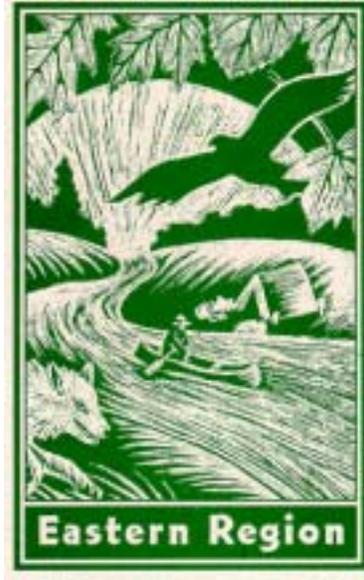


*Conservation Assessment
for
Bicknell's Thrush (Catharus bicknelli)*



USDA Forest Service, Eastern Region

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This document is undergoing peer review, comments welcome

This Conservation Assessment/Approach was prepared to compile the published and unpublished information on the subject taxon or community; or this document was prepared by another organization and provides information to serve as a Conservation Assessment for the Eastern Region of the Forest Service. It does not represent a management decision by the U.S. Forest Service. Though the best scientific information available was used and subject experts were consulted in preparation of this document, it is expected that new information will arise. In the spirit of continuous learning and adaptive management, if you have information that will assist in conserving the subject taxon, please contact the Eastern Region of the Forest Service Threatened and Endangered Species Program at 310 Wisconsin Avenue, Suite 580 Milwaukee, Wisconsin 53203.

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EXECUTIVE SUMMARY

Believed to number fewer than 100,000 individuals across its restricted northeastern U.S. and maritime Canada breeding range, Bicknell's Thrush (*Catharus bicknelli*) is one of eastern North America's most at-risk migratory songbirds. The species is an extreme habitat specialist, nesting in the U.S. only in fir-dominated montane forests above 900 m in elevation. These forests face numerous threats that include global climate change, atmospheric deposition of acidic ions and mercury, recreational ski area development, telecommunications tower construction, and wind turbine development. On its Greater Antillean wintering grounds, Bicknell's Thrush is largely restricted to moist, primary broadleaf forests, many of which have been severely reduced in extent. Most of the global wintering population probably occurs in the Dominican Republic, where few habitat protection measures are in place. The species is poorly monitored by traditional sampling methods, and estimates of breeding or wintering densities are unreliable at best. An elevation-based model of Bicknell's Thrush habitat in the U.S. indicates 110,934 ha of potentially suitable conifer-dominated montane forest habitat; of this, New Hampshire accounts for 45%, Maine 23%, the Adirondack Mountains of New York 23%, Vermont 8%, and the Catskill Mountains of New York <1%. Applying assumptions from recent research of an approximately 2:1 male:female breeding sex ratio and male densities of 0.4-0.79 birds/ha to the total calculated habitat of 110,934 ha yields very coarse estimates of 44,374 – 87,638 males and 22,187 – 43,819 females, or 66,561 – 131,457 total individuals in the U.S. It is almost certain that these upper limits significantly overestimate the U.S. breeding population of Bicknell's Thrush, as they further assume habitat saturation. Analyses of existing point count data from the Green and White mountains yield conflicting trends, with Bicknell's Thrush showing a non-significant annual increase of 3.9% in Vermont and a nearly-significant annual decline of -8.3% in New Hampshire. Power analyses indicate that at least 10-15 additional years of data collection will be needed to detect significant annual population declines of <5%.

INTRODUCTION

One of eastern North America's most rare and range-restricted breeding passerines, Bicknell's Thrush (*Catharus bicknelli*) is a natural disturbance habitat specialist. Historically, the species probably selected patches of regenerating forest caused by fir waves, wind throw, ice and snow damage, fire, and insect outbreaks, as well as chronically-disturbed, stunted altitudinal and coastal conifer forests (Ouellet 1993, Nixon 1999, Vermont Institute of Natural Science [VINS] unpubl. data). In addition to these natural successional habitats, Bicknell's Thrush has recently been discovered in areas anthropogenically disturbed by timber harvesting, ski trail and road construction, and other human activities (Ouellet 1993, Rimmer et al. 2001). Evidence of local declines and extinctions in "traditional" breeding habitats is suggested to indicate either a shift in habitat use or expanding populations (Ouellet 1993, 1996), but more likely reflects the species' opportunistic use of disturbed habitats. Extensive loss and degradation of the primary forests that Bicknell's Thrush appears to prefer in winter pose the greatest threat to the species' long-term viability.

Despite detailed studies by Wallace (1939), VINS and others, few concrete data are available by which to assess the conservation status of Bicknell's Thrush. The species is poorly

monitored by traditional sampling methods, and its unusual spacing and mating system makes estimation of breeding densities unreliable at best. Current rangewide population estimates represent little more than educated guesses. Knowledge of the species' wintering ecology and demography is fragmentary, and its migratory routes and stopover ecology are poorly known. Recent research on the breeding and behavioral ecology of Bicknell's Thrush has documented a strongly male-biased sex ratio, with 2-4 males feeding young at 75% of nests and multiple paternity of most broods (Rimmer et al. 2001). Possible sexual habitat or geographic segregation on wintering grounds may cause differential survivorship of females and promote a skewed breeding sex ratio, but firm evidence is lacking. Much work remains to be done on Bicknell's Thrush at all stages of its annual cycle and in all parts of its range.

The recent elevation of Bicknell's Thrush to full species status has heightened interest and concern among birders, scientists, land use planners, and conservationists. It is currently recognized as one of the most at-risk passerines in eastern North America. Partners in Flight ranks Bicknell's Thrush as the top conservation priority among Neotropical migrants in the Northeast (Pashley et al. 2000), while the International Union for the Conservation of Nature classifies the species as "vulnerable" on its worldwide list of threatened birds (BirdLife International 2000).

NOMENCLATURE AND TAXONOMY

Following its discovery in 1881 by E. Bicknell on Slide Mountain in New York's Catskill range, Bicknell's Thrush was classified as a subspecies (*Catharus minimus bicknelli*) of the Gray-cheeked Thrush (*C. m. minimus*; Ridgway). Wallace's (1939) classic natural history study and a careful taxonomic assessment by Ouellet (1993) led to specific recognition in 1995 (AOU 1995). Although reliable field identification of Bicknell's (now *C. bicknelli*) and Gray-cheeked (now *C. minimus*) thrush remains problematic, marked morphological, vocal and biochemical differences between the two taxa support this designation. The ranges are completely allopatric, with Gray-cheeked breeding further north (Newfoundland to Siberia) and wintering further south (Panama through northwestern Brazil and Colombia) than Bicknell's Thrush. No subspecies of Bicknell's Thrush are recognized.

There are indications of latitudinal variation in this species, both in size and dorsal coloration, but rigorous study is still needed (Todd 1963, Ouellet 1993). Todd (1963) proposed the possibility of a tawnier brown montane subspecies in New York and New England, and a colder olive-brown subspecies in the Canadian Maritime Provinces and southeastern Quebec. He further suggested that the brown versus olive color polymorphism seen in northern Vermont by Wallace (1939) represents contact between these forms. It is now unclear if the trend from brown birds in the south to olive birds in the north represents a true cline or if the two forms are intermixed throughout the range. It should be clarified whether this is true polymorphism, or only the separation of extremes in normal variation in dorsal color.

DISTINGUISHING CHARACTERISTICS

Bicknell's Thrush is a medium sized thrush (16-17 cm in length, 26-30 gm in weight), but smallish and slender for a *Catharus*. Although generally wary and hard to observe, the species occasionally sings on exposed song posts. Field identification is subtle and difficult under the best of circumstances. Plumage separation from the very similar Gray-cheeked Thrush relies on slight color differences and contrasts (e.g. tail versus lower back), which are less useful than soft part color and morphometrics (Ouellet 1993, Knox 1996). The body coloration of both species varies across their respective breeding ranges, obscuring differences in all but extreme variants. Most *bicknelli* show olive brown or brown dorsal coloration, most *minimus* olive gray or olive (Ouellet 1993). In comparison to Gray-cheeked, Bicknell's Thrush shows contrast between its chestnut-tinged tail and wings and the remaining upperparts. The species also shows warmer brown upperparts and a lighter buffy wash backing its spotted breast than continental subarctic Gray-cheeked Thrush (*C. m. aliciae*). This, combined with the bright yellow to yellow-orange basal half or more of its lower mandible, provides a subtle but generally reliable method of separating Bicknell's from *aliciae* Gray-cheeked Thrush. In *bicknelli*, color of the legs is purplish flesh, with the toes darker than tarsi and soles of the feet flesh to dull pale yellow; in *minimus*, the tarsi are a lighter flesh color, with the toes invariably much darker and soles of the feet brighter yellow than in *bicknelli* (Ouellet 1993).

Bicknell's Thrush is best identified in the hand on basis of its size and relative wing shape (Pyle 1997). Bicknell's is usually smaller than Gray-cheeked, although considerable overlap in measurements exists. Wing chords of adult *bicknelli* range 82-100 mm ($n = 415$; VINS), of *minimus* 93-109 mm ($n = 200$; Pyle 1997). Tail length of *bicknelli* ranges 60-75 mm ($n = 127$, VINS), of *minimus* 63-79 mm ($n = 185$; Pyle 1997). The majority of Gray-cheeked Thrushes have wings > 95 mm in length (Ouellet 1993), and 85% of Bicknell's have wings < 95 mm (VINS); however, most birds in the 94-98 mm overlap range (especially young female *C. m. minimus* and adult male *bicknelli*) are not safely identifiable. As befits a longer distance migrant, Gray-cheeked Thrush shows a more pointed wing morphology (Phillips 1991, Pyle 1997). Difference in length between primaries 8 and 6 averages 3-7 mm for *bicknelli* and 5-10 mm for *minimus*; primary 8 is 24-29 mm longer than primary 1 in *bicknelli*, 27-35 mm longer in *minimus* (Pyle 1997). The ratio of primary:tertial length may be useful in separating the two species: $\leq 1:1$ in *bicknelli*, $\geq 1:1$ in *minimus* (Lane and Jaramillo 2000).

