

Long-term Avian Research and Monitoring on Mt. Mansfield, Vermont

2012 Report to the Vermont Monitoring Cooperative



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

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

Part III. Migratory Movements and Distribution of Bicknell's Thrush (*Catharus bicknelli*): Gaining Insights from Geolocators

Submitted by:

Christopher C. Rimmer, Steven D. Faccio and Kent P. McFarland

Vermont Center for Ecostudies

P.O. Box 420

Norwich, VT 05055

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

Christopher C. Rimmer and Kent P. McFarland

During 2012, we completed our 21st consecutive field season of demographic monitoring of breeding montane forest birds on the Mt. Mansfield ridgeline. As in previous years, we focused our efforts on five target species: Bicknell's Thrush (*Catharus bicknelli*), Swainson's Thrush (*C. usutulatus*), Blackpoll Warbler (*Setophaga striata*), Yellow-rumped (Myrtle) Warbler (*S. coronata*), and White-throated Sparrow (*Zonotrichia albicollis*). However, as a means to more broadly assess population changes and the potential impacts of climatic warming, we expanded the scope of our work to encompass the entire avian community. This report presents a summary of data collected.

Methods

We used mist-netting and banding to sample breeding bird species on an established study plot on the Mt. Mansfield ridgeline between c. 1155-1190 m (3800-3900 ft) elevation. We conducted 14 banding sessions between 8 June and 22 September 2012, using 10-30 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 dusk and from dawn until late morning on the following day. Bicknell's Thrushes were captured both passively and through the use of vocal lures (recorded playbacks of 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. We recorded data on age, sex, breeding condition, fat class, ectoparasites, flight feather wear, and net site of capture. Standard morphometrics included wing chord, tail length, weight, tarsal length, culmen length, bill length from mid-nares, bill width, and bill depth. We collected a single tail feather of Bicknell's Thrush for future chemical analysis such as stable isotopes or other trace elements.

Results and Discussion

We accumulated 1,136.5 net-hours in 2012, with a mean of 81.2 ± 51.3 SD net-hours per day (range = 13–159). Inclement weather on several dates reduced the number of hours that nets could be operated. A total of 206 individuals of 17 species were captured and banded (Table 1.1), for a capture rate of 18.13 new birds/100 net-hours. Among the five target species, we captured 161 individuals (Table 1.1), for a capture rate of 14.17 birds/100 net hours (14.43 in 2011). We recaptured 16 individuals that had been banded in a previous year, including 8 Bicknell's Thrush, 5 Blackpoll Warblers, and 3 Yellow-rumped Warblers. Remarkable among the returning Bicknell's Thrushes was an 11 year-old male that had been banded as a yearling in June of 2003, recaptured the following year, then not encountered again until June of 2011. This individual's longevity equaled the species' record of a male banded on Stratton Mountain in 1997 and last recaptured there in 2008. Among new captures, Yellow-rumped Warbler ranked first in abundance ($n = 48$) but third in return captures ($n = 3$, 6.3% of total captured individuals), followed by Blackpoll Warbler (37, 5 [13.5%]). As usual, Bicknell's Thrush had the highest rate of return captures (29.6%), although this was lower than in previous years (e.g., 35.7% in 2011).

No previously banded Swainson's Thrushes or White-throated sparrows were recaptured in 2012.

Our 2012 results for Bicknell's thrush highlight both the high survivorship and strong breeding site fidelity of adults, but also the difficulty of obtaining complete population samples in a given year, underscoring the need for multiple-year sampling in order to obtain accurate demographic data for individual birds. The difficulty of intensively sampling all montane forest habitat on this study plot, due to constraints of terrain, accessibility, and weather, undoubtedly causes a significant portion of the breeding population to go undetected each year. This may be particularly true for females, with their smaller home ranges and more limited movements than males (Rimmer et al. 2001).

Acknowledgements

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Table 1.1. Numbers of new bandings on Mt. Mansfield in 2012, ranked by descending abundance.

Species	No. Individuals	Status
Yellow-rumped (Myrtle) Warbler	48	Breeding
Blackpoll Warbler	37	Breeding
White-throated Sparrow	37	Breeding
Bicknell's Thrush	27	Breeding
Dark-eyed (Slate-colored) Junco	19	Breeding
Swainson's Thrush	12	Breeding
Magnolia Warbler	8	Breeding
Ruby-crowned Kinglet	4	Breeding
American Robin	3	Breeding
Red-breasted Nuthatch	2	Breeding
Winter Wren	2	Breeding
Yellow-bellied Flycatcher	2	Breeding
Golden-crowned Kinglet	1	Breeding
Hermit Thrush	4	Transient
Black-throated Blue Warbler	1	Transient
Eastern Phoebe	1	Transient
Ovenbird	1	Transient

Table 1.2. Capture histories for previously banded birds recaptured on Mt. Mansfield in 2012.

Species	Band number	Sex	Total Captures	2012	2011	2010	2009	2008	2007	2006	2005	2004	2003
BITH	135192289	M	3	1	1							1	1
BITH	144128843	M	6	1	1	1		1	2				
BITH	144128885	M	1	1			1						
BITH	157105377	M	7	1	2		1	1	2				
BITH	157106563	M	6	2	4								
BITH	157106564	F	2	1	1								
BITH	157106571	F	6	2	1		3						
BITH	193176490	M	2	1	1								
BLPW	211080379	M	3	1		1	1						
BLPW	211080395	M	4	1	2	1							
BLPW	222096662	M	1	1	1								
BLPW	222096753	M	4	2		2							
BLPW	249089260	M	3	2			1						
MYWA	249089924	M	1	1		1							
MYWA	249089924	M	4	2	2								
MYWA	249089946	M	2	1	1								

BITH = Bicknell's Thrush; BLPW = Blackpoll Warbler, MYWA = Yellow-rumped (Myrtle) Warbler
M = male; F = female

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

Steven D. Faccio and Christopher C. Rimmer

In 2012, breeding bird surveys were continued at 2 permanent study sites on Mt. Mansfield. The Mt. Mansfield ridgeline was surveyed for the 22nd consecutive year while the Ranch Brook site was censused for the 17th year in 2012 (the 2004 survey was not completed due to inclement weather on attempted survey dates). Our permanent study sites at Underhill State Park and the Lye Brook Wilderness Area (LBWA) of the Green Mountain National Forest were not surveyed in 2012 due to observer schedule conflicts and inclement weather.

The Mt. Mansfield ridgeline site, at 1158 m (3800 ft), consists of montane fir-spruce, while the Ranch Brook site ranges between 975 and 1097 m (3200 and 3600 ft), and is dominated by a paper birch-fir canopy. Underhill State Park site consists of mature northern hardwoods ranging from 609 to 731 m (2000 to 2400 ft) elevation. The Lye Brook study site, located in Winhall, VT 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 32 monitoring sites in 9 different forested habitats primarily in Vermont, but with two sites just across the Connecticut River in New Hampshire. A complementary, volunteer-based, long-term monitoring program, called Mountain Birdwatch, was initiated in 2000 to collect census data on montane forest bird species throughout the Northeast (Scarl 2011). Also, through a cooperative agreement with the National Park Service, VCE coordinates breeding bird monitoring at 11 National Parks and Historic Sites in the Northeast. Initiated in 2006, National Park surveys are conducted at 29 study sites in New Jersey, Connecticut, New York, Massachusetts, Vermont, New Hampshire, and Maine (Faccio and Mitchell 2013).

Methods

In 2012, surveys were conducted by VCE staff biologists at the Mt. Mansfield Ridgeline and Ranch Brook. 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 and Discussion

After near record lows for relative abundance and species richness at both montane study sites on Mount Mansfield in 2011, these metrics recovered significantly on the ridgeline plot, but only slightly on the Ranch Brook transect. Overall, a combined total of 55 avian species have been detected during breeding bird surveys at three study sites on Mt. Mansfield from 1991-2012. Species richness was similar at both montane forest sites, with a total of 31 species detected at both the Mansfield ridgeline and 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 2.1 and 2.2).

