

Fall 2007

Status, distribution and broad scale habitat features associated with remnant populations of New England cottontails (*Sylvilagus transitionalis*)

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STATUS, DISTRIBUTION AND BROAD SCALE HABITAT FEATURES
ASSOCIATED WITH REMNANT POPULATIONS OF NEW ENGLAND
COTTONTAILS (*SYLVILAGUS TRANSITIONALIS*)

BY

JEFFREY P. TASH
Richard Stockton College of New Jersey, 1998

THESIS

Submitted to the University of New Hampshire
in Partial Fulfillment of
the Requirements for the Degree of

Master of Science
in
Natural Resources:
Wildlife

September, 2007

UMI Number: 1447910

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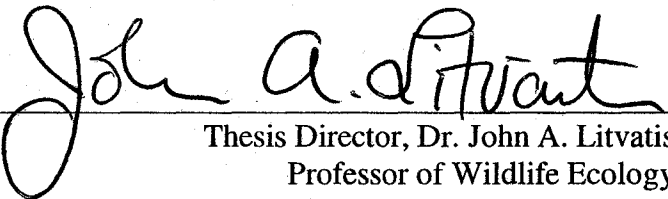
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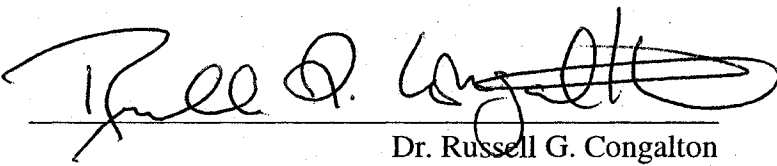
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ACKNOWLEDGEMENTS

I would like to thank my thesis advisor, John Litvaitis, for his support and advice throughout my time at the University of New Hampshire. I am especially grateful for his patience and great sense of humor. I would also like to thank my committee members, Russ Congalton and Mike Amaral for providing statistical advice, GIS software and early reviews of this thesis. Mr. Amaral was especially helpful in initiating this project.

Financial support was provided by U.S. Fish and Wildlife Service, McIntire-Stennis Cooperative Research Funds in Forestry, Maine Department of Inland Fisheries and Wildlife, New Hampshire Fish and Game Department, Vermont Department of Fish and Wildlife, Massachusetts Division of Fisheries and Wildlife, New York Department of Environmental Conservation, and Rachel Carson National Wildlife Refuge. New York Department of Environmental Conservation, Connecticut Department of Environmental Protection, and Rhode Island Department of Environmental Management also provided field support. Many individuals provided help throughout this study.

Special thanks to K. Boland, J. Cardoza, B. Clark, B. Conway, M. Cooper, M. Ellingwood, B. Drumm, H. Herrmann, A. Hicks, R. Innes, T. Goode, R. Hanson, W. Jakubas, J. Jaycox, B. Johnson, V. Johnson, J. Kanter, H. Kilpatrick, V. Kjoss, A. Kovach, M. Litvaitis, M. Marchand, K. Morris, P. Nye, M. Oberkrieser, K. O'Brien, J. Oehler, J. Organ, S. Parren, A. Shutt, B. Tefft, and R. Wood for their assistance with various aspects of this project. Methods used to capture and handle lagomorphs were approved by the University of New Hampshire Animal Care and Use Committee.

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ABSTRACT

STATUS, DISTRIBUTION AND BROAD SCALE HABITAT FEATURES ASSOCIATED WITH REMNANT POPULATIONS OF NEW ENGLAND COTTONTAILS (*SYLVILAGUS TRANSITIONALIS*)

By

Jeffrey P. Tash
University of New Hampshire September 2007

Since 1960 the range occupied by New England cottontails (NEC, *Sylvilagus transitionalis*) in the northeastern United States has declined dramatically. Populations in some regions are known to be vulnerable to extirpation, but little was known about the status of populations in most areas. A range-wide survey of NEC was conducted from 2000 to 2004 to determine the current distribution and status of remnant populations. Because NEC are sympatric with eastern cottontails (*Sylvilagus floridanus*) and snowshoe hares (*Lepus americanus*) throughout much of their historic range, identity of resident lagomorphs was based on DNA either extracted from tissue of live-captures or from fecal pellets collected in occupied patches of habitat. A total of 2301 patches of suitable habitat within 287 quads were searched for the presence of NEC. Of these, 162 patches and 87 quads were considered to be occupied. Five disjunct populations were identified in approximately 14% of the historic range of NEC. Forest maturation and fragmentation are the most plausible explanations for the widespread decline of NEC. Contraction of the historic distribution was toward eastern and southern edges where a

variety of anthropogenic disturbances (e.g., brushy edges of highways and railroad corridors and idle portions of agricultural fields) provided habitat. Land-use activities (expanding development and limited forest management) within the currently occupied range of NEC suggest a continued decline of suitable habitats.

Spatial information from the range-wide survey was incorporated into a geographic information system to examine habitat features associated with remnant populations of NEC at two spatial scales. The regional scale characterized habitats within survey sample units, 7.5 minute topographic quadrangles (quads, ~40 x 10 km) that were occupied by NEC or vacant. The landscape scale described habitats within a 1-km radius of occupied patches and an equivalent sample of vacant patches. At the regional scale, northeastern and southeastern populations were associated with human-dominated habitats with a greater abundance of developed and disturbed lands, less forest coverage, more edge habitats, and less snow fall than unoccupied quads. Landscapes occupied by NEC in these regions were characterized by a greater abundance of potential dispersal corridors than unoccupied landscapes. In contrast, quads occupied by NEC in the southwestern portion of the historic range were in rural areas that were dominated by forests and agricultural fields. At the landscape scale, southwestern populations were affiliated patches of habitat surrounded by more agricultural lands than patches that were not occupied by NEC. Logistic and autologistic regression models were then developed to identify habitats suitable for restoration or translocation within each region. The modeling effort identified approximately 740,000 ha of suitable habitat within the historic range of NEC. This included nearly 90,000 ha on conservation or other public lands. A

total of 1487 individual sites were identified as potential management targets with 155 ranked as having high value for restoration or translocation. The results suggest that initial restoration efforts be directed toward expanding existing populations of NEC. Next, habitat connections should be developed among these populations. Finally, new populations should then be established via translocation in portions of the historic range that are vacant. In addition to promoting New England cottontails, management of early-successional and shrub-dominated habitats in the northeastern United States will benefit other taxa of conservation concern that are dependent on these habitats.

CHAPTER I

INTRODUCTION

Concern over the conservation status of New England cottontails in the northeastern United States dates back to the early 1970's (Johnston 1972, Jackson 1973, Chapman and Morgan 1973). A comprehensive survey of southern (Johnston 1972) and northern (Jackson 1973) New England reveals a shift from a near continuous distribution throughout the region to spotty, disjunct populations in many areas. Litvaitis et al. (1999) later summarized existing information and indicated that the area once occupied by NEC had declined by more than 80% since 1960. This apparent decline has prompted several conservation organizations to take action. The Northeast Nongame Technical Committee listed the NEC as a priority for additional research on current abundance and factors limiting population growth (Therres 1999). In September 2000, this committee further recommended that NEC be elevated to highest conservation priority in the region. More recently, the U.S Fish and Wildlife Service (2006) designated the NEC a candidate for formal consideration as a threatened or endangered species. Although several hypotheses have been proposed to explain the apparent decline in the distribution and abundance of NEC (Litvaitis et al. 2007), two seem to be the most revealing: 1) habitat loss via forest maturation and 2) negative interactions (competition and hybridization) with expanding populations of eastern cottontails.

Role of habitat loss

Land use history of the northeastern United States has had a profound effect on the distribution and abundance of New England cottontails (Litvaitis 1993a, Litvaitis et al. 1999). Prior to the European settlement of the region, NEC likely occupied a variety of early-successional habitats including native thickets associated with rocky outcrops, riparian corridors and shrub-dominated wetlands, as well as regenerating forests created by small [e.g., inundation by Beavers (*Castor canadensis*), lightning strike, or windthrow] or large-scale (e.g., hurricanes, wildfires or fires intentionally set by native Americans) disturbances (Cronon 1983). The clearing of land for agriculture and subsequent abandonment made disturbance generated early-successional habitats widely available during the late 19th and early 20th centuries (Litvaitis 1993a, Litvaitis et al. 1999, Brooks 2003). Cottontail populations likely responded to this successional wave, reaching peak abundance during the early 1900s (Litvaitis 1993a). By the second half of the twentieth century, however, most forest stands in the region had matured into closed canopy second growth forests leading to the decline of NEC and other early-successional dependent species such as the golden-winged warbler (*Verminova chrysoptera*, Litvaitis 1993a). Litvaitis (1993a) modeled the succession of abandoned agricultural lands in New Hampshire in relation to changes in the abundance and distribution of NEC. According to the model, most early-successional habitats that resulted from abandoned farmland had matured into unsuitable mid-successional forests by 1960. Concurrent with this loss of habitat, the range occupied by NEC in New Hampshire had declined from a near continuous distribution throughout 60% of the state in 1950 to disjunct populations that currently span <20% of the state (Litvaitis 1993a). This pattern of population expansion

followed by contraction has likely occurred throughout the range of NEC.

As forests have continued to mature throughout the region (Brooks 2003), remnant populations of NEC have become increasingly isolated and fragmented. As a result, NEC are often found on small, sub-optimal patches of habitat embedded in human-dominated landscapes (Litvaitis and Villafuerte 1996). Small patches may have been typical of habitats occupied by NEC in pre-Columbian landscapes but have limited value in contemporary landscapes. In New Hampshire, Barbour and Litvaitis (1993) found skewed sex ratios and lower quality forage in small patches (<3 ha) versus large patches of habitat. Rabbits also foraged in sites further from escape cover, increasing the risk of predation on small patches (Barbour and Litvaitis 1993). This risk may be significantly heightened in highly altered landscapes because generalist predators are often abundant due to the increased access of food resources (Oehler and Litvaitis 1996). As a result, survival rates of cottontails on small patches in human dominated landscapes are so low that these patches functioned as demographic sinks (Barbour and Litvaitis 1993, Brown and Litvaitis 1995, Villafuerte et al. 1997).

Role of interactions with eastern cottontails (*Sylvilagus floridanus*)

Fay and Chandler (1955) suggested that rapidly expanding populations of eastern cottontails (*Sylvilagus floridanus*) may have contributed to the decline of NEC populations. Facilitated by frequent introductions by state wildlife agencies and private hunting clubs during the first half of the twentieth century (Johnston 1972, Chapman and Morgan 1973), the expansion of eastern cottontail populations into the northeastern United States was roughly simultaneous with the decline of NEC (Johnston 1972,

Jackson 1973). Because eastern cottontails are approximately 20% heavier than NEC (Litvaitis et al. 1991) and occupied similar habitats, it was assumed to be able to invade and exclude the smaller NEC from suitable patches of habitat (Chapman and Morgan 1973). However, a series of field trials using large enclosures failed to demonstrate a social hierarchy among NEC and eastern cottontails (Probert and Litvaitis 1996). Therefore, available evidence does not support the role of interference competition on the current state of NEC populations.

Scramble competition between NEC and eastern cottontails may offer an alternate explanation. Because much of the habitat of NEC and eastern cottontails is disturbance dependent (Litvaitis 1993a), patches of habitat effectively “shift” throughout a landscape as some succeed into unsuitable closed-canopy forests while natural and human disturbances generate new openings that become suitable habitat for cottontails. If eastern cottontails are able to colonize new habitats sooner than NEC, their populations would eventually become more abundant over time. Eastern cottontails are capable of occupying habitats with little cover, whereas NEC are dependent on dense understory vegetation to avoid predation (Barbour and Litvaitis 1993, Smith and Litvaitis 2000). As a result, patches of disturbance generated habitats must be approximately 10 years post disturbance to be suitable for NEC. In contrast, eastern cottontails can exploit newly disturbed patches almost right way. Because neither of these species is capable of dominating the other (Probert and Litvaitis 1996), eastern cottontails could maintain access to new disturbance generated habitats simply based on a system of “prior rights.”

This scenario was supported by a series of experiments conducted by Smith and Litvaitis (2000). They used large enclosures to manipulate access to food and food

quality relative to distance from escape cover. NEC consistently sacrificed food quality and body condition (measured by body weight) for safety by remaining in close proximity to cover. In contrast, eastern cottontails avoided low quality food by foraging in sites away from cover. Smith and Litvaitis (1999) speculate that eastern cottontails have morphological adaptations that allow it to occupy sites with limited cover where NEC would be vulnerable to intense predation. Specifically, eastern cottontails have a larger surface area of the eye exposed and are thus able to detect an approaching predator sooner than NEC. This adaptation may enable eastern cottontails to forage in areas distant from cover without succumbing to predation. Using the results from their feeding trials, Smith and Litvaitis (2000) estimated that NEC could exploit only 32% of the available habitat in a fragmented landscape without experiencing elevated rates of predation. Eastern cottontails, however, could effectively utilize 99% of the habitat.

In addition to competition, hybridization with eastern cottontails has been proposed as a possible explanation for the decline of NEC (Chapman and Morgan 1973). However, a range-wide survey did not reveal any evidence of hybridization between these two species (Litvaitis et al. 1997). It's worth noting that northern populations of NEC are sympatric with snowshoe hares (*Lepus americanus*) and that both species require dense understory vegetation. In Maine, Litvaitis et al. (2003) reported significant differences in habitat use at the landscape scale among NEC and snowshoe hares. However, some overlap in habitat preference was observed. Therefore, interactions between these two species may warrant additional research.

Goals and objectives

In summary, populations of NEC have responded to a dramatic increase and subsequent decline of early-successional habitats that followed the regional abandonment of agriculture (Litvaitis 1993a). Contemporary populations encounter landscapes that are substantially modified from pre-Columbian conditions. Remnant populations of NEC are often found on small, disjunct habitat patches in human-dominated landscapes (Barbour and Litvaitis 1993). Populations of generalist predators are especially abundant in these types of landscapes (Oehler and Litvaitis 1996) and may limit the ability of NEC to persist long-term (Brown and Litvaitis 1995). In the southern portions of their range, populations of NEC may be further hampered by the differential success of recently established populations of eastern cottontails (Probert and Litviatis 1996). As a result of these factors, populations of NEC are likely to continue to decline without human intervention. Any restoration or recovery effort would most likely include a translocation program and extensive habitat management (e.g., creation of early-successional habitat). The biggest challenge in designing such a program is identifying suitable sites where self-sustaining populations can be established or existing populations can be managed. However, information critical to this process is currently lacking. The status of extant populations in many areas of the historic range of NEC is unknown. Data on the current distribution is not only necessary for assessing the conservation status of NEC but will also provide a baseline of information to assess the effectiveness of management efforts. Although several authors have researched factors limiting New England cottontails, most have focused on patch specific features or have been limited in geographic extent. Relatively few have focused on larger scales

associated with metapopulations. My research aims to fill these critical information gaps.

Specifically, the objectives of this study are:

1. To determine the current distribution of New England cottontails (NEC) within its recent historic range.
2. To identify broad-scale habitat and climatic features associated with remnant populations of NEC.
3. To identify areas in New England where restoration, management and land acquisition would benefit populations of NEC.

Organization of the following chapters

The following chapters address each of the objectives as stated above. Chapter 2 summarizes the results of a range-wide survey of the distribution of NEC (Objective 1). Chapter 3 provides a geographic information systems based analysis of habitat features associated with remnant populations of NEC (Objective 2). In addition, Chapter 3 identifies potential restoration sites (Objective 3).

CHAPTER II

RANGE-WIDE SURVEY TO DETERMINE THE DISTRIBUTION OF NEW ENGLAND COTTONTAILS

Understanding the factors that influence the distribution and abundance of a species is essential if we are to respond to substantial changes in either of these parameters. Climate, habitat abundance, and interactions with other organisms (e.g., competition and predation) will influence the area that a species can occupy (MacArthur 1972). Consequently, changes in these environmental conditions through time and space can lead to shifts in the distribution of a species (Hengeveld 1990).

Determining the range boundaries of a species can be a difficult and subjective task. For example, the edge of a range can be defined as the outermost extent where the species is still found or where the density falls below an arbitrary level as compared to the center of the occupied range (e.g., 1/20 of the central density; MacArthur 1972). However, Gaston (1994) indicated that geographic ranges should be delineated in essentially two ways, the extent of occurrence or the area of occupancy. The extent of occurrence is the area contained by the outermost locations of a species, whereas the area of occupancy tends to be smaller because it is the area within which a species is actually found (Gaston 1994).

Most studies of geographic range changes have focused on the patterns and rates of expansion of introduced or invasive species (Channell and Lomolino 2000a). Relatively few studies have examined patterns of range contractions. Exceptions include

Investigations by Channell and Lomolino (2000b), Gabelli et al. (2004), Laliberte and Ripple (2004), and Scott-Morales et al. (2004). Initially, biogeographers suggested that the range contraction of a species was likely to occur toward the center of its geographic distribution (Shelford 1911, Kendeigh 1974, Hengeveld and Haeck 1981). More recently, however, others have noted that populations of many species that are in decline tend to persist at the margins of their historic range (Lomolino and Channell 1995, Channell and Lomolino 2000b). Regardless of the pattern of contraction, human influences have been implicated in range declines of a variety of species (e.g., Laliberte and Ripple 2004, Gabelli et al. 2004, Scott-Morales et al. 2004). Understanding the pattern, trajectory, and underlying causes of range contraction is necessary to develop effective conservation strategies for a species in decline (Simberloff 1986, Channell and Lomolino 2000a, Laliberte and Ripple 2004), such as the New England cottontail (NEC, *Sylvilagus transitionalis*).

