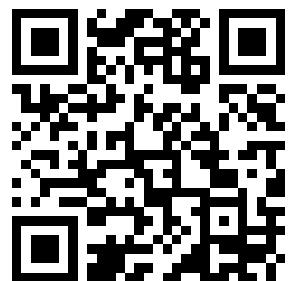


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# **Migratory Nongame Birds of Management Concern in the Northeast**

***Editors***

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**Cover Illustration (Artist, Mark McCollough):** Migratory nongame birds associated with a variety of habitats in the Northeast. Species (clockwise from lower left): common loon, American bittern, cerulean warbler, black tern, short-eared owl, and black rail.

## Preface

This publication is designed to provide a summary of information on the biology, status, and management of 20 migratory bird species of management concern in the northeastern U.S. These species were adapted from a national list of species of management concern compiled by the U.S. Fish and Wildlife Service, Office of Migratory Bird Management, in 1987. This compendium features species chosen primarily because of their small and/or declining population sizes resulting from a host of assaults such as habitat loss and degradation, human disturbance, and contaminants. The selection of these 20 species does not in any way imply that other bird species are not of management concern in the Northeast; instead these are chosen as a focus for land managers, researchers, and other resource specialists.

The species in this volume are representative of biological communities that are threatened in the Northeast. Wetlands continue to be modified and lost; contiguous forests are fragmented by buildings, roads, and powerlines; and fields are succeeding to shrubs and young trees. The species presented here show the most dramatic declines; other members of these communities are also losing ground.

It is our hope that land managers and biologists will use this information in their plans and activities. Life history accounts provide background information on species identification, reproduction, and habitat use. The authors define ranges worldwide, and maps are provided for North and Central America. Known population status is discussed for the Northeast and for each of the 13 states in the region. Limiting factors and threats are identified and the management potential of each species is discussed. Current monitoring and management programs are explained, and usable guidelines are given. Research needs are also identified. In all, each chapter provides a concise summary of the current status of a species of management concern.

This information is also incorporated into the Nature Conservancy's Biological Conservation Data (BCD) System as stewardship abstracts. These abstracts are updated as new information becomes available and will remain a ready source of current data. Abstracts can be obtained from state Natural Heritage Programs or the Nature Conservancy's Eastern Regional Office in Boston.

Many opportunities exist to integrate this information into population and habitat management plans. Land managers can see how their units fit into the range of these species and assess their units for their potential to provide habitat for these birds. If the species are present on the area, the land manager should identify at what time of year, and then accordingly provide optimal breeding, migrating, or wintering habitat.

**Conflicting land management uses should be identified and resolved based upon well-defined objectives. The species' distribution should be recorded and their status tracked through the monitoring programs presented in this volume. Responses to land management changes should be evaluated for these species. Information and actions accumulated over time may help to reverse the negative trends now being exhibited by these species in today's changing landscape.**

## Acknowledgements

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Much of the information in these chapters was provided by state fish and wildlife agencies and Natural Heritage Programs in the 13 northeastern states. Breeding bird atlas data were provided and/or reviewed by the following individuals: J. Baird, L. Bevier, D. Brauning, A. R. Buckelew, Jr., R. Enser, C. Foss, E. Hentcy, D. Hughes, A. Hutchinson, S. Laughlin, J. Ozard, R. Miller, S. Ridd, W. Sabin, C. Stihler, S. Sutcliffe, G. Therres, R. Wadja, and R. West. Data were transformed by K. Schneider and J. Ozard from quadrangles to counties for each state. Regional maps indicating species distributions were created by J. Ozard. Data from U.S. Fish and Wildlife Service Breeding Bird Surveys were provided by S. Droege with the Office of Migratory Bird Management. Rangewide maps were created by D. Pence with assistance from P. Graves. Thanks to M. McCollough for illustrating the cover of this volume.



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# **Wetlands**



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# COMMON LOON

## *Gavia immer*

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**Common loons nest on islands or backwater areas on lakes with adequate fish prey. Pairs typically raise one or two chicks, but failure is common and compensated sometimes by renesting. On their northern breeding grounds in Canada, common loon populations appear to be stable or increasing. However, at the southern edge of their range in the Northeast, breeding loons have gradually drawn northward in the past century. Many types of disturbances threaten loons at their breeding sites, including lakeshore development, human recreational disturbance, predation, fluctuating water levels at the nest site, entanglement in fishing gear, environmental pollutants, and acidic rain leading to loss of prey. Identification, monitoring and protection of nest sites in areas of human use are essential to the continued nesting success of this species. Wintering areas along the Pacific and Atlantic coasts also require protection from the damages of oil spills. More information is needed about wintering areas, distribution, and numbers.**

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### DESCRIPTION

#### Taxonomy

The common loon (*Gavia immer*, Brunnich), is one of five species in the single modern genus of Gaviidae, the only family in the order Gaviiformes. There are no recognized subspecies of *G. immer*. This species and the closely related yellow-billed loon (*G. adamsii*) constitute a superspecies and are considered conspecific by a minority of authors (American Ornithologists' Union 1983, 1985).

#### Morphology and Plumages

Although some overlap exists between the sexes, within pairs, male common loons are consistently larger than females (Barr 1973, McIntyre 1988a). Adults measure about 1 m in

length outstretched (Palmer 1962, McIntyre 1988a). Weight varies geographically and within populations, with adults ranging from 2.7 kg to more than 6.3 kg (McIntyre 1988a). Sexual dimorphism is most pronounced in bill length and depth (McIntyre 1975, Storer 1988). Some structures of the digestive tract are larger in males, suggesting possible food partitioning by pair members (Barr 1973).

The plumage differences between the sexes are indistinguishable. In alternate (breeding) plumage, the head and neck are velvety black with a slight greenish gloss. Across the throat is a prominent transverse bar of short, vertical white streaks; on either side of the neck is a collar of similar, longer white streaks, not quite meeting in the center of the throat or the back of the neck. The bill is black, and the iris is brownish ruby. The upperparts are black with a greenish gloss, heavily spotted with white, each feather (except

the unmarked upper tail coverts) with a pair of white, squarish subterminal spots, smallest on the upper mantle, back and rump, and largest on the scapulars. The underparts are mainly white, the sides of the breast are streaked black and white, and the flanks are black with small white spots. The tail is short, with 16-20 feathers, and entirely black. The wings are narrow and pointed, about 35 cm long (unflattened), with 11 primaries and 22-23 secondaries. The remiges are blackish with dark shafts, and the tips of the inner secondaries have single or paired white spots. The coverts are similar to the upperparts but have more circular subterminal spots. The wing lining is mostly white. The legs are black on the outer side paling to grayish or white inside; the webs of the feet are mostly white above, and black with white centers below (Palmer 1962, Jackson 1976, Johnsgard 1987, McIntyre 1988a).

The definitive alternate plumage is acquired by a complete molt between January and March, which renders birds flightless for nearly a month (Woolfenden 1972, McIntyre 1988a). This plumage is not acquired until the third or fourth summer (McIntyre 1986).

The basic (winter) plumage is acquired by a partial molt of contour feathers beginning in late summer and lasting through fall (although earlier in unsuccessful breeders and immatures). In this plumage the forehead, crown and back are grayish-brown, and the chin, throat and foreneck are white. The bill is brownish-gray to pale bluish-gray or horn colored. The iris is brown. The upperparts are brownish-gray, the feathers margined with paler gray, with a few black and white feathers occasionally retained in adults. The underparts are mainly white, with a brownish, streaked appearance on the sides of the breast and flanks. The tail is dark brown, tipped with white (Bent 1919; Johnsgard 1987; McIntyre 1986, 1988a).

The juvenal plumage begins to emerge at about one month. This plumage is similar to the adult basic plumage, although the upperparts have paler and more conspicuous feather margins than those of adults, and the throat and sides of the neck are more finely streaked with brown. Flight feather growth is completed at 12-13 weeks of age, and this entire plumage is worn until the following summer when a complete molt produces

another, more adult-like basic plumage (Palmer 1962, McIntyre 1988a).

Newly-hatched chicks undergo two successive changes of downy plumage. The first down is blackish, paler on the throat, upper breast and flanks, and white on the lower breast and belly. This is replaced at 10-14 days of age by the second downy plumage of primarily brownish-gray feathers, which are replaced by the juvenal contour feathers at 4 weeks (Palmer 1962, McIntyre 1988a).

## Vocalizations

Four basic types of loon calls are distinguishable. Hoots are short, one-syllable notes used as contact calls among individuals in close proximity to one another. These "conversational" calls are given between mates, between parents and young chicks, during social gatherings, and among members of migratory flocks in fall (Palmer 1962, Klein 1985, McIntyre 1988a).

Wails, or night calls, consist of three distinct types. One-note wails are single, unbroken notes of relatively unchanging tone. Two-note and three-note wails include tonal jumps and typically last several seconds. Wails are used to initiate interaction among members of a pair or family, such as for reestablishing contact after separation, signalling nest exchanges, and communicating messages to chicks. They are also used in night chorusing (Barklow 1979, Klein 1985, McIntyre 1988a).

Tremolos, or laughing calls, are frequency-modulated calls of varying lengths. They function primarily as alarm calls and are of three distinct types, reflecting graded frequencies in response to different intensities of threat. Type 3 tremolos are often accompanied by water-treading displays, signalling extreme distress. Tremolos are also given in flight, during pair duets, and during night chorusing (Barklow 1979, McIntyre 1988a).

Yodels are acoustically complex and variable calls thought to be given solely by males. They appear to correspond to male song in most passerine bird species, functioning primarily for territorial advertisement and defense, perhaps secondarily for mate attraction. Yodels appear to be unique to individual male loons and unvarying

over time, probably providing information to conspecifics on individual recognition (Sjolander and Agren 1972, Rummel and Goetzinger 1975, McIntyre 1988a, Miller 1988).

## HABITAT

### Breeding Habitat

Common loons select a diversity of lake types for breeding, ranging from oligotrophic to eutrophic, small to large, shallow to deep, clear to turbid, and remote to heavily developed (McIntyre 1975). Breeding has been documented on lakes as small as 4 ha but most often occurs on lakes of 20 ha or larger (McIntyre 1975). Lakes smaller than 80 ha generally support only a single loon pair, while lakes of several hundred hectares or more may support many nesting pairs (McIntyre 1988a). Typically, territory sizes of loons on large lakes are greater than on small lakes (Titus and VanDruff 1981, McIntyre 1988a).

Most loons breed on lakes containing both shallow and deep water areas (McIntyre 1975, 1988a; Strong 1985). In general, feeding by adults takes place in areas of relatively shallow depth (1-4 m) and moderate distance from land (51-150 m), often over rocky shoals that concentrate prey items (Strong 1985, McIntyre 1988a, Strong and Bissonette 1989). Brood-rearing areas are typically located in shallow coves of fairly uniform depth, sheltered from prevailing winds and wave action, and are independent of nest site location (McIntyre 1983, Strong 1985). Adults tending chicks prefer shallow water areas (< 2 m) close to land (< 150 m) (Strong 1985, Strong and Bissonette 1989). Deepwater areas (> 4 m) distant from land (> 250 m) are avoided by feeding adults and adults tending chicks, but are often used for social interactions (Strong 1985). Breeding adults usually feed outside of nursery areas (Strong 1985), occasionally outside of their territories (McIntyre 1983), and may visit nearby lakes for feeding (Miller and Dring 1988).

Water clarity is an important component of breeding habitat selection by common loons. Loons are visual predators and generally need clear visibility to at least 3-4 m (McIntyre 1988a),

although they can adapt to some conditions of low water clarity (McIntyre 1975). In studies comparing lakes with and without loons, higher turbidity has been suggested as a factor influencing lack of occupancy (Barr 1973, McIntyre 1988a).

Loons nest on both eutrophic (Vermeer 1973a, McIntyre 1975) and oligotrophic lakes (Barr 1973, Fox et al. 1980, Parker 1985). Oligotrophic lakes are less productive and generally support greater quantities of large than small fish. Because loons prefer small fish as food for their young, they tend to occupy larger oligotrophic lakes than eutrophic lakes to compensate for the reduced prey base per unit area (McIntyre 1975), or they occupy larger territories on these lakes (Klein 1985). While some studies have shown that loons occupy only those lakes that support fish (e.g., Vermeer 1973a, McIntyre 1975), others have documented loons on lakes without fish, where they feed aquatic insects and larvae to their chicks (e.g., Munro 1945, Parker 1985).

Availability of nest sites also influences loon habitat selection. Most studies have shown that small islands (usually < 2.5 ha) are strongly preferred over mainland nest sites (Olson and Marshall 1952, Vermeer 1973a, McIntyre 1975, Titus and VanDruff 1981, Strong 1985), and that quiet, backwater sites may be selected over mainland sites (Strong 1985). Loons have been found nesting in marshy portions of lakes in water depths no greater than 0.5 m (Alvo 1981). However, if natural islands are lacking or are unsuitable due to human disturbance or other factors, loons will readily use shoreline nest sites.

Nests are nearly always built at the water's edge. Substrates range from masses of aquatic and terrestrial vegetation, to moss, to bare soil, sand or rock, to depressions in old muskrat (*Ondatra zibethicus*) houses (McIntyre 1988a). Optimal nest sites, as measured by degree of success, include overhead cover to conceal eggs from predators, protection from wind and waves, good visibility by incubating adults, and a steep slope adjacent to the nest for adequate underwater approaches and exits (McIntyre 1975, 1983, 1988a).

## Wintering Habitat

Common loons winter primarily in coastal marine habitats, including bays, coves, channels, inlets and other shallow areas (Bent 1919, Palmer 1962, McIntyre 1988a). Some individuals overwinter on inland lakes and rivers, although this appears to be largely weather influenced (McIntyre 1988a). While shallow, inshore waters appear to be utilized more frequently than deeper, offshore waters (McIntyre 1978, Daub 1989), some loons use continental shelf waters up to 100 m in depth and 100 km from land (Haney 1990). In the southeastern U.S. (between 29° and 35°N latitude), wintering loons were most common in waters 0-19 m deep but were rare or absent in highly turbid waters 5-15 km from shore. Loon distribution shifted farther offshore during midwinter to avoid increases in these turbid water areas (Haney 1990). Feeding typically occurs in water depths of < 5 m, while maintenance activities (e.g., preening and drifting) take place in deeper water (McIntyre 1978, Daub 1989).

## BIOLOGY

### Reproduction

**Arrival and Territory Establishment.** - The timing of spring arrival is correlated with latitude and dictated primarily by ice-out phenology (McIntyre 1988a). In southern portions of the breeding range, pairs may reoccupy territories in March, while at northern latitudes arrival may be delayed until mid or late May (McIntyre 1988a). In Minnesota, an average of 8 days elapsed between ice break-up and loon arrival in an early ice-out year, 5 days in an average year, and 3 days in a late year (McIntyre 1975). Males typically return first, especially in southern breeding areas, and are followed by females (McIntyre 1975, 1988a; Sutcliffe 1980). However, pairs often arrive together at northern lakes (McIntyre 1988a). Territories are established immediately after arrival and may change in size as the breeding season progresses, expanding after chicks hatch and shrinking for failed pairs (McIntyre 1988a).

**Courtship.** - It is believed that common loon pairs remate each spring and that courtship serves primarily to renew the pair bond (McIntyre 1988a). Courtship begins shortly after territory reoccupation and involves quiet, shared displays, including simultaneous swimming, head posturing and short dives. Vocalizations are not extensive. Copulation sequences are stereotyped, typically last from 3-10 min, and take place on land (McIntyre 1988a). Some copulation sites become nest sites (McIntyre 1975).

**Nesting Period.** - Nest-building is conducted by both members of the pair and may immediately follow copulation, sometimes lasting over 4 days (McIntyre 1975, 1988a). Egg-laying begins 1-4.5 weeks after spring arrival, usually during mid-May in the south, and well into June farther north (Palmer 1962, McIntyre 1975). Eggs are typically laid at 2-day intervals (McIntyre 1975). Replacement clutches following failures of first nests are common (McIntyre 1975, 1988a). Renests have been reported to occur within 5 days of a nest loss (Olson and Marshall 1952), but intervals of 10-14 days appear to be most common (Olson and Marshall 1952, McIntyre 1975, Sutcliffe 1980). Up to three laying cycles have been recorded in a season (Olson and Marshall 1952, McIntyre 1975). Nests lost early in the season are more likely to be replaced than those lost later (McIntyre 1988a). Replacement clutches have been initiated as late as early July in Vermont (Kaveney and Rimmer 1989).

**Nests and Eggs.** - Nests are typically large, bulky structures composed of vegetation, if available, but may consist simply of scrapes in the bare soil or duff, or may be placed directly on rocky substrate (Bent 1919, McIntyre 1988a). Some are built on sedge or ericaceous bog mats, and occasionally partially rotted, semi-submerged logs are used (McIntyre 1988a). Nest materials often include clumps of partially decayed aquatic vegetation, roots and rhizomes (McIntyre 1975, 1988a). Nest diameters average 56-66 cm outside, 24.5-33 cm inside, and 3-7.6 cm deep (Olson and Marshall 1952, McIntyre 1975, Sutcliffe 1980). If waters rise during incubation, loons continue adding to the nest's height to prevent flooding (McIntyre 1988a). Replacement nests tend to

have smaller outside dimensions (McIntyre 1975). Nest bowls are often reused in subsequent years, and occasionally within years for replacement clutches (Strong et al. 1987).

Loon eggs are subelliptical to ovoid in shape and vary from deep-olive to light-brown in color, most being deep-olive brown with irregular dark brown or black spots. Dimensions average 86.5-91.5 mm by 54-57 mm, and weights average 140-160 g. Second eggs are smaller than first eggs, and eggs in replacement clutches are smaller than those in original clutches (McIntyre 1988a).

**Clutch Size and Incubation.** - Most clutches contain two eggs, and most one-egg clutches result from loss of the first egg (McIntyre 1975, Titus and VanDruff 1981). Three-egg clutches are very rare (Bent 1919, McIntyre 1988a), and only two, four-egg clutches have been reported (Nelson 1983, Zicus et al. 1983). Both pair members incubate, beginning with the laying of the first egg, for an average period of 28-29 days, ranging from 26-31 days (Bent 1919, Olson and Marshall 1952, Palmer 1962, McIntyre 1975). An adult is present at the nest 99% of the time, and the eggs hatch within a day of one another (McIntyre 1975).

**Chick Rearing.** - Chicks leave the nest within 24 hrs of hatching and are soon moved to nursery areas (McIntyre 1988a). In Saskatchewan, nurseries were located an average of 500 m from nest sites and occupied about 15% of territory size (McIntyre 1983). Both adults tend the young by feeding, carrying and defending them for several weeks. Chicks are carried on their parents' backs until they reach three weeks of age (McIntyre 1975). Although chicks are capable of short dives at the time of nest departure and may capture some fish by the second or third week (McIntyre 1975), they are fed largely by their parents until 8 weeks of age (McIntyre 1988a). Adults aggressively defend chicks underwater and on the surface (McIntyre 1988a). Most juveniles are capable of flight at 11-12 weeks (Barr 1973, McIntyre 1975), and some leave their small, natal lakes or parental territories shortly afterwards (McIntyre 1975).

**Nesting Success.** - Breeding success varies considerably among populations. Most breeding failures occur during incubation, from factors such as predation, flooding or stranding due to water level fluctuations, and human intrusion (e.g., Olson and Marshall 1952, McIntyre 1975, Wood 1979, Titus and VanDruff 1981, Rimmer and Kaveney 1988). Chick survival is relatively high, especially after chicks reach 2-3 weeks of age (McIntyre 1988a). However, Alvo et al. (1988) recently found higher mortality of older chicks on highly acidified lakes in Ontario, due to presumed starvation from an inadequate food base. Fledging success (percent of hatched chicks fledged) from a sample of 1,500 pairs across the breeding range averaged 80% (range = 67-94%) (McIntyre 1988a). Productivity (number of fledglings per pair) of this sample averaged 0.60 and varied widely between 0.22 for 9 pairs in Minnesota (McIntyre 1975) and 0.97 for 132 pairs in New York (Parker and Miller 1988).

**Site Fidelity.** - Common loons appear to be faithful to breeding territories. Banded adults have been recaptured on the same breeding territory in subsequent years (McIntyre 1974; Yonge 1981, cited in McIntyre 1988a; Eberhardt 1984, cited in McIntyre 1988a). Yearly reuse of nest sites and nursery areas has been documented (Strong et al. 1987), but it is not known whether the same individuals were involved. Sonograms of yodel calls suggest that individual males return to the same territory each year (McIntyre 1988a, Miller 1989). Little is known about mate fidelity of breeding pairs.

### Feeding Ecology

Loons are primarily piscivorous, but are opportunistic and will eat any suitable prey they can readily see and capture (McIntyre 1988a). Their primary food on breeding lakes is yellow perch (*Perca flavescens*), followed by other shallow, warmwater fish and minnows (Cyprinidae) (Olson and Marshall 1952, Palmer 1962, Barr 1973, McIntyre 1986). Salmonids are taken on lakes that have low populations of other fish species (McIntyre 1988a). On the Great Lakes, alewives (*Alosa pseudoharengus*) appear to

be the most common prey item (McIntyre 1988a). Crustaceans, especially crayfish (Decapoda), are commonly taken, and plant material is occasionally eaten (Palmer 1962, McIntyre 1988a). On lakes without fish, loons have been reported feeding on molluscs, insects, amphipods and amphibians (Munro 1945, Parker 1985). Young loons have a diversified diet consisting primarily of small fish and minnows, aquatic insects and crayfish (McIntyre 1988a).

Winter foods are reported to include flounder (*Pleuronectoidei*), rock cod (*Gadus morhua*), herring (*Clupea* spp.), menhaden (*Brevoortia patronus*), sea trout (*Salmo* spp.), sculpin (*Leptocottus armatus*), and crabs (Palmer 1962, McIntyre 1988a). A detailed study of winter feeding patterns and preferences has not been conducted.

## Migration

Common loons migrate along several routes. Atlantic pathways are offshore, along the continental shelf from Nova Scotia to the Carolinas, and farther inshore, following the coast across Cape Cod (Powers and Cherry 1983). Band recovery data document inland pathways along a broad southeast/northwest front between the Atlantic Coast and central Canada, funneling primarily through Lake Michigan and Georgian Bay on Lake Huron (McIntyre 1988a). Some birds migrate along a north/south line through the central U.S. to and from the Gulf of Mexico (McIntyre 1988a). Loons breeding in Alaska and western Canada appear to move along the western edge of the Rocky Mountains en route to and from the Pacific Coast (McIntyre 1988a).

Both adults and juveniles typically congregate on staging areas on large lakes following the breeding season (e.g., Bull 1974, McIntyre and Barr 1983, McIntyre 1988a). Large concentrations also build on the Great Lakes and other inland lakes during the fall migration. Loons may linger on freshwater lakes until freeze-up before moving to maritime wintering sites (McIntyre 1988a). The spring migration is direct and closely follows the northward retreat of ice (McIntyre 1988a). Loons are diurnal migrants, and most flights, whether coastal or overland, appear to be initiated in the early morning (Williams 1973, Kerlinger

1982, Powers and Cherry 1983). The movements of juveniles during their 3-4 years as nonbreeders are not well understood. Most appear to remain on the coast, but some may move hundreds of miles northward, possibly tracking fish movements (McIntyre 1988a).

## Wintering Biology

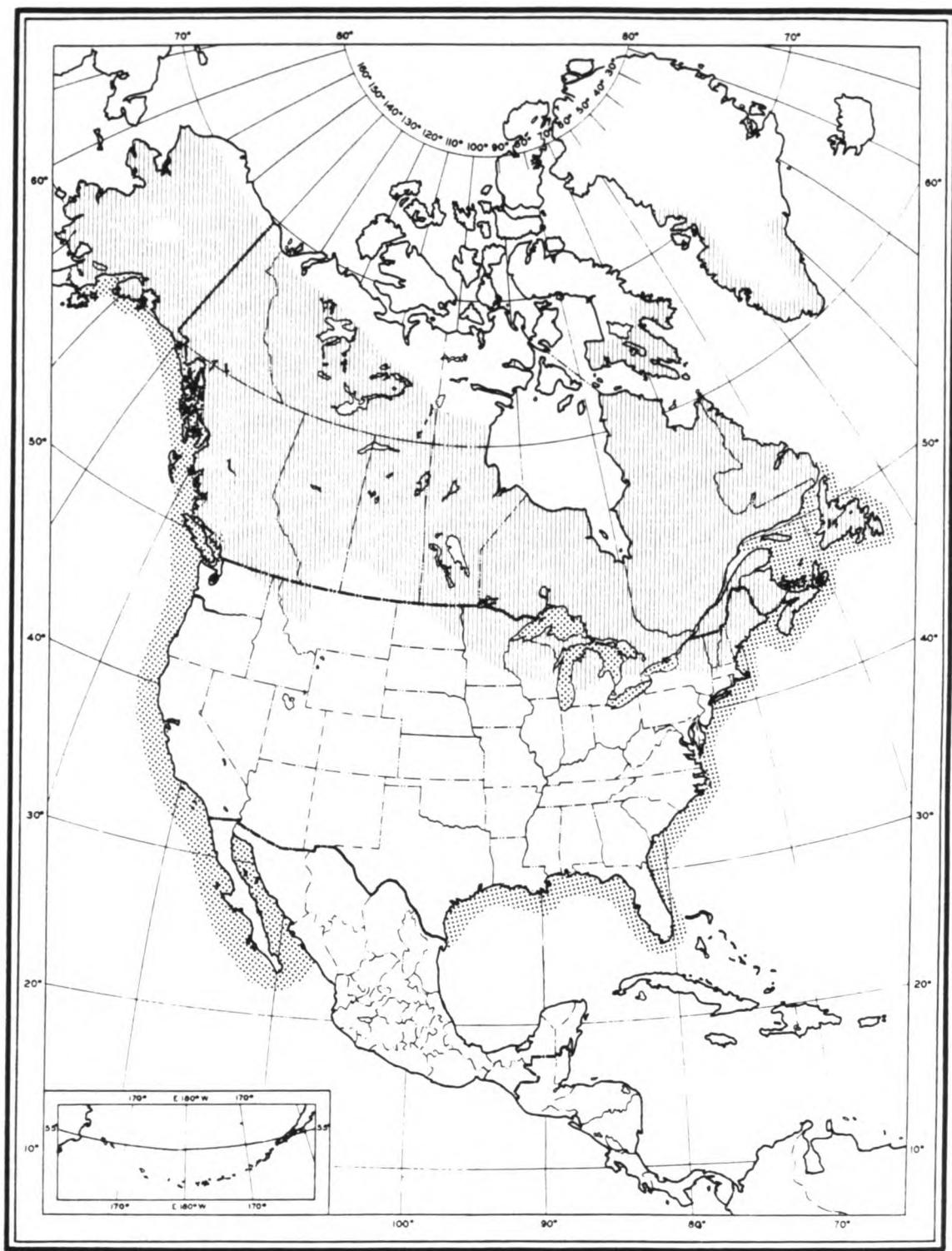
The ecology of wintering common loons has not been well studied. McIntyre (1978) found that loons off the Virginia coast maintained individual feeding territories of 4-8 ha during the day and rafted together at night. Activity patterns were significantly correlated with tidal changes. Maintenance behavior was greatest during the mid-period of tidal rise. Feeding activities peaked late in the flood tide and during the first half of the ebb tide. In Rhode Island, no winter feeding territories, feeding assemblages, or tide-correlated activity patterns were noted by Daub (1989).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

Common loons currently breed from a northern limit of western and central Alaska (south of the Brooks Range), northern Yukon, northwestern and southern MacKenzie, central Keewatin, northern Manitoba, northern Ontario, southern Baffin Island, northern Quebec, Labrador and Newfoundland, south to northeastern Washington, northwestern Montana, eastern Idaho, northwestern Wyoming, northeastern North Dakota, central Minnesota, central Wisconsin, central Michigan, southern Ontario, northern New York, central Massachusetts and Nova Scotia (Figure 1). The nesting range has contracted northward within the past 100-150 years from historic southern limits in northern California, northern Iowa, northern Illinois, northern Indiana, northern Ohio, northern Pennsylvania and southern New England. The species also breeds along the western and southern coasts of Greenland, in

Figure 1. The North American range of the common loon (*Gavia immer*) (American Ornithologists' Union 1983).



Breeding

Wintering

Iceland, and in Scotland (1970 record) (American Ornithologists' Union 1983, McIntyre 1988a).

Wintering in North America occurs along all three coasts, along the Pacific Coast from the Aleutians south to Baja California and Sonora, and along the Atlantic and Gulf Coasts from Newfoundland south to south Florida and west to southern Texas. Some common loons winter along the eastern Atlantic Coast from Norway south to northwestern Africa (Algeria), with the bulk of them being reported off Great Britain (American Ornithologists' Union 1983, McIntyre 1988a).

Common loons are ranked globally as G5 (demonstrably secure) (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

Although no precise continentwide estimate of common loon populations is available, some 500,000-600,000 adults probably inhabit the U.S. and Canada (J. McIntyre pers. comm.). Several states and provinces have begun annual breeding surveys (summarized in McIntyre 1988a). Estimated numbers of adults south of Canada number close to 18,000 and Alaska is thought to support about 34,000 common loons. Canada may be inhabited by half a million or more birds.

In the heart of their breeding range, common loon populations appear to be stable or increasing, although not without threats (McIntyre 1988a). Breeding Bird Survey (BBS) data from 1966-89 indicate a significant positive trend in loon numbers, both across the North American continent and in eastern North America (Table 1, Office of Migratory Bird Management unpubl. data, Laurel, Maryland). However, a northward range contraction has been documented within the last 100-150 years, and several states that once supported breeding loons have lost them (McIntyre 1988a). In response to these losses, the U.S. Fish and Wildlife Service considers the common loon a *migratory nongame bird of management concern* (U.S. Fish and Wildlife Service 1987).

Although recent population declines have been reported in several states, particularly in the Northeast Region, some recovery has occurred within the past decade, due primarily to focused conservation efforts. The species' current breeding distribution in the northeastern U.S., as reflected by Breeding Bird Atlas (BBA) data, is

**Table 1. Mean number of common loons (*Gavia immer*) per Breeding Bird Survey route and median percent annual change, 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).**

Region	No. routes <sup>a</sup>	$\bar{x}$ birds/route	% annual change <sup>b</sup>
<b>State</b>			
Maine	35	1.07	-5.7
New Hampshire	10	0.12	+4.3***
Vermont	0	-	-
Massachusetts	0	-	-
Rhode Island	0	-	-
Connecticut	0	-	-
New York	15	0.26	+4.3
Pennsylvania	0	-	-
New Jersey	0	-	-
Delaware	0	-	-
Maryland	0	-	-
Virginia	0	-	-
West Virginia	0	-	-
Eastern N. America	279	1.18	+1.8**
U.S.	159	0.33	+1.3
Canada	229	1.03	+2.3***
North America	388	0.82	+2.1***

<sup>a</sup>Number of routes on which common loons were detected (population trends may be invalid for states with < 10 routes).

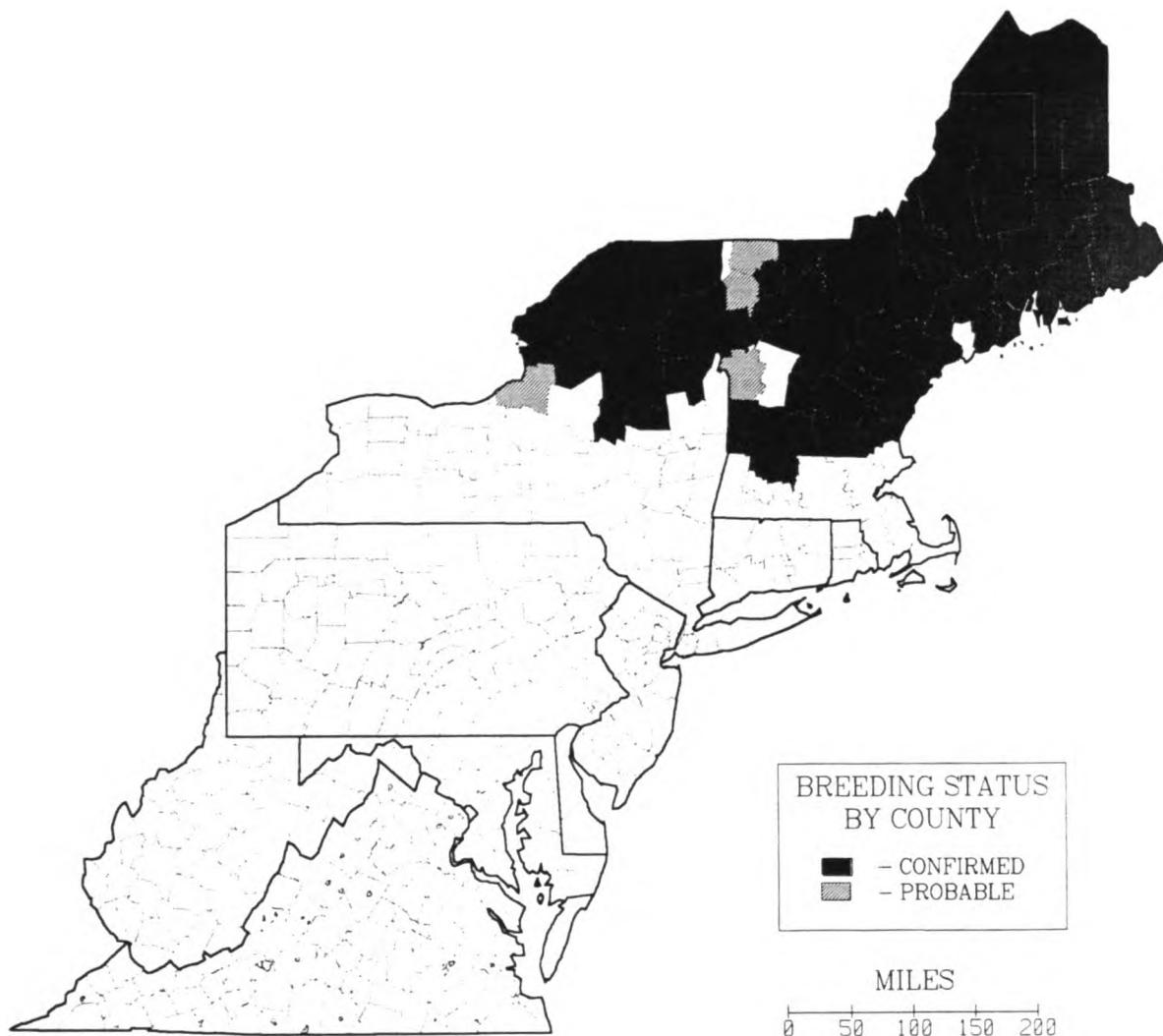
<sup>b</sup>Statistical significance of trend: \*\*  $P < 0.05$ , \*\*\*  $P < 0.01$ .

shown in Figure 2. Within each state, the common loon's legal status ranges from *endangered* to unlisted (although protected under migratory bird laws), and the state Natural Heritage Program ranks, which are standardized among states, range from S1 to SN (Table 2, The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

#### Distribution and Status in the Northeast Region

**Maine.** - Historic loon populations in Maine are imprecisely documented, but undoubtedly exceeded the fewer than 1,000 nesting pairs estimated by Knight (1908). Breeding loons are

**Figure 2.** Northeast breeding distribution of the common loon (*Gavia immer*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Kidd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



Iceland, and in Scotland (1970 record) (American Ornithologists' Union 1983, McIntyre 1988a).

Wintering in North America occurs along all three coasts, along the Pacific Coast from the Aleutians south to Baja California and Sonora, and along the Atlantic and Gulf Coasts from Newfoundland south to south Florida and west to southern Texas. Some common loons winter along the eastern Atlantic Coast from Norway south to northwestern Africa (Algeria), with the bulk of them being reported off Great Britain (American Ornithologists' Union 1983, McIntyre 1988a).

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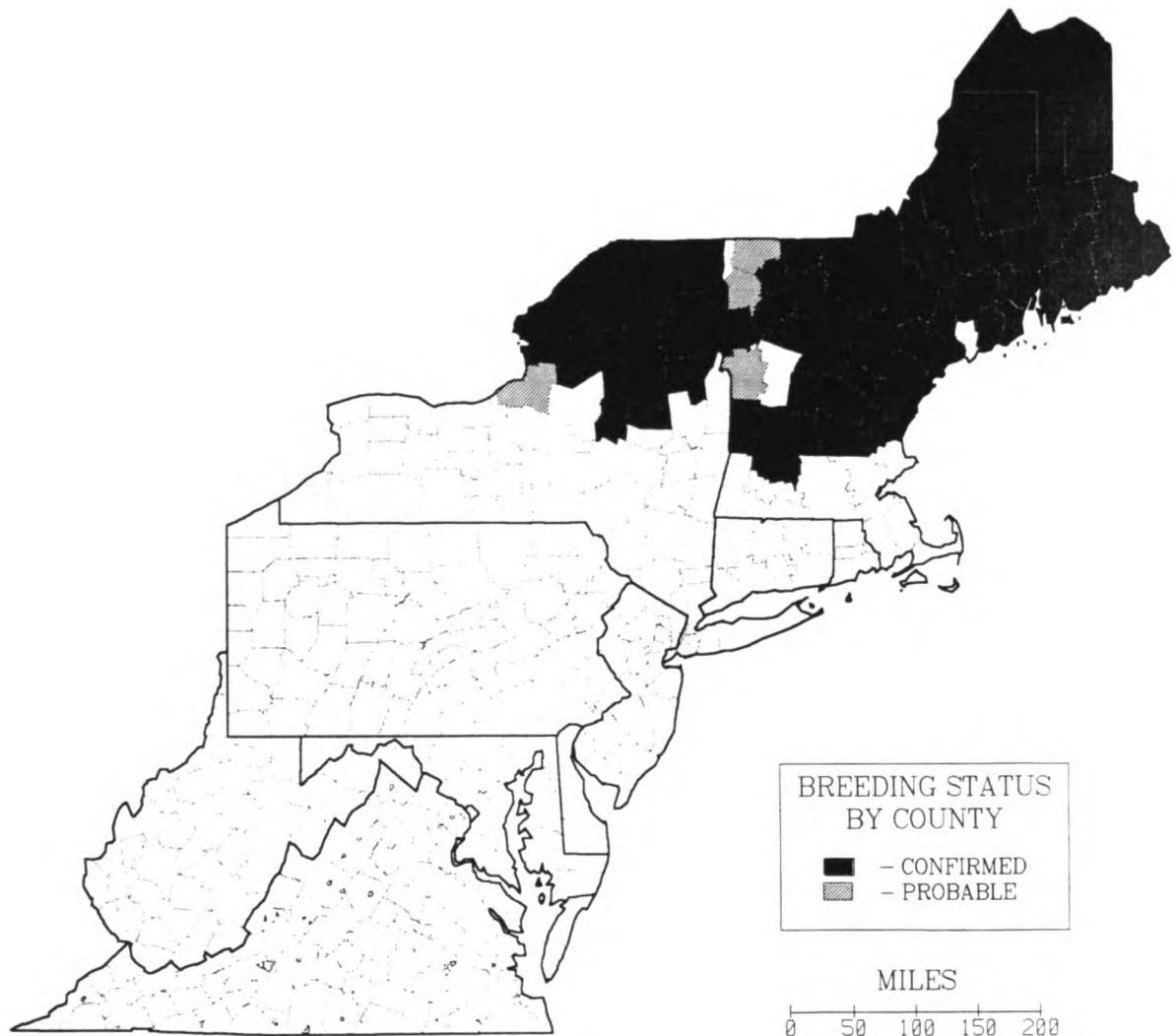
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**Table 2. Summary of state listing status and Natural Heritage Program state ranks for the common loon (*Gavia immer*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	S4
New Hampshire	threatened	S3
Vermont	endangered	S2
Massachusetts	special concern	S1
Rhode Island	-	SN
Connecticut	special concern	S1
New York	special concern	S4
Pennsylvania	-	SH
New Jersey	-	SN
Delaware	-	SN
Maryland	-	SN
Virginia	-	SN
West Virginia	-	SN

<sup>a</sup>SH = historical records only, but suspected extant

SN = regularly occurring but not breeding

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

distributed throughout the state, but are concentrated in northern regions (Arbuckle and Lee 1985) and nest only rarely along the coast (Palmer 1949; Maine Audubon Society 1989, 1990). Although no clear population trends are evident, development pressures in southern Maine appear to be threatening suitable habitat there (Christenson 1981) and may have caused a recent decline in the nesting population (Cross 1979, Strong and Bissonette 1985).

Survey efforts were initiated jointly by the Maine Audubon Society and the Maine Department of Inland Fisheries and Wildlife in 1977. A variety of survey techniques have been used, including game warden surveys in 1977 and 1978 (Cross 1979, Sawyer 1979), aerial and ground censuses of a stratified random sample of 200 lakes in 1983-85 (Arbuckle and Lee 1985), an annual volunteer loon count since 1983 (Lee and Arbuckle 1988), and an annual wintering loon

count in 1983-85 (Lee and Arbuckle 1988). Data collected in 1983-85 yielded a mean, statewide estimate of 3,498 adult loons, with 2,023 in northern Maine and 1,475 in the south (Arbuckle and Lee 1985). The 1990 statewide population estimate of 3,949 adults (1,828 in northern Maine and 2,121 in the south) indicates a steady recovery of the southern population (Stockwell 1990). Loon densities in southern Maine, calculated as the number of loons per 40 ha of surveyed lake surface, ranged in 1987 from 0.076 loons per 40 ha in Cumberland County to 1.12 in Lincoln and Ward counties (Lee and Arbuckle 1988). BBS data from 1966-89 point to a nonsignificant statewide decline in loon numbers (Table 1), but this may reflect an artifact of sampling technique more than an actual population decline.

Data on productivity are not extensive. Of 106 nests monitored in Maine in 1987 and 1988, chicks hatched from 71 (70%) (Lee and Stone 1988, Jacobs and Stockwell 1989), but fledging success was not documented. Strong (1985) documented hatching success at only 55% of nest sites on lakes with low human use. Sawyer (1979) reported an average of 1.38 chicks observed per reported nesting pair on a sample of 166 lakes, but cautioned that the data were imprecise and probably yielded overestimates.

Maine Audubon Society counts of wintering common loons along the coast ranged from 284-478 individuals in 1983-85. The areas of highest concentration were Saco Bay, Port Clyde to Spruce Head, and Mount Desert Island (Lee and Arbuckle 1988). National Audubon Society Christmas Bird Counts (CBCs) from 1984-88 averaged 556 individuals (range = 425-716), with highest densities typically on the Biddeford-Kennebunkport, Portland, Bath-Phippsburg, and Mount Desert counts (National Audubon Society 1984-88). Scattered individuals were regularly reported from inland locations during the 5 years.

**New Hampshire.** - Estimates of historic loon populations in New Hampshire are not precise, but a 1977 survey of 67 lakes with definite evidence of former loon nesting revealed only 34 lakes with breeding pairs, a 53% decline (Sutcliffe 1980). Unrestricted shooting near the turn of the century, followed by the direct and indirect impacts of increased human settlement, are

thought to have caused this decline (Sutcliffe et al. 1981). Many of the abandoned lakes in 1977 were located in the southern half of the state, in areas of more intense development pressure (Sutcliffe et al. 1981). Breeding loon pairs are currently distributed throughout New Hampshire, but are concentrated in the central and northern lakes regions and largely absent from higher altitudes (J. S. Fair pers. comm.).

Annual statewide surveys of the breeding population were initiated by the Loon Preservation Committee in 1976 (Sutcliffe 1979a). Survey data from the first 5 years indicated a stabilizing loon population of between 59-67 nesting pairs, with small increases in hatching and fledging success. The breeding population has since increased steadily to a record high of 105 nesting pairs on 60 lakes in 1990 (Fair 1990). Numbers of territorial, nonbreeding pairs and unpaired adults have also increased, and lake recolonization has shown a positive trend, from 58 lakes with territorial loon pairs in 1976 to 100 in 1990 (Fair 1990). These data suggest possible recruitment of young loons into the summering population (Wood et al. 1985). The total statewide population has increased 95% from its lowest point of 254 birds in 1978 to 495 in 1990 (Fair 1990). This increase is reflected in BBS data from 1966-89 which indicate a significant increase of 152% in loon numbers during the period (Table 1).

Productivity has also increased overall, although totals are slightly down from the peak year of 1983. An average of 51.6 chicks (range = 36-76) were fledged in 1976-82, while 71.9 (range = 60-83) fledged on average in 1983-90 (Fair 1990). Over the 15-year survey period, annual chick survival has averaged 79%, with 0.78 chicks fledging per nesting pair. Egg predation by raccoons (*Procyon lotor*) and unnatural water level fluctuations are primary causes of breeding failure. Lead poisoning from ingestion of fishing sinkers, entanglement in monofilament fishing line, and ingestion of fishing lures have been identified as causes of adult mortality in about one-half of the carcasses examined (Wood et al. 1985). Artificial nesting islands (rafts) have been used with great success in New Hampshire and have accounted for 20% of total chick production

(annual range = 5-32%) since their introduction in 1977 (Fair 1990).

The coastal New Hampshire CBC averaged 58 common loons (range = 19-144) from 1984-88 (National Audubon Society 1984-88). Excluding the high count in 1986, the mean number reported was 36 (range = 19-59). An average of 3.6 loons (range = 1-6) was recorded at inland locations during the 5 years.

**Vermont.** - Few historical data are available on breeding loon populations in Vermont. A number of lakes known to have formerly supported nesting loons, some within the last decade, are now unoccupied (Metcalfe 1979, Rimmer and Kaveney 1988). Breeding pairs are distributed primarily in the northeastern region, with a single pair consistently nesting on a large reservoir in southern Vermont (Metcalfe 1979, Hall and Rimmer 1990).

Surveys of the statewide breeding population have been conducted by the Vermont Institute of Natural Science since 1978. Thirteen years of data have documented an initially stable population of 15-19 known nesting pairs, a sharp decline to 8 pairs in 1983 and 1984, and a subsequent gradual recovery to 14 breeding pairs in 1989 and 1990 (Rimmer 1988, Kaveney and Rimmer 1989, Hall and Rimmer 1990). Population means for the 13-year survey period are 19.1 territorial pairs (24.5 in 1978-82, 16.3 in 1983-90) and 13.0 nesting pairs (17.0 in 1978-82, 11.0 in 1983-90). The statewide population, as estimated by an annual July census, has more than doubled since 1983 (127 birds in 1990). Increased numbers of nonbreeding adults and summering immatures are largely responsible for this increase, suggesting higher rates of immigration or recruitment (Rimmer 1988, Hall and Rimmer 1990).

Survival of Vermont loon chicks is high and has ranged from 81-92% in the 7 years for which data are available (Hall and Rimmer 1990). Productivity is also high, averaging 0.98 fledged chicks per nesting pair over the 13-year period, and has increased from 0.79 in 1978-82 to 1.07 in 1983-90 (Hall and Rimmer 1990). Most breeding failures occur during incubation (Rimmer 1988). Of 52 known nest losses between 1978-90, 22 (42%) resulted from water level fluctuations, and

12 (23%) were caused by predation, mostly by raccoons. The remaining nest failures resulted from human disturbance (8%) and unknown factors (27%) (Hall and Rimmer 1990).

Human-related threats to Vermont's breeding loons include increasing shoreline development, disturbance from boaters and fishermen, water level fluctuations, and raccoon predation. Severe intraspecific competition appeared to cause two breeding failures in 1989 and suggests that suitable (or optimal) loon habitat may somehow be limited (Kaveney and Rimmer 1989).

Lake Champlain is an important fall staging ground and stop-over site during the fall migration. Data from the *Records of Vermont Birds* (Vermont Institute of Natural Science unpubl. data) show peak concentrations of 361 and 247 common loons on northern Lake Champlain in early November of 1987 and late October of 1986, respectively. These probably represent a small percentage of the total loon population using Lake Champlain during peak migration periods. CBC data from 1984-88 show an average of 12.4 loons (range = 8-15) lingering on Lake Champlain in late December, mainly between Ferrisburg and Burlington. Only one individual was reported away from Lake Champlain during the 5-year period (National Audubon Society 1984-88).

**Massachusetts.** - Although probably never a common breeding species in Massachusetts, common loons are known to have nested in several counties (principally northern Worcester County) prior to 1900 (Blodget and Lyons 1988). No historic population estimates are available, but the species was extirpated by 1900, due apparently to direct human persecution (Bent 1919, McIntyre 1988a) and increased human pressures on breeding lakes. Although breeding was reported in Franklin County in 1943 (Allison and Allison 1943), nesting was not documented again until 1975, when a pair successfully raised two chicks on Quabbin Reservoir (Clark 1975, LaFontaine 1976).

Massachusetts' recovering loon population has been monitored since 1975, excluding 1978-80 when no data were collected (Blodget and Lyons 1988). Concerted survey and management efforts

were launched in 1983 by the Metropolitan District Commission, the Massachusetts Division of Fisheries and Wildlife, and the Massachusetts Audubon Society (Lyons and Erhardt 1984). Survey data indicate a single breeding pair at Quabbin Reservoir from 1975-77, followed by a gradual increase to five nesting pairs in 1984, including one on Wachusett Reservoir. Only three of seven territorial pairs nested in 1985, but the breeding population quickly rebounded and increased to nine pairs by 1987 (Blodget and Lyons 1988). In 1990, nine territorial pairs occupied four Massachusetts lakes, with Quabbin Reservoir (ca. 10,000 ha) supporting six pairs. Of these nine pairs, six nested and produced seven surviving chicks (B. Blodget pers. comm.). Blodget and Lyons (1988) suggested that this solid reestablishment may be the result of colonizing birds from stable or increasing populations further north.

Annual productivity has varied from 0.0 fledged chicks per nesting pair in 1982 to 1.67 in 1981; both were years in which 3 loon pairs nested (Lyons 1987). More recently, fledging rates have ranged from 0.50 chicks per nesting pair in 1986 to 1.17 in 1990. Since 1975 (excluding 1978-80), 0.87 chicks have fledged per nesting pair. Chick production peaked at nine fledglings in 1988 (B. Blodget pers. comm.). Most breeding failures have been attributed to water level fluctuations and predation by raccoons or American crows (*Corvus brachyrhynchos*). Direct human disturbance does not appear to be a serious problem at present. Rafts have been used successfully at Quabbin and Wachusett reservoirs, where loons have produced eight chicks from six platform nests since 1986 (Blodget and Lyons 1988; Monette 1988, 1989).

Based on CBC data, Massachusetts supports the largest wintering common loon population in Region 5 (U.S. Fish and Wildlife Service). Between 1984-88, a mean of 1,103 individuals (range = 819-1,736) was reported, nearly all from coastal locations (National Audubon Society 1984-88). Martha's Vineyard consistently supported the highest concentrations, with a mean of 477 birds (range = 292-630), followed by Nantucket Island, which reported the highest single count of 863 individuals in 1984. Offshore counts on Stellwagen Bank in 1987 and 1988

yielded 10 and 15 common loons, respectively. Scattered individuals were regularly reported from inland locations, with two on Quabbin Reservoir in 1988.

**Rhode Island.** - There appear to be no historic or recent records of nesting common loons in Rhode Island. It is possible that loons nested at low densities prior to extensive human settlement (e.g., Arbib 1963). CBC data indicate that the species is fairly common along the coast in winter. Counts between 1984-88 averaged 221 individuals (range = 134-364), with highest densities in the Point Judith-Narragansett area (National Audubon Society 1984-88).