Mount Mansfield

On the Mt. Mansfield ridgeline plot in 2012, both species richness and numerical abundance recovered dramatically from 2011's record low of 36 individuals of 11 species, to 76 individuals of 16 species (Table 2.1). The number of species detected in 2012 was the highest since 1997's count of 17 species, and, along with 1992's tally, was tied for the second highest species count. A Veery was detected for the first time in the count's 22-year history, while a Hermit Thrush was detected for just the third time. Numerical abundance was the highest since 2001, and was largely driven by relatively high counts of four species (White-throated Sparrow, Blackpoll Warbler, American Robin, and Swainson's Thrush) (Table 2.1). Of the 8 most commonly recorded species, half were above the 22-year average and half were below, although all except Bicknell's Thrush increased from 2011 counts. Still, seven species exhibited decreasing population trends. Although Blackpoll Warbler numbers increased dramatically from just a single bird in 2011 to nine in 2012, the species continued to exhibit a significant decline of 2.58% per year ($r^2 = 0.279$; $P \leq 0.018$). Bicknell's Thrush showed a weakly declining trend of 1.34% per year ($r^2 = 0.134$; $P = 0.093$), while American Robin increased significantly by 8.09% annually ($r^2 = 0.250$; $P \leq 0.018$).

At the Ranch Brook study site in 2012, species richness and relative abundance increased slightly from 2011 to 63 individuals of 13 species, but remained below the site's 17-year average (Table 2.2). Although no new species were detected, a single Hermit Thrush was found, just the third for the study site. Among the 8 most abundant species, five were at or above the 17-year mean. Overall, just two of these 8 species showed increasing trends, while 6 declined. Three species declined significantly (White-throated Sparrow, Yellow-bellied Flycatcher, and Blackpoll Warbler), although counts for all three increased from 2011. White-throated Sparrow, which had its highest count since 2003, continued a downward trend at a rate of 3.34% per year ($r^2 = 0.329$; $P \leq 0.012$). Although Blackpoll Warbler numbers bounced back after a dismal year in 2011 when no birds were detected at Ranch Brook for the first time, they continued an annual decline of 2.91% ($r^2 = 0.254$; $P = 0.039$). Swainson's Thrush numbers remained high for the second consecutive year.

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Table 2.1. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species(bold type) at Mt. Mansfield Ridgeline, 1991-2012.

Common Name	'91	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	'11	'12	Mean	SD	r ²	Annual Trend (%)	
Red Squirrel											1												0.05	0.21			
Sharp-shinned Hawk										1													0.05	0.21			
Hairy Woodpecker				1																			0.05	0.21			
Northern Flicker			1																				0.05	0.21			
Yellow-bellied Flycatcher			1		1	2	3		1	1	1	1	2	1		1	2	1	3				1.09	1.02			
Alder Flycatcher							1																0.05	0.21			
Red-eyed Vireo									1														0.05	0.21			
Blue Jay		1											1		1								0.14	0.35			
Common Raven			1			1			1	1		1	1	1		2		1					0.59	0.80			
Red-breasted Nuthatch	1	2	3	1	3	1		1	2		1				1		1						0.86	0.99			
Winter Wren	10	9	7	4	5	2	4	10	8	4	4	7	3	7	8	12	7	5	6	8	4	5	6.32	2.57	0.004	-0.38	
Golden-crowned Kinglet										1													0.05	0.21			
Ruby-crowned Kinglet		2			1							1	1								1	1	0.32	0.57			
Veery																						1	0.05	0.22			
Bicknell's Thrush	6	15	11	8	10	11	9	9	8	7	9	9	6	5	8	11	12	7	10	6	7	6	8.64	2.42	0.134	-1.34*	
Swainson's Thrush	3	8	1	1	3	6	7	5	4	3	3	2	2	1	2	5	1	5	3	5	1	6	3.50	2.11	0.011	-0.86	
Hermit Thrush											1	1										1	0.14	0.35			
American Robin	1	4	1	2	2	2	2	1	1	3	3	2	6	3	1	3	4	3	2	4	3	6	2.68	1.46	0.250	8.09**	
Cedar Waxwing		1	4				9						1										0.68	2.06			
Nashville Warbler	2					2	3	1	1		1					1					1		0.59	0.85			
Magnolia Warbler	1	2				3	1	1			1	3	1	4		1					1	2	0.95	1.17			
Yellow-rumped Warbler	9	11	8	9	8	12	10	13	11	9	11	14	10	13	9	9	7	12	12	8	5	7	9.86	2.27	0.047	-0.71	
Blackpoll Warbler	8	9	9	7	7	15	10	10	9	8	8	3	3	9	8	8	2	4	5	5	1	9	7.14	3.21	0.279	-2.58**	
Ovenbird			1						1														0.09	0.29			
Canada Warbler							1																0.05	0.21			
Lincoln's Sparrow	2					1																	0.14	0.47			
White-throated Sparrow	6	14	14	12	14	13	20	14	19	14	18	11	13	11	10	14	14	12	10	12	8	17	13.18	3.35	0.011	-0.38	
Dark-eyed Junco	3	9	6	2	5	5	9	8	7	2	7	6	5	7	4	5	4	6	6	6	6	3	4	5.41	1.99	0.026	-0.82
Purple Finch	2	4	1	2	3	2	2	1	4	2	3	4	4	2	1	2	2	4	3		2		2.27	1.24			
White-winged Crossbill					8		1	1															0.45	1.71			
Pine Siskin		1			1		2	1			11						5		1			3	1.14	2.53			
Evening Grosbeak		2																					0.09	0.43			
Species Richness^a	13	16	15	11	14	15	17	14	15	13	15	12	15	14	11	13	13	11	11	9	11	16	13.36	2.08			
Number of Individuals^a	54	94	69	49	71	78	94	76	78	56	80	61	61	63	56	62	62	60	61	55	36	76	65.32	13.44			

^a Does not include counts of Red Squirrel

* $P = 0.093$; ** $P \leq 0.018$

Table 2.2. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species (bold type) at Ranch Brook, 1995-2012. 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	'11	'12	Mean	SD	r ²	Annual Trend (%)
Eastern Chipmunk													1						0.06	0.24		
Red Squirrel					4		1		7				4						0.94	2.05		
Sharp-shinned Hawk				1							1								0.12	0.33		
Mourning Dove						1	1												0.12	0.33		
Ruby-throated Hummingbird						1													0.06	0.24		
Hairy Woodpecker	1																		0.06	0.24		
Pileated Woodpecker							2												0.12	0.49		
Yellow-bellied Flycatcher	4	4	4	3	3	4	2	4	4		3	2	4	3	2	1	2	3	3.06	0.97	0.352	-2.58**
Blue-headed Vireo												1							0.06	0.24		
Red-eyed Vireo				1															0.06	0.24		
Blue Jay	1										1	1		4					0.41	1.00		
Common Raven		4	3	4		4	2						1	1		1		2	1.29	1.57		
Black-capped Chickadee	1												1						0.12	0.33		
Red-breasted Nuthatch	7		2		6		2		2		4		5	1		5		2	2.12	2.39		
Winter Wren	8	3	7	10	9	10	5	5	9		10	11	6	8	5	9	7	8	7.65	2.23	0.009	0.53
Golden-crowned Kinglet				1	3	1		3			2	1		2		1			0.82	1.07		
Ruby-crowned Kinglet	3		3			3			1		1	1			1				0.76	1.15		
Bicknell's Thrush	5	6	7	5	5	6	2	8	1		8	2	5	5	2	7	4	4	4.82	2.13	0.049	-1.52
Swainson's Thrush	6	15	9	5	3	4	8	11	10		8	5	9	7	3	7	13	12	7.94	3.49	0.013	0.98
Hermit Thrush	1		3															1	0.29	0.77		
American Robin		2	2	2	1	1	1	1	3		4	5	2	2	3	6	4	2	2.41	1.58		
Cedar Waxwing				1			1				1								0.18	0.39		
Nashville Warbler		1	3	2	1	3		3	4		3	2	3	2	1	4	1		1.94	1.34		
Northern Parula									1										0.06	0.24		
Magnolia Warbler	2	4	4	2	3	5	4	2	4		2	3	1	2	2	6	1	1	2.82	1.47		
Black-throated Blue Warbler	1																		0.06	0.24		
Yellow-rumped Warbler	5	6	4	5	7	11	9	11	8		4	8	8	6	4	7	6	5	6.71	2.23	0.006	-0.43
Blackpoll Warbler	9	9	15	8	3	8	7	8	8		8	10	4	6	6	7		7	7.24	3.17	0.254	-2.91*
White-throated Sparrow	22	11	12	9	8	7	7	10	10		7	4	8	4	5	8	7	10	8.76	4.10	0.329	-3.34**
Dark-eyed Junco	9	5	3	2	5	2	5	4	4		7	5	1	4	1	5	6	6	4.41	2.15	0.007	-0.71
Purple Finch	2	1	4	4	2	4	4		6					2	1	5	1		2.29	1.93		
White-winged Crossbill	8		2		1		6												1.00	2.35		
Pine Siskin	12		1		7								1		1				1.29	3.24		
Species Richness ^a	19	13	18	17	16	17	18	12	15		17	15	16	16	14	15	11	13	15.41	2.24		
Number of Individuals ^a	107	71	88	65	67	75	69	82	82		74	61	62	59	37	79	52	63	70.18	15.55		