Since the early 1970s, wildlife biologists have noted that the abundance and distribution of NEC were declining (Linkkila 1971, Johnston 1972, Jackson 1973). In 1989, the U.S. Fish and Wildlife Service (1989) acknowledged that decline and included NEC as a candidate species (Category 2) for threatened or endangered status. Chapman et al. (1992) subsequently reclassified the taxonomy of NEC and identified two sister species. Populations east of the Hudson River drainage (including southeastern New York, all of Connecticut, Massachusetts and Rhode Island, much of Vermont, New Hampshire, and southwestern Maine) retained the name of New England cottontail. Those to the west and south were designated Appalachian cottontails (*S. obscurus*). To some degree, this reclassification increased concern for NEC because it designated a

much more restricted historic range for the species, and habitats suitable for NEC in the northeastern United States were known to be declining (Litvaitis 1993a). In 1996, the U.S. Fish and Wildlife Service (1996) announced that it would no longer maintain a list of category 2 candidate species. That action prompted the Northeast Nongame Technical Committee to prepare a list of declining species of regional concern (Therres 1999). Litvaitis et al. (1999) later summarized existing information and indicated that the area occupied by NEC had declined by more than 80% since 1960. Most recently, the U.S. Fish and Wildlife Service (2004) began formal consideration of NEC as a threatened or endangered species. In response to these concerns, a range-wide survey was initiated to quantify the current distribution of NEC as an essential step toward reversing recent declines in abundance and distribution. The goal of this survey is to inventory the historic range and determine to what extent that range was still occupied and identify factors that may limit long-term persistence.

Methods

The survey was designed to identify regions that still supported NEC and provide an estimate of the area of occupancy. Field manuals were prepared (Litvaitis et al. 2002) and training sessions were conducted for all survey participants to assure consistent application of the search methods.

Sample units

The survey was conducted throughout the known historic range of the New England cottontail, approximately 90,000 km² in the northeastern United States (Fig. 2-1). A template was overlaid onto the historic range of NEC and 7.5-minute USGS topographic quadrangles (quads, ~13.9 x 10 km) were considered potential sample units

(Fig. 2-1) because all survey participants were familiar with these maps. Due to the ephemeral nature of most habitats occupied by NEC (Litvaitis 2001), recent (since 1990) locations of hunter harvests, road kills, and field surveys of NEC that were collected by agency biologists served as starting points within each state (Fig. 2-1). Sampled quads were divided into four equal subunits (quarter quads) to facilitate systematic searches. Within each quarter quad, 2-4 patches (8-16 patches/quad) of suitable habitat were searched for presence of NEC. The limited availability of NEC habitat resulted in searches of less than 8 patches/quad in some areas. Surveys were restricted to periods when snow would likely facilitate detection of lagomorph activity (e.g., tracks and increased visibility of fecal pellets; Dec - Mar) during 2000 – 2001 through 2003 – 2004 field seasons.

Field protocol

Candidate patches of habitat were considered suitable for NEC if they had dense (>9,000 stems/ha), primarily deciduous, understory cover (Fay and Chandler 1955, Barbour and Litvaitis 1993, Probert and Litvaitis 1996). Litvaitis (1993) found that NEC occupied patches that ranged in size from 0.2 to >15 ha. Despite the inherent volatility of populations on small patches (Barbour and Litvaitis 1993), no minimum patch size was used during the survey because of the limited availability of habitat in many areas.

Patches of habitat were located by following roads in each quarter quad. Although roadside surveys are potentially biased (e.g., land uses likely differ immediately adjacent to roads and not all suitable habitats are visible from roads), they were considered necessary as a practical alternative to the expense of obtaining and reviewing aerial photographs. Additionally, most of the historic range of NEC has a dense network

of roads. Thus, the survey would miss only a small proportion of the habitat suitable for NEC. This assumption was evaluated during initial application of the field protocol in Maine where a biologist from the Maine Department of Inland Fisheries and Wildlife searched a portion of the study area for suitable habitat using aerial photographs (H. Givens, Maine Department of Inland Fisheries and Wildlife, personal communication). An independent inventory was conducted by driving through the same area. Upon completion of both efforts, 86% of the suitable sites identified with the aid of aerial photographs were searched during the roadside survey.

Next, candidate patches were searched for evidence of lagomorph activity, including tracks in snow, clipped twigs, or fecal pellets (Fig 2-2). Patches were searched for 20 minutes unless evidence was encountered before that time elapsed. This criterion was established during the initial surveys in which lagomorph sign was never found after the first 10 minutes of searching a new patch (B. Johnson, University of New Hampshire, unpublished data). Searches focused on the densest understory cover within a patch because these sites are selected by NEC (Barbour and Litvaitis 1993). However, an effort was made to distribute searches throughout all portions of a patch. Particular attention was directed to favored browse species, especially raspberry and blackberry (*Rubus* spp.), willow (*Salix* spp.), aspen (*Populus tremuloides*), and red maple (*Acer rubrum*) (Barbour 1993). The 45° cut on twigs clipped by rabbits or hares is easily distinguished from twigs clipped by deer (*Odocoileus virginianus*) or moose (*Alces alces*).

If lagomorph activity was detected, identity was determined using one of several approaches (Fig. 2-2). In some instances, fresh tracks in soft snow were sufficient to differentiate among sympatric NEC, eastern cottontails (*S. floridanus*), and snowshoe

hares (*Lepus americanus*). To differentiate between cottontail species or when trackable snow was not present, early efforts (2000-2001 field season) relied on livetraps (up to 12 traps/patch) to capture resident lagomorphs. Traps were set and monitored until at least one hare or rabbit was captured. Initial identity of captured individuals was based on pelage and morphological characteristics (Litvaitis et al. 1991), and tissue samples were collected from cottontails to subsequently confirm identity by using sequence patterns of extracted mitochondrial DNA (mtDNA). Field collection and laboratory analysis of tissue samples were described by Litvaitis and Litvaitis (1996).

To maximize sampling efficiency, live-trapping was abandoned in subsequent years and relied exclusively on mtDNA analysis of fecal pellets collected from occupied patches. Initial application of this technique indicated a high success rate (94%) in identifying the source (Kovach et al. 2003). At least 6 fecal pellets were collected from different areas of an occupied patch. Identity of resident lagomorphs was based on the first pellet that yielded useable DNA. This was comparable to the livetrapping efforts, where the first captured individual was assumed to be the resident species. A summary of the analytical procedures and limitations of this method were described by Kovach et al. (2003).

Geographic range delineation

The search was initiated in each state by verifying the presence of NEC in quads that were known to contain populations of NEC as far back as 1990 (Fig. 2-1). Quads that contained verified populations of NEC were classified as *occupied* and searches within that quad were discontinued. Searches then continued to all abutting quads (Fig. 2-3). If none of these quads were occupied, the search was extended to the next tier of

quads, where 50% were searched in an alternating pattern (Fig. 2-3). During these searches, any quad with a verified population of NEC was considered occupied and served as a new starting point for additional searches. The current range occupied by NEC was defined as the area contained within all occupied quads. The magnitude and rate of range decline were estimated by comparing the historic range (Hall and Kelson 1959, Godin 1977, Chapman et al. 1992) with the sum of the areas currently occupied by NEC.

Results

A total of 2,301 patches within 287 quads were searched for the presence of lagomorphs (Fig. 2-1). Of these, 162 (7.0%) patches (Appendix A for locations) in 87 quads were occupied by NEC. Additionally, eastern cottontails occupied 760 patches (33.0%), snowshoe hares occupied 277 patches (12.0%), and 895 patches (38.9%) were vacant. Lagomorph identity was not determined for 122 sites because of previously detected NEC within these quads; 9 samples were lost or mislabeled, and 76 pellet samples failed to yield sufficient mtDNA for genetic analysis. The failed samples did not include 44 patches in Connecticut that were first sampled late in the winter of 2003 when unseasonably warm temperatures and rain likely degraded the DNA and prevented species identification (Kovach et al. 2003). These patches were resampled during the 2003–2004 field season and successfully identified to achieve at least 8 samples/quad.

The estimated occupied range of NEC was determined to be 12,180 km². This represents a reduction of approximately 86% of the historic range (Fig. 2-4), and an annual rate of loss of 2% since 1960. Remnant populations of NEC are restricted to five

core regions: (1) seacoast region of southern Maine and New Hampshire, (2) Merrimack River Valley of New Hampshire, (3) a portion of Cape Cod, Massachusetts, (4) eastern Connecticut and Rhode Island, and (5) portions of western Connecticut, eastern New York, and southwestern Massachusetts (Fig. 2-4). The approximate area occupied by these populations ranged from 1260 (Population #2) to 4760 km² (Population #5).

Discussion

A number of possible explanations have been offered for the rapid decline of NEC populations, including habitat loss and fragmentation, competition with expanding populations of eastern cottontails, and genetic swamping via hybridization with eastern cottontails (Litvaitis et al. 2007). Of these explanations, habitat loss and fragmentation have had the most profound effects (Litvaitis 1993a, Litvaitis and Villafuerte 1996). Historically, NEC likely inhabited a variety of early-successional forests and native shrublands throughout the northeastern United States (Litvaitis 2001). Early-successional forests became widely available after widespread abandonment of farmlands in the late 19th and early 20th centuries (Litvaitis 1993a). By 1960, most of these second-growth forests were maturing into closed-canopy stands that lacked the necessary understory vegetation to support NEC (Barbour and Litvaitis 1993, Litvaitis 2001). As a result, populations quickly declined (Litvaitis 1993a). This apparent “rise and fall” of NEC populations in response to land use changes might imply that the observed range contraction is not cause for concern because populations have simply returned to levels that are more reflective of pre-Columbian conditions. However, the remaining, disjunct populations of NEC (Fig. 2-4) most likely do not represent a stable condition for long-

term persistence. Remaining habitats occupied by NEC are often fragmented by dense road networks and suburban developments (Litvaitis et al. 1999, Litvaitis 2003) and NEC are spatially structured as induced metapopulations (Litvaitis and Villafuerte 1996). In such an arrangement, small populations are able to persist only because surplus rabbits from one or more large populations regularly disperse to small patches of habitat. Intense predation by elevated population of generalist carnivores (Oehler and Litvaitis 1996) on small patches causes these habitats to function as demographic sinks (Barbour and Litvaitis, 1993, Villafuerte et al. 1997).

Pattern of range contraction

Differences in how the historic range and the current range were estimated could have potentially biased the comparisons between them. The historic range of NEC was most likely based on extent of occurrence (an outline of the most distant records), whereas the current range is based on the area of occupancy (sum of sample units actually occupied). Essentially, this is comparing a relatively liberal estimate of the historic range to a more conservative estimate of the existing distribution of NEC. However, the magnitude of change between the historic and current distributions is too large to be simply a consequence of using different estimators. Thus, the change in area occupied by NEC between 1960 and now represents a real and substantial decline.

The present distribution of NEC represents a contraction toward eastern and southern limits of the species' historic range (Fig. 2-4), indicating agreement with the contagion hypothesis proposed by Channell and Lomolino (2000*b*). That hypothesis was derived from the declining-species paradigm that suggests those factors that contribute to reductions in the abundance of a species (e.g., overexploitation, habitat degradation, and

introduced competitors or pathogens) do not spread uniformly through the geographic range of a species. As a result, the contagion hypothesis predicts that the last populations to be affected by extinction forces will persist the longest. Channel and Lomolino (2000b) contended that extinction forces spread across a region like a contagion and regions most isolated (i.e., along the edge of the historical range) should be the last places affected. This may be especially true for human-based disturbances, such as habitat degradation or fragmentation (Lomolino and Channel 1995, Laliberte and Ripple 2004). Does this explanation provide any insight into the observed range contraction by NEC? Based on the short-term suitability of NEC habitats (Litvaitis 2001), I suspected that the range contraction might reflect the pattern of habitat loss through forest maturation or land-use changes (e.g., suburbanization). Previously, Litvaitis (1993) argued that north-south contraction of NEC in New Hampshire could be explained by a more precipitous loss of suitable habitat in northern counties. However, the range contraction in Maine was not associated with differences in the abundance of early-successional forests in currently occupied areas versus vacant portions of the historic range (Litvaitis et al. 2003).

Although the historic range of NEC is small compared to other North American lagomorphs (Wilson and Ruff 1999), there are substantial variations in land-use patterns within this portion of the northeastern United States that may affect local and landscape suitability for NEC (Chapter 3). For example, based on patch-specific information collected in Maine, remnant populations of NEC were largely restricted to some of the most modified habitats within the historic range in that state. Average size of patches occupied by NEC in Maine was only 3.8 ha and 68% of occupied patches were <2.5 ha

(Litvaitis et al. 2003). Barbour and Litvaitis (1993) reported that habitat patches that small functioned as sink habitats for NEC. As a result, the persistence of NEC among small habitat patches in southern Maine was probably facilitated by frequent recolonization. It is worth noting that the major north–south road in Maine (Interstate Highway 95) spans the currently occupied range of NEC in Maine. Here, extensive stretches of Interstate 95 had shrub-dominated margins >10 m wide (Litvaitis et al. 2003). Additionally, rest areas and exit ramps had sufficient disturbance-generated habitats to support 1–2 cottontails (J. Litvaitis, personal observation). Bolger et al. (2001) found that the vegetated edges of highway rights-of-way were frequently used by a variety of small mammals, and these served as movement corridors between disjunct habitat patches. Although Interstate 95 poses a formidable barrier (up to 7 or 8 lanes of vehicle traffic) to east–west movement of rabbits, the substantial habitat associated with this corridor may facilitate north–south movement. Preliminary examination of landscape features (Chapter 3) indicated a similar pattern for the NEC population in the Merrimack Valley of southern New Hampshire (Fig. 2-4, Population #2). There, the majority of patches supporting NEC were associated with utility and railroad corridors. On the other hand, Population #5 (western Connecticut, southeastern New York, and southwestern Massachusetts) was largely associated with agricultural fields, and few multi-laned highways or railroad corridors occurred in that area (Chapter 3). In summary, populations of NEC declined in response to the range-wide loss of suitable habitat as a result of forest maturation and fragmentation. Remaining populations are associated largely with disturbance-generated habitats that are the result of a variety of contemporary land uses.

Conservation implications of range contraction and fragmentation

The survey revealed five distinct populations of NEC (Fig. 2-4). Edge-to-edge distances between neighboring populations ranged from 17 km for Population #1 and #2 to 50 km for Population # 3 and # 4 (Fig. 2-4). Preliminary examination of variation in microsatellite DNA from a sample of 82 cottontails collected throughout the currently occupied range indicated these populations may be genetically distinct (A. Kovach, unpublished data). This suggests that remnant populations are functioning as discrete units with little or no emigration/immigration among them. Whether current genetic structuring is a consequence of long-term isolation imposed by natural barriers (e.g., Connecticut River), a result of more recent landscape modifications (e.g., urbanization), or some combination of these features, is not known. What is obvious is that remaining populations are sufficiently isolated to warrant concern. Fragmentation at the local landscape scale has been demonstrated to substantially increase the risk of extinction of individual populations of NEC (Litvaitis and Villafuerte 1996). Fragmentation at a regional scale also increases the likelihood of extinction by eliminating the rescue effect of dispersal (Fagan et al. 2002).

Efforts to improve the long-term viability of NEC should occur at two spatial scales. At the population or landscape scale, efforts to enhance existing populations should consider the abundance of suitable habitat. Human densities and associated landscape modifications vary substantially among the areas that contain the five remaining populations (Chapter 3). As a result, habitat manipulations will need to be population or landscape specific (Litvaitis 2001, 2003). For example, in human-dominated landscapes, where suburban-urban developments with dense road networks

substantially influence natural disturbance regimes and affect animal dispersal opportunities, it may be most effective to dedicate moderate (>10-ha) to large (> 25-ha) tracts that would serve as core habitats (Litvaitis and Villafuerte 1996, Litvaitis 2001). In more rural landscapes, where forests remain largely intact, the resumption of natural disturbances may eventually generate sufficient habitat for early-successional species, including NEC. In many areas, however, forests are dominated by mid-successional stands where human intervention is necessary to provide sufficient early-successional habitats (Litvaitis 2003). Forest management activities (including even-aged management) could be used to offset current shortages of suitable habitats.

At the regional scale, some consideration must be given to facilitating exchanges between adjacent populations. The most practical approach in the short term may be the development of patches of habitat that are affiliated with existing land uses that provide potential dispersal corridors (e.g., powerline rights-of-way). Such “steppingstones” may be the only realistic solution, especially in areas with substantial habitat modifications.

Limitations of this approach

Perhaps the most common concern of any inventory method is the failure to detect a target organism when it is present (e.g., Azuma et al. 1990, MacKenzie et al. 2002). Detection rates for the range-wide survey were not estimated because the different survey methods used (live traps vs. fecal DNA) and variation in survey conditions encountered (e.g., searching for fecal pellets on snow vs. searching on leaf litter) would make the determination of detection rates a substantial undertaking (MacKenzie and Royle 2005, O’Connell et al. 2006). Additionally, the presence of eastern cottontails in central and southern portions of the historic range also would influence detection rates. In these

regions, NEC can be syntopic with eastern cottontails on large patches of habitat (H. Kilpatrick, Connecticut Department of Environmental Protection, personal communication). Under these conditions, it is possible that live-trapping or fecal pellet collections would have a low probability of detecting NEC among habitats that are occupied by both species if NEC were a small proportion of the resident cottontails. Species detection via live captures or collection of fecal pellets would be approximately proportional to their abundance if both techniques have the same detectability for sampling NEC and eastern cottontails. As a result, some occupied patches could have been misclassified. It is important, however, to remember that classification (occupied–vacant) of each sample unit (quads) was based on searching 8–16 patches. This approach can be considered to be conservative for several reasons. First, the majority of patches occupied by NEC among more intensively studied populations in Maine (Litvaitis et al. 2003) and New Hampshire (Litvaitis and Villafuerte 1996) were <3 ha. Subsequent to completing the survey, sampled patches in eastern Massachusetts (Population #3; Fig. 4) and many of those that comprised the largest population in western Massachusetts, southeastern New York, and western Connecticut (Population #5; Fig. 4) were revisited. There, too, the majority of occupied patches were <3 ha (J. A. Litvaitis and J. P. Tash, personal observations) and probably supported no more than 3–4 rabbits. Also, when trackable snow was present, cottontails and hares were never present in the same patch. Therefore, under these circumstances, accepting the identity of the first live capture or fecal pellet analyzed seemed reasonable. To attempt an exhaustive search of each patch would have severely limited the geographic area covered during the survey. On large patches, the possibility of misclassification did exist and small populations of NEC could

have been missed. In the study area, however, large patches of suitable habitat were rare (Litvaitis and Villafuerte 1996, Litvaitis et al. 2003). The problem of misclassification does warrant additional consideration, especially if monitoring programs are developed by individual states to track future changes in the distribution or abundance of NEC (MacKenzie and Royle 2005).