Subtle but clear distinctions in song help separate Bicknell's and Gray-cheeked thrush. The primary difference is a constant or slightly rising inflection at end of *bicknelli* song, whereas *minimus* songs fall to lower frequencies towards the end (Ouellet 1993). This difference is consistent across the breeding range of both species and is detectable in the field.

Males and females are indistinguishable in the field. Birds in first basic (winter) plumage are often separable from adults through their first full summer by retention of buffy-tipped juvenal feathers in the greater and median wing-coverts, occasionally in the scapulars and mantle. No appreciable seasonal changes in plumage occur after completion of the definitive prebasic molt.

DISTRIBUTION

Breeding range. Bicknell's Thrush occupies a restricted and highly fragmented breeding range (Map 1, Fig. 1). Breeding is documented north to southwestern Quebec in the Réserve La Verendrye, southeastern Quebec along the north shore of the St. Lawrence River and Gaspé Peninsula (Ouellet 1993, 1996), the Magdalen Islands, Quebec (probably extirpated; Ouellet 1996, McNair pers. comm.), northwestern and north-central New Brunswick (Erskine 1992, Nixon 1996), and Cape Breton Island, Nova Scotia, including the small, outlying St. Paul and Scaterie islands (Erskine 1992, D. Busby pers. comm.). Southern breeding limits are reached in the Catskill Mountains of New York (Atwood et al. 1996, Peterson 1988), the Green Mountains of southern Vermont (Atwood et al. 1996, Kibbe 1985), the White Mountains of central New Hampshire (Atwood et al. 1996, Richards 1994), the mountains of western and central Maine (Adamus 1988, Atwood et al. 1996), south-coastal New Brunswick (possibly extirpated; Erskine 1992, Christie 1993), and southwest-coastal Nova Scotia (probably extirpated; Erskine 1992, D. Busby pers. comm.). Possible but unconfirmed local and sporadic breeding has been documented in north-coastal Maine (Atwood et al. 1996, Rimmer and McFarland 1996).

Bicknell's Thrush is widespread at high elevations within both the Green and White Mountain National Forests (GMNF and WMNF, respectively). On the GMNF, presence/absence surveys conducted by VINS between 1992-2000 confirmed the species' presence on 38 of 60 (63%) surveyed peaks (i.e., identifiably discrete patches of montane forest; Table 1). These were distributed as far south as the Deerfield Ridge to as far north as Mt. Ellen (Fig. 2). On the WMNF, Bicknell's Thrushes were detected on 67 of 80 (84%) peaks surveyed in 1992-2000 (Table 2), from Sandwich Mtn. in the south to The Horn (northeast of Mt. Cabot) in the north (Fig. 3).

Winter range. The known wintering distribution of Bicknell's Thrush is confined to the Greater Antilles (Map 1). Specimen and field survey data indicate that the majority of wintering birds 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 2220 m (Rimmer et al. 1999). Few records exist from Haiti, where it is restricted to higher elevations, mainly in the southwest (Massif de la Hotte) and east (Massif La Visite; Wetmore and Swales 1931, Woods and Ottenwalder 1983, 1986). Bicknell's Thrush is uncommon and local in Jamaica, mainly in the Blue Mts. from 1200–2225 m elevation (R. and A. Sutton, unpubl.data; VINS). The species is a rare winter resident in eastern and southeastern Puerto Rico, known only from the Luquillo Mts. at 450-720 m elevation and Sierra de Cayey at 720 m (Arendt 1992; J. Wunderle unpubl.). Bicknell's Thrush has recently been recorded in eastern Cuba at 1600-1960 m in Sierra Maestra (Rompré et al. 2000, Y Aubry and G. Rompré pers. comm.); two October specimens from western Cuba (Havana) in the 1960s (Garrido and Garcia Montaña 1975) probably represent transients. There are no confirmed winter records elsewhere.

Historical changes. Local extirpations have been documented during the 20th century, but no clear evidence exists of a rangewide decline. However, few quantitative data are available to assess population changes. Historic breeding populations have disappeared on Mt. Greylock,

Massachusetts (10 pairs in 1950s, 0 in 1973; Veit and Petersen 1993); in the Magdalen Islands, Quebec (Ouellet 1996, D. McNair pers. comm.); on Seal and Mud islands, Nova Scotia (Wallace 1939, Erskine 1992, D. Busby pers. comm.); at Cape Forchu, Nova Scotia (Marshall 2001); in Fundy National Park, New Brunswick (Christie 1993); and on Grand Manan Island, 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.). In the U.S., Bicknell's Thrush presence was confirmed on 63/73 historic (pre-1992) breeding sites surveyed in 1992-1995 (Atwood et al. 1996), suggesting no largescale changes in recent distribution. The recently discovered occupancy of second-growth habitats in industrial forestry landscapes in Quebec, New Brunswick and Nova Scotia (Ouellet 1993, 1996; Holmes and Nixon 1997; D. Busby pers. comm.) has been suggested to indicate either a shift in habitat use or population increases (Ouellet 1993, 1996), but more likely reflects the species' specialization on disturbed habitats.

Changes on the wintering grounds are not well documented but seem likely due to extensive habitat loss and degradation throughout the Greater Antilles, including montane forests currently preferred by Bicknell's Thrush. Less than 1.5% of forest cover remains in Haiti and c. 10% in the Dominican Republic (Stattersfield et al. 1998). Jamaica has lost 75% of its original forest and Cuba 80-85% (Stattersfield et al. 1998). Of 14 identifiable historic (pre-1991) sites of occurrence in the Dominican Republic, Bicknell's Thrush was located at 7 of 11 surveyed in 1995-1997; several reported historic sites had been severely degraded to the point of being unrecognizable or unsuitable for the species' continued occupancy (Rimmer et al. 1999).

HABITAT

Breeding range: In the United States, Bicknell's Thrush is a habitat specialist restricted to montane forests dominated by balsam fir (*Abies balsamea*), with lesser amounts of spruce (*Picea rubra* and *P. mariana*), white birch (*Betula papyrifera* var. *cordifolia*), mountain ash (*Sorbus* sp.), and other hardwood species. At the southern extent of its range in the Catskill Mountains, Bicknell's Thrush generally breeds above 1100 m elevation; minimum elevations at which the species occurs decrease by 84 m per degree of latitude northward, with birds recorded as low as 750 m on several Maine peaks (Lambert et al. 2001). The lowest nest in Vermont was documented at 1006 m (Rimmer et al. 2001). The species is often associated with recently-disturbed areas undergoing vigorous succession, characterized by standing dead conifers and dense regrowth of balsam fir (Wallace 1939, Rimmer et al. 2001). Highest densities are typically found in chronically-disturbed (high winds, heavy winter ice accumulation) stands of dense, stunted fir on exposed ridgelines or along edges of human-created openings (e.g. ski trails), or in regenerating "fir waves" (cf. Sprugel 1976, Marchand 1984; Rimmer et al. 2001). In the White Mountains of New Hampshire, Sabo (1980) found Bicknell's Thrush at a mean elevation of 1290 m in exposed mid to upper slopes dominated by conifers (75% of foliage volume) with a mean canopy height of 4.8 m.

In Canada, Bicknell's Thrush occupies montane fir forests in southern Quebec and New Brunswick up to 1178 m elevation (Ouellet 1993, Rompré et al. 1997, Connolly 2000, Nixon

et al. 2001; D. Busby pers. comm.), coastal maritime spruce-fir forests in New Brunswick and Nova Scotia (Wallace 1939, Erskine 1992, D. Busby pers. comm.), and regenerating stands of mixed forest following forest fires or clear cutting in Quebec and New Brunswick, generally above 450 m (Ouellet 1993, Nixon 1996, Nixon et al. 2001).

In Quebec montane forests, occupied sites had significantly higher components of balsam fir than unoccupied sites (19,920 stems/ha vs. 7240 stems/ha; Connolly 2000); fir comprised 71.1, 75.1% and 88.5% of all stems recorded at 3 discrete geographic study areas (Rompré et al. 1997). Spruce and hardwoods species were significantly less abundant on occupied than unoccupied sites (Connolly 2000). Mean total stem density varied from 43.7-106.3/m² on occupied sites, and trees < 2.5 cm diameter at 20 cm height above ground were the dominant size class (Rompré et al. 1997). Occupied sites had a lower percentage of herbaceous ground cover, higher percentage of moss ground cover, more dead fallen trees, more snags and stumps, and higher overall tree density (stems > 2.5 cm diameter) than unoccupied sites (Connolly 2000). Mean canopy heights of occupied habitats ranged from 5.4 m in Parc de la Gaspésie, to 7.5 m in ZEC des Martres, to 14.1 m on Mont-Mégantic (Rompré et al. 1997).