^a Does not include counts of Eastern Chipmunk or Red Squirrel; * $P = 0.039$; ** $P \leq 0.012$

Table 2.3. Maximum counts of individual birds, and population trends from linear regression analysis for the 8 most common species (bold type) at Underhill State Park, 1991-2011. Note that surveys were not conducted in 2003, 2005, or 2012.

Common Name	'91	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	'11	'12	Mean	SD	r ²	Annual Trend (%)
Eastern Chipmunk							3	5					/	1	/		1						0.53	1.34		
Red Squirrel							1	3		1			/		/	1	1			1			0.42	0.77		
Broad-winged Hawk							1						/		/								0.05	0.23		
Mourning Dove									1				/	1	/								0.11	0.32		
Yellow-bellied Sapsucker		2		1	1		1	1	1		3		/	2	/	2	3	2					1.00	1.05		
Downy Woodpecker							1						/		/		1	1		1			0.21	0.42		
Hairy Woodpecker				1			1	1	2				/		/				2	2			0.47	0.77		
Northern Flicker			1										/		/								0.05	0.23		
Pileated Woodpecker	2	1	1			1							/		/								0.26	0.56		
Least Flycatcher													/		/		2						0.11	0.46		
Eastern Phoebe												1											0.05	0.23		
Blue-headed Vireo	1	2				1	1			1			/	1	/	2	1	1	3	3			0.89	0.99		
Red-eyed Vireo	3	4	4	6	9	8	7	6	10	8	8	7	/	5	/	7	8	6	2	5	4		6.16	2.14	0.009	-0.48
Blue Jay	2	1		1		2	2		1	1	2	1	/	1	/		1	1		1			0.89	0.74		
American Crow													/		/	1		1			1		0.16	0.37		
Common Raven				4	1				1	1			/	1	/					2			0.53	1.02		
Black-capped Chickadee		1	1		2	3	3		3	1	1		/		/	2	1	3		2	2		1.32	1.16		
Red-breasted Nuthatch							1						/		/								0.05	0.23		
White-breasted Nuthatch							1						/	1	/								0.11	0.32		
Brown Creeper				1					1	1		1	/	1	/	1	1						0.37	0.50		
Winter Wren		6	2	1	5	3	4	6	4	4	3	3	/	3	/	4	2	1			2		2.79	1.87	0.136	-2.73
Golden-crowned Kinglet								1					/		/	1							0.11	0.32		
Veery	1	1								1			/		/								0.16	0.37		
Swainson's Thrush		1		2	4	3		1	4	2	2		/		/	1			2				1.16	1.38		
Hermit Thrush		4	1	6	7	3	4	4	2		4	5	/	4	/	4	7	1	4	3	4		3.53	2.04	0.017	1.35
Wood Thrush	1	1											/		/								0.11	0.32		
American Robin	1				3	3	3	4	2	1	2	1	/	2	/		1				3		1.37	1.34		
Cedar Waxwing													/		/						1		0.05	0.22		
Nashville Warbler													/		/						1		0.05	0.22		
Magnolia Warbler	1				1								/		/	1							0.16	0.37		
Black-th. Blue Warbler	4	9	5	6	7	8	6	5	6	5	5	5	/	11	/	15	8	11	5	14	9		7.58	3.20	0.285	5.59*
Yellow-rumped Warbler			2	2		2	3	3	1	1	3	2	/		/	1		1	1	1	1		1.26	1.05		

Continued

Common Name	'91	'92	'93	'94	'95	'96	'97	'98	'99	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	'11	'12	Mean	SD	r ²	Annual Trend (%)
Black-th. Green Warbler	5	7	6	7	7	7	9	5	8	10	10	8	/	13	/	15	12	10	7	11	14	/	9.00	2.94	0.531	6.11**
Blackburnian Warbler											1	1	/	/	/	1				1		/	0.21	0.42		
Blackpoll Warbler						1	2						/	/	/							/	0.16	0.50		
Black-and-White Warbler		3	2	2	4	2	3	2	1	3	4	2	/	1	/	2	3		1	1	2	/	2.00	1.15		
American Redstart		4			1	1							/	/	/							/	0.32	0.95		
Ovenbird	4	10	11	11	13	12	12	10	13	10	13	6	/	11	/	11	15	14	7	14	10	/	10.89	2.81	0.064	1.13
Mourning Warbler													/	/	/		1	1				/	0.11	0.32		
Canada Warbler	3	4	4	6	2	4	4	2	2	3	2	2	/	/	/			1				/	2.05	1.81	0.732	-5.20***
Scarlet Tanager					1				1				/	/	/	1						/	0.16	0.37		
White-throated Sparrow	2		2	1	1		1					1	/	/	/	1		1		1		/	0.58	0.69		
Dark-eyed Junco		3	1	3	4	3	5	2	2	1	2	2	/	1	/	5	5	2	4	3	1	/	2.58	1.50	0.035	2.05
Rose-breasted Grosbeak	4	2		1	3	1	2		1				/	/	/	1					1	/	0.84	1.17		
Purple Finch						1		1			1		/	/	/	1	1		1			/	0.32	0.48		
White-winged Crossbill											2		/	/	/							/	0.11	0.46		
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	14	/	17.63	2.91		
Number of Individuals ^a	35	66	43	62	77	69	77	54	67	53	70	48	/	60	/	81	73	58	39	68	53	/	60.68	13.23		

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

* $P = 0.019$

** $P = 0.0004$

*** $P < 0.0001$

Table 2.4. 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, 2000-2011. Note that surveys were not conducted in 2012.