A second consideration is the assumption that NEC no longer exist in areas that were not searched during the study, especially portions of Vermont and eastern Massachusetts (Fig. 2-1). The decision not to search these regions was based on previously obtained information from both areas. Litvaitis (1993b) conducted a live-trapping survey of central and northern Vermont and encountered only eastern cottontails. Further, the most recent record of NEC in Vermont was from 1971 (Jackson 1973) and the nearest population identified since 1990 (western New Hampshire, Fig. 2-1) is now considered extinct (Fig. 2-4). In Massachusetts, Cardoza (1993) reported the geographic distribution of 967 cottontail skulls collected from hunters, road kills, and other sources from 1991 through 1993. Although such a sample clearly lacked design rigor, the distribution of these samples was representative of the large-scale distribution of cottontails in Massachusetts. In that sample, eastern cottontails (96% of the total) were detected in 13 of 14 counties, whereas NEC were collected in 6 counties and none were collected in the eastern portion of the state that was not searched. Additionally, the U.S. Fish and Wildlife Service collected rabbit pellets from thicket habitat in northeastern Massachusetts at Parker River National Wildlife Refuge. These samples were all identified as eastern cottontails (U.S. Fish and Wildlife Service 2005, unpublished data).

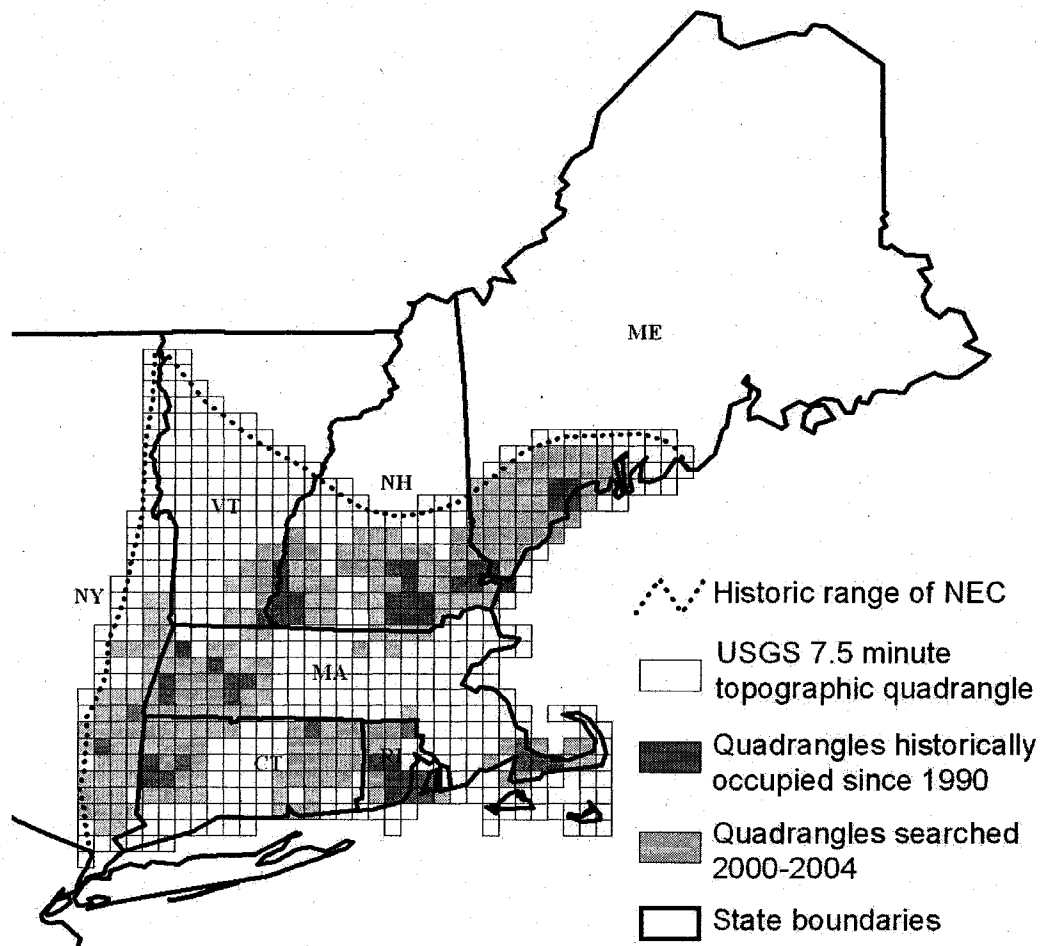


Figure 2-1. Historic range boundary of New England cottontails compiled from Hall and Kelson (1959), Godin (1977), and Chapman et al (1992). Recent locations (since 1990) of remnant populations were used as a starting point for the range-wide survey. Additional areas searched for the presence of NEC are indicated.

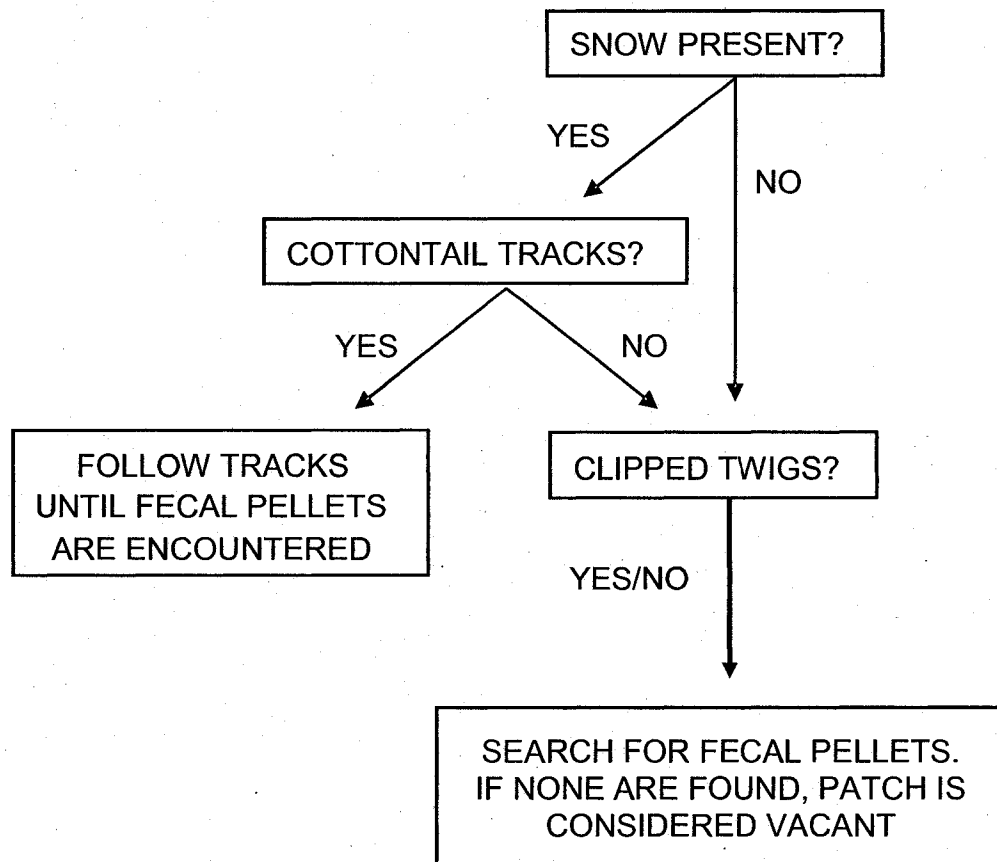


Figure 2-2. An example of the field protocol used to identify patches of habitat occupied by New England cottontails. In this instance, DNA extracted from fecal pellets was the method used to identify resident lagomorphs.

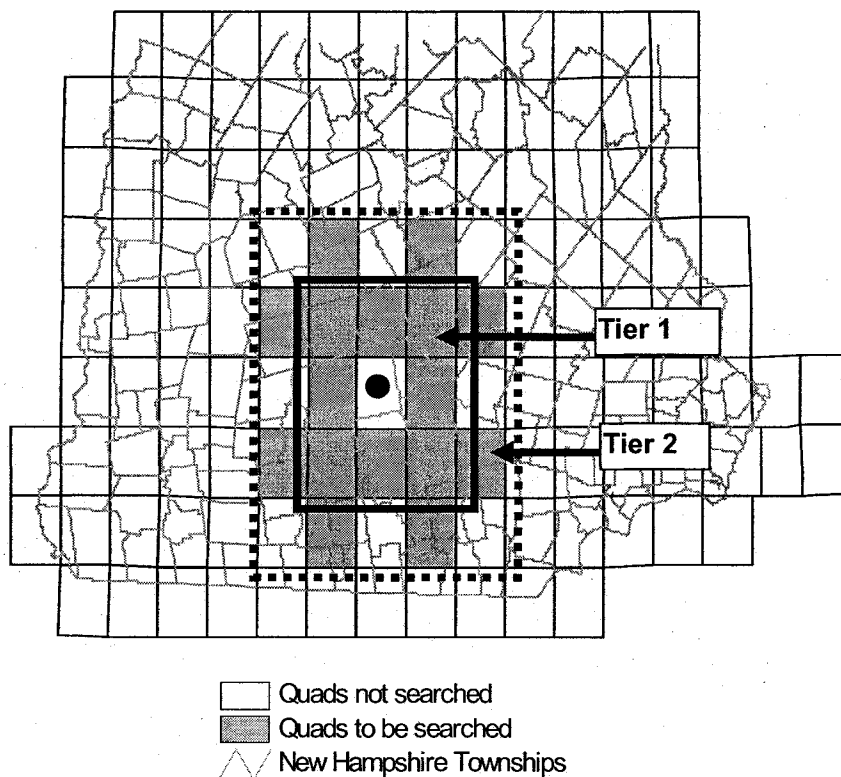


Figure 2-3. An example of the sampling template (based on USGS 7.5 minute topographic quadrangles) used in southern New Hampshire to survey of the historic range of New England cottontails. Field searches began at sites with verified records of cottontails since 1990 (black dot) and continued among immediately adjacent quads (tier 1) and then continued to a second tier where alternate quadrangles were searched. Any new population of New England cottontails encountered in tier 1 or 2 served as a new starting point for additional searches.

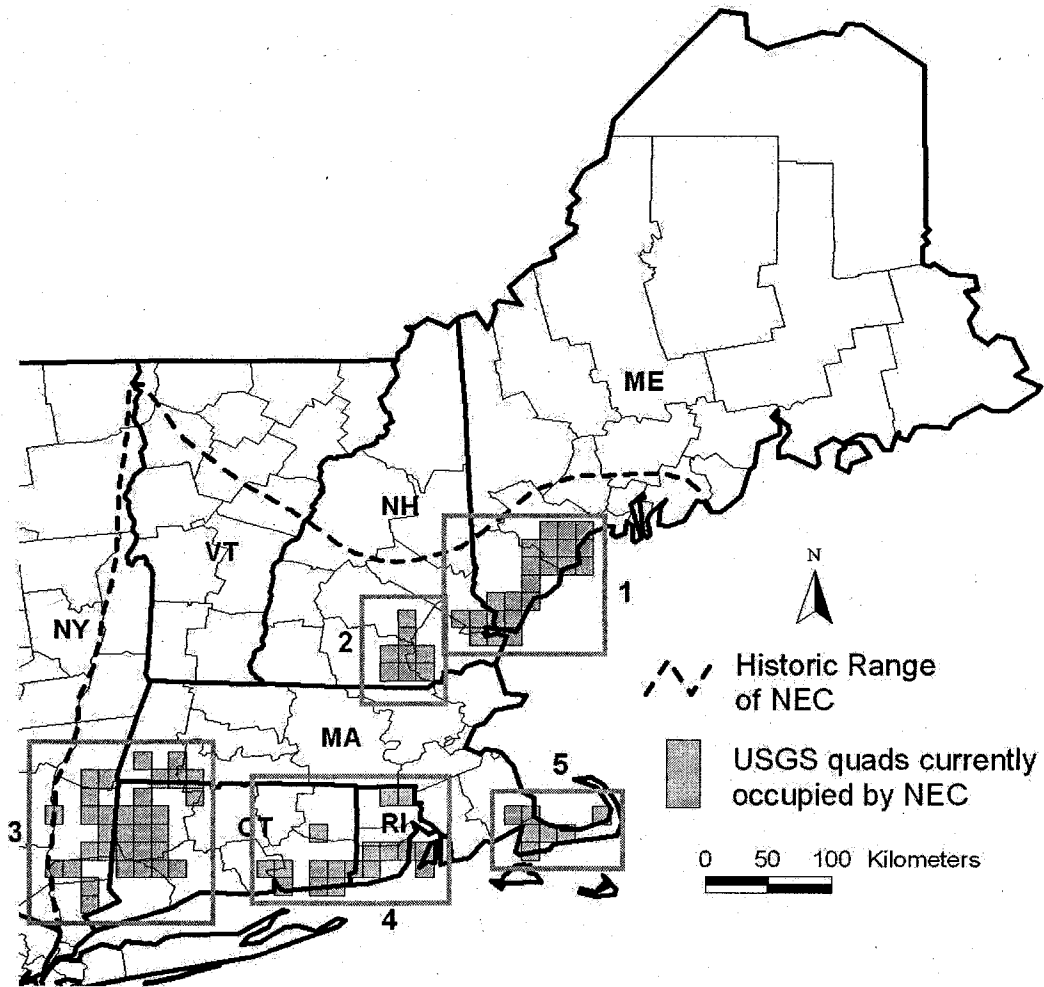


Figure 2-4. Current distribution of New England cottontails. Remnant populations are apparently restricted to five core regions: (1) seacoast region of southern Maine and New Hampshire, (2) Merrimack River Valley of New Hampshire, (3) Cape Cod, Massachusetts, (4) Connecticut east of the Connecticut River and Rhode Island, and (5) western Connecticut, eastern New York and southwestern Massachusetts.

CHAPTER III

BROAD SCALE HABITAT FEATURES ASSOCIATED WITH REMNANT POPULATIONS OF NEW ENGLAND COTTONTAILS

Quantifying the relationship between the distribution of a species and the biotic and abiotic factors that influence that distribution is essential for conservation and population management (Rushton et al. 2004). Most investigations into these relationships have focused on habitat features at specific locations [e.g. resting sites (Weir and Harestad 2003)], occupied patches (e.g., Woolf et al. 2002), or home ranges [e.g., multiple patches (Andrén 1990)]. However, wildlife populations have been shown to respond to habitat variables expressed at a variety of broad scales, from the activity site (Weins, 1973, Brown and Litvaitis 1996), landscape [e.g., multiple home ranges (Brown and Litvaitis 1996, Carroll et al. 1999)], and all the way up to the scale of a species' geographic range (Weins 1973, Laliberte and Ripple 2004). Additionally, an expanded spatial scale recognizes the role of regional and metapopulation influences (Hanski and Gilpin 1991). In metapopulations, local extinctions may occur frequently, but recolonization permits a species to exist at regional scales. Therefore, patterns of regional occupancy are ultimately determined by the balance between local colonization and extinction. For rare species with patchy distributions, examining habitat features at larger spatial scales associated with metapopulations may be especially important because these species are particularly vulnerable to declines in distribution and

abundance (Brown 1984).

Populations of New England cottontails (NEC, *Sylvilagus transitionalis*) have declined dramatically in recent decades (Litvaitis 1993a). A range-wide survey indicated that remaining NEC occupy approximately 14% of the species' historic range (Chapter 2). Although several hypotheses have been proposed to explain this decline, the most likely causes are the loss and fragmentation of preferred habitats (Litvaitis et al. 2007). In some areas, remnant populations now occupy small patches of habitat that are becoming increasingly isolated (Litvaitis and Villafuerte 1996). Previous research on factors limiting NEC have included an examination of patch-specific features (Barbour and Litvaitis 1993) and interactions with the sympatric eastern cottontail (*S. floridanus*) (Probert and Litvaitis 1996, Smith and Litvaitis 2000). These studies were somewhat limited in geographic extent (e.g., Barbour and Litvaitis 1993). Exceptions include Brown and Litvaitis (1995) who examined landscape-scale features associated with predation of NEC in southern New Hampshire. Their findings indicated that the composition of the landscape surrounding an occupied patch had an effect on the ability of a cottontail to avoid predators (Brown and Litvaitis 1995). Litvaitis et al. (2003) also found that variables measured at the landscape scale may have a substantial influence on the source-sink dynamics of NEC populations in southern Maine. Occupied landscapes were characterized by features that facilitated a balance between extinctions and recolonization within a highly fragmented region.