Winter range: The current preferred winter habitat of Bicknell's Thrush is mesic to wet broadleaf montane forests in the Dominican Republic (Rimmer et al. 1999), Haiti (Wetmore and Swales 1931, Woods and Ottenwalder 1983, 1986), Cuba (Rompré et al. 1999, Aubry and Rompré pers. comm.), Jamaica (R. and A. Sutton, pers. comm., VINS), and Puerto Rico (J. Wunderle unpubl.). In the Dominican Republic, the species is found at all elevations from sea level to 2200 m, although 62% of occupied sites were in forests > 1000 m elevation (Rimmer et al. 1999). The majority (75%) of occupied sites ($n = 24$) were in broadleaf-dominated forests ("cloud/montane broadleaf forest" and "submontane broadleaf rainforest"; Tolentino and Peña 1998) at all elevations, 19% were in mixed broadleaf-pine forests, and 6% occurred in pine-dominated forests. Primary, wet and/or mesic forests constituted 78% of all occupied sites; only 6% of occupied sites were in predominantly dry forests (Rimmer et al. 1999). The use of regenerating secondary forests (22% of occupied sites) in the Dominican Republic may indicate winter habitat flexibility or a recent shift from preferred primary broadleaf forest habitat, much of which has been lost or degraded.

In the Dominican Republic, some evidence exists for sexual habitat segregation, or segregation of sexes by geographic area (Rimmer and Goetz 2001, Rimmer et al. 2001). In Sierra de Bahoruco on the Haitian border, in predominantly undisturbed broadleaf montane forests, 29 of 33 (88%) birds mist-netted in November of 1998 and January of 2000 were males. At a smaller, more recently disturbed montane forest site in the Cordillera Septentrional in the north-central Dominican Republic, 9 of 11 birds captured in January of 2000 were female. At a similar site 23 km to the east, 4 females and 3 males were captured in January of 2000. These results are preliminary and may be an artifact of small sample sizes or habitat disturbance from human activities and/or Hurricane George in September of 1998; they warrant more intensive investigation.

LIFE HISTORY

Migration. Bicknell's Thrush is a nocturnal, long-distance migrant. Its migratory routes and timing are poorly documented due to difficulties of distinguishing *C. bicknelli* and *C. minimus* in the field. Examination of hand-held birds provides the only reliable means of separating migrants of the two species. Analysis of specimen and banding data, using wing chord as an identification criterion (< 94 mm = *bicknelli*, > 98 mm = *minimus*), suggests an elliptical southern portion of the migratory route between North American breeding grounds and the Greater Antillean winter range. Most southbound migrants may depart the East Coast from mid-Atlantic states or the Carolinas on an overwater flight to the Greater Antilles; fall records are scarce south of Virginia. The northward passage appears to be more concentrated through the Southeast, as spring specimens from Florida, Georgia, both Carolinas, and Virginia outnumber fall records nearly 2:1. The species' entire migration in both directions is concentrated east of the Appalachian Mountains (see Rimmer et al. 2001 for details).

Major food items. Invertebrates are primarily taken during the breeding season, mainly ants, beetles and lepidopteran larvae. Stomach contents of adults collected on Mt. Mansfield, Vermont ($n = 5$) and Slide Mt., New York ($n = 2$) in late June and early July contained an average of 34% beetles (range = 1-95%) and 29% ants (range = 0-55%); one bird contained 90% chrysomelid beetles (Wallace 1939). Animal matter comprised nearly 100% of these samples, but 2 birds showed small amounts of unidentified plant matter (Wallace 1939). Lepidopteran and other larvae constitute the bulk of food delivered to nestlings in Vermont, but beetles and adult Hymenoptera are important nestling prey items (Wallace 1939; A. Strong, unpubl. data).

Predators. Few predators have been documented to take adults. The remains of 2 radio-tagged females were found in or below an active Sharp-shinned Hawk (*Accipiter striatus*) nest in mid-elevation red spruce forest up to 2 km from known home ranges on Mt. Mansfield (Rimmer et al. 2001). Five other dead, radio-tagged adults found on hardwoods forest floor were probably depredated by Sharp-shinned Hawks; 2 of these were recovered at plucking posts of this species. A radio-tagged female with dependent fledglings was found cached underneath a rotting log; tooth marks in her skull suggested depredation by a long-tailed weasel (*Mustela frenata*). Occasional mobbing and chasing of Northern Saw-whet Owls (*Aegolius acadicus*) suggests that this species may depredate adults or free-flying young (Rimmer et al. 2001).

Of 7 radio-tagged fledglings known to have died, all were killed by predators. One was found at a Sharp-shinned Hawk plucking post, the others were apparently depredated by mammals. Juveniles are probably more susceptible to mammalian predation than adults, due to their less developed flight skills and conspicuous begging behavior.

Red squirrels (*Tamiasciurus hudsonicus*) are the only confirmed predator of eggs and nestlings (Wallace 1939, Rimmer et al. 2001). Other suspected or likely nest predators include Blue Jay (*Cyanocitta cristata*), Common Raven (*Corvus corax*), chipmunk (*Tamias sciurus*), boreal red-backed vole (*Clethrionomys gapperi*), deer mouse (*Peromyscus maniculatus*), and weasel (*Mustela* sp.; Wallace 1939; Rimmer et al. 2001). Other potential

predators observed in Bicknell's Thrush breeding habitat include red fox (*Vulpes fulva*), coyote (*Canis latrans*) and raccoon (*Procyon lotor*). Possible predators in winter include Sharp-shinned Hawk, Ridgway's Hawk (*Buteo ridgwayi*), mongoose (*Herpestes auropunctatus*), and rats (*Rattus* sp.).

Mating system and sex ratio. The mating system of Bicknell's Thrush is unusual and not easily categorized; it may be most similar to that of Smith's Longspur (*Calcarius pictus*), which has been termed "female-defense polygynandry" (Briskie 1992), in that both males and females mate with multiple partners, multiple paternity is common, and > 1 male often feeds nestlings (Goetz 2001). In Vermont, > 75% of broods are sired by multiple males; some males have offspring in two nests in the same breeding season. Of 13 broods in 1998 and 1999, 10 had 2 or more sires, 3 a single father (Goetz 2001).

The overall 4-yr mean male:female ratio on 3 Vermont study plots was 1.8:1.0 (annual range = 1.4-2.8:1.0; Rimmer et al. 2001). The cause of a male-biased sex ratio is not known, but may relate to the ratio at hatching, differential natal dispersal patterns, or events on the wintering grounds (e.g., differential male and survival due to winter habitat segregation).

Breeding phenology. The earliest known arrival date of a breeding male in Vermont is 16 May, of a female 23 May. Males arrive significantly earlier than females (mean difference = 1.7 days, 95% CI = 3.2 – 0.3). Mating activities probably begin shortly after female arrival, as evidenced by frequent singing and calling throughout the day in late May and early June (Rimmer et al. 1996). Mating associations are dynamic and probably tied to the stage of individual females' fertile periods, and are likely influenced by the availability of other mating opportunities and chick feeding by males.

The earliest confirmed nest construction date in Vermont is 1 June, with other extrapolated nest initiation dates of 2-4 June (Wallace 1939, Rimmer et al. 2001). In Vermont, 71% ($n = 89$) of clutches are initiated in the first 3 weeks of June. Later clutches probably represent renesting attempts. Clutch initiation dates in Vermont range from 7 June-14 July ($n = 89$), and in New Hampshire 21 June-14 July ($n = 5$; Wallace 1939, Foss 1994). Known hatching dates range from 23 June-29 July (70% by 6 July) in Vermont ($n = 68$; Wallace 1939, Rimmer et al. 2001). Known Vermont fledging dates range from 3 July-3 August (70% by 14 July, $n = 53$; Wallace 1939; Rimmer et al. 2001). Young stay in the nest 9-13 days (average 11.4 ± 1.3 , $n = 17$; Wallace 1939, Rimmer et al. 2001).

Second broods are rare, with only one confirmed instance in Vermont. A female that fledged 2 chicks on 2 July initiated a second clutch 5 days later, building her nest while feeding fledglings and continuing to feed her first brood during egg laying (Rimmer et al. 2001). Renesting attempts after early-season failures are common. The mean interval between loss of a first nest and initiation of a second clutch in Vermont is 6.8 days (range 5-12, $n = 5$). One female renested successfully on her third attempt, requiring only 2 days from loss of her second clutch to initiation of her third (Rimmer et al. 2001).

Nests and eggs. Nests are bulky, cup-shaped structures built primarily of twigs and moss. The exterior shell of most nests in montane forests of Vermont is constructed of twigs of balsam fir, occasionally of red spruce and paper birch, profusely interwoven with strands of moss (primarily *Pleurozium schreberi*, often lesser amounts *Sphagnum* spp.; Wallace 1939, Rimmer et al. 2001). Proportions of twigs and moss vary; some nests are reported to be almost entirely constructed of moss (Wallace 1939). Other materials found in nest walls include grasses, sedges, stalks of herbaceous flowering plants or ferns, dry leaves, bark strips, hair, and lichen (Wallace 1939; Rimmer et al. 2001). Interior walls consist of decayed vegetation, such as leaf mold. Inner linings of Vermont nests are invariably composed of threadlike, black rhizomorphs of Horsehair Fungus (*Marasimius androsaceus*; McFarland and Rimmer 1996); some nests may also be lined with fine stems of grasses or sedges (Wallace 1939, Rimmer et al. 2001). One nest on a ski area contained pieces of nylon rope woven in the cup (Rimmer et al. 2001).

The mean outside diameter of 79 Vermont nests in 1992-2000 was 11.3cm \pm 1.8 SD (range 5-16); inside diameter averaged 7.1cm \pm 1.3 SD (range 5.3-12); outside mean nest height was 8.1cm \pm 1.9 SD (range 1.6-14); inside depth averaged 4.4cm \pm 0.9 SD (range 2-6.5; Rimmer et al. 2001).

