to assess migratory connectivity (Rubenstein et al. 2002), this technique constitutes only a blunt instrument to assess migratory connectivity and can sometimes fail to provide clear, meaningful information (Hobson et al. 2001). Many migratory songbirds cross distances of over 5,000 km between breeding and wintering sites, yet the exact timing, pathways followed and migratory connectivity remain speculative.

The critical importance of migratory connectivity to understanding the fundamental biology of migratory birds has spurred a long history of research. Mark-recapture via bird-banding appears to have been the first and most broadly-used technique to address this issue for songbirds. However, despite banding millions of individual birds, there remains a relatively poor understanding of migratory connectivity for most species because of extremely low recapture rates (Webster et al. 2002). For example, only two Bicknell's Thrushes (*Catharus bicknelli*) have been recaptured on their Hispaniolan wintering grounds despite the banding of over 3,000 individuals throughout the species' North American breeding range (Rimmer and McFarland 2001, Townsend and Rimmer 2006). The recovery rate is much higher than that for other Nearctic-Neotropical songbird species, in which the numbers of individuals marked in one portion of the migratory range and recaptured in another is exceedingly small.

One of the most studied and publicized conservation problems in the past 20 years has been the population decline of Nearctic-Neotropical migratory songbirds (e.g., Robbins et al. 1989, Askins et al. 1990). Between 1993 and 2003, a core breeding population of Bicknell's Thrush in the White Mountain National Forest was estimated to number as few as 4,900 individuals (Hale 2006) and experienced annual declines of 7% per year along 40 survey routes (King et al. 2008, Lambert et al. 2008). Seven years of data from New Brunswick and Nova Scotia (2002-2008) indicate that Bicknell's Thrushes declined along established survey routes by as much as 20% annually (Campbell et al. 2007, Whittam and Campbell unpubl. data), while annual surveys at Mont Gosford, Quebec, from 2001-2007 showed a pronounced decline in the number of stations occupied by Bicknell's Thrush (Aubry unpubl. data). In addition, climate change projections derived from survey data indicate that suitable Bicknell's Thrush habitat may be lost from the U.S. following increases in summer temperatures that are projected to occur this century (Rodenhouse et al. 2008). Yet, incomplete information on fundamental aspects of the species' ecology and demographics throughout its annual cycle preclude a full understanding of the limiting factors that underlie these striking declines.

Understanding the timing and extent of avian population limitation and regulation is complicated in the case of migratory populations that spend different periods of their annual cycle in ecologically disparate regions. The "seasonal interaction hypothesis" was first put forth by Fretwell (1972), who argued that breeding density is determined by winter survival, which in turn is related to events that occur during the breeding cycle. Recent studies of American Redstarts (*Setophaga ruticilla*) support the seasonal interaction hypothesis (Marra et al. 1998, Norris et al. 2003). In this species, winter habitat quality determined physical condition and timing of spring migration departure, which influenced arrival date and physical condition on the breeding grounds. The quality of each individual's winter habitat was determined by measuring stable carbon isotope levels shortly after arrival on breeding territories. Subsequent monitoring of redstart breeding demographics revealed a profound interaction between seasons. Robust tests of the seasonal interaction hypothesis such as this require detailed knowledge of migratory connectivity.

Bicknell's Thrush is among eastern North America's most rare, range-restricted breeding passerines. Considered one of the Nearctic-Neotropical migrants at greatest risk of extinction and thus of highest continental conservation concern (Pashley et al. 2000, Rimmer et al. 2001, Rich et al. 2004, Wells 2007), Bicknell's Thrush is classified as globally "vulnerable" by the International Union for the Conservation of Nature (IUCN). At both ends of its migratory range, Bicknell's Thrush occupies a limited, highly fragmented distribution and faces multiple habitat threats. These include climate change (Rodenhouse et al. 2008), acid ion deposition (Johnson et al. 1992, Hames et al. 2002), mercury contamination (Rimmer et al. 2005), mountaintop development (Rimmer et al. 2001, 2004), forestry operations (Leonard and Chisholm 2008,

Gardiner 2006), and winter habitat loss and degradation (Rimmer 2005; Rimmer et al. 2001, 2005). With respect to the latter, Marshall (2001) postulated that loss of breeding populations at several sites in Nova Scotia was directly related to extensive deforestation of wintering habitat in Haiti, implying strong population connectivity. Overall, documented population declines in Canada and New Hampshire, combined with severe and ongoing habitat loss on Hispaniola, have heightened concern about the conservation status of Bicknell's Thrush. This concern catalyzed the 2007 formation of the International Bicknell's Thrush Conservation Group (IBTCG; see www.bicknellsthrush.org). To date, the IBTCG has identified elucidation of migratory connectivity as one of the species' most pressing conservation needs.

Breeding Distribution

Breeding from the northern Gulf of St. Lawrence and easternmost Nova Scotia southwest to the Catskill Mountains of New York, Bicknell's Thrush is estimated to number fewer than 100,000 individuals across its naturally fragmented breeding range (Rimmer et al. 2001). This range has been well documented (Wallace 1939, Ouellet 1993, Atwood et al. 1995, Rimmer et al. 2001), and recent habitat modeling has provided further refinements (Figure 1; Lambert et al. 2005, Aubry et al. 2009).

Numerous local Bicknell's Thrush extirpations have been documented. Historic breeding populations disappeared on Mt. Greylock, Massachusetts (10 pairs in 1950s, 0 in 1973; Veit and Petersen 1993); Magdalen Is., Quebec (Ouellet 1996, D. McNair pers. comm.); Seal and Mud Is., Nova Scotia (Wallace 1939, Erskine 1992, D. Busby pers. comm.); Cape Forchu, sw. Nova Scotia (J. Marshall pers. comm.); Fundy National Park, New Brunswick (Christie 1993); and Grand Manan I., New Brunswick (B. Dalzell pers. comm.). Further range contraction in the Canadian Maritime provinces is suggested by mid-1990s surveys that showed fewer occupied sites than during the 1986–1991 Breeding Bird Atlas (D. Busby pers. comm.) survey. The species' core U.S. range, however, appears to have remained stable overall, as Bicknell's Thrush was confirmed on 63 of 73 historic (pre-1992) breeding sites surveyed in 1992–1995 (Atwood et al. 1995).

Non-breeding Distribution

The stationary portion of the species' non-breeding period is believed to be confined to the Greater Antilles, mostly in mesic to wet broadleaf forest. Specimen and field-survey data indicate the bulk of wintering Bicknell's Thrushes occur in the Dominican Republic (Wetmore and Swales 1931, Ouellet 1993, Rimmer et al. 1997, 1999), where the species is widely distributed and locally common from sea level to 2,220 m (Rimmer et al. 1999, 2001). There are few records from Haiti where known populations are restricted to montane forest fragments in the southwest (Massif de la Hotte) and east (Massif La Visite; Wetmore and Swales 1931; Woods and Ottenwalder 1983, 1986; Rimmer et al. 2005, 2010). Bicknell's Thrush appears to be uncommon and local in Jamaica, known only from the Blue Mtns. at 1,200–2,225 m elevation (R.

and A. Sutton unpubl.; Rimmer unpubl.). The species is a rare winter resident in e. and se. Puerto Rico, in the Luquillo Mountains at 450–720 m elevation and Sierra de Cayey at 720 m (Arendt 1992, J. Wunderle unpubl.), and it has recently been reported from Vieques island off the east coast (S. Colon, pers. comm.). Thrushes have been found in e. Cuba at 1,600–1,960 m in Sierra Maestra (Rompré et al. 2000, Y. Aubry and G. Rompré pers. comm.). There are no confirmed winter records elsewhere. The remote locations and rough terrain of occupied winter habitats make it unlikely that we have yet fully documented all areas inhabited by Bicknell's Thrush. However, recent modeling of Bicknell's Thrush winter grounds has proven to be a robust predictor of its habitat in the Greater Antilles (VCE unpub. manuscript).

Migratory Routes

Analysis of the scant specimen and banding data, using wing-chord as identification criterion (<94 mm = Bicknell's, >98 mm = Gray-cheeked), suggests an elliptical southern portion of migratory route between the North American breeding grounds and Greater Antillean winter range (Rimmer et al. 2001). Most southbound migrants may depart the East Coast from the Mid-Atlantic States or Carolinas on an overwater flight to the Greater Antilles, as suggested by the scarcity of fall records south of Virginia. Northward passage appears to be more concentrated through the Southeast coast, as spring specimens from Florida, Georgia, both Carolinas, and Virginia outnumber fall records nearly 2 to 1. The entire migration in both directions appears to be concentrated east of the Appalachian Mountains. Stopover lengths during migration are not well documented by banding stations, but a few transients appear to linger at stopover sites during the fall. There is yet no evidence of spring stopovers.

No information exists on the timing of departure from the wintering grounds or the rate of spring or fall migration. We suspect that birds depart wintering areas in late April or early May, as individuals are still present in the Dominican Republic second week of April (VCE unpub. data) and there are no verifiable U.S. records prior to May. Furthermore, given recent data from Wood Thrushes (Stutchbury et al. 2009), we suspect the rate of migration to be about 200-250km/day. The northward migratory route of Bicknell's Thrush and its spring stopover locations largely remain a mystery.