Common Name	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	'11	'12	Mean	SD	r ²	Annual Trend (%)
Eastern Chipmunk	2			1										0.25	0.62		
Red Squirrel	1	1												0.17	0.39		
Ruffed Grouse	1					2								0.25	0.62		
Mourning Dove		1												0.08	0.29		
Yellow-Billed Cuckoo									1					0.08	0.29		
Barred Owl	1													0.08	0.29		
Chimney Swift	2													0.17	0.58		
Ruby-throated Hummingbird									1	1		1		0.25	0.45		
Yellow-bellied Sapsucker	5	6			2		2	2	5	8	11	2		3.58	3.48	0.119	23.32
Downy Woodpecker	1		1											0.17	0.39		
Hairy Woodpecker	2	1	2						1	1		5		1.08	1.44		
Unidentified Woodpecker	3													0.25	0.87		
Northern Flicker									1					0.08	0.29		
Pileated Woodpecker	1		3	1	4	1	1		2	1	2			1.33	1.23	0.021	-2.96
Eastern Wood-Pewee				1										0.08	0.29		
Yellow-bellied Flycatcher							1							0.08	0.29		
Least Flycatcher	2													0.17	0.58		
Great Crested Flycatcher				1										0.08	0.29		
Blue-headed Vireo		1	4	1		1				1		1		0.75	1.14		
Red-eyed Vireo	10	6	9	4	6	6	4	5	13	14	10	15		8.50	3.92	0.285	12.28*
Blue Jay		3		1		1			2	1	3	1		1.00	1.13		
Common Raven					1	1								0.18	0.40		
Black-capped Chickadee	1	1		2			1	2	1		1	1		0.83	0.72		
White-breasted Nuthatch						1	1					1		0.25	0.45		
Brown Creeper	1											2		0.25	0.62		
Winter Wren	7		1		3	1			2					1.17	2.08		
Ruby-crowned Kinglet						1						1		0.17	0.39		
Veery					1									0.08	0.29		
Swainson's Thrush	2		1	3	2		2	1	1	2				1.17	1.03		
Hermit Thrush	4	2	6	5	4	4	4	5	6	7	8	2		4.75	1.82	0.107	4.46
American Robin	1		1		3			1	1		2	2		0.92	1.00		
Cedar Waxwing	1								1					0.17	0.39		
Northern Parula				3	1									0.33	0.89		
Magnolia Warbler	1		3											0.33	0.89		
Black-throated Blue Warbler	9	7	10	9	8	12	11	8	8	8	5	7		8.50	1.88	0.161	-2.13
Yellow-rumped Warbler	2	1				0				1		1		0.42	0.67		
Black-throated Green Warbler	8	10	4	6	8	9	12	3	11	9	6	10		8.00	2.76	0.024	1.64

Common Name	'00	'01	'02	'03	'04	'05	'06	'07	'08	'09	'10	'11	'12	Mean	SD	r ²	Annual Trend (%)
Blackburnian Warbler	5													0.42	1.44		
American Redstart	2	1	3	1		4								0.92	1.38		
Ovenbird	15	13	19	11	14	13	12	12	8	12	10	10		12.42	2.81	0.471	-3.37**
Canada Warbler	1													0.08	0.29		
Scarlet Tanager	1		3	2	2	2			1		1			1.00	1.04		
White-throated Sparrow	2		2	4		2								0.83	1.34		
Dark-eyed Junco	2	3	1	1	1	4		1				2	1	1.33	1.23	0.136	-5.85
Rose-breasted Grosbeak	2	1												0.25	0.62		
Species Richness^a	28	15	17	17	16	17	11	11	18	13	12	17		16.00	4.55		
Number of Individuals^a	98	58	73	57	60	65	51	41	66	66	61	63		63.25	13.65		

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

* $P = 0.074$, ** $P = 0.014$

Part III. Migratory movements and distribution of Bicknell's Thrush (*Catharus bicknelli*): gaining insights from geolocators

Kent McFarland, Rosalind Renfrew and Chris Rimmer
Vermont Center for Ecostudies
PO Box 420, Norwich, VT 05091
www.vtecostudies.org

Jason Townsend
State University of New York College of Environmental Science and Forestry
1 Forestry Drive
Syracuse, NY 13210

Becky Whittam and Greg Campbell
Bird Studies Canada – Atlantic Region
PO Box 6227, Sackville, NB E4L 1G6

Yves Aubry and Sebastien Paradis
Canadian Wildlife Service – Quebec Region
Environment Canada
PO Box 10100
Ste-Foy, QU G1V 4H5

Peter Marra
Smithsonian Migratory Bird Center
National Zoological Park
PO Box 37012 MRC 5503
Washington, DC 20013-7012
www.si.edu/smbc

Vsevolod Afanaseyev and James W. Fox
British Antarctic Survey
High Cross, Madingley Road
Cambridge CB3 0ET
United Kingdom
www.antarctica.ac.uk/

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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 their migratory routes and stopovers linking the regions. An understanding of these factors, which could operate in breeding and/or during non-breeding periods, that 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 the breeding and wintering grounds. This is critical for understanding how limiting factors (e.g. habitat destruction, climate change, etc.) operate in different parts of the birds' annual cycle and for determining population size and local abundance.

Migratory connectivity is defined as the amount of population mixing between summer-breeding, 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), and although stable isotope measures have provided latitudinal gradients of habitat occupancy and a tool for assessing migratory connectivity (Rubenstein et al. 2002), this technique still provides only a blunt instrument to assess migratory connectivity and can sometimes fail to provide clear, meaningful information (Hobson et al. 2001). Many migratory songbirds travel distances of over 5,000 km between breeding and wintering sites, yet the exact timing, pathways 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 (Townsend and Rimmer 2006, Rimmer and McFarland 2001). The number of other Nearctic-Neotropical songbird species in which an individual marked in one portion of its range has been 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 (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 United States following increases in growing-season temperatures that are projected to occur this century (Rodenhouse et al. 2008). Yet, incomplete information on fundamental aspects of the ecology and demographics throughout the annual cycle preclude a full understanding of the limiting factors underlying 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 population connectivity as one of the species' most pressing conservation needs.

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 (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).

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, citation KPM). 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 unpub. data, 2005). 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.

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 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 remain a mystery.

New technology has the potential to shed light on these gaps in our understanding, with significant conservation implications for Bicknell's Thrush. The recent miniaturization of daylight level data recorders (geolocators) for tracking small animals over long periods of time has greatly increased our understanding for many migratory birds. Here we, (1) evaluate their utility for providing locations of forest understory species that may be shaded from direct sunlight; (2) determine the strength of population connectivity between breeding and winter regions; and (3) elucidate patterns of migratory movement and stopover.

Methods

Study Sites- During the 2009 breeding season at seven study areas scattered across the entire breeding range we capture adult male Bicknell's Thrushes by passive netting and using playback of conspecific vocalizations to attract birds into nylon mist nets (12 x 2.5-m and 6 x 2.5-m, 36-mm mesh). 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 more 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 (even sex ratio site).

Geolocators- 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 were recaptured up to four weeks after attachment were in good condition and within normal weight ranges. The total attachment weighed ~1.4g which corresponded to ~5% mean male body mass (28.18g; Rimmer et al. 2001).

Data analysis- We analyzed data from geologgers using the single threshold method (Lisovski et al. 2012) with the program suite BASTrak (BAS). We adjusted for clock drift when needed, and used a light threshold of 2. We visually inspected sunrise and sunset transitions, deleting those with very shallow slopes or with peaks in light intensity within thirty minutes of sunrise or

sunset. In cases when a transition was shaded just before reaching the light threshold, we manually inserted a sunrise or sunset where the slope would have reached the threshold.

For periods when birds were on relatively small breeding or winter home ranges, we calibrated each geolocator using data it collected while on the bird from 1 to 14 August, when birds are still on territory but no longer attending nests, yielding data that are relatively less shaded. For 6 birds, however, shading was excessive during this period, and instead we used the average of calibration values for all other birds (-3.658 , $n = 18$). For migration and wintering grounds, we calibrated location data using a static geolocator placed in the understory in a shaded area, to mimic BITH behavior in winter in known BITH habitat on the wintering grounds in PUVI, DR (lat 18.20461, long -71.53088) from 15 – 21 March.

We used only longitude data within 20 days of fall and spring equinox. We deleted an average of 42 days per geolocator due to shading, which varied substantially among units ($SD = 46.1$). We manually inserted an average of 16.4 ($SD = 8.7$) and 9.1 ($SD = 7.4$) sunrises and sunsets per geolocator, respectively.

We mapped point locations using both noon and midnight fixes. We mapped breeding areas using kernel density estimator analysis in the Spatial Analyst tool of ArcMap 10 (ESRI, Redlands, CA), using a search radius of 200 km (slightly less than mean latitudinal error on breeding grounds) and a grid cell size of 2 km (Bachler et al. 2010).

Results and Discussion

Overall return rates of geolocated birds on the breeding grounds were lower than expected, 26% versus the long-term average of 65% for Vermont-banded adults. Recoveries of geolocators themselves were low ($n = 21$; Table 1). We discovered that the Kevlar thread used to secure the 2009 backpack harnesses had abraded and broken with 2 of the 3 birds recovered having intact geolocators with 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 geolocator harnesses.