NEC occupy early-successional or shrub-dominated habitats that are characterized by dense understory vegetation (Barbour and Litvaitis 1993, Litvaitis, 2003). These types of habitats are known to be declining throughout the northeastern United States

(Brooks 2003), and it seems unlikely that remaining populations of NEC will continue to persist without active management of these habitats. Recently, the U.S. Fish and Wildlife Service (2006) has designated the NEC as a candidate for threatened or endangered status. Any restoration or recovery effort of NEC would most likely include extensive habitat management (e.g., creation of early-successional habitat) and translocations. The biggest challenge in designing such a program is identifying suitable sites where existing populations can be expanded or new (self-sustaining) populations can be established. A critical component of this process is an assessment of the quality, quantity, and distribution of potential habitats (Griffith et al. 1989). For NEC, this can be a difficult task because preferred habitats are often ephemeral as a consequence of being generated by some disturbance (Barbour and Litvaitis 1993) and influential patch-specific features such as understory density cannot easily be measured over large geographic areas. Thus, efforts to identify potential sites for restoration of NEC need to focus on landscape characteristics instead of patch-specific components of habitat quality. This approach assumes that necessary microhabitat features will be available or can be created within suitable landscapes (Niemuth 2003). Therefore, the goal of this research was to fill this critical information gap. Specifically, the results of a range-wide survey of New England cottontails (Chapter 2) was used in combination with remotely-sensed environmental data and multiple logistic regression to model the relationship between landscape characteristics and the presence of NEC at two broad spatial scales. The resulting models were then used to create spatially explicit maps depicting landscapes with similar characteristics to those currently occupied. A simple set of criteria was then used to identify and rank potential restoration sites where habitat management and translocations

would benefit the long-term survival of NEC.

Methods

Study Area

This study was conducted throughout the known historic range (circa 1960) of the New England cottontail (Hall and Kelson 1959, Godin, 1972 Chapman et al. 1992), an area covering approximately 90,000 km² (Fig. 3-1). It included much of the northeastern United States, from the Hudson River Valley in New York, to the east and north and contained all of Connecticut, Massachusetts and Rhode Island, much of Vermont, southern New Hampshire, and southern Maine (Fig. 3-1). Mean annual snowfall is quite variable, ranging from approximately 3,000 mm in the northwestern portion of the range to 500 mm in the southeast (Lull 1968). Unlike most areas of the United States, forests in this region have increased substantially over the past 100 years (Litvaitis 1993a). In southern areas and along the Atlantic coast, forest cover is dominated by oaks (*Quercus*), white pine (*Pinus strobus*), and hemlock (*Tsuga canadensis*). To the north and west, the most abundant forest types are northern hardwoods (*Acer*, *Betula*, and *Fagus*) at lower elevations and northern hardwoods mixed with spruces (*Picea*) and balsam fir (*Abies balsamea*) at higher elevations. Land use varies considerably throughout the region. In general, coastal areas are characterized by a mix of urban and suburban development, small woodlots, and scattered agricultural fields. Inland landscapes are dominated by large blocks of mid-successional forests, especially in the northern portion of the study area.

Current distribution of New England cottontails

Data on the current distribution of NEC used to develop habitat models came

from a range-wide survey conducted between 2000 and 2004 (Chapter 2). The basic sampling units for that survey were 7.5-minute USGS topographic quadrangles (quads, ~14 x 10 km) that were overlaid on the historic range of NEC. Within each sampled quad, a minimum of 8 patches of suitable habitat were searched for the presence of lagomorph activity (e.g., tracks in snow, clipped twigs, and fecal pellets). Because NEC are sympatric with eastern cottontails in the southern portion of their range and snowshoe hares (*Lepus americanus*) in northern areas, the identity of resident lagomorphs on occupied patches was determined by live-trapping early in the survey (2000-2001 field season) and later replaced by analysis of mitochondrial DNA extracted from fecal samples (2002 -2004 field seasons). A total of 2,301 patches within 287 quads were surveyed for the presence of lagomorphs. Of these, 162 patches in 87 quads were occupied by NEC that represented five populations (Fig. 3-1). Additional details of the survey are provided in Chapter 2.

Habitat variables

Initial attempts to identify habitats for NEC were based on models that incorporated the entire region-wide data set (Chapter 2). These attempts proved unsuccessful because of the variety of forest types, land uses, and climatic conditions encountered throughout the study area. Thus, regional or population-specific models were necessary. However, sample sizes for some populations were too small for multivariate statistical analysis (Tabachnick and Fidell 2001). Therefore, the five core populations were consolidated into three regions based on geographic proximity and similarities in environmental conditions (Fig. 3-1): northeast (populations a and b), southeast (populations c and d), and southwest (population e). For each region, habitat

variables thought to influence the distribution of NEC were inventoried at two spatial scales. At the region scale, environmental features associated with the current range of NEC were compared to features associated with the now vacant portion of the historic range. Potential predictor variables were summarized within occupied USGS quads and an equal number of randomly selected quads that were classified as vacant (Litvaitis et al. 2006a). The pool of vacant quads available for analysis was restricted to those within three quads of currently occupied quads.

Habitat features that influence the distribution of NEC within occupied regions were then examined at the landscape scale. For this analysis, habitat features were inventoried within a 1-km (3.14 km²) radius of NEC locations and compared to an equal number of sites determined to be vacant within the same quad. The 1-km distance was chosen because it is a reasonable approximation of the dispersal distance of NEC (Litvaitis and Villafuerte 1996). For both the region and landscape scales, all variables were summarized within a geographic information system and included information that described current land use and land cover, potential dispersal corridors for NEC, and winter severity (Table 3-1).

Land use and land cover. To examine the role of current land uses on the distribution of NEC, land cover data was obtained from the USGS National Land Cover Characterization Project (NLCD) for all states within the study area (Appendix B for summary of all GIS datasets). This dataset was derived from unsupervised clustering of 1992 Landsat Thematic Mapper satellite imagery (30-meter spatial resolution) using a systematic approach applied consistently throughout the United States (Vogelman et al. 2001). First, spectrally distinct clusters were assigned to one or more land cover classes

using high resolution aerial photographs as reference. Next, clusters that could not be distinguished with aerial photography were resolved using a combination of ancillary data sets (e.g., National Wetlands Inventory) and modeling. Finally, this basic classification was manually edited (e.g., on-screen digitizing) for NLCD classes that could not be separated via the modeling process (Vogelman et al. 2001). The result was a 21-class raster grid data set with an overall classification accuracy of approximately 46% for New England (Stehman et al. 2003). User's and producer's accuracies ranged from 24-97% and 2-92% respectively for individual land cover classes at this level (Stehman et al. 2003). For my analysis, the 21 classes were collapsed into five super-classes based on biological significance to NEC (Appendix C): forest, developed, agriculture, open/disturbed, and water/wetlands. This closely approximated NLCD Level I (7 classes) which has an overall classification accuracy of 80% and substantially improved error rates for individual classes (Stehman et al. 2003). The proportion of each land-use super class was calculated for all quads (occupied and vacant) in the regional analysis and all 1-km radius landscapes (occupied and vacant) for the landscape-scale analysis using the summarize zones function of ArcView 3.3 and Spatial Analyst 2.0 (Theobald 1999). Additionally, total forest edge and four edge types (forest-developed, forest-agriculture, forest-other open and forest-water/wetlands) was inventoried within each 1-km radius area. It is important to indicate that understory density was not part of the modeling efforts. Current remote sensing technology is not able to consistently identify/classify this habitat feature. However, generating dense understory vegetation at sites that are otherwise suitable is certainly feasible (Thompson and DeGraaf 2001, Oehler et al. 2006).

Potential dispersal corridors. Dispersal corridors can partially mitigate the deleterious effects of fragmentation by providing linkages among disjunct patches of suitable habitat (e.g., Bolger et al. 2000). Potential dispersal corridors for NEC include the brushy edges of major roadways, utility rights-of-way, and railroad corridors (Litvaitis et al. 2003). Digital data on roads, railroads, and miscellaneous transportation features (e.g., powerlines and pipelines rights-of-way) were extracted from United States Census Bureau's 2000 Topologically Integrated Geographic and Encoding Referencing system (TIGER; [U.S. Census Bureau 2003; Appendix B]) database for each county within the study area. TIGER line files were created with a variety of encoding techniques including automated map scanning and manual map digitizing (U.S. Census Bureau 2003). The horizontal positional accuracy of these files varies with source material used, but is generally consistent with USGS 1:100,000-scale map standards (U.S. Census Bureau 2003). To process the TIGER files, county tiles were combined into a single layer using the merge command in ArcView 3.3 (Theobald 1999). Duplicate records and other inconsistencies were removed and density (meters/km²) of major roads, local roads, rights-of-way, and railroad corridors were calculated for each quad and 1-km buffer associated with NEC occupied and vacant sites. Major roads were defined as primary (e.g., interstates and toll roads, TIGER A1 and A2 roads) and secondary highways (e.g., state and county highways, TIGER A3 roads). All other types were considered local roads (TIGER A4-A7 roads). In addition, the distance from occupied and vacant sites to all potential dispersal corridors was calculated for the landscape scale analysis.

Winter severity. Duration of snow coverage in the winter can have a profound

effect on the ability of a cottontail to avoid predation because the brown pelage of cottontails makes them more conspicuous in snow (Keith and Bloomer 1992). Brown and Litvaitis (1995) found that a greater proportion of marked New England cottontails were killed by predators during winters with prolonged snow coverage. Because information on the duration of snow coverage was not available for the study area, annual snowfall was used as a surrogate feature. Raster grid (2-km spatial resolution) data of mean annual snowfall was obtained from Spatial Climate Analysis Service at Oregon State University (Daly et al. 2001; Appendix B). These data were created using approximately 7,400 Weather Service Stations across the United States to develop a linear regression model of snowfall based on elevation for the period 1971-2000 (Daly et al. 2001). To control for the effects of other variables on snowfall, stations were assigned different weights based on numerous factors, including proximity to large water bodies and distance from the target grid cell (Daly et al. 2001). Data for each state were combined in ArcView 3.3 using the Spatial Analyst 2.0 extension and the mosaic command (Theobald 1999). All overlapping areas were smoothed by taking a weighted average of the two overlapping grid cells. The mean annual snowfall for each quad (occupied and vacant) and 1-km buffer (occupied and vacant) were then calculated.

Habitat models

Multiple logistic regression was chosen as the appropriate method of analysis because of the binary nature of the response variable (presence/absence of NEC). Models were constructed at the regional and landscape scales following the protocol of Hosmer and Lemeshow (2000). Potential predictor variables (Table 3-1) were initially screened by fitting univariate logistic regression models. The likelihood ratio test was examined

and variables with a P-value less than 0.25 were retained for further analysis. Variables retained from the univariate analysis were then entered into a Pearson correlation matrix to identify redundant variables. One variable among each correlated pair ($r \geq 0.7$) was dropped from further consideration based on biological significance for NEC. The remaining variables were entered into a stepwise logistic regression to determine variable entry or removal in the multivariate models.

Ignoring spatial autocorrelation in species distribution modeling could lead to false conclusions about habitat relationships (Lichstein et al. 2002). Thus, I constructed correlograms of Moran's I to test for the degree of spatial autocorrelation in the Pearson residuals of all logistic regression models (Klute et al. 2002). Because it could not determine *a priori* what neighborhood size was most appropriate for this analysis, multiple lag distances were considered. For region scale models, neighborhoods were defined by the spatial relationship between the target cell (USGS quadrangle) and its surrounding cells. Moran's I was calculated for 1st order (4 nearest neighbors), 2nd order (8 nearest neighbors), 3rd order (12 nearest neighbors) and 3-radius distance (all quads within a 3-radius distance) neighborhoods (Augustin et al. 1996). Lag intervals for landscape scale models were based on the median distance of 1.6 km between sampling points. At this scale, Moran's I was calculated at 10 distance intervals: 0 - 1.6, 1.6 - 3.2, ..., 14.4 - 16 km. For all Moran's I tests, one thousand random permutations of the data were calculated for each lag interval to determine if spatial autocorrelation of the Pearson's residuals significantly different from zero ($P < 0.05$).

The logistic regression models were used as a starting point for fitting autologistic models when significant spatial autocorrelation was detected in the model residuals at

any of the lag distances tested. When autocorrelation was present, an additional model term (called the autocovariate) was developed to account for spatial dependencies in the data. The autocovariate (AUTOCOV) represents the mean weighted probability of observing a New England cottontail at one sample location (USGS quadrangle for region scale; patch location for landscape scale) conditional on the presence of NEC at neighboring sample points within a given lag distance (Augustin et al. 1996, Klute et al. 2002) where:

$$autocov_i = \frac{\sum_{j=1}^{k_i} w_{ij} y_j}{\sum_{j=1}^{k_i} w_{ij}}$$

was the value of the autocovariate for sample location i . If NEC were present at location j then $y_j = 1$, otherwise $y_j = 0$. The weight (W_{ij}) assigned to location j was the inverse Euclidean distance between points i and j (Augustin et al. 1996). The information theoretic approach based on the Akaike Information Criterion (AIC) was used to identify the most parsimonious logistic (spatial autocorrelation not present) or autologistic model (Anderson et al. 2000, Burnham and Anderson 2002) from the group of candidate models.

The best landscape scale model according to AIC was used to predict and map the distribution of habitat potentially suitable for NEC within each region. Maps were generated on a pixel-by-pixel basis by applying a moving window algorithm and the logistic or autologistic regression coefficients to relevant habitat layers. Because the actual distribution of NEC is not known across the entire landscape, I relied on the predicted distribution of NEC rather than observed response values to derive the

autocovariate surface (Ferrier et al. 2002). In an iterative process described in Ferrier et al. (2002), the original logistic regression models (environmental predictors only) were used to predict the distribution of NEC across the landscape. Pixels with a probability >0.50 were considered “occupied” and used to calculate the initial autocovariate surface. The model was refitted, incorporating the autocovariate alongside the other environmental predictors, and then used to derive a new mapped probability surface from which new values for the autocovariate were calculated. This process was repeated iteratively until convergence (Augustin et al. 1998, Ferrier et al. 2002). For the final maps, pixels with a probability >0.5 were considered suitable for NEC.

Identification of potential restoration sites

A relatively straightforward approach was used to identify and rank potential restoration sites within each region of the study area. Although most of the habitats identified as suitable for NEC probably occur on private lands, initial management activities aimed at restoration will likely occur on public and non-profit conservation lands. Digital data on the distribution of conservation lands was obtained for all of the states within the study area and intersected with maps of landscapes that were classified as potentially suitable for NEC (probability of occurrence ≥ 0.5). The overlapping areas that resulted provided a pool of potential restoration sites. Because habitat patch size is an important determinant of NEC survival in fragmented landscapes (Barbour and Litvaitis 1993, Brown and Litvaitis 1995), parcels less than 15 ha were eliminated from the database of potential sites. Litvaitis and Villafuerte (1996) used simulation modeling to demonstrate that the long-term maintenance of NEC metapopulations may require a network of habitat patches at least 15-75 ha in size. The restoration potential of the

remaining parcels was determined using a simple set of criteria (Table 3-2): size of the parcel, proximity of the parcel to existing populations, and mean landscape suitability of the parcel as determined by the best local scale logistic regression model. For each of these variables, a numeric code was assigned based on its value towards sustaining populations of NEC (1 = low, 2 = medium, and 3 = high value). The restoration potential of each parcel was then totaled using the sum of all variables inventoried, resulting in scores ranging from 3 to 9. Parcels with a score of 7 to 9 were considered high value, 5 to 6 moderate value and 3 to 4 low value. This process-oriented approach is similar to the mechanistic approach used in creating habitat suitability models (e.g., Donovan et al. 1987). Although individual suitability variables may have a greater or less influence on cottontail fitness, our understanding of these variables and their interactions was insufficient to weigh them differently.

Results

At both spatial scales studied, NEC were associated with anthropogenic features in the northeastern and southeastern portions of their historic range and agricultural landscapes in the southwest (Table 3-3 and 3-4). After accounting for spatial autocorrelation at the landscape scale, approximately 740,000 ha of suitable habitat were identified within the three regions. This included nearly 90,000 ha on conservation or other public lands. A total of 1487 individual sites were identified as potential management targets with 155 (10.4%) ranked as having high value for restoration or translocation (Appendix C for locations). It is important to note that these habitats represent potential habitats based on the environmental features associated with existing

populations of NEC and that these habitats may or may not include suitable microhabitat features (including understory density).

Northeast region

Univariate logistic regression models identified 8 region scale variables as potentially important predictors of the presence/absence of NEC (Table 3-3). Pearson residuals of the initial multiple logistic regression model exhibited significant spatial autocorrelation across all neighborhoods tested (Fig 3-2). Thus, the autocovariate term (AUTOCOV) was incorporated into all subsequent models. The autologistic model that included the proportion of forested lands (PERFOR), mean annual snowfall (SNOW) and AUTOCOV (2nd order neighborhood) has the lowest AIC and was the best model according to the information theoretic approach (LL = -106.8, $P < 0.001$, $Rho^2 = 0.45$; Table 3-5). PERFOR and SNOW had negative parameter estimates suggesting conditions unfavorable to NEC occupation. The model correctly classified 87% of occupied and vacant quads within the northeast region.