**Connecticut.** - The history of common loons in Connecticut is not well documented. Limited breeding is reported to have occurred historically (Merriam 1877, Sage et al. 1913, Bent 1919, Bevier in press) with the last record near the West Haven-Milford border in 1890 (Sage et al. 1913). The Connecticut BBA project did not confirm breeding from 1982-86, although probable evidence was obtained in Litchfield and Hartford counties (Bever in press). Two reported nesting pairs from Litchfield County in 1977 (Vickery 1977), and reports from Norfolk in 1978 and 1985 are now believed to be misidentifications of other species (Bever in press). Adult common loons continue to summer on several of the larger lakes in northwestern Connecticut, but nesting has yet to be confirmed (Bever in press). Although suitable breeding habitat appears to exist (Bever in press), recreational use of lakes may inhibit expansion of the nesting range into Connecticut (Zeranski and Baptist 1990).

Common loons are fairly common migrants in coastal waters during spring and fall, and uncommon to fairly common inland, principally on larger lakes and rivers. Loons are rare in winter on western Long Island Sound but gradually increase eastward to become fairly common (Zeranski and Baptist 1990). CBC data from 1984-88 indicate a mean wintering population of 43 individuals (range = 23-56), with only a few scattered inland reports (National Audubon Society 1984-88).

**New York.** - New York's common loon population has been restricted primarily to the Adirondack region since the early 1900s (Arbib 1963, Peterson 1988). Historic records are incomplete but indicate that the statewide range and population levels have declined in the wake of increased human activities. In the late 1970s McIntyre (1979) documented that there were about 35% fewer occupied lakes than at the turn of the century.

The New York BBA Project located common loons in 369 5x5-km blocks, confirming breeding in 152 (41%) of these and finding probable evidence for nesting in 74 (20%) (Peterson 1988). Distribution was confined to the northern part of the state, primarily the Adirondack region, with additional breeding pairs also found in the St. Lawrence region (Parker et al. 1986, Peterson 1988).

Arbib (1963) recorded the first population estimate of common loons in New York. He reported 120 pairs nesting at more than 90 locations and estimated the statewide breeding population at about 240 pairs. A later field survey of 420 lakes larger than 10 ha between 1977-79 located 114 breeding pairs on 91 lakes and yielded an estimate of fewer than 200 pairs statewide (Trivelpiece et al. 1979). An intensive follow-up survey of 557 lakes in 1984 and 1985 resulted in a count of 157 breeding pairs on 128 lakes (Parker et al. 1986, Parker and Miller 1988). An additional 247 nonbreeding adults were located, a significantly higher number than found on earlier surveys. Although these results did not show an increase in the total breeding population or its productivity, Parker et al. (1986) suggested that the overall loon population might be increasing and is at least stable. BBS data indicate a positive, although statistically insignificant, median annual percent increase for the period 1966-89 (Table 1).

New York's breeding loon population occurs at low density but is high in productivity (Trivelpiece et al. 1979, Parker et al. 1986). This corresponds to results from other regions, where loon populations at low densities often experience higher productivity than populations at high densities (McIntyre 1988a). Parker et al. (1986) reported a fledging rate of 0.96 chicks per nesting

pair in New York, one of the highest in North America.

Conservation concerns for breeding loons in New York include increased recreational pressures, shoreline development, heavy metal and toxin contamination, and lake acidification (McIntyre 1979, Parker et al. 1986). Although Parker et al. (1986) found no significant relationship between lake acidity and loon breeding status or success, Parker (1985, 1988) documented problems for adults feeding chicks on highly acidified Adirondack Park lakes.

Lake Ontario, the Finger Lakes and some Catskill lakes have been identified as important sites for migrating loons (Arbib 1963, Parker and Milligan 1987). Bull (1974) lists common loons as common to abundant migrants, both on the coast and on larger inland bodies of water. As many as 700 loons have been observed on Lake Ontario off Rochester in late fall, and 550 were recorded on Oneida Lake on 12 April 1960. Off Long Island, 470 common loons were observed from the Montauk area on 6 November 1949, and 300 were recorded off Long Beach on 11 April 1933 (Bull 1974).

Wintering populations are primarily coastal. CBC data from 1984-88 documented a mean of 495 individuals (range = 418-587) statewide (National Audubon Society 1984-88). The Montauk count reported between 50-76% of all birds in each year, with a maximum of 418 in 1987. Loons were reported on scattered inland lakes during each CBC period, with counts in the Ithaca and Rochester areas ranging between 2-5 and 1-18 individuals, respectively.

**Pennsylvania.** - Although the breeding history of common loons in Pennsylvania has not been well documented, at least three nesting records have been recorded since 1900, all from the Pocono region. In 1908, a pair in Bushkill was reported to have hatched chicks that failed to survive (Harlow 1908). Breeding was not confirmed again until 1981 and 1983 at the Penn Forest Reservoir, when a pair with flightless young was observed in both years (Street and Wiltraut 1985). Adults also summered there in 1984 and 1985 and were observed on Wild Creek Reservoir in 1984. Although Poole (1964) reported nesting at Long Pond and Pocono Lake,

and formerly on the Susquehanna River near Marietta in southern Pennsylvania, no published accounts of such records exist. There appears to have been no nesting by common loons in Pennsylvania since 1983, although scattered reports of summering birds persist in *American Birds*.

Common loons are regular spring and fall transients on larger bodies of water (Todd 1940, Poole 1964, Leberman 1976). Counts as high as 250 individuals have been recorded on Lake Ontelaunee in April (Poole 1964). Wintering loons are regular but uncommon. CBC data from 1984-88 indicate an average statewide population of 23 individuals (range = 11-39), with the most consistent concentrations in the Erie and York areas (National Audubon Society 1984-88).

**New Jersey.** - There are no documented historic records of common loons nesting in New Jersey. Although a pair was reported to breed at Lake Hopatacong in the late 1800s (Stone 1894), this record was never confirmed (Stone 1908). Fables (1955) reported that a few nonbreeding individuals regularly summer in the northern part of the state. Loons are common migrants along the coast and on Delaware Bay, and less common inland (Stone 1894, Fables 1955). They regularly occur along the coast in winter, with CBC data from 1984-88 indicating a mean of 68 individuals (range = 45-133), scattered along the coast and occasionally reported from inland sites (National Audubon Society 1984-88).

**Delaware.** - There are no published accounts of historical nesting by common loons in Delaware. The species is an uncommon to fairly common transient and winter visitor, primarily along the coast (Hess et al. in press). Winter numbers are variable, and no trend is detectable during the past 20-25 years. CBC data from 1984-88 indicate a mean statewide wintering population of 20 individuals (range = 12-34), with highest densities consistently in the Rehoboth area (National Audubon Society 1984-88).

**Maryland and the District of Columbia.** - There are no published records of nesting by common loons in Maryland or the District of Columbia. Stewart and Robbins (1958) list the species only

as casual in nearshore coastal waters during summer. Loons are common coastal migrants, and uncommon to fairly common transients at inland freshwater sites. Maximum counts in spring have included 90 at Point Lookout, and in fall, 200 in the Ocean City area (Stewart and Robbins 1958). Common loons are fairly common during winter in southern coastal areas, uncommon along the northern coasts and in the upper Chesapeake Bay, and casual in the interior (Stewart and Robbins 1958). CBC data from 1984-88 indicate a variable early winter population averaging 117 individuals (range = 65-224), with Ocean City consistently recording the highest count (mean = 72 individuals, range = 37-156) (National Audubon Society 1984-88). Scattered individuals are reported from inland lakes and reservoirs. The CBC in Washington, D.C. recorded up to three common loons between 1984-88.

**Virginia.** - Common loons are not known to have nested historically in Virginia. Loons are occasional in summer on the coast, and common during migration and in winter. The species is uncommon as a transient and winter visitor inland, and rare west of the Blue Ridge (Murray 1952). Virginia supports a fairly large wintering loon population, as indicated by CBC data. Between 1984-88, an average of 628 common loons (range = 364-994) was recorded, with peak concentrations typically off Cape Charles and Chincoteague (National Audubon Society 1984-88). Exceptionally high, recent CBCs have recorded 641 at Chincoteague in 1970, and 612 on the Back Bay National Wildlife Refuge count in 1986. Individuals are regularly reported on inland lakes in winter, with 11 on the CBC at J.H. Kerr Reservoir in 1988.

**West Virginia.** - There are no published records of nesting common loons in West Virginia. The species is an uncommon to common spring migrant and an uncommon to very common fall transient (Hall 1983). Although migrating throughout the state, common loons are most numerous in northern West Virginia. Over 1,000 birds have been recorded in a single day on Cheat Lake during fall (Hall 1983). Loons are rare winter visitors where open water persists.

CBC data between 1984-88 indicate a mean of only 2 individuals statewide (range = 0-5) (National Audubon Society 1984-88). Hall (1983) reported that common loons have declined drastically in recent years for unknown reasons. Although "frequent and numerous" as migrants until the 1960s, common loons are now rarely observed.

### Summary

Although the historic nesting range of common loons in the Northeast has retreated northward since 1900, recolonization of some former breeding areas appears to be underway. Loon populations are stable or slowly increasing in New York and the northern New England states, in an apparent reversal of marked declines noted only 10-15 years ago. Massachusetts now supports nine nesting pairs after a long absence, and confirmation of breeding seems imminent in Connecticut. Based on CBC data, coastal wintering populations, while variable from year to year in a given area, show no detectable upward or downward trends.

## LIMITING FACTORS AND THREATS

### Habitat Loss and Degradation

Direct and indirect effects of shoreline development may reduce the suitability of lakes for nesting loons. Although radical shoreline alteration and cottage construction appear to only rarely inhibit loon nest site selection, increased human activity around developments often does (McIntyre 1988a). In New Hampshire, a state that has experienced heavy lakeshore development and increased recreational use, a 1979 survey indicated a 53% decline in lake utilization by loons over 50 years (Sutcliffe 1980). In Wisconsin, the density of houses on lakes with breeding loons was lower than on lakes not used by loons (Zimmer 1979, in Strong and Bissonette 1985). Hatching success decreased as the number of cottages within 150 m of loon nests increased on lakes in central Ontario (Heimberger et al. 1983). McIntyre (1988b) found that the number

of lakes with territorial loons decreased with increasing shoreline development and recreational use in Minnesota between 1971-86.

Acid precipitation may reduce the quality of loon nesting lakes. Alvo et al. (1988) monitored reproductive success on 68 small lakes (5.3-75 ha) near Sudbury, Ontario that varied in pH from 4.0-8.4. Nesting success was higher on lakes with high alkalinity, which was negatively correlated with pH, than on low-alkalinity lakes. Unsuccessful breeding resulted primarily from brood mortalities on acidic lakes and appeared to result from lower prey fish densities. One pair with a chick on a fishless, acidic lake fed the chick benthic algae, and possibly invertebrates, but fed themselves on other lakes. The chick died at 1-4 weeks of age. On acidified lakes in New York's Adirondack Park, chicks were fed prey much smaller or larger than those typically preferred (Parker 1988). On one fishless, low-pH lake, adults fed aquatic insects to their chick in lieu of small fish and spent two to four times longer feeding the chick than did loons on lakes with fish. Although loon nesting success was not correlated with lake acidity in this study, potentially severe feeding problems for chicks were created by reduced prey diversity and quality (Parker 1988).

Oil spills pose a serious, although localized, threat to common loon habitat. Most spills have occurred on marine wintering areas, and reports of loon mortalities from coastal spills are common (McIntyre 1988a). Of 387 loon carcasses collected within 6 months of the 1989 Exxon Valdez oil spill in Alaska, 203 were identified as common loons (Tankersley unpubl. data). Spills on inland waters, particularly on staging grounds, could be disastrous to migrating loons (McIntyre 1988a).

Water-level fluctuations resulting from human-made dams can also reduce the suitability of a lake for breeding loons. Fair (1979) documented nest abandonment and predation following lake drawdowns in New Hampshire. On Stillwater Reservoir in New York, McIntyre (1988b) found that loons lost nests to inundation when water levels rose more than 20 cm. Breeding loons in an area of regulated water levels in Minnesota had significantly lower productivity than other populations on naturally fluctuating lake systems (Reiser 1988). Nesting

may be delayed by water levels that fail to recede after snowmelt (Fair 1979, Strong 1985). Although poorly regulated lake levels can lead to nest losses, creation of reservoirs has increased the availability of suitable loon nesting habitat in some areas (e.g., McIntyre 1988b).

### Human Disturbance

Recreational pressures may have contributed to declines in some loon populations, but loons generally can acclimate to moderate recreational lake use. While Ream (1976) suggested that disturbance of nest sites by canoeists in the Boundary Waters Canoe Area (BWCA) in Minnesota was the primary factor limiting loon productivity, Titus and VanDruff (1981) later found few negative impacts from recreational activities in the BWCA. However, territory locations had shifted from island centered sites to more peripheral ones, a response, they suggested, to competition with campers for centrally positioned islands. They also noted that productivity was higher on lakes with motor boat prohibition and low human use than on lakes with motor boats and high levels of human use. Smith (1981, cited in McIntyre 1988a) reported identical productivity on both remote lakes and on lakes with established canoe routes in Alaska. In Maine, no significant difference in breeding success was found for loons on high human-use versus low human-use lakes (Christenson 1981). However, loon densities may be lower on heavily developed than on relatively undeveloped lakes (McIntyre 1988a).

When incubating loons leave nests because of disturbance, they may not return for an hour or more, leaving the eggs vulnerable to predation and cooling (McIntyre 1975, Titus and VanDruff 1981). Loons exhibit behavioral modifications in response to moderate recreational activity on many lakes. On high human-use lakes, loons flush at shorter distances (Smith 1981, cited in McIntyre 1988a; Titus and VanDruff 1981), flush less readily and less vigorously, vocalize less once flushed, and return to the nest more quickly than loons on remote lakes (Titus and VanDruff 1981).

Motorboats may impact loons more negatively than canoes overall, depending on the stage of the loon's breeding cycle. On BWCA

lakes, motorboat use is heaviest early in the season, when loons are nesting, while canoe use peaks in August after the nesting season (McIntyre 1988a). Loons are more easily able to avoid canoes than motorboats, and chicks are less likely to be separated from their parents by canoes. Separation can lead to chick mortality from predators or neighboring territorial loons (McIntyre 1988a). Motorboat wakes in combination with high water levels may cause nest destruction (Vermeer 1973b).

### Predation

The major predators on loon nests and chicks include scavengers such as American crows, common ravens (*Corvus corax*), herring gulls (*Larus argentatus*) and raccoons, all of which have increased due to the proliferation of garbage dumps and other human refuse (McIntyre 1988a). Disturbance that flushes incubating loons may increase the likelihood of egg destruction by watchful avian predators (McIntyre 1988a). Raccoons caused 75-80% of nest losses on New Hampshire's two largest lakes in 1977 (Sutcliffe 1980). Other predators include snapping turtles (*Chelydra serpentina*), northern pike (*Esox lucius*), muskellunge (*Esox masquinongy*), walleye (*Stizostedion vitreum*), red fox (*Vulpes fulva*), mink (*Mustela vison*), skunk (*Mephitis mephitis*), and occasionally other mammalian carnivores (Olson and Marshall 1952; McIntyre 1975, 1988a). Otters (*Lutra canadensis*) and bald eagles (*Haliaeetus leucocephalus*) have been reported as possible predators (Titus and VanDruff 1981, McIntyre 1988a). Harassment or disturbance by herring gulls, beavers (*Castor canadensis*) and muskrats may lead to nest abandonment (Titus and VanDruff 1981). Predation on adult loons appears to be rare, although an unusual attack by a coyote (*Canis latrans*) was reported (Barklow and Chamberlain 1984), and a recent report of predation on seabirds by sea otters (*Enhydra lutris*) listed three predation events on common loons (Riedman and Estes 1988).

### Diseases and Parasites

Common loons are susceptible to epidemics of both type C and type E botulism (McIntyre 1988a). Type E botulism has killed up to 3,570 loons on Lake Michigan in a single year (Fay 1966, in McIntyre 1988a), probably through ingestion of alewives during migration. Loons are also susceptible to aspergillosis from airborne *Aspergillus* spp. spores, which destroy the functioning of air sacs, particularly in stressed birds (Okoniewski and Stone 1987, McIntyre 1988a). An extensive list of internal parasites has been documented in common loons (McIntyre 1988a). Loon carcasses in emaciated condition from a massive die-off along the Gulf coast in 1983 had abnormally high numbers of microphallid trematodes (flukes), as well as tapeworms, spiny-headed worms, renal trematodes and renal coccidia (Stroud and Lange 1983). These parasites are believed to have caused hemorrhagic enteritis and contributed to the pronounced emaciation of many dead birds. Loons are afflicted by a host-specific black fly (*Simulium euryadminiculum*), which may act as a vector and transmit a blood parasite (McIntyre 1975, 1988a).

### Competition

Intraspecific competition may limit productivity. Sibling aggression can be severe, especially during food shortages, and may result in the death of the subordinate, usually younger, chick (Dulin 1987, cited in McIntyre 1988a). Chicks that wander into adjacent territories may be killed by neighboring adult loons (McIntyre 1988a). Severe fighting by adults has been documented, presumably over territorial ownership, and can lead to injury or nest abandonment (e.g., Kaveney and Rimmer 1989). Competition with aggressive, non-native mute swans (*Cygnus olor*) has been documented in Michigan (McPeek and Evers 1989).

## Entanglement in Fishing Lines, Nets and Traps

Loon mortality is known to occur from entanglement in monofilament sports fishing line and in commercial fishing nets. In the Northwest Territories, 5,662 loons of all 4 species--common, yellow-billed, Pacific (*G. pacifica*), and arctic loons (*G. arctica*)--were reported to have drowned in gill nets between mid-May and mid-June in 1960 and 1961 (Vermeer 1973b). Six of 29 common loons necropsied in New York between 1972-86 drowned in gill nets on Lake Ontario (Okoniewski and Stone 1987). Commercial fish traps and nets in the Great Lakes pose a serious, although unquantified, threat to loons (McIntyre 1986, 1988a). Loons are also caught in nets used during coastal fishing operations (McIntyre 1978). Most mortality from these sources undoubtedly goes unreported.

## Environmental Pollutants

Organochlorines and their residues have been detected in common loon eggs and carcasses. DDE levels in loon tissue from Minnesota in the 1960s may have had adverse, sublethal effects (Ream 1976, McIntyre 1988a). Loon eggs with higher levels of DDE residues tend to have thinner shells than eggs with lower residue levels (Vermeer 1973b, McIntyre 1975, Sutcliffe 1978, Fox et al. 1980), although no studies have demonstrated evidence of shell breakage. There appears to be no documentation of lowered productivity in loons as result of elevated pesticide loads (Fox et al. 1980), and organochlorine levels have generally declined in loon tissue in recent years (Frank et al. 1983, cited in McIntyre 1988a).

Heavy metal contaminants may pose the most widespread, irreversible and deadly threat to loons (McIntyre 1988a). Methylmercury poisoning has been implicated in lowered productivity (Barr 1986) and winter mortality (Stroud and Lange 1983, Alexander 1985). Mercury is released into the environment during the operation of chlor-alkali and wood pulp plants, and through treatments of agricultural seeds (McIntyre 1988a). Lake acidification may

accelerate the release of mercury into the water column, hastening its uptake through the aquatic food chain (Barr 1986, McIntyre 1988a). In Ontario, Barr (1986) found significantly higher mercury residues and lower successful use of territories in loons on lakes within 160 km downstream of a chlor-alkali plant. High mercury levels in many necropsied loons following a large winter die-off along the Gulf Coast of Florida in 1983 (as many as 7,500 birds: Alexander 1985) may have contributed to the emaciation and subsequent death of these individuals. In New York, Okoniewski and Stone (1987) tentatively diagnosed mercury intoxication in 3 of 29 common loon carcasses examined between 1972-86.

Other heavy metals, such as lead, cadmium and selenium, are actual or potential hazards to loons (McIntyre 1988a). Lead poisoning from ingestion of lead fishing sinkers is implicated as a cause of loon mortality by Locke et al. 1982 and Okoniewski and Stone 1987. Recent post-mortem analyses (Pokras and Chafel in press) of 31 common loons from New England indicate that 16 birds (52%) had ingested lead sinkers. Eight of these 16 were observed pre-mortem, and displayed clinical signs of lead toxicosis such as grounding on land, weakness, gaping, and head tilting. Toxicological analyses of liver tissues indicated lead levels of 5.03-18.0 ppm ( $n = 4$ ) in birds with ingested lead sinkers, and < 0.05 ppm ( $n = 10$ , excepting one bird with 0.11 ppm) in birds which died from other causes.

## Direct Killing by Humans

Although the sport shooting that impacted loon populations around 1900 is now illegal, loons continue to be intentionally killed on occasion, primarily by sport and commercial fishermen who consider the birds to be direct competitors (McIntyre 1988a). Of 29 dead common loons necropsied in New York from 1972-86, 3 had been shot (Okoniewski and Stone 1987). Loons are still taken for food by Indians and Inuits. In northern Quebec, the annual harvest ranges from 2,500-6,500 loons, most of them common loons (Desgranges and Laporte 1979, cited in McIntyre 1988a). This harvest is thought to be too high to

support the region's current population of 12,000 pairs.

## MANAGEMENT POTENTIAL

The ability of common loons to habituate to moderate levels of lakeshore and recreational use indicates that loon populations may continue to survive if suitable breeding, staging, stopover and wintering habitats are available. Loons are currently increasing in Vermont, New Hampshire and Massachusetts, while populations appear to be stable in New York and Maine. Just as human-induced habitat changes and recreational pressures probably caused the widespread declines noted prior to the 1970s, integrated management programs have contributed to the recovery of common loons in much of their northeastern U.S. breeding range. The potential for continued recovery is favorable.

## LAND PROTECTION AND PRESERVE DESIGN

Because common loons are vulnerable to habitat loss or degradation, lakes that support breeding loons or serve as important migratory stopover sites need protection. Identification and protection of known nesting areas is an important strategy because loons exhibit strong year-to-year fidelity to old nest sites (Strong and Bissonette 1985). When possible, two or three alternate sites with characteristics of preferred nesting areas (e.g., islands, deadwaters, marshes, protected coves) should be protected on each breeding lake. Small islands (< 5 ha) and deadwaters should receive complete protection from development. Undeveloped buffer zones of at least 150 m should be left on either side of mainland nest sites or deadwater entrances (Strong and Bissonette 1985). Shoreline areas adjacent to known traditional nursery areas should also be protected, with minimum undeveloped buffer zones of 150 m from both ends of the nursery (Strong and Bissonette 1985). Purchase of known nesting areas or suitable lakeshore breeding habitat by state or private conservation

organizations, or acquisition of options to protect such lands from development (e.g., easements and zoning ordinances), may preserve loon nesting habitat (McPeek and Evers 1989).

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

It is important to continue monitoring population trends and productivity in all states where common loons nest. Censusing of adult populations can begin shortly after arrival on the breeding grounds, although such counts may also include northbound transients. Populations of breeding and nonbreeding adults are most accurately estimated by repeated, standardized surveys after territory establishment in mid-May. In states such as Vermont, New Hampshire and Massachusetts, it may be possible to census all or nearly all suitable breeding habitat and to accurately determine population levels and productivity totals. In the larger states of New York and Maine, a specified sample of lakes should be surveyed annually, or every few years, to document population trends. Strong and Bissonette (1985) suggested that aerial counts of loons on 50-100 randomly selected lakes in Maine should be conducted every 3-5 years. To monitor breeding status and productivity, they recommended the establishment of a more regularly surveyed network of lakes. In New York, presence, reproductive success, and density of loons were compared on a subsample of 384 "core lakes" from a larger sample during 1977-79 and 1984-85 surveys (Parker et al. 1986, Parker and Miller 1988). A repeat survey is planned for the early 1990s.

Breeding can be verified by the presence of a recently used nest or of flightless young. Lake and island perimeters, bog mats, and floating vegetation can be carefully checked to find nests. Precautions should be taken not to flush incubating birds, since it is not necessary to view nest contents to confirm nesting. Nest searches should not be conducted during cold or wet weather. Follow-up visits can be made to determine breeding chronology and outcome.

Chicks should not be considered fledged until at least one month old (McIntyre 1988a).

Volunteer assistance can often be enlisted on a regular basis for surveys and monitoring of known breeding pairs, especially in more developed areas. Coordinated censuses, in which volunteer observers are assigned a lake or portion thereof to monitor during a specified time period on a specified date, provide an additional means to estimate loon populations. These simultaneous counts provide an index of lake occupancy and productivity and can be used to refine statewide population totals. They are typically conducted in midsummer, just after the peak period of chick hatching.

CBCs should be continued to monitor trends in distribution and abundance of common loons in early winter. These data should ideally be complemented by those from more standardized, repeated surveys at specific concentration points throughout the winter (e.g., Lee and Arbuckle 1988). A coastwide winter loon watch would provide important information on wintering patterns (McIntyre 1986). Data on wintering loons could also be collected during aerial midwinter waterfowl surveys conducted by the U.S. Fish and Wildlife Service (D. Pence pers. comm.).

### Management Procedures and Programs

Techniques for protecting common loons have focused on the breeding season and have involved primarily management of both habitat and people. Loons have responded successfully to management by private conservation groups and state agencies. Most of the organizations that conduct statewide monitoring programs also coordinate management efforts.

Several authors have reviewed management techniques and made the following recommendations in recent years (e.g., Titus and VanDruff 1981, Strong and Bissonette 1985, McIntyre 1986, Hands et al. 1989, McPeek and Evers 1989).

- ***Control of Water Levels.*** - Nest losses caused by flooding and stranding can be reduced by maintaining constant water levels during the peak nesting period (Fair 1979, Wood 1979),

usually mid-May to mid-July in New York and New England. Rises in water level are more detrimental than drawdowns, and small drawdowns may be acceptable if distances between nests and the water's edge are not greatly increased (Strong and Bissonette 1985). In areas of recent flooding, all flooded timber should be cut and removed to minimize the amount of driftwood on shorelines (Strong and Bissonette 1985).

- ***Use of Artificial Nesting Islands.*** - On lakes that lack natural islands and have poor shoreline nesting habitat, fluctuating water levels, or a history of low productivity, artificial nesting islands may improve nesting success. These rise and fall with water levels and can counteract extreme fluctuations on lakes where loons are not considered in water management plans (Wood 1979). Platforms may reduce predation on shoreline nests and provide more secure nest sites on lakes where optimal habitat has been lost or human disturbance lowers nesting success (Sutcliffe 1979b). Artificial islands have increased nesting success in Minnesota (McIntyre and Mathisen 1977), New Hampshire (Sutcliffe 1979b, Fair 1989), Vermont (Rimmer and Kaveney 1988), and Massachusetts (Lyons 1987). Artificial nesting islands alone are unlikely to induce nesting on unoccupied lakes or territories (McIntyre and Mathisen 1977, Sutcliffe 1979b) and should not be used as mitigation to development or water level manipulation (Strong and Bissonette 1985). Plans for the construction of loon platforms are available from the North American Loon Fund.
- ***Predator Management.*** - Predators can be the major cause of breeding failure in some areas. Removal of raccoons by hunting or trapping has met with some success in New Hampshire, but is labor-intensive and expensive (Wood 1979), and probably of only temporary benefit. Artificial nesting islands may reduce mammalian predation, which often increases after water levels drop (Fair 1979, Sutcliffe 1979b). Improved methods of garbage disposal could reduce nest predation

by crows, gulls and raccoons, which are often attracted to human refuse (Hands et al. 1989).

- **Modification of Commercial Fish Traps.** - Loon mortalities on the Great Lakes from commercial fishing operations could be reduced by using traps that open at the top to allow loons to escape and also by more frequently checking traps for captured loons. A cooperative program must be developed between commercial fishermen and government agencies (McIntyre 1986).
- **Protection of Breeding Habitat.** - Strong efforts to acquire and protect breeding habitat must be continued (see *Land Protection and Preserve Design*).
- **Population Monitoring.** - Long-term state surveys must be continued. Data on population trends, breeding status and productivity must be collected and evaluated to determine the necessary direction of conservation efforts.
- **Public Education.** - Human disturbance during the nesting and chick-rearing periods can be controlled in many ways. Activities such as boating, fishing, swimming, camping and picnicking should be prohibited near nest sites and in nursery areas. Closures of nest and brood-rearing sites through posting are recommended only in situations of heavy boat traffic where the closure can be reinforced through steady monitoring by law enforcement officials or volunteers (Wood 1979, J. Fair pers. comm.). Interpretive signs posted at boat ramps, beaches, campgrounds and other public access points should inform recreationists about the natural history and conservation needs of loons. The number of visitors to wilderness areas could be restricted and specific travel routes established. Visitors could be required to attend an educational program about loons before entering a recreational area with breeding loons (Titus and VanDruff 1981). No camping should be allowed on small islands, and other uses should be discouraged

(Titus and VanDruff 1981) or, if necessary, prohibited. Boat engine horsepower limitations or speed limits should be established on smaller breeding lakes or in designated areas of large lakes.

Informational brochures, posters, press releases and other educational paraphernalia should be produced and distributed (Strong and Bissonette 1985, McIntyre 1986). Educational programs, including filmstrips and slide lectures, should be presented to schools, lake associations and other groups. Personal contacts with lakeshore residents should be established and maintained. Monitoring by volunteers may be especially important on lakes with relatively low human use where the presence of law enforcement officials may not be feasible. Involvement of volunteers in lake patrols and population surveys may stimulate public interest in loons and reduce levels of human disturbance.

### Research Needs

The North American Loon Fund annually provides small grants to researchers studying a diversity of topics on loon biology, behavior and conservation. A list of current and past projects that have received funding, as well as grant application guidelines, can be obtained by contacting the Loon Fund.

Many aspects of the annual cycle, ecology, behavior and conservation of common loons need further research. These have been discussed by many authors (e.g., Titus and VanDruff 1981; Strong and Bissonette 1985; McIntyre 1986, 1988a; Hands et al. 1989; McPeek and Evers 1989) and include:

- **Wintering Distribution and Ecology.** - Detailed information on the precise wintering distribution and abundance of common loons is lacking. Little is known about the distribution of discrete breeding populations, the ecological requirements and social structure of wintering loons, the relationship between adults and juveniles, causes and rates of mortality, the impacts of environmental contaminants or oil spills, the effects of weather, the impacts of commercial

fishing, the site preferences of individual loons or different age and sex classes, and the mobility of loons during winter. In addition to research addressing these topics, monitoring programs such as a coordinated coastal winter watch or midwinter aerial loon survey should be initiated.

- ***Migration Routes and Ecology.*** - More information is needed on migration routes and staging areas of common loons in spring and fall. Little is known about the habitat and feeding requirements of loons during migration, the residence times of individual loons on staging areas, the effects of weather or human disturbance on migrating loons, and age and sex differences in the timing, route selection and ecology of migrant loons. Coordinated migration watches should be conducted at strategic coastal and inland sites.
- ***Distribution and Ecology of Juveniles.*** - Studies are needed on the life history of juveniles between fledging and their return to northern lakes. Little is known about their ecological needs and habitat use, diet, migration routes, wintering distribution, movements during their two or more years as nonbreeders, causes and rates of mortality, social relationships and behavior, age at which the definitive alternate plumage is acquired, age at first breeding, and degree of philopatry to natal lakes.
- ***Breeding Ecology.*** - Research is needed on the energetic requirements of adults and young, recruitment patterns of young and nonbreeders into breeding populations, effects of intraspecific competition on breeding status and success, site fidelity and territory turnover patterns, duration of pair bonds, and patterns of lake colonization or recolonization.
- ***Impacts of Environmental Contaminants.*** - Levels of chemical contaminants in common loons and their eggs should be monitored on a regular basis. Studies should attempt to determine the biological consequences of

chemical and heavy metal toxification on loons so that discharge practices can be modified if necessary. Monitoring should be continued on the effects of lake acidification on breeding loons.

- ***Impacts of Commercial Fishing.*** - Studies should be undertaken to quantify and assess the impacts on loons of entrapment in commercial fishing nets and traps.
- ***Causes and Effects of Diseases and Parasites.*** - Research is needed on the causes of type E botulism and how outbreaks can be prevented or minimized. Little is known about the impacts of diseases and parasites on loon populations.
- ***Development of Marking Techniques.*** - The answers to many important questions on loon movements, behavioral ecology and demography require the banding and marking of individual birds. The development of improved capture methods is essential. Protocols must be designed for individually marking discrete populations on both the breeding and wintering grounds. Feasibility studies should be initiated to design and test radio transmitters that can be used on loons.

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# PIED-BILLED GREBE

*Podilymbus podiceps*

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Breeding pied-billed grebes in the Northeast occur in ponds, sloughs, and marshes, along marshy edges of rivers, lakes, and reservoirs, and occasionally in estuarine wetlands. Preferred wetlands are > 5 ha with abundant aquatic-bed vegetation and open water interspersed with robust emergent vegetation. Breeding locations are scattered through much of the Northeast, but become extremely local in Virginia and West Virginia. Populations are more localized and less abundant in the Northeast than in other regions of the U.S. or Canada. No significant population trends were evident from Breeding Bird Survey data for the Northeast during 1966-89. Published accounts suggest long-term population declines have occurred in southern and central New England, New Jersey, Delaware, and possibly New York. Standardized surveys that adequately monitor population and habitat trends are needed. Loss and alteration of wetland habitats through draining, filling, pollution, and siltation are the primary threats to pied-billed grebes in the Northeast. Preservation of large, deepwater fresh and brackish marshes is the most important management need. Managed waterfowl impoundments often provide prime nesting and migration habitat, and future efforts to protect or create wetlands significant to waterfowl could further benefit pied-billed grebes.

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## DESCRIPTION

### Taxonomy

The pied-billed grebe (*Podilymbus podiceps*, Linnaeus 1758) is 1 of about 20 extant species of grebes in 6 genera worldwide. The grebes, Order

Podicipediformes, are an ancient group known as far back as the early Miocene, about 25 million years ago (Feduccia 1980). Fossil remains of the pied-billed grebe indicate its occurrence in the middle Pleistocene in Florida (Steadman 1984) and the late-Quaternary in New York (Steadman 1986). Three subspecies of pied-billed grebes are

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recognized (Cramp et al. 1977): *P. p. podiceps* in North America, *P. p. antillarum* in the Antilles, and *P. p. antarcticus* in South America. The following discussion is restricted to *P. p. podiceps*.

### Morphology and Plumages

The pied-billed grebe is a small, stocky, and poorly buoyant waterbird, 31-38 cm in length (Cramp et al. 1977), with small, narrow wings, feet placed far back, and a blunt-ended posterior. Diagnostic field characters include a short, stout, chicken-like bill with a broad, black band in the middle, large head and elongated neck, white orbital ring, black throat patch and forehead, and drab, brownish plumage throughout except for white under the tail (Palmer 1962, Cramp et al. 1977). During the nonbreeding period, the bill is unmarked, the throat is white, and the white rear becomes more conspicuous. As adults, sexes are alike, whereas juveniles are distinguished by the lack of a white orbital ring, an unmarked bill, darker brown sides of the head and neck, and a whiter underbelly (Palmer 1962). Downy chicks have a striking, zebra-like pattern of black and white stripes, interspersed with reddish-brown spots, and a bare loral area (Palmer 1962, Storer 1967).

### Vocalizations

During territorial defense, male pied-billed grebes emit a distinctive, prolonged call, a loud "cow-cow-cow-cow-cow-cowp...cowp...cowp..." This call is reminiscent of cuckoos (Palmer 1962), and enables grebes to communicate over several hundred meters in nesting habitats dominated by dense, visually restricting vegetation. A variety of other calls are also produced during the breeding season (see Palmer 1962), but during the nonbreeding season the species is generally silent.

### HABITAT

In the eastern U.S., breeding pied-billed grebes occur in ponds, sloughs, and marshes, in marshy inlets and along edges of rivers, lakes, and reservoirs, and occasionally in estuarine wetlands

(Palmer 1962, Chabreck 1963, Cramp et al. 1977, Andrie and Carroll 1988). In studies in Maine, pied-billed grebes occurred only in wetlands > 5 ha in size, and were more common on wetlands impounded by beavers (*Castor canadensis*) or humans than in wetlands of glacial origin (Gibbs et al. in press, Gibbs and Melvin 1990). Wetlands used by grebes had, on average, more aquatic-bed (submerged and floating vegetation) and emergent vegetation than did unused wetlands.

In an Iowa study, pied-billed grebes were observed at only 44 seasonal and semi-permanent wetlands among > 500 ponds and lakes surveyed (Faaborg 1976). Pied-billed grebes used wetlands of intermediate size (0.6-7.0 ha), but seemed to avoid either smaller or larger wetlands (Faaborg 1976). Another Iowa study suggested that the occurrence of pied-billed grebes was dependent on wetland area (Brown and Dinsmore 1986) because grebes occurred regularly only at wetlands > 5 ha. Wetlands as small as 0.3 ha, however, were used by pied-billed grebes in Manitoba (Nudds 1982). In eastern Washington, 80% of broods were found at potholes of only 0.4-2.0 ha (Yocum et al. 1958). Microhabitats of pied-billed grebes at Manitoba wetlands included the densest and tallest stands of emergent vegetation available, particularly those in deeper portions of ponds (Nudds 1982). In Iowa, pied-billed grebes were always associated with dense stands of emergent, littoral vegetation, and avoided wetlands with 100% open water (Faaborg 1976). On moist-soil impoundments in Missouri, habitat use by pied-billed grebes was associated with water > 25 cm deep and vegetative cover characterized as "open, sparse, or short" (Fredrickson and Reid 1986). Grebe use was not associated with shallower waters or "dense" or "rank" emergent vegetative cover (Fredrickson and Reid 1986). Habitat data specific to migratory pied-billed grebes is unavailable.

Grebes seek habitats in winter similar to breeding areas (Cramp et al. 1977), but many shift to more exposed areas on brackish, estuarine waters or sheltered inlets on large lakes, rivers, and salt water (Palmer 1962). Root (1988) noted that the densest overwintering populations of pied-billed grebes occur on wide rivers and large lakes.

## BIOLOGY

### Feeding Ecology

Wetmore (1924) analyzed stomach contents of 174 pied-billed grebes collected during different seasons from localities throughout North America. The diet was dominated by fish (24% by volume, including catfish, eels, perch, sunfish, suckers, carp, and minnows), crustaceans (31%), and insects (46%). Most crustaceans taken were crayfish, and insect food was predominantly Odonata (dragonflies and damselflies), Heteroptera (bugs), and Coleoptera (beetles). A strong seasonal shift in diet was observed; fish were important foods during the nonbreeding season, but were relatively unimportant during nesting. Odonates, only 8% of the overall annual diet, constituted 34% of the diet during July and August.

Eight stomachs of grebes from British Columbia contained mostly fish, while Odonates comprised most of the contents of stomachs from three downy young (Munroe 1941). Palmer (1962) reported that pied-billed grebes fed principally on leeches during the breeding season in South Carolina. Stomachs of pied-billed grebes in Pennsylvania contained fish, frogs, aquatic insects, especially beetles, and aquatic plants (Warren 1890). The stout bill and heavy jaw musculature of pied-billed grebes (Zusi and Storer 1969) may be adaptations that enable pied-billed grebes to take larger fish than other sympatric species of grebes (Forbes 1989).

Feather-eating is an unusual aspect of the pied-billed grebe's diet. Wetmore (1924) observed feathers in 52% of the 174 stomachs he examined, and adults sometimes feed feathers to their chicks (Ehrlich et al. 1988). Feather-eating may pad and protect the stomach and trap fish bones so that bones can dissolve slowly in the stomach rather than passing directly into the fragile intestine (Storer 1961). Also, hard, indigestible materials, such as chitin and bones, may be felted together with feathers prior to regurgitation as pellets (Storer 1961).

Pied-billed grebes have a number of morphological adaptations for pursuing prey underwater. Their toes are lobed and their tail is

short and rudder-like. Their feet are situated far back on the body and can be pivoted high above the back to permit quick forward propulsion and a high degree of maneuverability underwater (Fjeldsa 1975, Cramp et al. 1977). Their small, narrow wings also are used for underwater swimming (Forbush 1925). The eyes of pied-billed grebes have cone-dense retinas that permit detection of small prey at close range in shallow waters, compared to more deeply diving grebes that have rod-dense retinas (Begin and Handford 1987). Grebes compress their feathers to expel trapped air and submerge more easily, and, compared to other birds, generally have many more feathers (ca. 20,000), which enable grebes to stay waterborne continuously (Fjeldsa 1975).

In Florida, pied-billed grebes may stir up sediments with their feet to bring prey into view (King 1974). The duration of 154 foraging dives by pied-billed grebes in California ranged from 1-15 s (average 7.6 s, Bleich 1975), with grebes moving only 3.7 m, on average, between dives. Escape dives to avoid disturbance lasted longer and ended farther away than foraging dives (Bleich 1975). Pied-billed grebes frequently sunbathe between foraging dives; this may be an important means of heat conservation, especially for females (Ryan and Heagy 1980). Pied-billed grebes sometimes associate with other birds to enhance foraging opportunities. Pied-billed grebes have been observed in mutualistic foraging associations with snowy egrets (*Egretta thula*) in Virginia (Leck 1971), snowy egrets and tricolored herons (*Hydranassa tricolor*) in North Carolina (Mueller et al. 1972), and boat-tailed grackles (*Quiscalus major*) in Mississippi (Jackson 1985).

### Reproduction

Pied-billed grebes are solitary nesters and defend relatively small territories of as little as < 2 ha (Glover 1953) that provide food, cover, and nest sites. Territorial birds also sometimes forage outside their defended areas. Pied-billed grebes are highly territorial, and usually only one pair nests at a wetland (Faaborg 1976, Sealy 1978). Wetlands > 5 ha, however, may support > 1 pair (Palmer 1962, Faaborg 1976), and large marshes with suitable habitat have supported > 100 pairs (Chabreck 1963). Pied-billed grebes may be

aggressive toward other bird species with similar diet and habitat requirements, and the presence of pied-billed grebes is sometimes considered by wetland managers to be detrimental to waterfowl production (Kilham 1954, Kirby 1976).

**Nesting Period.** - Initiation of nesting activity varies throughout the pied-billed grebe's range, occurring as early as April and as late as June, and peaking in May in most areas. Examples of nesting periods are 3 May-10 September for 107 nests in Louisiana (Chabreck 1963), 2 May-8 August for 138 nests in Iowa (Glover 1953), and 3 May-22 August in Ontario (Peck and James 1983, cited in Johnsgard 1987). Although some pairing may occur on wintering areas (Palmer 1962), courtship begins soon after ice-out following the grebe's arrival at nesting areas. Courtship behavior of pied-billed grebes is mutual and less formalized than other species of grebes (Palmer 1962).

**Nests and Eggs.** - Pied-billed grebes build sodden, floating nests of rotting and green plant material and mud. Nests are often anchored to growing, emergent plants. Both sexes build nests and may add plant material and mud as the season progresses and as nests slowly sink (Fjeldsa 1975). Air-pockets in green plants and trapped gases generated by the fermenting and rotting vegetation give the nest buoyancy. In Iowa, 138 nests averaged 38 cm in diameter (Glover 1953). Nests have a hollow to hold the eggs, and may extend 90 cm below the surface but only 8 cm above (Glover 1953). Nests are typically built in shallow water surrounded by dense vegetation, especially cattail (*Typha* spp.) and bulrush (*Scirpus* spp.), and are farther from shore than from open water (Glover 1953, Stewart 1975, Faaborg 1976, Sealy 1978, Forbes et al. 1989). Wind and waves are major threats to the grebe's floating nests, and surrounding emergent vegetation acts as a wave break, anchors the nest, and conceals the nest from predators (Forbes et al. 1989). Because the direction of wind and waves shifts frequently during the nesting season, sheltered nesting sites can be limiting (Faaborg 1976). In Nova Scotia, pied-billed grebes avoided nesting on edges of stands of emergent vegetation that were exposed to wave action, and nest-site selection was related

to structure but not type of vegetation available (Forbes et al. 1989). In comparison to randomly chosen marsh locations, nests were characterized by greater distance from shore, increased proximity to open water, and deeper water (Forbes et al. 1989). The grebe's floating, rotting nest generates substantial quantities of heat, and many aspects of reproduction in pied-billed grebes may be related to their use of a warm, humid nest (Davis et al. 1985). Pied-billed grebe eggs have a threefold increase in pore density, compared to other birds' eggs, which enables eggs to lose sufficient water within the humid confines of the nest prior to hatching (Davis et al. 1985). When leaving the nest, adult grebes cover their eggs with plant material, and the rotting nest, where temperatures may remain 11-13°C higher than the surrounding water, can provide enough heat to incubate the eggs in the adults' absence (Davis et al. 1985). Time constraints imposed by incubation may thereby be lessened, providing adult pied-billed grebes with more time for foraging and territory defense.

Eggs are elliptical to subelliptical, ca. 44.3 x 30.1 mm, and smooth and nonglossy (Harrison 1978, Arnold 1989). Although white or tinted bluish or buff when laid, eggs gather a heavy, brown stain from the wet, organic matter comprising the nest. Pied-billed grebes are single- or double-brooded and lay 2-10 eggs, usually 6-8 eggs per clutch (Sealy 1978, Forbes et al. 1989). Eggs are laid daily. Incubation is initiated after the fourth egg is laid, and occurs during about 90% of a given day (Forbes and Ankney 1988a). Incubation is shared equally between sexes during laying and post-laying periods, although females spend more time incubating around hatching (Forbes and Ankney 1988a), which occurs at about 23 days (Palmer 1962). Human disturbance can greatly disrupt patterns of nest attendance and incubation (Forbes and Ankney 1988b, Davis et al. 1985).

Unlike many species of birds, pied-billed grebes begin incubation before completing the clutch (Cramp et al. 1977), which leads to considerable hatching asynchrony among the brood. Two to four eggs generally hatch on the first day of hatching, and the remaining eggs hatch daily over a period of 3-7 days (Forbes and Ankney 1987, 1988a). A detailed description of

the hatching muscle and its development in embryonic pied-billed grebes is given by Fisher (1961). The first eggs laid are about 8% lighter than subsequent eggs within clutches (Forbes and Ankney 1988b), but variation in egg weight probably has little effect on the vigor of individual hatchlings (Arnold 1989).

**Nesting Success.** - High nest success has been reported in many areas: 70% of 138 nests in Iowa (Glover 1953), 77% of 150 nests in Wisconsin (Otto 1983), 90% of 107 nests in Louisiana (Chabreck 1963), and 90% of 115 nests in Nova Scotia (Forbes et al. 1989) hatched  $\geq 1$  egg. Wind and high waves, fluctuating water levels, and predation can be significant sources of nest loss. Of 42 nests in Manitoba, 69% failed, mostly due to flooding from high waves (Sealy 1978). Half of total nest loss in Iowa was due to wave action or water level fluctuation and 25% to raccoon (*Procyon lotor*) predation (Glover 1953). In Nova Scotia, nest loss resulted from predation, including crows (*Corvus brachyrhynchos*) and poor weather (Forbes et al. 1989). Females are indeterminate layers (Fugle and Rothstein 1977), and frequently replace lost clutches, usually renesting within 50 m of destroyed nests (Glover 1953, Forbes et al. 1989).

**Chick Rearing.** - Adult pied-billed grebes usually divide broods and provision chicks with a variety of small-sized prey, including dragonfly naiads, dytiscid beetle larvae, leeches, and salamanders (Forbes and Ankney 1987). Chicks usually remain near parents, and frequently ride on the backs of adults, even during foraging dives (Forbush 1925). Initial size disparities of chicks, due to asynchronous hatching, influence food allocation within broods. Aggression among chicks is high when rates of food-delivery by adults are low, and larger chicks win more disputes over food than smaller chicks (Forbes and Ankney 1987). The bare loral area of chicks changes from dull-colored to bright crimson in hungry chicks, however, and adults may use this indicator of nutritional status to allocate food among members of a brood (Forbes and Ankney 1987). Two unusual forms of chick provisioning occur: for unknown reasons, chicks are occasionally fed by adults other than pair

members (Forbes 1987), and young grebes from first broods may feed young from second broods (Cramp et al. 1977).

Age at first flight has been estimated at 35 days (Kirby 1976, Forbes and Ankney 1987). In comparison to clutch sizes, observations of relatively small broods, e.g., averaging 4.4 (Chabreck 1963) and 2.9 (Yocum et al. 1958), suggest that substantial chick mortality occurs. Snapping turtles (*Chelydra serpentina*) may represent important predators of young pied-billed grebes (Coulter 1957). Age at first breeding may be as early as 13 months (MacVean 1990).

**Molt and Migration.** - Molt in pied-billed grebes is poorly known. Palmer (1962) noted that a complete molt into basic plumage takes place in autumn, with considerable individual variation in its timing and duration (Cramp et al. 1977). Flight feathers are lost simultaneously, prior to loss of body feathers. Complete molt takes 1-2 months. Basic molt and late nesting may occur simultaneously (Otto and Strohmeyer 1985). A partial molt into alternate plumage occurs in spring (Palmer 1962).

Fall migration is protracted and begins in August, with the majority of migrants moving south between September and November (Cramp et al. 1977). Migration is usually at night. Freezing temperatures sometimes force birds to move short distances southward during mid-winter. During winter, pied-billed grebes may occur in large, dispersed flocks occasionally of  $> 1,000$  birds and commonly 100 birds. Return migration begins in March, and arrival at nesting areas in April and early May is dependent on timing of spring thaw (Cramp et al. 1977).

## POPULATION DISTRIBUTION, STATUS AND TRENDS

### Distribution and Status in North America

The pied-billed grebe is the most widespread Nearctic breeder in its family, and breeds locally throughout North America south of central British Columbia, Alberta, Saskatchewan,

Manitoba, Ontario, southwestern Quebec, southern New Brunswick, and Prince Edward Island (Figure 1, Palmer 1962). Breeding occurs patchily throughout the northeastern states, except in Virginia and West Virginia, where breeding is extremely local (Figure 2).

Data from U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; Robbins et al. 1986) suggest that during the period 1966-89, U.S. and Canadian populations of pied-billed grebes showed no detectable trends (Table 1). During this period in the U.S., however, a small but significantly ( $P <$

**Table 1.** Mean number of pied-billed grebes (*Podilymbus podiceps*) per Breeding Bird Survey route and median percent annual change, 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; see Robbins et al. 1986 for details).

Region	No. routes <sup>a</sup>	$\bar{x}$ birds/route	% annual change <sup>b</sup>
<b>State</b>			
Maine	3	0.02	+3.1
New Hampshire	3	0.02	+0.6
Vermont	4	0.04	-1.1
Massachusetts	0	-	-
Rhode Island	0	-	-
Connecticut	2	0.01	-2.4
New York	12	0.02	-0.6
Pennsylvania	3	0.01	6.6
New Jersey	0	-	-
Delaware	0	-	-
Maryland	3	0.00	-0.2
Virginia	0	-	-
West Virginia	0	-	-
Eastern U.S.	198	0.09	-0.5
U.S.	390	0.21	-0.7 <sup>c</sup>
Canada	129	0.36	-1.0

<sup>a</sup>Number of routes on which pied-billed grebes were detected (population trends may be invalid for states with < 10 routes).

<sup>b</sup>All population trends were not significant ( $P > 0.10$ ).