In this study we used miniature daylight level data recorders (geolocators), recently developed by the British Antarctic Survey (Afanasyev 2004), to track locations of individual Bicknell's Thrushes during their annual cycle to help elucidate migratory connectivity between breeding, stopover and winter sites.

Methods

Study Sites and Geolocators

During the 2009 breeding season, we captured adult male Bicknell's Thrushes in nylon mist nets (12 x 2.5-m and 6 x 2.5-m, 36-mm mesh) at seven study areas across the entire breeding range, using passive netting and playback of conspecific vocalizations to attract birds. Each individual was fitted with a uniquely-numbered Canadian Wildlife Service / U.S. Fish and Wildlife Service leg band and in some cases a unique combination of three plastic colored leg bands. We attempted to recover and deploy additional geolocators during the 2010 breeding season at each site. During February and March of 2010, we added a south-north component to this study by attaching geolocators to overwintering thrushes at two ecologically distinct sites in the Dominican Republic. One was at Pueblo Viejo in Sierra de Bahoruco (a strongly male-dominated forest habitat), while the second was at Loma Guaconejo in Cordillera Septentrional (a mid-elevation, moderately disturbed site with even sex ratios).

We deployed Mk10S light level geolocators developed and manufactured by the British Antarctic Survey (BAS). Light sensors were mounted on a stalk ~15mm in length and at 20 degrees to horizontal to better clear plumage. These loggers take consistent readings of daylight timing for up to two years. The recovered data are then interpreted to determine latitude and longitude of the individual bird for every day the logger was attached and exposed to suitable sunlight.

We attached geolocators to Bicknell's Thrushes using a leg-loop backpack harness (Rappole and Tipton 1991) that has been deployed successfully on over 250 Bicknell's Thrushes with 1.2 g radio transmitters, as well as on many other passerine species. During 2009 and winter 2010, we used size 600 Kevlar thread for the harnesses, while in June and July of 2010 we switched to 3/16-inch Teflon ribbon (Stutchbury et al. 2009). Both methods resulted in geolocators being attached without logistic problems of any kind or any detectable short-term effects on thrushes. Several birds recaptured up to four weeks after attachment were in good condition and within normal weight ranges. The total attachment weighed ~1.4 g which corresponded to ~5% mean male body mass (28.18 g; Rimmer et al. 2001).

Light level data were downloaded following recovery of geolocators, and data were examined for clock drift. Very little or no clock drift was detected (i.e., drift <2 minutes), so we did not apply linear drift corrections. Mk10S geolocators use a sensor to measure light intensity every minute and record the maximum measurement in each 10-min interval. We used a light level threshold of two to define light transition events because of the mountainous terrain and heavy shading encountered by Bicknell's Thrush. We used a 2-week post deployment period as calibration data for the breeding grounds. Additionally, we used a 1-week deployment of a fixed geocator placed in the forest understory on a Bicknell's Thrush territory at Pueblo Viejo as calibration data for wintering grounds (mean sun elevation angle = -2.9). The annual cycle was divided into four seasons: breeding (June – August), fall migration (Sept.- Oct.), winter (Nov. – March), and spring migration (April – May). Exact seasonal transitions are not known for this

bird, but we believed this to be a conservative estimate of annual timing. We used mean breeding sun elevation data for birds released on winter grounds and the above winter sun elevation estimate for those released on the breeding grounds. Sun elevation estimates for migration periods are problematic. We used the winter estimate of sun elevation because we believed that Bicknell's Thrush likely inhabited heavily shaded forest understory during migration stopover, while on the breeding grounds birds often call and sing from elevated perches during transition events.

For each recovered geolocator, light transition events were visually assessed and assigned a confidence level using program TransEdit (BAS). Non-linear or apparent shading events during the transition periods were scored lower. Data for 15 days before and after spring and fall equinox were excluded from latitude calculations.

Geolocator light levels are referenced to an internal clock/calendar and from these the Greenwich Meridian Time (GMT) of local sunrise and sunset are estimated. The GMT midway between the times of sunrise and sunset is local noon GMT and local midnight GMT, from which longitude can be determined. The day length on a particular date determines the latitude. Longitude tends to be much more precise than latitude. Location estimates were calculated and plotted with BirdTracker software (BAS). Further analyses were completed in ArcGIS 10.

We derived a mean and 95% confidence interval for longitude using all winter locations and examined the intersection with modeled Bicknell's Thrush winter habitat to derive an estimated wintering location for each individual. We used only high confidence transition events when attempting to derive locations using both latitude and longitude throughout the annual cycle.

Results and Discussion

During 2009, VCE and collaborators from the Smithsonian Migratory Bird Center (SMBC), SUNY Environmental School of Forestry, Canadian Wildlife Service, Bird Studies Canada, and University of New Brunswick deployed geolocators on 70 Bicknell's Thrushes: 36 in the U.S. and 34 in Canada (Table 1). During the winter of 2009-10 we attached geolocators to 15 individuals at Pueblo Viejo and 13 at Loma Guaconejo (Table 1).

Overall return rates of geolocated birds from 2009 were lower than expected, 30% versus the long-term average of 65% for Vermont-banded adults. Recoveries of geolocators themselves were low ($n = 17$). We discovered that the Kevlar thread used to secure the 2009 backpack harnesses had abraded and broken – 2 of the 3 birds recovered with intact geolocators had at least some broken strands of Kevlar fibers. However, in Quebec, where Dacron fly fishing line was used for the harnesses, all 8 recovered birds still had fully intact geolocators.

Table 1. Numbers of geolocators attached and recovered on Bicknell's Thrushes across the migratory range, 2009 and 2010.

Site	Deployed		Recovered in 2010	Recaptures missing 2009 geolocators
	2009	2010		
Plateau Mt. (NY)	5	0	0	–
Slide Mt. (NY)	17	0	1	4
Mt. Mansfield (VT)	14	22	2	4
Mt. Washington (NH)	0	4	–	–
Mt. Jefferson (NH)	0	4	–	–
Mt. Gosford (QC)	10	0	3	0
Massif-du-Sud (QC)	6	0	3	0
Gaspesie (QC)	9	0	2	0
New Brunswick Highlands	5	7	0	–
Cape Breton Highlands (NS)	4	7	2	1
Pueblo Viejo (DR) ^a	15	0	2	2
Loma Guaconejo (DR) ^a	13	0	2	0
Total	98	41	17	11

^a geolocators attached in February and March of 2010, recovered in November

During November of 2010, we returned to both study sites in the Dominican Republic for geocator recovery. At Pueblo Viejo, we recaptured 4 of the 15 birds to which we had attached geolocators in the previous February or March; 2 of these retained their geolocators. At Loma Guaconejo, both recovered thrushes still had their geolocators securely attached.

During the summer of 2010, in addition to recovering 2009 geolocators, we concentrated on attachment of new devices. We used Teflon webbing to attach geolocators to 30 Bicknell's Thrushes: 17 ASY male and 5 ASY female Bicknell's Thrushes on Mt. Mansfield, 4 ASY males on Mt. Washington, and 4 ASY males on Mt. Jefferson. Additionally, 14 geolocators were deployed by our cooperators in New Brunswick and Nova Scotia.

The rangewide total of 17 recovered geolocators during 2010, while falling short of expectations, provides an invaluable store of data. However, analyses and interpretation of the retrieved data have proven extremely challenging to date. The data showed extreme shading for much of the year, indicating that individuals are often under heavy forest canopy cover during daily sun transitions. After initial tests and examinations of data, we have been able to partially analyze five geolocators thus far (Nova Scotia = 2, Mt. Mansfield = 2, and Slide Mountain, NY = 1).

We have been able to complete analysis using only high confidence transition events for one male from Mt. Mansfield, Vermont. Geolocator data yielded few reliable locations due to heavy shading events (Fig. 1). During fall migration between 19-26 October 2009, this bird stopped in the eastern Bahamas for approximately six days. During northward spring migration it stopped over in the western Bahamas or southeast Florida from 6-8 May 2010. On 15-16 May the bird progressed northward to the coastal Carolinas. It arrived at its Mt. Mansfield breeding site on 24 May, and we recaptured it on 2 June. While on migration from 8-15 May from a presumed position in southeast Florida to coastal South Carolina, this bird traveled 850 km (121 km/day). It then traveled 1,360 km at a rate of 170 km/day from 16-24 May to arrive on the breeding grounds. Stutchbury et al. (2009) reported that spring migrating Wood Thrushes traveled 242 km/day from winter to breeding sites (range of 5 birds: 159-279 km/day). This Bicknell's Thrush migrated nearly twice as slowly, covering only 123 km/day from its apparent arrival in southeastern Florida to its breeding site. The general migratory pathway of this individual helps to confirm the pattern previously documented from migration banding records, nocturnal migration calls, tower kills and other reliable sightings (Rimmer et al. 2001), indicating that fall migrants tend to pass from coastal Virginia and North Carolina across open waters and through the eastern Bahamas to the wintering grounds, while spring migration appears to be through eastern Florida and northward an inland route to the breeding grounds.