During November of 2010, we returned to both study sites on the winter grounds in the Dominican Republic for geolocator 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.

The retrieved data showed extreme shading for much of the year, indicating that individuals are often under heavy forest canopy cover during daily sun transitions. We used winter longitude estimates to determine approximate wintering locations. All individuals were predicted to be on Hispaniola based on the mean longitude location (Figure 1). Even using 95% confidence limits, only half of the individuals had predicted locations that included other islands (e. Cuba and Puerto Rico).

As an example, 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. 2). During fall migration between 19-26 October 2009, it stopped in the eastern Bahamas for approximately six days. During spring migration northward 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.

As shown by longitude estimates, at least seven thrushes appeared to have substantial stopover periods during fall migration (Fig. 3). We cannot determine the exact locations without more precise latitude estimates, but it appears that thrushes resided for up to two weeks in either small islands in the eastern Bahamas or far eastern Cuba in late October and early November before proceeding to their winter grounds on Hispaniola.

Estimates of breeding areas of four thrushes from the wintering grounds yielded more precise data (Fig. 4). Thrushes presumably have less shading events during critical sunrise and sunset periods in the breeding grounds as they are often more exposed while singing and calling from perches. Three of the individuals were estimated to be in Canada, while one individual from Pueblo Viejo was estimated to be in the northern Green Mountains, Vermont.

As previously described from banding data (Rimmer et al. 2001), the overall migration pattern did appear to have a "loop" pattern with fall migration farther to the east and spring migration through Florida and then northward. We estimated mean spring migration to last 20 days (± 3.6 SD) (Fig. 5), not significantly longer than Wood thrushes at 17.9 days (± 7 SD) (Stutchbury et al. 2009). Mean fall migration was estimated to last 32 days (± 11.3 SD) (Fig. 6), slightly shorter than Wood thrushes at 37.3 days (± 12.3 SD) (Stutchbury et al. 2009).

Our findings suggest that Bicknell's thrush may have a unique and important stopover area near their overwinter grounds. This phenomenon may be an important consideration for future conservation initiatives and warrants further investigation.

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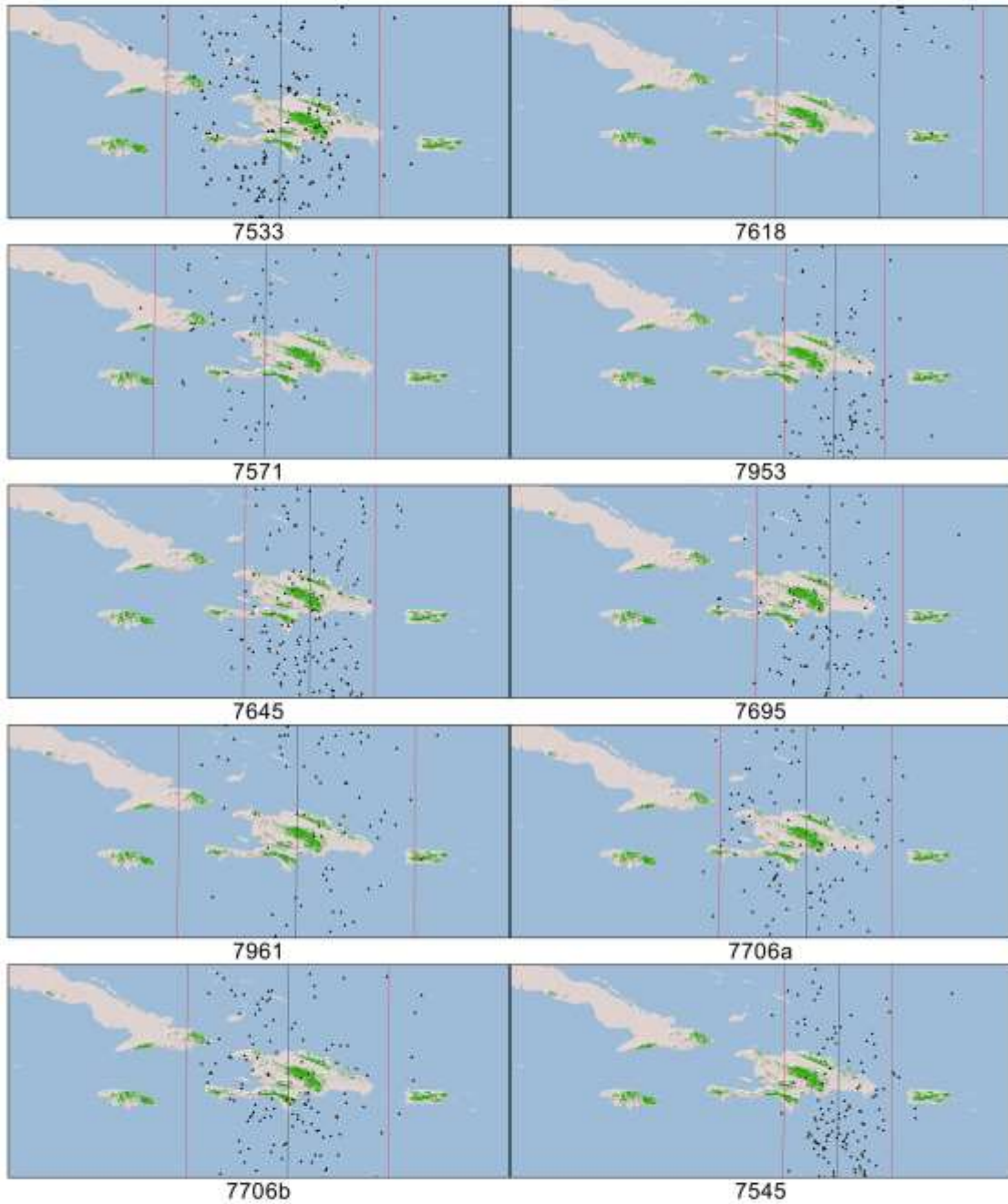
Table 1. Geolocator deployment and recovery locations for Bicknell's Thrushes, 2009-2011.