At the landscape scale, 14 habitat variables were identified by the univariate logistic regression models as potentially important predictors of the presence/absence of NEC (Table 3-4). Significant spatial autocorrelation was detected in the Pearson residuals of the initial multiple logistic regression model at the 6 smallest lag-distance classes (Fig. 3-2). The model that incorporated the distance to potential dispersal corridors (PDCDIS), the proportion of forest land use (PERFOR), the proportion of open and disturbed land use (PEROD) and AUTOCOV (0 – 6.4 km lag) had the lowest AIC and was the best model according to the information theoretic approach (LL = -110.9, $P < 0.001$, $Rho^2 = 0.49$; Table 3-6). PEROD had a significant positive parameter estimate,

suggesting favorable contributions to the presence of NEC. All other parameter estimates were negative. The model correctly classified 78% of occupied and vacant patches in the northeast region. Approximately 200,000 hectares of habitat potentially suitable for NEC (probability of occurrence > 0.5) was identified, covering 10.6% of the northeastern portion of the study area (Fig. 3-3). Almost 30,000 hectares or 15% of suitable habitat occurs on public or non-profit conservation lands. A total of 326 potential restoration sites were identified. Of these, only 19 (5.8%) were considered highly ranked for possible restoration or translocation (Fig. 3-3).

Southeast region

Like the northeast region, univariate logistic regression models identified 8 region scale variables as potentially important predictors of the presence/absence of NEC (Table 3-3). Pearson residuals of the initial multiple logistic regression model exhibited significant spatial autocorrelation for three of the four neighborhoods tested (Fig. 3-2). The autologistic model that included the density of potential dispersal corridors (PDCTOT), the proportion of forested lands (PERFOR), mean annual snowfall (SNOW) and AUTOCOV (3rd order neighborhood) has the lowest AIC and was the best model according to the information theoretic approach (LL = -34.6, $P < 0.001$, $Rho^2 = 0.31$; Table 3-5). PDCTOT had a positive parameter estimate indicating favorable conditions for NEC occupation. PERFOR and SNOW had negative parameter estimates. The model correctly classified 83% of occupied and vacant quads in the southeast region.

At the landscape scale, 13 habitat variables were identified by the univariate logistic regression models as potentially important predictors of the presence/absence of NEC (Table 3-4). Significant spatial autocorrelation was detected in the Pearson

residuals of the initial multiple logistic regression model at the 5 lag distance classes (Fig. 3-2). The model that incorporated the density potential dispersal corridors (PDCTOT), the proportion of forest land use (PERFOR), the proportion of open and AUTOCOV (0 – 8.0 km lag) had the lowest AIC and was the best model according to the information theoretic approach (LL = -48.5, $P < 0.001$, $Rho^2 = 0.39$; Table 3-6). PDCTOT had a significant positive parameter estimate whereas PERFOR was negative. The model correctly classified 71% of occupied and vacant patches within the southeast region. Nearly 250,000 hectares of habitat potentially suitable for NEC (probability of occurrence > 0.5) was identified, covering 14.6% of the southeastern portion of the study area (Fig. 3-4). Approximately 38,000 hectares or 15.2% of suitable habitat occurs on public or non-profit conservation lands. A total of 647 potential restoration sites were identified. Of these, 70 (10.8%) were considered highly ranked for possible restoration or translocation (Fig. 3-4).

Southwest region

Univariate logistic regression models identified 5 region scale variables as potentially important predictors of the presence/absence of NEC (Table 3-3). Pearson residuals of the initial multiple logistic regression model exhibited significant spatial autocorrelation across all neighborhoods tested (Fig. 3-2). The autologistic model that included the proportion of forested lands (PERFOR), the proportion of agricultural land use (PERAG) and AUTOCOV (2nd order neighborhood) has the lowest AIC and was the best model according to the information theoretic approach (LL = -19.4, $P < 0.01$, $Rho^2 = 0.26$; Table 3-5). Unlike the northeast and southeast region, PERFOR had a positive parameter estimate indicating favorable conditions for NEC in this region. PERAG was

also positive. The region scale model correctly classified 76% occupied and vacant quads within the southwest region.

At the landscape scale, 8 habitat variables were identified by the univariate logistic regression models as potentially important predictors of the presence/absence of NEC (Table 3-4). Significant spatial autocorrelation was detected in the Pearson residuals of the initial multiple logistic regression model at one lag distance class (Fig. 3-2; 0 – 9.6 km). The model that incorporated the distance to potential dispersal corridors (PDCDIS), the proportion of forest land use (PERFOR), and the proportion of agricultural lands (PERAG) had the lowest AIC and was the best model according to the information theoretic approach ($LL = -52.7$, $P < 0.001$, $Rho^2 = 0.32$; Table 3-6). The parameter estimate for PDCDIS was negative, while PERFOR and PERAG were positive. The model correctly classified 67% of occupied and vacant patches in the southwest region. Nearly 290,000 hectares of habitat potentially suitable for NEC (probability of occurrence > 0.5) was identified, covering 20.3% of the southwestern portion of the study area (Fig. 3-5). Only 21,000 hectares or 7.4% of suitable habitat occurs on public or non-profit conservation lands. A total of 514 potential restoration sites were identified. Of these, 66 (12.8%) were considered highly ranked for possible restoration or translocation (Fig. 3-5).

Discussion

Wildlife-habitat models created with remotely sensed data, multivariate statistics and GIS have been used to direct conservation efforts for a variety of taxa (Ruston et al. 2004). In recent years, these types of models have been shown to be an effective tool for understanding species-habitat relationships at multiple spatial scales (e.g., Carroll et al.

1999), for mapping the distribution and abundance of potential habitat (e.g., Dettmers and Bart 1999) and for identifying potential restoration sites for declining species (e.g., Niemuth 2003). Despite the popularity and widespread use of these methods, many researchers fail to acknowledge the limitations of their models (Rae 2002) where interpretation of results is dependent on the statistical method chosen, the quality of the species and habitat information and the spatial scale of analysis (Weir and Harestad 2003).

Limitations of the habitat models

The use of logistic regression to model potential habitat for wildlife assumes that presence/absence has been determined without error (Keating and Cherry 2004). The most common concern with the use of presence/absence data for habitat modeling is the potential for false absences (Williams 2003). False absences in wildlife surveys can occur when a species is present but not detected by the observer. Many factors can influence the detectability of a species including sampling design, environmental conditions, species-specific characteristics (e.g., appearance or behavior), habitat type and population density of the species (Bayley and Peterson 2001, Tyre et al. 2001). Gu and Swihart (2004) cautioned that habitat models resulting from field surveys where specific features influence detection could result in models that incorrectly identify suitable habitats. For NEC, detectability within a patch of habitat is likely influenced by the survey method employed (livetraps versus fecal DNA), prevailing weather conditions (e.g., searching for fecal pellets on snow versus searching on leaf litter), the presence of eastern cottontails or snowshoe hares. As a result, some occupied patches could have been misclassified as “vacant” in the region-wide survey (Chapter 2) and subsequently used in the modeling

process. However, more than 8 patches were used to classify most quads, so misclassification at the region scale were not likely (Chapter 2). Classification accuracy at the landscape scale (1-km radius) was more problematic because the detection rate for the range-wide survey was never determined (Chapter 2). Concern over the misclassification of patches because of the presence of a sympatric lagomorph was tempered by results from intensively studied populations in Maine (Litvaitis et al. 2003) and New Hampshire (Litvaitis and Villafuerte 1996). In these areas, the majority of patches surveyed were <3 ha and therefore not likely to be occupied by more than one species of lagomorph (Litvaitis et al. 2003). Site visits to sampled patches in eastern Massachusetts (Population #c, Fig. 3-1) and many of those that comprised the largest population in western Massachusetts, southeastern New York, and western Connecticut (Population #e, Fig. 3-1) revealed that the majority of occupied patches were also <3 ha (J. Litvaitis and J. Tash, personal observations) and probably supported no more than 3-4 rabbits. Under these circumstances, accepting the identity of the first live capture or fecal pellet analyzed seemed a reasonable compromise given the large geographic area that was sampled (Chapter 2).

With the relative ease of use and widespread availability of GIS data, the variety of factors related to the use of spatial data that can influence the ability of a model to accurately predict the probability of occurrence of a wildlife species can be easily overlooked. Several sources of uncertainty have been identified when dealing with GIS-derived habitat variables including the loss of information from spatial generalization, positional errors and misclassification (Stoms et al. 1992). Errors in GIS data tend to propagate throughout the modeling process causing composite map accuracies to decline

exponentially with increasing number of data layers (Veregin 1989). Consequently, species-specific habitat maps derived from multiple GIS data sets may have unacceptable error rates (both positional and classification errors) even though individual layers are within acceptable error limits as defined by the user. Because of the broad geographic extent of this research, I relied on readily available GIS layers rather than creating custom data sets (e.g., derived from photo interpretation) to derive habitat variables. Thus, I was unable to control for errors and uncertainty in model input data. Additionally, model outputs have not been thoroughly field checked. As a result, maps derived from these models need to be interpreted with care and are not recommended for site specific management applications. Instead, they should guide landscape and region level conservation planning for NEC.

The choice of spatial scale in constructing species-habitat models can determine the patterns of association that are detected between habitat variables and species presence/absence (Wiens 1989). In this study, habitat variables used in model development were inventoried at two broad spatial scales (region [USGS quadrangle] and local [1-km radius]) and do not account for important fine-grained elements of habitat suitability measured at patch-specific scales. Despite this obvious limitation, my approach is justified from a practical and management perspective. The most important patch scale feature for NEC, understory stem density, cannot easily be obtained from satellite imagery or aerial photographs. Thus, it would be impossible to map vegetation density throughout the historic range of NEC and include it in model development. Further, it is easier for managers to create fine-grained habitat elements within suitable landscapes than to alter a landscape surrounding a patch with fine-grained habitat but

unsuitable landscape characteristics. The greatest value of these models is in providing wildlife managers with a potential starting point in identifying areas where management activities could be directed towards creating suitable patches of dense understory vegetation that would support NEC.

Perhaps the biggest limitation of these models is that they may not identify habitats that are ideal for NEC but describe areas that are similar to habitats that are currently occupied. When locations with particular habitat characteristics are associated with the occurrence of a species, it is often assumed that the locations are quality habitat. The probability of occupancy is thus used as a surrogate for habitat quality, when in fact the occupied habitat may not be “good” habitat (Tyre et al. 2001, VanHorne 1983). This may be especially true for a species like the NEC whose populations are often spatially structured as induced metapopulations (Litvaitis and Villafuerte 1996). In landscapes where species exhibit source-sink population dynamics, species occur both in good or source habitat (where birth rates exceed death rates) and in sink habitat (where death rates exceed birth rates). During the modeling process, no effort was made to determine the habitat quality of occupied patches and all were assumed to be good habitat.

Factors influencing the distribution of New England cottontails

The modeling efforts revealed important regional differences in habitats occupied by remnant populations of NEC. As a result, no single factor appears to explain the current distribution throughout the species historic range. In the northeast and southeast regions, NEC continues to persist in heavily developed, human dominated landscapes (Table 3-3). Most likely, this relationship is driven by the pattern of early-successional habitat loss through forest maturation that has occurred throughout the region (Brooks

2002). Previously, Litvaitis (1993) argued that the north-south contraction of NEC in New Hampshire could be explained by a more drastic loss of suitable early-successional habitats in northern counties compared to southern counties as abandoned agriculture matured into unsuitable second growth forests. Concurrent with this loss of habitat, the range occupied by NEC in New Hampshire had declined from a near continuous distribution throughout 60% of the state in 1950 to disjunct populations that span <20% of the state (Litvaitis 1993a). This pattern of habitat loss through forest succession is supported by the regional and landscape scale models for northeastern and southeastern populations of NEC. Here, remnant populations are negatively associated with percent forest cover at both spatial scales studied (PERFOR; Table 3-5 and 3-6). Large blocks of mid-successional forest lack the understory necessary to support populations of NEC and may be a significant barrier to movement between suitable patches of early-successional habitat.

In contrast to populations in the northeast and southeast, NEC occupying the southwest region are associated with rural landscapes characterized by the positive relationship between percent forest cover (PERFOR; Table 3-6) and the presence of NEC. Here, regional populations are likely sustained by the relative abundance of agricultural lands (PERAG) compared to other regions of the study area (Table 3-3). At both spatial scales studied, the percentage of the landscape composed of agricultural land-uses was a significant positive predictor of the presence of NEC (Table 3-6). The idle edges of agricultural fields, especially pastures (J. Tash, personal observation), may function as both habitat for resident cottontails and as dispersal corridors. Additionally, NEC in the southwest region were often observed in young forests associated with an

abundance of recently abandoned agricultural lands.

Within all three regions of the study area, the abundance of potential dispersal corridors (PDCTOT, Table 3-6) or the distance to dispersal corridors (PDCDIS, Table 3-6) was a significant predictor of the presence of NEC at the landscape scale. The brushy edges of these linear features may provide important linkages among disjunct patches of early-successional habitat (Bolger et al. 2000). This may be especially important for maintaining local populations of NEC given the transitory nature of required habitats (Tiebout and Anderson 1997). Species that are narrowly adapted for a given seral stage, like NEC, must disperse from resident patches of habitat when successional maturation renders them unsuitable. Therefore, population viability depends upon successful colonization of new sites. Some species are highly mobile and can readily cross expanses of unsuitable habitat to reach new sites, while others possess constraints in mobility that significantly limits dispersal opportunities. NEC certainly represents the latter and thus requires connectivity among suitable habitats in order to persist (Tiebout and Anderson 1997).

Management recommendations

This research has identified habitats within the regions surveyed that are suitable for NEC and has specifically located those lands that were ranked high for restoration or translocation throughout the species' historic range. Based on these results, efforts to enhance populations of NEC should consider habitat management and possible translocation sites in the context of regional land-uses and local landscape features. Initially, efforts should be directed towards expanding existing populations. At these sites, local habitat manipulations should be guided by land use patterns within the region

(Litvaitis 2001, 2003). Next, habitat restoration efforts should be directed at the vacant areas that separate the five identified populations (Fig. 3-1). Suitable habitats in these areas should be surveyed and understory density modified if needed (Litvaitis 2003). These vacant habitats could serve as stepping stones to connect populations (Chapter 2) and as sites for possible translocations. Finally, new populations of NEC should be established in regions that are no longer occupied (e.g., southern Vermont and southwestern New Hampshire; Fig. 3-1).

Northeast and Southeast Regions. Efforts to create or maintain early-successional habitats in human-dominated landscapes (e.g., northeast and southeast regions) are confronted with a variety of limitations, such as dense road networks and suburban developments that limit dispersal of NEC. Cottontails in these landscapes are dependent on the balance between extinction and recolonization (Litvaitis et al. 2003). Small populations are able to persist only because surplus rabbits from one or more larger populations regularly disperse to small patches of habitat. Therefore, it may be most effective to establish and maintain large patches of habitat that can serve as “core” habitats. Research by Litvaitis and Villafuerte (1996) demonstrates that the long-term maintenance of NEC metapopulations may require a network of habitat patches at least 15-75 ha in size. Large tracts would be less susceptible to the limitations of the surrounding landscape matrix and allow population of NEC to withstand short-term perturbations. Unfortunately, identifying high quality restoration sites on public lands will be difficult, especially in the northeast region where only 12.5% of suitable habitat identified during model development occurs on conservation lands or easements. To address this limitation, it may be possible for wildlife managers to establish core

populations on degraded sites that already have an abundance of early-successional habitats, such as abandoned gravel and sand mines. The distribution of NEC in the northeast and southeast regions was clearly influenced by the abundance of open and disturbed habitats (PEROD; Table 3-3). These tracts would require only modest initial efforts and periodic maintenance. Clustering core habitats and placing them adjacent to existing land-uses that include early successional habitats (e.g., utility corridors) might facilitate exchanges among neighboring populations.

Most of the sites in these regions identified as suitable for potential restoration occur in human dominated landscapes. Although these areas are similar to those currently occupied by NEC, they might not represent a viable long-term solution. In fragmented landscapes, generalist predators are often more abundant due to increased foraging efficiency associated with human land uses such as agriculture and suburban developments (Litvaitis and Villafuerte 1996). NEC may be especially vulnerable to increased predation in altered landscapes because they are often restricted to small patches of habitat (Litvatis et al. 2003) where predator densities are high (Andren and Angelstram 1988). NEC occupied patches in southeastern New Hampshire ranged from 0.2 to greater than 15 ha, but very small patches (less than or equal to 2 ha) were inherently vulnerable because of intense predation (Barbour and Litvaitis 1993, Villafuerte et al. 1997). Not surprisingly, predation was found to be the most common mortality factor among NEC populations in New Hampshire, especially by coyotes (*Canis latrans*) and foxes (*Vulpes vulpes*) (Barbour and Litvaitis 1993, Brown and Litvaitis 1995, Villafuerte et al. 1997). Therefore, long-term efforts aimed at managing for NEC should consider options outside of highly developed landscapes.

Southwestern Region. In the southwest region, NEC are associated with forested landscapes. As such, habitat management programs that resemble natural disturbance regimes may be more appropriate (Seymour et al. 2002). Where feasible, restoration of native shrublands should be a management priority. In regions where the historic abundance of shrublands and barrens was limited, timber harvests will provide a practical approach to diversify stand age distributions. Here, a sliding scale approach would be appropriate in mid-successional forests where natural disturbances are rare (Litvaitis 2003). Initially, the size of the timber harvests would be larger than natural disturbances to offset the shortfall in early-successional habitat that currently exists. Once established, some of these openings could be maintained by active management (e.g., cutting, mowing, or controlled fires). As forests mature, management efforts could then be patterned after canopy gaps (Runkle 1991) or modified to specific silvicultural practices of a region if other forms of early-successional habitats (e.g., native shrublands and beaver impoundments) are not adequately represented.

Additionally, populations of NEC were associated with the abundance of agricultural lands in the southwest region. To manage for NEC on these lands, resource managers need to establish partnerships with private landowners. Oehler (2003, 2006) summarized the government funding programs that are available to assist private landowners interested in managing early-successional habitats in the northeastern United States.