Eggs are subelliptical, bluish green with variable amounts of light brown spotting, and smooth to semi glossy in surface texture. Twenty-nine eggs from 8 clutches on Mt. Mansfield in 1935 had a mean length of 21.9 mm (range 21.0-23.0) and mean breadth of 16.6 mm (range 16.0-17.5; Wallace 1939). Ten eggs from Vermont in the late 1990s had a mean length of 22.38 mm \pm 0.78 SD (range 20.48-23.6), and 8 eggs had mean breadth of 16.29 mm \pm 1.64 SD (range 12.36-17.5; Rimmer et al. 2001).

First clutches invariably contain 3-4 eggs. Of 13 Mt. Mansfield nests examined in 1935, 7 contained 3 eggs, 6 contained 4 (Wallace 1939). Of 59 known or probable first clutch nests examined on Mt. Mansfield and Stratton Mtn. in 1992-2000, mean clutch size was 3.6 \pm 0.49 SD (range 3-4; Rimmer et al. 2001). Nests initiated earlier in the season tend to have 4 eggs, later nests 3 (Wallace 1939; Rimmer et al. 2001). The mean clutch size of 13 known second attempts was 3.1 \pm 0.28 SD (range 2-4; Rimmer et al. 2001). One known third attempt contained 3 eggs.

Nest Selection process. Little information is available, but nest sites are probably selected solely by females. Females build nests 17-1,344 m apart in successive years (mean = 182.9 m \pm 267.8, $n = 26$; Rimmer et al. 2001). No statistical difference exists between distances for females of failed vs. successful previous year's nest, although large movements tend to follow failures. One ASY (after second-year) female moved 1,344 m and another 540 m after failing the prior year; these distances were more than twice those between any other successive year's nests. One female in 2000 nested 1,715 m away from a nest she built in 1998 as a yearling. Renesting attempts averaged 52.7 m \pm 28.5 SD from first nests ($n = 7$, range = 19-87; Rimmer et al. 2001).

Nest microhabitat. Nests are usually located in dense stands of young to mid-successional fir or “krummholz”, uncommonly in more mature, open forests (Wallace 1939, Rimmer et al. 2001), and are often found in dense regrowth along natural or artificially created edges. On 2 ski areas (Stowe and Stratton) in the Green Mountains, nests averaged $10.8 \text{ m} \pm 8.97 \text{ SD}$ from ski trail edges (range = 0-33, $n = 26$; Rimmer et al. 2001).

On nest-centered 5-m radius plots ($n = 103$) in Vermont, the mean density of large woody stems (<8.0 cm diameter at 10 cm above ground) was $163.4 \pm 107.34 \text{ SD}$ (Rimmer et al. 2001). Balsam fir accounted for 67% of all live woody stems < 8.0 cm diameter within 5 m of nests, followed by white birch (11.7%), dead stems (9%), mountain ash (6.1%), mountain-holly (*Nemopanthus mucronata*; 1.9%), and red spruce (1.1%); 11 other species each accounted for <1%. Leaf litter depth ranged from 1.5-21.5cm (mean = 5.1 ± 2.9 , $n = 74$). On nest-centered 11.3-m radius plots ($n = 103$), the mean density of live trees 8-23 cm DBH was $33.4 \pm 18.7 \text{ SD}$ (range = 5-89); dead standing trees 8-23 cm DBH averaged $11.9 \pm 8.2 \text{ SD}$ (range = 0-34). The mean density of live trees >23 cm DBH was $3.25 \pm 4.95 \text{ SD}$ (range = 0-30), while standing dead trees >23 cm DBH averaged 2.3 ± 2.9 (range = 0-22). Canopies were dominated by balsam fir at 81 of 103 nests (79%), balsam fir and white birch were co-dominant at 9 nests, a mix of balsam fir and mountain ash predominated at 5 nests, white birch was dominant at 4 nests, a mix of several species predominated at 2 nests, balsam fir and red spruce were co-dominant at 1 nest, and red spruce was dominant at 1 nest. Mean canopy heights within 11.3 m of nests ranged from 1.2-17.9 m (mean = $5.4 \pm 2.9 \text{ SD}$, $n = 103$). Slopes ranged from 0-46 degrees (mean = 18.7 ± 10.4 , $n = 101$).

Nest site characteristics. Vermont nests are typically built at the base of 1-4 horizontal branches against the trunk of a small tree (70%, $n = 105$), occasionally up to 3 m from the trunk on horizontal branches of larger trees (Rimmer et al. 2001). Support branches averaged 1 cm diameter (range 0.1 – 5.25, $n = 93$). Some nests were supported between two closely spaced trees (23%, $n = 105$). One nest was built inside the cavity of a balsam fir snag; another was perched on a shelf created by a broken snag. Most nests (103 of 118; 87%) are built in balsam fir, but also in red spruce ($n = 10$), paper birch ($n = 3$), and dead standing fir ($n = 2$; Wallace 1939, Rimmer et al. 2001). Average nest tree height was $3.2 \text{ m} \pm 1.55 \text{ SD}$ (range 0 – 11m, $n = 102$) and mean DBH was $5.7 \text{ cm} \pm 5.24 \text{ SD}$ (range 1-31.5 cm, $n = 102$). Nest orientation in relation to the trunk averaged 161 degrees ($n = 27$ in southeast quadrant, 22 in southwest quadrant, 15 in northwest quadrant, 13 in northeast quadrant). Of 118 Vermont nests, the mean height above ground was $2.05 \text{ m} \pm 1.18 \text{ SD}$ (range = 0.46-10 m; Wallace 1939, Rimmer et al. 2001). Mean vegetation concealment in a 25cm diameter circle around 98 nests, estimated from 1 m away, was $74.7\% \pm 24 \text{ SD}$ overhead, $62.7\% \pm 27.4 \text{ SD}$ to the north, $64.9\% \pm 29.3 \text{ SD}$ to the south, $63.8\% \pm 27.4 \text{ SD}$ to the east, and $67\% \pm 27.1 \text{ SD}$ to the west.

Incubation and nestling care. In Vermont, the incubation period is 9-14 days (mean 12 ± 1.6 , $n = 8$; Wallace 1939, Rimmer et al. 2001). Chicks generally hatch within 24 hours of each other. Only females brood, but both sexes feed chicks. At 25 Vermont nests observed by videography, one female fed at each nest, with 2 provisioning males most common (60%), followed by 1 male (20%), 3 males (16%) and 4 males (4%). Four males were documented to feed at more than one nest within a single breeding season, three feeding two broods

simultaneously (nests 186-443 m apart). One male simultaneously provisioned at two nests 443 m apart; he shared feeding of nestlings at the first nest and was the sole male feeder at the second nest. His first nest fledged 3 days after the second nest hatched; the male then left care of fledglings to the other male and fed his second brood at nearly twice the rate as he had fed young at the first nest. Individual female, total male and total adult provisioning rates did not differ between nests with single and multiple male feeders. Some males did not feed at nests in which they sired young, and some males fed at nests in which they sired no young. Male feeding rates increased with nestling age until day 7-8 and then decreased until fledging. (Rimmer et al. 2001)

Fledgling stage. Nestlings fledge 9-13 days after hatching (mean 11.4 ± 1.3 , $n = 17$ known to the exact day; Wallace 1939, Rimmer et al. 2001). Fledglings may remain with adults up to 14 days after leaving the nest. Adults often split broods, although one known case exists of 2 males splitting a brood, apparently emancipating the female. In another case, a female and one of 2 male feeders split the brood, while the second male continued to feed nestlings in another nest. Movements of family groups are not well documented, but adults with dependent fledglings have been found up to 280 m away from known nest sites. (Rimmer et al. 2001)

Movements and habitat use during the post-fledging period of independence are poorly known. Of 11 Vermont fledglings radio-tagged in 2000, 7 were known to have been depredated (mean survival 8.1 ± 6.6 days after fledging, range 1-19), 2 disappeared after 8 and 19 days, respectively, and 2 survived until their transmitter batteries expired (40 and 31 days, respectively). Of these latter 2 birds, one remained within 275 m of its natal nest site in montane fir forest, while the other moved nearly 1 km downslope after about 10 days to hardwood-dominated forest at elevations of 700-900 m and remained there. One free-flying juvenile banded on 25 July stayed within a 100-m radius of its banding location in stunted fir forest at 1150-1175 m elevation until 22 August, then disappeared. (Rimmer et al. 2001)

POPULATION BIOLOGY AND VIABILITY

Age at first breeding; intervals between breeding. Bicknell's Thrushes breed at approximately 1 year and annually thereafter. Of known-age female breeders at 85 Vermont nests in 1994-1999, ASY females outnumbered SY (second-year) females 73 to 12 (85.9% to 14.1%). Of 25 Vermont males with known paternity at 1998 and 1999 nests, only 2 (8%) were SY birds, while the SY age class comprised about 25% of entire male study population (Goetz 2001). Highly irregular settlement patterns further suggest that some SY males may fail to achieve fertilizations. (Rimmer et al. 2001)

Annual and lifetime reproductive success. In Vermont, annual reproductive success among males is skewed but generally low. Paternity data from 12 broods monitored in 1998 and 1999 suggest considerable reproductive skew. Of 19 males with known paternity, 12 (63%) males sired 1 young, 2 (11%) sired 2 young, 4 (21%) sired 3 young, and 1 (5%) sired 4 young; these are minimum estimates (Goetz 2001).