We used winter longitude locations for all five birds to determine approximate wintering locations (Table 2, Fig. 2). All four birds were predicted to be in the Dominican Republic based on the mean longitude location. However, significant variation in longitude locations places confidence intervals to Cuba and Puerto Rico. We hope to make significant improvements by filtering heavily shaded dates.

Table 2. Longitude location estimates for geolocators on Bicknell's Thrush during the winter period (November – March).

Geolocator	Mean	SD
Catskills, NY	-69.93772	-1.03
Mt. Mansfield, VT	-70.20977	-2.16
Nova Scotia (7533)	-71.76242	-1.75
Nova Scotia (7618)	-68.92296	-2.91

Future Work

During the 2011 breeding season, VCE staff and collaborators will undertake intensive efforts to recover geolocators in Vermont, New Hampshire, Nova Scotia, and New Brunswick. With the improved attachment method used in 2010, we expect to recover ~20 individuals.

We will continue to refine analytical methods to improve interpretation of data, and we are optimistic about achieving more results from the four Dominican Republic recoveries given the behavior of the bird during the breeding season at daily transition times. We will combine geolocator data from both 2010 and 2011 for a comprehensive analysis, which we expect to result in a peer-reviewed paper by the end of 2011.

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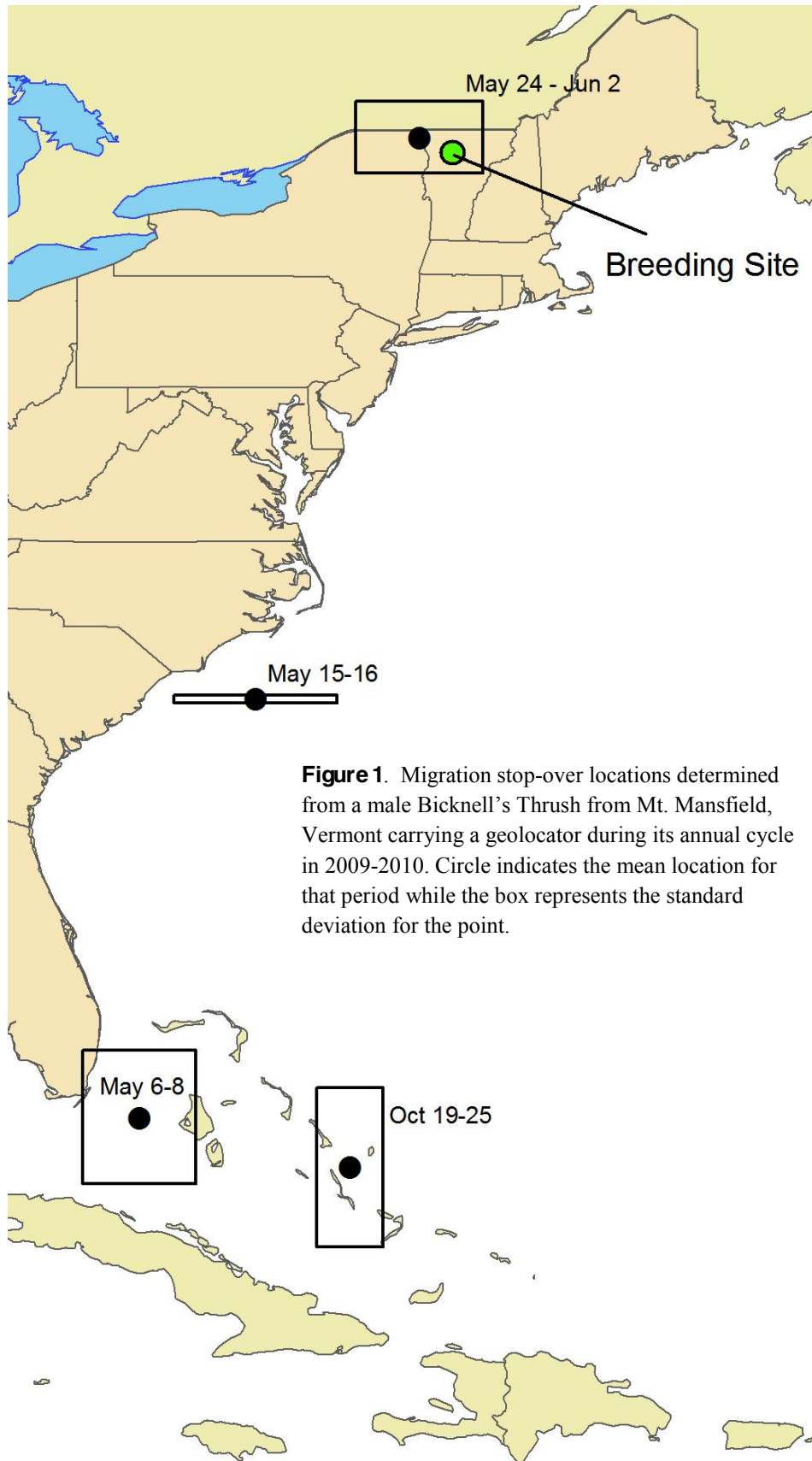


Figure 1. Migration stop-over locations determined from a male Bicknell's Thrush from Mt. Mansfield, Vermont carrying a geolocator during its annual cycle in 2009-2010. Circle indicates the mean location for that period while the box represents the standard deviation for the point.

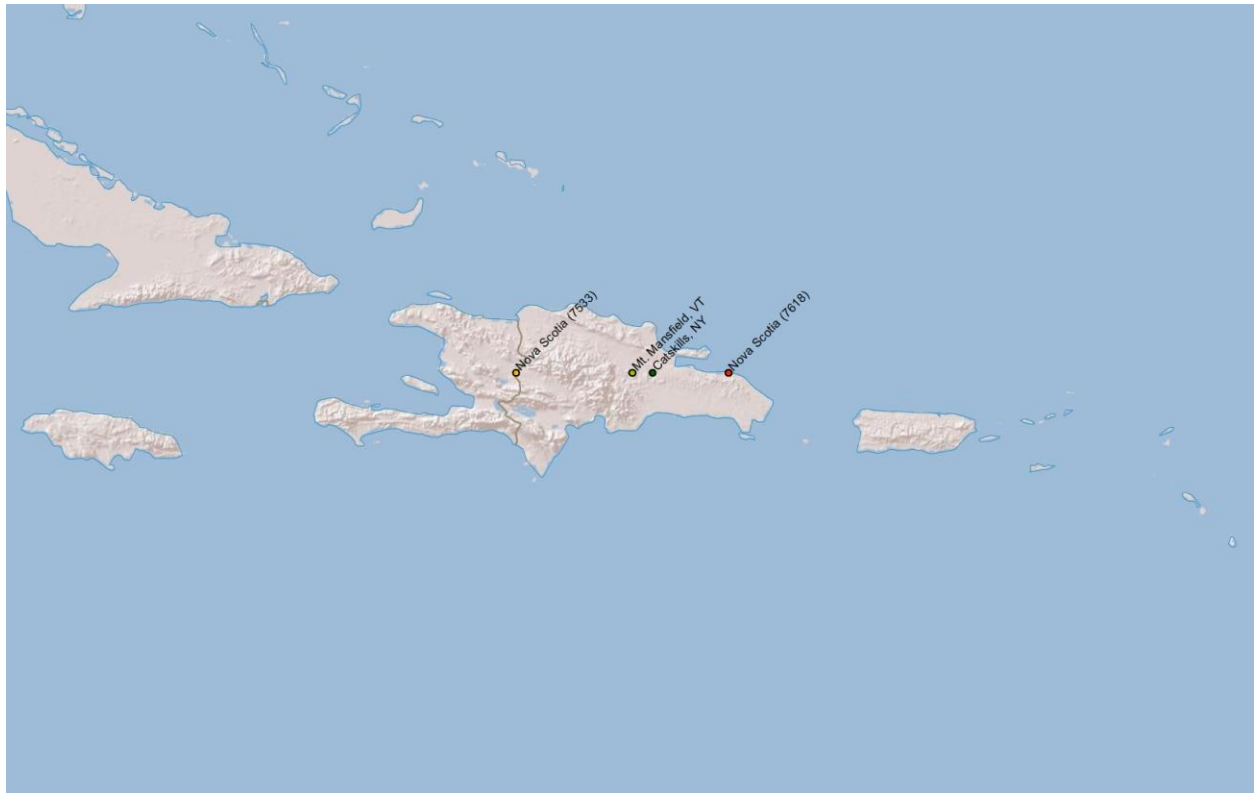


Figure 2. Mean of winter longitude estimates from Bicknell's Thrush geolocators. Each bird could have been located along any portion of the longitude and the associated error of that mean.