<sup>c</sup>A significantly ( $P < 0.01$ ) greater proportion of routes showed increasing grebe populations (0.51) than decreasing (0.44).

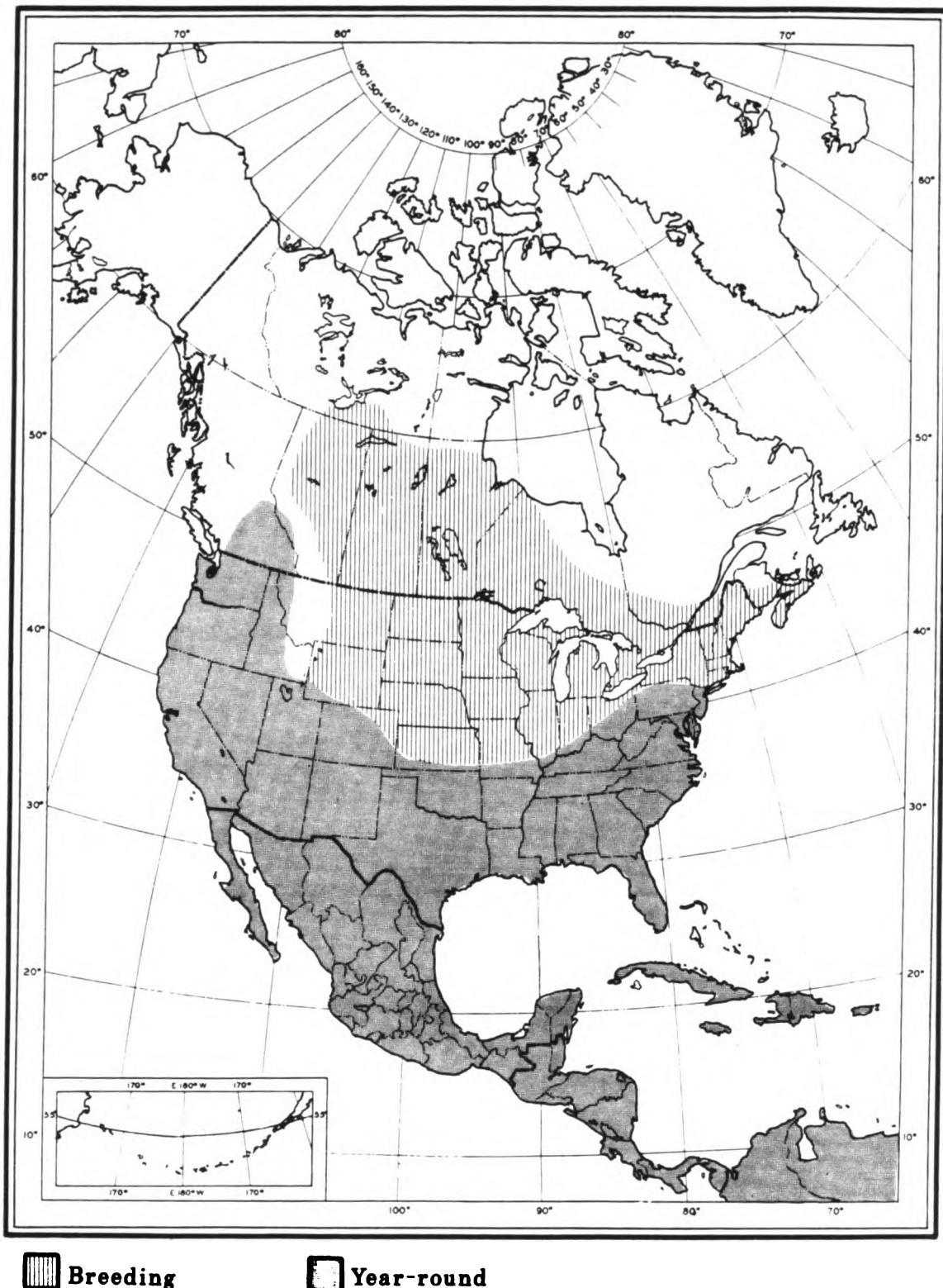
0.01) larger proportion of BBS routes showed increases in grebe populations (0.51) than decreases (0.44). No significant population trends were evident for the eastern U.S., but data were too sparse to assess populations on a state-by-state basis in the northeastern region, except for New York (Table 1). An earlier analysis covering the period 1966-86 suggested that populations were in slight ( $P < 0.10$ ) decline. The pied-billed grebe was not included on either the 1982 or 1987 list of *migratory nongame birds of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987), but was considered a *local problem species* by the National Audubon Society in 1982 because of severe declines in New Jersey (Tate and Tate 1982). The pied-billed grebe has a Natural Heritage Program ranking of G5, i.e., demonstrably secure globally, although possibly rare in parts of its range (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

Northernmost populations are migratory, and the species' winter distribution extends over a wide inland area (Root 1988) and overlaps with nonmigratory populations (Cramp et al. 1977). Dense wintering populations occur at inland regions bordering the Atlantic coast (south of Virginia, especially southern Florida), the Gulf of Mexico coast (especially southern Louisiana and southern Texas), at Great Salt Lake, Lake Mead, and the San Joaquin Valley in California (Root 1988). Overwintering populations also range south into Mexico (Cramp et al. 1977). The northern limit for overwintering populations is determined by availability of ice-free, freshwater habitats (in areas where average January temperature  $> 6^{\circ}\text{C}$ : Root 1988), and may change within and between years due to temperature fluctuation.

#### Distribution and Status in the Northeast Region

**Maine.** - The pied-billed grebe is considered an uncommon but widely occurring summer resident of freshwater wetlands in Maine (Palmer 1949). Surveys for the Maine Breeding Bird Atlas (BBA) (Adamus 1988) indicated that pied-billed

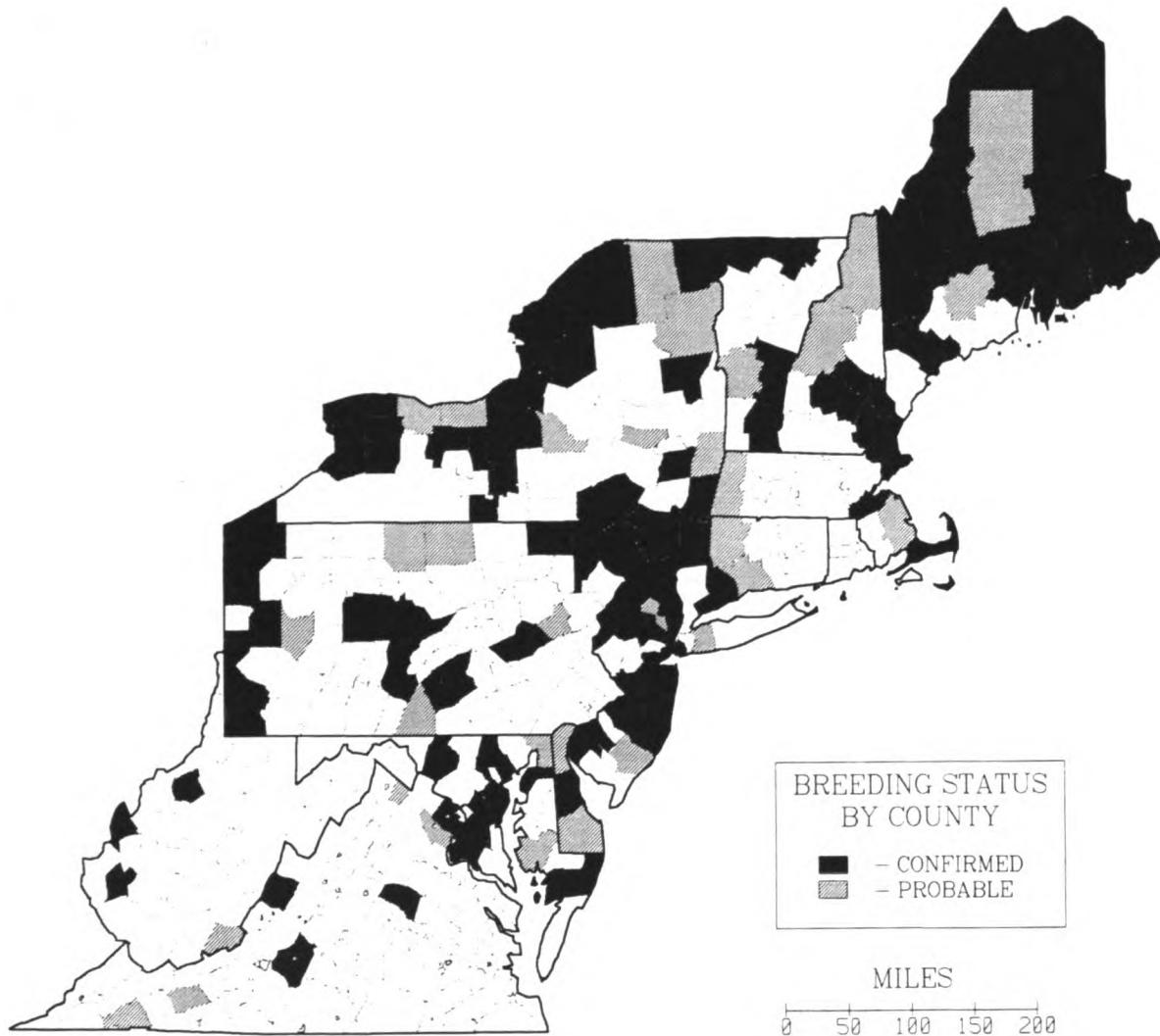
**Figure 1.** The North American range of the pied-billed grebe (*Podilymbus podiceps*) (American Ornithologists' Union 1983, National Geographic Society 1987).



Breeding

Year-round

**Figure 2.** Northeast breeding distribution of the pied-billed grebe (*Podilymbus podiceps*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



grebes were present during the breeding season in 11 minimum coverage blocks (22% of total). The species was sparsely but widely distributed in the state (Adamus 1988, Figure 2). A recent, 12-year study of wetlands in central and eastern Maine located summer residents at 17% of 87 wetlands surveyed (Gibbs et al. in press). Pied-billed grebes were encountered on too few BBS routes to assess population trends in Maine during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S4 (apparently secure) in Maine, and has no special listing with the state (Table 2).

**Table 2. Summary of state listing status and Natural Heritage Program state ranks for the pied-billed grebe (*Podilymbus podiceps*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	S4
New Hampshire	endangered	S2
Vermont	special concern	S3
Massachusetts	threatened	S1
Rhode Island	extirpated	SH
Connecticut	endangered	S1/S2
New York	-	S3
Pennsylvania	-	S3
New Jersey	endangered <sup>c</sup>	S1
Delaware	-	S2
Maryland	in need of conservation <sup>b</sup>	S3
Virginia	-	S3
West Virginia	-	S2

<sup>a</sup>SH = historical records only, but suspected extant

S1 = critically imperiled; ≤5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

<sup>b</sup>Proposed

<sup>c</sup>Breeding population only

**New Hampshire.** - Allen (1909) considered the pied-billed grebe to be a rare, local summer resident in New Hampshire, and perhaps most

abundant in the southern part of the state. Recent, statewide BBA surveys (1981-86) found pied-billed grebes present during the breeding season at 21 locations, mostly in the central and southern part of the state (C. Foss pers. comm., Figure 2). Pied-billed grebes were encountered on too few BBS routes to assess population trends in New Hampshire during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S2 in New Hampshire (imperiled because of rarity), and is listed as an *endangered* species by the New Hampshire Department of Fish and Game (Table 2).

**Vermont.** - Although once a fairly common summer resident in Vermont (Perkins and Howe 1901), the pied-billed grebe is now considered relatively rare (Laughlin and Kibbe 1985). Surveys for the Vermont BBA found pied-billed grebes during the breeding season at 12 priority blocks (7% of total: Laughlin and Kibbe 1985; Figure 2). Records were scattered among small patches of wetland in northeastern, central, and southeastern Vermont, but were conspicuously rare in the extensive and seemingly suitable marshlands of Lake Champlain and Lake Memphremagog (Laughlin and Kibbe 1985). A recent survey of five of Vermont's major marshes (D. Kibbe 1989) located six pairs of pied-billed grebes at Lake Memphremagog, but detected only a single bird in extensive marshes along Lake Champlain. Pied-billed grebes were encountered on too few BBS routes to assess population trends in Vermont during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S3 in Vermont (i.e., rare or uncommon, Table 2). The species is currently listed as a *species of special concern* by the Vermont Fish and Wildlife Department (Table 2), due in part to the small size of extant populations relative to the availability of nesting habitats, although no immediate threats to the species' continued presence are perceived (Laughlin and Kibbe 1985).

**Massachusetts.** - Howe and Allen (1901) described the pied-billed grebe as a local breeder throughout the state. In the Concord area, Griscom (1949) considered the pied-billed grebe a breeder whose numbers varied annually in

response to habitat availability (absent in dry years and when woody vegetation overtook nesting areas). Griscom and Snyder (1955) described the pied-billed grebe as a local summer resident throughout Massachusetts, but absent from the coastal plain and outer islands (Griscom and Folger 1948). Griscom and Snyder (1955) reported that numbers had declined greatly in Massachusetts since the 1850s. Statewide BBA surveys from 1974-79 found pied-billed grebes present during the breeding season at 13 locations, all in coastal or extreme western portions of the state (J. Baird pers. comm., Figure 2). Pied-billed grebes were encountered on too few BBS routes to assess population trends in Massachusetts during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of *S1* (critically imperiled) in Massachusetts, and is listed as *threatened* by the Massachusetts Division of Fisheries and Wildlife because of its rarity in the state (Table 2).

**Rhode Island.** - The pied-billed grebe has been extirpated as a breeding bird in Rhode Island. Nesting is well documented up to the 1950s, and occurred mostly in coastal, brackish marshes and at one inland site. Populations may have been in decline for many decades. At the turn of the century Howe and Sturtevant (1899) stated that the pied-billed grebe was an uncommon, declining summer resident in the state. Breeding was not recorded during recent, statewide BBA surveys (R. Enser pers. comm.). Pied-billed grebes were not encountered on BBS routes in Rhode Island during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of *SH* (historical records only, but suspected extant) in Rhode Island, and is listed as extirpated by the Rhode Island Department of Fish and Wildlife (Table 2).

**Connecticut.** - The pied-billed grebe may never have been a common nesting species in Connecticut (Zeranski and Baptist 1990). Merriam (1877) stated that the pied-billed grebe was a summer resident in Connecticut, but gave no indication of its abundance or distribution. The few records in Sage et al. (1913) indicated that nesting occurred sparsely within the state. The pied-billed grebe colonized several nesting

localities in southwestern Connecticut from 1934-39, but thereafter became only transient again in the region (Saunders 1950). MacKenzie (1961) noted that the pied-billed grebe had nested during the 1940s but was declining in the Guilford area. More recently, breeding activity has been restricted to areas west of the Connecticut River (Figure 1). Surveys for the Connecticut BBA found pied-billed grebes during the breeding season in 24 blocks (4% of total) in 6 counties, with reports of confirmed or probable nesting all occurring in Fairfield, Litchfield, and New Haven counties (Bevier in press). The species has a Natural Heritage Program ranking of *S1/S2* in Connecticut (imperiled, possibly critically, because of rarity), and is listed as an *endangered* species in the state (Table 2). Pied-billed grebes were not encountered on BBS routes in Connecticut during the period 1966-89 (Table 1). Despite supporting few nesting grebes, coastal brackish marshes and inland waters in Connecticut provide important stopover areas for migrants and, occasionally, winter residents (Zeranski and Baptist 1990).

**New York.** - Surveys for the New York BBA indicated that the pied-billed grebe is a widespread but locally rare to uncommon breeder in the state (Andrie and Carroll 1988). Pied-billed grebes were recorded during the breeding season on 182 (3% of total) survey blocks, and breeding was confirmed on 53 blocks. Most records came from Hudson River Valley marshes and protected and managed wetlands at state and national wildlife refuges throughout the state. Comparatively few records came from mountainous and upland regions in the central and southwestern parts of the state where wetland habitats are limited (Figure 2). During a 1989 survey for black terns at wetlands in western and northern New York, pied-billed grebes were observed at 18 of 95 sites visited; all but 1 occupied site were at state or federal wildlife management areas (Novak 1990). Data from BBS routes indicate that pied-billed grebe populations showed no significant trend in New York during the period 1966-89 (Table 1). In the New York City region, Bull (1974) suggested that both overwintering and breeding populations had increased because of milder winters and the creation of wildlife refuges. The species has a

Natural Heritage Program ranking of S3 (rare or uncommon) in New York, and has no special protective status in the state (Table 2).

**Pennsylvania.** - Records from the late 1800s indicate that the pied-billed grebe was never an abundant breeding bird over much of Pennsylvania. Warren (1890) considered the pied-billed grebe to be a common spring and fall visitor during migration, but a rare summer resident (see also Sutton 1928). In parts of western Pennsylvania, Todd (1940) considered the species to be a common and regular breeder, e.g., at Pymatuning Lake, Conneaut Lake, and Erie Bay, but rare and scattered elsewhere. Poole (1964) described the pied-billed grebe as a fairly common but inconspicuous summer resident.

Recent surveys for the Pennsylvania BBA found pied-billed grebes during the breeding season at 86 locations in the state (Brauning in press). The species generally is most abundant in the southeastern and northwestern corners of the state, but it nests sparsely elsewhere, where marshland is limited (Figure 2). Unlike other marsh birds in Pennsylvania, the pied-billed grebe does not appear to have undergone major declines in the state (Gill 1985), perhaps because it readily colonizes artificial impoundments and frequents lacustrine habitats that are less vulnerable to drainage than shallow marshes in the state. The species has a Natural Heritage Program ranking of S3 (rare or uncommon) and no special protective status with the Pennsylvania Game Commission (Table 2). Pied-billed grebes were not encountered on BBS routes in Pennsylvania during the period 1966-89 (Table 1).

**New Jersey.** - Stone (1908) considered the pied-billed grebe a regular but local breeder in the state. Formerly a common summer resident of freshwater ponds and marshes in many areas of New Jersey, the pied-billed grebe underwent population declines during the 1970s (Leck 1984), and is now considered a very uncommon breeding bird. Bull (1974) listed eight breeding localities known from the 1940s along the northern New Jersey coast, but by 1981 only two breeding sites were known in the state, at Kearney Marsh and Pedricktown (Leck 1984), prompting the National Audubon Society to list the pied-billed grebe as a

*local problem species* (Tate and Tate 1982). Recent, statewide surveys for a BBA (D. Hughes pers. comm., Figure 2), however, indicate that the species is still widely distributed in the state. Although salt and brackish marshes are abundant in New Jersey, the pied-billed grebe generally prefers freshwater areas, and Leck (1984) suggests that the combination of extensive emergent vegetation and adjacent open waters favored by the species may be scarce at most inland locations. Pied-billed grebes were not encountered on BBS routes in New Jersey during the period 1966-89 (Table 1). The pied-billed grebe has a Natural Heritage Program ranking of S1 (critically imperiled) in New Jersey, and the breeding population is listed as *endangered* by the New Jersey Division of Fish, Game, and Wildlife (Table 2).

**Delaware.** - Few data are available on the abundance and population trends of the pied-billed grebe in Delaware, where the species occurs year-round on freshwater ponds and brackish marshes (Harding and Harding 1980). Recent surveys for the Delaware BBA found pied-billed grebes present during the breeding season at 14 locations in central and northern portions of the state (Hess et al. in press, Figure 2). Summer populations are restricted to Delaware tidal marshes, are currently estimated at between 10-100 individuals, and appear to have declined. Pied-billed grebes were not encountered on BBS routes in Delaware during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S2 (imperiled because of rarity) in Delaware (Table 2).

**Maryland and the District of Columbia.** - The pied-billed grebe is considered to be an uncommon breeder in the tidewater areas of the Eastern Shore and Western Shore, and a rare breeder in the interior of the Eastern Shore and Western Shore, Upper Chesapeake, and Piedmont sections (Stewart and Robbins 1958). Recent surveys for the BBA of Maryland and the District of Columbia (G. Therres pers. comm.) recorded grebes present during the breeding season at 32 locations in the state (Figure 2). Pied-billed grebes were encountered on too few BBS routes to assess population trends in Maryland during

the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S3 (rare or uncommon) in Maryland, and is currently proposed as a species in need of conservation by the Maryland Forest, Park and Wildlife Service. Wetland habitats of pied-billed grebes in Maryland are currently threatened by destruction from filling, ditching, and draining and indirectly through degradation of water quality and reduction of wetland productivity.

**Virginia.** - The pied-billed grebe is a rare breeder on weedy and marshy ponds and sloughs in Virginia, with 21 nesting records known from the coastal plain (Chincoteague, Virginia Beach, Norfolk, Newport News, and Charles City counties), the Piedmont (Amherst County), and mountains and valleys west of the Piedmont (Rockingham and Russell counties) (Virginia Society Ornithology 1979; S. Ridd and R. Wadja pers. comm.: BBA data, 1984-89). Pied-billed grebes were not encountered on BBS routes in Virginia during the period 1966-89 (Table 1). The pied-billed grebe is much more abundant during the nonbreeding season, and is considered a common migrant and winter resident throughout the state (except west of the Piedmont). The species has a Natural Heritage Program ranking of S3 (rare or uncommon) in Virginia, and no special listing by the Virginia Commission of Game and Inland Fisheries (Table 2).

**West Virginia.** - Hall (1983) considered the pied-billed grebe to be a well distributed but rare and local summer resident in West Virginia. Although there are few historical reports of nesting within the state, the species may be frequently overlooked and more numerous than reports indicate (Hall 1983). Recent surveys for a state BBA recorded pied-billed grebes during the breeding season at 16 locations (S. Ridd and R. Wadja pers. comm., Figure 2). Habitat availability imposes a severe limit to the size of breeding populations in West Virginia. Only 0.3% of the state is under water, primarily in unvegetated lake-like situations behind impoundments. Malaria control may have eliminated much of the weedy growth along wetland margins that the grebe required

(Anderson et al. 1981). Pied-billed grebes were not encountered on BBS routes in West Virginia during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S2 (imperiled because of rarity) in West Virginia, but no special listing by the state (Table 2).

## Summary

Based on BBS data, no obvious population trends were evident during the period 1966-89 for pied-billed grebe populations in the eastern U.S. (Table 1). Only one state (New York) had an adequate sample of routes, however, to permit reliable assessment of state-specific population trends. A notable feature of these data is that, on the relatively few BBS routes on which pied-billed grebes occurred in the northeastern states (Table 1), the species was encountered at a low frequency ( $\leq 0.04$  birds/route) in comparison to the U.S. average (0.21 birds/route) and Canadian average (0.36 birds/route). Thus, nesting populations in the northeastern states seem both local (i.e., detected on few routes per state) and sparse (i.e., few individuals recorded per route) compared to elsewhere in the species' range.

Nesting populations in many northeastern states, however, probably underwent substantial declines long before BBS efforts were initiated. Published accounts suggest long-term population declines in New Hampshire, Massachusetts, Rhode Island, Connecticut, New Jersey, and Delaware, and possibly in New York and Vermont. Hunting for plumes and sport resulted in a significant, initial reduction in grebe populations throughout the region in the late 1800s to early 1900s (Warren 1890, Forbush 1925), prior to the occurrence of massive losses of marshlands in the mid- to late 1900s (Tiner 1984). Currently in some states (e.g., Vermont), grebe populations seem well below levels available habitat could support (Laughlin and Kibbe 1985).

Many prime nesting and migration areas are now protected at state- and federally-managed waterfowl impoundments (e.g., Maryland, New York, and Vermont), and future efforts to protect or create wetlands significant to waterfowl in the northeastern states could further benefit the pied-billed grebe. The continued presence of the species in the northeastern U.S., however, is

contingent upon preventing further loss and degradation of inland and brackish marshes.

## LIMITING FACTORS AND THREATS

The greatest threat to pied-billed grebe populations in the northeastern region is alteration and loss of wetland habitat through draining, dredging, filling, pollution, acid rain, agricultural practices, and siltation. Palustrine emergent wetlands, including inland freshwater and brackish marshlands frequented by pied-billed grebes, are among the most threatened wildlife habitats in the U.S. Over 4.75 million acres (1.92 million ha) of such wetlands were destroyed in the U.S. between the mid-1950s and mid-1970s, and losses continue at > 160,000 ac/year (64,777 ha) (Tiner 1984).

Pollution and environmental contaminants may degrade wetland ecosystems and impair the reproductive capacity of pied-billed grebes in industrialized portions of their range. Carbamate pesticides have had lethal effects on pied-billed grebes in New York (Stone 1979). Although acidification of wetland nesting habitats could potentially reduce food supplies, pied-billed grebes usually occupy wetlands of circumneutral pH with dense growths of emergent vegetation (Gibbs et al. in press, Gibbs and Melvin 1990) that may provide effective chemical buffering against acidification. In agricultural areas, siltation resulting from erosion of farmlands and run-off containing insecticides may degrade nesting habitats and reduce availability of invertebrate foods.

Little is known about sources of mortality in pied-billed grebes. Avian botulism, avian cholera, and gizzard worms are known to occur in grebes (Friend 1987). Pied-billed grebes are highly susceptible to oil toxicosis, although this does not pose a major threat to overwintering pied-billed grebes because the species occurs in small groups and favors sites at inland, fresh waters generally well-protected from large, ocean-borne oil spills (Clapp et al. 1982). Predators of pied-billed grebes include cottonmouths (*Agkistrodon piscivorus*) (Leavitt 1957), peregrine falcons (*Falco peregrinus*) (Buckalew 1948), and American alligators (*Alligator mississippiensis*) (Delany

1986). Coulter (1957) reported substantial predation on pied-billed grebes by snapping turtles in Maine. One pied-billed grebe died after choking on a fish (Behrstock 1981). Pied-billed grebes sometimes mistake wet roads for water bodies from which they are unable to take off.

Human exploitation historically has had an important impact on grebe numbers. Warren (1890) reported that large numbers of pied-billed grebes were shot and sold to milliners and furriers who fashioned ear-muffs and hat ornaments from the silver-white breast and abdomen feathers. The pied-billed grebe has long been persecuted because of the challenge it poses as a target for hunters (Bent 1919, Forbush 1925, Palmer 1949). The species' predilection for using managed impoundments during migration may predispose it to accidental or malicious shootings by duck hunters. Human presence in marshes disrupts incubation activities of female pied-billed grebes (Forbes and Ankney 1988a).

## MANAGEMENT POTENTIAL

Pied-billed grebes will colonize artificial wetlands created by surface-mining (Perkins and Lawrence 1985) and abandoned industrial settling ponds (Rickard et al. 1981). Also, pied-billed grebes readily use artificial wetlands at managed impoundments (e.g., at state and federal waterfowl refuges: Andrle and Carroll 1988, Gibbs et al. in press). Artificial impoundments are thought to have reversed population declines in some areas and led to local population increases in others (Bull 1974). The pied-billed grebe's ability to renest following nest loss and raise two broods per year, its relatively large clutch size (up to seven eggs), and its tolerance of a wide range of habitat conditions, suggest that, given a stable habitat base, management potential for populations is high.

## LAND PROTECTION AND PRESERVE DESIGN

Minimum wetland area is an important consideration in preserve design. Pied-billed

the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S3 (rare or uncommon) in Maryland, and is currently proposed as a species in need of conservation by the Maryland Forest, Park and Wildlife Service. Wetland habitats of pied-billed grebes in Maryland are currently threatened by destruction from filling, ditching, and draining and indirectly through degradation of water quality and reduction of wetland productivity.

**Virginia.** - The pied-billed grebe is a rare breeder on weedy and marshy ponds and sloughs in Virginia, with 21 nesting records known from the coastal plain (Chincoteague, Virginia Beach, Norfolk, Newport News, and Charles City counties), the Piedmont (Amherst County), and mountains and valleys west of the Piedmont (Rockingham and Russell counties) (Virginia Society Ornithology 1979; S. Ridd and R. Wadja pers. comm.: BBA data, 1984-89). Pied-billed grebes were not encountered on BBS routes in Virginia during the period 1966-89 (Table 1). The pied-billed grebe is much more abundant during the nonbreeding season, and is considered a common migrant and winter resident throughout the state (except west of the Piedmont). The species has a Natural Heritage Program ranking of S3 (rare or uncommon) in Virginia, and no special listing by the Virginia Commission of Game and Inland Fisheries (Table 2).

**West Virginia.** - Hall (1983) considered the pied-billed grebe to be a well distributed but rare and local summer resident in West Virginia. Although there are few historical reports of nesting within the state, the species may be frequently overlooked and more numerous than reports indicate (Hall 1983). Recent surveys for a state BBA recorded pied-billed grebes during the breeding season at 16 locations (S. Ridd and R. Wadja pers. comm., Figure 2). Habitat availability imposes a severe limit to the size of breeding populations in West Virginia. Only 0.3% of the state is under water, primarily in unvegetated lake-like situations behind impoundments. Malaria control may have eliminated much of the weedy growth along wetland margins that the grebe required

(Anderson et al. 1981). Pied-billed grebes were not encountered on BBS routes in West Virginia during the period 1966-89 (Table 1). The species has a Natural Heritage Program ranking of S2 (imperiled because of rarity) in West Virginia, but no special listing by the state (Table 2).

## Summary

Based on BBS data, no obvious population trends were evident during the period 1966-89 for pied-billed grebe populations in the eastern U.S. (Table 1). Only one state (New York) had an adequate sample of routes, however, to permit reliable assessment of state-specific population trends. A notable feature of these data is that, on the relatively few BBS routes on which pied-billed grebes occurred in the northeastern states (Table 1), the species was encountered at a low frequency ( $\leq 0.04$  birds/route) in comparison to the U.S. average (0.21 birds/route) and Canadian average (0.36 birds/route). Thus, nesting populations in the northeastern states seem both local (i.e., detected on few routes per state) and sparse (i.e., few individuals recorded per route) compared to elsewhere in the species' range.

Nesting populations in many northeastern states, however, probably underwent substantial declines long before BBS efforts were initiated. Published accounts suggest long-term population declines in New Hampshire, Massachusetts, Rhode Island, Connecticut, New Jersey, and Delaware, and possibly in New York and Vermont. Hunting for plumes and sport resulted in a significant, initial reduction in grebe populations throughout the region in the late 1800s to early 1900s (Warren 1890, Forbush 1925), prior to the occurrence of massive losses of marshlands in the mid- to late 1900s (Tiner 1984). Currently in some states (e.g., Vermont), grebe populations seem well below levels available habitat could support (Laughlin and Kibbe 1985).

Many prime nesting and migration areas are now protected at state- and federally-managed waterfowl impoundments (e.g., Maryland, New York, and Vermont), and future efforts to protect or create wetlands significant to waterfowl in the northeastern states could further benefit the pied-billed grebe. The continued presence of the species in the northeastern U.S., however, is

contingent upon preventing further loss and degradation of inland and brackish marshes.

## LIMITING FACTORS AND THREATS

The greatest threat to pied-billed grebe populations in the northeastern region is alteration and loss of wetland habitat through draining, dredging, filling, pollution, acid rain, agricultural practices, and siltation. Palustrine emergent wetlands, including inland freshwater and brackish marshlands frequented by pied-billed grebes, are among the most threatened wildlife habitats in the U.S. Over 4.75 million acres (1.92 million ha) of such wetlands were destroyed in the U.S. between the mid-1950s and mid-1970s, and losses continue at > 160,000 ac/year (64,777 ha) (Tiner 1984).

Pollution and environmental contaminants may degrade wetland ecosystems and impair the reproductive capacity of pied-billed grebes in industrialized portions of their range. Carbamate pesticides have had lethal effects on pied-billed grebes in New York (Stone 1979). Although acidification of wetland nesting habitats could potentially reduce food supplies, pied-billed grebes usually occupy wetlands of circumneutral pH with dense growths of emergent vegetation (Gibbs et al. in press, Gibbs and Melvin 1990) that may provide effective chemical buffering against acidification. In agricultural areas, siltation resulting from erosion of farmlands and run-off containing insecticides may degrade nesting habitats and reduce availability of invertebrate foods.

Little is known about sources of mortality in pied-billed grebes. Avian botulism, avian cholera, and gizzard worms are known to occur in grebes (Friend 1987). Pied-billed grebes are highly susceptible to oil toxicosis, although this does not pose a major threat to overwintering pied-billed grebes because the species occurs in small groups and favors sites at inland, fresh waters generally well-protected from large, ocean-borne oil spills (Clapp et al. 1982). Predators of pied-billed grebes include cottonmouths (*Agkistrodon piscivorus*) (Leavitt 1957), peregrine falcons (*Falco peregrinus*) (Buckalew 1948), and American alligators (*Alligator mississippiensis*) (Delany

1986). Coulter (1957) reported substantial predation on pied-billed grebes by snapping turtles in Maine. One pied-billed grebe died after choking on a fish (Behrstock 1981). Pied-billed grebes sometimes mistake wet roads for water bodies from which they are unable to take off.

Human exploitation historically has had an important impact on grebe numbers. Warren (1890) reported that large numbers of pied-billed grebes were shot and sold to milliners and furriers who fashioned ear-muffs and hat ornaments from the silver-white breast and abdomen feathers. The pied-billed grebe has long been persecuted because of the challenge it poses as a target for hunters (Bent 1919, Forbush 1925, Palmer 1949). The species' predilection for using managed impoundments during migration may predispose it to accidental or malicious shootings by duck hunters. Human presence in marshes disrupts incubation activities of female pied-billed grebes (Forbes and Ankney 1988a).

## MANAGEMENT POTENTIAL

Pied-billed grebes will colonize artificial wetlands created by surface-mining (Perkins and Lawrence 1985) and abandoned industrial settling ponds (Rickard et al. 1981). Also, pied-billed grebes readily use artificial wetlands at managed impoundments (e.g., at state and federal waterfowl refuges: Andrle and Carroll 1988, Gibbs et al. in press). Artificial impoundments are thought to have reversed population declines in some areas and led to local population increases in others (Bull 1974). The pied-billed grebe's ability to renest following nest loss and raise two broods per year, its relatively large clutch size (up to seven eggs), and its tolerance of a wide range of habitat conditions, suggest that, given a stable habitat base, management potential for populations is high.

## LAND PROTECTION AND PRESERVE DESIGN

Minimum wetland area is an important consideration in preserve design. Pied-billed

grebes typically occur on wetlands above a minimum threshold size, although this "minimum area" varies regionally. Minimum area requirements appear to be as large as 5 ha in the Midwest and Northeast (Brown and Dinsmore 1986, Gibbs et al. in press, Gibbs and Melvin 1990), although sites in the Midwest as small as 0.5 ha are used occasionally (Faaborg 1976). If provided with suitable habitat, large breeding populations can be supported on a single, managed wetland, e.g., up to 107 pairs have nested on an 81-ha impoundment in Louisiana (Chabreck 1963).

Moderately deep (0.25-2 m), stable waters represent an important feature of habitats for nesting grebes. Key physical features of breeding areas also include large areas of aquatic-bed vegetation and open water, which can serve as deepwater feeding sites, interspersed with dense growth of robust emergents, with some > 100 m from shorelines that can serve as predator-free nest sites. Protection from human disturbance is important, i.e., from boats whose wakes can flood nests and recreationists whose presence can disturb incubating birds. Preserves also should be protected from upland run-off that may transport silt and contaminants and thereby lower wetland productivity and reduce food supplies.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Development of standardized surveys to monitor pied-billed grebe populations is critical because there are currently few population data on which to base management decisions. Standard monitoring procedures for bird populations encounter pied-billed grebes at frequencies too low to assess population trends in most areas. For example, in only 1 state in the northeastern U.S. were pied-billed grebes detected on an adequate number of BBS routes (> 10) to assess population trends (Table 1). Pied-billed grebes use localized wetland habitats that usually occur away from roads, and the grebe's seasonal peak of vocal activity (May) tends to occur earlier in the year than BBS routes are run.

Surveys of pied-billed grebe populations should be conducted during the peak nesting season, and can be readily justified if coordinated with surveys of other marshbirds (Swift 1987, Manci and Rusch 1988, Gibbs and Melvin 1990), many of which also are of management concern (e.g., least bittern (*Ixobrychus exilis*), American bittern (*Botaurus lentiginosus*), and black tern (*Chlidonias niger*)). Surveys can evaluate population responses to habitat features in addition to monitoring population trends. Annual surveys (for 2-3 years) of many sites could provide baseline data on the distribution and abundance of pied-billed grebes in a region. Subsequent surveys (perhaps every 5 years) could then be used to assess population trends. Broadcast of tape-recorded calls of pied-billed grebes is useful for locating territorial birds (Kibbe 1989, Gibbs and Melvin 1990). Standardization of survey methodology is necessary for results to be comparable between years and among areas, and would be facilitated by having a single regional or national agency or organization responsible for directing and coordinating survey efforts.

### Management Procedures and Programs

Preservation of relatively large (> 10 ha) wetlands with an interspersion of dense, robust emergents, submergent vegetation, and open water, is the most urgent management need for pied-billed grebes in the Northeast. Wetlands used for breeding also need to be protected from chemical contamination, siltation, eutrophication, and other forms of pollution that harm pied-billed grebes or their food supplies. Vegetative features of habitats preferred by pied-billed grebes represent a particular stage of wetland succession ("hemi-marsh" stage of Weller and Spatcher 1965). Wetland managers, therefore, need to periodically reverse vegetative succession and open up extensive stands of emergent vegetation while maintaining suitable habitats nearby to serve as alternate nesting areas for pied-billed grebes during wetland manipulations. In the northeastern states, wetlands characterized as deep, fresh marshes are perhaps best suited for nesting pied-billed grebes. For states with substantially reduced populations of pied-billed

grebes (e.g., Rhode Island, Connecticut, and New Jersey), creation of nesting habitats may be necessary to restore viable nesting populations. Managed impoundments could also bolster nesting populations in areas where marshlands were scarce originally (e.g., central Pennsylvania, central and western Virginia, and West Virginia).

Because pied-billed grebes occur in many wetlands managed for waterfowl by state and federal agencies, there is ample opportunity for making minor alterations to existing management schemes to improve nesting habitat for pied-billed grebes. For example, portions of dense stands of cattail and bulrush, which are often removed with cutting, burning, or flooding treatments to improve waterfowl habitat, should be retained as nesting sites for pied-billed grebes. Maintaining stable water levels during the nesting season prevents flooding of nests (a major source of reproductive failure) and predator access. Pied-billed grebes are heavily dependent on aquatic-bed vegetation (floating-leaved and submergent), and management activities that promote establishment and growth of such vegetation will benefit pied-billed grebes. Manipulation of water levels (Fredrickson and Taylor 1982) provides a cost-effective method for establishing dense stands of emergent vegetation while retaining open-water areas preferred by pied-billed grebes.

Given the dependence of nesting pied-billed grebes on Odonates and small fish (Wetmore 1924), complete drying during wetland drawdowns should be avoided to prevent die-offs of dragonflies, many of which overwinter in late instars rather than in drought-resistant eggs (Orians 1980), and fish. The presence of carp may significantly lower availability of invertebrate foods for grebes and other waterbirds. Liming and fertilizing dikes and adjacent fields can increase the productivity and raise the pH of nutrient-poor, acidic wetlands in the northeastern region. Population monitoring programs should accompany any management activities to determine the success or failure of wetland manipulations. Floating nests of grebes are easily washed over and capsized by wave action, and thus large, motorized boats should be excluded from marshes with nesting grebes. Also, recreation activity should be restricted during the

nesting season to avoid disturbance to incubating grebes.

### Research Needs and Programs

- Conduct surveys to better determine relative abundance and distribution of pied-billed grebes in the Northeast.
- Develop standardized survey techniques and implement programs to monitor trends in populations and habitat availability.
- Conduct detailed studies of the floristic and structural composition of wetland vegetation, water levels and water quality, and minimum wetland area associated with the occurrence of pied-billed grebes during nesting and migration.
- Determine the effects of diseases, parasites, contaminants and weather on pied-billed grebes.
- Investigate effects of altering management strategies at wetland impoundments managed primarily for waterfowl in order to benefit pied-billed grebes.
- Monitor contaminant levels in pied-billed grebes and their eggs in agricultural and industrialized regions of the northeastern region.
- Determine major migration stop-over sites and study the over-wintering habitats and biology of pied-billed grebes.
- Evaluate the effects of invasion of phragmites (*Phragmites australis*) and purple loosestrife (*Lythrum salicaria*) on pied-billed grebes.

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# AMERICAN BITTERN

## *Botaurus lentiginosus*

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American bitterns inhabit freshwater wetlands with more diverse and less dense vegetation and shallower water depths than the sympatric least bittern. American bitterns are more abundant in Canada than the U.S. and populations decline sharply south of northern New England and New York. Breeding Bird Survey data suggest a 2.4% annual decline in U.S. populations between 1966 and 1989, although no significant trends were evident for populations in the eastern U.S. or Canada. Other sources suggest that declines have occurred in portions of New York and in southern New England, Pennsylvania, New Jersey, and Delaware. Standardized surveys that adequately monitor populations and habitats are needed. Preservation of shallow freshwater wetlands with abundant emergent vegetation interspersed with patches of open water and aquatic-bed vegetation is the most urgent management need. Wetlands  $\geq 2.5$  ha may support nesting, while smaller wetlands may serve as alternate foraging sites. American bitterns readily use wetlands created by impoundments, a trait that could facilitate establishment of populations in areas where natural wetlands have been destroyed. Slow, incomplete drawdowns of managed wetlands can promote emergent vegetation and create favorable water levels, while maintaining prey populations of small fish, amphibians, and aquatic invertebrates.

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### DESCRIPTION

#### Taxonomy

The American bittern (*Botaurus lentiginosus*) is a member of the heron family (Ardeidae)

within the large order of wading birds, the Ciconiiformes. The bittern subfamily, Botaurinae, may represent the oldest and most highly derived of the four groups within the heron family (Payne and Risley 1976). In comparison to other members of the heron family, bitterns are more

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solitary, and have shorter legs and a highly compressed trunk as adaptations for moving through densely vegetated habitats. Also, bitterns are more dependent on vocal than visual means of communication, hence the "booming" calls characteristic of bitterns.

The Botaurinae is comprised of eight species of small bitterns (*Ixobrychus* spp.) and four species of larger bitterns (*Botaurus* spp.). Among larger bitterns, the two Old World species (Eurasian bittern, *B. stellaris*, and Australian bittern, *B. poiciloptilus*) may represent a single superspecies, whereas the two New World species, the American bittern (*B. lentiginosus*) and South American bittern (*B. pinnatus*), differ sufficiently in skeletal characteristics from one another and from the Old World bitterns to be considered separate species (Payne and Risley 1976). The American bittern is considered a monotypic species, i.e., no subspecies have been designated (Cramp 1977). A similar synopsis of the natural history and conservation of the American bittern has recently been completed by Hands et al. (1989) for midwestern U.S. populations.

### Morphology and Plumages

The American bittern is a brown, medium-sized heron, 60-85 cm long, with a stout body and neck and relatively short legs (Palmer 1962, Cramp 1977, Hancock and Kushlan 1984). Adult plumage is all brown above (finely flecked with black) and heavily streaked with brown and white below. The crown is rusty-brown. An elongated, black patch extends from below the eye down the side of the neck, a characteristic unique among herons (Hancock and Kushlan 1984). The throat is white. Sexes are similar, except that the male is slightly larger (Palmer 1962). Juveniles differ only in lacking black neck patches, which are obtained in the first winter. Plumage does not change seasonally. In flight, the American bittern seems hurried, ungraceful, and stiff. When disturbed American bitterns often freeze in an upright, concealing posture, with head and bill upturned.

### Vocalizations

During the breeding season, male American bitterns repeat from 2-10 times a distinctive, far-carrying call, rendered as "*pump-er-lunk*" by Palmer (1962), and which is often preceded by a series of clicking and gulping sounds. A detailed, anatomical description of how American bitterns vocalize, through inflation of the esophagus, was provided by Chapin (1922). American bitterns rely on their resounding calls to communicate among the dense, visually restricting emergent vegetation that dominates nesting habitats. Low frequency sounds, such as the American bittern's "booming," attenuate less rapidly and are audible at greater distances in dense marsh vegetation than high frequency sounds (Cossens and Falls 1984). When flushed from a marsh, American bitterns often emit a hoarse "*kok-kok-kok*" or nasal "*haink*" (Palmer 1962).

### HABITAT

The American bittern breeds primarily in freshwater wetlands with tall emergent vegetation. Sparsely vegetated wetlands and dry grassy uplands are sometimes used, as are tidal marshes in some areas (Stewart and Robbins 1958, Swift 1987). In comparison to the sympatric least bittern (*Ixobrychus exilis*), the American bittern uses a wider variety of wetland cover types, less densely vegetated sites, shallower water depths, and primarily freshwater habitats.

Wetlands used by American bitterns in Maine were dominated by emergent and aquatic-bed (floating-leaved and submergent) vegetation, had a high diversity of vegetative life forms, and a high degree of cover/water interspersion (Gibbs et al. in press; Gibbs and Melvin 1990). Portions of wetlands used by American bitterns were dominated by sedges (*Carex* spp.), broad-leaved cattail (*Typha latifolia*), and ericaceous shrubs. In a study of Quebec lakes, American bitterns preferred lakes with patches of floating-leaved plants, emergent growth along shorelines, and abundant amphibian populations (DesGranges and Houde 1989).

At Horicon Marsh in Wisconsin, American bitterns responded to tape-recorded calls only from shallow water cattail and dry cattail habitats, and seemed to avoid deepwater cattails (Manci and Rusch 1988). At moist soil impoundments in Missouri, American bitterns were associated with water depths < 10 cm and vegetative cover characterized as "rank, dense, or sparse." Habitat use was not associated with "open" or "short" vegetative cover or water of depths > 10 cm (Fredrickson and Reid 1986). In Minnesota, 7 breeding territories of American bitterns had a mean water depth of 10 cm, vegetation height of 1.3 m, and density of sedge and grass stems of 117 stems/m<sup>2</sup> (Hanowski and Niemi 1986).

American bitterns in Maine inhabited wetlands from < 1 to > 25 ha in size, but were more abundant in larger than smaller wetlands, and preferred impoundments and beaver (*Ondatra zibethicus*)-created wetlands to wetlands of glacial origin (Gibbs et al. in press, Gibbs and Melvin 1990). Eaton (1910) suggested that the American bittern in New York occurred at marshes > 4 ha. In Iowa, Brown and Dinsmore (1986) observed American bitterns only on wetlands > 11 ha and suggested that occurrence of the species at wetlands was possibly area-dependent.

Migrant American bitterns were flushed at 25 sites during spring in Missouri with mean water depth of 26 cm, vegetation height of 63 cm, and stem density of 157 stems/m<sup>2</sup>. Characteristics of 35 flush sites in fall were similar, except that vegetation was taller (118 cm) (Reid 1989). American bitterns overwinter in areas where temperatures stay above freezing and waters remain open, especially in coastal regions where the ocean moderates climate (Root 1988). Wintering habitat is much like breeding habitat, and overwintering American bitterns are heavily dependent on managed wetland areas, such as those occurring at wildlife refuges (Root 1988). Overwintering bitterns occasionally use brackish coastal marshes (Hancock and Kushlan 1984).

## BIOLOGY

### Feeding

The trunk of American bitterns is highly compressed to facilitate movement through dense vegetation. The American bittern's repertoire of feeding behaviors is relatively small. Of 28 recognized heron foraging behaviors (Kushlan 1978), only 4 are used by American bitterns: standing in place, neck swaying (which may enable bitterns to overcome glare or permit a quicker strike by having muscles in movement when strike begins), walking slowly, and walking quickly. The American bittern is a solitary feeder and relies more on stealth than pursuit to capture prey. The species' coloration, particularly its ventral stripes, provides camouflage in dense, vertical marsh vegetation, complements its inactive feeding repertoire, and permits solitary foraging (Kushlan 1978). Crypsis in the American bittern is thought to function mostly to reduce visibility to prey and competitors rather than to predators (Kushlan 1978).

The American bittern's long, thick bill enables it to take a variety of large and small prey (Kushlan 1978), a conclusion supported by analyses of stomach contents. Based on 160 specimens (133 with food remains) collected throughout North America, stomach contents included insects (23% of items, including many Odonates), frogs and salamanders (21%), fish (21%, catfish, sunfish, yellow perch, suckers, killifishes, and sticklebacks), crayfish (19%), mice and shrews (10%), snakes (5%), and small quantities of crabs, spiders, and unidentified invertebrates (Cottam and Uhler 1945). Nine stomachs of American bitterns from Pennsylvania contained fish, frogs, crayfish, watersnakes, snails, beetles, and grasshoppers (Warren 1890). American bitterns have been observed intercepting dragonflies in midair (Dudones 1983). Garter snakes (*Thamnophis sirtalis*) feeding on frogs at pond margins are occasionally eaten by American bitterns (Ingram 1941). A captive American bittern ate 23.9 g (dry weight) of food (mice) per day, and required 22 hours to digest a meal and eject a pellet (Rhoades and Duke 1975).

## Nesting

American bitterns arrive at nesting areas in the northeastern U.S. in mid to late March, about a month before the more southerly wintering least bittern (Bent 1926), but peak numbers of birds may not arrive until mid-April (Palmer 1962). Nesting occurs solitarily (non-colonially) on all-purpose territories that provide both feeding and nesting sites. Polygyny is suspected to occur (Palmer 1962). Mating rituals are elaborate, and involve display of a tuft of white nuptial feathers located on the sides of the neck (see Baker 1980, Johnsgard 1980). *Botaurus* bitterns may undergo a change in iris color from yellow to orange during courtship (Wood 1986).

The American bittern's nest consists of a platform of reeds, sedges, cattail, or other available emergent vegetation, and is lined with fine grasses. Nests are usually placed on the ground or a tussock, are surrounded by water, and have dense, overhead cover (Bent 1926, Vesall 1940, Harrison 1978). American bitterns also may nest in grassy uplands (Vesall 1940, Duebbert and Lokemoen 1977). Nests are usually accessed by two, well-beaten pathways (Gabrielson 1914).

Egg-laying begins in late April or early May, about a month after the bitterns' arrival at nesting areas, and continues until mid-June (Bent 1926). Eggs measure 49 by 37 mm and are elliptical, buff-brown to deep olive-brown, smooth, and slightly glossy (Harrison 1978). Clutch size ranges from 2-7 eggs, usually 3-5 (Duebbert and Lokemoen 1977, Gruber et al. 1978). Incubation, by the female only, begins with the first egg (Palmer 1962). Hatching occurs after 24-28 days (Burns 1915, Bent 1926, Vesall 1940), and chicks remain at or near the nest for 2 weeks (Gabrielson 1914, Vesall 1940). Chicks are fed only by the female, and are given fish, frogs, snakes, crayfish, and mice (Gabrielson 1914, Byers 1951). Nests often become fouled with food debris (Vesall 1940). Post-fledging behavior of young is unreported.

American bitterns apparently are single-brooded (Palmer 1962). In upland habitats in North and South Dakota, 57% of 72 nests hatched at least 1 egg (Duebbert and Lokemoen 1977). Little is known about sources of egg, chick or post-juvenile mortality, age at fledging or first

breeding, or territory size. Age at fledging for the closely-related Eurasian bittern is 50-55 days (Cramp 1977). One banded American bittern lived at least 8 years (Clapp et al. 1982). American bitterns probably undergo extensive, post-breeding dispersal, which has resulted in numerous sightings, mostly between September and December, at locations as distant as Iceland, Norway, and Great Britain (Cramp 1977). Information on molt is incomplete. Adults undergo a complete post-breeding molt from August to November, and possibly a pre-breeding renewal of body plumage (Palmer 1962, but see Bent 1926).