The annual Mayfield daily survival rate of nests (probability of nest surviving 1 day without failure) on Stratton Mtn. was 0.98 ± 0.014 SE ($n = 39$ nests), and on Mt. Mansfield 0.96 ± 0.007 SE ($n = 56$ nests). Daily survival rates of Vermont nests are strikingly biennial in response to balsam fir cone production and red squirrel population cycles. From 1994-2000, fall cone crops were very high in even-numbered years, resulting in high red squirrel populations during the following springs and summers, with consequent low productivity for Bicknell's Thrush due to nest depredation. In odd-numbered years, fall cone production was invariably lower, spring and summer squirrel populations reduced, and thrush nesting success markedly higher. (Rimmer et al. 2001)

The average number of young fledged/nest in Vermont was 2.1 ± 1.37 SD (range = 0-4, $n = 30$) on Stratton Mtn. and 1.5 ± 1.59 SD (range = 0-4, $n = 46$) on Mt. Mansfield (Rimmer et al. 2001). The percentage of females that raised one brood to independence each year on Stratton Mtn. was 85.7 (1997), 88.8 (1998), 0 (1999), and 90.9 (2000). On Mt. Mansfield, 62.5 of females raised at least one fledgling to independence in both 1999 and 2000 (Rimmer et al. 2001).

Life span and survivorship. The longevity record for a banded male Bicknell's Thrush is 8 years and for a female 7 years. The annual survival rate of ASY birds captured on Vermont breeding grounds, based on a Cormack-Jolly-Seber model (Cooch and White 1998, White and Burnham 1999), was not dependent on time or sex on four study plots. To account for uncertainty in model selection, the range of mean parameter estimates was averaged over all 16 models in the candidate set for each study plot, weighted by Akaike model weights, and the most parsimonious model was used (Burnham and Anderson 1998, Bertram et al. 2000). Survivorship on the Mt. Mansfield ridgeline in 1992-1999 was $54.7\% \pm 6.5\%$ SE with mean parameter estimates for all models ranging from 54% - 55.8%; on Mt. Mansfield's east slope in 1995-1999, $74.8\% \pm 8.6\%$ SE, mean estimates 71.9% - 79.1%; on VINS' Stratton Mtn. ski area plot in 1997-1999, $73.9\% \pm 10.1\%$ SE, mean estimates 75.6% - 88.3%; and on VINS' Stratton Mtn. natural plot in 1997-1999, $94.6\% \pm 28.4\%$ SE, mean estimates 86.1% - 94%. There was no difference in survivorship between Stratton Mtn. ski area and natural area plots. Survival rate of juveniles are poorly known due to apparent natal dispersal; only 3 of 115 (2.6 %) nestlings and dependent fledglings and 9 of 62 (14.5 %) independent juveniles banded in Vermont 1992-1998 were documented to return to their natal mountain. Two nestlings that returned were females from the same nest. On Mt. Mansfield in 2000, only 2 of 11 (18.2%) radio tagged fledglings were known to have survived beyond 30 days. The annual survival rate of wintering individuals captured at a montane broadleaf forest site in Sierra de Bahoruco, Dominican Republic, based on Cormack-Jolly-Seber model estimates, was not time dependent in 1994-1999: $72.9\% \pm 14.3\%$ SE, with mean parameter estimates for all models ranging 68.4% - 79.7%. (Rimmer et al. 2001)

POPULATION STATUS

Numbers. Bicknell's Thrush is one of the most rare, range-restricted breeding species in eastern North America. Densities are difficult to ascertain because of the species' unusual mating system and its occupancy of rugged terrain and dense habitats. Previously published density estimates based on spot-mapping of presumed territorial males (Rimmer et al. 1996)

are now known to be too high, due to large and significantly overlapping male home ranges. Mark-recapture analyses of mist netted adults are thought to provide a more accurate measure of density. Recent studies on Mt. Mansfield and Stratton Mtn. indicate maximum densities of 0.4-0.79 males/ha (VINS unpubl. data). Several factors, however, confound extrapolations of estimated rangewide numbers from these or any other field data. These include (1) differing calculations of suitable montane coniferous forest habitat in the U.S. (see Table 3); (2) lack of adequate data to assess the extent to which potential habitat is saturated, i.e., how breeding densities vary among and within habitat patches across the range; (3) uncertainty about overall breeding sex ratios; and (4) lack of sufficient knowledge about the species' breeding status in Canada, particularly in regenerating industrial forests. Further, high densities may not necessarily reflect high-quality habitat (e.g., Van Horne 1983, Vickery et al. 1992) and may lead to overestimates of actual breeding abundance for Bicknell's Thrush.

Calculations of the amount of conifer-dominated montane forest habitat available for occupancy by Bicknell's Thrush must account for both elevation and latitude. The combined effects of these two parameters on Bicknell's Thrush distribution in the U.S. is strongly linear, in that the lower limit of occupied habitat drops 84 m in elevation for every one-degree increase in latitude (Lambert et al. 2001). Factoring this relationship into an elevation-based model of Bicknell's Thrush habitat, an estimate of potentially suitable habitat within the U.S. is 110,934 ha (Table 3; VINS unpubl. data). We believe this to be a more realistic estimate than those derived strictly from uncorrected elevation data above 915 m (Atwood et al. 1996; Table 3), spruce-fir cover data above 793 m (Miller-Weeks and Smoronk 1993; Table 3), or multiple logistic regression of habitat data derived from Enhanced Thematic Mapper satellite imagery and Bicknell's Thrush density derived from fixed radius point counts in New Hampshire (Hale 2001; Table 4). According to our model, New Hampshire contains the largest area of conifer-dominated montane forest habitat in the U.S. (49,733 ha; 45% of total), followed in descending order by Maine (26,048 ha; 23%); the Adirondack Mountains of New York (26,037 ha; 23%), the Green and Taconic mountains of Vermont (8,610 ha; 8%), and the Catskill Mountains of New York (506 ha; < 1%) (Table 3; VINS unpubl. data).

We do not believe that it is possible to accurately estimate the rangewide breeding population of Bicknell's Thrush at this time. Even crude estimates are tenuous, as they rely on numerous unverifiable assumptions. Foremost among these is that conifer-dominated montane forest habitat is saturated, a hypothesis that our field investigations have repeatedly falsified. Further, assumptions of an approximately 2:1 male:female breeding sex ratio and male densities of 0.4-0.79 birds/ha can not currently be validated across the species' range. Applying these two assumptions to the total calculated habitat of 110,934 ha from our model yields a very coarse estimate of 44,374 – 87,638 males and 22,187 – 43,819 females, or 66,561 – 131,457 total individuals in the U.S. We strongly suspect that these upper limits significantly overestimate the U.S. breeding population of Bicknell's Thrush, as they are derived from our maximum density estimate over several years at a single site (Mt. Mansfield) and assume saturation of habitat. We believe that a more judicious and conservative estimate is simply that there are likely fewer than 100,000 individuals inhabiting the species' U.S. breeding grounds. More precise estimation than this will require detailed field determination of actual densities and their habitat correlates across a geographically

representative cross-section of breeding sites, incorporating different natural successional and disturbance regimes.

Estimates of an effective population size of 800,000 females, derived from mitochondrial DNA control region genealogies and assuming a relatively slow mutation rate (Ellison 2001), are much higher than ours. At a higher mutation rate, effective female population size is 80,000 (Ellison 2001), nearly twice the upper limit of our crude estimates. Estimates based on effective population size may be high relative to actual population sizes because (1) the high rate of gene flow documented in Bicknell's Thrush (Ellison 2001) may artificially maintain a higher than theoretically expected amount of genetic variation in the species, or (2) the species has recently experienced a sharp population decline and has not yet lost substantial variation to genetic drift (W. Ellison, pers. comm.). Whatever the explanation, we believe that actual U.S. population is comprised of fewer than 100,000 individuals.

A probability-based habitat model for the WMNF constructed from satellite imagery data estimated a total of 67,000 "singing males" on 90,114 ha of forest habitat (Hale 2001), an overall density (0.74 males/ha) close to our maximum estimate. However, Hale's (2001) model included lower elevation forests with a high hardwoods component. Bicknell's Thrush may occasionally occur in these habitats, but our field data indicate that such forests are rarely, if ever, used for nesting. This model effectively estimated probability deciles that described the chance of detecting at least one vocalizing Bicknell's Thrush in five 50-m radius point counts from 1993-1997. While Hale (2001) took this model one step further and attempted to estimate densities in each decile, he strongly advocated the proper calibration of density estimates with intensive demographic work in representative areas of each decile (Hale, pers. comm.). His method, leading to development of a Habitat Suitability Index map, and combined with careful demographic studies, may be the best means to determine the overall population size of Bicknell's Thrush.

More than 90% of the global breeding population is believed to occur within the U.S, with only an estimated 2,000-2,500 pairs breeding in Canada (Nixon 1999). This estimate, however, seems low and likely has a high margin of error, as no rigorous estimations of densities have been made in representative habitat types occupied by Bicknell's Thrush in Canada.

Trends. Few population trend data are available from any part of the breeding range, due to lack of adequate baseline data. This species is virtually unsampled by the North American Breeding Bird Survey of the USGS Patuxent Wildlife Research Center. The only data currently available are point count data collected annually at 37 montane forest sites containing 311 individual points in the White Mountains from 1993-2000 (Audubon Society of New Hampshire and New Hampshire Fish and Game Department unpubl. data) and those collected at 11 sites consisting of 101 points in the Green Mountains from 1991-2000 (University of Vermont Spatial Analysis Lab and VINS unpubl. data).

These two datasets were analyzed using the program ESTEQN (Collins 1997). This program derives an estimate of an overall population trend as a weighted average of the trend seen on individual sites, using estimating equations (Link and Sauer 1994). The sum of the number of

Bicknell's Thrushes detected on all points for each point count series (route) was used as an index of annual relative abundance. Because each route in Vermont was covered twice during each breeding season, the higher total count in each year was used. Of the 37 routes censused in New Hampshire, only the 27 that had been completed in every year were included in the analysis. Within each route only those points that were completed in each year were analyzed.