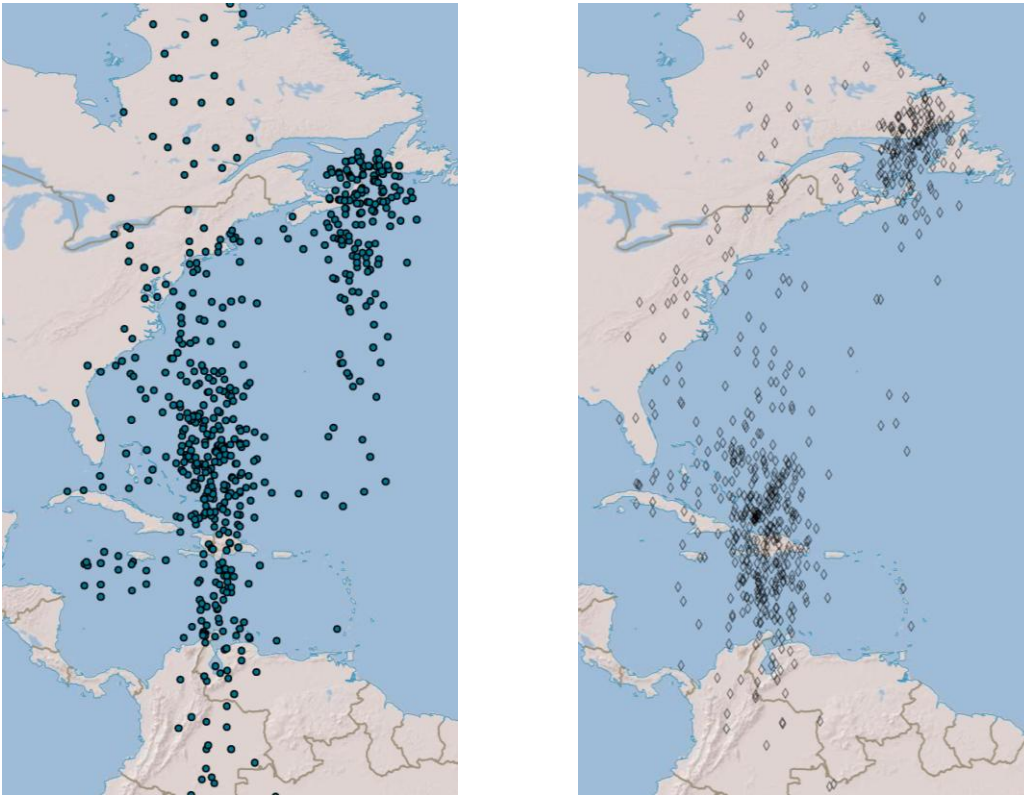


Figure 3. Geolocator from Nova Scotia Bicknell's Thrush using the mean sun elevation from summer data (left) compared to the winter elevation from a test geolocator in the Dominican Republic (left).

Part III. A Dynamic Occupancy Model of Bicknell's Thrush Breeding Habitat

Introduction

In previous studies (Lambert et al. 2005, Hart et al. 2009, McFarland and Rimmer 2009), we used the Quantreg library in R software (<http://lib.stat.cmu.edu/R/CRAN>) to estimate the 0.05 quantile regression (Cade and Noon 2003) of elevation as a linear function of latitude for locations where Bicknell's Thrush (*Catharus bicknelli*) were found during targeted surveys. This produced an elevation threshold above which 95% of the detections occurred. We converted the linear threshold into an elevation mask, formed as a raster data set of 30 x 30 m cells (Lambert et al. 2005) or 90 x 90 m cells (Hart et al. 2009, McFarland and Rimmer 2009). The elevation mask was then placed over a digital elevation model (DEM) of the northeastern U.S. Summits, ridgelines, and slopes emerged above the mask as a vast complex of high-elevation habitat units predicted to be occupied by Bicknell's Thrush during the breeding season.

In this study we extended our model using detection-non detection (presence-absence) data collected from 2001-2009 for Bicknell's Thrush across its U.S. breeding range through a volunteer-based monitoring project, Mountain Birdwatch (Hart and Lambert 2007) and Forest Bird Monitoring Program survey routes (see section IV).

Methods

We used detection-non detection (presence-absence) data collected from 2001-2009 across the U.S. breeding range of Bicknell's Thrush through the volunteer-based Mountain Birdwatch (MBW) project (Hart and Lambert 2007). We removed all sites with fewer than six years of data for a final sample size of n=113. Each site consisted of a ~1 km transect of five points 200-250m apart. Transect locations were placed through prioritization of high-elevation habitat units, although trail locations and volunteer availability ultimately determined which sites were to be surveyed and where at each site the transect was placed (Hart and Lambert 2007).

As many as four surveys were conducted at each site in each year, although each site was not always surveyed all four times nor surveyed in all years. These missing observations did not affect our analysis, as the parameters associated with a missed survey were simply not estimated using that survey's data (MacKenzie et al. 2006). The four detection/non-detection surveys conducted at each site included: 1) the evening before a morning point count; 2) a 10-minute morning (between 4:30-6:30 h EST) point count at each of the five points along the transect; 3) a one-minute playback followed by a two-minute listening period at each of the five points along the transect immediately following the point count surveys; and 4) a follow-up playback survey every 100 m along the transect within two weeks of the previous surveys and/or before 15 July.

For each survey at each site, a detection or non-detection was recorded based on whether a Bicknell's Thrush was heard or seen at or between any of the five points along the transect.

A multi-season robust occupancy model framework was used, following MacKenzie et al. (2003) to estimate the probability of extinction and colonization. The robust model examines the state variable, occupancy, and how it changes over time at the site level. The four parameters estimated in this model are probability of detection (ρ), probability of initial site occupancy (ψ), probability of site colonization (γ), and probability of site extinction (ϵ). Analyses were conducted using program Presence (Hines 2006). Detection probability (ρ) was modeled as a function of survey number.

We determined Bicknell's Thrush breeding habitat across the northeastern U.S. using our predictive model (Hart et al. 2009, McFarland and Rimmer 2009) and the National Land Cover Database 2001 (<http://www.mrlc.gov/>; Homer et al. 2004). Raster cells that were defined as conifer within areas identified as potentially containing Bicknell's Thrush were considered Bicknell's Thrush breeding habitat.

We calculated the amount of habitat within a radius of 1, 5, 10, and 25 km around MBW sample points using 'isectpolrst' command in Geospatial Modeling Environment (<http://www.spatial ecology.com/gme/>) in ArcGIS 9.3.1. We used 'zonal statistics' (command v.what.rast) in GRASS GIS 6.4 to assign a value for patch size, habitat within 1, 5, 10, and 25 km to each sample site. We standardized values for patch size, habitat within 1, 5, 10, and 25 km to obtain z-scores [(x-mean)/standard deviation].

Because our dataset was relatively sparse due to missing surveys, we kept candidate models comparatively simple. We selected 19 models *a priori* to compare the importance of patch size, habitat within 1, 5, 10, and 25 km in explaining the variability of three population parameters: 1) probability of initial site occupancy (ψ_1); 2) local site colonization (γ); and 3) local site extinction (ϵ). We used Program Presence 2.4 to run 19 dynamic occupancy models and rank them each by their AIC score. We then obtained the beta (describing intercept and effect size of habitat within 1 km) and parameter estimates and standard error for each site based on the highest ranked model. Maximum likelihood techniques were used to estimate parameters, where ψ_1 refers to the initial occupancy in the first period and ϵ and γ determine ψ_t in the following seasons (MacKenzie et al. 2003). A total of 19 models were compared using AIC model selection procedures (Burnham and Anderson 2002). Models with an AIC value within 4.0 of the minimum AIC were considered plausible models (Burnham and Anderson 2002).

We used command r.neighbors in GRASS GIS 6.4 to calculate the amount of habitat within 1 km for each raster cell in the Bicknell's Thrush breeding habitat model. We then used GRASS GIS 6.4 to calculate the probability of occupancy, colonization, and extinction for each cell within the

US distribution model based on the amount of habitat within 1 km of each cell and beta estimates from the highest ranked model.

Results and Discussion

The amount of habitat within a 1-km radius was a prominent factor in determining whether a site occupied by Bicknell's Thrush became locally extinct or a vacant site was colonized (Table 1). The highest ranked model, $\psi(\text{hab1km})$, $\gamma(\text{hab1km})$, $\varepsilon(\text{hab1km})$, $p(\text{survey})$, was >6 AIC points lower than the next model. The top model expressed both γ and ε as a function of the effect of the amount of habitat within a 1-km radius. Initial occupancy (ψ_1) was a function of the amount of habitat within a 1-km radius in the top 10 models.

There were 192.7 ha of habitat within 1 km of the Mt Mansfield ridgeline survey route (MANS) and 167.7 ha around the Ranch Brook survey route (RABR). Based on the highest ranked model, the initial occupancy was 0.94 for MANS and 0.91 for RABR (Fig. 1). Extinction and colonization probabilities were 0.01 and 0.43 for MANS and 0.02 and 0.39 for RABR (Figs. 2 and 3). Only the smallest and most isolated peaks within the U.S. breeding range of Bicknell's Thrush had an extinction probability above 0.2 (Fig. 3).

The U.S. breeding range of Bicknell's Thrush contained 173,354.3 ha of habitat. Of this total, Maine had 61,353.9 ha (35.4%), New Hampshire 64,696.2 ha (37.3%), Vermont 12,820.3 ha (7.4%), and New York 34,471.8 ha (19.9%). Massachusetts, where the species has been extirpated since the early 1970s (Rimmer et al. 2001), had just 12.06 ha (0.01%).