Other Management Considerations. The success of habitat models is ultimately related to the existence of strong and predictable associations between species and habitat variables (Cardillo et al. 1999). In reality however, habitat is not the only factor that

determines the distribution of species. Interspecific interactions, such as predation and competition, may have a significant effect on the distribution and abundance of certain species (Morin 1981). The distribution of present-day populations of NEC is substantially affected by heterospecific interactions that were not included in the models. Previous investigators have suggested that competition with expanding populations of eastern cottontails may reduce the suitability of habitats where this species is sympatric with NEC (Fay and Chandler 1955, Reynolds 1975). Both species utilize similar habitats and foods (Dalke and Sime 1941, Linkkila 1971, Johnston 1972) and the decline of NEC populations roughly coincided with the expansion of eastern cottontail populations into large portions of the range of NEC (Probert and Litvaitis 1996). If eastern cottontails are able to colonize disturbance-generated patches sooner than NEC, they may be able to maintain access to these habitats simply on a system of "prior rights" (Probert and Litvaitis 1996). Eastern cottontails have behavioral (Smith and Litvaitis 2000) and morphological traits (Smith and Litvaitis 1999) that enable the species to occupy habitats where NEC would be vulnerable to intense predation. Thus, initial translocations of NEC should occur in habitats devoid of eastern cottontails.

A preliminary investigation of genetic variation based on microsatellites of nuclear DNA from 82 individuals sampled across the current range of NEC supported the separation of distinct populations (Kovach et al., unpublished data). Notably, cottontails collected in eastern Massachusetts diverged from all other subpopulations. It is unclear whether this is the result of isolation of from other populations of NEC or a response to local environmental conditions. Additional investigation is warranted, including a comparison with historically collected specimens to determine if current geographic

structure of genetic variation is a recent consequence of population decline and isolation. Such information may be especially relevant if translocations are used to establish additional populations in vacant habitats. If genetic variation is a response to regional or local environmental differences, translocation efforts may be best restricted to moving animals within one region.

Table 3-1. Habitat and climatic features thought to influence the distribution of New England cottontails.

Variable	Definition	Scale
MAJRDS	Density of class 1 and class 2 roads in m/km ²	Region and landscape
LOCRDS	Density of local roads in m/km ²	local
ROW	Density of rights-of-way (railroad and power line) in m/km ²	Region and landscape
PDCTOT	Density of all potential dispersal corridors (ROW and MAJRDS) in m/km ²	Region and landscape
PDCDIST	Distance to potential dispersal corridors (ROW and MAJRDS) in m	Local
PERFOR	Proportion of landscape consisting of forest cover	Region and landscape
PERDEV	Proportion of landscape consisting of developed lands	Region and landscape
PERAG	Proportion of landscape consisting of agricultural lands	Region and landscape
PEROD	Proportion of landscape consisting of other open lands and disturbed lands	Region and landscape
PERWET	Proportion of landscape consisting of open water or wetlands	Region and landscape
FORDEV	Total forest-developed land edge in m	Local
FORAG	Total forest-agricultural land edge in m	Local
FOROPEN	Total forest-open land edge in m	Local
FORWET	Total forest-water/wetland edge in m	Local
FOREEDGE	Total forest edge in m	Local
SNOW	Mean annual snowfall 1960-2000 in mm	Region and landscape

Table 3-2. Summary of variables used to rank potential restoration sites for New England cottontails on existing conservation lands.

Variable	Low	Mid	High
Habitat quality (probability of occurrence)	0.50 – 0.65	0.65 – 0.85	0.85 – 1.00
Size of parcel (area in hectares)	15 – 45	45 – 75	> 75
Proximity to existing populations (distance in kilometers)	> 5	2.5 – 5	< 2.5

Table 3-3. Mean differences in habitat characteristics of USGS quadrangles occupied by New England cottontails and quadrangles classified as vacant (Region scale).

Variable	Northeast (n = 30)			Southeast (n = 24)			Southwest (n = 33)		
	NEC	Vacant	p	NEC	Vacant	p	NEC	Vacant	p
MAJRDS	759	465	<0.01	737	466	<0.01	482	536	0.74
ROW	179	120	0.01	146	107	0.07	78	161	<0.001
PDCTOT	938	585	<0.01	883	566	<0.01	560	697	0.35
PERFOR	67.9	84.0	<0.001	66.1	74.5	0.03	78.9	61.0	<0.01
PERDEV	17.2	3.9	<0.001	18.6	10.9	<0.01	6.2	23.8	<0.001
PERAG	7.9	8.7	0.76	3.6	7.1	0.12	12.5	9.8	0.06
PEROD	3.5	1.0	<0.001	7.1	3.6	<0.01	1.5	4.3	<0.001
PERWET	3.5	2.4	0.02	4.5	3.7	0.26	0.9	1.3	0.27
SNOW	1616	1960	<0.001	746	1182	<0.001	1303	1290	0.98

Table 3-4. Differences in habitat characteristics associated with 1-km radius landscapes surrounding patches occupied by New England cottontails and patches classified as vacant (Landscape scale)

Variable	Northeast (n = 81)			Southeast (n = 35)			Southwest (n = 38)		
	NEC	Vacant	p	NEC	Vacant	p	NEC	Vacant	p
MAJRDS	952	577	<0.01	483	404	0.51	515	379	0.01
LOCRDS	2180	1892	0.18	2601	1802	<0.01	1950	1838	0.73
ROW	232	113	0.05	273	0	<0.001	64	9	0.06
PDCTOT	1184	691	<0.01	756	404	<0.01	579	388	0.16
PDCDIST	364	777	<0.001	542	881	0.06	698	1600	<0.001
PERFOR	52.2	74.5	<0.001	55.9	70.6	<0.001	75.0	80.0	0.02
PERDEV	20.2	8.9	<0.001	18.7	13.2	<0.001	5.7	7.2	0.31
PERAG	10.7	11.3	0.82	5.9	7.7	0.34	14.2	7.8	<0.001
PEROD	13.6	4.6	<0.001	12.7	6.3	<0.001	2.9	2.6	0.40
PERWET	3.3	0.7	<0.01	6.8	2.4	<0.01	2.2	2.1	0.39
FORDEV	8892	5359	<0.001	8739	4885	<0.01	3398	3272	0.92
FORAG	4541	6234	0.05	2938	4671	0.07	7599	3897	<0.001
FOROPEN	4165	1517	<0.001	4149	1949	<0.001	1016	857	0.56
FORWET	3630	1774	<0.001	3778	2810	0.14	1723	1646	<0.83
FOREEDGE	21229	14854	<0.001	19605	14315	<0.01	13736	9672	<0.01
SNOW	1608	1645	0.59	738	755	0.77	1333	1300	0.67

Table 3-5. Logistic regression models that best discriminated USGS quadrangles occupied by NEC from USGS quadrangles classified as vacant (Region scale).

Variable	Northeast (n = 30)			Southeast (n = 24)			Southwest (n = 33)		
	Parameter	Odds Ratio	p	Parameter	Odds Ratio	p	Parameter	Odds Ratio	p
CONSTANT	16.39118		<0.001	3.68336		0.05	-7.24395		<0.01
PDCTOT				0.00217	1.00217	0.01			
PERFOR	-0.05346	0.94795	0.01	-0.00997	0.98763	0.01	0.06062	1.06250	0.01
PERAG							0.20244	1.22438	<0.001
SNOW	-0.00773	0.99149	<0.001	-0.00813	0.99339	<0.001			
AUTOCOV	2.77002	1.20998	<0.01	6.99544	1.12335	0.05	1.99658	1.19658	<0.001

Table 3-6. Logistic regression models that best discriminated landscapes surrounding patches occupied by NEC from landscapes surrounding patches classified as vacant (Landscape scale).

Variable	Northeast (n = 81)			Southeast (n = 35)			Southwest (n = 38)		
	Parameter	Odds Ratio	p	Parameter	Odds Ratio	p	Parameter	Odds Ratio	p
CONSTANT	5.84182		<0.001	2.77113		<0.01	-1.15833		<0.01
PDCTOT				0.00077	1.00084	0.05			
PDCDIST	-0.00032	0.99934	0.03				-0.00149	0.99851	0.01
PERFOR	-0.10974	0.93722	<0.001	-0.06133	0.94051	<0.01			
PERAG							0.11093	1.11731	<0.001
PEROD	0.07291	1.15935	<0.01						
AUTOCOV	4.23590	1.23587	<0.001	3.93631	1.17543	<0.01			

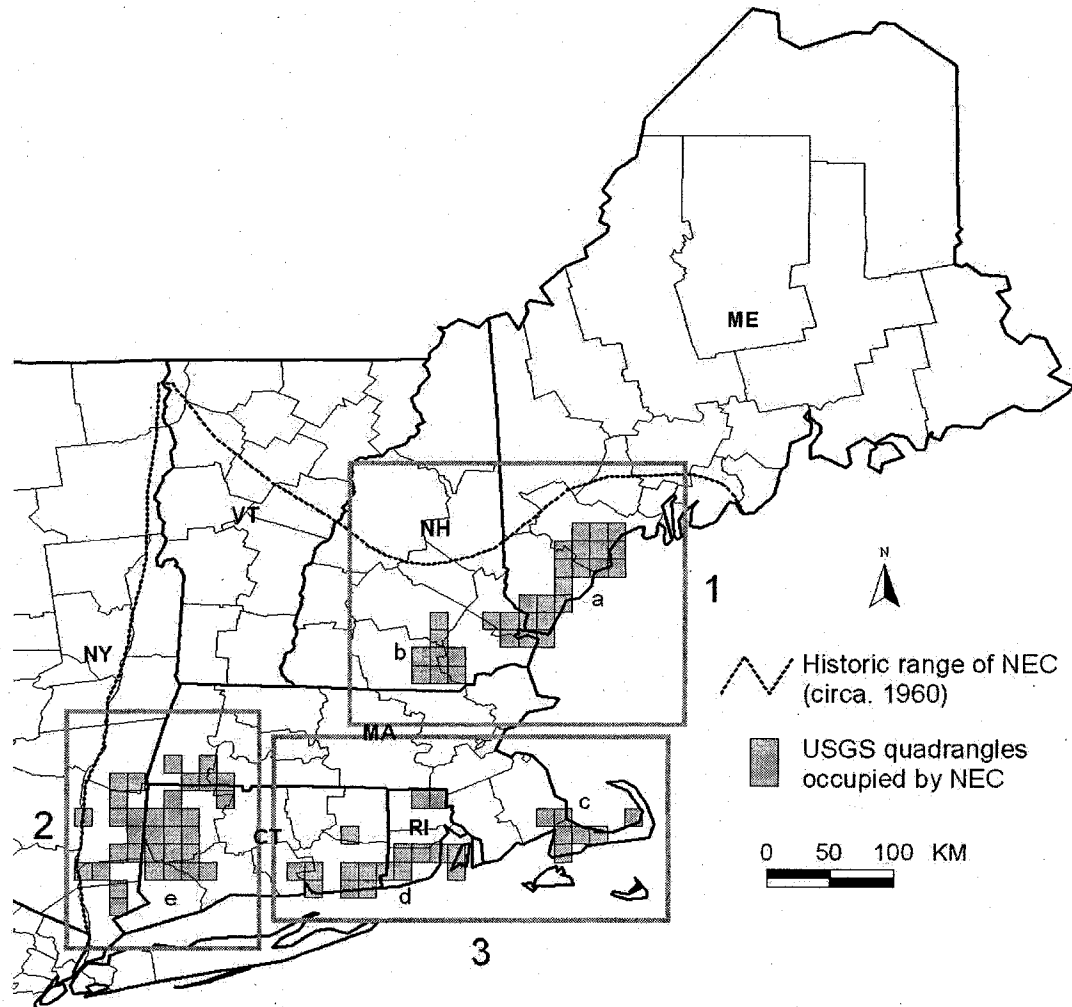
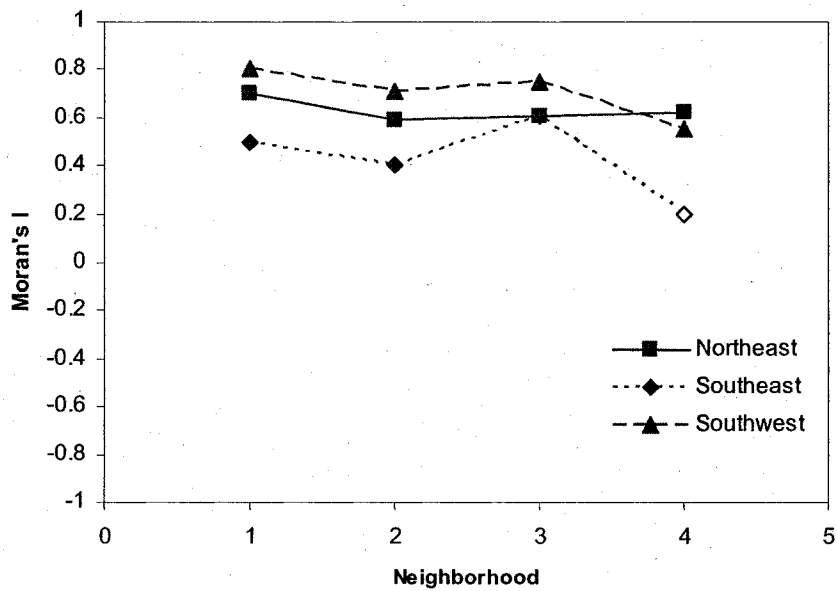
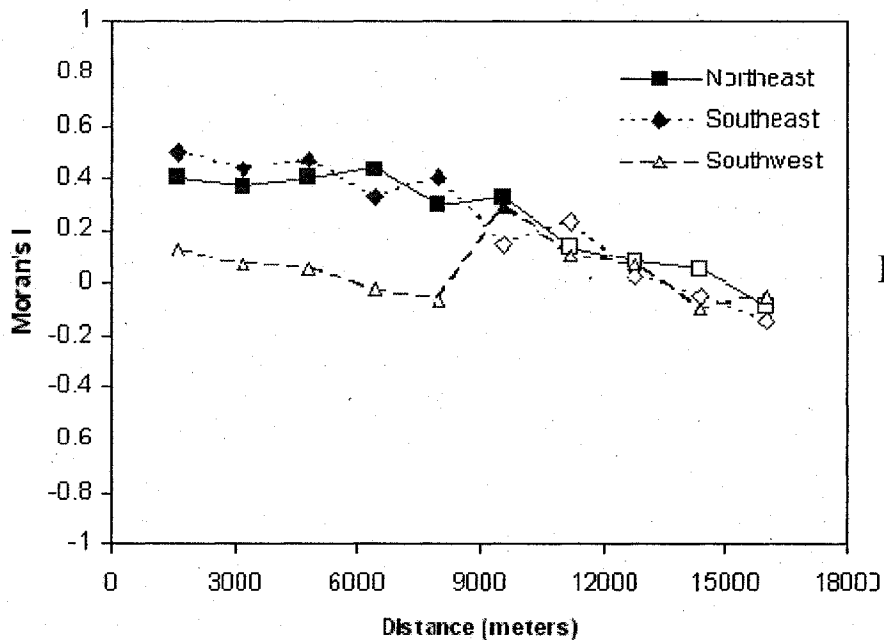


Figure 3-1. Historic and current distribution of New England cottontails. Remnant populations are apparently restricted to five core populations: (a) seacoast region of southern Maine and New Hampshire, (b) Merrimack River Valley of New Hampshire, (c) Cape Cod, Massachusetts, (d) Connecticut east of the Connecticut River and Rhode Island, and (e) western Connecticut, eastern New York and southwestern Massachusetts. For modeling purposes, the five core populations were consolidated into three regions of the study area based on the current distribution of New England cottontails: (1) northeast, (2) southwest, and (3) southeast regions.



A



B

Figure 3-2. Moran's I correlograms of logistic regression model residuals where I has an expected value near zero for no spatial autocorrelation, with negative and positive values indicating negative and positive autocorrelation, respectively. Each point represents the value of I calculated for all neighborhood sizes with neighborhoods defined by nearest neighbor relationships for the region scale models (A) and regular lag distances (1.6 km) for the landscape scale models (B). Closed symbols indicate values of I that are significantly different from 0 ($R < 0.05$).