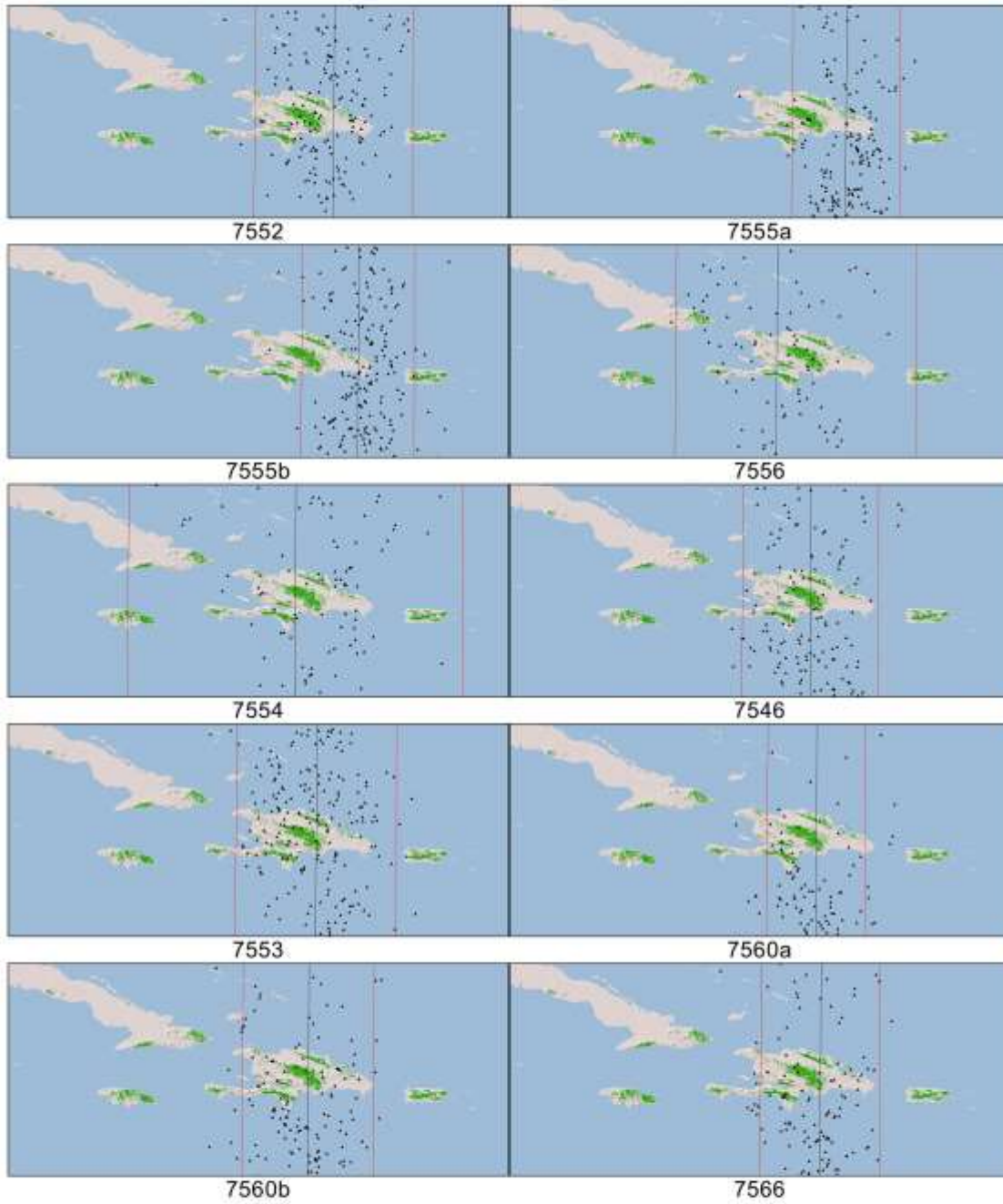
Site	Deployed	Recovered	Missing Geolocator	Deployed	Recovered	Recovered	TOTAL Recovered
	2009	2010	2010	2010	2011	2011 (with 2 yrs of data)	
Plateau Mt. (NY)	5	0	0				0
Slide Mt. (NY)	17	1	4				1
Mt. Mansfield (VT)	14	2	4	22	2 ^b	1	6
Mt. Washington (NH)				4	0		0
Mt. Jefferson (NH)				4	1		1
Mt. Gosford (QC)	10	3	0				3
Massif-du-Sud (QC)	6	3	0			1	4
Gaspesie (QC)	9	2	0			1	3
New Brunswick Highlands	5	0	0	7	1		1
Cape Breton Highlands (NS)	4	2	1	7	1		4
Pueblo Viejo (DR) ^a	15	2	2				2
Loma Guaconejo (DR) ^a	13	2	0				2
Total	98	17	11	44	6	3	27

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

^b two geolocators recovered on Mansfield in 2011 stopped working in winter grounds. No spring migration data.

Figure 1. Estimated winter locations of Bicknell's thrush. Black line is the mean longitude with the 95% confidence area indicated by red lines.







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Figure 2. 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.

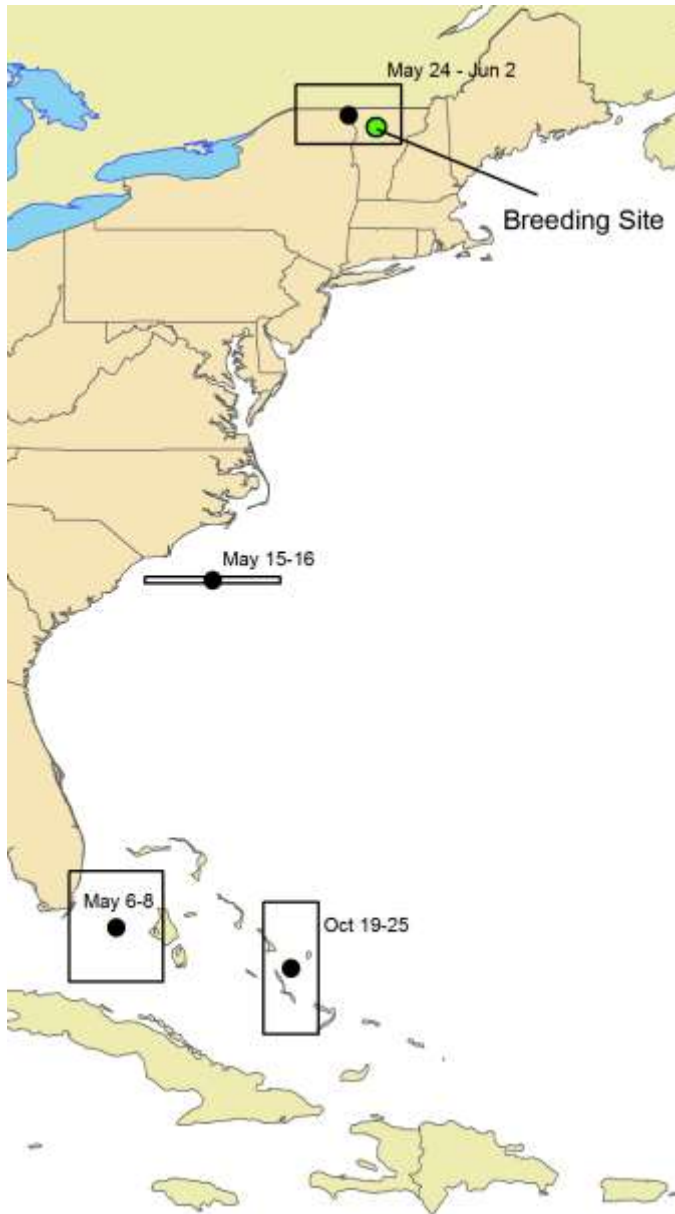


Figure 3. Potential fall stopover regions of Bicknell's thrush ($n=7$ individuals) during late October and early November. Light black dotted line is the mean longitude of stopover location with the 95% confidence area indicated by red dotted lines. The estimated winter location is indicated by the black line farther to the east in all cases.