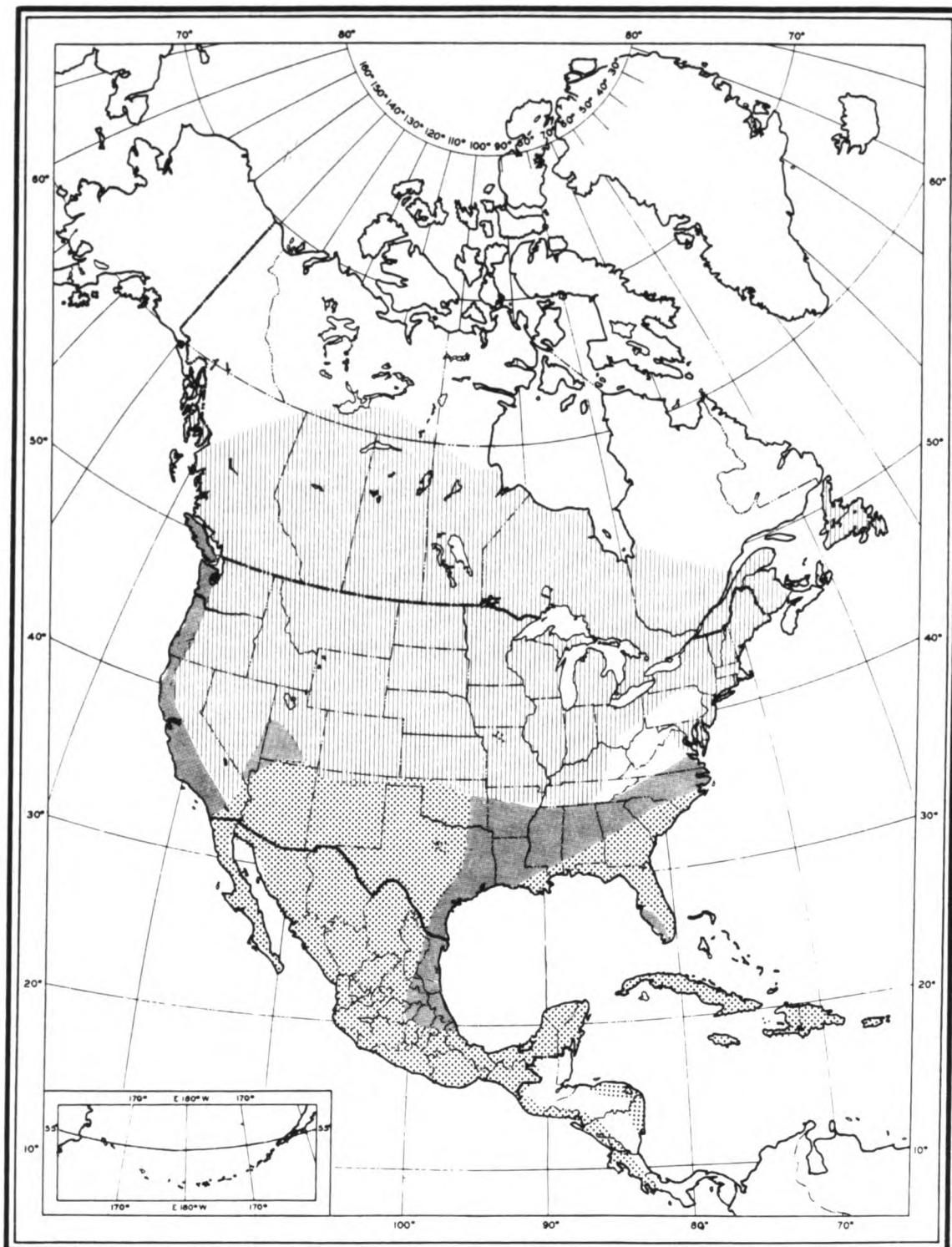
## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

The breeding range of the American bittern extends from the mid-U.S. into northern Canada (Figure 1). The northern limit spans from about 55° in British Columbia, north to Great Slave Lake in the Northwest Territories, along the southern shores of Hudson Bay, and east to Newfoundland. Breeding is discontinuous south of Pennsylvania, Ohio, Indiana, Illinois, Missouri, Kansas, Colorado, Utah, Nevada, and California, occurring locally south to Texas, Louisiana, Florida, and Mexico (Hancock and Kushlan 1984). In the northeastern U.S., breeding is continuous throughout much of New England, northern and western New York, and New Jersey, and extremely patchy or absent elsewhere (Figure 2).

During the period 1966-89, data from U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; Robbins et al. 1986) showed a significant ( $P < 0.05$ ) decline (-2.4 % per year) in American bittern populations in the U.S. (Table 1). Canadian populations showed no significant trends and populations in the eastern states also remained stable (Table 1). Notably, American bitterns were encountered, on average, 3.2 times more frequently on BBS routes in Canada (0.82

Figure 1. The North American range of the American bittern (*Botaurus lentiginosus*) (American Ornithologists' Union 1983, Hancock and Kushlan 1984).

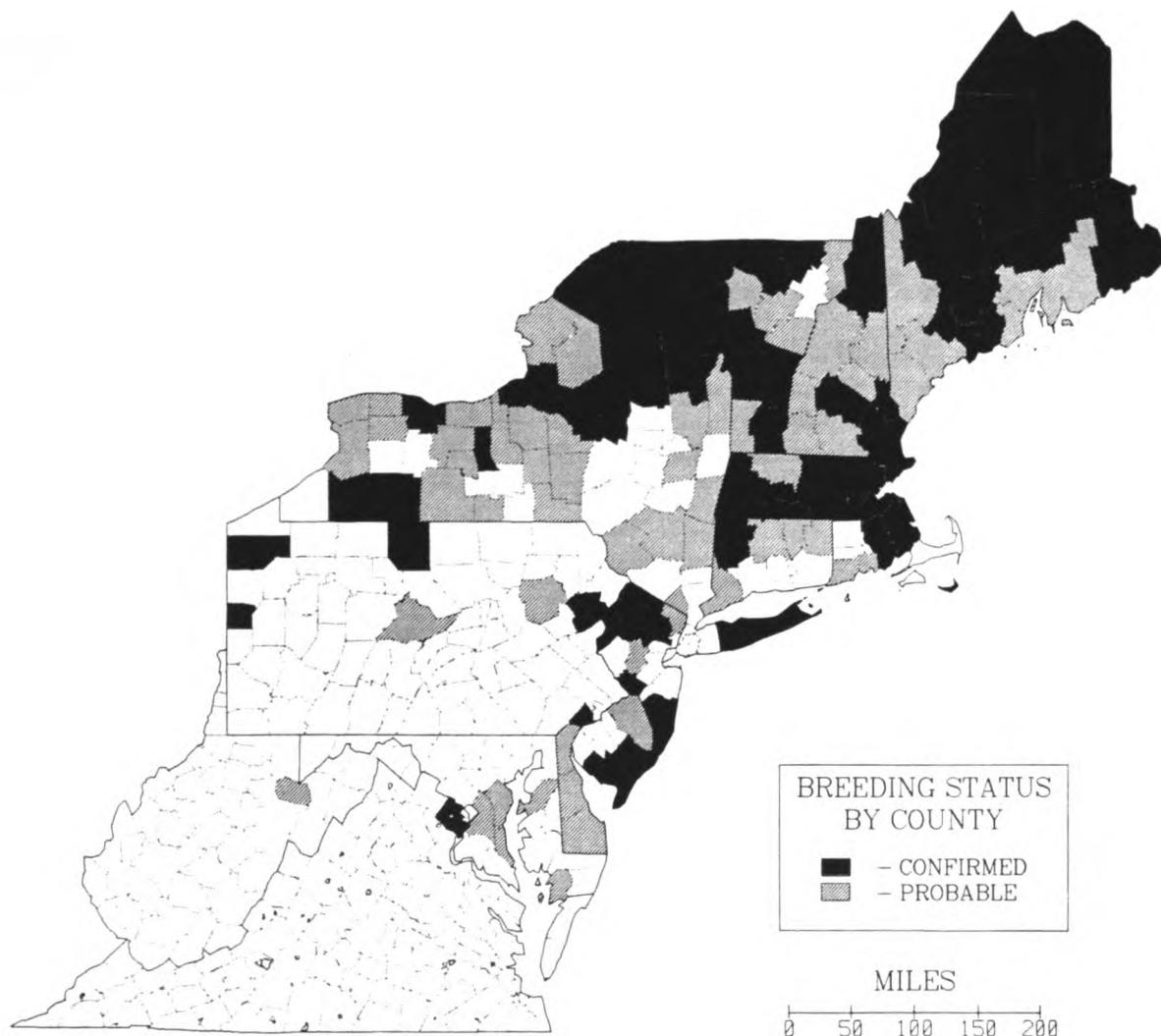


Breeding

Year-round

Wintering

**Figure 2.** Northeast breeding distribution of the American bittern (*Botaurus lentiginosus*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



**Table 1.** Mean number of American bitterns (*Botaurus lentiginosus*) per Breeding Bird Survey Route and median percent annual change, 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; see Robbins et al. 1986 for details).

Region	No. routes <sup>a</sup>	$\bar{x}$ birds/route	% change/year <sup>b</sup>
<b>State</b>			
Maine	27	0.25	+1.1
N. Hampshire	19	0.38	+2.5
Vermont	14	0.23	-1.3
Massachusetts	7	0.04	-0.9
Rhode Island	0	-	-
Connecticut	2	0.03	-3.1***
New York	45	0.12	+0.2
Pennsylvania	7	0.01	-1.3
New Jersey	5	0.03	-7.5
Delaware	3	0.02	-1.6
Maryland	4	0.01	+0.8
Virginia	0	-	-
West Virginia	0	-	-
Eastern U.S.	407	0.42	+0.2
U.S.	401	0.26	-2.4**
Canada	238	0.82	-1.0

<sup>a</sup>Number of routes on which American bitterns were detected, or at a national scale, used for analysis.

<sup>b</sup>Statistical significance of trend: \*\*  $P \leq 0.05$ , \*\*\*  $P < 0.01$ .

birds/route) than in the U.S. (0.26 birds/route), indicating a latitudinal trend in abundance.

The American bittern was listed as a *migratory nongame bird of management concern* by the U.S. Fish and Wildlife Service in 1982 (Regions 3 and 5) and again in 1987 (Regions 3, 4, 5, and 6) because a nationwide, downward trend in populations was then evident and the species is patchily distributed and dependent on rapidly disappearing wetland habitats (U.S. Fish and Wildlife Service 1987). Causes of population declines identified in the report by respondents to a survey were habitat loss (100% of respondents), human disturbance (8%), and pesticides/contaminants (17%). The American bittern has been *blue-listed* by the National Audubon Society from 1976-86 (Tate 1986). Birdwatchers had

reported population declines in mid-continent areas by the late-1970s, in the Atlantic coastal region by 1980, and in the entire continent by 1986 (Tate 1986). The American bittern has a Natural Heritage Program ranking of *G4*, indicating the species is apparently secure globally (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

Overwintering populations occur along the southern coastal plain in areas where temperatures remain above freezing (Root 1988). Concentrations of overwintering American bitterns occur in wetlands on the coast of Virginia and North Carolina, Okefenokee Swamp in Georgia and Florida, the Everglades of southern Florida, Noxubee National Wildlife Refuge in Mississippi, swamps along the Louisiana coast (including Sabine and Lacassine National Wildlife Refuges and Rockefeller and Russel Sage state refuges), and Honey Lake, the Salton Sea National Wildlife Area, and the San Joaquin River in California (Root 1988). Some populations are nonmigratory, particularly in milder zones in southern parts of the range, and a small population overwinters at the Ottawa National Wildlife Refuge on Lake Erie (Root 1988). American bitterns also migrate to the Greater Antilles and as far south as Panama (Palmer 1962). In the northeastern states, American bitterns overwinter north to about Delaware (Bull 1974).

## Distribution and Status in the Northeast Region

**Maine.** - Forbush (1925) considered the American bittern an uncommon local summer resident in Maine. Palmer (1949) described the American bittern as an uncommon to fairly common summer resident throughout the state in marshes, swamps, swales, moist grassy meadows, and alder (*Alnus* spp.) thickets. American bittern populations along BBS routes in Maine showed no trends during the period 1966-89 (Table 1). Encounter rates along BBS routes in Maine were among the 3 highest reported for the 13 northeastern states (Table 1). Surveys for the Maine Breeding Bird Atlas (BBA) (Adamus 1988) reported confirmed or probable breeding in 24

minimum coverage blocks (58% of total) distributed in all counties of the state (Figure 2). A 12-year study in central and eastern Maine observed American bitterns during summer at 53% of 87 wetlands studied (Gibbs et al. in press). The relative abundance of American bitterns (Table 1), their extensive distribution in the state (Adamus 1988, Gibbs et al. in press), and the existence of state laws protecting major (> 10 ac) wetland breeding habitats suggest that currently there are no major local threats to the continued presence of the species in Maine. The species is ranked as S3 (rare or uncommon) in Maine by the Maine Natural Heritage Program, and has no special listing in the state (Table 2).

**Table 2. Summary of state listing status and Natural Heritage Program state ranks for the American bittern (*Botaurus lentiginosus*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	S3
New Hampshire	-	S3
Vermont	-	S3
Massachusetts	special concern	S1
Rhode Island	endangered	S1
Connecticut	endangered	S1
New York	-	S4
Pennsylvania	threatened	S1
New Jersey	threatened <sup>b</sup>	S3
Delaware	-	S3
Maryland	special concern	S3
Virginia	-	SU
West Virginia	-	S1

<sup>a</sup>S1 = critically imperiled; ≤ 5 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

SU = status uncertain

<sup>b</sup>Breeding status only

**New Hampshire.** - The American bittern was characterized as a "not uncommon" breeder at freshwater marshes throughout New Hampshire

by Allen (1903). Forbush (1925) considered the American bittern an uncommon local summer resident in the state. Lake Umbagog has figured historically as an important breeding area (Allen 1903, Palmer 1949). Recent surveys (1981-86) for the New Hampshire BBA (C. Foss pers. comm.) recorded evidence of breeding in all major drainages of the state (Figure 2). American bitterns were reported from 74 locations well-distributed in the state, except in the White Mountains. Populations along BBS routes in New Hampshire did not change significantly during the period 1966-89 (Table 1), although a significantly greater proportion of routes showed decreases in bittern numbers (0.68) than increases (0.32) during the period 1966-89. Encounter rates of American bitterns were highest in New Hampshire among all northeastern states (Table 1). The species has a Natural Heritage Program ranking of S3 in New Hampshire (rare or uncommon), and has no special listing in the state (Table 2).

**Vermont.** - American bitterns were originally described as fairly common summer residents in the marshlands of Vermont by Perkins and Howe (1901). Forbush (1925) later considered American bitterns as uncommon summer residents in the state, and Fortner et al. (1933) described them as uncommon to rare; thus, a decline in American bittern populations in Vermont may have occurred during the early part of the century. Trends of American bittern populations along BBS routes in Vermont during the period 1966-89 were stable (Table 1), and encounter rates of American bitterns were among the 3 highest reported for the 13 northeastern states (Table 1). Surveys for the Vermont BBA (Laughlin and Kibbe 1985) reported American bitterns present during the breeding season in 42 priority blocks (23% of total). Most records were from extensive marshlands in the Lake Champlain lowlands. Others were scattered throughout the state, except in the Taconic Mountains, North and East Central regions, and Northeast Highlands where open wetland habitat is relatively uncommon (Figure 2). The wide distribution of breeding records suggests that American bitterns in Vermont use a variety of types and sizes of wetlands (Laughlin and Kibbe 1985). The species

has a Natural Heritage Program ranking of S3 in Vermont (rare or uncommon), and has no special listing in the state (Table 2).

**Massachusetts.** - Howe and Allen (1901) described the American bittern as a common, regular summer resident in many parts of Massachusetts. Forbush (1925) considered the American bittern a common summer resident of marshy areas of the state, particularly in river marshes, but noted that hunting had reduced populations in many parts of Massachusetts and New England where bitterns had been formerly common. Bagg and Eliot (1937) also considered the American bittern to be declining in the Connecticut River Valley (as an uncommon resident in an "*interim of scarcity*"), due, in part, to hunting. Griscom and Snyder (1955) considered American bitterns to be common summer residents in freshwater marshes at lower altitudes throughout the state, but rare in the salt marshes on Cape Cod. In the Concord area, Griscom (1949) stated that there had been no perceptible trend in American bittern populations since 1886.

American bitterns were encountered on too few BBS routes in Massachusetts to assess population trends (Table 1). Forbush (1925) characterized American bittern populations in Massachusetts as being more abundant than those in Vermont, New Hampshire, and Maine, whereas the reverse is clearly true today (Table 1). BBA data collected from 1974-79 (J. Baird pers. comm.) showed confirmed, possible, or probable breeding in all counties of the state, except for Cape Cod. Thus, the American bittern remains fairly abundant and well-distributed in Massachusetts, but seems to have undergone a significant population decline in many parts of the state (e.g., the Connecticut Valley: Bagg and Eliot 1937) but apparently not in all parts (e.g., Concord: Griscom 1949). The species has a Natural Heritage Program ranking of S1 in Massachusetts (critically imperiled), and is considered a *species of special concern* in the state because of its relative rarity and uncertain status (Table 2).

**Rhode Island.** - Howe and Sturtevant (1899) considered the American bittern to be an uncommon summer resident in Rhode Island, but

a common migrant. Nesting populations are thought to have declined steadily in the state since the 1970s, and BBA surveys during 1982-87 (R. Enser pers. comm.) found evidence of breeding at five locations (Figure 2), including two at Block Island. American bitterns were not detected along BBS routes in Rhode Island during the period 1966-86 (Table 1). The species has a Natural Heritage Program ranking of S1 in Rhode Island (critically imperiled), and is considered an *endangered species* in the state (Table 2).

**Connecticut.** - Merriam (1877) considered the American bittern a "common" summer resident in Connecticut, although Sage et al. (1913) described it as being rare. MacKenzie (1961) stated that the American bittern was a regular but uncommon breeder in the Guilford area prior to 1955, but had undergone a decline due to drainage of wetland habitats. Recent surveys for a statewide BBA (Bevier et al. in press) recorded American bitterns present during the breeding season at 34 locations (6% of blocks surveyed). Records were concentrated mostly in Litchfield County in northern Connecticut, although breeding also was evident in northeastern, central, and southwestern portions of the state (Figure 2). American bitterns were encountered on only two BBS routes in Connecticut (Table 1), too few to assess population trends. The species is now considered "...a rare and very local nester at large inland marshes" (Zeranski and Baptist 1990). Draining, filling and disturbance of wetlands and environmental contamination (especially by chlorinated hydrocarbons) are suspected as causes of the species' decline in Connecticut (Zeranski and Baptist 1990). The species has a Natural Heritage Program ranking of S1 in Connecticut (critically imperiled), and is listed as an *endangered species* in the state (Table 2).

**New York.** - Surveys for the New York BBA indicate that the American bittern is widely distributed across the state (Andrie and Carroll 1988), except for the eastern Appalachian Plateau, where habitat is limited (Figure 2). American bitterns were present during the breeding season at 534 blocks (10% of total surveyed). Concentrations of atlas records came from the Great Lakes Plain, south of the Finger and

minimum coverage blocks (58% of total) distributed in all counties of the state (Figure 2). A 12-year study in central and eastern Maine observed American bitterns during summer at 53% of 87 wetlands studied (Gibbs et al. in press). The relative abundance of American bitterns (Table 1), their extensive distribution in the state (Adamus 1988, Gibbs et al. in press), and the existence of state laws protecting major (> 10 ac) wetland breeding habitats suggest that currently there are no major local threats to the continued presence of the species in Maine. The species is ranked as S3 (rare or uncommon) in Maine by the Maine Natural Heritage Program, and has no special listing in the state (Table 2).

**Table 2.** Summary of state listing status and Natural Heritage Program state ranks for the American bittern (*Botaurus lentiginosus*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

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Vermont	-	S3
Massachusetts	special concern	S1
Rhode Island	endangered	S1
Connecticut	endangered	S1
New York	-	S4
Pennsylvania	threatened	S1
New Jersey	threatened <sup>b</sup>	S3
Delaware	-	S3
Maryland	special concern	S3
Virginia	-	SU
West Virginia	-	S1

<sup>a</sup>S1 = critically imperiled; ≤ 5 occurrences

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<sup>b</sup>Breeding status only

**New Hampshire.** - The American bittern was characterized as a "not uncommon" breeder at freshwater marshes throughout New Hampshire

by Allen (1903). Forbush (1925) considered the American bittern an uncommon local summer resident in the state. Lake Umbagog has figured historically as an important breeding area (Allen 1903, Palmer 1949). Recent surveys (1981-86) for the New Hampshire BBA (C. Foss pers. comm.) recorded evidence of breeding in all major drainages of the state (Figure 2). American bitterns were reported from 74 locations well-distributed in the state, except in the White Mountains. Populations along BBS routes in New Hampshire did not change significantly during the period 1966-89 (Table 1), although a significantly greater proportion of routes showed decreases in bittern numbers (0.68) than increases (0.32) during the period 1966-89. Encounter rates of American bitterns were highest in New Hampshire among all northeastern states (Table 1). The species has a Natural Heritage Program ranking of S3 in New Hampshire (rare or uncommon), and has no special listing in the state (Table 2).

**Vermont.** - American bitterns were originally described as fairly common summer residents in the marshlands of Vermont by Perkins and Howe (1901). Forbush (1925) later considered American bitterns as uncommon summer residents in the state, and Fortner et al. (1933) described them as uncommon to rare; thus, a decline in American bittern populations in Vermont may have occurred during the early part of the century. Trends of American bittern populations along BBS routes in Vermont during the period 1966-89 were stable (Table 1), and encounter rates of American bitterns were among the 3 highest reported for the 13 northeastern states (Table 1). Surveys for the Vermont BBA (Laughlin and Kibbe 1985) reported American bitterns present during the breeding season in 42 priority blocks (23% of total). Most records were from extensive marshlands in the Lake Champlain lowlands. Others were scattered throughout the state, except in the Taconic Mountains, North and East Central regions, and Northeast Highlands where open wetland habitat is relatively uncommon (Figure 2). The wide distribution of breeding records suggests that American bitterns in Vermont use a variety of types and sizes of wetlands (Laughlin and Kibbe 1985). The species

has a Natural Heritage Program ranking of S3 in Vermont (rare or uncommon), and has no special listing in the state (Table 2).

**Massachusetts.** - Howe and Allen (1901) described the American bittern as a common, regular summer resident in many parts of Massachusetts. Forbush (1925) considered the American bittern a common summer resident of marshy areas of the state, particularly in river marshes, but noted that hunting had reduced populations in many parts of Massachusetts and New England where bitterns had been formerly common. Bagg and Eliot (1937) also considered the American bittern to be declining in the Connecticut River Valley (as an uncommon resident in an "*interim of scarcity*"), due, in part, to hunting. Griscom and Snyder (1955) considered American bitterns to be common summer residents in freshwater marshes at lower altitudes throughout the state, but rare in the salt marshes on Cape Cod. In the Concord area, Griscom (1949) stated that there had been no perceptible trend in American bittern populations since 1886.

American bitterns were encountered on too few BBS routes in Massachusetts to assess population trends (Table 1). Forbush (1925) characterized American bittern populations in Massachusetts as being more abundant than those in Vermont, New Hampshire, and Maine, whereas the reverse is clearly true today (Table 1). BBA data collected from 1974-79 (J. Baird pers. comm.) showed confirmed, possible, or probable breeding in all counties of the state, except for Cape Cod. Thus, the American bittern remains fairly abundant and well-distributed in Massachusetts, but seems to have undergone a significant population decline in many parts of the state (e.g., the Connecticut Valley: Bagg and Eliot 1937) but apparently not in all parts (e.g., Concord: Griscom 1949). The species has a Natural Heritage Program ranking of S1 in Massachusetts (critically imperiled), and is considered a *species of special concern* in the state because of its relative rarity and uncertain status (Table 2).

**Rhode Island.** - Howe and Sturtevant (1899) considered the American bittern to be an uncommon summer resident in Rhode Island, but

a common migrant. Nesting populations are thought to have declined steadily in the state since the 1970s, and BBA surveys during 1982-87 (R. Enser pers. comm.) found evidence of breeding at five locations (Figure 2), including two at Block Island. American bitterns were not detected along BBS routes in Rhode Island during the period 1966-86 (Table 1). The species has a Natural Heritage Program ranking of S1 in Rhode Island (critically imperiled), and is considered an *endangered* species in the state (Table 2).

**Connecticut.** - Merriam (1877) considered the American bittern a "common" summer resident in Connecticut, although Sage et al. (1913) described it as being rare. MacKenzie (1961) stated that the American bittern was a regular but uncommon breeder in the Guilford area prior to 1955, but had undergone a decline due to drainage of wetland habitats. Recent surveys for a statewide BBA (Bevier et al. in press) recorded American bitterns present during the breeding season at 34 locations (6% of blocks surveyed). Records were concentrated mostly in Litchfield County in northern Connecticut, although breeding also was evident in northeastern, central, and southwestern portions of the state (Figure 2). American bitterns were encountered on only two BBS routes in Connecticut (Table 1), too few to assess population trends. The species is now considered "...a rare and very local nester at large inland marshes" (Zeranski and Baptist 1990). Draining, filling and disturbance of wetlands and environmental contamination (especially by chlorinated hydrocarbons) are suspected as causes of the species' decline in Connecticut (Zeranski and Baptist 1990). The species has a Natural Heritage Program ranking of S1 in Connecticut (critically imperiled), and is listed as an *endangered* species in the state (Table 2).

**New York.** - Surveys for the New York BBA indicate that the American bittern is widely distributed across the state (Andrie and Carroll 1988), except for the eastern Appalachian Plateau, where habitat is limited (Figure 2). American bitterns were present during the breeding season at 534 blocks (10% of total surveyed). Concentrations of atlas records came from the Great Lakes Plain, south of the Finger and

Oneida Lakes, along the St. Lawrence River, and the Lake Champlain Valley. Many records were from protected wetlands at state and national wildlife refuges. Other concentrations of records came from riverine wetlands in the Adirondacks and tidal marshes along the Hudson River, although Swift's (1987) study suggests that American bitterns rarely breed in Hudson River tidal marshes, but depend on them heavily during migration.

American bittern populations along BBS routes in New York during the period 1966-89 appeared stable (Table 1). Despite the large number of BBS routes in New York on which the species was detected ( $n = 45$ , Table 1), sighting frequencies were about half those observed among other states along the northern U.S. border (Maine, New Hampshire, and Vermont). Although BBS data have shown no significant trends (Table 1), historical declines in populations have, nevertheless, occurred in portions of the state. Populations in the Cayuga Lake basin declined from 1940-70 due, perhaps, to loss of marshlands there (Temple and Temple 1976). Jorde et al. (1989) described rates of wetland loss in the agricultural area of central New York, where Temple and Temple's (1976) analysis was focused, as among the highest in the north Atlantic states. Also, historical writings (ca. 1900) describe populations of American bitterns in New York seemingly more abundant than at present (Eaton 1910). In western New York, Beardslee and Mitchell (1965) reported that numbers of American bitterns had "*markedly decreased*." In the New York City area, however, Bull (1974) stated that no obvious change in size or status of American bittern populations had occurred. Thus, American bittern populations have evidently declined in portions of New York State during this century, but the magnitude and geographic extent of the decline is not clear. The species has a Natural Heritage Program ranking of S4 in New York (apparently secure, Table 2), but has no special state listing.

**Pennsylvania.** - Warren (1890) considered the American bittern an occasional summer resident in Pennsylvania, with reports of breeding concentrated in the extreme northwestern corner of the state and scattered elsewhere, a pattern

similar to that observed today (Gill 1985). Sutton (1928) described the species as being fairly common in the state. Poole (1964) considered the American bittern to be a regular but uncommon breeder at large marshes below 2,000 ft (610 m) in elevation over much of the state, but most common in Crawford, Erie and Mercer counties (see also Todd 1940). Recent BBA surveys found evidence of breeding in six counties scattered throughout the state (Brauning in press, Figure 2).

American bitterns were encountered on too few BBS routes in Pennsylvania to assess population trends (Table 1). The species is listed as threatened in Pennsylvania (Table 2), and continued loss of marshlands, which were originally scarce in the state (Todd 1940), may lead to extirpation of already diminished and isolated nesting populations (Gill 1985). The species has a Natural Heritage Program ranking of S1 in Pennsylvania (critically imperiled, Table 2).

**New Jersey.** - The American bittern has been a regular but local breeder in freshwater marshes in New Jersey throughout this century (Stone 1908, Leck 1984, Figure 2). At Cape May, wetland drainage evidently caused declines in local nesting populations (Stone 1937), and inland, freshwater wetlands continue to be threatened by development (Tiner 1984). Data from a recent BBA project indicate that extant populations occur mainly in the northern and southern portions of the state (D. Hughes pers. comm., Figure 2). American bitterns were encountered on too few BBS routes in New Jersey to assess population trends (Table 1). The American bittern has a Natural Heritage ranking of S3 in New Jersey (rare or uncommon), and a state listing of *threatened* (breeding population only) (Table 2).

**Delaware.** - American bitterns are probably more abundant in Delaware in winter than summer (Harding and Harding 1980), but few data are available to assess statewide status or population trends. Christmas Bird Counts suggest that overwintering populations have declined by 70% over the last 25 years. Recent surveys (1983-87) for a state BBA found American bitterns

during the breeding season at 10 locations, mostly in central and southern Delaware (Hess et al. in press, Figure 2). Whereas nesting formerly occurred at both inland and tidewater marshes, loss and disturbance of inland wetlands now restricts nesting to estuarine areas, where total summer populations are probably less than 100 birds. American bitterns were encountered on only two BBS routes in Delaware (Table 1), too few to assess population trends. The species has a Natural Heritage Program ranking of S3 in Delaware (rare or uncommon), and no special listing in the state.

**Maryland and the District of Columbia.** - Unlike in many other states, the American bittern breeds chiefly in tidal wetlands in Maryland. Stewart and Robbins (1958) considered the species as fairly common in marshes along the southeastern reaches of Chesapeake Bay (Somerset, Wicomico, and Dorchester counties), but uncommon along the Eastern Shore, Western Shore, and the Upper Chesapeake, and rare in the Allegheny Mountain section. Recent surveys for the first BBA of Maryland and the District of Columbia (G. Therres pers. comm.) found American bitterns during the breeding season at 5 counties surrounding Chesapeake Bay (Figure 2). American bitterns were encountered on too few BBS routes in Maryland to assess population trends (Table 1). The species has a Natural Heritage Program ranking of S3 in Maryland (rare or uncommon), and a state listing of *special concern* (Table 2).

**Virginia.** - The American bittern is considered a rare summer resident in tall marshes and wet fields along the coastal plain (Virginia Society Ornithology 1979). Several summer occurrences are recorded for counties west of the Piedmont (Virginia Society Ornithology 1979). Although a regular breeder along the coastal plain 80 years ago, nesting within the last 20 years was only recently confirmed (1989) near Alexandria during the State's BBA survey (S. Ridd and R. Wadja pers. comm.). American bitterns were not detected along BBS routes in Virginia during the period 1966-86 (Table 1). During migration and in winter, the American bittern is considered uncommon or rare throughout the state (Virginia

Society Ornithology 1979). The species has a Natural Heritage Program ranking of SU in Virginia (possibly in peril but status uncertain), and no special listing in the state (Table 2).

**West Virginia.** - In West Virginia, at the southern edge of its breeding range, the American bittern is considered a rare to uncommon summer visitor (Hall 1983). Although there exist widespread reports of summer residents from 13 counties, few breeding records are known (Hall 1983), and many summer residents might be nonbreeders that failed to migrate. Recent statewide BBA surveys (1984-88) observed breeding activity at just five locations in the state (C. Stihler pers. comm.). Availability of marshland severely limits the distribution and abundance of American bitterns in West Virginia, where only 0.3% of the state's area is under water, primarily at unvegetated impoundments (Hall 1983). Summer records are primarily restricted to boreal bogs in the state (Anderson et al. 1981), and the species may disappear if the relatively few bogs at which it nests are disturbed. American bitterns were not detected along BBS routes in West Virginia during the period 1966-86 (Table 1). The species has a Natural Heritage Program ranking of S1 in West Virginia (critically imperiled), but no special listing by the state (Table 2).

## Summary

Few quantitative data are available to assess past trends in size and distribution of American bittern populations. In parts of the northeastern U.S., however, populations appear relatively large and stable, e.g., in Maine, New Hampshire, and Vermont. Substantial populations also occur in New York, but have declined markedly in parts of the state during this century (Beardslee and Mitchell 1965, Temple and Temple 1976). Elsewhere, populations seem to be heavily reduced, and perhaps still declining, e.g., in Massachusetts, Rhode Island, Connecticut, Pennsylvania, New Jersey, and Delaware. Populations in Maryland are small but seem stable. American bitterns in Virginia and West Virginia are rare and at the southern edge of their breeding range.

Loss of wetland habitats is clearly the major cause of declines, and resulted in population reductions in some states, e.g., Massachusetts, as early as 100 years ago (Forbush 1925, Bagg and Eliot 1937). Hunting also may have substantially reduced populations in many areas prior to the early 1900s (Forbush 1925). The continued presence of American bitterns in many states depends on preventing further loss and degradation of marshlands. Many important nesting areas are now protected as state and federally-managed waterfowl impoundments (e.g., in New York), and current efforts to protect and create additional wetland habitat for waterfowl in the northeastern states should benefit the American bittern.

The dependence of the American bittern on inland, freshwater marshlands suggests that this ancient species may be a relict over much of the U.S. (Payne and Risley 1976). The species' range may historically have been slowly shifting northward, tracking the distribution of wetlands created by retreating glaciers. Based on BBS sighting frequencies (an index of population size), the abundance of American bitterns is substantially greater in Canada than the U.S., and declines sharply in the U.S. below the northern border states (Table 1). Thus, the American bittern may be primarily adapted to northern climates and wetlands, and now occurs sparsely in the U.S. over what represents the southern edge of its breeding range. Declines in American bittern populations in the U.S. may represent a hastening of an ongoing northern retreat by the species.

## LIMITING FACTORS AND THREATS

The most serious factor limiting American bittern populations is availability of wetland habitat. The American bittern's entire life cycle is dependent on wetlands, yet over half the original wetlands in the conterminous U.S. have already been destroyed (Tiner 1984). The most serious losses have occurred among palustrine emergent wetlands, of which about 4.75 million acres (1.92 million ha) were lost between the mid-1950s and mid-1970s (Tiner 1984). Inland, freshwater wetlands, the most important nesting

and wintering areas of American bitterns, are among the most threatened habitats (Tiner 1984). Larger wetlands ( $> 10$  ha) may support large portions of regional nesting populations, and loss of these wetlands can be critical to populations in many areas. Small wetlands ( $< 5$  ha) may serve as important alternate feeding sites for American bitterns and as "stepping stones" during movements between larger wetlands, but receive no legal protection in most states. Habitat quality also may be limiting. Eutrophication, siltation, chemical contamination, and human disturbance may seriously reduce habitat quality, even at large protected wetlands. Wetland eutrophication, caused by inflow of nitrates and phosphates from urban and agricultural areas, in combination with silting and turbidity, has damaged the aquatic fauna that serve as prey for endangered Eurasian bitterns in Great Britain (Day and Wilson 1978). Marshland invasion by purple loosestrife (*Lythrum salicaria*) and phragmites (*Phragmites australis*) may substantially alter waterbird habitats in the northeastern U.S., but its effects on American bitterns have not been assessed.

Acid deposition is a potential threat to American bitterns given their dependence on wetland habitats vulnerable to acidification, the high proportion of amphibians in their diet (Cottam and Uhler 1945), and the large numbers of bitterns and large proportion of the breeding range in regions receiving rainfall of reduced pH (the northeastern U.S. and central and eastern Canada). Amphibians are typically vulnerable to strong reductions in water pH (Freda 1985), and the Eurasian bittern avoids waters with pH  $< 4.5$  (Cramp 1977), probably because such waters lack amphibian and fish prey. However, American bitterns usually occupy habitats with substantial growths of emergent vegetation, even in more northerly parts of their range (DesGranges and Houde 1989, Gibbs et al. in press.). Such areas typically are of circumneutral pH and are chemically buffered against strong shifts in acidity.

Little is known about the effects of contaminants on American bitterns. However, heavy metals, PCBs, and organochlorines have been found in tissues of a number of heron species (Fleming et al. 1983), despite bans on use of some of these chemicals (i.e., organochlorines) since the 1970s. Agricultural chemicals may have

significant, indirect effects on American bitterns by entering wetlands via runoff from upland areas and reducing prey populations. Many prey of American bitterns, including aquatic insects, crayfish, and amphibians, are vulnerable to agricultural pesticides.

Too little is known of rates of predation on nests and adult American bitterns to infer the importance of predation as a factor limiting populations. Similarly, only minimal information is available on the effects of parasites and disease on American bitterns. Wading birds are known to be susceptible to oil toxicosis (moderately), Type C botulism (occasionally), avian cholera (infrequently), chlamydial infections (frequently), sarcocystis (rarely), and aspergillosis (infrequently) (Friend 1987). American bitterns are known to harbor lice and flies (Peters 1936) and a number of species of nematodes (Boyd 1966). A nematode parasite, *Eustrongylides*, contracted from small fish, can devastate wading bird populations, and thrives in waters polluted with nutrients and silt; bittern populations at wetlands and impoundments receiving stormwater and runoff from residential and agricultural areas may be particularly vulnerable to epizootics of Eustrongyliosis (P. Frederick pers. comm.).

Illegal shooting incidental to upland gamebird or waterfowl hunting has an unknown impact on American bittern populations. Records maintained by the Massachusetts Division of Fisheries and Wildlife indicate that at least three American bitterns were shot and killed on or adjacent to wildlife management areas in that state during pheasant hunting seasons between 1986-90 (B. Blodget and W. Easte pers. comm.). Historically, hunting may have been a greater source of mortality than it is today. Audubon (1940) reported that American bitterns were "common" in markets of New Orleans in the early 1800s. Several of the species' common names, such as Indian Hen and Meadow Hen, may reflect its popularity as food for native Americans and colonists. In Europe, illegal hunting has a significant impact on Eurasian bittern populations and has led to declines and prevented population recovery in many areas (Day 1981).

Human disturbance in or along the margins of wetlands has unknown impacts on the reproduction, feeding ecology or site fidelity of

American bitterns. In Great Britain, recreational boaters are thought to limit the availability of undisturbed feeding sites for Eurasian bitterns at protected wetlands (Bibby 1981).

## MANAGEMENT POTENTIAL

Because *Botaurus* bitterns undergo extensive post-breeding dispersal (Cramp 1977), they are able to colonize new habitats and persist as small, isolated populations. For example, the Eurasian bittern recolonized Great Britain in the 1940s, where it was extirpated in the 1870s, and now persists in 6 small populations (50 pairs total) located distantly from one another (Bibby 1981, Day 1981). American bitterns readily use artificial wetlands created by impoundments at waterfowl refuges (Andrle and Carroll 1988, Gibbs et al. in press.), a trait that could facilitate restoration of populations in regions where natural, inland freshwater wetlands have been destroyed (Connecticut, Rhode Island, central New York, New Jersey, and Maryland) or were scarce originally (central Pennsylvania and West Virginia). Also, American bitterns seem adaptable to a wide range of wetland habitats, ranging from margins of boreal lakes in Quebec (DesGranges and Houde 1989) to dense cattail marshes in New York (Andrle and Carroll 1988), and can thrive at wetlands of many types as long as suitable prey and adequate cover are available (Gibbs et al. in press). Information is too scarce, however, on the structure and potential growth rates of populations to speculate about the recovery potential of the species.

## LAND PROTECTION AND PRESERVE DESIGN

Wetland area is a consideration for preserve design because both the occurrence (Brown and Dinsmore 1986) and abundance (Gibbs et al. in press., Gibbs and Melvin 1990) of American bitterns is greater on larger than smaller wetlands. We suggest a minimum area of 2.5-5 ha as being sufficient to support nesting activity (see Eaton 1910, Brown and Dinsmore 1986, Gibbs et al. in

press). Smaller wetlands adjacent to large wetlands used for nesting may serve as important, alternate foraging sites for American bitterns, which are seen regularly flying between wetlands during the nesting season in Maine (J. Gibbs pers. obs.). Vegetative features of wetland preserves should include a high diversity of vegetative lifeforms and an abundance of emergent vegetation well-interspersed with patches of open water and aquatic-bed vegetation. Water levels should be < 10 cm deep (Fredrickson and Reid 1986). Retaining dense, woody riparian vegetation may provide a visual barrier that reduces human disturbance of nesting bitterns and also buffers a wetland ecosystem against upland runoff that may contain silt, pesticides, and other contaminants.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

At present there is no program in place to systematically monitor American bittern populations in the northeastern states. American bitterns are encountered too infrequently on Breeding Bird Surveys to assess population trends in most states (Table 1). American bitterns use localized habitats that usually occur away from roads, and the species' seasonal peak of vocal activity (April-May) tends to occur earlier than when BBS routes are run. Broadcast of tape-recorded calls of American bitterns can increase detectability by eliciting responses from these rarely seen birds (Manci and Rusch 1988, Gibbs and Melvin 1989, Kibbe 1989). Broadcast surveys can be coordinated with surveys of other marshbirds (Swift 1987, Manci and Rusch 1988, Gibbs and Melvin 1989), many of which also are of management concern (e.g., pied-billed grebe (*Podilymbus podiceps*), least bittern, and black tern (*Chlidonias niger*)). Surveys initially should be conducted annually (for 2-3 years) at many sites to provide baseline data on population distributions and abundance in a given region. Repeated surveys (e.g., once every 3-5 years) can then assess population trends. Surveys should be conducted during the peak of the breeding season.

Standardization of survey methodology and coordination of state and regional surveys is necessary for results to be comparable between years and among areas, and might best be accomplished by having a single agency or organization responsible for coordinating survey efforts.

### Management Procedures and Programs

Preservation of wetland habitats, particularly large (> 10 ha), shallow wetlands with dense growths of robust emergents, is the most urgent management need. Wetlands used for breeding by American bitterns also need to be protected from chemical contamination, siltation, eutrophication, and other forms of pollution that harm American bitterns or their food supplies. Day and Wilson (1978) emphasized that merely preserving habitats for endangered populations of Eurasian bitterns is not adequate to ensure their longterm viability; management of vegetation and monitoring water quality at protected areas also is required.

Vegetative features of habitats preferred by American bitterns represent a particular stage of wetland succession. Wetland managers therefore need to periodically reverse vegetative succession while maintaining suitable habitats nearby to serve as alternate nesting areas during wetland manipulations. The concentration of both nesting and overwintering populations of American bitterns at protected and managed wetlands, e.g., state and national wildlife refuges (Andrle and Carroll 1988, Root 1988) emphasizes the need to develop and implement habitat management procedures that benefit American bitterns. Minor alterations of existing management schemes could greatly improve nesting habitat for American bitterns. Where littoral vegetation is scarce, moist soil plant management (Fredrickson and Taylor 1982) provides a cost effective method involving water-level manipulation to reestablish and promote growth of dense stands of emergent vegetation preferred by American bitterns.

Complete drawdowns should be avoided so that populations of small fish, amphibians, and dragonfly larvae, which make up a large part of the American bittern's diet, are conserved for the following season. Slow, rather than rapid, drawdowns emulate natural water recession

patterns, concentrate foods, and help prevent oxygen depletion (Fredrickson and Reid 1986). Drawdowns can be used to create favorable water levels (< 10 cm deep, Fredrickson and Reid 1986) for foraging American bitterns. Liming and fertilizing dikes and adjacent fields can increase the productivity and raise the pH of many nutrient-poor, acidic wetlands in the northeastern region (Jorde et al. 1989). Control of infestations of purple loosestrife and phragmites may improve habitats for American bitterns in many northeastern states.

### Research Needs

- Develop standardized methodologies for monitoring populations of American bitterns in the northeastern U.S.
- Conduct surveys to determine the abundance and distribution of American bitterns in the northeastern U.S.
- Monitor trends in populations and habitat availability for the American bittern in the northeastern U.S.
- Conduct a detailed, autecological study of the American bittern that examines basic features of the breeding biology of the species, including diet, home range, mating systems, ability to renest, sources and rates of mortality in adults, juveniles, nestlings, and eggs, and juvenile dispersal patterns and philopatry. This information could be gathered through radio-telemetry and banding studies.
- Evaluate habitat requirements of American bitterns, including the vegetative characteristics, water quality, and minimum area of wetlands used by nesting, migrant, and overwintering American bitterns. Also, evaluate effects of riparian zone management on wetland use by American bitterns.
- Identify migration routes, major stopover sites, and major overwintering areas used by American bitterns.
- Examine the effects of contaminants, parasites, diseases, predation, water pollution, acid rain, human disturbance, and severe weather on American bittern populations.
- Develop wetland management strategies that benefit nesting and migrant American bitterns. Because the distribution of nesting American bitterns is closely tied to protected wetlands at state and national wildlife refuges in many parts of the Northeast, assess the long-term viability of these sites as core-breeding areas for regional populations.
- Monitor contaminant levels in American bitterns and their eggs in industrialized and agricultural regions of the northeastern states.
- Evaluate the effects of marsh invasion by phragmites and purple loosestrife on American bitterns.

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# LEAST BITTERN

## *Ixobrychus exilis*

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Breeding and wintering least bitterns inhabit freshwater and brackish wetlands with dense, tall emergent vegetation such as cattail or bulrush. In the Northeast, least bitterns breed primarily in wetlands along lakes, rivers, and estuaries on the coastal plain, but are generally absent from the Appalachian highlands and mountainous parts of New York and northern New England. Historical records suggest that declines in breeding populations may have occurred in New York, Delaware, Maryland, and Virginia. The species is probably often overlooked and may be more common than reports indicate, but few data are available to assess population trends or serve as a baseline for future surveys. Loss of wetland habitat is likely the greatest threat to least bitterns in the Northeast. Preservation of wetlands > 5 ha with dense, tall (> 1 m) emergent vegetation over relatively deep water (10-50 cm) interspersed with patches of open water is the most important management need. Least bitterns readily use artificial impoundments, and management of these wetlands to optimize least bittern habitat could enhance recovery of populations in areas where losses of natural wetlands have been high. Special surveys for least bitterns need to be developed to better determine population distribution and assess trends and responses to habitat features.

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### DESCRIPTION

### Taxonomy

The least bittern (*Ixobrychus exilis*) is a member of the bittern subfamily, Botaurinae,

which is included in the heron family, Ardeidae, within the large order of wading birds, the Ciconiiformes. In comparison to other herons, bitterns are more solitary and rely more heavily on auditory than visual communication. Bitterns also have a laterally compressed trunk, short legs,

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and short, outer toes and long, curved toenails that enable them to travel through and grasp the dense, emergent vegetation typical of nesting habitats (Kushlan 1978, Hancock and Kushlan 1984). The Botaurinae consists of four species of large bitterns (*Botaurus* spp.) and eight species of small bitterns (*Ixobrychus* spp.). Small size, yellow color, and a dark crown are characteristics that distinguish least bitterns from all other bitterns and herons (Hancock and Kushlan 1984). The eight congeneric species of *Ixobrychus* found worldwide are morphologically similar, and the least bittern (*I. exilis*) of the Americas, little bittern (*I. minutus*) of western Eurasia and Africa, and the yellow bittern (*I. sinensis*) of the Far East probably represent a single superspecies (Payne and Risly 1976, Hancock and Kushlan 1984). The least bittern is comprised of five superficially similar subspecies distributed throughout North and South America (Hancock and Kushlan 1984). The following discussion is restricted to the eastern North American race, *I. exilis exilis*.

### Morphology and Plumages

As the smallest member of the heron family, the least bittern ranges between 28-36 cm in length (Cramp 1977). Diagnostic field characteristics include a vivid, greenish-black crown, back, and tail; brownish and white neck, sides, and underparts; and chestnut-colored wings with conspicuous, contrasting, pale-colored wing patches. The head is slightly crested. Whitish, highly visible lines border the scapular feathers. Sexes are similar in size, but plumage is dimorphic. The crown and back of the female is purple-chestnut, whereas those of the male are black, and the neck of the female is darkly streaked (Palmer 1962). Juvenile plumage is similar to that of the adult female, but the crown of the juvenile is paler and more brown, and the breast and throat are browner with heavier streaking. The only seasonal difference in plumage is a high gloss in spring (Weller 1961). A rare, darker morph of the least bittern occurs, known as "Cory's bittern," in which paler areas of typical plumage appear chestnut-colored (Palmer 1962, Hancock and Kushlan 1984). Young least bitterns are covered with long, soft, ochre-colored

down above and whiter down beneath (Palmer 1962, McVaugh 1975).

### Vocalizations

The least bittern is quite vocal and has a varied repertoire of calls. The male's advertisement call most frequently heard in spring is a dove-like cooing characterized as "uh-uh-uh-uh-uh-oo-oo-oooo-oo-ooah" (Palmer 1962). Females may respond with "ticking" calls (Hancock and Kushlan 1984). When alarmed, three calls may be uttered: a loud, shrieking "quoh," a hissing "hah," or cackling "tut-tut-tut" (Palmer 1962, Hancock and Kushlan 1984).

### HABITAT

Breeding habitats of least bitterns vary throughout North America, but nesting usually occurs among dense, tall growths of emergent vegetation (particularly cattail (*Typha* spp.), sedge (*Carex* spp.), bulrush (*Scirpus* spp.), or common reed (*Phragmites australis*)), interspersed with some woody vegetation and open, fresh water (Weller 1961, Palmer 1962, Kushlan 1973, Swift 1987, Frederick et al. 1990). Both fresh and brackish marshes are used (Palmer 1962, Swift 1987, Andrle and Carroll 1988). The occurrence of least bitterns has been associated particularly with cattail, vegetated edges along deep, open waters (Weller 1961), and nutrient-rich microhabitats (Kushlan 1973). Least bitterns spend nearly all the diurnal period in dense, grass-like vegetation; open habitats such as mats of emergent vegetation are rarely used (Frederick et al. 1990).

Weller and Spatcher (1965) found more least bittern nests at two Iowa marshes during years when ratios of emergent vegetative cover to open water were equal (the "hemi-marsh" condition). More densely or sparsely vegetated wetlands contained fewer nests, and interspersion of water and cover may thus be an important characteristic of breeding habitats. In Wisconsin, least bitterns were restricted to deep-water and shallow-water cattail habitats, apparently avoiding areas of dry cattail, river bulrush (*Scirpus fluviatilis*), and sedge

(Manci and Rusch 1988). Least bitterns used variably-sized wetlands in Maine with dense, tall stands of cattail, which were often associated with relatively stable water regimes at managed impoundments and coves on lakes (Gibbs and Melvin 1990).

On moist-soil impoundments in Missouri, least bitterns were associated with waters up to 50 cm deep and rank, dense vegetative cover bordering open water (Fredrickson and Reid 1986). Least bitterns were not associated with open, sparse, or short vegetative cover or muddy openings (Fredrickson and Reid 1986). Least bitterns in Iowa occurred most regularly on wetlands > 5 ha, suggesting that wetland area alone may be an important feature of least bittern habitat (Brown and Dinsmore 1986). Gibbs and Melvin (1990), however, observed territorial least bitterns on wetlands as small as 0.4 ha in Maine.