In the Green Mountains, Bicknell's Thrush showed a non-significant annual increase of 3.9% ($P=0.57$, $CI=-8.9-18.5\%$; Fig. 4). Point count data from WMNF, however, indicated an annual decline of -8.3% ($P=0.063$, $CI=-16.1-0.2\%$; Fig. 4).

The program MONITOR version 6.2 (Gibbs 1995) was used to examine the statistical power to detect population trends for both New Hampshire and Vermont. A common problem in trend detection is that sources of annual variance in counts obscure actual trends. The probability that a monitoring program will detect a real trend in sample counts, despite variance in count data, represents its power. Although statistical power is central to every monitoring effort, it is rarely assessed. Consequences of ignoring it include collection of count data insufficient to make reliable inferences about population trends, and collection of data in excess of those needed. Generally, monitoring programs should try to attain power estimates that exceed 0.80 (Cohen 1988). In other words, a monitoring program whose power estimates exceeded 0.80 would detect trends, should they occur, >80% of the time. With only 11 routes, the Vermont data have low power to detect even large declining trends over a 10-year period (Fig. 5). After 15 years, we would be able to detect a 6% annual decline and after 20 years, about a 4% annual decline. Because of the higher number of routes, the New Hampshire survey has greater statistical power. Ten years of New Hampshire data would be sufficient to detect a 5% annual decline; annual decreases of 3% and 2.5% would be detectable in 15 years and 20 years, respectively (Fig. 6). The New Hampshire routes have been discontinued as of 2001.

Mountain Birdwatch (MBW), a long-term, volunteer-based point count program, was initiated by VINS in 2000 to assess the status of Bicknell's Thrush and four other montane forest breeding species in the Northeast. Combining data from 44 MBW surveys (point count series) completed during the project's 2000 pilot season in Vermont with data from 9 longer-term Vermont routes indicates that a minimum of five years would be necessary to obtain $\geq 90\%$ power to detect an annual decline of 5% in Bicknell's Thrush, using 100 survey routes (Lambert et al. 2001). A minimum of nine years would be required to detect a 2% annual decline. MBW was expanded in 2001 to include 125 high-elevation survey routes throughout New York, Vermont, New Hampshire, and Maine. Data are currently being analyzed.

Anecdotal evidence suggests that breeding populations have declined on several small Vermont peaks in the past decade (VINS). Most of these are geographically isolated montane forest patches that probably never supported more than a few birds. Capture rates of migrant "Gray-cheeked" Thrushes ($n=3252$, including known *bicknelli* and *minimus*) in coastal Virginia declined significantly from 1968 to 1995 (Wilson and Watts 1997). No meaningful quantitative trend data are available from the wintering grounds.

Formal Conservation Status: Bicknell's Thrush is protected by federal law under the Migratory Bird Treaty Act, but it was a former C2 candidate for listing by the U.S. Fish and Wildlife Service prior to the abolishment of that category in 1996. The United States National Conservation Status Rank and the Canadian National Conservation Status Rank are both listed as N4B, indicating "apparently secure" breeding populations (NatureServe 2001). The Natural Heritage Global Rank, and the Natural Heritage Rounded Global Conservation Status are both given as G4, also indicating that the species is "apparently secure... apparently not vulnerable in most of its range, but possibly a cause for concern" (NatureServe 2001). Stronger concern is expressed by the International Union for the Conservation of Nature, which classifies Bicknell's Thrush as "vulnerable" on its worldwide list of threatened birds (BirdLife International 2000). The Committee on the Status of Endangered Wildlife in Canada lists the status of Bicknell's Thrush as "Special Concern" (Nixon 1999, NatureServe 2001).

Although not listed as Endangered or Threatened in any Northeast states, Bicknell's Thrush is a Species of Special Concern in New York, Vermont, and Maine. The species is included on a watch list of "special concern" species in New Hampshire (J. Kanter, pers. comm.). Natural Heritage state ranks follow (see Appendix for explanations): NY (S2S3B), VT (S3B, SZN), NH (S2S3B), ME (S3B), MA (SN, SHB). Natural Heritage ranks for Canadian provinces include New Brunswick (S2B), Nova Scotia (S1S2B), Ontario (SZN), Prince Edward Island (S1?B), and Quebec (S4).

POTENTIAL THREATS

Pesticides and other contaminants/toxics. Blood mercury (Hg) levels were examined in 37 adults from 6 breeding sites across the northeastern U.S. in 1999 and 2000. Mean blood Hg was $0.113 \text{ ppm} \pm 0.134 \text{ SD}$ (range 0.013-0.795), with no consistent age, sex or geographic differences (Rimmer and McFarland 2001). Mean feather Hg levels, reflecting the chronic body burden, were $0.765 \text{ ppm} \pm 0.362 \text{ SD}$ ($n = 29$, range 0.171-1.61); these differed significantly across a west-east geographic gradient, progressively declining from Whiteface Mtn. in the Adirondacks ($1.209 \pm 0.385 \text{ SD}$) to Mt. Mansfield ($0.756 \pm 0.254 \text{ SD}$) to the Bigelow Range in Maine ($0.286 \pm 0.09 \text{ SD}$; Rimmer and McFarland 2001). Among known-aged birds, significantly higher feather Hg levels were found in older (2+ year-old) birds (mean $0.942 \text{ ppm} \pm 0.315 \text{ SD}$; $n = 19$) than in yearling birds (mean $0.445 \text{ ppm} \pm 0.12 \text{ SD}$, $n = 4$), suggesting that annual inputs of Hg exceed outputs. No significant differences in blood or feather Hg levels were found between adult males and females. The mean ratio of toxic methylmercury to total mercury in Bicknell's Thrush blood was 0.949 ± 0.155 ($n = 18$; range 0.541-1.182), suggesting that methylation rates in montane forests may be high. Effects of mercury toxicity are not known in this or other terrestrial insectivorous bird species, because thresholds have not been established.

Degradation of habitat. Decline of high elevation forests in the northeastern U.S. during the 1960s and 1970s has been well documented (e.g., Johnson and Siccama 1984, Eager and Adams 1992). Red spruce dieback was especially pronounced, but extensive and widespread mortality of balsam fir was also documented (Miller-Weeks and Smoronk 1993), although most of this resulted from naturally-occurring fir waves. Atmospheric deposition of acidic

ions from industrial sulfur and nitrogen oxides has been strongly, although not conclusively, implicated as a causal factor in red spruce decline (Johnson et al. 1992, NAPAP 1992). Increased winter freezing injury of spruce, possibly mediated through reductions in calcium reserves, may be directly linked to high levels of acidic deposition (DeHayes et al. 1999). Despite declining trends in atmospheric sulfate concentrations resulting from mandates of the 1990 Clean Air Act amendments, acidity of precipitation in northeastern North America does not appear to be decreasing (Scherbatskoy et al. 1999).

Heavy metal toxicity from airborne pollutants has also been implicated as a contributing cause of high elevation forest decline in the northeastern U.S., particularly in the Adirondack and Green Mountains (Gawel et al. 1996). However, several recent studies indicate that lead concentrations in the forest floor are rapidly decreasing (Friedland et al. 1992, Miller and Friedland 1994, Wang and Benoit 1997). Little information exists on other heavy metals in montane forests.

Atmospheric deposition of airborne mercury is 2-5 times higher in montane forests of Mt. Mansfield than in surrounding low elevation areas (Lawson 1999). Methylation rates and possible mechanisms of uptake in the terrestrial food chain of montane forests have not been documented.

Global climate change may exert profound, long-term impacts on balsam fir forests. The average global surface temperature could rise 1.6-6.3°F by 2100, with significant regional variation (EPA 2000). A modeling effort using USDA Forest Service Forest Inventory Data, numerous environmental variables, and equilibrium climate variables provided by five Global Circulation Models (assuming doubling of atmospheric carbon dioxide) predicts an average reduction of 96% in the total area occupied by balsam fir in the eastern U.S. (Iverson et al. 1999, Prasad and Iverson 1999).

Recreational and commercial development in montane forests contribute to increased habitat fragmentation and loss, but their cumulative effects are poorly known. In Vermont, 13 mountains > 915 m in elevation are developed for recreational skiing; many of these offer mountain biking programs during the summer. Ski area development pressures are similar in New Hampshire and Maine, less so in the Catskill and Adirondack mountains of New York. Preliminary data indicate that nesting Bicknell's Thrushes are able to tolerate moderate levels of human traffic and that nesting success is not adversely impacted near ski trails (VINS unpubl. data), but more study is needed to investigate this.

Proliferation of telecommunications towers on mountaintops of the northeastern U.S., as well as development of wind power generation facilities, may further fragment montane breeding habitat and introduce disturbance from construction and servicing activities. However, few quantitative data are available to evaluate the impacts of towers and wind power facilities on Bicknell's Thrush. Systematic daily surveys beneath four transmission towers on Mt. Mansfield during the fall of 1997 failed to document any avian casualties, suggesting that the mountain's existing tower configuration did not routinely cause extensive injury or mortality to migrating birds during autumn (Rimmer et al. 1997). Studies conducted at New England's only high elevation wind turbine facility in Searsburg, Vermont (825-885 m elevation) failed

to document any fatalities or other obvious negative effects on breeding or migrant songbirds, including Bicknell's Thrush (Kerlinger 2000). However, this site is below the elevation at which Bicknell's Thrush typically occurs in the Green Mountains.