Bicknell's Thrush habitat occupancy involves a complexity of factors with intricate links between landscape and local scale features of the habitat (Frey 2008). Simply preserving large tracts of habitat may not be sufficient to ensure future persistence, but could minimize local extinction risk. Because it is well known that Bicknell's Thrush is a natural disturbance, mid-succession specialist (Rimmer et al. 2001), careful consideration of local habitat attributes such as natural disturbance regimes or anthropogenic uses and management that closely mimic these processes will be vital to the continued persistence of Bicknell's Thrush.

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Table 1. AIC model selection results for determining the effects of patch size, habitat within 1, 5, 10, and 25 km of survey site on initial occupancy (ψ), colonization (γ), and local extinction (ϵ). Detection probability (p) was modeled as a function of survey number for all models. Each model is ranked by its AIC score, which represents how well the model fits the data. A lower delta AIC value is indicative of a better model. Only models within 4 AIC points of the top model were considered plausible. The probability that the model (of the models tested) would best explain the data is indicated by AIC weight. Model likelihood is the ratio of each model's AIC weight to the top model's AIC weight.

Model	AIC	delta AIC	AIC weight	Model Likelihood	Parameters
psi(hab1km),gamma(hab1km),eps(hab1km),p(survey)	1252.23	0	0.9528	1	10
psi(hab1km),gamma(hab1km),eps(hab5km),p(survey)	1258.25	6.02	0.047	0.0493	10
psi(hab1km),gamma(hab1km),eps(ps),p(survey)	1268.69	16.46	0.0003	0.0003	10
psi(hab1km),gamma(hab1km),eps(hab10km),p(survey)	1275.5	23.27	0	0	10
psi(hab1km),gamma(hab1km),eps(.),p(survey)	1288.1	35.87	0	0	9
psi(hab1km),gamma(hab1km),eps(hab25km),p(survey)	1288.9	36.67	0	0	10
psi(hab1km),gamma(hab5km),eps(.),p(survey)	1293.2	40.97	0	0	9
psi(hab1km),gamma(ps),eps(.),p(survey)	1293.88	41.65	0	0	9
psi(hab1km),gamma(hab10km),eps(.),p(survey)	1295.01	42.78	0	0	9
psi(hab1km),gamma(.),eps(.),p(survey)	1295.61	43.38	0	0	8
psi(hab1km),gamma(hab25km),eps(.),p(survey)	1296.6	44.37	0	0	9
psi(ps),gamma(.),eps(.),p(survey)	1304.01	51.78	0	0	8
psi(hab5km),gamma(.),eps(.),p(survey)	1304.91	52.68	0	0	8
psi(hab1km),gamma(year),eps(year),p(survey)	1306.05	53.82	0	0	22
psi(hab10km),gamma(.),eps(.),p(survey)	1315.24	63.01	0	0	8
psi(hab25km),gamma(.),eps(.),p(survey)	1319.56	67.33	0	0	8
psi(.),gamma(.),eps(.),p(survey+ps)	1321.39	69.16	0	0	8
psi(.),gamma(.),eps(.),p(survey)	1323.22	70.99	0	0	7
psi(.),gamma(.),eps(.),p(.)	1374.14	121.91	0	0	4

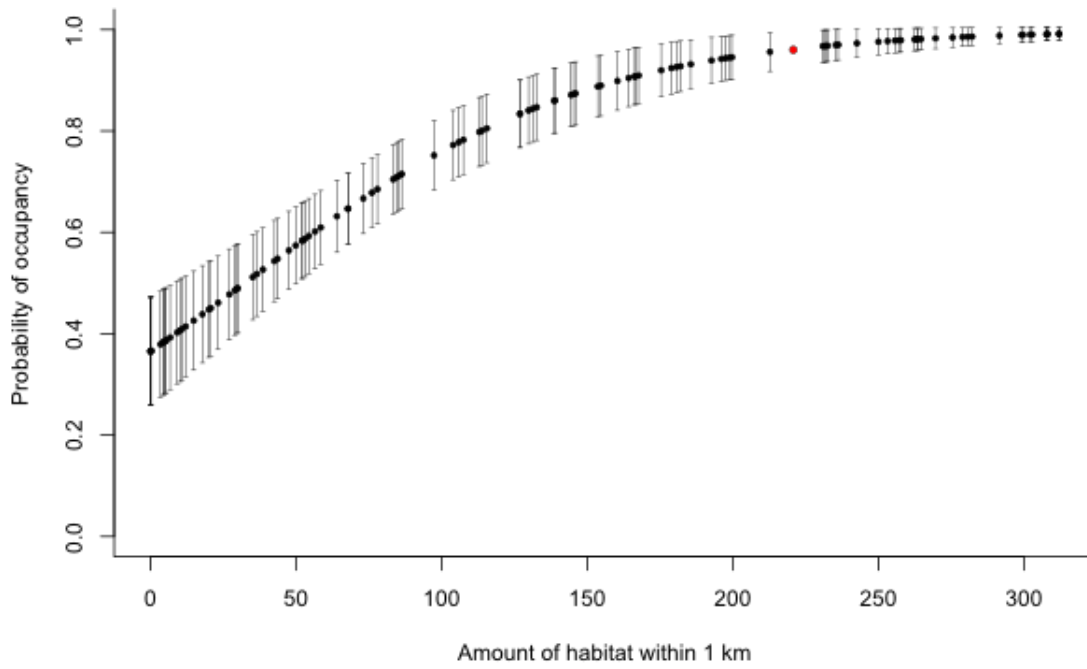


Figure 1. Estimated probability (± 1 SE) of initial occupancy (ψ) as a function of the amount of habitat (ha) within a 1-km radius from the top ranked model: $\psi(\text{hab1km}), \gamma(\text{hab1km}), \epsilon(\text{hab1km}), p(\text{survey})$.

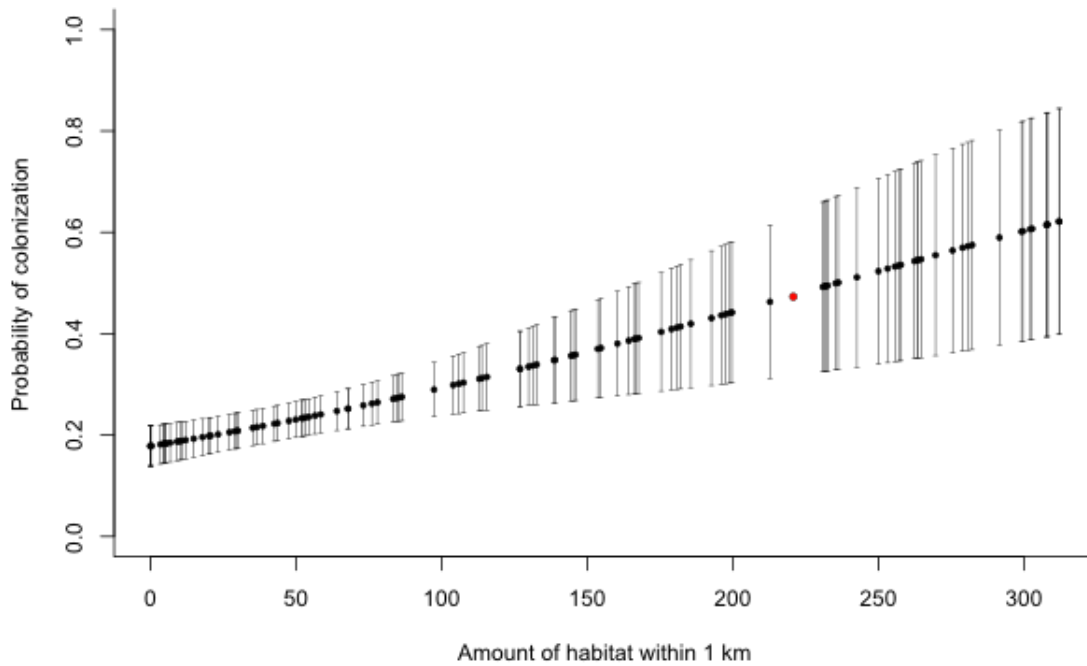


Figure 2. Estimated probability (± 1 SE) of local site colonization (γ) as a function of the amount of habitat (ha) within a 1-km radius from the top ranked model: $\psi(\text{hab } 1\text{km}), \gamma(\text{hab } 1\text{km}), \epsilon(\text{hab } 1\text{km}), p(\text{survey})$.