Among six tidal marshes along the Hudson River in New York, Swift (1987) reported that the presence of least bitterns was related to the extent of tall bulrush-cattail cover and site elevation (i.e., depth of tidal flooding). Least bitterns favored sites with tall, dense vegetation and low-lying, "wetter" sites (peak water depths averaged 70 cm on sites used by least bitterns), perhaps because nests are usually placed over water or near open water (Weller 1961, Palmer 1962, Kushlan 1973, Aniskowicz 1981). In tidewater areas of Chesapeake Bay, Stewart and Robbins (1958) reported that the species was especially abundant in narrow-leaved cattail (*Typha angustifolia*) marshes, common in other coarse marsh types and weak-stemmed brackish marsh types, but sparse in salt marshes.

In the Florida Everglades, least bitterns were observed from airboats at densities (birds/km) of 0.13 in canals, 0.04 in open grasslands, and 0.37 along airboat trails (Frederick et al. 1990). Most individuals were seen in mixed sawgrass (*Cladium jamaicensis*)/cattail (29% of birds), homogeneous sawgrass (23%), and homogeneous cattail (9%), and, within these vegetative associations, were seen twice as frequently in dense than sparse stands (Frederick et al. 1990). Open sloughs, rush prairies, mats of emergent vegetation, and burn areas were used infrequently (Frederick et al. 1990, but see Kushlan 1973). The strong association between cattail and least bitterns in

northern regions (e.g., Weller 1961, Swift 1987, Manci and Rusch 1988, Gibbs and Melvin 1990) may occur because cattail is the most common tall plant growing in dense stands above deep water in most northern areas (Frederick et al. 1990).

Sites where least bitterns were flushed during spring migration in Missouri ( $n = 61$ ) had a mean water depth of 23 cm, vegetation height of 64 cm, and stem density of 287 stems/m<sup>2</sup> (Reid 1989), and were dominated by burreed (*Sparganium eurycarpum*), water smartweed (*Polygonum coccineum*), and cattail. Reid (1989) observed a shift in habitat use to taller (121 cm), sparser (165 stems/m<sup>2</sup>) stands in fall ( $n = 15$  flush sites). Feeding platforms constructed by least bitterns ( $n = 52$ ) occurred over water averaging 29 cm in depth and among stands of emergent vegetation averaging 199 stems/m<sup>2</sup>.

Overwintering birds occur in brackish and saline swamps and marshes (Palmer 1962, Hancock and Kushlan 1984), but little is known about wintering habitats of least bitterns.

## BIOLOGY

### Feeding

The repertoire of feeding behaviors used by the least bittern is relatively small (Kushlan 1978), and among 28 recognized behaviors used by herons, least bitterns employ only 4: standing in place; walking slowly; neck swaying (to overcome glare, to increase camouflage, or to have muscles in movement when strike begins); and wing-flicking, which involves quick, repeated extension and retraction of wings that may startle prey from hiding (Sutton 1936). The least bittern's small size and highly compressed trunk enable it to move easily through dense vegetation. Least bitterns forage by stalking along branches and reeds, and often cling to clumps of vegetation above water-level, aided by their short, outer toes and long, curved claws. Least bitterns may build foraging platforms of bent reeds at productive feeding sites (Weller 1961, Reid 1989), and frequent these platforms during late-incubation and brood-rearing periods (Reid 1989). Clinging to emergent vegetation and constructing platforms

enables this smallest of North American herons to forage at water depths as great as those of the largest North American herons (25-60 cm depth).

The small, thin bill of least bitterns is probably an adaptation for securing small, fast-moving prey (Kushlan 1978), an observation supported by dietary analyses. Contents of 93 stomachs of least bitterns from Florida included 40% fish (top minnows, mud minnows, yellow perch, and sunfish), 21% dragonflies, 12% other aquatic insects, and 10% crustaceans (mainly crayfish: Howell 1932). Six stomachs from Pennsylvania contained small mammals, beetles, fish, a snake, and vegetable matter (Warren 1890). Nesting least bitterns consumed damselflies, grasshoppers, other insects, and tadpoles in Iowa (Weller 1961). Least bitterns also may prey on the eggs and young of yellow-headed blackbirds (*Xanthocephalus xanthocephalus*) (Roberts 1936, cited in Hancock and Kushlan 1984). Other foods include frogs, lizards, molluscs, and leeches (Bent 1926).

## Reproduction

Least bitterns arrive at nesting areas in the northeastern U.S. in early to mid-April or early May (Bent 1926); arrival may be delayed until late April in more northern states, including Maine (Palmer 1949) and New York (Swift 1987). Although few data are available on territoriality and mating systems of least bitterns, most least bitterns nest solitarily and probably form seasonal, monogamous pair-bonds (Weller 1961, Palmer 1962). Occasionally least bitterns breed colonially near abundant food sources (Audubon 1840; Weller 1961; Wood 1951, cited in Palmer 1962; Kushlan 1973). Descriptions of courtship and territorial behavior are given by Weller (1961).

In six tidal marshes along the Hudson River in New York, Swift (1987) observed an average of up to 0.5 least bitterns/ha during playback surveys using tape-recorded vocalizations in 1986 and 1987. Other reports of nesting densities in North America vary widely: 15 nests/ha ( $n = 15$  nests: Wood 1951, cited in Palmer 1962), 1 nest/ha ( $n = 26$  nests: Beecher 1942, cited in Palmer 1962), and 1 nest/ha ( $n = 19$  nests: Kent 1951). Approximately equal numbers of male and female least bitterns were observed by Frederick et al.

(1990) during the breeding season in the Florida Everglades.

**Nests and Eggs.** - Nests are usually built over shallow water (0.1-1.0 m deep) (Palmer 1962, Kushlan 1973, Aniskowicz 1981) and tend to be near (< 10 m) open water (Weller 1961). Least bitterns make a nesting platform with a canopy by pulling down and crimping surrounding emergent vegetation, such as cattail or bulrush (Weller 1961).

Copulation occurs on the nest before and during incubation (Weller 1961). Female least bitterns lay 1 egg daily to complete clutches of 2-7 eggs, usually 4-5 eggs (Trautman 1940, Kent 1951, Weller 1961, Hansen 1984). Clutch-size seems to increase with latitude in North America (Palmer 1962). Least bitterns can probably rear second broods (Kent 1951, Weller 1961). Peak nesting (including first and second broods) occurs in the northeastern U.S. from late-May to early-July (Palmer 1962, Swift et al. 1988). In the Florida Everglades, most nesting was initiated after 18 May, with large numbers of fledgling least bitterns appearing after 15 June (Frederick et al. 1990).

Early clutches, laid before mid-June, may be smaller than later clutches (Weller 1961). Eggs are elliptical, pale blue or pale green, smooth and nonglossy, averaging 31 by 24 mm (Bent 1926, Harrison 1978). Incubation by both sexes begins upon deposition of the first or second egg, and lasts 17-20 days (Weller 1961, Aniskowicz 1981). The semi-altricial young hatch asynchronously over a 3-day period (Weller 1961, Palmer 1962).

**Nesting Success.** - Success of 38 nests was estimated by Weller (1961) to be 84% (nests in which  $\geq 1$  eggs hatched), with an average of 3.0 young hatched/nest. Kent (1951) observed that eggs in 70% of 20 nests hatched, and listed abandonment, predation, cannibalism, and disease as causes of nest loss. In Kansas, four of five nests studied by Hansen (1984) were depredated, at least one by a raccoon (*Procyon lotor*). High water also may destroy least bittern nests (McVaugh 1975), marsh wrens (*Cistothorus palustris*) may puncture least bittern eggs (Bent 1926), and mink (*Mustela vison*) may take eggs and nestlings (de Vore 1968, cited in Hancock and Kushlan 1984). Young least bitterns, brooded

for several days, leave the nest after 6-15 days but return occasionally to be fed regurgitated food by their parents, mostly by the male (Gabrielson 1914, Weller 1961). Hansen (1984) and Nero (1950) noted that the youngest hatchling among broods grew the slowest because it received less food. Detailed descriptions and illustrations of the plumage development of young least bitterns are provided by McVaugh (1975). Young least bitterns may linger at nesting areas until October before beginning their southward migration (Palmer 1962). Sources of mortality of chicks and adults include predation by raptors, crows (*Corvus* spp.), raccoons, mink, snakes, snapping turtles (*Chelydra serpentina*), and bullfrogs (*Rana catesbeiana*) (Bent 1926; Trautman 1940; Weller 1961; de Vore 1968, cited in Hancock and Kushlan 1984). Age at fledging and age of first breeding are unknown (Palmer 1962).

**Least versus American bitterns.** - Although both least bitterns and American bitterns are diurnal and often breed at the same wetland, competition may be minimal between the two species due to differences in microhabitat used for foraging (Gibbs and Melvin 1990, Reid 1989), prey preferences (Howell 1932, Cottam and Uhler 1945), nest-site requirements (Harrison 1978), and breeding phenology (Bent 1926). Generally speaking, least bitterns prefer more densely vegetated, deeper-water habitats than do American bitterns for foraging and nesting, and take smaller prey. Also, least bitterns arrive at nesting areas about a month later than American bitterns, perhaps because least bitterns overwinter in more southerly regions (see Palmer 1962). Because least bitterns are highly insectivorous, delayed breeding also may be related to the life cycles of aquatic prey. Emergences of aquatic insects in temperate zone wetlands begin in April but peak in June (Orians 1980) when food demands of parenting least bitterns probably are highest.

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

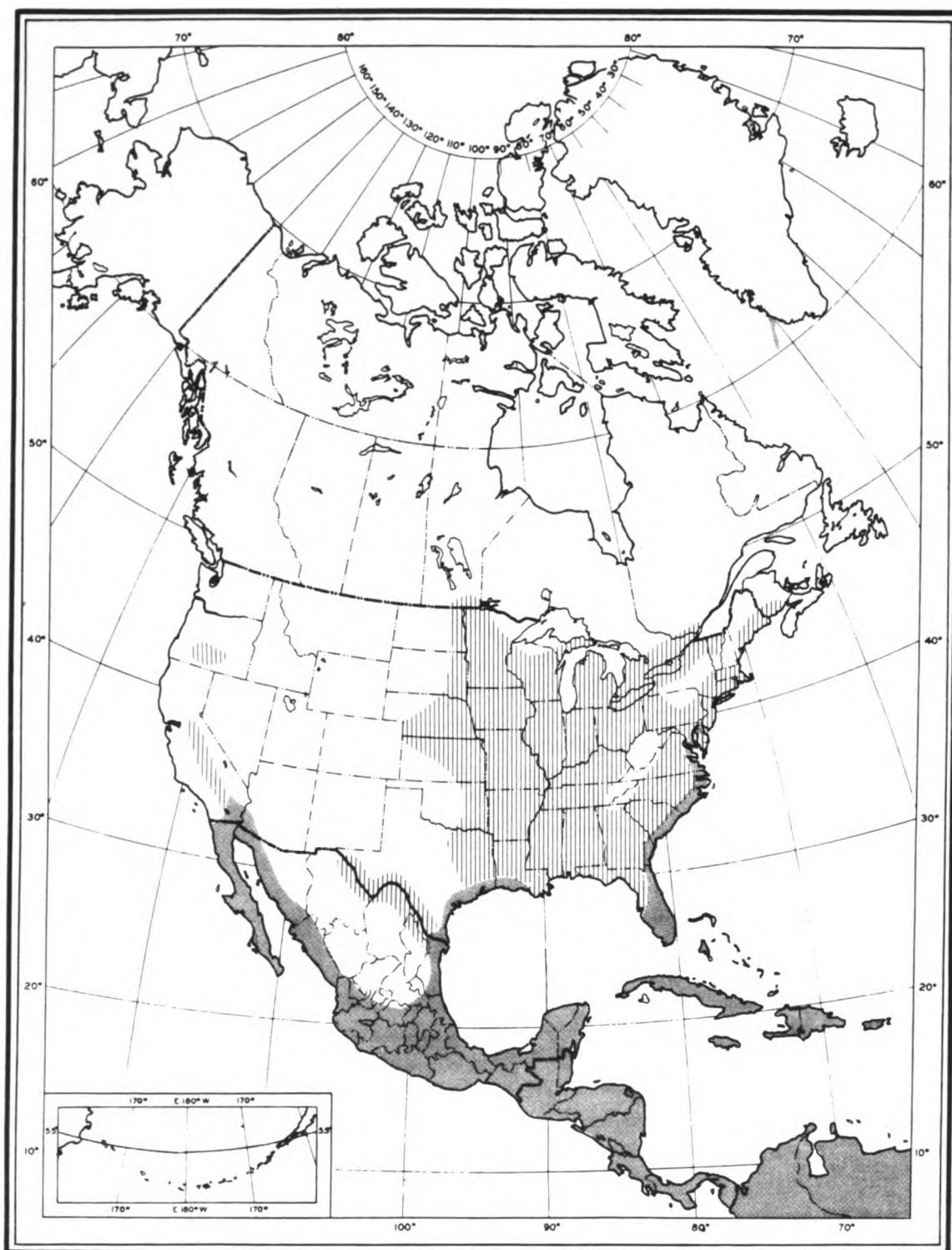
### Distribution and Status in North America

The least bittern's breeding range in North America extends from southeastern Canada through the U.S. and Mexico (Figure 1). Least bitterns breed throughout the coastal plain of the northeastern U.S., but are absent from the Appalachian highlands and mountainous portions of New York, Vermont, New Hampshire, and Maine (Figure 2). The species' distribution seems associated with low-altitude wetlands that occur along lakes, rivers, and estuaries in the northeastern U.S. (Figure 2); the absence of least bitterns from many interior areas is notable, given the abundance of glacial wetlands and wetlands created by beavers (*Castor canadensis*) in many of these areas. The northern and eastern limit of the species' breeding range is reached in New England and the Canadian Maritimes. The dark color morph, "Cory's bittern," is now rarely seen in its former breeding areas in Ontario and southern Florida (Hancock and Kushlan 1984).

The least bittern is likely overlooked in many bird survey efforts, and data on population trends are contradictory and unclear. Data from U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; Robbins et al. 1986) indicate that least bittern populations have remained stable during the period 1966-89 in the eastern region of the U.S. ( $n = 53$  routes). No trends were evident for the U.S. as a whole, although least bitterns were observed on just 62 routes during the period 1966-89. Data are too few, however, to assess populations in any state or province in North America, except Florida (i.e., all had  $< 10$  useable routes). Least bitterns have been on the National Audubon Society *blue list* since 1979 (Tate 1986) because birdwatchers had reported the species as reduced over much of its range and extirpated in some areas.

The Office of Migratory Bird Management included the least bittern on its list of *migratory nongame birds of management concern* in 1982

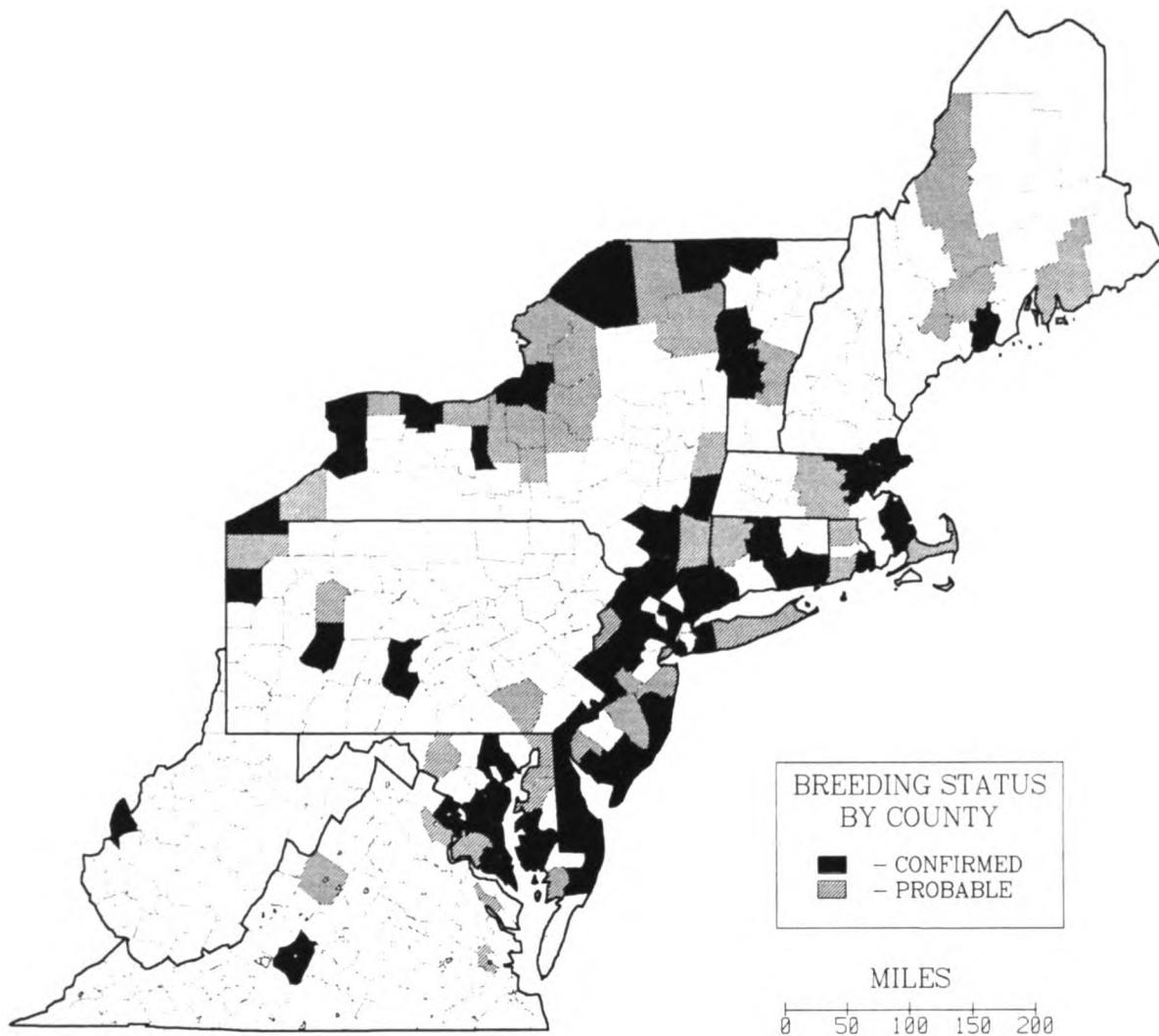
Figure 1. The North American range of the least bittern (*Ixobrychus exilis*) (American Ornithologists' Union 1983, Hancock and Kushlan 1984).



Breeding

Year-round

**Figure 2.** Northeast breeding distribution of the least bittern (*Ixobrychus exilis*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliff unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



(Regions 1, 3, 4, 5, and 6) and 1987 (Regions 3, 4, 5, and 6) because the species was of restricted distribution and dependent on rapidly disappearing wetland habitats (U.S. Fish and Wildlife Service 1987). (The northeastern states are included in Region 5.) The causes of population declines listed by 7 respondents to the survey used to compile this list were habitat loss (100% of respondents) and human disturbance (14%). The least bittern has a Natural Heritage Program global ranking of G5, i.e., demonstrably secure globally, although it may be rare in parts of its range (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

Overwintering populations occur south of areas with prolonged winter frosts (Hancock and Kushlan 1984), and are distributed along the Atlantic coastal plain from Maryland and Virginia south to Louisiana and Texas, with peak numbers occurring in southern Florida (Bull 1974, Root 1988). Audubon (1840) reported observing large flocks (> 20 birds) of least bitterns along the Gulf Coast during winter. Many birds also overwinter in the Greater Antilles (Palmer 1962). The northern limit of overwintering least bitterns occurs considerably farther south than that of the hardier American bittern (Root 1988).

### Distribution and Status in the Northeast Region

**Maine.** - Forbush (1925) considered the least bittern a rare summer resident of southern Maine. Palmer (1949) also emphasized that nesting was restricted to coastal marshes in southern Maine, and cited summer reports from most coastal counties and one inland county. Although Palmer (1949) seemed skeptical that the species still bred in Maine, surveys for the Maine Breeding Bird Atlas (BBA) (Adamus 1988) found evidence of breeding activity at 5 minimum coverage blocks (12% of the total, Figure 2), and a recent survey of freshwater wetlands in southern, central, and eastern Maine (Gibbs and Melvin 1990) detected territorial males at 15 sites. The range of the least bittern in Maine extends along the entire coast and about 50 km inland (Adamus 1988, Gibbs and Melvin 1989). Least bitterns, however, were not encountered on any BBS routes in

Maine during the period 1966-89. Although nesting populations are sparsely distributed in the state, a population decline or range contraction in recent years is not evident, and nesting areas remain largely overlooked in the state. The species has a Natural Heritage Program ranking of S2 in Maine (imperiled because of its rarity), and is considered of *indeterminate status* by the Maine Department of Inland Fisheries and Wildlife because of limited data on distribution and abundance (Table 1).

**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the least bittern (*Ixobrychus exilis*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	indeterminate	S2
New Hampshire	-	S1
Vermont	special concern <sup>b</sup>	S2
Massachusetts	threatened	S1
Rhode Island	state interest	S2
Connecticut	threatened	S2
New York	special concern	S3
Pennsylvania	threatened	S2
New Jersey	declining	S3
Delaware	-	S3
Maryland	special concern	S2
Virginia	undetermined	S2
West Virginia	-	S1

<sup>a</sup>S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

<sup>b</sup>Proposed

**New Hampshire.** - Allen (1903) considered the least bittern a rare summer resident in the extreme southern part of New Hampshire, but knew of no breeding records. Forbush (1925) stated that the least bittern "probably breeds" in New Hampshire. Recent statewide BBA surveys (1981-86) observed breeding activity at only two locations in southern New Hampshire (C. Foss

pers. comm.), suggesting that the least bittern has remained rare in the state throughout this century. This paucity of records is notable given the availability of seemingly suitable marshland in the southern part of the state. Least bitterns were not encountered on any BBS routes in New Hampshire during the period 1966-89. The species has a Natural Heritage Program ranking of S1 in New Hampshire (critically imperiled), but has no special listing by the Fish and Game Department (Table 1).

**Vermont.** - The least bittern was considered as rare in Vermont by Forbush (1925). Surveys for the Vermont BBA (Laughlin and Kibbe 1985) detected breeding activity at seven survey blocks (4% of total). The species' distribution was patchy, and most records came from the Champlain lowlands and near Rutland. Other suitable habitats, and perhaps overlooked nesting populations, occur at Lake Memphremagog on the Canadian border and in marshes along the Connecticut River (Laughlin and Kibbe 1985). More recent surveys of marshlands indicated that the majority of Vermont's nesting least bitterns reside in the Lake Champlain lowlands (Kibbe 1989). It is not clear whether any significant population decline has occurred within the state (Laughlin and Kibbe 1985), but the least bittern is classified as a *species of special concern* by the Vermont Fish and Wildlife Department because of its dependence on restricted, vulnerable wetland habitats (Table 1). Least bitterns were not encountered on any BBS routes in Vermont during the period 1966-89. The species has a Natural Heritage Program ranking of S2 in Vermont (imperiled because of rarity, Table 1).

**Massachusetts.** - Howe and Allen (1901) considered the least bittern in Massachusetts a "not uncommon," very local summer resident, mainly near the coast. Forbush (1925) reported the least bittern as an uncommon or rare and local summer resident in Massachusetts, and later Griscom and Snyder (1955) considered the species only as rare and local, but potentially "badly overlooked." Least bitterns occurred mainly near the coast or in river valleys in the early 1900s (Forbush 1925), but probably did not breed on Cape Cod and the outer islands (Griscom and

Folger 1948, Griscom and Snyder 1955). In the Concord area, Griscom (1949) considered the least bittern a local, easily overlooked breeder, and noted that it was absent during dry years when grasses rather than cattail and rushes dominated nesting areas.

Unpublished records from statewide BBA surveys from 1974-79 showed nesting activity at 10 locations in the state, mostly in coastal regions (J. Baird pers. comm., Figure 2). Historical trends in populations are unclear, although a gradual decline in least bitterns within the state is plausible. Bagg and Eliot, Jr. (1937) report numerous nesting locations from the Connecticut Valley region that, if resurveyed, could provide a more definitive, historical perspective on population trends in Massachusetts. Least bitterns were not encountered on any BBS routes in Massachusetts during the period 1966-89. The species has a Natural Heritage Program ranking of S1 in Massachusetts (critically imperiled), and is considered a *threatened* species in the state (Table 1).

**Rhode Island.** - Howe and Sturtevant (1899) considered the least bittern to be a common summer resident in Rhode Island, breeding "...in all the reedy marshes of the state." Forbush (1925) also considered the least bittern a "rather common" summer resident in Rhode Island. Recent, statewide BBA surveys (1982-87), however, recorded breeding activity at only seven locations in coastal regions, suggesting that the species is a regular but rare summer resident (R. Enser pers. comm.). Least bitterns were not encountered on any BBS routes in Rhode Island during the period 1966-89. If the population assessments of Howe and Sturtevant (1899) and Forbush (1925) were valid, least bitterns evidently have undergone a substantive decline in Rhode Island. The species has a Natural Heritage Program ranking of S2 in Rhode Island (imperiled because of its rarity), and is considered a *species of state interest* by the Rhode Island Division of Fish and Wildlife (Table 1).

**Connecticut.** - The least bittern was at one time a "pretty regular" summer resident (Merriam 1877) and abundant throughout the state in 1875. Sage et al. (1913), however, commented that the

species was a "rather rare" resident of marshes along the coast, occasionally occurring inland, and was formerly "common." The species was considered "not very common" in the state by Forbush (1925). MacKenzie (1961) considered the least bittern a rare breeder in the Guilford area. This series of descriptions suggests that least bittern populations declined in Connecticut during the turn of this century.

The least bittern continues to be absent from most parts of the state (Zeranski and Baptist 1990). Recent surveys for the Connecticut BBA (1982-86) recorded breeding activity at 19 locations scattered both along the coast and inland (Bevier in press, Figure 2). Least bitterns were encountered on only two BBS routes in Connecticut during the period 1966-89, too few to assess population trends in the state. The main cause of population declines appears to be drainage and filling of wetlands (Zeranski and Baptist 1990). The species has a Natural Heritage Program ranking of S2 in Connecticut (imperiled because of rarity), and is listed as a *threatened* species by the Connecticut Department of Environmental Protection (Table 1).

**New York.** - The least bittern is an uncommon to rare breeder in New York. Surveys for the New York BBA recorded least bitterns on 142 blocks (3% of total surveyed) (Andrle and Carroll 1988). Most records came from the Great Lakes plain, and along the valleys of the St. Lawrence River, Lake Champlain, and the Hudson River (Figure 2). Scattered records occurred elsewhere in the state except on the Appalachian Plateau and in the Catskill Mountains. Andrle and Carroll (1988) suggest that populations have declined because of loss of wetlands in the state, although the historical range of the species has remained intact. Least bitterns were not encountered on any BBS routes in New York during the period 1966-89, and perhaps were overlooked frequently during the BBA surveys (Andrle and Carroll 1988).

Least bitterns evidently are locally common, however, in some areas of the state. Swift (1987) found least bitterns to be 1 of the 10 most common nesting species at Hudson River tidal marshes, accounting for 2-5% of all observations of breeding birds in most areas. In western New

York, Beardslee and Mitchell (1965) reported that the status of the least bittern had gone unchanged over the last 60 years. Although data are sparse for assessing population trends of least bitterns in New York, Bull (1974) lists a number of historical breeding localities (known from the 1940-60s) that, if resurveyed, could provide insight into population trends. The species has a Natural Heritage Program ranking of S3 in New York (rare or uncommon), and is considered a *species of special concern* by the New York Department of Environmental Conservation (Table 1).

**Pennsylvania.** - Warren (1890) and Sutton (1928) considered the least bittern to be a rare summer resident, occurring mainly in Crawford, Erie, and Northampton counties. Todd (1940) listed breeding records also for Mercer and Butler counties. Poole (1964) considered the least bittern a very rare and local breeder in Pennsylvania whose range was restricted to the tidewater Delaware Valley and glacial plain near Lake Erie. A population decline may be suggested by a shift in the species' statewide characterization from rare (Warren 1890) to very rare (Poole 1964).

Surveys for the Pennsylvania BBA found evidence of breeding at 29 locations scattered throughout the state (Brauning in press). The current distribution of nesting populations in Pennsylvania (Gill 1985) is similar to the historical distribution, i.e., restricted to the extreme southeastern and northwestern parts of the state, with a few, small populations occurring irregularly elsewhere (Warren 1890, Sutton 1928, Poole 1964). Least bitterns were not encountered on any BBS routes in Pennsylvania during the period 1966-89. The species is now listed as *threatened* in the state because further loss of marshlands may lead to its extirpation (Gill 1985, Table 1). Least bitterns have a Natural Heritage Program ranking of S2 in Pennsylvania (imperiled because of rarity, Table 1).

**New Jersey.** - Least bitterns nest in some of New Jersey's large freshwater and brackish marshes (e.g., Manahawkin, Trenton, and Kearny marshes), but apparently have declined and are conspicuously absent from many former breeding localities (Leck 1984). Stone (1908) believed the

least bittern was a locally common breeder in New Jersey, but Leck (1984) considered it local and uncommon. At Cape May, drainage of wetlands apparently led to declines in local nesting populations (Stone 1937). The species may still be locally common at some locations, given that 50 males were heard calling in 1973 in the Great Swamp (Leck 1984). BBA survey data indicate the least bittern is currently distributed throughout much of the state (D. Hughes pers. comm., Figure 2). Least bitterns were not encountered on any BBS routes in New Jersey during the period 1966-89. The least bittern is considered a *declining* species in New Jersey, and has a Natural Heritage Program ranking of S3 in the state (rare or uncommon, Table 1).

**Delaware.** - In Delaware, the least bittern is currently considered a common breeder in estuarine impoundments and brackish portions of tidewater marshes from New Castle to Cape Henlopen (Figure 2). Recent surveys for the Delaware BBA (Hess et al. in press) recorded breeding activity at 19 locations scattered throughout the state. Construction of impoundments and a weed control program that replaces phragmites with cattail and wild rice (*Zizania aquatica*) has evidently benefitted nesting populations. Least bitterns were not encountered on any BBS routes in Delaware during the period 1966-89. The species has a Natural Heritage Program ranking of S3 (rare or uncommon) in Delaware (Table 1).

**Maryland and the District of Columbia.** - Stewart and Robbins (1958) considered the least bittern to be a common breeding species in fresh and brackish marshes of the Eastern Shore, Western Shore, and Upper Chesapeake Bay, but largely absent from interior portions of the state. Recent surveys for the first BBA of Maryland and the District of Columbia (G. Therres pers. comm.) found evidence of breeding in 83 survey blocks, mostly in counties bordering Chesapeake Bay. The species has a Natural Heritage Program ranking of S2 in Maryland (imperiled because of rarity), and is proposed as a species in need of conservation by the Maryland Forest, Park and Wildlife Service (Table 1). Least bitterns were encountered on only four BBS routes during the

period 1966-89, too few to assess population trends in the state.

**Virginia.** - The least bittern is considered to be an uncommon, local breeder at freshwater and brackish marshes in the state (Virginia Society of Ornithology 1979), but rare and very local farther inland according to BBA data, 1984-89 (S. Ridd and R. Wadja pers. comm., Figure 2). Continued loss of wetlands threatens nesting populations, particularly inland where nontidal wetlands receive little regulatory protection. The least bittern is common in spring and fall along the coastal plain (Virginia Society of Ornithology 1979), where brackish marshes represent an important stop-over site for migrants and overwintering birds. The species has a Natural Heritage Program ranking of S2 in Virginia (imperiled because of rarity), and no special status in the state (Table 1). Least bitterns were not encountered on any BBS routes in Virginia during the period 1966-89.

**West Virginia.** - The least bittern is an uncommon, very local summer resident in West Virginia, and breeds regularly in only three to four locations in the state (Hall 1983). The most frequently used areas are at Boaz Marsh (Wood County) and at McClintic Wildlife Station (Mason County). Scattered summer records of least bitterns suggest that the species is more widespread than present reports indicate (Hall 1983), but low availability of large, palustrine wetlands in the state clearly limits nesting populations. Recent, statewide BBA surveys found evidence of breeding at only five locations in the state (C. Stihler pers. comm.). Least bitterns were not encountered on any BBS routes in West Virginia during the period 1966-89. The species has a Natural Heritage Program ranking of S1 in West Virginia (critically imperiled), but no special listing with the West Virginia Department of Natural Resources (Table 1).

## Summary

Existing breeding bird survey programs are inadequate to assess trends in least bittern populations. For example, in the entire eastern region of the U.S., least bitterns were encountered

on only 53 BBS routes during the period 1966-89, and on only 6 routes in the northeastern states. The species has probably always been overlooked and is more common than reports indicate, but few quantitative data are available to assess past population trends or to serve as a baseline for future surveys. Thus, there is a critical need to develop special survey programs and methodologies that focus directly on the least bittern.

Historical records suggest that breeding population declines may have occurred in Massachusetts, Rhode Island, Connecticut, Pennsylvania, and New Jersey, whereas historical information is too sparse for Maine, New Hampshire, or Vermont to speculate about population trends. Availability of marshland habitat in West Virginia may severely limit least bittern nesting populations, but New York, Delaware, Maryland, and Virginia seem to have sizable, relatively secure nesting populations.

## LIMITING FACTORS AND THREATS

Palmer (1962) stated that "...unquestionably, parts of [the least bittern's] range have been affected adversely in recent years by marsh drainage, pollution, spraying of insecticides, and other activities of man." Loss of wetland habitat is likely the greatest threat to least bitterns in the northeastern region. Palustrine emergent wetlands, including inland, freshwater and brackish wetlands frequented by least bitterns, are among the most threatened wildlife habitats in the U.S., and over 4.75 million acres (1.92 million ha) were lost between the mid-1950s and mid-1970s (Tiner 1984). Wetland losses in the northeastern states are primarily caused by draining, dredging, filling, pollution, acid rain, agricultural practices, siltation, and urbanization (Jorde et al. 1989).

Pollution and environmental contaminants may impair the reproductive capacity of least bitterns and predispose them to disease in industrialized and agricultural portions of their range. Relatively high concentrations of dieldrin have been detected in least bittern eggs in Louisiana (Causey and Graves 1969). Organochlorines, heavy metals, and PCBs have been found in many other species of herons, and

some contaminants (DDE, dieldrin) have persisted in tissues of herons long after their use was banned in the early 1970s (Fleming et al. 1983). Although acid rain could potentially reduce food supplies, least bitterns usually occupy wetlands of circumneutral pH with dense growths of emergent vegetation that may provide chemical buffering against acidification. Siltation resulting from erosion of farmlands and run-off containing insecticides may degrade nesting habitats and reduce food supplies in agricultural areas. Threats to birds nesting in Atlantic coast tidal marshes include high mercury levels and DDT residues in marsh soils (Meanley 1985). Marshland invasion by purple loosestrife (*Lythrum salicaria*) and phragmites may alter and degrade least bittern habitats.

Little is known about the effects of disease and parasites on the reproduction of least bitterns. Friend (1987) reported that wading birds were susceptible to the following diseases: oil toxicosis (moderate), type c botulism (occasional), avian cholera (infrequent), chlamydial infections (frequent), sarcocystis (rare), and aspergillosis (infrequent). The least bittern is known to host trematode worms (Font et al. 1984), two species of lice, and one species of mite (Peters 1936). Least bittern populations were thought to be reduced by an unknown disease during the nesting season at an Iowa wetland (Kent 1951). A nematode parasite, *Eustrongylides*, contracted from small fish, can devastate wading bird populations and thrives in waters polluted with nutrients and silt; least bittern populations at wetlands and impoundments receiving stormwater and run-off from residential and agricultural areas may be particularly vulnerable to epizootics of Eustrongylidosis (P. Frederick pers. comm.).

The least bittern seems relatively resistant to human disturbance, and may persist in highly urbanized areas if wetlands remain relatively undisturbed (Bent 1926, Palmer 1962). Because least bitterns fly low to the ground, collisions with motor vehicles, barbed wire fences, and transmission lines can be a mortality factor (Forbush 1925, Guillory 1973). Collisions with airboats also kill nesting and overwintering least bitterns in the Florida Everglades; 3% of 607 least bitterns flushed by airboats were struck (Frederick et al. 1990).

## MANAGEMENT POTENTIAL

Least bitterns readily use wetlands created by artificial impoundments (Andrie and Carroll 1988, Gibbs and Melvin 1990), and seem adaptable to a wide range of wetland habitats, e.g., brackish, coastal marshes, artificial impoundments, and natural freshwater, palustrine, lacustrine, and riverine emergent wetlands. These traits could facilitate restoration or expansion of populations in regions where marshland losses have been high (Connecticut, Rhode Island, central New York, New Jersey, and Maryland) or where marshlands were scarce originally (central Pennsylvania and West Virginia). Information on reproduction and population structure of least bitterns is too scarce, however, to state anything definitive about the potential for management to stabilize or increase populations.

## LAND PROTECTION AND PRESERVE DESIGN

Wetland area is a consideration for preserve design because frequency of occurrence of least bitterns is greater on larger than smaller wetlands (Brown and Dinsmore 1986). A minimum area of  $\leq 5$  ha may be sufficient to support nesting activity (Brown and Dinsmore 1986, Gibbs and Melvin 1990). Vegetative features of wetland preserves should include dense ( $> 100$  stems/m<sup>2</sup>) and tall ( $> 1$  m) stands of emergent vegetation (e.g., cattail or bulrush) in deep-water (10-50 cm) well-interspersed with patches of open water (Weller and Spatcher 1965, Fredrickson and Reid 1986, Reid 1989). Flooded vegetation is likely required for nesting (Weller 1961). Stable water levels will prevent nest flooding, may reduce predation, and can promote dense growth of emergents. If a dynamic wetland system is stabilized over a long period, however, productivity of a site can decline. Retaining riparian vegetation can buffer the wetland ecosystem against upland runoff that may contain silt and contaminants.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Least bitterns were detected too infrequently on BBS routes during the period 1966-89 to assess population trends in any state or province in North America except Florida. Least bitterns breed in localized habitats that usually occur away from roadsides, and the species' seasonal peak of vocal activity (May) usually occurs when BBS routes are not run. Extremely low sighting frequencies of least bitterns on National Audubon Christmas Bird Counts prompted Butcher (1989) to state that a special monitoring program for this species should be initiated immediately.

Broadcast of tape-recorded calls of least bitterns is useful for eliciting responses from these rarely seen birds and surveying their populations (Manci and Rusch 1988, Swift et al. 1988, Gibbs and Melvin 1990, Kibbe 1989). Such surveys can be coordinated with surveys of other marshbirds (Swift 1987, Manci and Rusch 1988, Gibbs and Melvin 1990), many of which are also of management concern (e.g., pied-billed grebe, American bittern (*Botaurus lentiginosus*), and black tern (*Chlidonias niger*)). If so designed, surveys can be used to evaluate population responses to habitat features in addition to assessing population trends. Surveys should initially be conducted annually (for 2-3 years) at many sites to provide baseline data on population distributions and abundance in a given region. Repeated surveys (e.g., once every 3-5 years) can then be used to determine population trends. Surveys should be conducted during the peak of nesting activity, which may be later than for other marsh birds, e.g., in early June (Swift et al. 1988, Frederick et al. 1990). Local populations of least bitterns should be exposed to  $\leq 3$  surveys/year to avoid undue harassment of nesting birds. Standardization of survey methodology and coordination of state and regional surveys is necessary for results to be comparable among years and areas. Coordination of surveys might best be accomplished by having a single regional or national agency responsible for administering survey efforts.

## Management Procedures and Programs

Preservation, protection, and improvement of wetland habitats for least bitterns, particularly large (> 5 ha), shallow wetlands with dense growths of robust, emergent vegetation, is the most urgent management need. Wetlands used for breeding also need to be protected from chemical contamination, siltation, eutrophication, and other forms of pollution that directly harm least bitterns or their food supplies. Equal ratios of cover to open water are preferred by least bitterns, so wetland managers may need to periodically reverse vegetative succession while maintaining suitable habitats nearby to serve as alternate nesting areas during wetland manipulations (e.g., at other wetlands in a complex).

Because least bitterns occur in many states at wetlands managed by state and federal agencies for waterfowl, there is ample opportunity for making minor alterations to existing management schemes to improve nesting habitat for least bitterns. For example, dense stands of cattail and bulrush, often eliminated with cutting, burning, or flooding treatments to improve waterfowl habitat, can be partially retained as habitat for least bitterns. Maintaining stands of deep-water (10-30 cm) cattail is important because water levels at or below the base of emergent vegetation may reduce nesting activity by least bitterns (Weller 1961) and least bitterns prefer foraging over deep water (10-50 cm).

Where littoral vegetation is scarce, moist-soil plant management (Fredrickson and Taylor 1982) provides a cost-effective method involving water-level manipulation to reestablish and promote growth of dense stands of emergent vegetation. Complete drawdowns, sometimes undertaken for waterfowl management, should be avoided so that populations of small fish and dragonfly larvae, which make up the majority of the least bittern's diet, are conserved for the following season. Liming and fertilizing dikes and adjacent fields can increase the productivity and raise the Ph of many nutrient-poor, acidic wetlands in the northeastern region. Infestations of purple loosestrife, which are detrimental to least bitterns, can be controlled with herbicides, physical removal, and burning.

## Research Needs

- Initiate standardized monitoring programs for least bitterns to determine abundance and distribution and to evaluate trends in populations.
- Conduct studies of wetland vegetation, water levels, water quality, and minimum wetland area associated with the occurrence of least bitterns during nesting, migration, and over-wintering.
- Conduct studies on the breeding biology of least bitterns to examine movements and patterns of habitat use, causes and rates of juvenile and adult mortality, sources of nest failure, ability of least bitterns to renest, juvenile dispersal patterns, mating systems and philopatry, and diet. This information could be obtained partly through radio-telemetry and banding studies. Radio-telemetry studies are contingent upon development of appropriate capture techniques for least bitterns.
- Identify major stop-over sites for overwintering and migrating least bitterns.
- Determine the effects of diseases, parasites, contaminants, and weather on breeding and overwintering least bitterns.
- Monitor contaminant levels in least bitterns (or their eggs) in agricultural and industrialized regions.
- Evaluate the effects of open-water management and mosquito-ditching programs at brackish and salt marshes on least bitterns in the northeastern states.
- Evaluate the effects of marshland invasion by phragmites and purple loosestrife on least bitterns.

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# NORTHERN HARRIER

*Circus cyaneus*

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**Northern harriers breed in a wide range of open habitats across much of the Northeast, including abandoned fields, upland maritime heaths, wet hayfields, saltmarshes, and cattail marshes. Nesting sites are chosen based on the availability of appropriate habitat and the abundance of prey in adjacent areas. Harriers feed their young primarily small mammals and birds. When prey is abundant, harriers tend to nest in greater densities and form polygynous mating associations. From August to November, harriers migrate to their wintering grounds in the saltmarshes of the Atlantic coast where they may form communal roosting flocks. Population declines have been documented in several northeastern states and have been attributed primarily to habitat loss from reforestation, the filling of wetlands, and urban and industrial development in coastal areas. Populations should be monitored across the Northeast by checking previously used or historic nesting sites for evidence of breeding at least every other year. Wintering populations should also be observed by surveying suitable hunting habitats. Both breeding and wintering habitats should be protected and managed to provide a complex of several different undisturbed habitat types for nesting and hunting.**

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## DESCRIPTION

### Taxonomy

The northern harrier (*Circus cyaneus*), a member of the Order Falconiformes, Family Accipitridae, was first described by Linnaeus in 1766 (American Ornithologists' Union 1983). The genus *Circus* has worldwide distribution (Brown and Amadon 1968). The number of species, subspecies or races of *C. cyaneus* has been debated by various authorities (Brown and Amadon 1968, Nieboer 1973, Watson 1977, Bildstein and Collopy 1990). The American

Ornithologists' Union (1983) separates *C. cyaneus* into two groups: the northern harrier (*C. c. hudsonius*), which is distributed primarily in North America, and the hen harrier (*C. c. cyaneus*), which is distributed generally in Eurasia. Despite some morphological differences, the two populations share similar reproductive behaviors (Watson 1977, Simmons et al. 1987).

### Morphology and Plumages

Members of the genus *Circus* are slim, medium-sized hawks with long, broad wings and long legs and tails. In most harrier species, adult

males and females are sexually dimorphic with respect to size and color; in size, females are larger than males. Harriers have a characteristic facial ruff, which gives them an owl-like appearance (Brown and Amadon 1968). In the northern and hen harriers, adults and immatures of both sexes have a distinctive, white rump patch.

The following description refers to the northern harrier. The adult female is dark brown above and buffy below, with some streaking on the underparts. The tail is barred. The mean weight of the adult female is 529.9 g (Hamerstrom 1986), total length varies from 48-61 cm and wingspread ranges from 110-137 cm (Bildstein 1988). The adult male is pale grey above and white below with reddish spots on the underparts. The wingtips are edged with black. Males up to 3-4 years of age have brown markings dorsally (Bildstein 1988). The mean weight of the adult male is 367.4 g (Hamerstrom 1986), total length ranges from 44-51 cm, and wingspread varies from 102-114 cm (Bildstein 1988). The immature harrier appears similar in color to the adult female, but has a cinnamon-colored breast and darker brown back and wings (Bent 1937, Brown and Amadon 1968). This distinctive immature plumage is retained throughout the first winter into the following spring and, in some cases, summer. During spring and early summer it is difficult to discriminate between brown harriers (i.e., immatures from adult females) (Bildstein 1988).

### Vocalizations

The call given by adult and immature harriers when they are alarmed or excited has been described as a "rapid chattering," "ke-ke-ke," or "chek-ek-chek-ek" (Brown and Amadon 1968). The begging call has been described as a "wailing squeal," given by the female to the male, and juveniles to adults when begging for food. This call is also used during courtship by the male and female (Bent 1937, Brown and Amadon 1968, Balfour and MacDonald 1970, Watson 1977). When incubating the female may utter a "quip quip quip" (Brown and Amadon 1968).

## HABITAT

### Breeding Habitat

In the Northeast, harriers breed in a wide range of open habitats and vegetative associations. Nests are placed on the ground, usually in dense cover. Harrier nesting sites have included abandoned fields in dense stands of meadowsweet (*Spiraea latifolia*) or red-osier dogwood (*Cornus stolonifera*) in New Hampshire (Serrentino 1987), upland maritime heaths comprised of northern bayberry (*Myrica pensylvanica*), black huckleberry (*Gaylussacia baccata*) and wild rose (*Rosa spp.*) in Massachusetts (Holt and Melvin 1986), and in wet hayfields dominated by reed canary grass (*Phalaris arundinacea*) in Vermont (Laughlin and Kibbe 1985). Breeding sites in New Jersey saltmarshes on the Atlantic coast have been found in pure stands of common reed (*Phragmites australis*), as well as in salt hay grass (*Spartina patens*) and smooth cordgrass (*Spartina alterniflora*) (Dunne 1984). On Long Island, nests were found in stands of common reed and poison ivy (*Toxicodendron radicans*) (England 1989).

Other habitats frequented by nesting harriers in the Northeast are cattail marshes (Laughlin and Kibbe 1985, Serrentino 1989), bogs (Hall 1983, Laughlin and Kibbe 1985, Andrle and Carroll 1988), native grassland prairies (Genoways and Brenner 1985), and dwarf conifer forest (England 1989). In other regions of North America, harriers nest in a variety of upland and wetland habitats such as willow (*Salix spp.*) swales and meadows (Hamerstrom and Kopeny 1981), pure stands of blackberry (*Rubus spp.*) (Toland 1985), hayfields and cropland (Duebbert and Lokemoen 1977, Follen 1986) and undisturbed grass/legume vegetation (Duebbert and Lokemoen 1977).

### Wintering Habitat

In the Northeast, harriers winter in the greatest numbers in the saltmarshes of the Atlantic coast, with the winter population exhibiting a tendency to increase from north to south (National Audubon Society 1971-74, 1982-83, 1985-87). Although harriers appear to prefer

coastal regions in the Northeast, they will range inland during the winter when suitable open habitats are available (Root 1988), though avoiding the mountainous interior. Other habitats used by harriers during the nonbreeding season in both coastal and inland areas include agricultural fields (croplands, hayfields, and pastures), abandoned fields, and freshwater wetlands. Elsewhere in North America, wintering harriers have been observed in habitats similar to those in the Northeast (Craighead and Craighead 1956, Bildstein 1978, Temeles 1986, Collopy and Bildstein 1987, Littlefield and Thompson 1987).

### Winter Roosts

Harriers may form communal roosting flocks during the nonbreeding season, beginning in October and often breaking up at the onset of spring migration (Bildstein 1979). Harriers roost on the ground in open habitats such as agricultural and abandoned fields, and saltmarshes (Weller et al. 1955, Mumford and Danner 1974, Bildstein 1979, Bosakowski 1983). The same roost may be used for several nights or for several months (Bent 1937, Craighead and Craighead 1956). The number of birds using roosts varies from several to 60 individuals, and roosts may be shared with short-eared owls (*Asio flammeus*). Roost sites may be abandoned during periods of flooding or heavy snow (Bildstein 1979) or when prey becomes depleted in areas adjacent to roosts (Craighead and Craighead 1956, Bildstein 1979).

### Hunting Range Size

The sizes of hunting ranges vary widely during the breeding season in different areas, presumably because of differences in habitat types, availability of prey species, distribution of nest sites, and stage of the breeding cycle (Craighead and Craighead 1956, Balfour and MacDonald 1970, Balfour and Cadbury 1979). In two midwestern studies (Table 1), the range sizes for harrier pairs varied from 2.6-5.5 km<sup>2</sup> (Breckenridge 1935, Craighead and Craighead 1956).

Males usually have larger hunting ranges than females (Hecht 1951, Schipper 1977, Watson

Table 1. Hunting range sizes of breeding northern harriers (*Circus cyaneus hudsonius*) and hen harriers (*C. c. cyaneus*).

Study	Hunting range size (km <sup>2</sup> )
Breckenridge (1935)	2.6 (2 pairs)
Schipper (1977)	
Terschelling	1.8-4.1 (4 males) < 1.0 (4 females)
Flevoland	7.2-12.3 (2 males) 0.8-5.4 (2 females)
Craighead and Craighead (1956)	5.5 (1 pair)
Martin (1987)	1.1 (2 females) 9.7-17.7 (4 males)
Serrentino (1987)	1.4-4.2 (3 females)

1977). Schipper (1977) reported range sizes from 1.8-12.3 km<sup>2</sup> for males in the Netherlands, and Martin (1987) observed range sizes from 9.7-17.7 km<sup>2</sup> for males in Idaho. Hunting range sizes of approximately 0.8-5.4 km<sup>2</sup> for females were observed in both the Netherlands and New Hampshire (Schipper 1977, Serrentino 1987).

Little data are available on the range sizes of nonbreeding harriers. In Michigan, wintering birds flew up to 8 km from roost sites to hunting ranges (Craighead and Craighead 1956). The number of harriers hunting in a particular area decreased with increasing distance from the roost, and hunting range size varied from 0.12-2.6 km<sup>2</sup>. In the Netherlands, harriers appeared to have fixed hunting ranges of unidentified sizes (Schipper et al. 1975).