Industrial forestry practices in Canada, such as clear-cutting and pre-commercial thinning, may cause adverse, short-term impacts on Bicknell's Thrush breeding habitat, but the effects of these activities are unknown.

MANAGEMENT POTENTIAL

Little specific information exists. Vegetation management of montane forest breeding sites currently developed for recreational skiing can enhance habitat for Bicknell's Thrush, or minimize possible adverse impacts. Maintenance of low fir-spruce thickets in 3-7 m wide bands of gradually increasing height along ski trail edges can provide nesting and foraging sites. Maintaining forested "islands" of maximum size between ski trails, minimizing width of trails and maximizing connectivity of habitat in developed areas may increase suitability. Vegetation management or construction at breeding sites should be conducted outside nesting season. Preliminary guidelines for habitat management on ski areas are summarized in VFWD (1999).

PRIORITIES FOR FUTURE RESEARCH AND MONITORING

Many aspects of the breeding and wintering ecology, demography, and behavior of Bicknell's Thrush remain poorly known. Lack of baseline population data and logistical difficulties hinder attempts to clarify this species' conservation status. A standardized, regionwide monitoring program, currently in its early stages, is needed to determine breeding population trends and distributional changes. Similar efforts are warranted on the wintering grounds, where limiting factors may be most severe. Development of accurate methods to census populations and estimate densities are needed in both areas. Accurate calculations of total population size, based on GIS projections of occupied habitats and spatially-explicit density estimates, are needed throughout the breeding range. Development of a Habitat Suitability Index and its incorporation in a spatially-explicit Population Viability Analysis would be a critical step in developing ecological risk assessments and sound conservation planning for Bicknell's Thrush.

Many landscape-level questions about the species' ecology and population dynamics require focused research. Information is needed on reproductive success, demographics, and site persistence in habitat patches of different size and isolation; on the existence of source/sink population dynamics; on patterns of natal dispersal and breeding recruitment; on levels of population interchange among habitat patches. The apparent male-biased breeding sex ratio requires rangewide investigation; its causes and demographic/ecological correlates must be determined. Accurate estimates of breeding population density in different habitat types across the species' range are needed. The species' breeding status and ecology in regenerating clearcuts in both montane and low elevation forests of Maine and Maritime Canada should be investigated.

Research is needed on potential effects of food availability and its temporal-spatial variability on breeding system structure and reproductive success; relative diets of adults, nestlings and fledglings; post-fledging dispersal and habitat use; post-breeding movements and habitat use of adults; effects of human activities (e.g. recreational development, telecommunications towers) on spacing patterns and reproductive success.

In winter, distribution and habitat use of Bicknell's Thrush in Cuba and Haiti, and to a lesser extent Jamaica, need to be better understood. The protected status of core wintering areas must be carefully assessed, and needs for further protection specifically identified. Occupancy of primary vs. second-growth winter habitats needs study, as does existence of possible sexual habitat segregation. Demographic studies are needed to investigate microhabitat use, overwinter survival and site persistence by age and sex, between-winter site fidelity and survivorship. Spacing patterns and movements of age and sex classes throughout winter need further study, as do possible seasonal shifts in diet and body condition.

Stopover ecology is virtually unknown. Studies of banded, transient individuals are needed to determine stopover lengths, physiological condition, diet, and habitat use. A thorough study (currently underway by VINS; see preliminary summary in Rimmer et al. 2001) of available banding and specimen data would help establish migratory routes and timing, and might identify specific geographic areas of importance to stopover migrants.

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Table 1. Peaks surveyed for Bicknell's Thrush in the Green Mountain National Forest, 1992-2000.

<u>Mountain</u>	<u>Elevation (ft)</u>	<u>Observed 1992-2000</u>
Baker Peak	2835	no
Bald Mountain (Woodford)	2857	no
Battell Mountain	3482	yes
Bloodroot Mountain	3485	yes
Blue Ridge Mountain	3278	no
Boyce Mountain	3323	yes
Bread Loaf Mountain	3835	yes
Burnt Hill	3040	no
Cape Lookoff Mountain	3420	yes
Corporation Mountain	3142	no
Cutts Peak	4022	yes
Deer Leap Mountain	2782	no
Farr Peak	3522	yes
Gillespie Peak	3366	yes
Glastenbury Mountain	3748	yes
Goshen Mountain	3292	no
Haystack Mountain (Mount Snow)	3445	yes
Haystack Mountain (Hazens Notch)	3223	yes
Kirby Peak	3140	no
Lincoln Peak	3975	yes
Monastery Mountain	3224	yes
Mother Myrick Mountain	3361	no
Mount Abraham	4006	yes
Mount Carmel	3365	yes
Mount Ellen	4083	yes
Mount Grant	3623	yes
Mount Horrid	3216	yes
Mount Roosevelt	3540	yes
Mount Snow	3556	yes
Mount Tabor	3043	no
Mount Wilson	3760	yes
Nancy Hanks Peak	3812	yes
Peru Peak	3429	yes
Philadelphia Peak	3203	no
Round Mountain	3342	yes
Spruce Peak (Arlington)	3033	no
Stratton Mountain	3940	yes
Styles Peak	3394	yes
Unnamed #1 Deerfield Ridge	3356	yes
Unnamed #1 N of Farr Peak	3384	yes

Table 1 (continued).

<u>Mountain</u>	<u>Elevation (ft)</u>	<u>Observed 1992-2000</u>
Unnamed #2 N of Farr Peak	3382	yes
Unnamed #1 N of Glastenbury Mountain	3399	no
Unnamed #1 N of Kid Gore	3047	no
Unnamed #2 N of Kid Gore	3303	no
Unnamed #3 N of Kid Gore	3412	no
Unnamed #1 W of Glastenbury Mountain	3423	no
Unnamed #2 W of Glastenbury Mountain	3365	yes
Unnamed E of Goshen Mountain	3120	yes
Unnamed E of Story Spring	3100	no
Unnamed N of Stratton Pond	2940	no
Unnamed N of Styles Peak	3206	yes
Unnamed NE of Mount Roosevelt	3348	yes
Unnamed S of Branch Pond	3080	no
Unnamed S of Gillespie Peak	3200	yes
Unnamed S of Glastenbury Mountain	3331	yes
Unnamed S of Mount Abraham	3549	yes
Unnamed SE of Glastenbury Mountain	3150	no
Unnamed SW of Stratton Mountain	3491	yes
White Rocks	2756	no
Worth Mountain	3234	yes

Table 2. Peaks surveyed for Bicknell's Thrush in the White Mountain National Forest, 1992-2000.

<u>Mountain</u>	<u>Elevation (ft)</u>	<u>Observed 1992-2000</u>
Big Attitash Mountain	2936	no
Black Mountain (Jackson)	3236	no
Bondcliff	4265	yes
Cannon Mountain	4040	yes
Carter Dome	4832	yes
Carter Notch	3388	yes
Eastman Mountain	2936	no
East Osceola	4156	yes
East Royce Mountain	3114	yes
Galehead Mountain	4024	yes
Gulf Peak	4692	yes
Hurricane Mountain	3015	no
Imp Mountain	3165	yes
Jennings Peak	3500	yes
Kearsarge North	3268	no
Kinsman Mountain (North Peak)	4293	yes
Kinsman Mountain (South Peak)	4358	yes
Little Haystack	4800	yes
Middle Carter Mountain	4584	yes
Middle Moat Mountain	2805	no
Middle Moriah	3755	yes
Mount Adams	5774	yes
Mount Avalon	3442	yes
Mount Bond	4698	yes
Mount Carrigain	4580	yes
Mount Chocorua	3475	yes
Mount Clay	5532	yes
Mount Crawford	3119	yes
Mount Davis	3819	yes
Mount Field	4321	yes
Mount Flume	4328	yes
Mount Garfield	4480	yes
Mount Guyot	4508	yes
Mount Hale	4054	yes
Mount Hight	4675	yes
Mount Isolation	4005	yes
Mount Jackson	3952	yes
Mount Jefferson	5712	yes
Mount Lafayette	5249	yes
Mount Liberty	4459	yes
Mount Madison	5367	yes

Table 2 (continued).

<u>Mountain</u>	<u>Elevation (ft)</u>	<u>Observed 1992-2000</u>
Mount Passaconaway	4060	yes
Mount Pemigewasset	2557	no
Mount Resolution	3428	yes
Mount Tecumseh	4004	yes
Mount Tom	4051	yes
Mount Tremont	3371	no
Mount Tripyramid (Middle)	4140	yes
Mount Tripyramid (North)	4180	yes
Mount Tripyramid (South)	4100	yes
Mount Washington	6288	yes
Mount Waumbek	4005	yes
Mount Webster	3910	yes
Mount Whiteface	3994	yes
Mount Willey	4302	yes
Mount Wolf	3478	yes
Noon Peak	2976	yes
North Carter Mountain	4589	yes
North Moat Mountain	3196	no
North Twin Mountain	4761	yes
Owls Cliff	2940	no
Sandwich Mountain	3993	yes
Shelburne Moriah Mountain	3735	yes
South Carter Mountain	4420	yes
South Moat Mountain	2749	no
South Twin Mountain	4902	yes
Speckled Mountain	2906	yes
Stairs Mountain	3460	yes
Stinson Mountain	2840	no
Table Mountain	2675	no
The Cannon Balls	3769	yes
The Sleepers	3870	yes
West Bond	4540	yes
West Royce Mountain	3116	yes
Wildcat A	4422	yes
Wildcat D	4062	yes
Zealand Mountain	4260	yes

Table 3. Estimates of the extent of suitable Bicknell’s Thrush habitat in the U.S. Figures given as number of hectares.