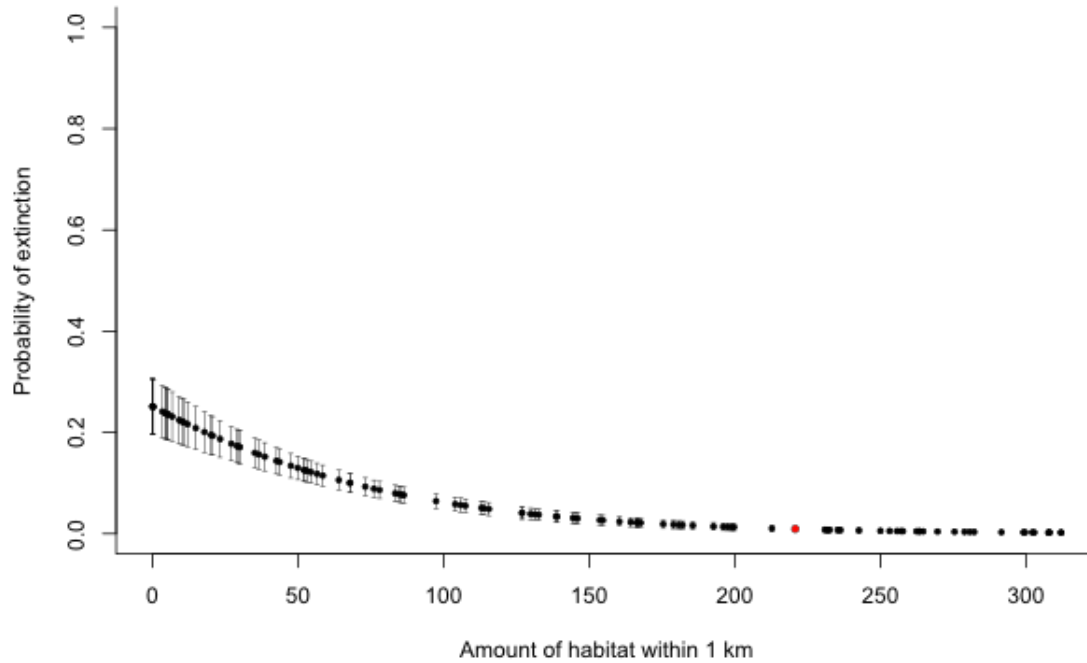


Figure 3. Estimated probability (± 1 SE) of local site extinction (ϵ) as a function of the amount of habitat (ha) within a 1-km radius from the top ranked model: $\psi(\text{hab } 1\text{km}), \gamma(\text{hab } 1\text{km}), \epsilon(\text{hab } 1\text{km}), p(\text{survey})$.

Part IV. Forest Bird Surveys on Mt. Mansfield and Lye Brook Wilderness Area

In 2010, breeding bird surveys were continued at 3 permanent study sites on Mt. Mansfield, and on a single site at the Lye Brook Wilderness Area (LBWA) of the Green Mountain National Forest. The Mt. Mansfield ridgeline was surveyed for the 20th consecutive year, while the Ranch Brook site was censused for the 15th year in 2010 (the 2004 survey was not completed due to inclement weather on attempted survey dates). Our permanent study site at Underhill State Park was surveyed for the 18th year in 2010 (the site was not surveyed in 2003 or 2005). The LBWA was surveyed for the 11th consecutive year in 2010.

The Underhill State Park site consists of mature northern hardwoods ranging from 609 to 731 m (2000 to 2400 ft) elevation, while the Mansfield ridgeline site, at 1158 m (3800 ft), consists of montane fir-spruce. The Ranch Brook site ranges between 975 and 1097 m (3200 and 3600 ft), and is dominated by a paper birch-fir canopy. The Lye Brook study site, located in Winhall, just north of Little Mud Pond, is characterized by mature northern hardwoods at an elevation of 701 m (2300 ft).

These four study sites are part of VCE's long-term Forest Bird Monitoring Program (FBMP). This program was initiated in 1989 with the primary goals of conducting habitat-specific monitoring of forest interior breeding bird populations in Vermont and tracking long-term changes (Faccio et al. 1998). As of 2008, VCE had established 39 monitoring sites in 9 different forested habitats in Vermont, with additional montane sites in New York, New Hampshire, Maine, and Massachusetts. A complementary, volunteer-based, long-term monitoring program, called Mountain Birdwatch, was initiated in 2000 to collect census data on five common montane forest bird species throughout the Northeast. Also, through a cooperative agreement with the National Park Service, VCE is coordinating breeding bird monitoring at 9 National Parks and Historic Sites in the Northeast. Initiated in 2006, annual surveys are conducted at 19 study sites in New Jersey, Connecticut, New York, Massachusetts, Vermont, New Hampshire, and Maine.

Methods

In 2010, surveys were conducted by VCE staff biologists at the Mt. Mansfield Ridgeline and Ranch Brook, and by volunteer observers at the Lye Brook and Underhill sites. Each study site consisted of 5 point count stations. Survey methods consisted of unlimited distance point counts, based on the approach described by Blondel et al. (1981) and used in Ontario (Welsh 1995). The count procedure was as follows:

1. Counts began shortly after dawn on days where weather conditions were unlikely to reduce count numbers (i.e., calm winds and very light or no rain). Censusing began shortly (< 1 min) after arriving at a station.
2. Observers recorded all birds seen and heard during a 10-min sampling period, which was divided into 3 time intervals: 3, 2, and 5 mins. Observers noted in which time interval each bird was first encountered, and placed birds into one of 2 distance categories (within or beyond 50 m). To reduce duplicate records, individual birds were mapped on standardized field cards, and known or presumed movements were noted. Different symbols were used to record the status of birds encountered (i.e., singing male, pair observed, calling bird, etc.).
3. The number of surveys at each site was dependent on elevation; montane fir-spruce sites were sampled once, while LBWA and Underhill were sampled twice during the breeding season, the first during early June (ca. 2-12 June) and the second during late June (ca. 14-30 June). Observers were encouraged to space their visits 7-10 days apart. For each site visit, all stations were censused in a single morning and in the same sequence.

In summarizing data for analysis, the maximum count for each species was used as the station estimate for each year. All birds seen or heard were each counted as 1 individual unless a family group or active nest was encountered, in which case they were scored as a breeding pair, or 2 individuals. Population trends were calculated for the 8 most commonly encountered species at each site using simple linear regression. For each species, the annual population trend was calculated by plotting the maximum count against year, and then calculating the mean annual rate of change of a linear trendline inserted through the points (e.g. Percent Annual Trend = slope ÷ y intercept x 100). Regression and correlation analyses were done using SYSTAT 10.2.

Results

A combined total of 55 avian species were detected during breeding bird surveys at three study sites on Mt. Mansfield from 1991-2010. Species richness was similar at both montane forest sites, with a total of 30 species detected at the Mansfield ridgeline and 31 at Ranch Brook. Surveys at Ranch Brook continue to average a greater number of individuals and species per year than the higher elevation and more exposed Mansfield ridgeline site (Tables 1 and 2). Surveys at the mid-elevation, northern hardwood study sites at Underhill State Park and Lye Brook Wilderness showed similar species composition, with Underhill averaging 17.83 species per year compared to Lye Brook's 15.91 (Tables 3 and 4).

Mount Mansfield

On the Mt. Mansfield ridgeline plot in 2010, both species richness and numerical abundance were well below average, with 55 individuals of 9 species detected, the lowest species count in the survey's 20-year history (Table 1). Of the 8 most commonly recorded species, 4 were below the 20-year average, and 4 were above. Five species exhibited decreasing population trends, with one species, Blackpoll Warbler, showing a significant decline of 2.75% per year ($r^2 = 0.306$; $P = 0.011$). Three species showed non-significant increasing trends. The number of Bicknell's Thrush dropped from last year's count of 10 individuals to just 6, the lowest since 2004.

At the Ranch Brook study site in 2010, species richness was slightly below the 15-year average, while numerical abundance rebounded from last year's record low count of just 37 individuals to 79 (Table 2). Among the 8 most abundant species, half were below the 15-year mean and half were above. Overall, just 2 of these 8 species showed increasing trends, while 6 declined. Two species declined significantly; White-throated Sparrow, which continued a downward trend at a rate of 4.35% per year ($r^2 = 0.465$; $P = 0.005$), and Yellow-bellied Flycatcher, which declined at a rate of 2.91% per year ($r^2 = 0.359$; $P = 0.018$). Counts of Bicknell's Thrush, Swainson's Thrush, and Dark-eyed Junco all recovered from last year's record low numbers.

At Underhill State Park in 2010, total number of individuals rebounded from last year's record low count of 39 individuals to 68 birds of 17 species (Table 3). Among the 8 most common species at the site, half were above the 18-year mean, and half were below. Overall, 6 species showed increasing population trends, including significant increases for Black-throated Blue Warbler (6.27%; $r^2 = 0.289$, $P = 0.022$) and Black-throated Green Warbler (5.44%; $r^2 = 0.459$, $P = 0.002$). After a single Canada Warbler was detected in 2008, the first in 5 years, none were encountered in 2009 or 2010, continuing its declining trend at 5.30% per year ($r^2 = 0.714$, $P < 0.001$). This was also the second consecutive year that no Winter Wrens were detected at Underhill State Park.

Lye Brook Wilderness

At Lye Brook Wilderness, species richness and numerical abundance were both below the 11-year average, with 61 individuals of 12 species detected (Table 4). Among the 8 most common species, five were above the 10-year average, while three (all warblers) were below. Of these 8 species, five exhibited increasing population trends, while three showed declines. Among significant trends, Ovenbird declined at a rate of 3.52% ($r^2 = 0.434$; $P = 0.028$), and Hermit Thrush increased at 12.90% per year ($r^2 = 0.519$; $P = 0.012$). The maximum counts for two species (Yellow-bellied Sapsucker and Hermit Thrush) were the highest in site's 11-year history.