### Hunting Habitat

Selection of hunting habitat by the northern harrier is affected by several parameters including proximity to the nest site (Schipper 1977, Martin 1987, Serrentino 1987), sex and age of the individual (Schipper et al. 1975, Bildstein 1978,

Marquiss 1980), prey abundance and availability (Schipper et al. 1975), vegetation structure (Schipper et al. 1975, Temeles 1986), and the presence of competitors (Temeles 1986). During the breeding season, females often hunt in areas adjacent to the nest site (Schipper 1977, Martin 1987, Serrentino 1987). Males hunt farther from the nest where they may encounter habitat types different than those located adjacent to nests.

Differences in habitat selection have been observed among adult females, adult males, and juveniles. In Ohio, intersexual differences in habitat selection were related to prey choice (Bildstein 1978). Females were observed significantly more often than males in fallow fields where small mammals were common. Adult males preferred corn stubble where avian prey was predominant. Males took more birds than females (40% vs. 4%), while females were principally small mammal specialists (93% for females versus 56% for males). Unsexed juveniles relied primarily on mammals.

Harriers select habitats on the basis of the availability and abundance of prey species. In the Netherlands, harriers preyed upon common voles (*Microtus arvalis*) in agricultural areas when voles were accessible and populations were high (Schipper et al. 1975). However, when voles became concealed by heavy snowfall, harriers hunted in reedbeds where avian prey was common.

Vegetation structure affects harrier habitat selection and hunting behavior. Harriers often increase flight altitude with increasing vegetation height, enabling them to "see" into the vegetation (Schipper 1977, Serrentino 1987). In a freshwater marsh in Florida, harrier hunting success was negatively affected by the dense marsh grasses that concealed their primary prey, the cotton rat (*Sigmodon hispidus*) (Collopy and Bildstein 1987).

During the nonbreeding season harriers may defend hunting territories (Temeles 1986). In California, females defended hunting territories against other females and males, and aggressively excluded males from preferred hunting habitats such as fallow fields. The substantial size difference between male and female harriers is probably responsible for female dominance of males.

## BIOLOGY

### Feeding Ecology

During the breeding season, harriers feed their young primarily small mammals and birds. In Pennsylvania, a variety of birds, mostly juvenile, were important prey for the young, and included northern flickers (*Colaptes auratus*), eastern meadowlarks (*Sturnella magna*), red-winged blackbirds (*Agelaius phoeniceus*), bobolinks (*Dolichonyx oryzivorus*), American robins (*Turdus migratorius*), and mourning doves (*Zenaida macroura*) (Randall 1940). Other prey taken were several species of mice (*Microtus*, *Peromyscus*, and *Zapus* spp.), frogs (*Rana* spp.), and garter snakes (*Thamnophis sirtalis*). In New Hampshire, microtine rodents and small and medium-sized birds were the most common prey of harriers (Serrentino 1987). When prey could be identified to species, the following were noted; shorttail shrew (*Blarina brevicauda*), meadow vole (*Microtus pennsylvanicus*), meadow jumping mouse (*Zapus hudsonius*), ruffed grouse (*Bonasa umbellus*), northern flicker, American robin, bobolink, and garter snake. In a population breeding on the barrier beaches of Long Island, New York, the meadow vole was the primary prey, with avian species being of secondary importance (England 1989). Waders and passerine birds were the avian groups represented in greatest frequency in prey remains.

Small mammals and birds were also the most important prey taken in other North American studies (Breckenridge 1935, Hecht 1951, Craighead and Craighead 1956, Toland 1985). Harriers in New Brunswick concentrated on meadow voles early in the breeding season, with juvenile birds becoming the most common prey item during the mid and late nestling stages (Barnard et al. 1987).

In Michigan and Ohio, prey taken by harriers in winter consisted primarily of meadow voles (Craighead and Craighead 1956, Bildstein 1978). In Pennsylvania, several rodent species (*Microtus*, *Peromyscus*, and *Zapus* spp.), cottontail rabbits (*Sylvilagus* spp.), and birds were common in the fall and winter diets of harriers (Randall 1940). At a freshwater marsh in Florida, the birds preyed

primarily on cotton rats (Collopy and Bildstein 1987). Godfrey and Fedynich (1987) reported that harriers appeared to take waterfowl opportunistically in Texas.

### Reproductive Biology

**Breeding Chronology.** - Spring arrival dates on breeding grounds in the Northeast range from mid-March to early April, and laying dates range from mid-April to mid and late June (Bent 1937, Hall 1983, Laughlin and Kibbe 1985). Clutches are laid over a period of about 9 days, with eggs being laid at 2-day intervals (Hamerstrom 1969). Clutch size usually varies from 4-6 eggs (Bent 1937, Hamerstrom 1969, Duebbert and Lokemoen 1977). Eggs are pale blue at laying and turn white in a few days; brown markings may occur (Hamerstrom 1969). Incubation is 30-32 days and begins before the last egg is laid; as a result, hatch is asynchronous (Breckenridge 1935, Hamerstrom 1969). The nestling period varies from 30-41 days (Urner 1925, Hamerstrom 1969, England 1989). Juveniles stay near the nest and are dependent on their parents for food for an additional 3-4 weeks (Breckenridge 1935, Hamerstrom 1969).

**Courtship and Breeding Behavior.** - The distinctive courtship flight of harriers has been called a "sky dance." The flight is performed by the male, and occasionally by the female, and consists of a series of nose dives or U-shaped dives (Bent 1937). Hamerstrom (1986) illustrated the behavior as a series of circular flights when viewed from the front. Males have been observed "sky-dancing" during migration (Hamerstrom 1969). Copulation occurs either on the ground or on a short perch (Brown and Amadon 1968, Clark 1972). The female apparently solicits the male with the begging or food call. Harriers do not appear to mate for life (Hamerstrom 1969).

Harriers of both sexes may breed in their first year (Watson 1977); however, more females usually breed as yearlings than males (Hamerstrom et al. 1985, England 1989). In Wisconsin, the majority of females bred at 1 year of age (Hamerstrom et al. 1985). The number of harriers breeding as yearlings (both males and females) in Wisconsin (Hamerstrom et al. 1985) and New Brunswick (Simmons et al. 1986a)

increased during periods of high meadow vole abundance.

Although the male, female, or both may choose the nest site, the female appears to do the majority of nest building. Both parents bring nest material (Watson 1977, Toland 1985). The nest is built on the ground and is composed of dead grasses, weeds, and small twigs (Urner 1925, Bent 1937, Hecht 1951). Nests are frequently placed in dense vegetation (Duebbert and Lokemoen 1977, Hamerstrom and Kopeny 1981, Toland 1985, Serrentino 1987). Larger and deeper nests are often built in wet or flood-prone areas (Urner 1925, Sealy 1967). Sealy (1967) measured 12 nests in upland and wetland habitats; depths of nests ranged from 5.0-24.0 cm, and diameters varied from 39.0-63.0 cm. Harriers may use the same patch of shrubs, field, or general area for several years (Sealy 1967, Balfour and Cadbury 1979, Serrentino 1987, England 1989).

Incubation and feeding of the young is done by the female only (Hamerstrom 1969). During incubation and the early portion of the nestling stage, the female rarely leaves the nest. At this time she is supplied with food by the male, accomplished by a "food pass" in which the male drops the prey to her in mid-air over or near the nest (Breckenridge 1935, Hecht 1951). When she is absent, he drops the prey into the nest and usually leaves immediately (Breckenridge 1935). When the nestlings are about 2 weeks old, the female leaves the nest to hunt more frequently (Hecht 1951, Schipper 1973).

**Nestling Development.** - Newly-hatched harriers are covered with a layer of white down. Their eyes open a few hours after hatching. Mean weight at hatching is 19.8 g (Scharf and Balfour 1971). For the first 5 days, the nestlings are almost continually brooded by the female (Hecht 1951). Between 2 and 3 weeks of age the young begin to make tunnels in the vegetation adjacent to the nest. These tunnels may be used as escape routes (Balfour and MacDonald 1970). Between the third and fourth weeks, the young lose most of their down and acquire their distinctive juvenal plumage (Watson 1977). The young are usually able to fly at 30 days, and have become fairly proficient flyers at 35 days (Hammond and Henry 1949). The lightest individuals and those with the

most well developed flight feathers fledge first, usually the males (Scharf and Balfour 1971).

Because hatching occurs asynchronously, the nest contains young of varying sizes. The smallest nestlings often do not survive because of competition for food with their larger nest-mates (Breckenridge 1935, Balfour and MacDonald 1970). Female nestlings are larger than males for most of the nestling period (Scharf and Balfour 1971, Picozzi 1980).

**Breeding Success.** - Hatching success for harriers varies greatly both among years and study areas. In Michigan, yearly hatching success varied from 0-78% (Craighead and Craighead 1956) and in Canada from 23-89% (Sealy 1967). Fledgling production of harriers, i.e., the number of young fledged per female (monogamous females only) varied as follows: 1.5-2.3 for all nests, including those that failed (Craighead and Craighead 1956, Picozzi 1978, Balfour and Cadbury 1979, Hamerstrom et al. 1985, England 1989), and 2.7-3.1 for successful nests (Picozzi 1978, Balfour and Cadbury 1979, Hamerstrom et al. 1985, England 1989). For data on fledgling production for polygynous females, see Balfour and Cadbury (1979), Hamerstrom et al. (1985), Simmons et al. (1986a,b), and England (1989).

The frequency of renesting after nest failure is low in the harrier. Renesting has been documented in populations studied in New Brunswick (Simmons 1984), Michigan (Craighead and Craighead 1956), and the Dakotas (Duebbert and Lokemoen 1977). In Wisconsin (Hamerstrom 1969) and New York (England 1989), harriers did not lay replacement clutches. In Wisconsin, the adults left the study area within 24 hours of nest failure (Hamerstrom 1969).

**Polygyny.** - Polygyny has been well documented in both the northern and hen harriers (Breckenridge 1935; Hecht 1951; Clark 1972; Balfour and Cadbury 1979; Hamerstrom et al. 1985; Simmons et al. 1986a,b; England 1989). In mainland Scotland, the frequency of polygynous matings was low (Table 2) (Picozzi 1978). However, in the Orkney Islands in Scotland, polygyny accounted for a majority of the mating associations in some years (Balfour and Cadbury 1975, 1979; Picozzi 1984a). In Wisconsin and

**Table 2. Frequency of polygyny in the northern harrier (*Circus cyaneus hudsonius*) and hen harrier (*C. c. cyaneus*).**

Author(s), location and length of study	Frequency of polygyny (% of all nests)
Picozzi (1978) Scotland 5 years	23% (mean)
Picozzi (1984a) Scotland (Orkney Islands) 6 years	74-91%
Simmons et al. (1986b) New Brunswick 4 years	11-43%
Hamerstrom et al. (1985) Wisconsin 25 years	24% (mean)
England (1989) New York 5 years	50% (mean)

New Brunswick, the occurrence and frequency of polygyny was related to meadow vole abundance. High vole numbers led to increases in (1) the number of harriers nesting, (2) the number of yearlings nesting, and (3) the occurrence of polygyny. Simmons et al. (1986a) concluded that the frequency of polygynous matings increased during high vole years because males were able to provision more females successfully.

In the Orkney Islands and on Long Island, New York, the occurrence of polygyny was related to an unbalanced sex ratio that resulted in a shortage of male breeders (Picozzi 1984a, England 1989). Balfour and Cadbury (1979) noted that polygyny was fairly rare until the 1950s, when the population began to increase from a low point. Between the 1950s and 1960s more females than males were reared in the population (Picozzi 1984a); during recent years, more males than females have fledged.

**Breeding Density and Dispersion.** - The breeding density and dispersion of northern harriers are affected by the abundance of prey species (Hamerstrom 1979), the occurrence of polygyny (Balfour and Cadbury 1979, Simmons et al. 1986b), nest site fidelity (Sealy 1967, Balfour and Cadbury 1979), and habitat quality (Picozzi 1984b, Simmons and Smith 1985). The number of nesting harriers increases during high meadow vole abundance in those populations that prey primarily on voles (Hamerstrom 1969, 1979; Clark 1972; Simmons et al. 1986a). In Wisconsin (Hamerstrom 1979) and New Brunswick (Simmons et al. 1986a), nest densities increased more than twofold when vole abundance rose from low to high (Table 3).

Polygyny tends to increase the degree of nest clumping in breeding populations (Balfour and Cadbury 1979, Simmons et al. 1986b, England 1989). In Orkney, the distance between nests decreased with increasing harem size (Picozzi 1984b). Harriers often occupy the same nest sites or nesting territories, but not the nest itself, for several years (Sealy 1967, Balfour and Cadbury 1979, Serrentino 1987, England 1989). In New Hampshire, a pair of harriers nested in the same field for a minimum of 5 years (Serrentino 1987).

Both Picozzi (1984b) and Simmons and Smith (1985) noted that high densities of breeding harriers in some areas were probably a reflection of habitat quality. At a moorland site in mainland Scotland (Picozzi 1978), nest densities were much lower than densities on the Orkney Islands (Table 3) (Balfour and Cadbury 1979). Differences between the two areas included an abundance of prey in the farmlands and wetlands at Orkney (Picozzi 1984b). Simmons and Smith (1985) postulated that harrier nest densities at a predominantly wet marsh in New Brunswick were higher than those reported in other areas because of increased availability of nest sites and high densities of meadow voles.

### Migration

During migration, harriers move in a broad front at heights up to 770 m (Kerlinger 1989). They have been observed traveling along both coastal and inland ridges at numerous sites in eastern North America (Nagy 1977, Bildstein et

**Table 3. Densities of breeding northern harriers (*Circus cyaneus hudsonius*) and hen harriers (*C. c. cyaneus*) in North America and Europe.**

Author(s), location, size of study area (km <sup>2</sup> ), length of study	Density (no. females per 10 km <sup>2</sup> )
Picozzi (1978), Scotland, 120 km <sup>2</sup> , 5 years	0.8 <sup>a</sup> (4-yr mean)
Balfour and Cadbury (1979), Orkney Islands, Scotland, 80 km <sup>2</sup> , 7 years	6.4 <sup>b</sup> (4-yr mean)
Hamerstrom (1979), Wisconsin, 160 km <sup>2</sup> , 20 years	1.5 <sup>c</sup> (3-yr mean: high vole density) 0.6 (3-yr mean: low vole density)
Hamerstrom et al. (1985), Wisconsin, 160 km <sup>2</sup> , 25 years	0.8 <sup>d</sup> (25-yr mean)
Toland (1985), Missouri, 8.5 km <sup>2</sup> , 1 year	8.2 (1-yr mean)
Simmons et al. (1986a), New Brunswick, 60 km <sup>2</sup> , 5 years	4.0 (5-yr mean) 5.6 <sup>e</sup> (2-yr mean: high vole density) 2.0 (1-yr mean: low vole density)
England (1989), New York, 26 km <sup>2</sup> 5 years	4.8 <sup>f</sup> (4-yr mean)

<sup>a</sup>Derived from Table 1 (Picozzi 1978)

<sup>b</sup>Derived from Table 2 (Balfour and Cadbury 1979)

<sup>c</sup>Derived from Table 1 (DDT years omitted) (Hamerstrom 1979)

<sup>d</sup>Derived from Table 1 (Hamerstrom et al. 1985)

<sup>e</sup>Derived from Table 1 (Simmons et al. 1986a)

<sup>f</sup>Derived from Table 1 (England 1989)

al. 1984, Heintzelman 1986, Dunne and Sutton 1986). Sattler and Bart (1984) stated that harriers may not be as conspicuous as other raptors during migration because of their frequent use of flapping flight versus soaring, and their tendency to migrate individually or in small flocks. Harriers do not appear to avoid crossing water during migration. They have been seen "island hopping" in Maine, and have been observed in Bermuda, the Bahamas, and the West Indies (Bildstein 1988, Kerlinger 1989).

Most of the information on harrier migration comes from the autumn season. In the Northeast, the peak of fall migration occurs during September in Maine (Appell 1986), from the last week in September to the first week in October in Rhode Island (C. Raithel pers. comm.), and from early October to mid-November in Maryland (Stewart and Robbins 1958). In West Virginia, one or two harriers are usually observed migrating over the eastern mountain ridges from September to early October (Hall 1983). At Hawk Mountain in Pennsylvania, the peak period for harrier migration is October and November, although their entire migratory period ranges from August to November (Broun 1939, Spofford 1969, Nagy 1977). During August and September, immature harriers are the most common, followed by both adults and immatures in varying numbers in October. Adult males are predominant later in the season (Nagy 1977). Using data from four raptor banding stations (Cape May Point and Kittatinny Mountain, New Jersey; Hawk Cliff, Ontario; and Duluth, Minnesota), Bildstein et al. (1984) found similar trends in the timing of migration with respect to sex and age as that observed by Nagy (1977) at Hawk Mountain.

During the spring migration in central Wisconsin, adult males preceded both adult females and immatures, and adult females arrived before immatures at potential breeding areas (Hamerstrom 1969). On Long Island, New York, males also arrived at breeding sites before females (England 1989).

### Recoveries

Hammond and Henry (1949) banded 150 nestling harriers in North Dakota and received returns on 12 (8%). The birds had dispersed in a

general southern direction, with recoveries from North Dakota (2), Kansas (1), Texas (6), Louisiana (1), Mexico (1), and British Columbia (1). More than half the birds were recovered within 1 year (7 of 12).

Between 1959 and 1977, 12 returns were received from harriers banded as nestlings on the Buena Vista Marsh in Wisconsin (F. Hamerstrom pers. comm., cited by Beske 1982). The birds had migrated in a general south-southeasterly direction and all were recovered during their first fall and winter. Returns were from the following states: Wisconsin (1), Michigan (2), Illinois (2), Tennessee (1), Mississippi (1), Alabama (1), Georgia (1), Florida (1), South Carolina (1), and North Carolina (1).

### Longevity

The average life span of the northern harrier is approximately 7 years in the wild (Brown and Amadon 1968). The maximum age reported for a hen harrier was 12 years for a female and 13 years for a male (Watson 1977). The longest life span of a banded, free-ranging, northern harrier was 16 years, 5 months (Clapp et al. 1982).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

The northern harrier breeds in North America from Alaska south to California and Texas, east to Hudson Bay and the Gulf of St. Lawrence, south to Virginia, and rarely to Florida (Figure 1) (Bent 1937, Terres 1980). Its winter range extends from southwestern Canada eastward to southern New England, southward to Central America and the Caribbean, and rarely to northern South America (Bent 1937, Terres 1980).

The authors of several historic references on the natural history of North American birds described the northern harrier as an abundant and widely distributed raptor (Baird et al. 1860, Bendire 1892, Coues 1892, Bent 1937). In the 1800s and early 1900s, harriers were common

throughout the continent in saltmarshes, inland wetlands, grasslands, and agricultural areas.

Declines in breeding harriers have been observed in parts of North America in recent years (Arbib 1973, Evans 1982, Robbins et al. 1986, Tate 1986, Serrentino and England 1989). This raptor has been on the *blue list* from 1972-86 (Arbib 1973, Tate 1986) which is comprised of species showing severe and noncyclical population declines. In addition, harriers have been placed on official state endangered and threatened species lists throughout most of the Northcentral and Northeast regions (Hands et al. 1989; Serrentino and England 1989; The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia), and are considered a *migratory nongame bird of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987). Inland populations may be threatened in the Southeast (Bildstein and Collopy 1990). In most regions, the decreases in harrier populations have been attributed primarily to the loss and degradation of open habitats and changes in agricultural practices (Evans 1982, Laughlin and Kibbe 1985, Hamerstrom 1986, Serrentino and England 1989, Bildstein and Collopy 1990).

Breeding Bird Survey (BBS) results from 1966-87 and 1980-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland) can be used to look at several regional trends. First, the regions analyzed are composed of the following: East--all areas east of the Mississippi River; Central--the area between the Mississippi River and the Rocky Mountains; and West--the area west of the Rockies, excluding Mexico and Alaska (Robbins et al. 1986). Second, trends in bird species abundance are noted in the subregions recognized by the U.S. Fish and Wildlife Service (USFWS), including: Region 1, Far West (WA, OR, ID, CA, NV); Region 2, Southwest (AZ, NM, OK, TX); Region 3, Northcentral (MN, IA, MO, WI, IL, MI, IN, OH); Region 4, Southeast (AR, LA, KY, TN, MS, AL, NC, SC, GA, FL); Region 5, Northeast (ME, NH, VT, MA, RI, CT, NY, PA, NJ, DE, MD, VA, WV); Region 6, Northern Plains (MT, ND, WY, SD, UT, CO, NE, KS).

The median percent annual changes for the northern harrier in two periods, 1966-87 and

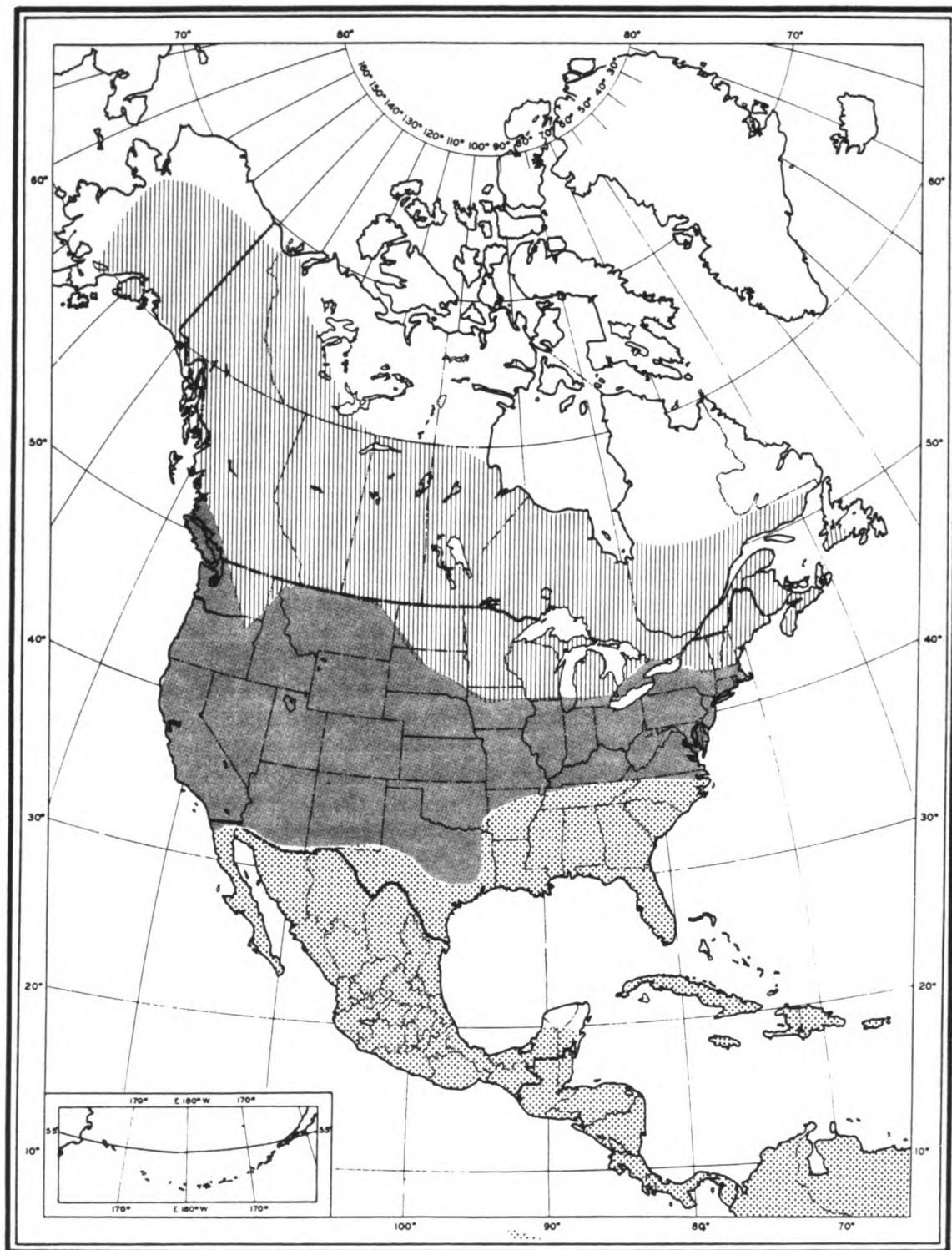
1980-89, are indicated in Table 4. BBS data for the Southeast region were not analyzed because the number of routes on which birds were observed was too low to determine breeding trends (S. Droege pers. comm.). Harriers breed sparsely or not at all in most southeastern states (Bildstein and Collopy 1990). Harrier breeding populations exhibited different trends depending on the period (i.e., 22-year vs. 10-year) and the region analyzed. During the 22-year period, harriers showed a slight, significant increase in the East ( $P < 0.05$ ), whereas a significant decline was observed in the Central region ( $P < 0.01$ ). From 1980-89, harriers showed significant decreasing trends in the Central ( $P < 0.10$ ) and Western ( $P < 0.05$ ) regions.

For USFWS subregions, only the 22-year period was analyzed. Harriers exhibited significant declines in the Southwest (Region 2,  $P < 0.01$ ) and the Northern Plains (Region 6,  $P < 0.05$ ). These data suggest that harrier populations in the East, and more specifically the Northeast, may be stable or increasing slightly. However, breeding harriers appear to be declining in the Central and Western regions, especially in the Southwest and Northern Plains subregions.

BBS data can provide federal and state agencies and private conservation organizations with information on the general population trends occurring in some North American avifauna. Some of the biases involved in the collection of BBS data should be noted: (1) harriers and other raptors may not be as detectable during surveys as other birds because of their wide-ranging habits; (2) survey routes are run in the early morning (Robbins et al. 1986) when harriers may not be as active; and (3) harriers are generally less conspicuous during the incubation stage of the nestling cycle, when the female rarely leaves the nest (Serrentino 1987, 1989). Therefore, harriers may be more difficult to count accurately during surveys conducted at these times. Robbins et al. (1986) point out the problems associated with distinguishing between breeders and nonbreeders.

Currently, wintering populations of harriers may be more stable than breeding populations. Few analyses of the trends of wintering populations have been conducted. Brown (1973) analyzed Christmas Bird Count (CBC) data from 45 states between 1952 and 1971. Harriers

**Figure 1.** The North American range of the northern harrier (*Circus cyaneus hudsonius*) (Bent 1937, Terres 1980).



Breeding

Year-round

Wintering

**Table 4.** Mean number of northern harriers (*Circus cyaneus hudsonius*) per Breeding Bird Survey route and median percent annual change, 1966-87 and 1980-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).

Region <sup>a</sup>	1966-87		1980-89			
	Median % annual change <sup>b</sup>	$\bar{x}$ birds/route <sup>c</sup>	No. routes <sup>d</sup>	Median % annual change	$\bar{x}$ birds/route	No. routes
East	+1.19** <sup>e</sup>	0.14	320	+1.0	0.17	223
Central	-1.85***	0.62	267	-3.2*	0.61	190
West	-0.93	0.58	284	-2.5** <sup>f</sup>	0.70	250
Region 1 <sup>f</sup>	-0.14	0.47	141	-- <sup>g</sup>	--	--
Region 2	-2.73***	0.09	62	--	--	--
Region 3	+0.46	0.15	166	--	--	--
Region 5	+0.04	0.05	70	--	--	--
Region 6	-1.77**	0.86	220	--	--	--

<sup>a</sup>For explanation of different regions, see *Distribution and Status in North America*.

<sup>b</sup>Rate of change in the population or trend, expressed as median percent annual change.

<sup>c</sup>Mean number of birds per route in the analysis region.

<sup>d</sup>Number of routes used in the analysis.

<sup>e</sup>Statistical significance of trend: \*  $P < 0.10$ , \*\*  $P < 0.05$ , \*\*\*  $P < 0.01$ .

<sup>f</sup>BBS data for Region 4 (Southeast) were not analyzed.

<sup>g</sup>BBS data were not available for the period 1980-89 for USFWS subregions.

showed a downward trend from 1952-66 and an increase from 1966-69; however, the increase was primarily attributable to populations in California. In the Northeast, harriers continue to winter in states where breeding populations have either been extirpated (Connecticut) or have exhibited severe declines (Massachusetts and Rhode Island)(Serrentino and England 1989). Bildstein and Collopy (1990) analyzed CBC data for 12 southeastern states (AL, AR, FL, GA, KY, LA, MS, NC, SC, TN, VA, WV) from 1962-64 and 1971-85 and found that harrier wintering populations appeared to have been fairly stable since the early 1960s. Wintering birds were more common in coastal areas than inland.

Analyses of migratory trends of harriers contradict the declines exhibited by breeding harriers in most of the northeastern states (see *Distribution and Status in the Northeast Region*). Dunne and Sutton (1986) assessed migratory data from Cape May Point, New Jersey, from 1976-85 and concluded that harrier numbers had doubled during this period. In two studies conducted at Hawk Mountain, Pennsylvania, harrier numbers

increased slightly from 1934-66 (Spofford 1969), and no definitive trend was observed from 1934-75 by Nagy (1977).

Using Hawk Mountain data from 1934-86, Bednarz et al. (1990) concluded that several raptor species showed population trends coincident with the declines exhibited by these species during the DDT era, 1946-72 (e.g., these species included bald eagles (*Haliaeetus leucocephalus*), sharp-shinned hawks (*Accipiter striatus*), Cooper's hawks (*Accipiter cooperii*), and peregrine falcons (*Falco peregrinus*)). However, harriers did not show any population trend during the DDT period. Harriers exhibited a slightly positive, significant trend overall from 1934-86, and no trend was detected from 1971-86.

Many investigators have warned about the problems associated with migration data, such as the effect of weather on migrants, differences in methods of data collection among observatories, variation in abilities among observers, and observer fatigue (Sattler and Bart 1984, Bednarz and Kerlinger 1989). It is unclear why a discrepancy occurs between migratory assessments

of population trends versus observed declines in harrier breeding populations in the Northeast. The difference may be the result of one or more of the following: (1) harrier declines have not been observed in all regions of North America, and their declines have not been as severe as those observed in some raptors, e.g., bald eagles, ospreys (*Pandion haliaetus*), and peregrine falcons; (2) harriers may not have been as sensitive to DDT poisoning as other raptors because their diet includes a wide range of prey species (e.g., both mammals and birds); and (3) harrier populations in eastern Canada may be masking the declines exhibited by populations breeding in the eastern U.S. (assuming that the birds seen at Hawk Mountain and Cape May could be from eastern Canada). More information is needed to clarify the relationship between migratory trends in harriers and the status of breeding populations in the Northeast.

### Distribution and Status in the Northeast Region

Information on the distribution, abundance, and status of harriers in the Northeast was collected from many sources. Historic information on breeding and wintering birds was gathered primarily from regional and state accounts of bird populations published from the late 1800s to the 1940s (e.g., Howe and Sturtevant 1899, Palmer 1949), and from Christmas Bird Counts (National Audubon Society 1947-50). Christmas Bird Counts from the 1970s and 1980s were used to provide general estimates (e.g., common, uncommon, and rare) of the abundance of current wintering populations in the Northeast. Current information on breeding harriers was gathered from sources such as recent state accounts of bird populations (e.g., Hall 1983, Leck 1984), various published and unpublished reports, papers, and theses, published and unpublished Breeding Bird Atlases (BBA) accounts, state biologists, and BBS data from 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).

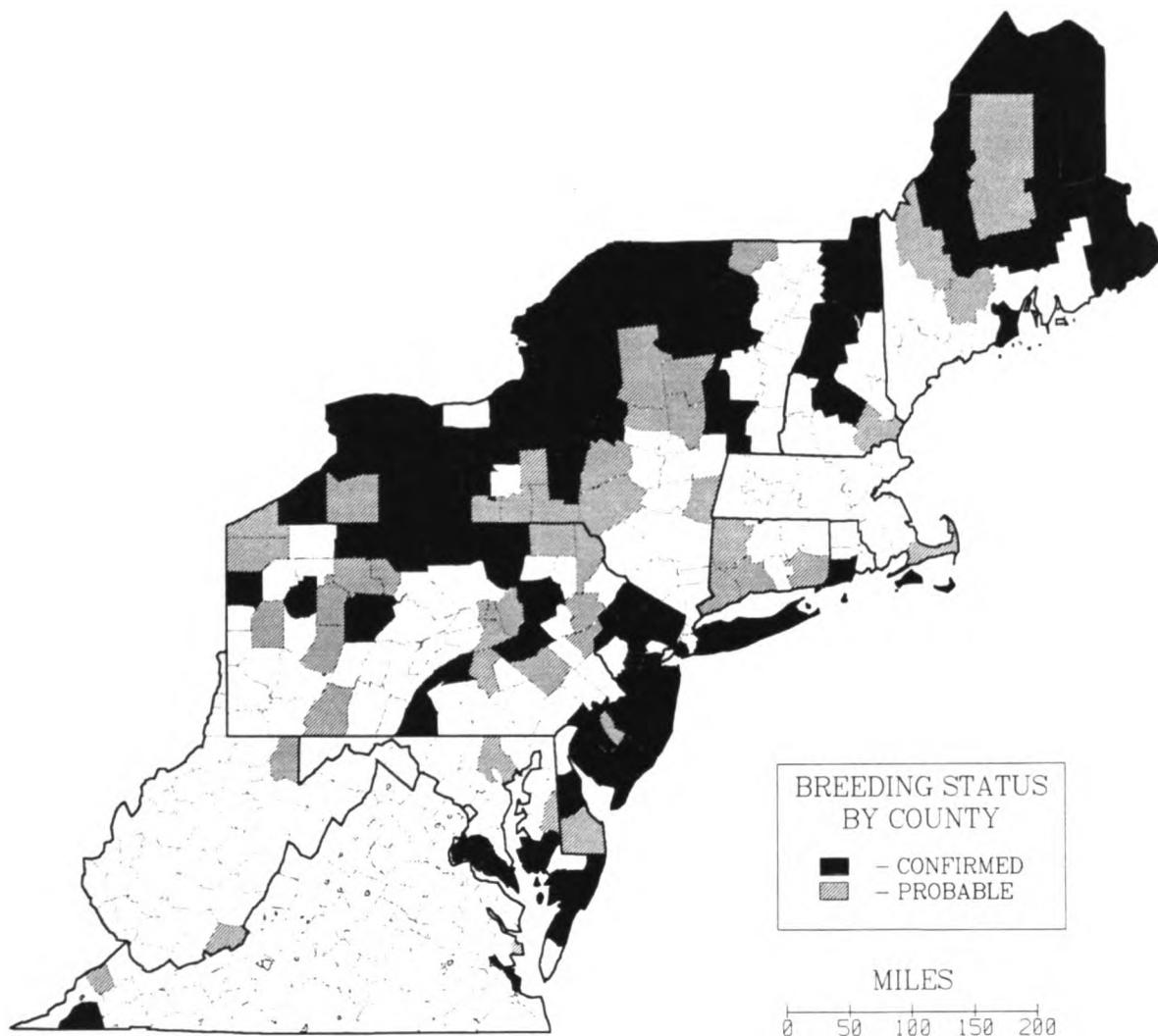
BBA data are used to provide rough estimates of relative abundance and breeding distribution. BBA data for northern harriers by

county for each northeastern state are illustrated in Figure 2. BBS data are used only when sample sizes are adequate (i.e., number of routes are > 10) for analyses of harrier population trends for a state; thus these data are discussed for Maine and New York only.

The state status of the northern harrier in the Northeast is summarized in Table 5. The official state status of harriers was provided by various state agencies in a previous report (Serrentino and England 1989), and was updated with information from The Nature Conservancy's *Central Scientific Databases* (Arlington, Virginia). The status of the harrier as determined by Natural Heritage Program S ranks in each northeastern state is also contained in Table 5. Because the two methods used to determine breeding status are not identical, differences in the status of the bird may occur between state lists and S ranks.

**Maine.** - Presently harriers are extremely rare winter residents in Maine and their historic distribution appears to have been similar (National Audubon Society 1947-48, 1971-73, 1985-86). Most winter sightings are from the southern coast (York and Cumberland counties), where suitable open habitats such as saltmarshes occur. Palmer (1949) noted that harriers were transient and common residents throughout Maine in the fall and spring. Harriers were common summer residents in open habitats throughout most of Maine (Knight 1908, Palmer 1949), but by the 1940s the birds had begun to decline (Palmer 1949). BBS data indicate that the northern harrier showed a nonsignificant, increasing trend from 1966-89. Data collected during the Maine BBA show that the breeding range of this raptor appears to have become more restricted, with most records from the northern and northeastern sections of the state (Adamus 1988, Figure 2). Nesting birds were associated with agricultural lands, recently-cut forest, and various wetland cover types, notably shallow marsh, wooded, and saltmarsh. Pierson and Pierson (1981) describe harriers nesting in the saltmarshes of southern Maine and the blueberry (*Vaccinium* spp.) barrens in eastern sections. The harrier does not have any official state status in Maine other than that afforded to all migratory

**Figure 2.** Northeast breeding distribution of the northern harrier (*Circus cyaneus hudsonius*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



**Table 5. Summary of state listing status and Natural Heritage Program state ranks for the northern harrier (*Circus cyaneus hudsonius*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	S3
New Hampshire	threatened	S2
Vermont	special concern	S1/S2
Massachusetts	threatened	S1
Rhode Island	endangered	S1
Connecticut	endangered	SH
New York	threatened	S4
Pennsylvania	-	S2
New Jersey	endangered <sup>b</sup>	S2
Delaware	-	S1
Maryland	-	S2
Virginia	-	S1/S2
West Virginia	scientific interest	S2

<sup>a</sup>SH = historical records only, but suspected extant

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

<sup>b</sup>Breeding population only

birds (A. Hutchinson pers. comm.), and is given a state Natural Heritage ranking of S3 (Table 5).

**New Hampshire.** - Harriers were previously described as "uncommon, local summer residents" breeding throughout New Hampshire, with the exception of heavily forested areas (Allen 1903). Historic breeding sites were located in Hillsboro, Strafford, Belknap, Merrimack, and Coos counties (Allen 1903, Smith and Choate 1985). Forbush (1927) noted that harriers were casual winter residents in southern New Hampshire. The current winter range of the harrier coincides with its historic range (National Audubon Society 1947-48, 1971-73, 1985-86).

The breeding population in New Hampshire has declined in response to both the reforestation of open habitats that occurred in the early 1900s and the recent destruction and degradation of suitable upland and wetland habitats (Smith and

Choate 1985). Data collected from several sources (Serrentino 1987; C. Foss, E. Hentcy, and S. Sutcliffe pers. comm.:BBA data) indicate that a population of approximately 10 pairs nests in the northern portion of the state, with several pairs reported in central and southern regions. Approximately 20-30 pairs breed statewide (C. Foss pers. comm.). The core breeding population in New Hampshire is associated with agricultural habitats, wetlands, and abandoned fields (Serrentino 1987). This raptor is listed as threatened in New Hampshire and has a Natural Heritage ranking of S2.

**Vermont.** - Historically, harriers were described by Forbush (1927) as casual winter residents in southern Vermont. Data from Laughlin and Kibbe (1985) and recent Christmas Bird Counts (National Audubon Society 1971-73, 1985-86) show that harriers currently winter in low numbers in the Champlain Lowlands.

The harrier has undergone a decline throughout Vermont in the last 150 years that coincides primarily with the reforestation of previously open habitats during the last 50 years coupled with the loss of wetlands (Laughlin and Kibbe 1985). The harrier is presently listed as a species of special concern by the Vermont Fish and Wildlife Department and its Natural Heritage ranking is S1/S2. Currently, harriers breed primarily in regions with the largest amount of agricultural and wetland habitats--the Champlain Lowlands, and North Central and Northeast Highlands (Laughlin and Kibbe 1985). The breeding status of harriers is not well known in Vermont. Laughlin and Kibbe (1985) consider the population stable, but low, using BBA data.

**Massachusetts.** - Howe and Allen (1901) described the harrier as a "...common summer resident and spring and autumn migrant." Prior to 1959, the known breeding records were restricted to the eastern half of the state on the mainland (T. French pers. comm.). Harriers were "rare" in winter on the Massachusetts coast and "very rare" in interior regions (Forbush 1927). Currently, harriers are fairly common winter residents in coastal Massachusetts and the offshore islands, and rare inland (National Audubon Society 1971-73, 1985-86, B. Nikula and D. Holt unpubl. data).

Harrier breeding populations have declined in Massachusetts in response to reforestation and losses in wetland and agricultural habitats (B. Nikula and D. Holt unpubl. data). Breeding birds appear confined to Cape Cod and the offshore islands, where they are associated primarily with maritime heath habitats (J. Baird pers. comm.: BBA data). In recent years there have been four scattered reports of nesting at inland sites and one in Boston Harbor (T. French pers. comm.). Harriers continue to be threatened by development pressures in their remaining strongholds (B. Nikula and D. Holt unpubl. data). The harrier has a Natural Heritage ranking of *S1* and is listed as *threatened* in Massachusetts.

**Rhode Island.** - The harrier was previously considered a "common migrant and summer resident" in Rhode Island, although declines were observed during the late 1800s (Howe and Sturtevant 1899). Prior to the 1960s there were reports of breeding birds from Washington and Newport counties (R. Enser pers. comm.: BBA data). Historically, the harrier was a rare winter resident, with records primarily from coastal areas (Forbush 1927). The current wintering population is considered "moderate," and the highest number of birds occurs in coastal regions and on Block Island (C. Raithel pers. comm., National Audubon Society 1971-73, 1985-86).

In Rhode Island, harriers have suffered from similar patterns of habitat loss and degradation that have been observed in other New England states, primarily from the development of coastal areas, reforestation, and destruction of wetlands (Serrentino and England 1989). The status of the harrier in Rhode Island is *endangered* and its Natural Heritage ranking is *S1*. Currently, the breeding population appears confined to Block Island, where 5-10 pairs nest (Bowen 1982, Serrentino 1989). Breeding sites on the island were located in shrub uplands, shrub wetlands, and cattail (*Typha* spp.) marshes (Serrentino 1989).

**Connecticut.** - Forbush (1927) stated that harriers were rare winter residents in coastal Connecticut. Today they continue to winter in coastal areas with inland records less frequent (National Audubon Society 1971-73, 1985-86;

Zeranski and Baptist 1990). Harriers were considered common summer residents in the 1800s and up to the early 1900s, occurring primarily in saltmarshes and the Connecticut River Valley (Zeranski and Baptist 1990). Previous breeding habitats included fresh and saltwater wetlands and upland meadows (Dowhan and Craig 1976).

The harrier breeding population in Connecticut has declined as a result of losses of open habitats and pesticide poisoning (Dowhan and Craig 1976). No nests were confirmed during the Connecticut BBA, indicating that harriers may have been extirpated from the state as a breeding species (Bevier in press). The last observed harrier nest was in the 1960s, with an attempted nesting during 1989 (Connecticut Geological and Natural History Survey n.d.). The State of Connecticut lists the harrier as *endangered* and its state Natural Heritage ranking is *SH*.

**New York.** - The harrier was historically a widespread and abundant breeder from the Adirondack Mountains to the saltmarshes on Long Island and the lower Hudson River Valley (Eaton 1914). Wintering birds were observed in coastal regions and the Hudson River Valley (Eaton 1914). Currently, harriers are common winter residents in the coastal marshes of Long Island and Staten Island, and the lower Hudson River Valley, with fewer reports from inland areas (National Audubon Society 1971-73, 1985-86). During the New York BBA, major nesting concentrations were associated with agricultural and wetland habitats located in the central, western, and northern portions of upstate New York, and in saltmarshes and shrub habitats on Long Island (Andrle and Carroll 1988).

The harrier breeding population began to decline in New York between the early and mid-1900s (Bull 1974, cited by Andrle and Carroll 1988). This raptor has been negatively affected by changes in agricultural practices and losses of open habitats in coastal and inland areas (Andrle and Carroll 1988, England 1989). Harriers continue to breed in lower numbers throughout their historic range (Andrle and Carroll 1988, Serrentino and England 1989). BBS data indicate that harriers in New York exhibited a nonsignificant decreasing population trend from

1966-89. From 1980-89, a significant positive trend was observed ( $P < .05$ ). The harrier is currently listed as *threatened*, and has a Natural Heritage ranking of S4.

**Pennsylvania.** - Sutton (1928) noted that harriers were common summer residents in Pennsylvania, with most observations restricted to wetland habitats. Harriers commonly nested in several northwestern counties. However, Todd (1940) noted that birds were "...scarcely known outside of the flat country of the northwest counties." Harriers may have bred historically in the wetland habitats associated with the Pocono Plateau and in the fresh and saltwater marshes adjacent to the Delaware River (Serrentino and England 1989). Wintering records from the 1940s showed that harriers occurred in low numbers primarily in southeastern and northwestern sections of the state (National Audubon Society 1947-48).

Presently harriers are rare to fairly common winter residents in historic localities, as well as in the central and southwestern counties (National Audubon Society 1971-73, 1985-86). This raptor does not have any official state status in Pennsylvania and has a Natural Heritage ranking of S2. The species is considered "stable or declining slowly" because of the loss of wetland habitats and monotypic farming practices (Pennsylvania Natural Diversity Inventory: *Vertebrate Characterization Abstract*, Middletown, Pennsylvania). Current breeding locations are concentrated in the native prairie grasslands and barrens in northwestern sections of the state and in wetlands associated with the lower Delaware River (Genoways and Brenner 1985). During the Pennsylvania BBA, breeding records (probable and confirmed) were also received from eastern portions of the state (Brauning in press).

**New Jersey.** - Currently, harriers winter throughout New Jersey, with large concentrations of birds occurring in the southern coastal marshes and Delaware Bay (National Audubon Society 1971-73, 1985-86). Historic wintering localities are similar to the present (National Audubon Society 1947-48). Large winter roosts of more than 100 harriers were observed in Hunterdon

County in the 1950s (Leck 1984), but winter roosts of this size are now rare.

Harriers were fairly common breeding birds in the fresh and saltwater marshes of New Jersey in the late 1800s and early 1900s (Urner 1925, Bent 1937, Stone 1937). Between the 1950s and 1960s, the number of breeding birds in New Jersey began to decrease (Dunne 1984). The destruction of coastal wetlands, pesticide poisoning, human disturbance, and mosquito control techniques have been implicated in their decline (Dunne 1984). Presently, the breeding population of harriers is listed as *endangered* in New Jersey and the species has a Natural Heritage ranking of S2.

Several surveys of breeding harriers were conducted in coastal areas between 1979 and 1986 (Dunne 1984, 1986). The coastal breeding population is considered stable provided that habitat availability remains constant (Serrentino and England 1989). During the New Jersey BBA, harriers were confirmed as breeders throughout most of the state (D. Hughes pers. comm.).

**Delaware.** - Today, as in the past, large numbers of harriers winter in Delaware, primarily in coastal areas (National Audubon Society 1947-48, 1971-73, 1982-83, 1985-86; Hess et al. in press). Historically, harriers probably nested on the Delaware Bay marshes on the eastern shore (Serrentino and England 1989). This raptor has been described as a rare breeder in Delaware, with the current population estimated at < 10 pairs (Hess et al. in press).

During the Delaware BBA, two nests were confirmed and eight were probable, with all sites located on the Delaware Bay marshes (Hess et al. in press). In 1987 a third nest was found at an inland site in a field of barley (Thomas 1987). Currently, the harrier does not have any official state status in Delaware and its Natural Heritage ranking is S1.

**Maryland and the District of Columbia.** - Historically, harriers were fairly common to common winter residents on the eastern and western shores, the Piedmont, and the Upper Chesapeake Bay (National Audubon Society 1947-48, Stewart and Robbins 1958). These same areas currently support high densities of wintering

harriers (National Audubon Society 1971-73, 1982-83, 1985-86).

Harriers have declined in Maryland since the 1950s, with loss of habitat cited as the primary cause (Robbins and Boone 1985). Previous breeding records were from the Allegheny Plateau and the eastern shore (e.g., Dorchester, Wicomico, and Somerset counties) (Stewart and Robbins 1958). During the Maryland and District of Columbia BBA, confirmed breeding records for harriers were located on the eastern and western shores (G. Therres pers. comm.). During a 1990 survey of the coastal marshes in Dorchester, Somerset, and Wicomico counties, biologists found between 10 and 15 pairs of harriers nesting (S. Smith pers. comm.). The status of the inland harrier population appears unknown. In Maryland the northern harrier has a Natural Heritage ranking of S2.

**Virginia.** - Harriers historically occurred in Virginia in low numbers in scattered localities during the winter (National Audubon Society 1947-50). Current CBC records indicate that this raptor remains a scarce to common winter resident, with most birds observed in coastal areas (National Audubon Society 1973-74, 1986-87).

The Virginia BBA classifies the harrier as a "*local to rare breeder*" with current nesting sites located primarily in coastal areas (S. Ridd pers. comm.). Approximately five pairs of birds appear to breed statewide (S. Ridd pers. comm., cited by Bildstein and Collopy 1990). Murray (1952) noted that harriers were never common breeders in Virginia, where they were confined to the coastal plain. Presently, this raptor does not have any official state status and its Natural Heritage ranking is S1/S2.

**West Virginia.** - In West Virginia, breeding harriers are closely associated with wetland habitats (Hall 1983). Historic breeding records from the 1930s-50s were from the mountain bogs of Preston and Tucker counties. Currently, the harrier is described as a "...rare, *local summer resident and rare to uncommon local winter visitant*" (Hall 1983). Harriers appear to nest in wetland habitats on the Allegheny Front and in Jefferson County. However, during the West Virginia BBA there were no confirmed breeding

records for the harrier and only 2 probable and 11 possible records (C. Stihler and A. R. Buckelew, Jr. pers. comm.). Because this raptor has strict habitat requirements and West Virginia is located on the southern edge of its breeding range, the harrier is listed as a *species of scientific interest* (West Virginia Division of Natural Resources 1987). The state Natural Heritage ranking for the harrier is S2.

Both recent and historical data suggest that harriers winter in very low numbers in scattered localities, primarily in the northeastern and northwestern sections of the state (National Audubon Society, 1947-50, 1973-74, 1986-87).

## Summary

Declines in northern harrier breeding populations have been well documented in several northeastern states (Dowhan and Craig 1976, Dunne 1984, Laughlin and Kibbe 1985, Robbins and Boone 1985, Smith and Choate 1985, England 1989, Serrentino and England 1989, B. Nikula and D. Holt unpubl. data). Their downward trend has been attributed primarily to habitat loss from reforestation, the filling of wetlands, and urban and industrial development in coastal areas. In the Northeast, breeding birds may have been extirpated in Connecticut, are restricted primarily to offshore islands in Massachusetts and Rhode Island, or remain as small and scattered populations in Vermont and New Hampshire.

Little information is available on breeding populations in Maine, Pennsylvania, Delaware, and Virginia. In West Virginia harriers were probably never common (Hall 1983, Bildstein and Collopy 1990). New York, Pennsylvania, New Jersey, and Maryland may support the largest breeding populations in the region. Although coastal populations in New Jersey (Dunne 1984, 1986), New York (England 1989), and Maryland (S. Smith pers. comm.) have been studied, the trend of the harrier in these states remains largely unknown.