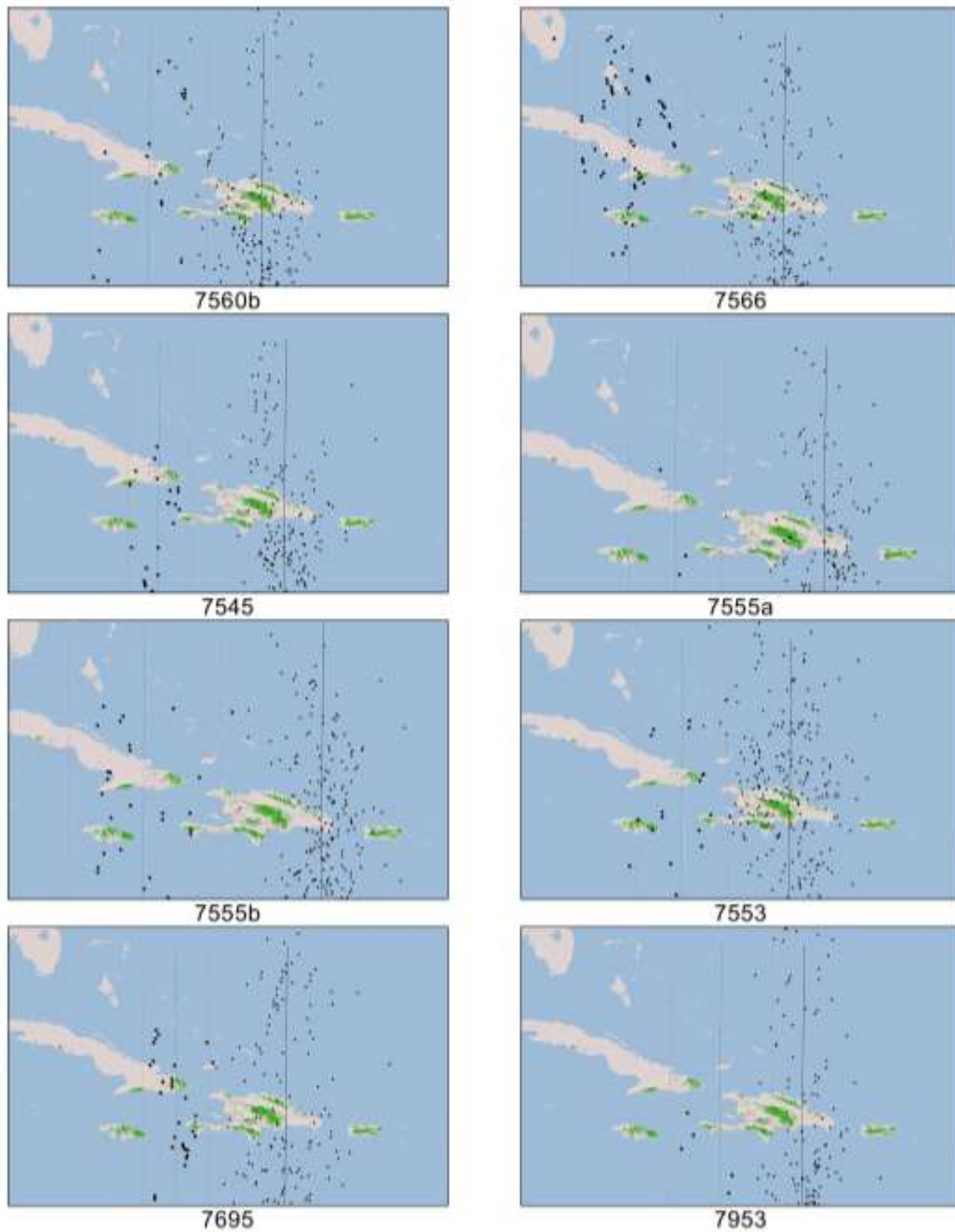


Figure 4. Estimated breeding locations of four Bicknell's Thrushes that wintered on Hispaniola. Green areas are potential Bicknell's thrush breeding habitat.

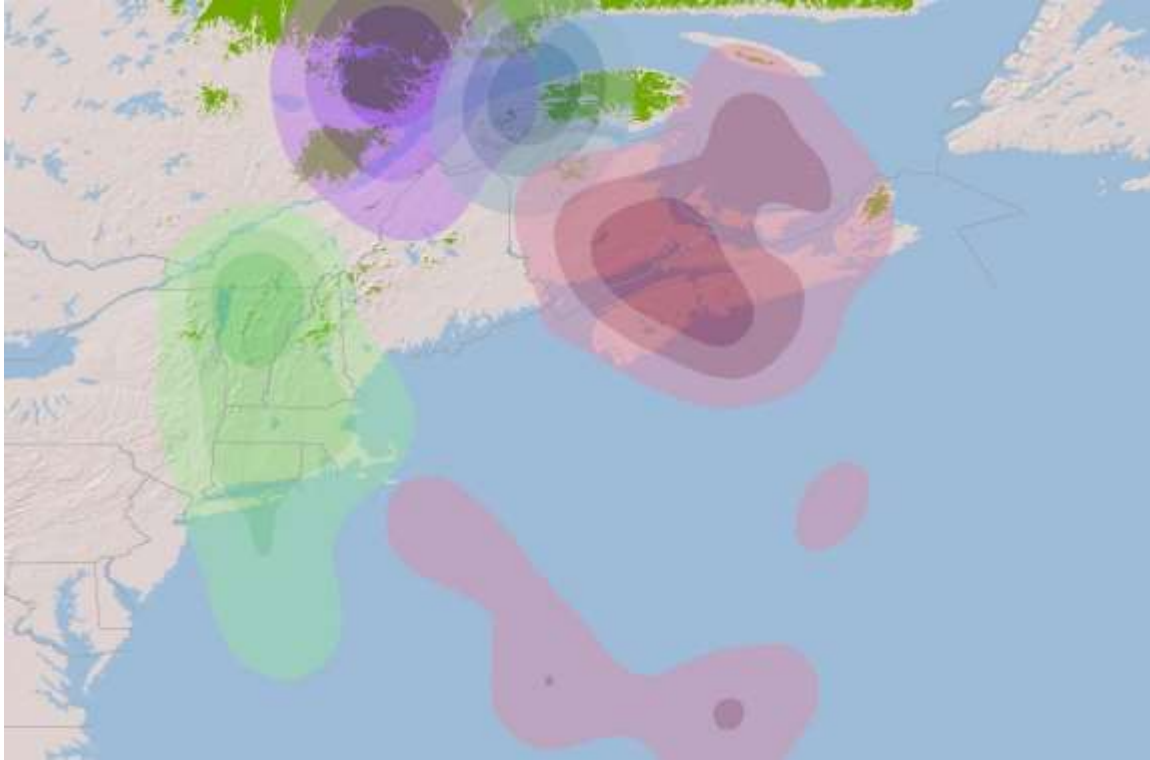


Figure 5. Estimated length of spring migration by Bicknell's thrush compared to latitude of breeding site.