Discussion

Although the linear regression trend for Blackpoll Warbler showed a significant decline for the fourth consecutive year, the species appears to be rebounding from its record low count of 2 individuals on the Mt. Mansfield Ridgeline site in 2007. Interestingly, 2007 was also a low year for Blackpolls at the Ranch Brook site, suggesting that low counts may represent an accurate index to the population.

At the Ranch Brook site, White-throated Sparrow continued its declining trend for the fourth consecutive year. However, the unusually high maximum count of 22 White-throats recorded in 1995 is largely responsible for driving the trend's statistical significance. The mean count at Ranch Brook was 8.8 over the 15-year study period, and 7.9 without the 1995 outlier. So, while the biological significance of the White-throated Sparrow decline observed at Ranch Brook appears to be low, it bears continued scrutiny.

It's encouraging to note that two of the most commonly detected species at Underhill State Park (Black-throated Green Warbler and Black-throated Blue Warbler) show significantly increasing trends, suggesting that the conditions of their preferred breeding habitat has improved over the last 10 years or so. Considering that Black-throated Green Warbler numbers increased after 1999 and the upward trend for Black-throated Blue began in 2004, it's possible that impacts from the 1998 ice storm may have resulted in changes to forest structure and composition, benefitting both species. Black-throated Blue Warbler prefers hardwood forests with a dense understory of hobblebush (*Viburnum lantanoides*), while Black-throated Green prefer stands with a mix of hardwood and coniferous trees. The ice storm resulted in broken limbs and main stems, primarily of hardwood canopy trees, which left many small- to medium-sized canopy gaps (Faccio 2003). Since the storm had a disproportionate effect on hardwoods over conifers (which can better withstand ice accumulation due to their structural morphology), percent conifer cover may have increased, favoring Black-throated Green Warbler. In addition, canopy gaps allowed more sunlight to reach the forest floor, which, after a lag-time of 2-3 years, increased the density of the understory layer, including hobblebush (Faccio 2003), possibly creating additional habitat for Black-throated Blue Warbler.

The site-specific trend estimates presented for the Mt. Mansfield and Lye Brook sites must be interpreted carefully as these data are from a limited geographic sample with low power. Year to year changes in survey counts may simply reflect natural fluctuations, variable detection rates, and/or a variety of dynamic factors, such as prey abundance, overwinter survival, and habitat change. Continued data collection, their correlation with other VMC data, and comparison with survey data from other ecologically similar sites will be necessary to elucidate meaningful population trends of various species at these sites.

Acknowledgements

Many thanks to Bobbie Jean Booth and Zoe Richards for conducting bird surveys at the Lye Brook Wilderness Area and Underhill State Park, respectively.

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Table 1. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species (bold type) at Mt. Mansfield Ridgeline, 1991-2010.

Common Name	'91	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	Mean	SD	r ²	Annual Trend (%)
Red Squirrel											1										0.05	0.22		
Sharp-shinned Hawk										1											0.05	0.22		
Hairy Woodpecker				1																	0.05	0.22		
Northern Flicker			1																		0.05	0.22		
Yellow-bellied Flycatcher			1		1	2	3		1	1	1	1	2	1		1	2	1	3		1.05	0.94		
Alder Flycatcher							1														0.05	0.22		
Red-eyed Vireo									1												0.05	0.22		
Blue Jay		1												1		1					0.15	0.37		
Common Raven			1		1				1	1		1	1	1		2		1			0.50	0.61		
Red-breasted Nuthatch	1	2	3	1	3	1		1	2		1				1		1				0.85	0.99		
Winter Wren	10	9	7	4	5	2	4	10	8	4	4	7	3	7	8	12	7	5	6	8	6.50	2.63	0.003	0.41
Golden-crowned Kinglet										1											0.05	0.22		
Ruby-crowned Kinglet		2			1							1	1							1	0.30	0.57		
Bicknell's Thrush	6	15	11	8	10	11	9	9	8	7	9	9	6	5	8	11	12	7	10	6	8.85	2.43	0.072	-1.10
Swainson's Thrush	3	8	1	1	3	6	7	5	4	3	3	2	2	1	2	5	1	5	3	5	3.50	2.06	0.019	-1.20
Hermit Thrush											1	1									0.10	0.31		
American Robin	1	4	1	2	2	2	2	1	1	3	3	2	6	3	1	3	4	3	2	4	2.50	1.32	0.153	5.51
Cedar Waxwing		1	4				9							1							0.75	2.15		
Nashville Warbler	2					2	3	1	1		1					1					0.60	0.88		
Magnolia Warbler	1	2				3	1	1			1		3	1	4		1				0.90	1.21		
Yellow-rumped Warbler	9	11	8	9	8	12	10	13	11	9	11	14	10	13	9	9	7	12	12	8	10.25	1.97	0.006	0.23
Blackpoll Warbler	8	9	9	7	7	15	10	10	9	8	8	3	3	9	8	8	2	4	5	5	7.35	3.03	0.306	-2.75*
Ovenbird			1						1												0.10	0.31		
Canada Warbler							1														0.05	0.22		
Lincoln's Sparrow	2					1															0.15	0.49		
White-throated Sparrow	6	14	14	12	14	13	20	14	19	14	18	11	13	11	10	14	14	12	10	12	13.25	3.19	0.010	-0.40
Dark-eyed Junco	3	9	6	2	5	5	9	8	7	2	7	6	5	7	4	5	4	6	6	6	5.60	1.98	0.000	-0.53
Purple Finch	2	4	1	2	3	2	2	1	4	2	3	4	4	2	1	2	2	4	3		2.40	1.19		
White-winged Crossbill					8		1	1													0.50	1.79		
Pine Siskin		1			1		2	1			11						5		1		1.10	2.61		
Evening Grosbeak		2																			0.10	0.45		
Species Richness^a	13	16	15	11	14	15	17	14	15	13	15	12	15	14	11	13	13	11	11	9	13.35	2.03		
Number of Individuals^a	54	94	69	49	71	78	94	76	78	56	80	61	61	63	56	62	62	60	61	55	66.25	12.14		

^a Does not include counts of Red Squirrel* $P = 0.011$

Table 2. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species (bold type) at Ranch Brook, 1995-2010. Note that a survey was not conducted in 2004.

Common Name	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	Mean	SD	r ²	Annual Trend (%)
Eastern Chipmunk													1				0.07	0.26		
Red Squirrel					4		1		7					4			1.07	2.15		
Sharp-shinned Hawk				1							1						0.13	0.35		
Mourning Dove						1	1										0.13	0.35		
Ruby-throated Hummingbird						1											0.07	0.26		
Hairy Woodpecker	1																0.07	0.26		
Pileated Woodpecker							2										0.13	0.52		
Yellow-bellied Flycatcher	4	4	4	3	3	4	2	4	4		3	2	4	3	2	1	3.13	0.99	0.359	-2.91*
Blue-headed Vireo												1					0.07	0.26		
Red-eyed Vireo				1													0.07	0.26		
Blue Jay	1										1	1		4			0.47	1.06		
Common Raven		4	3	4		4	2						1	1		1	1.33	1.63		
Black-capped Chickadee	1												1				0.13	0.35		
Red-breasted Nuthatch	7		2		6		2		2		4		5	1		5	2.27	2.49		
Winter Wren	8	3	7	10	9	10	5	5	9		10	11	6	8	5	9	7.67	2.38	0.016	0.87
Golden-crowned Kinglet				1	3	1		3			2	1		2		1	0.93	1.10		
Ruby-crowned Kinglet	3		3			3			1		1	1				1	0.87	1.19		
Bicknell's Thrush	5	6	7	5	5	6	2	8	1		8	2	5	5	2	7	4.93	2.25	0.030	-1.41
Swainson's Thrush	6	15	9	5	3	4	8	11	10		8	5	9	7	3	7	7.33	3.24	0.046	-1.67
Hermit Thrush	1		3														0.27	0.80		
American Robin		2	2	2	1	1	1	1	3		4	5	2	2	3	6	2.33	1.63		
Cedar Waxwing				1			1				1						0.20	0.41		
Nashville Warbler		1	3	2	1	3		3	4		3	2	3	2	1	4	2.13	1.30		
Northern Parula									1								0.07	0.26		
Magnolia Warbler	2	4	4	2	3	5	4	2	4		2	3	1	2	2	6	3.07	1.39		
Black-throated Blue Warbler	1																0.07	0.26		
Yellow-rumped Warbler	5	6	4	5	7	11	9	11	8		4	8	8	6	4	7	6.87	2.33	0.002	0.35
Blackpoll Warbler	9	9	15	8	3	8	7	8	8		8	10	4	6	6	7	7.73	2.74	0.165	-2.35
White-throated Sparrow	22	11	12	9	8	7	7	10	10		7	4	8	4	5	8	8.80	4.35	0.465	-4.35**
Dark-eyed Junco	9	5	3	2	5	2	5	4	4		7	5	1	4	1	5	4.20	2.21	0.088	-2.51
Purple Finch	2	1	4	4	2	4	4		6					2	1	5	2.53	1.92		
White-winged Crossbill	8		2		1		6										1.13	2.47		
Pine Siskin	12		1		7								1		1		1.47	3.42		
Species Richness^a	19	13	18	17	16	17	18	12	15		17	15	16	16	14	15	15.87	1.92		
Number of Individuals^a	107	71	88	65	67	75	69	82	82		74	61	62	59	37	79	71.87	15.68		

^a Does not include counts of Eastern Chipmunk or Red Squirrel

* $P = 0.018$; ** $P = 0.005$

Table 3. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species (bold type) at Underhill State Park, 1991-2010. Note that surveys were not conducted in 2003 or 2005.