Wintering populations in the Northeast may not have declined as severely as breeding populations (Serrentino and England 1989). However, little data are available on the status of nonbreeding populations, and further decreases in

habitat quality and quantity on the wintering grounds are likely to have negative effects on nonbreeding populations. Harriers continue to winter in areas where breeding populations have been extirpated locally (Massachusetts and Rhode Island) or statewide (Connecticut). The coastal areas of New York, New Jersey, Delaware, Maryland, and Virginia support the highest number of wintering birds in the Northeast (National Audubon Society 1971-74, 1982-83, 1985-87).

## LIMITING FACTORS AND THREATS

### Habitat

The most significant threat to northern harriers in the Northeast is the continuing loss of open habitats. Habitat loss has occurred in primarily two phases (Serrentino and England 1989). In the Northeast, the reforestation or development of previously open lands (largely farmland) took place from approximately the late 1800s to the mid-1900s (Laughlin and Kibbe 1985, Serrentino 1987, Brooks 1989). Secondly, between 1954 and 1978 the rate of loss of coastal wetlands doubled from urban and industrial development (Tiner 1984). In the Northeast, residential development has been most dramatic in New York and New Jersey. Tiner (1984) also noted that wetlands in the Poconos of Pennsylvania were threatened by peat mining and resort development. Connecticut has lost over half of its original wetlands. The reduction of estuarine wetlands in portions of Rhode Island was high prior to 1971, when coastal wetland protection laws were passed by this state (Porter 1990). The enforcement of existing federal and state wetland protection laws would help prevent further fragmentation and destruction of harrier breeding and wintering habitats.

The ditching of saltmarshes for mosquito control may have negative effects on harrier breeding populations (Serrentino and England 1989). Approximately 90% of Atlantic coastal marshes were grid-ditched during the 1930s, a practice that results in a general drying of the marsh (Clarke et al. 1984). Ditching can also

increase the likelihood of flooding during high spring tides (when harriers may be incubating eggs) and may decrease avian saltmarsh populations (Clarke et al. 1984). The effects of ditching on small mammal populations are unknown (P. Dunne pers. comm., cited by Serrentino and England 1989).

Agricultural practices in the midwestern and western U.S. have contributed to declines in breeding harriers (Duebbert and Lokemoen 1977, Toland 1985, Hamerstrom 1986). In the Northeast, the value of agricultural areas to harriers has declined as a result of increased use of cropland versus hayfields (Andrie and Carroll 1988) and earlier haying and harrowing (Laughlin and Kibbe 1985). In addition, residential development continues to threaten the remaining populations of harriers in Massachusetts and Rhode Island (B. Nikula and D. Holt unpubl. data, C. Raithel pers. comm.).

### Food

During the breeding season, high prey densities have been associated with increased breeding success (Hamerstrom et al. 1985, Simmons et al. 1986a). The number of breeding harriers increased during periods of high meadow vole abundance in Wisconsin (Hamerstrom 1979, Hamerstrom et al. 1985) and New Brunswick (Simmons et al. 1986a). Hamerstrom et al. (1985) noted that nesting success showed a slightly positive relationship with vole abundance. In New Brunswick, clutch size was positively correlated with vole indices and high provisioning rates by males were associated with an increase in the number of harriers surviving to fledging (Simmons et al. 1986a). The starvation of nestlings was more common during vole population lows.

Harrier density and distribution may be affected by prey abundance during the winter. The number of harriers observed at winter roosts increased during winters when meadow vole abundance was high (Weller et al. 1955, Craighead and Craighead 1956). Bildstein (1979) also observed that the placement of roost sites was related, in part, to the density of prey in the

surrounding areas. Roost sites were commonly situated in the center of the birds' hunting areas.

### Diseases and Parasites

Raptors are susceptible to a number of bacterial and viral diseases. Numerous species of endoparasites, blood protozoans, and ectoparasites have been recorded in raptors. Little is known of the effects of diseases and parasites on wild populations because of the scarcity of data and the difficulty associated with separating the direct causes of death from diseases and parasites with the indirect causes (e.g., birds weakened by disease may succumb to death from predation, starvation and severe weather) (Newton 1979).

Rosen and Morse (1959) documented the death of harriers and short-eared owls from the ingestion of mice contaminated with the fowl cholera bacterium, *Pasteurella multocida*. During an outbreak of cholera among waterfowl in California, both mountain voles (*Microtus montanus*) and *Peromyscus* spp. had ingested portions of dead birds.

External and internal parasites have been found on free-living northern harriers. Peters (1936) found four species of external parasites on birds collected from Alaska, New Hampshire, Pennsylvania, and South Carolina: three species of biting louse (*Colpocephalum flavescens*, *Degeeriella fusca*, and *Philopterus taurocephalus*) and a Hypoboscid fly (*Lynchia americana*). In Wisconsin, another Hypoboscid fly, *Ornithoica vicina*, was found on an adult male and female harrier (Scharf 1966). Nestlings were eaten alive by carrion beetles (Silphidae) in Wisconsin (Hamerstrom 1969). Internal parasites have been observed in harriers, including two species of nematodes: *Cardiofilaria inornata* in Ontario (Anderson and Freeman 1969) and *Tetramerites robusta* (Tetrameridae) in Louisiana (Pence 1973).

### Contaminants

The effects of a variety of organochlorines on raptors have been well documented and include eggshell thinning, reproductive failure, and death (Newton 1979). Harrier populations in North America (Hamerstrom 1969) and Europe

(Bijleveld 1974, cited by Watson 1977) have been negatively affected by organochlorines. On Long Island, New York, high levels of DDE were found in one harrier egg from an abandoned nest (Foehrenbach et al. 1970), although no evidence of eggshell thinning was observed. In North America, a mean decrease of 15% in eggshell thickness was noted between 1947 and 1969 in eggs of the northern harrier (Anderson and Hickey 1974, cited by Newton 1979). In Illinois, a bird collected in 1975 had dieldrin levels of 4.5 ppm in its brain tissue (Havera and Duzan 1986), a level high enough to either cause the direct death of the individual or to have increased the bird's susceptibility to other forms of mortality.

In Wisconsin, harriers have been studied since 1959 (Hamerstrom 1969, 1979; Hamerstrom et al. 1985). Between 1960 and 1968 the number of harriers observed during migration declined markedly (Hamerstrom 1969). In addition, the number of nesting harriers did not increase in response to a high vole population in 1966 and 1967. Several behavioral changes, such as atypical transfer of prey from males to females and a sharp decrease in the frequency of "sky dancing" in migrating and breeding males were observed. The declines of both breeding and migrating harriers and the occurrence of behavioral changes coincided with the heavy use of DDT at the Wisconsin study site and elsewhere in North America.

Pesticide use in Vermont (Laughlin and Kibbe 1985), Connecticut (Dowhan and Craig 1976), and New Jersey (Dunne 1984) has been implicated in the decrease of harriers. A variety of biocides (DDE, dieldrin, PCBs) have been found in harrier eggs collected during the 1980s on Long Island, New York; however, the results of the study are not yet available (M. England pers. comm.). Hands et al. (1989) point out that few studies have been conducted on the long term effects of DDT and other biocides on harriers, with the exception of Hamerstrom (1969).

### Predation and Human Disturbance at Nest Sites

Because harriers nest on the ground, their eggs and young are vulnerable to destruction from

both human-made and natural causes. A number of mammalian and avian predators (e.g., skunks (*Mephitis spp.*), mink (*Mustela vison*), raccoons (*Procyon lotor*), dogs, and other raptor species) may prey upon eggs and young, and nests have been trampled by deer (*Odocoileus virginianus*) and cattle (Craighead and Craighead 1956, Hamerstrom 1969, Toland 1985, England 1989). Farming activities such as mowing and harrowing may cause nest abandonment by adults (Hamerstrom 1969, Follen 1986) and destruction of nests and young (Craighead and Craighead 1956, Hamerstrom 1969, Thomas 1987). Predation of harrier young has occurred when predators followed humans to nests (Watson 1977, Toland 1985).

### Weather

Cold or rainy weather may negatively affect harrier breeding success. Egg-laying may be delayed by cold weather (Watson 1977, Schipper 1979). Prolonged periods of rainy weather, particularly during the incubation and nestling stages, may cause nest desertion or death of nestlings from exposure (Follen 1986, Simmons et al. 1986a). Abnormally high tides have destroyed nests in coastal areas (Dunne 1986).

### Mortality

Juvenile mortality has been attributed to starvation and malnutrition (Craighead and Craighead 1956). In Great Britain, mortality was highest for first-year birds (Balfour and Cadbury 1975, Watson 1977). In the Orkney Islands, survival of hen harriers increased from 32% in the first year to 70% in the following year (Balfour and Cadbury 1975).

During the first half of the 1900s, mortality from shooting was common (Craighead and Craighead 1956). Out of 12 recoveries of harriers banded in North Dakota, 10 had been shot (Hammond and Henry 1949). In North America, death from shooting is no longer a serious threat to harriers (Bildstein 1988). In Great Britain, Watson (1977) reported that human-related mortality remained one of the most frequent causes of death for the hen harrier. Deaths also

resulted from starvation and collisions with automobiles and overhead wires.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Northern harrier populations should be monitored in areas where decreases have occurred or when the status of the population is unknown (Serrentino and England 1989). Surveys of suitable habitats should be undertaken to look for previously unknown populations of birds. These areas can be chosen from analyses of aerial photographs, investigations of historical breeding sites, the results of Breeding Bird Atlases and Christmas Bird Counts, and contacts with local birders. States that currently monitor harrier populations during the breeding season are New Hampshire, Massachusetts, Rhode Island, New York, New Jersey, and Maryland. However, monitoring programs differ among these states with respect to their intensity and scope.

To monitor breeding populations, the density and fledgling production of breeding birds should be collected. Biologists should check previously used or historic nesting sites for evidence of breeding activity since harriers may occupy the same site for several years. Because harriers may be polygynous, the total number of nests or females should be used to determine density and breeding success, and not the number of pairs. Fledgling production should be expressed as the number of fledglings produced per nest for all nests, including those that failed. Surveys should be conducted at a minimum of every other year. In areas where severe declines have occurred, yearly surveys may be required. Because harrier nest density may track small mammal populations, a decrease in the number of nesting birds may be the result of a low vole year and not the beginning of a serious decline, provided that habitat availability remains constant (Serrentino and England 1989).

Wintering populations should also be monitored. Information provided by Christmas Bird Counts can be used to identify areas that support high numbers of birds. Surveys of

suitable hunting habitats should be conducted by experienced observers throughout the winter. Data collected on surveys should include: (1) the number of birds observed, and sex and age (juvenile vs. adult) when possible; (2) weather variables, since flight activity is affected by weather conditions (Bildstein 1978); and (3) if time permits, hunting habitat and roost site selection.

### Management Procedures and Programs

The implementation of a management plan for harriers in the Northeast requires more detailed and accurate data on abundance and current population trends, habitat and area requirements, and more specific information, such as prey selection and predation rates. The following discussion of management recommendations for the northern harrier is based, in part, on Serrentino and England (1989); however, most of the information necessary to design a concise and scientific plan is currently lacking.

**Habitat Preservation.** - Habitat loss and degradation are most likely the primary causes of the decline of the harrier in the northeastern region (Dowhan and Craig 1976, Laughlin and Kibbe 1985, Robbins and Boone 1985, England 1989, Serrentino and England 1989). If the current trend of land use patterns in the Northeast continues, open land will decline and the amount of developed area will increase (Brooks 1989). Habitat preservation is imperative in states where remnant populations are (1) confined to islands or small areas of suitable habitat on the mainland, or (2) threatened by development of coastal areas, drainage of wetlands, reforestation, and other forms of habitat loss (Serrentino and England 1989). The passage of a bill to restrict the development of habitats critical to endangered wildlife would benefit harriers and other species in need of protection throughout the Northeast (M. England pers. comm.).

In addition to the passage of laws, the protection of key harrier breeding and wintering areas can be accomplished by land acquisition and conservation easements. These two methods

ensure the greatest amount of preservation (Therres 1989). When neither of these options are available, other forms of protection are possible, such as voluntary landowner agreements that do not require acquisition (Therres 1989). Under this category, signed cooperative agreements, registry programs, and tax incentives are recommended. Voluntary agreements may be most suitable for parcels which are located outside of protected areas (e.g., refuges and private conservation land) and are unavailable for acquisition. These agreements should stress the prevention of unnecessary disturbances to nesting pairs on private property and, where possible, the conservation and maintenance of hunting habitats associated with breeding and wintering birds (Serrentino and England 1989).

The amount of land required to maintain a viable population of harriers has not been established. Breeding and wintering densities will vary between areas because of differences in habitat types, availability and density of prey species, availability and distribution of suitable nesting sites, and the frequency of polygyny (Craighead and Craighead 1956, Clark 1972, Picozzi 1978, Balfour and Cadbury 1979, Hamerstrom et al. 1985). F. Hamerstrom (pers. comm., cited by Hands et al. 1989) recommended that an adequate amount of habitat should be protected to support more than four females during the breeding season.

The results of several long-term studies of harriers can provide land managers with general estimates of the size requirements necessary for the maintenance of breeding populations. At a 160-km<sup>2</sup> site in Wisconsin, between 4 and 34 harrier nests were found per year (Hamerstrom et al. 1985). In New Brunswick, a 60-km<sup>2</sup> portion of the reclaimed Tantramar Marsh supported 12-37 nests over a 5-year period (Simmons et al. 1986a,b). At a 120-km<sup>2</sup> site in Scotland, between 5 and 15 nests per year were found (Picozzi 1978). A 26-km<sup>2</sup> portion of barrier beach on Long Island, New York, supported from 9-14 nests each year (England 1989).

**Habitat Management.** - Harriers will breed and winter in a wide range of open lands provided that the following are available: (1) suitable breeding sites (e.g., cattail marshes, wet meadows,

and shrub uplands and wetlands; (2) hunting habitats, such as early successional fields, grasslands, and wet meadows; and (3) an adequate prey base, comprised primarily of small mammals and birds. Harriers breed in areas managed for prairie chickens (*Tympanuchus* spp.) that consist of early successional fields and dry and wet habitats (Hamerstrom 1969, 1979; Hamerstrom et al. 1985). Harrier breeding habitat in New Brunswick is composed of cattail marshes, wet meadows, upland hayfields, and abandoned fields (Simmons and Smith 1985). In northern New Hampshire, harriers continue to nest in an area where dairy farming and timber harvesting are the primary forms of land use (Serrentino 1987). Crops are rarely grown, and hayfields, abandoned fields, and shrub wetlands are common. Harriers breed in areas managed for waterfowl in the Dakotas, where fields are maintained in early successional stages composed of planted grass and legume species (Duebbert and Lokemoen 1977).

Management areas consisting of a complex of several different habitat types, such as dense shrublands and grasslands (dry and wet) and marshes may benefit breeding harriers. Nests have been found in areas where the dominant shrubs ranged from 0.5-2.0 m (Holt and Melvin 1986, Toland 1985, Serrentino 1987). In grasslands in the Dakotas, harriers nested in vegetation ranging in height from 30-cm to more than 60-cm (Duebbert and Lokemoen 1977). The dead vegetation remaining from previous growing seasons was an important component in nest site selection. At least one harrier nest was present in fields ranging in size from 11-54 ha. Harriers have nested in wet meadows comprised of bluejoint (*Calamagrostis canadensis*) and prairie cordgrass (*Spartina pectinata*) (Simmons and Smith 1985). Cattail marshes are also important nest sites and, in some studies, predation rates were lower in these habitats compared to drier areas (Sealy 1967, Simmons and Smith 1985).

Harrier hunting habitats must be capable of providing an adequate prey base for breeding, wintering, and migrating birds. The maintenance of early successional stages is recommended. Small mammals prefer abandoned fields and other disturbed habitats with vegetation cover consisting of dense grasses and weeds (Birney et al. 1976, Baker and Brooks 1981). In contrast, extensive

croplands and hayfields that are subjected to several annual cuttings may depress small mammal populations. Burning, grazing, mowing, and disking may be used to encourage early successional stages. The timing and frequency of these treatments would depend on characteristics of the particular site (e.g., location or successional stage).

At Cape May, New Jersey, a management program has been established to provide avian prey for migrating raptors, including harriers (Niles and Clark 1989). Some of the methods used included the following: (1) plowing and disking to maintain annual seed-producing plants; (2) planting fields with perennial pasture mixture and mowing every 2-3 years; (3) placing hedgerows between fields and providing islands of cover, such as shrubs, for prey species; and (4) plowing or disking perennial herbaceous species every 2-3 years. (See Niles and Clark (1989) for specifics of this management program.)

**Protection of Nest Sites.** - Harrier nests should be protected from disturbance by recreational activities (e.g., offroad vehicle use), timber operations (cutting and bulldozing), certain agricultural operations (mowing, plowing, etc.), and unnecessary nest visitations from both researchers and the public. Nest visitations should be avoided during the early part of the nesting cycle, especially from the pre-laying and egg-laying stages up to hatching (Hamerstrom 1969, Fyfe and Olendorff 1976). Predation of harrier young has occurred when predators followed humans to nests (Watson 1977, Toland 1985). In agricultural areas, haying and tilling has destroyed nests and young (Craighead and Craighead 1956, Thomas 1987).

The use of buffer zones around nest sites may be necessary in areas where human-related disturbance is likely to occur. The size of these buffers has not been established and would most likely vary with the habitat type surrounding the nest, nature of the disturbance, stage of the nesting cycle, and the individual behavior of the pair (White and Thurow 1985, P. Serrentino pers. obs.). Harriers will tolerate some agricultural activities during the breeding season. In New Hampshire, harriers nested in shrub wetlands and uplands adjacent to hayfields (Serrentino 1987).

The birds did not seem disturbed by haying operations, perhaps because they were familiar with the activity. In addition, the availability of suitable nesting habitat adjacent to agricultural fields may have prevented the birds from using the hayfields as nest sites. In Wisconsin, farmers left the area around nests unmowed or unplanted (Follen 1986). Follen also reported that nests in agricultural fields were abandoned or females left eggs or young for long periods during rainy weather when farmers were plowing.

***Regional Management Program.*** - A regional management program should be implemented in the Northeast for species such as the harrier that remain threatened by continued habitat loss. It may also be possible to design a management plan that would include other threatened species with similar habitat requirements such as the short-eared owl (S. Melvin and G.R. Tate pers. comm.). Particular issues which need to be addressed before a regional plan can be implemented include: (1) can harrier populations be maintained in areas where agriculture is declining; (2) should management efforts concentrate on breeding and wintering populations that occur in traditional habitats such as inland and coastal wetlands; and (3) how can key habitats be preserved in areas where the cost of land acquisition may already be prohibitive? In addition, cooperation and communication among biologists studying harriers in the Northeast are urgently needed to prevent the duplication of research efforts and to disseminate information on current harrier protection efforts.

***Education of the General Public.*** - A public education campaign would benefit harriers by increasing the awareness of this raptor's status in the region. Tate and Melvin (1987) used several methods to inform the public and private sectors on the conservation of the short-eared owl in Massachusetts. Their program included the presentation of current research and management efforts to local citizen groups and city or town officials, broadcasts on local cable television stations, and articles published in newspapers.

### Research Needs

- Determine the amount and type of disturbances that breeding harriers will tolerate, especially for populations located in coastal areas with high human densities (Serrentino and England 1989).
- Investigate the relationship between harrier wintering distribution and abundance in coastal Massachusetts, Rhode Island, and Connecticut, and the decline or lack of breeding birds in these same areas.
- Determine the effect of saltmarsh ditching on harrier populations and their major prey species (Serrentino and England 1989).
- Collect data on hunting habitat and roost site selection in various habitats such as saltmarshes, freshwater wetlands, agricultural habitats, and maritime heaths.
- Conduct analyses of pellets and prey remains found at roost sites to determine the prey selection of nonbreeding harriers.
- Determine the causes of breeding failure and mortality in young and adult harriers.
- Monitor the current levels of biocides in harriers and compare with the results of previous studies.
- Determine the sizes of hunting ranges of birds during the breeding and nonbreeding season at sites with varying densities of harriers and habitat types.
- Implement accurate and standardized survey methods to determine the trends of harrier populations on a regional level. Currently, the status of most populations in the Northeast is unknown because of the lack of reliable estimates of their abundance.
- Conduct studies on the techniques used to maintain early successional habitats. Comparisons between treatments and the

cost-effectiveness of each treatment are especially needed.

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# BLACK RAIL

## *Laterallus jamaicensis*

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**Black rails are secretive inhabitants of salt and brackish marshes, as well as wet meadows and the margins of freshwater wetlands. Short, dense vegetation and saturated or shallowly-flooded soils are characteristic of breeding areas. In the Northeast, the species is most common in the tidal marshes of Maryland, Delaware, and southern New Jersey. The significant loss of tidal marshes to dredging and filling, and alterations due to impoundments and pollutants, have all led to severe restrictions in the amount of available habitat for black rails. Their usage of drier wetlands, which are more easily and readily developed, has also contributed significantly to their localized, small numbers. Management programs for black rails must rely upon restoring natural water regimes and vegetational compositions on lands owned by state and federal agencies. Land-use practices such as ditching, impounding, dredging, and burning may be detrimental to black rails and should be studied in greater detail to determine optimal habitat management practices. Few data are available on population trends of black rails, although New Jersey and Maryland have recently begun monitoring black rails using tape-recorded vocalizations to elicit responses.**

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### DESCRIPTION

#### Taxonomy

The black rail (*Laterallus jamaicensis*) was first discovered in 1760 on the island of Jamaica (Allen 1900). Formerly called the little black rail, previous authors have recorded the scientific name of this species as *Porzana jamaicensis*, *Creciscus jamaicensis*, and *Rallus jamaicensis* (Allen 1990, Griscom 1923, Ripley 1977). Six subspecies are known: *L. j. jamaicensis* in eastern North America, Jamaica, and Cuba; *L. j. coturniculus* in western North America; *L. j.*

*murivagans* in western Peru; *L. j. salinasi* in Chile and western Argentina (Ripley 1977); a recently described *L. j. tuerosi* in central Peru; and *L. j. spilonotus*, which had been considered a separate species, *L. spilonotus*, in the Galapagos Islands (Fjeldsa 1983). Although this report concerns the nominate *L. j. jamaicensis*, the only subspecies known to occur in northeastern North America, pertinent data obtained from studies of *L. j. coturniculus* are included.

## Morphology and Plumages

The black rail is the smallest member of the family Rallidae breeding in North America. The sexes are similar in size, averaging 10-15 cm in length and weighing 29-38 g (Weske 1969, Ripley 1977, Todd 1977). Males are blackish above with a chestnut-brown nape patch and small white spots on the lower back, wings, rump, and tail. The head, breast, and upper abdomen are blackish-gray. The lower abdomen, flanks, and undertail coverts are blackish with narrow white barring. Adults have blackish-gray bills and red eyes (Ridgeway 1941). One study suggests most females are noticeably lighter-gray below and whitish in the area of the throat (W. R. Eddleman pers. comm.). Leg color descriptions include greenish or yellowish-green, "pinkish-horn," and dark grayish-brown (Ridgeway 1941, Wilds 1986). Whether this variation can be attributed to differences in age or subspecies is not documented. Several observers studying captive black rails in the Northeast agree on a brownish-gray or "grayed-off blackish-brown" description of the leg color (Meanley and Stewart 1960, Post and Enders 1969, Weske 1969). Juveniles are duller overall than adults and have brown eyes which change to red within the first 6 months, usually by October (W. R. Eddleman pers. comm.).

## Vocalizations

Black rails make a variety of calls. The most commonly heard vocalization, the primary advertisement call, is made by adult males, but may be given occasionally by females (H. Wierenga pers. comm.). The call has been transcribed as "kic-kic-kerr" (Kellogg 1962), "kickee-doo" (Robbins et al. 1983), and "ki-ki-krr" (Weske 1969). Although this call typically contains 3 notes, the 2 introductory "ki" notes are sometimes increased to 3-4, and extremes of 0-11 have been heard (H. Wierenga pers. comm.). This advertising call is usually repeated every 3-6 seconds (Reynard 1974), although lethargic singers may call much more slowly, and excited individuals often call as frequently as once per second (H. Wierenga pers. comm.). Advertising birds may call incessantly for hours.

Another call, which has been attributed to females, is a cuckoo-like "croo-croo-croo" or "who-whoo" (Reynard 1974). Black rails are also known to vocalize a transcribed "tic-tic-tic-McGreer" (H. Wierenga pers. comm.), a mystery song long debated in the literature (Kellogg 1962, Post and Enders 1969) and similar to a song frequently delivered by Virginia rails (*Rallus limicola*) and possibly other rails (H. Wierenga pers. comm.). Other vocalizations, including growls, barks, and clucks (H. Wierenga pers. comm., Kerlinger and Wiedner 1990), and a series of "kik" notes similar to a call of the yellow rail (*Coturnicops noveboracensis*) (Reese 1975) have been described.

Black rails in the Northeast usually call between an hour or two after sunset to an hour or two before sunrise (Reynard 1974). They also call, although less frequently and less persistently, during daylight hours (Reynard 1974, Reese 1975, Armistead 1990, H. Wierenga pers. comm.). Black rails in one southern New Jersey wetland are thought to call only during the day (Kerlinger and Wiedner 1990).

## HABITAT

Black rails in northeastern North America breed primarily in salt and brackish marshes. However, wet meadows and freshwater areas of narrow-leaved cattail (*Typha angustifolia*) and river bulrush (*Scirpus fluviatilis*) have also been documented (Griscom 1923, Proctor 1981, Armistead 1990). In salt or brackish marshes, home ranges generally include dense stands of saltmeadow cordgrass (*Spartina patens*) mixed with saltwater cordgrass (*S. alterniflora*), big cordgrass (*S. cynosuroides*), marsh spikegrass (*Distichlis spicata*), black needlerush (*Juncus roemerianus*), black rush (*J. gerardi*), or olney's threesquare (*Scirpus olneyi*) (H. Wierenga pers. comm., Kerlinger and Wiedner 1990). Black rails also occur in the dryer, upland edges of these marshes where saltmeadow cordgrass mixes with marsh elder (*Iva frutescens*) and groundsel tree (*Baccharis halimifolia*) in the saltbush community and with common reed (*Phragmites australis*) in disturbed areas (Kerlinger and Wiedner 1990).

Research in wetlands along the lower Colorado River has revealed that water depth is an important and perhaps key habitat component. Black rails there are found typically where the water depth is less than 2-4 cm (R. Flores pers. comm.). Other significant habitat factors may include vegetation density, distance to open water, and water regime stability (R. Flores pers. comm.). Black rails nest in the highest sections of the marsh, which have mesic to hydric soils and are flooded by only the highest tides (Todd 1977, Andrle and Carroll 1987). The area around the nest also typically includes lower wet areas, such as shallow pools and potholes (Andrle and Carroll 1987; W. Burt, W. R. Eddleman, and H. Wierenga pers. comm.).

## BIOLOGY

### Arrival, Nesting, and Departure Periods

Most black rails in the Northeast are probably migratory. Some of the earliest reported arrival dates are 10 April in New Jersey, 12 April in Maryland, and 19 April in New York (Bull 1964, Armistead 1990, Bull 1985). The peak calling period in the Elliott Island marshes of Maryland is thought to occur in early to mid-May (H. Wierenga pers. comm.), while peak calling in southern New Jersey marshes may occur from late April to mid-May (Kerlinger and Wiedner 1990).

The peak nesting period is from June to mid-July (Bull 1964, Kerlinger and Sutton 1989). The earliest egg date in the Northeast is a 16 May record of a nest with six eggs in Virginia (Bailey 1927). Other egg dates range from 20 May to 8 August in Maryland (Stewart and Robbins 1958, W. Burt pers. comm.) and from 30 May to 15 August in New Jersey (Bent 1926, Kerlinger and Sutton 1989). In Maryland, birds have been found still on territory until late September (H. Wierenga pers. comm.). Southward migration is thought to occur from late September to mid-October (Bailey 1913, Todd 1977). Some of the latest fall records are 7 November in Maryland, 29 October in New York, and 1 November in New Jersey (H. Wierenga pers. comm.; Bull 1985, 1964; respectively).

### Territory Size

Telemetry studies conducted on a resident population of black rails on the lower Colorado River showed an average nesting home range size of 0.43 ha with a significant core area of 0.10 ha (R. Flores pers. comm.). In this study, black rails nested in fringe marsh lining a lake. Home ranges in this habitat may be significantly different in size and shape than for black rails breeding in extensive saltmeadow cordgrass-dominated marshes in the Northeast. Although not determined through telemetry studies, territory size of black rails in the Elliott Island marshes of Dorchester County, Maryland is estimated to be 3-4 ha (J. S. Weske pers. comm.).

### Nest and Clutch Description

Black rail nests are constructed in dense vegetation within a few centimeters of the ground (H. Wierenga pers. comm.). Individuals breeding along the lower Colorado River sometimes built nests over water which was typically less than 2 cm deep (R. Flores pers. comm.). According to Pough (1951), nests are sometimes supported by a mat of the previous year's vegetation. The nest consists of a cup loosely woven from surrounding live and dead vegetation and is often covered by a domed arch of grass. Frequently, a ramp of dead grasses leads from the ground to an opening in the side of the nest. The nest has been compared to that of the eastern meadowlark (*Sturnella magna*) (Allen 1900, Harrison 1975).

Clutch size averages 6-10, and the oval eggs are buffy-white with fine brown dots (Harrison 1975). Incubation lasts for approximately 16-20 days and is performed by both sexes (Ehrlich et al. 1988). The clutch hatches synchronously, and the chicks leave the nest within approximately 24 hours (W. Burt pers. comm., Todd 1977). After hatching, the black downy young are precocial, but continue to be fed by the parents for an undetermined length of time (Ehrlich et al. 1988).

### Diet

The principal diet consists of aquatic invertebrates, especially insects, and the seeds of

aquatic vegetation (Ehrlich et al. 1988). The stomach contents of one black rail, collected at Elliott Island, Maryland in June of 1958, were analyzed and found to contain fragments of larval and adult aquatic beetles (Coleoptera). Three genera of the family Hydrophilidae were represented: *Enochrus*, *Hydrochara*, and *Tropisiterus*. Also identified was a curculionid, or weevil, from the genus *Calendra* (Spangler 1959).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

The distribution of black rails in North America is poorly known (Figure 1). Ripley (1977) gives the breeding range as extending along the East Coast from Massachusetts to Florida, and inland through West Virginia, Ohio, Indiana, Illinois, and Wisconsin. Breeding bird atlas (BBA) data from the Midwest, however, do not confirm any nesting there, although a few black rails were recorded in spring and fall outside of the migratory periods (Hands et al. 1989). Ripley (1977) also includes a restricted coastal area in California as supporting a resident population, as well as along the lower Colorado River in California and Arizona (Eddleman and Flores 1987). The species breeds also along the Gulf coast from Florida to Texas (Hunter 1990: pp.44-45); in Veracruz, Mexico; Belize; western Peru; Chile; and western Argentina (American Ornithologists' Union 1983).

Black rails winter primarily along the Gulf coast from Florida to Texas, and in Cuba, Jamaica, and Puerto Rico, southern California, and the Central and South American breeding areas (Ripley 1977, American Ornithologists' Union 1983, Hunter 1990).

Black rails are considered a *migratory nongame bird of management concern* in the U.S. because of their disjunct, restricted distributions and small populations (U.S. Fish and Wildlife Service 1987). The species has been most recently proposed as a *candidate (category 2)* for listing as a federally endangered or threatened species (U.S.

Fish and Wildlife Service 1991). The Natural Heritage global ranking is *G3*.

### Distribution and Status in the Northeast Region

According to Weske (1969), a few winter records exist in the Middle Atlantic and New England coastal saltmarshes. Maryland has had two winter reports, in 1973 and 1975, from the same marsh in Dorchester County (Bystrak 1974, Reese 1975). A third winter report came from coastal Worcester County in 1991 (H. Wierenga pers. comm.). Virginia's first winter report was in 1960 at Norfolk (Scott 1960). Since then, at least six winter reports have come from coastal Virginia marshes (reports from *The Raven*).

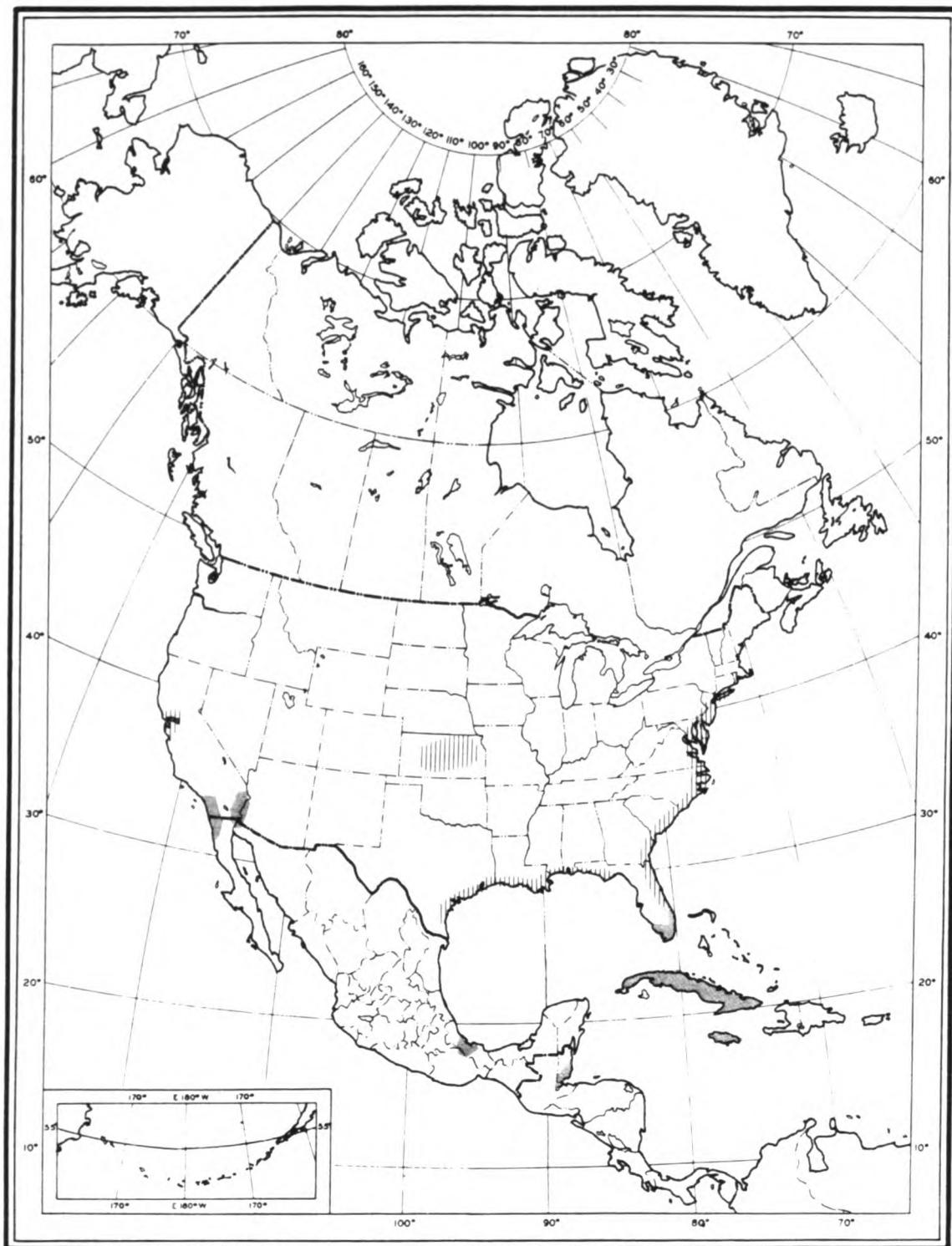
Surveys during state BBAs have located some black rails (Figure 2), but a more accurate overview of the status of this species in each state is available through reviewing literature documentation. The following discussion also reviews the black rail's state agency listing status and its Natural Heritage Program state rank (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia; Table 1). Unlike the legal status assigned by a state agency, the Heritage "S ranks" carry no added protection, however, they are based on standard ranking criteria from state to state. In the Northeast, black rails have been reported from Maine, Massachusetts, Rhode Island, Connecticut, New York, New Jersey, Pennsylvania, Delaware, Maryland, West Virginia, and Virginia. The following is a summary of the historical records and current breeding status of this species in each of these states.

**Maine.** - The only report of a black rail in Maine was an 1881 record from Scarboro on 4 October (Bent 1926).

**New Hampshire.** - Black rails have not been reported from New Hampshire.

**Vermont.** - Black rails were not recorded during the Vermont BBA (Laughlin and Kibbe 1985).

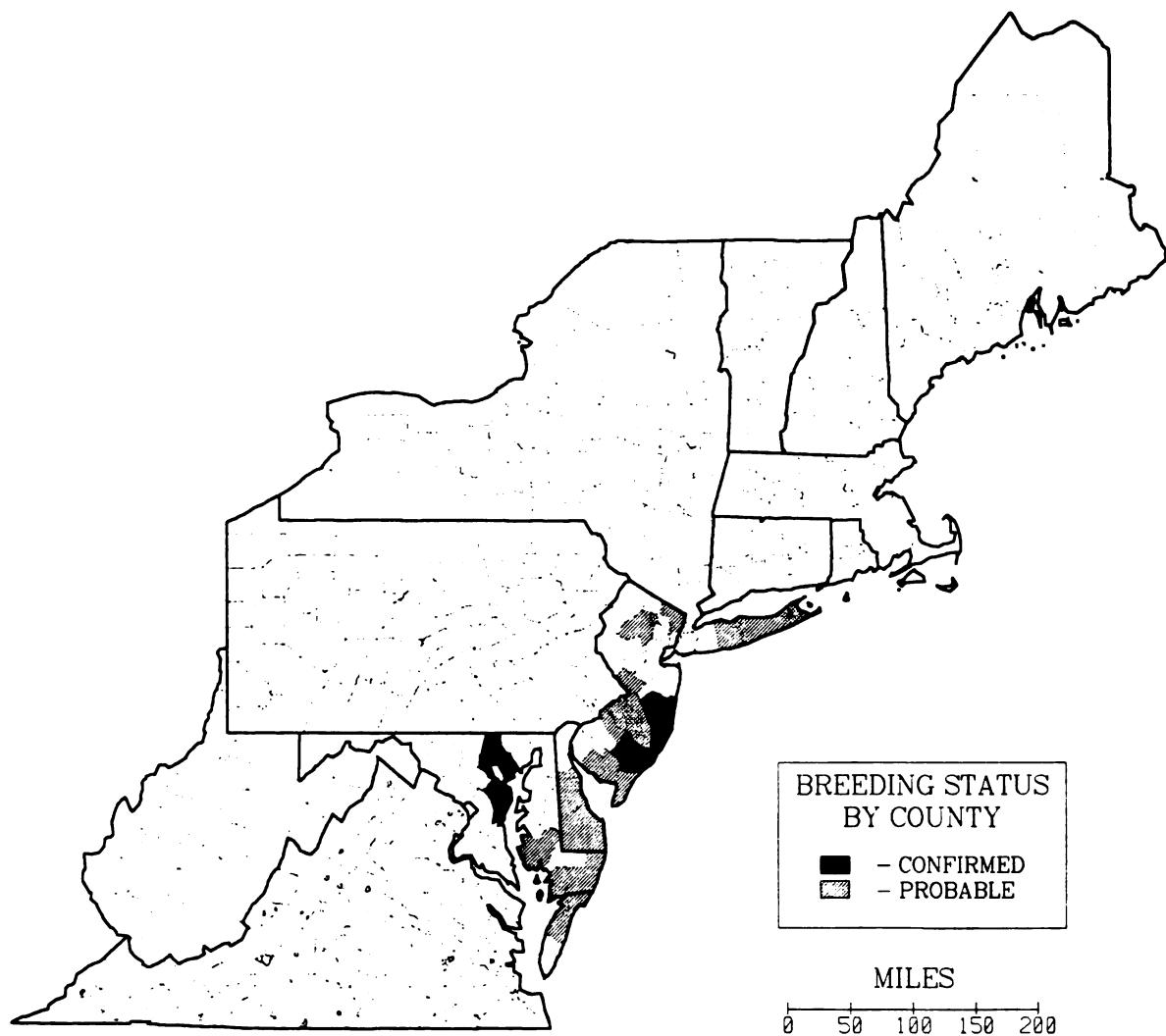
Figure 1. The North American range of the black rail (*Laterallus jamaicensis*) (American Ornithologists' Union 1983, National Geographic Society 1987).



Breeding

Year-round

**Figure 2.** Northeast breeding distribution of the black rail (*Laterallus jamaicensis*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the black rail (*Laterallus jamaicensis*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	-
New Hampshire	-	-
Vermont	-	-
Massachusetts	-	-
Connecticut	threatened	SH
Rhode Island	-	-
New York	special concern	S1
Pennsylvania	-	SA
New Jersey	threatened	S3
Delaware	-	S1?
West Virginia	-	-
Maryland	in need of conservation	S2
Virginia	uncertain status	SU

<sup>a</sup>SH = historical records only, but suspected extant

SA = accidental

SU = possibly in peril, but status uncertain

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

**Massachusetts.** - The earliest accounts of black rails in Massachusetts were of individuals found at Plymouth Harbor, Clark's Island, in August 1869, and Boston on 20 September 1874 (Howe and Allen 1901). The first confirmed breeding records were from Chatham in July 1884 and May 1885, and from Hazardville near the turn of the century (Howe and Allen 1901). However, Bull (1985) claims these breeding reports are erroneous. Other historic black rail reports from this state were on 16 May 1904 from Milton and on 5 May 1913 from Boston (Bent 1926). Both are listed as spring migrants. Two recent reports include an uncorroborated record of a calling bird on 24 May 1976 from Rowley Marsh at Stilt Pond, Essex County, and a specimen found dead in a yard on Nantucket Island on 31 March 1978 (W. Peterson pers. comm.). Presently, the breeding status of black rails in Massachusetts is

unknown and questionable. This species is considered an irregular vagrant, rather than a rare breeder (H. Woolsey and W. Peterson pers. comm.).

**Rhode Island.** - At least three black rail records have been documented in Rhode Island (D. Emerson pers. comm.). The earliest was of a bird collected 28 September 1970 from East Matunick Marshes near South Kingston, Washington County. A calling black rail was tape-recorded by D. Krauss (pers. comm.) at Green Hill, Washington County. This bird was present 8-13 June 1975. The third record was on 16 September 1988 when a bird was flushed from a marsh at Quicksand Pond, Newport County.

**Connecticut.** - The earliest confirmed black rail breeding records in Connecticut were of nests located on 13 July 1876 at "Cove meadows" and 6-13 June 1884 at Great Island (Clark 1884), both near Saybrook at the mouth of the Connecticut River (Sage et al. 1913). An 1897 nest was found on the western shore of the Connecticut River at Old Saybrook (Bent 1926). Other probable breeding reports came from the Quinnipiac Marshes, North Haven, in 1894 and 1904, and from Essex in 1904 (Sage et al. 1913). Unsubstantiated reports in the mid-1940s to late-1950s came from Guilford Beach, Great Harbor, and Stratford.

More recent reports include the collection of a young bird from along the Housatonic River near Milford Point on 5 September 1973 (Proctor 1981). On 25 June 1980 a black rail was caught and photographed at Dead Man's Swamp, near Cromwell, and an adult was located 15 September of the same year at Nell's Island marsh, near Milford (Proctor 1981). The Connecticut BBA (1982-86) turned up one undocumented breeding season record from Stonington, New London County (Bevier in press).

The Connecticut Department of Environmental Protection lists black rails as *threatened* in the state.

**New York.** - Although several reports of black rails in New York are from inland locations, no inland reports are substantiated by a preserved specimen or photograph (Bull 1985). The only

area in the state with accepted records is Long Island. The earliest reports were of specimens from Jamaica in the spring of 1879 and from Canarsie and South Oyster Bay in 1884 (Griscom 1923). Birds were found at Jones Beach in May of 1914, 1920, and 1922, and at Mastic in May 1920 (Griscom 1923). Although these records were assumed to refer to nesting individuals, confirmed breeding was not documented until 1937 when nests were located at Oak Beach on 20 June and Long Beach on 30 June (Bull 1985). The next report of a nest was at Lido Beach on 12 July 1940. Birds reported from the Moriches Bay area in early to mid-September of 1939 and 1953 may have bred locally (Bull 1985).

Since the mid-1960s, the Oak Beach area has been more intensively surveyed. In 1968, four black rails were banded, and observers estimated at least three pairs were present (Post and Enders 1969). Possibly five pairs were at Oak Beach in 1971, and three or four birds were present in May of 1974 (Bull 1985). Only one bird was found in the area in 1977, 1978, 1980, and 1983 (Andrle and Carroll 1987). Oak Beach Marsh, Long Island, is the only location for black rails reported to the New York BBA (1980-85) (Andrle and Carroll 1988).

The black rail currently is listed by the Department of Environmental Conservation as a *species of special concern* in New York. Under the Environmental Conservation Law, this category confers no added legal protection; however, it documents the concern which exists for the species' continued welfare in the state.

**Pennsylvania.** - The first U.S. black rail record was of an adult male and four young collected near Philadelphia on 22 July 1836 that were sent to Audubon (Allen 1900). Reports of August and September specimens taken in the late 1800s came from Bethlehem, the Delaware River near Chester City, and Lebanon County (Warren 1890). In western Pennsylvania, several were reported from Conneaut Lake on 5-7 September 1917 and one was seen at Crystal Lake, Hartstown, on 7 September 1925 (Todd 1940). However, the validity of the report from Conneaut Lake has been questioned (Grimm 1952). The most recent, and perhaps most reliable, breeding season reports are from the

Pennsylvania BBA (1984-88). According to B. Haas (fide E. A. Blom pers. comm.), black rails were reported to the atlas from Quarryville, Lancaster County in 1985 and from Colyer Lake, Centre County in 1986. Both reports were of at least one bird in small freshwater cattail (*Typha* spp.) marshes in cow pastures.

**New Jersey.** - The earliest confirmed black rail breeding records in New Jersey were from 1844 and 1845 when three nests were found at Beesley's Point near Great Egg Harbor River (Kerlinger and Sutton 1989). Specimens were taken at the mouth of Big Timber Creek in September of 1886 and at Salem (no date given) (Stone 1894). Bent (1926) reports nests located in 1912 at Brigantine and in 1919 at an island near Beach Haven. Since that time, many nests have been reported from southern New Jersey, both along the Atlantic Coast and the Delaware Bay. Only two breeding season reports of black rails came from north of the Raritan River (Kerlinger and Sutton 1989).

Before the 1950s, most nests were found in saltmarshes behind coastal barrier islands. Black rails were less well studied in the marshes of the Delaware Bay, with most records coming from the Turkey Point area of Cumberland County. During a 1988 census in southern New Jersey, observers found black rails at 14 of 59 sites, with 75% of 24 individuals located along Delaware Bay, predominately in Cumberland County. Cape May County had the second largest number of sites (Kerlinger and Sutton 1989). Black rails were absent from many historical sites in Atlantic barrier island marshes, and observers attribute their disappearance to intense wetland alteration and human intrusion (Kerlinger and Sutton 1989).

The New Jersey Department of Environmental Protection currently lists the black rail as *threatened* in the state.

**Delaware.** - The earliest record of black rails in Delaware was on 15 July 1939, when a nest and fledglings were found at Bombay Hook, Kent County. This is apparently the only documented nest record (G. K. Hess pers. comm.). Since the 1960s, breeding season reports have come from six localities. The following data are from *Birds of Delaware* (Hess et al. in press): Indian River Inlet, Essex County, had one report in the 1960s;

Port Mahon, Kent County, had one each in the 1970s and 1980s; Little Creek Wildlife Area, Kent County, had five reports in the 1960s, but none since; approximately six reports came from Bombay Hook National Wildlife Refuge, all in the 1980s; finally, the 1960s and 1970s each had four reports for Broadkill Beach, Essex County, but only one report came from there in the 1980s. Of these five locations, the largest reported number of individuals counted in one evening was 13 at Broadkill Beach in 1968. The sixth location is a 1988 report from a freshwater cattail and water lily (*Nymphaea odorata*) marsh near Delaware City, New Castle County (Armistead 1990). During the period of the Delaware BBA (1983-87), Kent County had four probable breeding records and a fifth came from Sussex County (Hess et al. in press).

Armistead (1990) reports a decline in black rails and other high marsh species, such as sedge wrens (*Cistothorus platensis*) and Henslow's sparrows (*Ammodramus henslowii*), in the Prime Hook National Wildlife Refuge/Broadkill Beach area in the early 1970s. Only two reports of black rails have come from that area since 1974 (Hess et al. in press). However, the low number of reports may be misleading since the amount of observer effort is not recorded.

The black rail is not included on any official state list as warranting special concern or protection. However, the Delaware Natural Heritage Program considers this species extremely rare in the state.

**Maryland and the District of Columbia.** - The earliest black rail record in Maryland was a specimen taken at Piscataway on 25 September 1877 (Brewster 1907), while the first nest was found 16 June 1931 in Dorchester County (Stewart and Robbins 1958). From the 1950s to 1980s, breeding season reports have come from at least 15 locations in 9 counties, all in the Coastal Plain region. Of these, the largest number of reports are from marshes in Dorchester and Somerset counties (seasonal reports from *Maryland Birdlife*). Black rails have been recorded more than five separate years from each of the following locations: Sandy Point State Park, Anne Arundel County; Black Marsh, Baltimore County; Elliott Island, Dorchester County; and

Deal Island Wildlife Management Area and Irish Grove Sanctuary, both in Somerset County. Most of the remaining 10 locations have had only one or two records (seasonal reports from *Maryland Birdlife*).

During the Maryland and District of Columbia BBA (1982-87), Chesapeake Bay marshes had the largest number of black rail reports. Several records were from the coastal marshes of Sinepuxent Bay and Chincoteague Bay. Of the five counties with black rail reports, only two, Anne Arundel and Baltimore counties, had confirmed breeding records (E. A. Blom pers. comm.: BBA data). Since the atlas period, breeding has been confirmed in Dorchester County (H. Wierenga pers. comm.).

The best known location is the extensive brackish marsh area called Elliott Island. Black rails have been found there for over 30 years, and the highest count reported was of at least 100 individuals in 1954 (Stewart and Robbins 1958, Armistead 1990). Other high counts at Elliott Island have included approximately 80 individuals one year in the early 1970s (H. Wierenga pers. comm.) and several counts, over a period of 20 years, numbering in the 30-40s (seasonal reports from *Maryland Birdlife*). Several factors, including weather, tide, phase of the moon, and amount of observer effort, may account for much of the annual differences in numbers counted (H. Wierenga pers. comm.).

Under the Nongame and Endangered Species Conservation Act, the Maryland Department of Natural Resources currently lists the black rail as a *species in need of conservation*.

**Virginia.** - The earliest breeding records of black rails in Virginia were of nests located at fresh ponds on Cobb Island and Hog Island, Northampton County, in 1911 and 1917 (Bailey 1927). In 1926, Bent reported this species from Wallops Island; however, Bailey (1927) believed this report was erroneous. Since then, spring location records from eastern Virginia include Sandbridge, near Cape Henry, in 1932 (Gray 1950); Rogue Island, Northampton County, in 1942 (S. Ridd pers. comm.); Dameron Marsh on Northern Neck in 1954 (Scott 1969); and Back Bay National Wildlife Refuge in 1956 (Perkins 1956). Dead individuals have been found in May

and October on the Chesapeake Bay Bridge and Tunnel islands (reports from *American Birds*, vols. 26, 33, and 35). It is unclear whether the black rail found 3 March 1973 at Fishermans Island, Northampton County, represents an early spring migrant or overwintering individual (Scott 1974).