	Catskill Mts.	Adirondack Mts.	Vermont	New Hampshire	Maine	Total
VINS	506	26,037	8,610	49,733	26,048	110,934
Atwood et al. (1996)	17,035	41,674	16,731	54,480	24,803	154,723
Miller-Weeks and Smoronk (1993)	na	28,393	4275	34,154	na	na

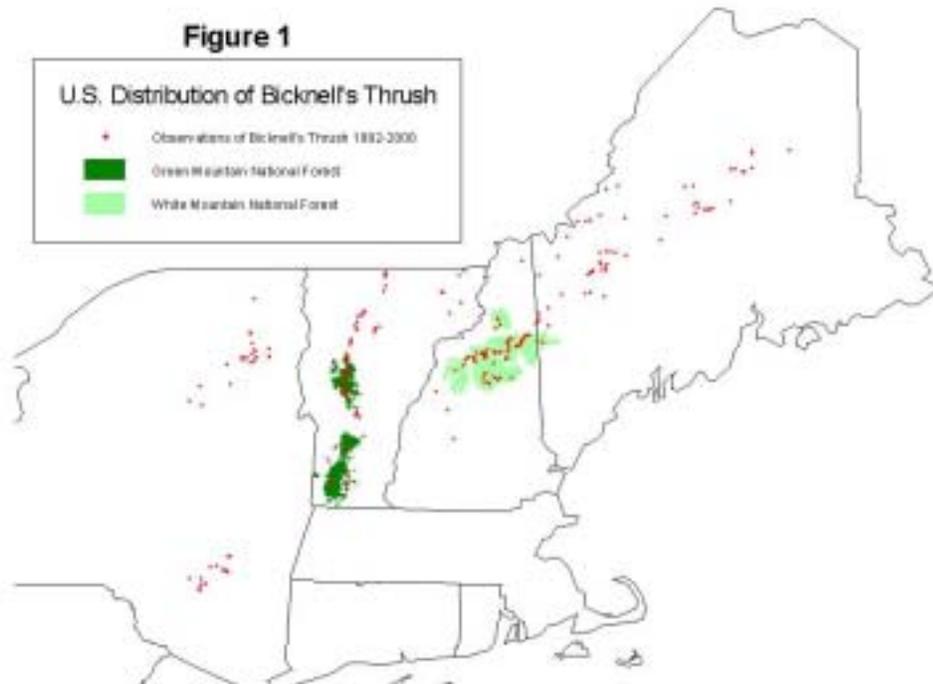
Table 4. Area results of a distribution model describing the probability of detecting at least one vocalizing Bicknell’s Thrush in five 50-m radius point counts from 1993-1997 (Hale 2001).

PROBABILITY DECILE	Area (Ha)	% Area
0.0 - <0.1	1,587	1.7
0.1 - <0.2	30,805	33.7
0.2 - <0.3	35,332	38.7
0.3 - <0.4	13,791	15.1
0.4 - <0.5	5,816	6.4
0.5 - <0.6	2,783	3.0
Total	90,114	100.0

APPENDIX

Explanations of Natural Heritage state and provincial ranks for Bicknell's Thrush.

- S1:** Very rare, generally 1 to 5 occurrences believed to be extant and/or some factor(s) making it especially vulnerable to extirpation from the state/province
- S2:** Rare, generally 6 to 20 occurrences believed to be extant and/or some factor(s) making it vulnerable to extirpation in the state/province
- S3:** Uncommon, believed to be more than 20 occurrences and/or there is some threat to it in the state/province
- S4:** Apparently secure in state/province, often with more than 100 occurrences
- SH:** Known from historical records only
- SZ:** Not of practical conservation concern because there are no definable occurrences
- ?:** Denotes questionable breeding status
- B:** Breeding status; e.g. S12B is a rare to very rare breeder
- N:** Nonbreeding status; e.g. SN, SHB is a nonbreeder known only to breed historically, and SZN is a migrant that occurs in an irregular, transitory, and/or dispersed manner



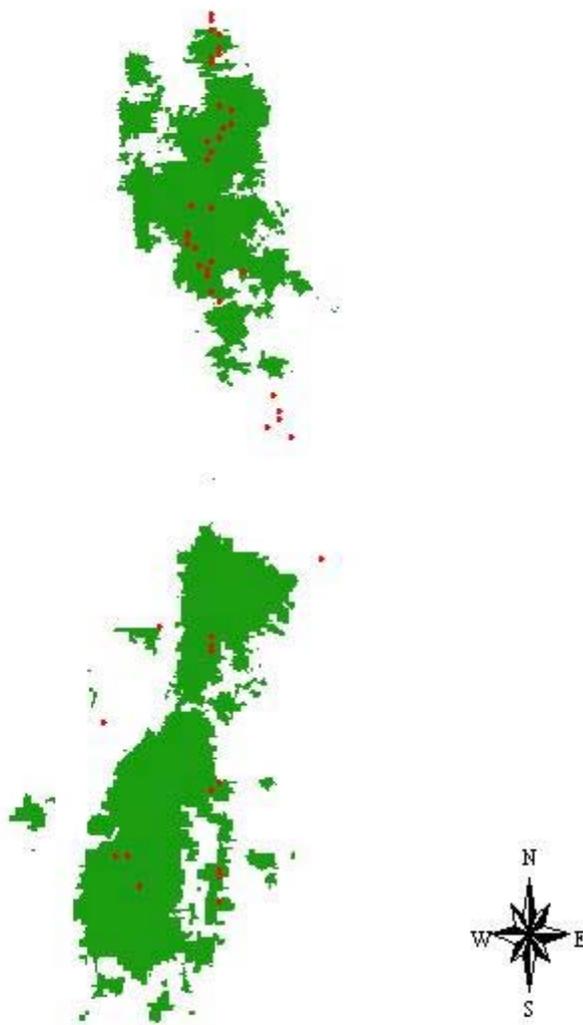
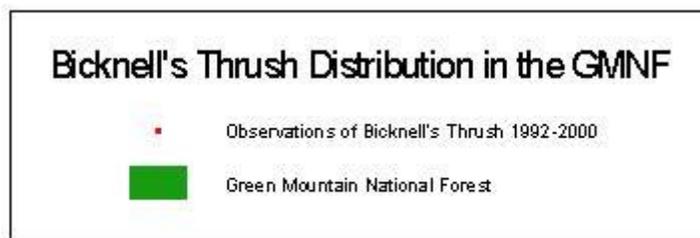


Figure 2



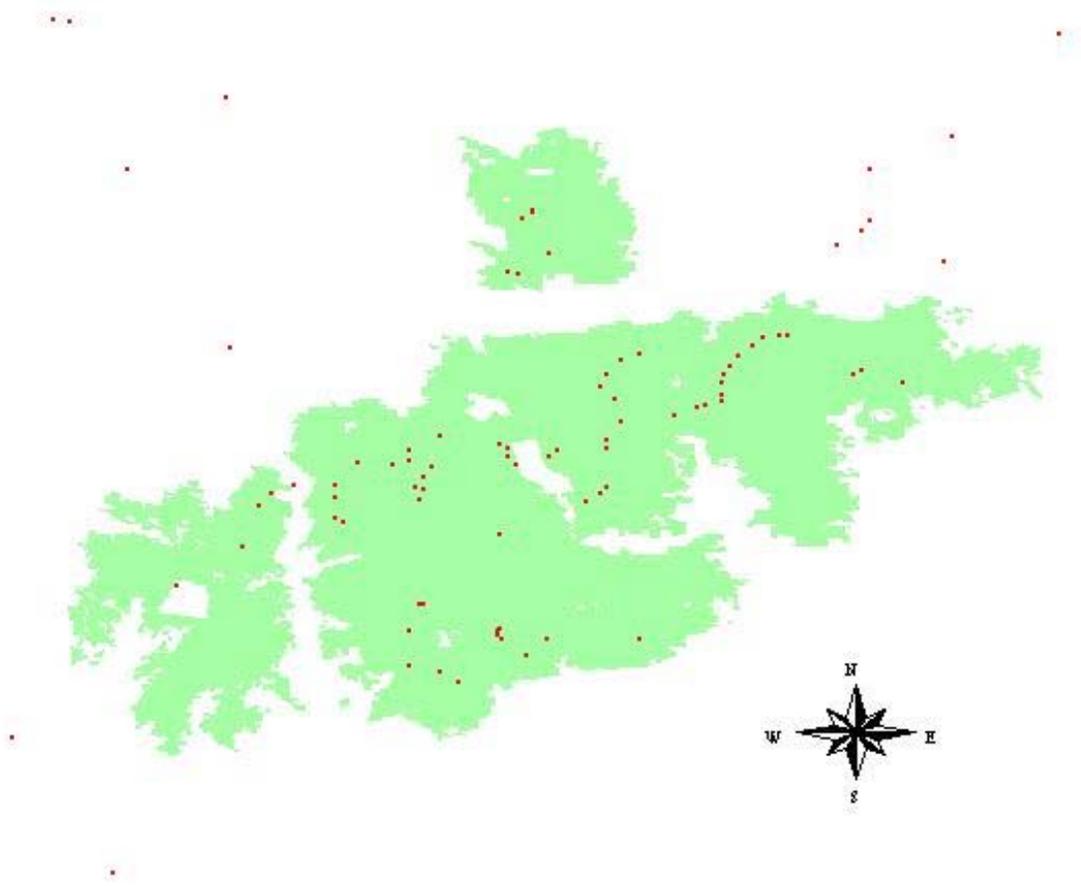


Figure 3