Common Name	'91	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	Mean	SD	r ²	Annual Trend (%)	
Eastern Chipmunk							3	5					/	1	/		1				0.56	1.34			
Red Squirrel							1	3		1			/		/		1	1		1	0.44	0.78			
Broad-winged Hawk							1						/		/						0.06	0.24			
Mourning Dove									1				/	1	/						0.11	0.32			
Yellow-bellied Sapsucker		2		1	1		1	1	1		3		/	2	/		2	3	2		1.06	1.06			
Downy Woodpecker							1						/		/		1	1		1	0.22	0.43			
Hairy Woodpecker				1			1	1	2				/		/				2	2	0.50	0.79			
Northern Flicker			1										/		/						0.06	0.24			
Pileated Woodpecker	2	1	1			1							/		/						0.28	0.57			
Least Flycatcher													/		/		2				0.11	0.47			
Eastern Phoebe												1	/		/						0.06	0.24			
Blue-headed Vireo	1	2				1	1			1			/	1	/		2	1	1	3	3	0.94	1.00		
Red-eyed Vireo	3	4	4	6	9	8	7	6	10	8	8	7	/	5	/		7	8	6	2	5	6.28	2.14	0.000	0.01
Blue Jay	2	1		1		2	2		1	1	2	1	/	1	/		1	1		1	0.94	0.73			
American Crow													/		/		1	1			0.11	0.32			
Common Raven				4	1				1	1			/	1	/					2	0.56	1.04			
Black-capped Chickadee		1	1		2	3	3		3	1	1		/		/		2	1	3	2	1.28	1.18			
Red-breasted Nuthatch							1						/		/						0.06	0.24			
White-breasted Nuthatch							1						/	1	/						0.11	0.32			
Brown Creeper				1					1	1		1	/	1	/		1	1			0.39	0.50			
Winter Wren		6	2	1	5	3	4	6	4	4	3	3	/	3	/		4	2	1		2.83	1.92	0.129	-2.84	
Golden-crowned Kinglet								1					/		/		1				0.11	0.32			
Veery	1	1								1			/		/						0.17	0.38			
Swainson's Thrush		1		2	4	3		1	4	2	2		/		/		1		2		1.22	1.40			
Hermit Thrush		4	1	6	7	3	4	4	2		4	5	/	4	/		4	7	1	4	3	3.50	2.09	0.014	1.33
Wood Thrush	1	1											/		/						0.11	0.32			
American Robin	1				3	3	3	4	2	1	2	1	/	2	/		1			3	1.44	1.34			
Magnolia Warbler	1				1								/		/		1				0.17	0.38			
Black-th. Blue Warbler	4	9	5	6	7	8	6	5	6	5	5	5	/	11	/		15	8	11	5	14	7.50	3.28	0.289	6.27*
Yellow-rumped Warbler				2	2		2	3	3	1	1	3	2	/	/		1		1	1	1	1.28	1.07		
Black-th. Green Warbler	5	7	6	7	7	7	9	5	8	10	10	8	/	13	/		15	12	10	7	11	8.72	2.76	0.459	5.44**
Blackburnian Warbler										1	1		/		/		1			1	0.22	0.43			
Blackpoll Warbler						1	2						/		/						0.17	0.51			

Continued on next page

Common Name	'91	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	Mean	SD	r^2	Annual Trend (%)
Black-and-White Warbler		3	2	2	4	2	3	2	1	3	4	2	/	1	/	2	3		1	1	2.00	1.19		
American Redstart		4			1	1							/	/							0.33	0.97		
Ovenbird	4	10	11	11	13	12	12	10	13	10	13	6	/	11	/	11	15	14	7	14	10.94	2.88	0.095	1.53
Mourning Warbler													/	/			1	1			0.11	0.32		
Canada Warbler	3	4	4	6	2	4	4	2	2	3	2	2	/	/					1		2.17	1.79	0.714	-5.30***
Scarlet Tanager					1				1				/	/		1					0.17	0.38		
White-throated Sparrow	2		2	1	1		1					1	/	/		1		1		1	0.61	0.70		
Dark-eyed Junco		3	1	3	4	3	5	2	2	1	2	2	/	1	/	5	5	2	4	3	2.67	1.50	0.102	4.16
Rose-breasted Grosbeak	4	2		1	3	1	2		1				/	/		1					0.83	1.20		
Purple Finch						1		1			1		/	/		1	1		1		0.33	0.49		
White-winged Crossbill											2		/	/							0.11	0.47		
Pine Siskin					1						1		/	/							0.11	0.32		
American Goldfinch	1												/	1	/						0.11	0.32		
Species Richness^a	15	19	14	18	20	20	23	16	21	16	20	16	/	17	/	22	18	17	12	17	17.83	2.85		
Number of Individuals^a	35	66	43	62	77	69	77	54	67	53	70	48	/	60	/	81	73	58	39	68	61.11	13.48		

^a Does not include counts of Red Squirrel or Eastern Chipmunk

* $P = 0.022$

** $P = 0.002$

*** $P < 0.001$

Table 4. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species (bold type) at Lye Brook Wilderness Area, 2000-2010.

Common Name	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	Mean	SD	r ²	Annual Trend (%)
Eastern Chipmunk	2			1								0.27	0.65		
Red Squirrel	1	1										0.18	0.40		
Ruffed Grouse	1					2						0.27	0.65		
Mourning Dove		1										0.09	0.30		
Yellow-Billed Cuckoo									1			0.09	0.30		
Barred Owl	1											0.09	0.30		
Chimney Swift	2											0.18	0.60		
Ruby-throated Hummingbird									1	1		0.18	0.40		
Yellow-bellied Sapsucker	5	6			2		2	2	5	8	11	3.73	3.61	0.227	83.80
Downy Woodpecker	1		1									0.18	0.40		
Hairy Woodpecker	2	1	2					1	1	1		0.73	0.79		
Unidentified Woodpecker	3											0.27	0.90		
Northern Flicker									1			0.09	0.30		
Pileated Woodpecker	1		3	1	4	1	1		2	1	2	1.45	1.21	0.001	0.65
Eastern Wood-Pewee				1								0.09	0.30		
Yellow-bellied Flycatcher							1					0.09	0.30		
Least Flycatcher	2											0.18	0.60		
Great Crested Flycatcher				1								0.09	0.30		
Blue-headed Vireo		1	4	1		1				1		0.73	1.19		
Red-eyed Vireo	10	6	9	4	6	6	4	5	13	14	10	7.91	3.51	0.143	7.26
Blue Jay		3		1		1			2	1	3	1.00	1.18		
Common Raven					1	1						0.18	0.40		
Black-capped Chickadee	1	1		2			1	2	1		1	0.82	0.75		
White-breasted Nuthatch						1	1					0.18	0.40		
Brown Creeper	1											0.09	0.30		
Winter Wren	7		1		3	1			2			1.27	2.15		
Ruby-crowned Kinglet						1						0.09	0.30		
Veery					1							0.09	0.30		
Swainson's Thrush	2		1	3	2		2	1	1	2		1.27	1.01		
Hermit Thrush	4	2	6	5	4	4	4	5	6	7	8	5.00	1.67	0.519	12.90*
American Robin	1		1		3			1	1		2	0.82	0.98		
Cedar Waxwing	1								1			0.18	0.40		
Northern Parula				3	1							0.36	0.92		
Magnolia Warbler	1		3									0.36	0.92		
Black-throated Blue Warbler	9	7	10	9	8	12	11	8	8	8	5	8.64	1.91	0.110	-1.95
Yellow-rumped Warbler	2	1				0				1		0.36	0.67	1	
Black-throated Green Warbler	8	10	4	6	8	9	12	3	11	9	6	7.82	2.82	0.003	0.60
Blackburnian Warbler	5											0.45	1.51		
American Redstart	2	1	3	1		4						1.00	1.41		
Ovenbird	15	13	19	11	14	13	12	12	8	12	10	12.64	2.84	0.434	-3.52**
Canada Warbler	1											0.09	0.30		
Scarlet Tanager	1		3	2	2	2			1		1	1.09	1.04		
White-throated Sparrow	2		2	4		2						0.91	1.38		
Dark-eyed Junco	2	3	1	1	1	4		1			2	1.36	1.29	0.141	-6.50
Rose-breasted Grosbeak	2	1										0.27	0.65		
Species Richness^a	28	15	17	17	16	17	11	11	18	13	12	15.91	4.76		
Number of Individuals^a	98	58	73	57	60	65	51	41	66	66	61	63.27	14.31		

^a Does not include counts of Red Squirrel or Eastern Chipmunk* $P = 0.012$, ** $P = 0.028$