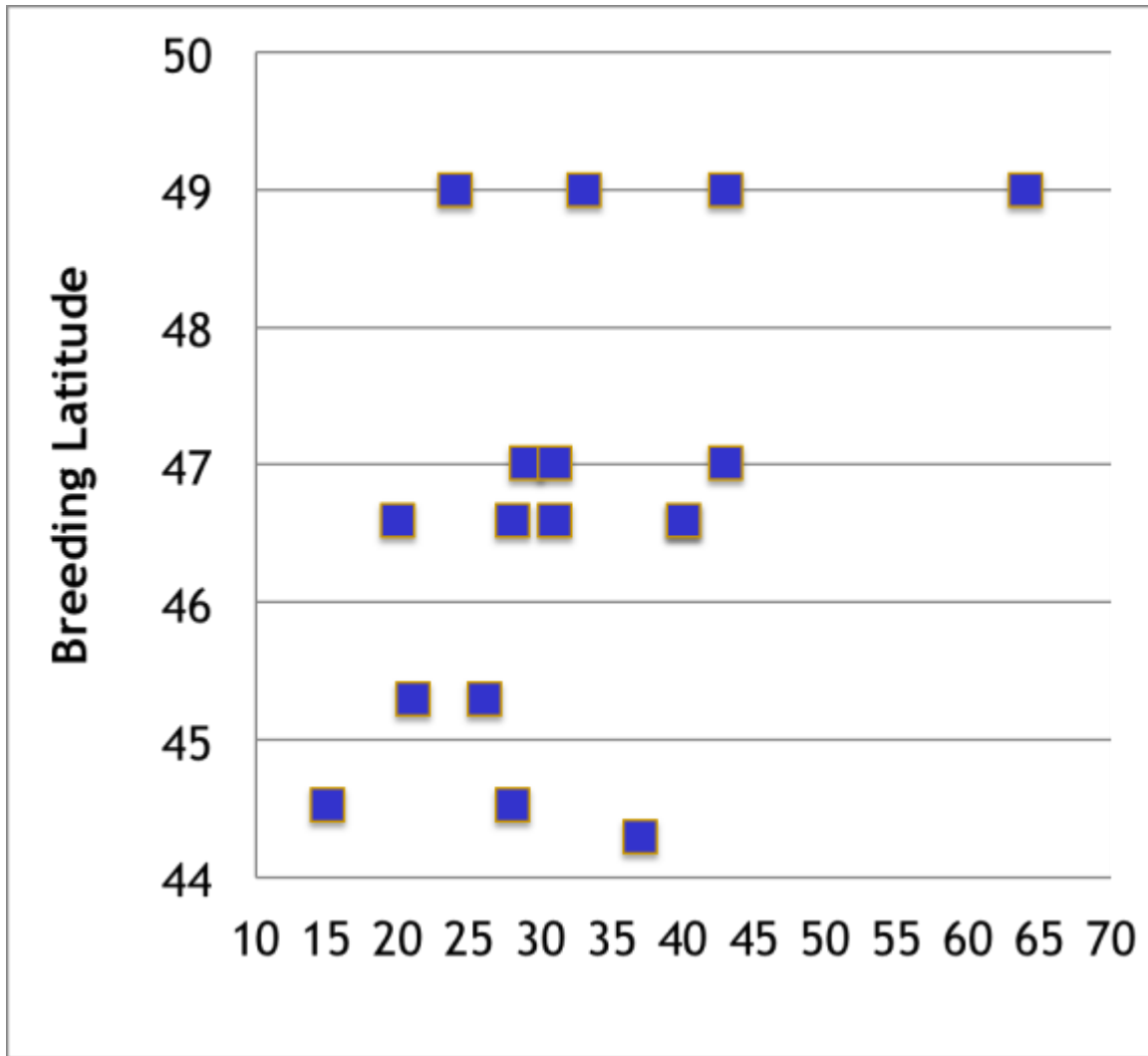
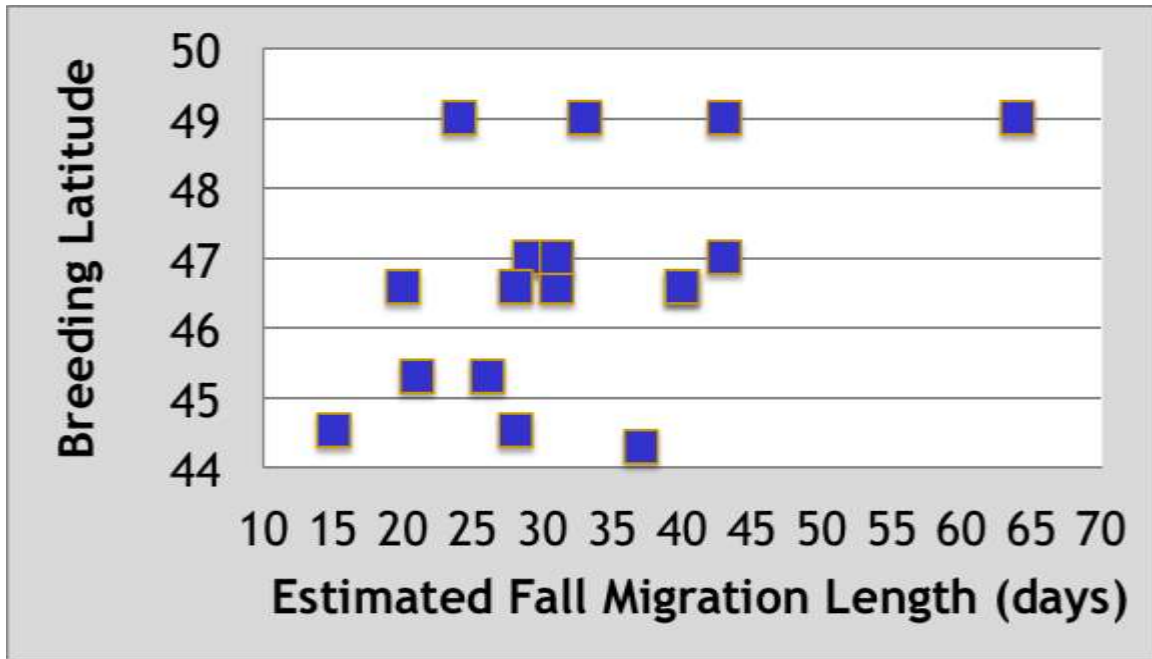


Figure 6. Estimated length of fall migration by Bicknell's thrush compared to latitude of breeding site.



Appendix A. VCE peer-reviewed papers published in 2011-2013 utilizing data from Mount Mansfield, Vermont.

Studds, C. E., McFarland, K. P., Aubry, Y., Rimmer, C. C., Hobson, K. A., Marra, P. P. and Wassenaar, L. I. 2012. Stable-hydrogen isotope measures of natal dispersal reflect observed population declines in a threatened migratory songbird. *Diversity and Distributions* 18: 919–930. <http://www.vtecostudies.org/PDF/bithnataldispersal2012.pdf>

Abstract: Measuring dispersal is crucial for estimating demographic rates that inform conservation plans for rare and threatened species. We evaluated natal dispersal patterns in Bicknell's thrush (*Catharus bicknelli*) across most of the breeding range using a 10-year data set of stable-hydrogen isotope ratios in feathers (δ^2HF) grown on the natal area and sampled 1 year later at the first breeding site. We used δ^2HF values of adult thrushes sampled at 25 breeding sites as prior information for assigning first-time breeders to their natal site. We calculated the minimum distance birds moved from their natal to first breeding site and fit these data to three statistical distributions for characterizing the importance of long-distance dispersal: the exponential, Weibull and half-Cauchy. Finally, we assessed differences in the probability of dispersal across the breeding range and through time to understand spatio-temporal variation in demographic connectivity. The δ^2HF values of first-time breeders were lower compared with those of adults, a difference that was greater at the southern compared with northern breeding range extreme. Assignment tests accounting for age differences in δ^2HF suggested that most birds dispersed < 200 km from their natal area and within the centre of the breeding range, whereas comparatively few individuals dispersed up to 700 km. A Weibull distribution provided the best fit to these data. Two of three corrections for age differences in δ^2HF indicated that natal dispersal probability declined by 30–38% from 1996 to 2005. Our findings suggest that estimating natal dispersal with δ^2HF measurements may contribute to understanding the resilience of geographically isolated Bicknell's thrush populations. Declining natal dispersal may be symptomatic of observed population declines and could compound this trend by limiting demographic exchange between habitat patches predicted to be increasingly isolated by natural and anthropogenic habitat changes.

Frey, S. J. K., A. M. Strong, and K. P. McFarland. 2011. The relative contribution of local habitat and landscape context to metapopulation processes: a dynamic occupancy modeling approach. *Ecography* 34:001-009. <http://tinyurl.com/nnyck4l>

Abstract: Changes in site occupancy across habitat patches have often been attributed to landscape features in fragmented systems, particularly when considering metapopulations. However, failure to include habitat quality of individual patches can mask the relative importance of local scale features in determining distributional changes. We employed dynamic occupancy modeling to compare the strength of local habitat variables and metrics of landscape patterns as drivers of metapopulation dynamics for a vulnerable, high-elevation species in a naturally fragmented landscape. Repeat surveys of Bicknell's thrush (*Catharus bicknelli*) presence/non-detection were conducted at 88 sites across Vermont, USA in 2006 and 2007. We used an organism-based approach, such that at each site we measured important local-scale habitat characteristics and quantified landscape-scale features using a predictive habitat model for this species. We performed a principal component analysis on both the local and landscape

features to reduce dimensionality. We estimated site occupancy, colonization, and extinction probabilities while accounting for imperfect detection. Univariate, additive, and interaction models of local habitat and landscape context were ranked using AICc scores. Both local and landscape scales were important in determining changes in occupancy patterns. An interaction between scales was detected for occupancy dynamics indicating that the relationship of the parameters to local-scale habitat conditions can change depending on the landscape context and vice versa. An increase in both landscape- and local-scale habitat quality increased occupancy and colonization probability while decreasing extinction risk. Colonization and extinction were both more strongly influenced by local habitat quality relative to landscape patterns. We also identified clear, qualitative thresholds for landscape-scale features. Conservation of large habitat patches in high-cover landscapes will help ensure persistence of Bicknell's thrushes, but only if local scale habitat quality is maintained. Our results highlight the importance of incorporating information beyond landscape characteristics when investigating patch occupancy patterns in metapopulations.

Rimmer, C.C. and K.P. McFarland. 2013. Bicknell's Thrush: a twenty-year retrospective on the Northeast's most vulnerable songbird. Bird Observer 41:9-16.
[http://www.vtecostudies.org/PDF/BITH Bird Obs CCR-KPM Feb13.pdf](http://www.vtecostudies.org/PDF/BITH%20Bird%20Obs%20CCR-KPM%20Feb13.pdf)