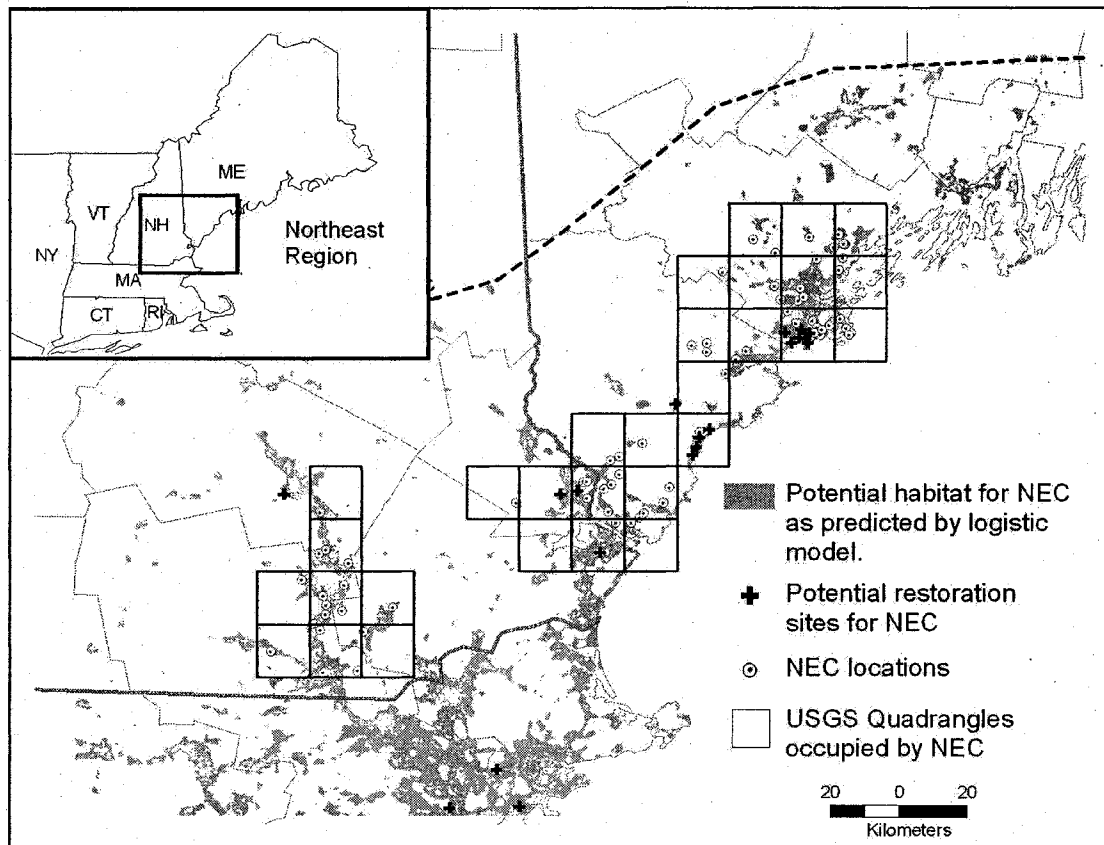


Figure 3-3. Distribution of all potential habitats for New England cottontails in the northeast region as predicted by the best fit landscape-scale logistic regression model. Among these, high ranked parcels (based on quality, size, and proximity to known populations of New England cottontails) on existing conservation lands are identified.

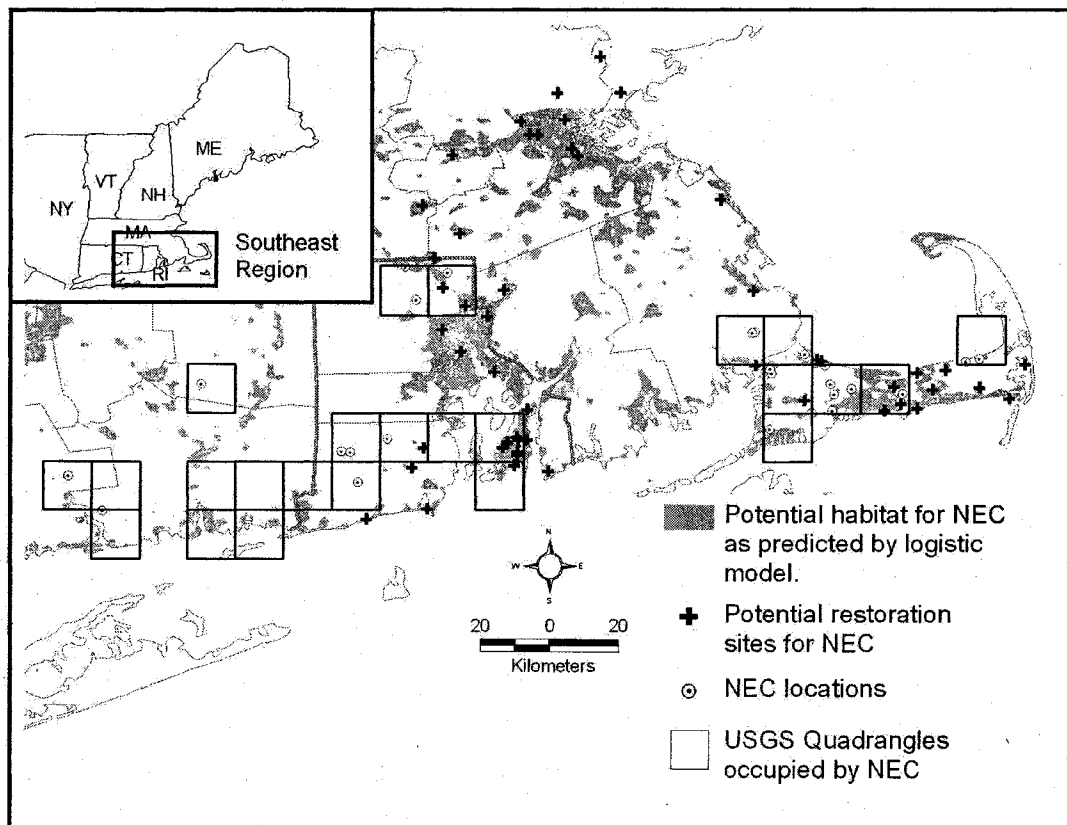


Figure 3-4. Distribution of all potential habitats for New England cottontails in the southeast region as predicted by the best fit landscape-scale logistic regression model. Among these, high ranked parcels (based on quality, size, and proximity to known populations of New England cottontails) on existing conservation lands are identified.

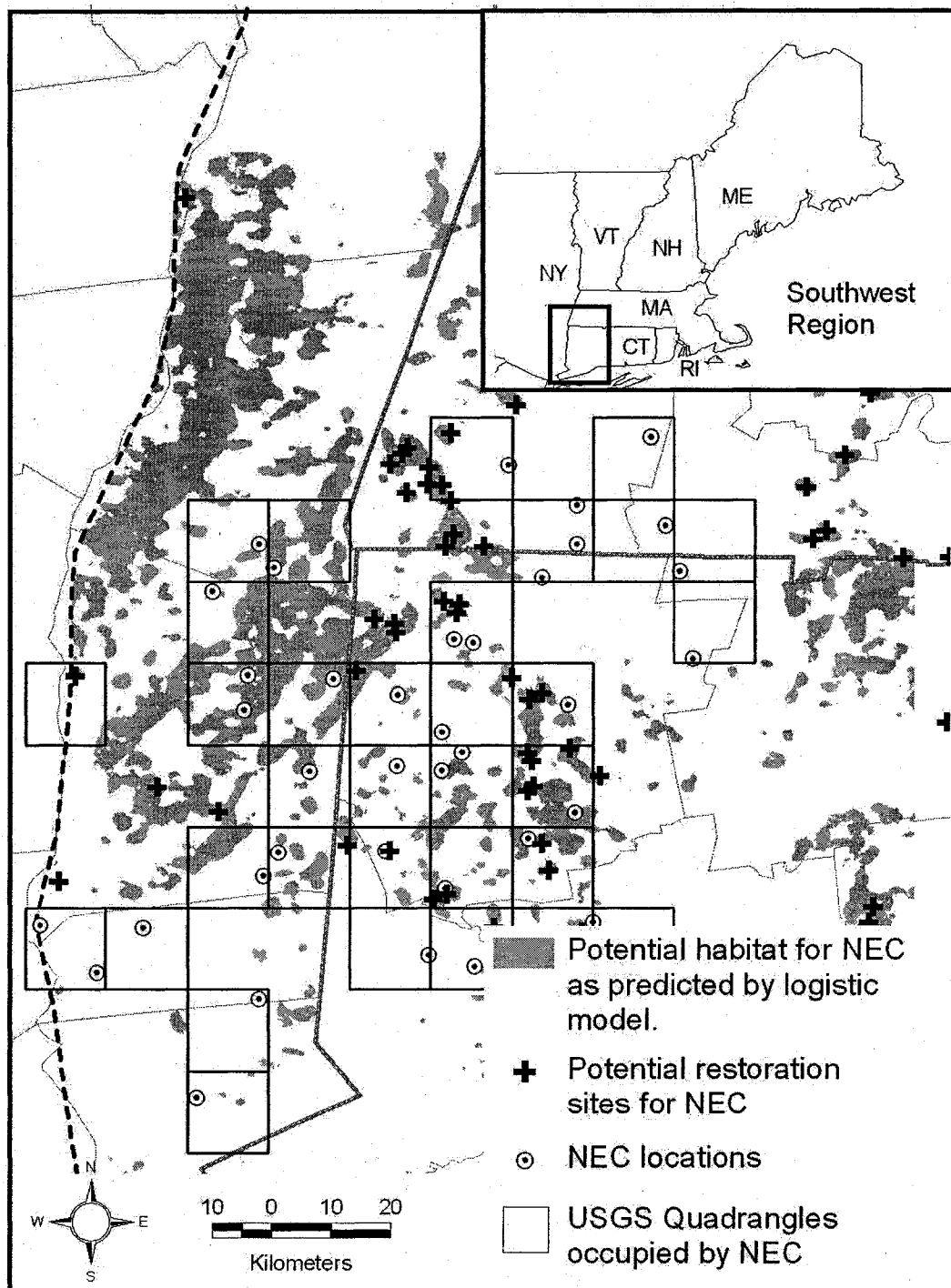


Figure 3-5. Distribution of all potential habitats for New England cottontails in the southwest region as predicted by the best fit landscape-scale logistic regression model. Among these, high ranked parcels (based on quality, size, and proximity to known populations of New England cottontails) on existing conservation lands are identified.

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APPENDICES

APPENDIX A – Sites occupied by New England cottontails as determined by the region-wide survey. Coordinates are in decimal degrees.

State	County	Municipality	Quad Name	Latitude	Longitude
CT	Fairfield	Brookfield	Danbury	41.4266	73.3767
CT	Fairfield	Newbury	Southbury	41.4095	73.2325
CT	Fairfield	Newtown	Newtown	41.4090	73.3066
CT	Hartford	Hartland	West Granville	42.0151	72.9893
CT	Litchfield	Bridgewater	Roxbury	41.5302	73.3512
CT	Litchfield	Cornwall	South Canaan	41.9044	73.3081
CT	Litchfield	Cornwall	South Canaan	41.9095	73.3376
CT	Litchfield	Kent	Kent	41.7166	73.4269
CT	Litchfield	New Hartford	New Hartford	41.8812	72.9693
CT	Litchfield	New Milford	New Milford	41.5855	73.4435
CT	Litchfield	Norfolk	South Sandisfield	42.0053	73.2027
CT	Litchfield	Sharon	Ellsworth	41.8261	73.4254
CT	Litchfield	Torrington	West Torrington	41.8108	73.1629
CT	Litchfield	Warren	New Preston	41.7384	73.3256
CT	Litchfield	Warren	New Preston	41.7102	73.3567
CT	Litchfield	Warren	Cornwall	41.7691	73.3568
CT	Litchfield	Watertown	Litchfield	41.6455	73.1513
CT	Litchfield	Woodbury	Woodbury	41.6057	73.2247
CT	Middlesex	East Haddam	Deep River	41.4637	72.4322
CT	New Haven	Oxford	Naugatuck	41.4777	73.1232
CT	New London	Lyme	Old Lyme	41.3734	72.3442
CT	New London	Montville		41.4331	72.2290
CT	New London	Salem		41.4607	72.2686
CT	Windham	Scotland	Scotland	41.7001	72.0880
MA	Barnstable	Barnstable	Sandwich	41.6873	70.3981
MA	Barnstable	Barnstable	Sandwich	41.6319	70.4458
MA	Barnstable	Barnstable	Sandwich	41.6722	70.4428
MA	Barnstable	Barnstable	Hyannis	41.6588	70.2713
MA	Barnstable	Bourne	Pocasset	41.7263	70.6029
MA	Barnstable	Bourne	Pocasset	41.7387	70.6070
MA	Barnstable	Brewster	Orleans OE W	41.7653	70.0648
MA	Barnstable	Brewster	Orleans OE W	41.7574	70.1010

MA	Barnstable	Falmouth	Falmouth	41.5832	70.6068
MA	Barnstable	Mashpee	Sandwich	41.6260	70.4992
MA	Barnstable	Sandwich	Sandwich OE N	41.7529	70.4888
MA	Barnstable	Sandwich	Sagamore	41.7658	70.5120
MA	Barnstable	Sandwich	Sagamore	41.7674	70.5211
MA	Barnstable	Sandwich	Sandwich OE N	41.7534	70.4918
MA	Barnstable	Sandwich	Sagamore	41.7676	70.5160
MA	Barnstable	Sandwich	Sandwich	41.7484	70.4652
MA	Barnstable	Sandwich	Sandwich	41.6985	70.4522
MA	Barnstable	Sandwich	Sagamore	41.7758	70.5179
MA	Barnstable	Yarmouth	Hyannis	41.6750	70.2644
MA	Berkshire	Monterey	Great Barrington	42.1771	73.2553
MA	Berkshire	Otis	Monterey	42.2199	73.0354
MA	Berkshire	Sandisfield	South Sandisfield	42.1148	73.1497
MA	Berkshire	Sandisfield	South Sandisfield	42.0559	73.1493
MA	Hampden	Tolland	South Sandisfield	42.0843	73.0131
MA	Plymouth	Plymouth	Wareham	41.8316	70.6472
MA	Plymouth	Plymouth	Wareham	41.8307	70.6548
ME	Cumberland	Cape Elizabeth	Prouts Neck	43.5932	70.2539
ME	Cumberland	Cape Elizabeth	Cape Elizabeth	43.5954	70.2460
ME	Cumberland	Cape Elizabeth	Cape Elizabeth	43.6010	70.2335
ME	Cumberland	Cape Elizabeth	Cape Elizabeth	43.5778	70.2208
ME	Cumberland	Cape Elizabeth	Cape Elizabeth	43.5616	70.2122
ME	Cumberland	Cape Elizabeth	Cape Elizabeth	43.5808	70.2489
ME	Cumberland	Cumberland	Cumberland	43.7962	70.3063
ME	Cumberland	Cumberland	Cumberland Center	43.7711	70.2503
ME	Cumberland	Cumberland	Yarmouth	43.7785	70.2232
ME	Cumberland	Cumberland	Yarmouth	43.7803	70.2268
ME	Cumberland	Cumberland	Yarmouth	43.8031	70.2368
ME	Cumberland	Falmouth	Portland West	43.7426	70.2895
ME	Cumberland	Falmouth	Portland East	43.7173	70.2379
ME	Cumberland	Falmouth	Portland East	43.7441	70.2286
ME	Cumberland	Gorham	Standish	43.7114	70.5141
ME	Cumberland	Gorham	Gorham	43.6774	70.3916

ME	Cumberland	Gorham	Gorham	43.6791	70.3893
ME	Cumberland	Portland	Portland West	43.6734	70.3283
ME	Cumberland	Portland	Portland West	43.6530	70.3213
ME	Cumberland	Scarborough	Prouts Neck	43.5711	70.2776
ME	Cumberland	Scarborough	Prouts Neck	43.6184	70.3582
ME	Cumberland	Scarborough	Prouts Neck	43.5947	70.3010
ME	Cumberland	Scarborough	Prouts Neck	43.5798	70.2889
ME	Cumberland	Scarborough	Prouts Neck	43.5621	70.2805
ME	Cumberland	Scarborough	Prouts Neck	43.5741	70.3351
ME	Cumberland	Scarborough	Prouts Neck	43.5897	70.3393
ME	Cumberland	Scarborough	Prouts Neck	43.5590	70.3038
ME	Cumberland	Scarborough	Prouts Neck	43.5468	70.3128
ME	Cumberland	Scarborough	Prouts Neck	43.5670	70.2953
ME	Cumberland	Scarborough	Portland West	43.6451	70.3748
ME	Cumberland	Scarborough	Prouts Neck	43.5743	70.2778
ME	Cumberland	Scarborough	Prouts Neck	43.5845	70.2594
ME	Cumberland	South Portland	Portland West	43.6440	70.3331
ME	Cumberland	Westbrook	Gorham	43.6809	70.3772
ME	Cumberland	Westbrook	Gorham	43.6835	70.3815
ME	Cumberland	Windham	North Windham	43.7581	70.3870
ME	Cumberland	Windham	North Windham	43.7910	70.4399
ME	York	Berwick	Somersworth	43.2638	70.7807
ME	York	Berwick	Somersworth	43.2730	70.7608
ME	York	Biddeford	Kennebunk	43.4697	70.5074
ME	York	Dayton	Bar Mills	43.5426	70.5527
ME	York	Dayton	Bar Mills	43.5214	70.5537
ME	York	Dayton	Bar Mills	43.5371	70.5899
ME	York	Eliot	Portsmouth	43.1160	70.7689
ME	York	Eliot	Dover East	43.1424	70.7860
ME	York	Kittery	Kittery	43.1161	70.7323
ME	York	Kittery	York Harbor	43.1272	70.7127
ME	York	Saco	Old Orchard Beach	43.5237	70.4578
ME	York	Saco	Old Orchard Beach	43.5037	70.4838
ME	York	South Berwick	Dover East	43.1957	70.7973

ME	York	South Berwick	Dover East	43.2298	70.7598
ME	York	South Berwick	Dover East	43.2057	70.7789
ME	York	Wells	North Berwick	43.3043	70.7071
ME	York	Wells	Wells	43.3323	70.5591
ME	York	Wells	Wells	43.3312	70.5698
ME	York	York	York Harbor	43.1626	70.6557
ME	York	York	York Harbor	43.2002	70.6425
ME	York	York	York Harbor	43.1389	70.7024
NH	Hillsborough	Amherst	South Merrimack	42.8113	71.5907
NH	Hillsborough	Bedford	Manchester South	42.9436	71.4708
NH	Hillsborough	Bedford	Manchester South	42.9420	71.4663
NH	Hillsborough	Bedford	Manchester South	42.9209	71.4589
NH	Hillsborough	Bedford	Pinardville	42.9802	71.5170
NH	Hillsborough	Hudson	Nashua North	42.7653	71.3836
NH	Hillsborough	Litchfield	Nashua North	42.8614	71.4733
NH	Hillsborough	Manchester	Manchester South	42.9669	71.4158
NH	Hillsborough	Merrimack	Manchester South	42.8767	71.4910
NH	Hillsborough	Merrimack	Manchester South	42.8979	71.4621
NH	Merrimack	Bow	Suncook	43.1402	71.4734
NH	Merrimack	Hooksett	Manchester North	43.0432	71.4744
NH	Merrimack	Hooksett	Manchester North	43.0512	71.4591
NH	Merrimack	Hooksett	Manchester North	43.0201	71.4057
NH	Rockingham	Derry	Derry	42.9143	71.2990
NH	Rockingham	Londonderry	Windham	42.8572	71.3688
NH	Rockingham	Londonderry	Manchester South	42.9063	71.4212
NH	Strafford	Dover	Dover West	43.1814	70.8979
NH	Strafford	Dover	Dover West	43.1850	70.8946
NH	Strafford	Dover	Dover East	43.1723	70.8376
NH	Strafford	Lee	Barrington	43.1627	71.0056
NH	Strafford	Rollinsford	Dover East	43.2119	70.8451

NH	Strafford	Rollinsford	Dover East	43.2141	70.8371
NY	Columbia	Gallatine	Ancram	42.0568	73.6382
NY	Columbia		Copake	42.0204	73.6154
NY	Dutchess		Millbrook	41.8551	73.6557
NY	Dutchess	Amenia	Amenia	41.8500	73.5239
NY	Dutchess	Dover	Dover Plains	41.7092	73.5600
NY	Dutchess	Pawling	Poughquag	41.5483	73.6317
NY	Dutchess	Pawling	Pawling	41.5844	73.6092
NY	Dutchess	Pine Plains	Pine Plains	41.9832	73.7100
NY	Dutchess		Hyde Park	41.8024	73.6607
NY	Dutchess		West Point	41.4717	73.9755
NY	Putnam	Putnam Valley	Oscawana Lake	41.4680	73.8168
NY	Putnam	Southeast	Croton Falls	41.3597	73.6389
NY	Putnam		West Point	41.3990	73.8870
NY	Westchester		Mount Kisco	41.2079	73.7349
RI	Newport	Newport	Newport	41.4954	71.2497
RI	Newport	Portsmouth	Prudence Island	41.5400	71.2581
RI	Providence	Cumberland	Pawtucket	41.9813	71.4466
RI	Providence	North Smithfield	Georgiaville	41.9958	71.5579
RI	Providence	Smithfield	Georgiaville	41.9139	71.5290
RI	Washington	Exeter	Slocum	41.5563	71.6047
RI	Washington	Hopkinton	Hope Valley	41.5249	71.7242
RI	Washington	Hopkinton	Hope Valley	41.5246	71.7241
RI	Washington	Hopkinton	Hope Valley	41.5238	71.7014
RI	Washington	Richmond	Carolina	41.4466	71.6812
RI	Washington	Richmond	Carolina	41.4471	71.6801

APPENDIX B. Summary of GIS datasets used for modeling habitat and identifying potential restoration sites for New England cottontails

Category	Data Layer(s)	Source	Citations/download
<u>Habitat Models</u>			
Land Use Land Cover	National Land Cover Data	United State Geological Survey	Yang et al. (2001) Vogelman et al. (2001) http://landcover.usgs.gov/natl/landcover.php
Potential Dispersal Corridors	TIGER Line Files	United States Census Bureau	U.S. Census Bureau 2003 http://www.census.gov/geo/www/tiger/
Winter Severity	Mean Annual Snowfall	Oregon State University – Climate Analysis Service	Daly et al. (2001) http://www.climatesource.com/
<u>Conservation Lands</u>			
New Hampshire	New Hampshire Conservation/Public Lands	New Hampshire Geographically Referenced Analysis and Information Transfer System (NH GRANIT)	http://www.granit.sr.unh.edu/
Maine	Maine Conservation Lands	Maine Office of GIS	http://apollo.ogis.state.me.us/
Massachusetts	Massachusetts Protected and Recreational Open Space	Massachusetts Geographic Information Systems	http://www.mass.gov/mgis/

Connecticut	DEP Property, Federal Property, Municipal and Private Open Space	University of Connecticut Map and Geographic Information Center	http://magic.lib.uconn.edu/
New York	NYS Public Lands Boundaries	NYS Office of Cyber Security and Critical Infrastructure Coordination	http://www.nysgis.state.ny.us/
Rhode Island	Local and NGO Conservation Lands and Parks, State Conservation and Park Lands, Federal Conservation and Park Lands	Rhode Island Geographic Information System	http://www.edc.uri.edu/rigis/

APPENDIX C. The original twenty-one land cover types reclassified into five super classes based on biological significance for New England cottontails.

Code	NLDC Level I	NLDC Level II	Inventoried Level II
11	Open Water	Water	Water / wetlands
21	Low Intensity Residential	Developed	Developed
22	High Intensity Residential	Developed	Developed
23	Commercial/Industrial/Transportation	Developed	Developed
31	Bare Rock/Sand/Clay	Barren	Open
32	Quarries/Strip Mines/Gravel Pits	Barren	Open
33	Transitional	Barren	Open
41	Deciduous Forest	Forest	Forest
42	Coniferous Forest	Forest	Forest
43	Mixed Forest	Forest	Forest
51	Shrubland	Shrubland	Open
61	Orchards/Vineyards/Other	Non-natural Woody	Agriculture
81	Pasture/Hay	Herbaceous Planted/Cultivated	Agriculture
82	Row Crops	Herbaceous Planted/Cultivated	Agriculture
84	Fallow	Herbaceous Planted/Cultivated	Agriculture
85	Urban/Recreational Grasses	Herbaceous Planted/Cultivated	Open
91	Woody Wetlands	Wetlands	Forest
92	Emergent Herbaceous Wetlands	Wetlands	Water / Wetlands

APPENDIX D. Location of potential restoration sites for NEC that were classified as “highly ranked” according to the model that incorporates the size of the parcel, the distance to the nearest known population of NEC and habitat suitability as determined by logistic regression modeling. Coordinates are in decimal degrees.

Region	State	County	Latitude	Longitude	Hectares
Northeast	ME	Cumberland	43.57634	70.32676	132
Northeast	ME	Cumberland	43.56935	70.36404	471
Northeast	ME	Cumberland	43.56646	70.30756	25
Northeast	ME	Cumberland	43.55654	70.33056	91
Northeast	ME	Cumberland	43.55420	70.31521	89
Northeast	ME	Cumberland	43.54444	70.34926	95
Northeast	ME	Cumberland	43.54436	70.31021	18
Northeast	ME	York	43.39868	70.62716	358
Northeast	ME	York	43.33931	70.54604	93
Northeast	ME	York	43.31939	70.57060	112
Northeast	ME	York	43.29719	70.57651	103
Northeast	ME	York	43.27832	70.58632	156
Northeast	NH	Strafford	43.19298	70.86010	22
Northeast	NH	Merrimack	43.18404	71.55856	218
Northeast	NH	Strafford	43.18374	70.89951	20
Northeast	NH	Rockingham	43.04539	70.80565	131
Northeast	MA	Essex	42.53011	71.05071	143
Northeast	MA	Essex	42.44054	70.99759	211
Northeast	MA	Middlesex	42.43809	71.16243	76
Southeast	CT	Hartford	41.78626	72.57673	78
Southeast	MA	Barnstable	41.75242	69.94753	78
Southeast	MA	Barnstable	41.75623	70.05455	130
Southeast	MA	Barnstable	41.69250	70.06475	78
Southeast	MA	Barnstable	41.72429	70.34803	1599
Southeast	MA	Barnstable	41.76523	70.48609	52
Southeast	MA	Barnstable	41.76112	70.47427	21
Southeast	MA	Barnstable	41.76630	70.55645	93
Southeast	MA	Barnstable	41.68615	70.18553	52
Southeast	MA	Barnstable	41.73167	70.42258	129
Southeast	MA	Barnstable	41.69482	70.28658	20
Southeast	MA	Barnstable	41.70095	70.37207	77

Southeast	MA	Barnstable	41.65125	70.18323	49
Southeast	MA	Barnstable	41.64016	70.22521	65
Southeast	MA	Barnstable	41.65095	70.26784	15
Southeast	MA	Barnstable	41.63495	70.31162	58
Southeast	MA	Barnstable	41.65102	70.45308	108
Southeast	MA	Barnstable	41.65914	70.51649	382
Southeast	MA	Barnstable	41.61853	70.54053	100
Southeast	MA	Barnstable	41.57187	70.61264	110
Southeast	MA	Barnstable	41.55959	70.63621	206
Southeast	MA	Barnstable	41.75532	70.12639	156
Southeast	MA	Barnstable	41.73706	70.15326	112
Southeast	MA	Barnstable	41.73133	70.22710	100
Southeast	MA	Barnstable	41.71662	70.22387	146
Southeast	MA	Barnstable	41.66443	69.98861	85
Southeast	MA	Barnstable	41.55690	70.50109	207
Southeast	MA	Bristol	41.94034	71.30141	80
Southeast	MA	Hampden	42.03895	72.57318	112
Southeast	MA	Hampshire	42.34945	72.53634	303
Southeast	MA	Hampshire	42.31488	72.60797	341
Southeast	MA	Middlesex	42.37043	71.14306	103
Southeast	MA	Middlesex	42.36600	71.25707	105
Southeast	MA	Middlesex	42.33096	71.21314	75
Southeast	MA	Middlesex	42.33343	71.23471	81
Southeast	MA	Middlesex	42.28094	71.43418	77
Southeast	MA	Norfolk	42.08135	71.41560	84
Southeast	MA	Norfolk	42.02170	71.47980	95
Southeast	MA	Plymouth	42.16812	70.73736	111
Southeast	MA	Plymouth	41.93974	70.64909	79
Southeast	MA	Plymouth	41.77576	70.72958	82
Southeast	MA	Plymouth	41.74972	70.64442	69
Southeast	MA	Plymouth	41.68620	70.72128	83
Southeast	MA	Plymouth	42.04386	70.63843	114
Southeast	MA	Providence	41.94676	71.45842	45
Southeast	MA	Suffolk	42.29880	71.12432	113
Southeast	MA	Suffolk	42.27895	71.10804	120
Southeast	MA	Worcester	42.15409	71.51176	82

Southeast	RI	Bristol	41.73250	71.32520	142
Southeast	RI	Newport	41.60592	71.26648	95
Southeast	RI	Newport	41.56529	71.26712	76
Southeast	RI	Newport	41.55741	71.24132	46
Southeast	RI	Newport	41.55602	71.26307	19
Southeast	RI	Newport	41.54902	71.27203	74
Southeast	RI	Newport	41.55235	71.29197	66
Southeast	RI	Newport	41.53824	71.30306	48
Southeast	RI	Newport	41.52729	71.26625	62
Southeast	RI	Newport	41.51836	71.26502	57
Southeast	RI	Newport	41.47633	71.18497	77
Southeast	RI	Newport	41.49097	71.27299	19
Southeast	RI	Newport	41.48479	71.24558	56
Southeast	RI	Newport	41.63641	71.23905	86
Southeast	RI	Providence	41.90100	71.40012	90
Southeast	RI	Providence	41.87543	71.34298	162
Southeast	RI	Providence	41.83967	71.45978	114
Southeast	RI	Providence	41.78324	71.41213	144
Southeast	RI	Washington	41.53671	71.50933	64
Southeast	RI	Washington	41.48612	71.53895	259
Southeast	RI	Washington	41.37876	71.49968	121
Southeast	RI	Washington	41.35368	71.65749	138
Southwest	CT	Fairfield	41.59835	73.50308	63
Southwest	CT	Litchfield	41.97025	73.35389	365
Southwest	CT	Litchfield	41.96642	73.33054	95
Southwest	CT	Litchfield	41.95286	73.33470	137
Southwest	CT	Litchfield	41.94295	73.46094	107
Southwest	CT	Litchfield	41.93654	73.43077	207
Southwest	CT	Litchfield	41.92286	73.42803	91
Southwest	CT	Litchfield	41.86517	73.48912	86
Southwest	CT	Litchfield	41.85444	73.24986	112
Southwest	CT	Litchfield	41.83161	73.20240	94
Southwest	CT	Litchfield	41.82130	73.22316	52
Southwest	CT	Litchfield	41.80835	73.18475	144
Southwest	CT	Litchfield	41.75369	73.44829	292
Southwest	CT	Litchfield	41.74672	73.15971	380

Southwest	CT	Litchfield	41.72808	73.21895	457
Southwest	CT	Litchfield	41.74001	73.22603	54
Southwest	CT	Litchfield	41.70561	73.11249	151
Southwest	CT	Litchfield	41.70577	73.38449	54
Southwest	CT	Litchfield	41.68898	73.21594	112
Southwest	CT	Litchfield	41.68244	73.22597	179
Southwest	CT	Litchfield	41.66927	73.25706	159
Southwest	CT	Litchfield	41.60109	73.20306	93
Southwest	CT	Litchfield	41.58791	73.43697	46
Southwest	CT	Litchfield	41.55915	73.19221	62
Southwest	CT	Litchfield	41.52334	73.34906	267
Southwest	CT	Litchfield	41.51497	73.36834	81
Southwest	CT	Middlesex	41.50234	72.69111	84
Southwest	CT	Middlesex	41.47833	72.69717	549
Southwest	CT	New Haven	41.47032	73.27691	107
Southwest	CT	New Haven	41.46341	73.14047	63
Southwest	MA	Berkshire	42.64977	73.09693	219
Southwest	MA	Berkshire	42.64375	73.26387	257
Southwest	MA	Berkshire	42.59793	73.10770	275
Southwest	MA	Berkshire	42.52454	73.20828	93
Southwest	MA	Berkshire	42.51632	73.21121	86
Southwest	MA	Berkshire	42.31556	73.27678	395
Southwest	MA	Berkshire	42.27081	73.24336	279
Southwest	MA	Berkshire	42.22735	73.34410	127
Southwest	MA	Berkshire	42.20686	73.41046	508
Southwest	MA	Berkshire	42.19670	73.41812	168
Southwest	MA	Berkshire	42.18033	73.43723	765
Southwest	MA	Berkshire	42.17863	73.23829	103
Southwest	MA	Berkshire	42.17515	73.37657	272
Southwest	MA	Berkshire	42.14950	73.37906	118
Southwest	MA	Berkshire	42.14904	73.35630	115
Southwest	MA	Berkshire	42.13624	73.41203	127
Southwest	MA	Berkshire	42.12421	73.34314	82
Southwest	MA	Berkshire	42.07377	73.33884	126
Southwest	MA	Berkshire	42.05507	73.29193	84
Southwest	MA	Berkshire	42.05478	73.35208	147

Southwest	MA	Franklin	42.65005	72.67270	283
Southwest	MA	Hampden	42.14484	72.79379	431
Southwest	MA	Hampden	42.07897	72.76318	98
Southwest	MA	Hampden	42.06492	72.78285	253
Southwest	MA	Hampden	42.03674	72.64515	235
Southwest	MA	Hampshire	42.40899	72.92729	152
Southwest	MA	Hampshire	42.28965	72.69636	269
Southwest	MA	Hampshire	42.19372	72.73571	144
Southwest	NY	Dutchess	41.97904	73.72726	93
Southwest	NY	Dutchess	41.85712	73.92154	149
Southwest	NY	Dutchess	41.68733	73.79528	299
Southwest	NY	Dutchess	41.64964	73.70167	234
Southwest	NY	Dutchess	41.54213	73.94669	236
Southwest	NY	Rensselaer	42.58630	73.75175	134

UNIVERSITY OF NEW HAMPSHIRE

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LAST NAME	Litvaitis	FIRST NAME	John
DEPT	Natural Resources Department, 213 James Hall	APP'L DATE	8/7/2001
OFF-CAMPUS ADDRESS (if applicable)	Natural Resources Department, 213 James Hall	IACUC #	010802
PROJECT TITLE	Landscape features affecting dispersal among populations of New England cottontails: Phase I - application of molecular genetics		

All cage, pen or other animal identification records must include your IACUC Protocol # as listed above.

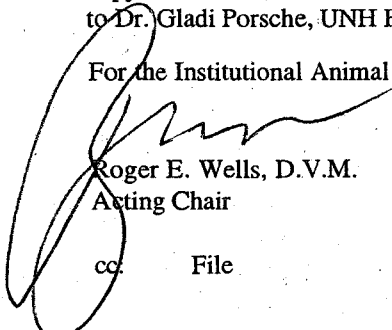
The Institutional Animal Care and Use Committee has reviewed and approved the protocol submitted for this study under Category C on Page 4 of the "Application for Review of Animal Use in Research or Instruction" – the research potentially involves minor short-term pain, discomfort or distress which will be treated with appropriate anesthetics/analgesics or other assessments. The IACUC made the following comments on this protocol -- *comments* are usually minor editorial changes or clarifications that do not affect approval status (unlike contingencies, which require investigator action for initial or continuing approval):

1. In Section VI, d (number of animals: capture), second row, Committee inserted "Juvenile" in second column.

Approval is granted for a period of three years from the approval date above. Continued approval throughout the three year period is contingent upon completion of annual reports on the use of animals. At the end of the three year approval period you may submit a new application and request for extension to continue this project. Requests for extension must be filed prior to the expiration of the original approval.

Please note: Use of animals in research and instruction is approved contingent upon participation in the UNH Occupational Health Program for persons handling animals. *Participation is mandatory* for all principal investigators and their affiliated personnel, employees of the University and students alike. A Medical History Questionnaire accompanies this approval; please copy and distribute to all listed project staff who have not completed this form already. Completed questionnaires should be sent to Dr. Gladi Porsche, UNH Health Services. Thank you.

For the Institutional Animal Care and Use Committee,


Roger E. Wells, D.V.M.
Acting Chair

cc: File