Inland records exist for three counties in the southwestern corner of the state. One was seen 17 June 1946 at Abingdon, Washington County (Murray 1952). Reports during the breeding season from marshes at Blacksburg, Montgomery County, include one bird in 1939 and 1941, and four or more pairs in 1940. This species also was present from 7 November to 17 December 1943 (Murray 1974; Handley 1941a, 1941b). Roanoke County records came from Murray's Pond on 23 September 1936 and in mid-May of 1945 (Murray 1952, English and Hawkins 1945).

The Virginia BBA (1984-89) reported black rails from Saxis Marsh, Accomack County, in 1988 and 1989, and several have been found there in 1990 (S. Ridd pers. comm.). According to H. F. Day (pers. comm.), most of the barrier island marshes are not easily accessible. Until this area can be thoroughly surveyed, the species' distribution and relative abundance in the state will remain undetermined. The black rail is currently considered a species of uncertain status by the Virginia Department of Game and Inland Fisheries.

**West Virginia.** - The only report of a black rail in West Virginia was of an individual seen at Bluefield, Mercer County, on 28 April 1955. This report is not substantiated by a specimen or photograph (Hall 1983).

## Summary

Black rails have been reported from 11 of the 13 states in the Northeast. Of those 11, only 8 have reliable breeding season records: Rhode Island, Connecticut, New York, Pennsylvania, New Jersey, Delaware, Maryland, and Virginia. Four of these, Connecticut, New York, Pennsylvania, and Virginia, have breeding season records from only one or two locations in the 1980s. Four states, New York, Connecticut, New Jersey, and Maryland, have recognized this species as

deserving attention or protection by placing it on respective state threatened and endangered species lists (Table 1).

The black rail's stronghold in the Northeast are the tidal marshes of Maryland, Delaware, and southern New Jersey. The number of black rail breeding pairs in these three states is unknown; however, the combined total of all black rail breeding areas reported in the 1980s from this region is approximately 40. This number may be misleading because of the ambiguity of the term "breeding area." The information contained in this paper is compiled from various sources and may not be directly comparable because of the lack of a standardized definition of "breeding area" or any similar term. However, this term is used by the author to refer to separate place names used by observers in reporting black rails during the breeding season. The majority of these 40 breeding areas come from 5 counties: Cumberland and Cape May counties in New Jersey; Kent County, Delaware; and Dorchester and Somerset counties in Maryland.

## LIMITING FACTORS AND THREATS

Limiting factors are those which increase mortality rates and/or decrease reproductive success. Although black rail mortality rates and longevity are undetermined, many natural and human-created causes of mortality have been documented.

### Predation

Predation by various species, especially during extreme high tides when black rails are forced from their dense cover, is well documented in California (Evens and Page 1986). The most common predators included great egrets (*Casmerodius albus*), great blue herons (*Ardea herodias*), and northern harriers (*Circus cyaneus*). Other predators include great horned owls (*Bubo virginianus*), short-eared owls (*Asio flammeus*), ring-billed gulls (*Larus delawarensis*), domestic cats, and possibly loggerhead shrikes (*Lanius ludovicianus*) (Orr 1947, Weske 1969, Evens and Page 1986). Although undocumented, predation

by other species such as foxes, snakes, snapping turtles (*Chelydra serpentina*), feral dogs, and raccoons (*Procyon lotor*) is likely.

### Weather

Severe weather during migration can also increase mortality rates. Strong winds or low cloud ceilings may cause migrants to land in hostile environments (Cobb 1906) and probably increase collisions with human-made structures, such as tall towers and buildings (Browne and Post 1972, Hands et al. 1989). Storm tides and abnormally high water levels can reduce reproduction by flooding nesting areas (Bailey 1927, Todd 1977). Although undocumented, weather may also decrease reproductive success by reducing the invertebrate prey base during extreme drought or flooding conditions.

### Human Disturbance

Collisions with human-made structures, such as lighthouses, towers, buildings, and wires, are a well-documented mortality source (Emerson 1904, Browne and Post 1972, Todd 1977, Hands et al. 1989). Humans have also directly increased mortality levels through various other means, including hunting (Bailey 1913, L. Bevier and H. Wierenga pers. comm.); automobile strikes (Orr 1947); trampling by birdwatchers (Evens and Page 1986); decapitation by mowers (Clark 1884); and possibly trapping (Eddleman et al. 1988).

### Contaminants

Chemical contamination is another potential limiting factor. Ingestion of lead shot by soras (*Porzana carolina*), a close relative of the black rail, has been documented in Maryland, and lead residues at levels lethal to waterfowl were discovered in the tissues of some of these birds (Stendell et al. 1980). Although undetermined, black rails may also be contaminated by pesticides which are applied to saltmarshes or leached into wetlands from nearby agricultural fields.

### Habitat Loss

Probably the single greatest threat to the black rail is the loss and degradation of its wetland habitat (Todd 1977, Kerlinger and Sutton 1989, Kerlinger and Wiedner 1990). According to Tiner (1984), only 46% of the original 87 million ha of wetlands in the U.S. remained by the mid-1970s. Between the mid-1950s and the mid-1970s, 7,300 ha of estuarine and 178,000 ha of palustrine wetlands were lost each year. Most of this national wetland loss was attributed to agricultural development.

Two northeastern states, New Jersey and New York, suffered significant wetland loss from dredge and fill residential development in coastal areas (Tiner 1984, Kerlinger and Sutton 1989). Also, approximately half of the original wetland acreage in Connecticut has been lost (Tiner 1984). The coastal marshlands of several states in the Northeast diminished by hundreds to thousands of hectares each year in the 1970s (Tiner 1984).

Tiner (1984) lists at least nine different direct and indirect human-induced threats to wetlands. These include drainage, dredging, filling, impounding, mining, and pollutant discharge. He also includes five "natural" threats: drought, subsidence, hurricanes and other storms, erosion, and biotic effects (goose, muskrat (*Ondatra zibethicus*), and nutria (*Myocastor coypus*) "eat-outs"). The latter has been included as one of the probable causes for the loss of over 5,000 acres (2,024 ha) of marsh at Blackwater National Wildlife Refuge, Maryland (Jayne 1990).

Another potential threat to black rail habitat is marsh burning. Some saltmarshes in Maryland are burned annually, typically in late winter (S. Dawson pers. comm.). While black rails have been known to nest on top of a mat of dead vegetation from the previous season (Pough 1951), the importance of dead vegetation to this species is unknown. The effect of marsh burning on black rail reproductive success and habitat quality is unknown and subject to vigorous debate (H. Wierenga pers. comm.).

In many areas, common reed (*Phragmites*) is a weedy, invasive species which threatens the integrity of native wetland communities. Disturbed and stressed wetlands are more susceptible to *Phragmites* invasion than

undisturbed wetlands (Marks 1986). The effect of *Phragmites* invasion and dominance in black rail breeding habitat on the species' mortality rate and reproductive success is unstudied, yet some believe it to be negative (H. Wierenga and P. Kerlinger pers. comm., Hess et al. in press).

## MANAGEMENT POTENTIAL

Black rails in the Northeast have been rarely studied. Very little quantitative data exists on the basic biologic or population parameters of this species, and the factors influencing population change and reproductive success are poorly known. Without this information, it is difficult to assess the potential for restoring black rails to wetlands from which they are presumed extirpated or greatly reduced. In New Jersey and Maryland, black rails may no longer breed in certain wetlands that have been altered or degraded because of human activities (Kerlinger and Wiedner 1990, H. Wierenga pers. comm.). Restoration of viable black rail populations to these areas may be dependent upon restoring natural water regimes and vegetation compositions.

## LAND PROTECTION AND PRESERVE DESIGN

The minimum size of habitat required to sustain a breeding pair of black rails is undocumented and may be larger than home range or territory size alone. For example, many species of passerines require forests much larger than their territory size to successfully reproduce (Robbins 1979). This has been rarely studied in marsh birds.

An important element in designing a preserve to protect black rails would be to include enough area to protect their habitat from alteration, especially from changes in water regimes or vegetation composition. In an ideal situation, the primary ecological boundary (the smallest habitat delimiter needed to ensure a sustainable population) should include the entire marsh from which one or more black rails were

located during the breeding season. This would be the most effective way to protect the area's hydrology, an important factor in determining wetland structure and composition.

A protection boundary smaller than the entire marsh could be considered if the wetland were too extensive to be included entirely in a preserve or if black rails were utilizing only a small portion of a large wetland. In these situations, a minimum preserve boundary should at least include the entire territories of all black rails present. However, it is difficult to determine the exact boundaries of breeding territories from the locations of advertising adults.

Site fidelity should be another consideration when designing a preserve smaller than the entire marsh. Even though black rails have been known to return to the same location over consecutive years (H. Wierenga pers. comm.), their degree of loyalty to a particular site in a large marsh over a number of years is undetermined. A preserve established to protect breeding black rails must be large enough to allow for annual movement of territories within the preserve boundary. Thus, a larger preserve increases the chances of future black rails breeding inside the boundary.

To provide adequate buffering against potential threats and adjacent activities, an outer, or secondary, ecological boundary should include larger areas of remaining marshland and nearby upland areas. Upland areas may provide necessary cover for black rails during extreme high tides and flooding conditions (Evens and Page 1986, W. R. Eddleman pers. comm.) and may provide a buffer against land management activities near the marsh.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Only two states in the Northeast have implemented recent black rail studies. In 1988, a preliminary study was conducted for the New Jersey Department of Environmental Protection's Endangered and Nongame Species Program to determine the distribution of black rails in southern New Jersey (Kerlinger and Sutton 1989).

This study was expanded the following year to include analyses of seasonal and daily patterns in black rail vocalizations and of the habitats utilized by vocalizing birds (Kerlinger and Wiedner 1990).

In 1990, a multi-year study was initiated by the Maryland Department of Natural Resources to investigate the status and distribution of the black rail, to determine the environmental factors influencing vocalization, and to develop a standardized censusing technique (H. Wierenga pers. comm.). Qualified researchers, familiar with the many types of black rail vocalizations, used tape-recordings in an attempt to induce responses. Surveys in Maryland were conducted at night, usually between 10 p.m. and 4 a.m., and listening points were located primarily along roads in areas of appropriate habitat. Preliminary results indicate this species is more widespread than previously known. In addition to locating black rails at new sites in both Dorchester and Somerset counties, researchers discovered a sizable population in coastal Worcester County (H. Wierenga pers. comm.).

Breeding bird atlases conducted throughout the Northeast have added greatly to the known distribution of most land birds. However, wetland species require much more intensive survey efforts before an adequate level of information can be obtained. Both New Jersey and Maryland have initiated such studies for the black rail. Other states in the Northeast, especially Delaware and Virginia, need to conduct similarly intensive surveys to more accurately determine population levels and trends.

#### Management Procedures and Programs

No state in the Northeast presently implements any management procedures or programs specifically designed to protect or enhance black rail populations or breeding areas. Many black rail breeding areas in the region are located on state wildlife areas and national wildlife refuges. However, large areas of high marsh on these public lands have been impounded and are managed primarily for waterfowl. Water levels in these areas are typically too high to support the habitat required by black rails.

Methods for managing wetlands which would promote both waterfowl and most species of

North American rallids are known (Eddleman et al. 1988). However, black rails require much drier wetland areas than nearly all other North American rallids. Waterfowl management procedures which are compatible with the maintenance of habitats for most rallids may still be detrimental to black rails. Until more detailed information becomes available on black rail habitat and breeding requirements, management actions designed to promote this species should include preserving black rail breeding areas, restoring high marsh areas flooded by impoundments, securing adequate upland buffer for marshes near agricultural lands, minimizing human disturbance in known breeding areas during the breeding season, and possibly developing appropriate predator control programs (Todd 1977, Hands et al. 1989).

#### Research Needs and Programs

As stated earlier, few black rail studies have been conducted in the Northeast, and very little is known about even their most basic biological requirements. A fairly comprehensive list of research needs has been compiled by Hands et al. (1989). Although the list was developed for black rails in the northcentral U.S., it also applies to this species in the Northeast. Some of the basic research needs include determining reproductive success and productivity; longevity; nestling, fledgling, and adult mortality rates and causes; diet; diseases and parasites; site and mate fidelity; home range size; wintering, migratory, and breeding locations, and relative abundances at these locations; detailed habitat requirements along migratory routes and at breeding and wintering areas; analysis of contaminant levels and their effects on mortality and reproduction; and interspecific and intraspecific competition levels.

Some of the most important research needs are those which would have direct implications for land management. As discussed previously, certain land-use practices, including ditching, impounding, dredging, and burning, are considered potential threats to black rail habitat. The effects of these practices on habitat quality and quantity should be studied, and the degree of threat both to the habitat and to black rail populations should be evaluated. Land manage-

ment techniques designed to restore, maintain, and enhance black rail habitat should be developed and implemented on state and federal lands throughout the Northeast.

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# GULL-BILLED TERN

## *Sterna nilotica*

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Gull-billed terns nest in colonies with other waterbirds along the Atlantic coast from New York south to the Gulf coast. In the Northeast region, the species is scarce from New York to Maryland, and the largest colonies in Virginia have declined considerably. Loss of upland foraging sites and island nesting sites has probably been the greatest threat to gull-billed tern populations, although predation and competition with gulls, and human disturbance at colonies have also contributed to population declines. Protection of nesting tern colonies by posting, gull control, and creation of artificial nesting islands are management techniques that could assist with recovery of populations. Annual nesting surveys should also be conducted to monitor population status. Research activities should address foraging requirements and movement of populations among colonies as well as the ecology of the species during migration and in winter.

### DESCRIPTION

#### Taxonomy

The gull-billed tern, *Sterna nilotica*, is placed in the genus *Sterna* in North America, whereas Europeans have retained it in the monotypic genus *Geochelidon* (American Ornithologists' Union 1983, Cramp 1985). The gull-billed tern is a nearly cosmopolitan species with six subspecies, only one of which, *S. nilotica aranea*, occurs along the east coast of the U.S. south to the West Indies and Mexico. A second subspecies, *S. nilotica vanrossemi*, occurs on the Pacific coast.

#### Morphology and Plumages

The gull-billed tern is a stout, white, bluntnosed tern that feeds in the marshes and adjacent coastal uplands of the southern and Gulf coasts of the U.S. (Forbush 1939, Harrison 1983, Cramp 1985). It is similar in size to other medium-sized terns, but lacks the long tail streamers of common terns (*Sterna hirundo*), Forster's terns (*S. forsteri*), and roseate terns (*S. dougallii*) terns. The gull-billed tern is also longer-legged and broader-winged than these species. It can be told from the sandwich tern (*S. sandvicensis*) by its shorter beak and tail, shallower wingbeat in flight, and more upright posture while sitting. This tern's flight is

usually more buoyant and gull-like than that of other terns (Vinicombe and Harris 1989).

While in breeding plumage, a black cap extends from the lores of the gull-billed tern, including the eyes, to the nape. The rest of the upper parts, wings, and tail are a pale pearl gray. The side of the head, underparts, wing linings, tail, and underwing coverts are white. The primaries are grayish-black underneath and frosty-gray above. The legs and feet are black. The heavy black bill lacks the sharp tip of other terns and is stouter and proportionately shorter. The sexes are similar in appearance. Winter plumage is similar to the breeding plumage except that the black cap is nearly absent, with only some remnant spotting near the rear of the crown. A blackish patch extends from the eye to the auriculars, although the extent of this is quite variable (Harrison 1983, Cramp 1985).

Recently fledged juveniles are similar to adults in winter plumage except that the head is darker with more blackish spots and the gray back and upper wing are edged in tan, giving the back and wing a buff-colored appearance when the bird is in flight.

The downy young are variable in appearance but generally cream, buff, or peach colored, with darker down on the dorsal surfaces. Young usually have two dorsal stripes on the crown, nape, and back and a distinctive dark smudge behind the eye. The bill is typically light pink at hatching and darkens with age. The feet are light pink and darken to an orange-brown with age (Harrison 1983, Cramp 1985).

### Vocalizations

The typical call of the gull-billed tern is a nasal "tee-hee-hee" or "kat-y-did" (Bent 1921). Terns attacking terrestrial predators will frequently utter a harsh "grack" call during defensive dives (Sears 1981). Other calls are described by Bent (1921), Sears (1981), and Cramp (1985).

## HABITAT AND BIOLOGY

### Nesting

Wilson (1840) and Stone (1908) reported that the gull-billed tern was a marsh-nesting species along the coast of New Jersey. Bent (1921) concluded that this species had been driven to nest on barrier beaches because of hunting at sites on inner dunes, saltmarshes, and islands. Nesting sites are presently confined to sandy barrier islands, beaches, sandy shores of saline lagoons and marshes, and artificially-produced dredge spoil islands (Clapp et al. 1983). Regional differences in nest sites occur, with the percentage of gull-billed terns nesting on spoil islands ranging from 28% in New Jersey to 60-80% in North Carolina and 70-84% in Texas (Clapp et al. 1983).

The gull-billed tern is a colonial nester and typically forms interspecies colonies with common terns and black skimmers (*Rynchops niger*). Least terns (*Sterna antillarum*), royal terns (*S. maxima*), sandwich terns, and caspian terns (*S. caspia*) may also be present. Most of the gull-billed tern colonies studied in Virginia and North Carolina were small (mean = 45 birds) (Erwin 1978).

Nests are generally located close to landmarks, such as plants or pieces of driftwood, and are usually slight depressions containing the eggs with rims of dried straw and/or shell fragments (Harrison 1975, Sears 1978). Some nests are more elaborate piles of accumulated shell fragments, which may serve to provide protection from drifting sands (Sears 1978). The appearance of the nest lining and perimeter varies greatly between nests and from day to day in the same nest, depending on the individual, the time available for placing the lining, and weather conditions (Sears 1976, 1978).

Distances between nests ranges from 2-114 m, with a mean of 21 m (Sears pers. comm. cited by Cramp 1985), although the inter-nest distance may vary as a function of colony size (Møller 1982). Vegetation in nesting areas is sparse, e.g., approximately 15% of the ground was covered in a North Carolina colony (Soots and Parnell 1975).

Clutch size is one to three eggs (Bent 1921, Sears 1978, Møller 1981) and occasionally four (Bent 1921, Forbush 1939, Pemberton 1927,

Harrison 1978). Clutch size in one study was found to be significantly greater in larger colonies (Möller 1981). Renesting attempts usually result in smaller clutches (Sears 1978).

The background color of the eggs may vary from buff to olive and the mottling is somewhat finer grained than the pattern of the common tern egg. The eggs have a characteristic "frosty" appearance which also distinguishes gull-billed tern eggs from those of the common tern (Bent 1921, Harrison 1975). The eggs of the gull-billed tern are cryptically colored, and the immediate area of the nest is not white-washed.

Like many terns, the gull-billed tern has a highly ritualized courtship behavior which includes aerial flights, a variety of terrestrial displays, and courtship feeding (Bent 1921, Lind 1963, Sears 1976, 1981).

Gull-billed terns display a variety of behaviors which reduce the risk of nest predation. For instance, they walk a short distance from the nest to defecate (Sears 1978). They also frequently remove eggshells from the nest after hatching (Cullen 1960), but this response is not as strong as it is with other tern species, probably because the cryptically-colored young leave the nest several days after hatching (Sears 1978).

The eggs are incubated by both the male and the female for 22-24 days (Harrison 1975), and only one brood is raised per season (Bent 1921, Forbush 1939). The major factors that determine the time at which the young leave the nest are the age of the chicks, the proximity of vegetation, and disturbance (Sears 1978). The young fledge at 4-5 weeks of age (Harrison 1975).

Nesting success appears to be low, with many colonies producing no young at all (Blus and Stafford 1980). The main causes of breeding failure are flooding of low-lying colonies, disturbance by humans, and predation (Clapp et al. 1983).

## Foraging

Foraging normally takes place within 10 km of the colony (Sears 1976). Unlike other species of terns, gull-billed terns rarely feed over open water (Ashmole 1971); instead they forage by flying low over marshes, open fields, shrubs, and

agricultural fields, with a gull-like flight. Upon locating food, they stall briefly, then drop to the ground, vegetation, or substrate to capture their prey (Cramp 1985). Their feeding behavior is similar to that of the laughing gull, *Larus atricilla*, with which they are frequently observed feeding over agricultural fields (Meanley 1981).

Gull-billed terns sometimes forage over quiet estuaries and ponds, where they pick fish and invertebrates from the surface. Unlike other species of terns, they rarely dive. They sometimes feed with black skimmers on small fish concentrated in drying marsh pools (J. Via pers. obs.). Gull-billed terns rarely feed in conspecific groups because many of their invertebrate prey tend to be dispersed rather than aggregated like the fish schools exploited by other species of terns (Erwin 1978).

Major dietary items include arthropods, locusts, grasshoppers, dragonflies, insects, spiders, and marine life such as fiddler crabs, crustacea, crabs, and sand bugs (Wilson 1840, Bent 1921, Sprunt 1954, Rohwer and Woolfenden 1968, Cramp 1985, Quinn and Wiggins 1990). Vertebrate fauna consumed by the gull-billed tern include fish, frogs, toads, lizards, small mammals (Bent 1921, Dement'ev et al. 1951, Bannerman 1962), green anoles (*Anolis carolinensis*) (Rohwer and Woolfenden 1968), and mice (Hobbs 1976).

Gull-billed terns are occasional opportunistic predators on the downy chicks of other beach-nesting birds including gull-billed (Zubakin 1975) and least terns (Densmore 1990), and possibly piping plovers, *Charadrius melanodus* (R. Cross pers. comm.).

## Migration

Gull-billed terns usually arrive at their breeding colonies in mid-April and initiate nesting in mid to late May. Young hatch in mid-June. Adults may begin the migration south upon the completion of nesting, usually in late July and early August (Erwin 1979).

## Winter

Little is known about the winter biology of this species in Central and South America.

Coastlines, estuaries, and marshes are primary habitats, but some birds are also found along freshwater lakes, marshes, and agricultural fields (Clapp et al. 1983).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### World Distribution and Status

The gull-billed tern has disjunct breeding populations in Europe (Denmark, southern France, the Balkans, southern Russia); Asia (southern Iran, Pakistan, southeast China, and elsewhere in southeast Asia); northwest Africa; Australia; North America; and South America (Dement'ev et al. 1951, Cramp 1985).

Møller (1975) estimated the total breeding population in 1972 as 24,000 for Europe, Africa, and Asia. The main breeding areas are the Banc d'Arguin, Mauretania, with 13,000 nesting birds in 1951 and the Soviet Union with 4,000 (Møller 1975). No global population trends are apparent although local increases and decreases are noted for different subpopulations (Møller 1975, Cramp 1985). The species' Natural Heritage rank is G5 (demonstrably secure) (The Nature Conservancy, Central Scientific Databases, Arlington, Virginia).

In the New World, gull-billed terns are found along the Pacific coast from southern California and Mexico south to Ecuador, on the Atlantic coasts of Brazil, Uruguay, and Argentina, and on the Atlantic and Gulf coasts of the southern U.S., south to Mexico (American Ornithologists' Union 1983). Population estimates are not available for most of this range.

### Distribution and Status in North America

Two disjunct breeding populations of gull-billed terns occur in the U.S. (Figure 1). One small inland population breeds on the salt flats of the Salton Sea in California and the second population is centered along the Gulf and southeastern coasts, nesting in diminishing numbers north to New York (American Ornithologists' Union 1983). There seems to be little information on the inland California

population, *Sterna nilotica vanrossemi*, on the Salton Sea (Pemberton 1927, Small 1974), except that the breeding population may be only 100 pairs and thus a candidate for *endangered* status (C. Molina pers. comm.). In the Southeast, Clapp and Buckley (1984) estimated a population of 3,019 pairs of gull-billed terns, which they believed to represent 95% of the U.S. population and perhaps 10% of the world population. Two states held most of the 3,472 individuals reported by Clapp et al. (1983): North Carolina (26%) and Texas (56.5%).

The wintering grounds of the gull-billed tern include the Gulf coast from northern Florida through Central America, the West Indies, and South America to northern Argentina on the Atlantic Coast. On the Pacific Coast, the species winters from northern Mexico to Peru (American Ornithologists Union 1983). Kress et al. (1983) indicated that the U.S. population of gull-billed terns is declining. The U.S. Fish and Wildlife Service (1982) listed the gull-billed tern as a species with a decreasing population and more recently placed it on the 1987 list of *migratory nongame birds of management concern* (U.S. Fish and Wildlife Service 1987). On the other hand, Clapp and Buckley (1984) felt that there had been little change in gull-billed tern populations in North Carolina and Texas between the early 1900s and 1984.

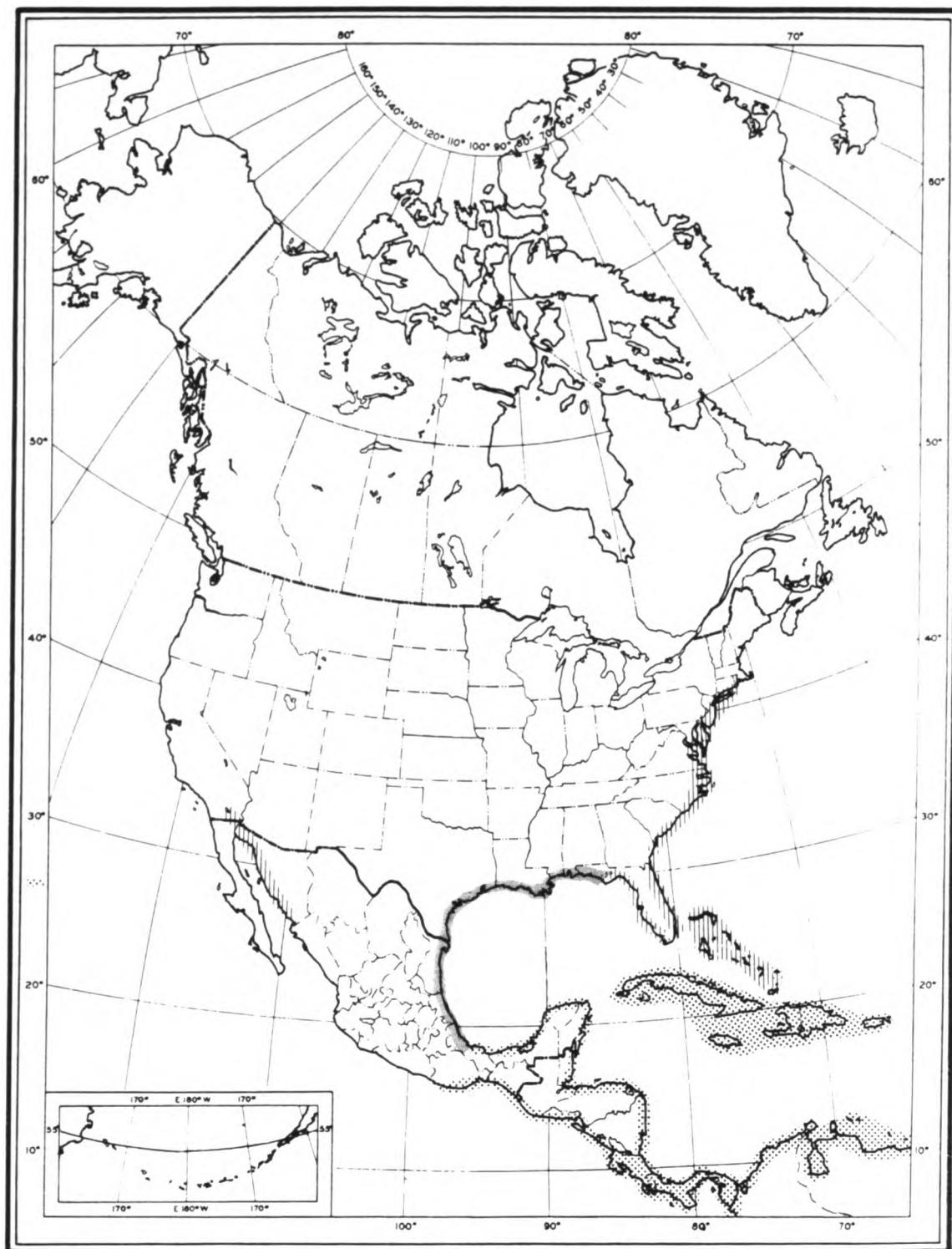
### Distribution and Status in the Northeast Region

The gull-billed tern has probably never been very common in the northern portion of the Northeast region compared with areas farther south. Numbers may be much reduced from those of the early 19th century, before the advent of market and plume hunting, although no documentation exists.

More recently, Nisbet 1983 (in Kress and Nisbet 1983) reported 2,000 pairs in the Northeast in 1975-76, and 1,000 pairs in 1980-82. Buckley and Buckley (1984) also estimated the gull-billed tern population in the Northeast region to be 1,000 pairs in 1982.

Population changes in the Northeast are probably responses to changes in populations

**Figure 1.** The North American range of the gull-billed tern (*Sterna nilotica*) (American Ornithologists' Union 1983).



Breeding



Year-round



Wintering

farther south. Expanding populations or breeding ranges in northern states might reflect "overflow" birds from increasing southern populations (Buckley and Buckley 1984). Similarly, decreases in the northern population might reflect reduction in numbers at southern colonies, as well as other factors such as destruction of northern breeding habitats or increased mortality of gull-billed terns on their wintering grounds. Distinguishing among these and other possible causes of decline is difficult and probably requires long-term data on breeding and wintering numbers throughout the range. Counts from single colonies or even single regions may be impossible to interpret in isolation. Data from state breeding bird atlases (BBAs) indicate population distribution in the Northeast (Figure 2).

**Maine.** - Gull-billed terns do not breed in Maine and sightings of vagrants are rare (P. Vickery pers. comm.).

**New Hampshire.** - No breeding or sight records exist for New Hampshire (D. Abbott pers. comm.).

**Vermont.** - There are no breeding records (Laughlin and Kibbe 1985) or sightings for Vermont (N. Murton pers. comm.).

**Massachusetts.** - In early summer and after hurricanes this species has been an uncommon but regular visitor to Massachusetts (Hill 1965, Veit and Petersen in press). Brewster reported nesting on Nantucket Island in the 1800s, but there are no recent breeding records of gull-billed terns in Massachusetts (Veit and Petersen in press). Adult gull-billed terns have appeared in mixed-species waterbird colonies in Massachusetts, so nesting might be expected in the future (Buckley and Buckley 1984).

**Rhode Island.** - The gull-billed tern is listed as a "... rare, irregular occurrence, not seen every year" in Rhode Island, with records from mid-July to mid-September and no indication of nesting (Conway 1979).

**Connecticut.** - There are nine sightings of gull-billed terns in Connecticut, mostly in late

summer, and no evidence of breeding (Zeranski and Baptist 1990).

**New York.** - The first New York specimen of the gull-billed tern collected was in 1882 (Bull 1985). The species was a rare visitor (Bull 1985) until 1975, when two pairs of gull-billed terns nested on Jones Beach along the southern barrier islands of Long Island (Buckley et al. 1975). Since then, there have been one or two nesting attempts each year in mixed colonies. New York represents the northern extent of the breeding range in the eastern U.S. (Figure 1). This species is ranked as S1 (critically imperiled) by the New York Natural Heritage Program (Table 1). Its

**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the gull-billed tern (*Sterna nilotica*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	-
New Hampshire	-	-
Vermont	-	-
Massachusetts	-	-
Rhode Island	-	-
Connecticut	-	-
New York	-	S1
Pennsylvania	-	-
New Jersey	-	S3
Delaware	-	-
Maryland	threatened	S1
Virginia	threatened <sup>b</sup>	S2
West Virginia	-	-

<sup>a</sup>S1 = critically imperiled; ≤ 5 occurrences

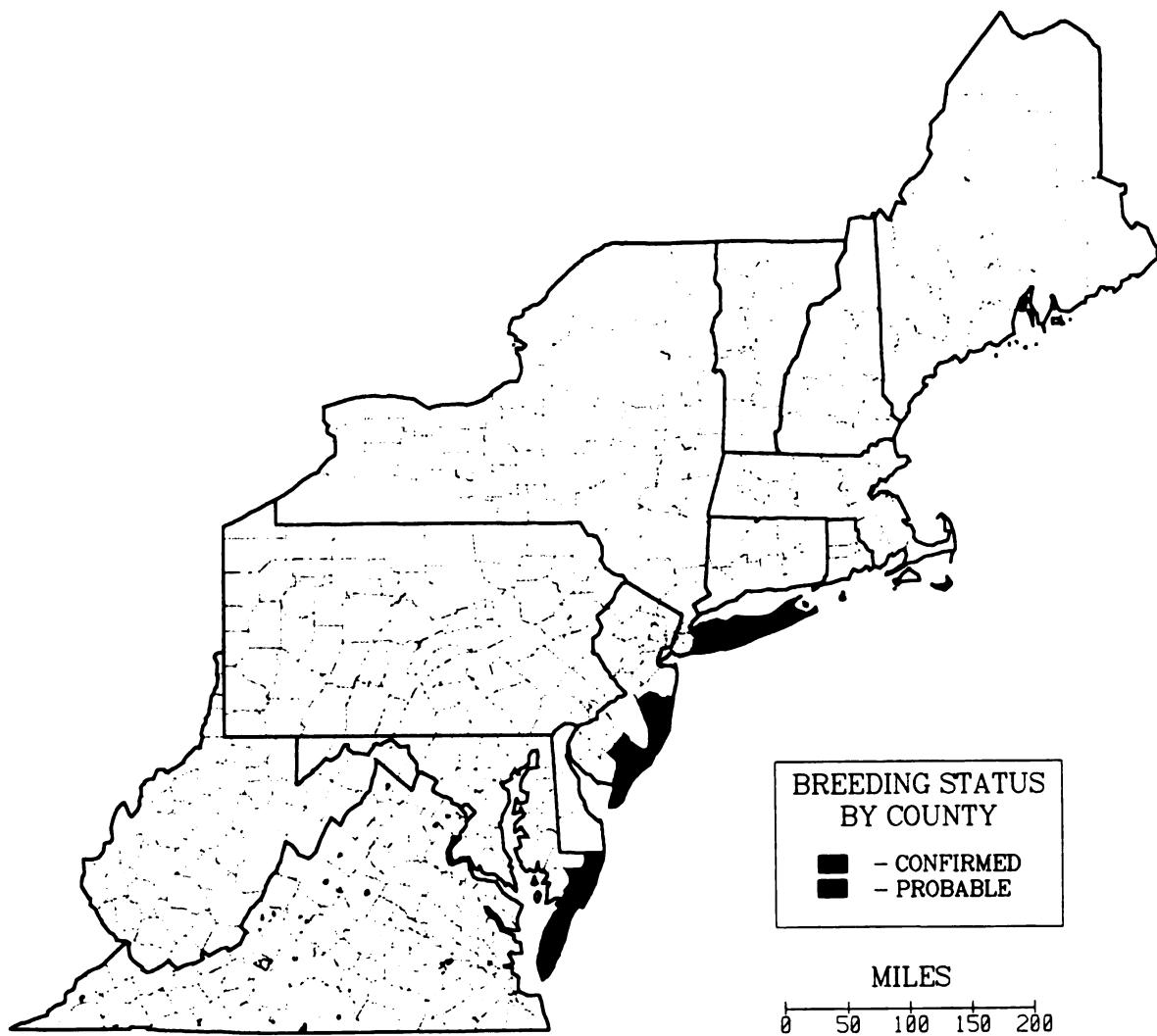
S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

<sup>b</sup>Recommended

breeding was confirmed from three blocks in Nassau and Suffolk counties, Long Island, during the New York State BBA (Andrle and Carroll 1988, Figure 2).

**Figure 2.** Northeast breeding distribution of the gull-billed tern (*Sterna nilotica*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



**Pennsylvania.** - The gull-billed tern has not been recorded from Pennsylvania (B. Haas pers. comm.).

**New Jersey.** - In New Jersey the number of gull-billed terns declined during the 19th century and by 1890 none nested in the state (Stone 1937). The next nest was found at Stone Harbor in 1926 (Stone 1937). Kane and Farrar (1977) reported 19 nesting pairs in 1977; Buckley (1979) reported 18-19 pairs nesting on dredge-spoil islands. Subsequently there have been 15-25 nests per year on saltmarsh islands (J. Burger pers. comm.) There are many records of non-breeding gull-billed terns congregating along coastal beaches or feeding over cranberry bogs (Leck 1984). The New Jersey Natural Heritage Program ranks this species as S3 (rare or uncommon) and data from the New Jersey BBA suggest that the gull-billed tern is a rare summer resident and breeder on the coastal marshes in southern New Jersey (D. Hughes pers. comm., Figure 2).

**Delaware.** - The gull-billed tern is a rare summer visitor in Delaware. Since 1958, 31 observations of the species have been verified in the state, but only one nesting record has been confirmed. Three clutches of eggs were collected at Rehoboth Bay in May, 1958 (Hess et al. in press).

**Maryland and the District of Columbia.** - This species was first recorded as breeding in Maryland in 1945 (Stewart and Robbins 1958). Gull-billed tern colonies are restricted to Worcester County in southeast Maryland (Stewart and Robbins 1958; Maryland Natural Heritage Program: *Vertebrate Characterization Abstracts*, Annapolis) but are unstable (Therres et al. 1978). Erwin (1979) noted that the species had almost disappeared as a breeder since the 1950s, from a peak of only about 35 pairs in 1951 to none in 1976. More recent data from 1985-91 (D. Brinker pers. comm.) indicate a population ranging from 33 in 1986 to none in 1988 (Figure 2).

The species is also a casual visitor to other tidewater areas of the state (Stewart and Robbins 1958). The State of Maryland lists the gull-billed tern as a *threatened* species and the Natural Heritage Program ranks it S1 (critically

imperiled), because of the small number of breeding pairs and the fragile nature of its breeding habitat.

**Virginia.** - Because of its southern location and its abundant barrier island beaches, Virginia has the largest population of gull-billed terns in the region. Numbers have fluctuated greatly over the past century. Austin (1932) showed that the Cobb Island population of gull-billed terns decreased from approximately 1,000 in 1901 to 16 in 1909 as a result of hunting for the millinery trade. Bailey (1913) estimated that there were probably no more than 12 pairs in the entire state by 1913.

With increased protection, the Cobb Island population recovered to 26 birds in 1932 (Austin 1932) and to approximately 100 in the 1950s (Erwin 1979). At barrier islands south of Wallops Island, the population plummeted from 2,228 to 214 during the period from 1975-90 (Williams et al. in press). Numbers at the largest colony, Metompkin Island, went from 500 in 1975 to no breeding birds in 1990 (Meanley 1981, Virginia Society of Ornithology 1985, Williams et al. in press).

During the same period, there has been almost a ten-fold increase in the numbers of herring (*Larus argentatus*) and great black-backed gulls (*L. marinus*), species which may usurp nesting sites or prey on gull-billed terns (Williams et al. in press). Nesting common terns have abandoned the barrier islands for a more protected, human-made island in the Chesapeake Bay (Williams et al. in press) and this may have left the remaining gull-billed terns at greater risk from predation. The State of Virginia has recommended the gull-billed tern for *threatened* status. It is ranked S2, indicating 6-20 breeding occurrences statewide.

**West Virginia.** - There are no breeding or vagrant records for the state (Hall 1983).

## Summary

Most colonies of gull-billed terns in the northeastern states are small and marginal outliers from the bulk of the population to the

south. It is only in Virginia that significant numbers of gull-billed terns nest and the decline of the species in this state may be the greatest cause for concern.

## LIMITING FACTORS AND THREATS

### Habitat Loss

Habitat loss, both through destruction of marshes and nesting sites, and indirectly through human activities that result in unacceptable levels of disturbance, has probably been the most important problem in the past for gull-billed terns and other waterbirds (Erwin 1980). However only limited data document this. The past effects of marsh ditching on gull-billed tern nesting and foraging are unknown, as are the historical population levels before market hunting and marsh drainage. Today, however, most northeastern marshes are either publicly owned or subject to laws that, at least in theory, protect them from further damage.

### Environmental Contamination

At the mouths of major estuaries, gull-billed terns may be exposed to a wide variety of water-borne contaminants, including crude oil and other petroleum products. However, analyses of gull-billed tern eggs from South Carolina indicated that levels of hydrocarbons were sufficient to induce eggshell thinning and reproductive failure, but that the effects of these compounds seemed to be negligible (Blus and Stafford 1980).

### Human Disturbance

Human disturbance may include a wide variety of human activities ranging from walking near nesting colonies to more severe forms of disturbance, such as vehicular traffic and slaughter of waterbirds (Parnell et al. 1988). Disturbance may result in desertion of nesting areas or exposure of eggs and chicks to extreme temperatures, rain, wind, and numerous predators, particularly gulls. Human disturbance has its most disastrous effects on gull-billed tern nesting

in the period immediately following hatching, because chicks may leave the nest prematurely and suffer excessive mortality (Sears 1978). Erwin (1980) compared barrier island nest sites in New Jersey with those in Virginia and found that the New Jersey sites were less frequently used by terns because of human disturbance.

While there are no guidelines for the minimum distance at which gull-billed terns will flush, Erwin (1989) has proposed a minimum distance of 200 m for posting colonies of black skimmers and common terns, species with which they often nest. However, because gull-billed terns are extremely sensitive to disturbance during the nesting period (Sears 1978), even these distances may be insufficient.

### Predation

Potential mammalian predators of gull-billed terns include raccoons (*Procyon lotor*), red foxes (*Vulpes fulva*), and rats (*Rattus spp.*). Suspected rat predation on the eggs and nests of gull-billed terns was reported in two studies (Sears 1978, Blus and Stafford 1980). Both pet and feral dogs may also pose a problem.

The primary avian predators of terns in the Northeast region include laughing, herring, and great black-backed gulls (Buckley and Buckley 1972). Blus and Stafford (1980) suggested that predation by laughing gulls caused nearly complete nesting failure from 1972-75 in South Carolina. Nocturnal avian predators of common terns that are also likely to prey on gull-billed terns include great horned owls (*Bubo virginianus*) and black-crowned night-herons (*Nycticorax nycticorax*) (Nisbet 1975, Morris and Wiggins 1986). Potential diurnal avian predators in addition to gulls include the fish crow (*Corvus ossifragus*) and northern harrier (*Circus cyaneus*).

## MANAGEMENT POTENTIAL

With the possible exception of the Virginia population, it is unlikely that gull-billed tern populations in the Northeast region will be large enough in the near or mid-term future to manage directly at the species-level. Fluctuations in gull-

billed populations in the Northeast are probably heavily influenced by changes in the main population in the Southeast Region.

Protecting colonies of the waterbird species with which gull-billed terns nest may be the most effective management strategy in the Northeast. Protecting wetlands where they feed is also important, although at present not enough is known about the foraging requirements of gull-billed terns to recommend such management.

In Virginia, several colonies have historically been large enough to merit management for gull-billed terns. Management might include manipulating vegetation and specifying the siting, shape, and contours of spoil islands for use as colony sites. Again, the effect of these efforts might be overshadowed by changes in the gull-billed tern population of North Carolina.

## LAND PROTECTION AND PRESERVE DESIGN

Møller (1981) suggested that successful gull-billed tern colonies need both a safe and acceptable nesting site and suitable nearby foraging areas. Gull-billed terns may be more vulnerable to human actions than are terns that feed at sea because of their dependence on terrestrial foraging sites. For example, if upland foraging areas around a protected wetland are being converted from agricultural use to housing, or if fields are succeeding to second-growth forests, there simply may not be enough foraging area to support a colony, no matter how suitable the colony site itself is. There may also be a fundamental difference in foraging suitability of saltmarshes of the "New England" and the "Coastal Plain" types (Nixon 1982) for gull-billed terns. The former, located from Connecticut to Maine along a subsiding coast, are usually limited in area, so the south shore of Long Island, the northernmost extension of the Coastal Plain marsh, may also mark the northern limit of breeding for the gull-billed tern.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Annual nesting surveys of gull-billed terns should be conducted within each state where they occur. Surveys should be conducted during mid to late incubation to avoid colony desertion during early incubation and mortality of young during late surveys. Surveys should be completed before mid-morning to prevent overheating of unintended eggs. Counts at hatching should never be done because they may cause massive loss of chicks.

Total counts of all tern species are best done by aerial survey, but accurate surveys for gull-billed terns are probably best conducted on foot because of the difficulty in distinguishing gull-billed terns from other terns when viewed from an aircraft (Erwin 1979). The best method for counting gull-billed terns is to flush adults and count them in the air where their broad wings and white general appearance allow separation from other terns. Total counts of gull-billed terns are not needed or practical in large colonies where subcounts are sufficient to determine the ratio of gull-billed to other terns. Surveys should also note presence of nesting gulls and other potential predators. Counts from single colonies or regions need to be interpreted in the context of total regional or national counts of gull-billed terns.

Surveys of foraging areas are at least as important as surveys of nesting sites. The land-use category of foraging areas and surrounding drainage areas, i.e. marsh, open-water, agricultural areas, and other land types should be noted and the relative use of each should be measured seasonally.

### Management Procedures and Programs

Colonies of nesting gull-billed terns should be posted with 200-m buffers following the recommendations of Erwin (1989), and the posting should be reinforced by appointment of "tern wardens" to patrol the colonies.

Areas where gulls are competing with gull-billed terns for nesting spaces or where gulls pose a potential predation problem should be

considered for some form of gull control or removal. Routine predator control should not be undertaken unless a problem is severe because such efforts may themselves generate considerable disturbance for other nesting waterbird species.

Cooperation with the U.S. Army Corps of Engineers and other dredging operations can lead to the siting and physical preparation of dredge islands to make them suitable for gull-billed terns and associated nesting species.

### Research Needs

Virtually all aspects of the life cycle and biology of gull-billed terns need further research, but particular attention should be paid to foraging requirements and population movements of this species. How far do the birds feed from colonies? In what habitats? What foods are taken from such habitats and do they affect nesting success?

Dispersal of gull-billed terns between colonies also needs to be measured, either through banding or biochemical genetic studies. These data would tell us which colonies are responsible for colonizing the northern states and whether local changes in populations are caused by changes in local reproductive success or immigration.

As a southern species intruding into the Northeast region, the gull-billed tern may be a sensitive indicator of climate change and global warming (cf. Duffy and Nettleship in press). Shifts in breeding range or population distribution might also be early warnings of shifts in marsh habitats. Similarly, because of their terrestrial and marsh foraging habits, gull-billed terns may respond to local differences in pesticide levels more closely than do other terns that feed at sea.

Finally, almost nothing is known of the ecology or distribution of this species during migration or on the wintering grounds. Changes in numbers on the breeding grounds may be caused by events thousands of miles away. These basic natural history questions remain to be addressed.

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# BLACK TERN

## *Chlidonias niger*

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Black terns breed in freshwater marshes across much of the northern U.S. and Canada. Sites comprised of 50% emergent vegetation and 50% shallow open water appear to be preferred as nesting areas. Nests are typically located in loose colonies, upon a floating base substrate. Eggs are often placed just above water level, and consequently are easily flooded during incubation, when most reproductive failure occurs. Water levels in impoundments should be stabilized from May through July to alleviate these losses. The species' poor reproductive success has undoubtedly contributed to population declines reaching 5.6% per year across its breeding range. Contaminants accumulated during migration and on the wintering grounds could also be contributing to population declines. Loss and disturbance of wetland nesting areas, including the invasion of exotic plants, has rendered some areas unsuitable for nesting. Colony surveys in New York, Vermont, and Maine should be conducted annually to count the number of breeding pairs and to monitor population trends. Occupied sites should be protected and enhanced for breeding black terns.

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### DESCRIPTION

#### Taxonomy

The black tern belongs to the Order Charadriiformes, Family Laridae (gulls and terns). Two subspecies of the black tern are recognized: *Chlidonias niger surinamensis*, which breeds in North America, and *C. n. niger*, the Eurasian subspecies. The black tern is the only member of the genus *Chlidonias* that breeds in North America. Most other North American terns belong to the genus *Sterna*. The white-winged

tern (*Chlidonias leucopterus*), which is closely related to the black tern, breeds in Europe and is a rare visitor along the east coast from New Brunswick to Georgia (Farrand 1983).

#### Morphology and Plumages

The adult black tern measures 23.0-26.5 cm. In the alternate (breeding) plumage, the head and body are black, fading to gray on the rump. The undertail coverts are white. The upper surface of the wings and tail are dark gray, and the wing linings pale gray. The leading margin of the wing

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from the body to the first digit is white. The bill is black and the feet are a dark reddish-purple (Goodwin 1960, Farrand 1983). Females are somewhat duller black than males, but this difference is often difficult to distinguish in the field (Goodwin 1960).

The prebasic (postbreeding) molt begins in late June when eggs begin to hatch. White feathers appear first around the eyes and cheeks, then on the forehead, the neck, the throat and breast, and finally on the abdomen. Heavily molting adults take on a peculiar, piebald appearance. The prebasic molt is completed during fall migration (Goodwin 1960).

In the basic (winter) plumage, the underparts are pure white except for a small, dark patch on each side of the breast. The back becomes a shade of gray similar to the wings and tail. A blackish cap joins black ear coverts on the otherwise white head (Goodwin 1960, Farrand 1983). The juvenal plumage is similar to the basic plumage, but the feathers of the back are darker and the wing coverts and cap are barred and scalloped brown (Goodwin 1960, Farrand 1983).

### Vocalizations

Vocalizations include shrill, somewhat metallic alarm notes, described as "kik" or "keek," depending upon intensity and level of motivation, and a complex of contact calls described as "kyew," followed by one to four additional syllables, as "kyew-dik, kyew-dik-ik, etc." (Goodwin 1960).

The "kik" call commonly serves as a signal of impending danger in the nesting area. It may also be given during the ascent portion of the courtship flight. The "keek" call is similar to, but more shrill and forceful than the "kik" call, and is given during aggressive attacks on enemies in close proximity to the nest. The frequency of repetition increases as the terns become more aggressive. The "kyew" calls are given as parents approach and leave the nest, during foraging flights, by adults accompanied in flight by young, by parents calling to young at or near the nest, by parents at the nest during incubation, brooding and feeding, and during the courtship flights (Goodwin 1960).

## HABITAT

### Breeding

Black terns nest in a variety of freshwater wetland types including marshes associated with lakes, ponds and rivers, prairie sloughs, and various types of water impoundments. Cattails (*Typha* spp.), rushes (*Scirpus* spp.), burreed (*Sparganium* spp.) and reed (*Phragmites* spp.) are commonly present in nesting areas (Bent 1921, Cuthbert 1954, Goodwin 1960, Bailey 1977, Firstencel 1987, Novak 1990). Although the species of vegetation may not be an important factor in habitat selection, an interspersion of emergent vegetation and open water appears to be critical. In Iowa, Weller and Spatcher (1965) found the greatest number of nests when the emergent vegetation to open-water ratio was approximately 50:50. Similarly, Tilghman (1979) found nests in areas where emergent vegetation covered 51-75% of the marsh, while nests in British Columbia were located in areas with 33% open water, 42% matted vegetation, and 25% standing vegetation (Mosher 1987). Areas of shallow water (< 2 m) are chosen for nesting, although deeper water may be used for foraging. Exposed perches, such as channel marker posts, floating logs, fallen trees, and old dock or fence posts are utilized as stations for feeding recently fledged young, for resting and for copulation (Cuthbert 1954, Novak 1990). In addition, marsh size and proximity to other wetlands may be important factors in habitat selection. In Iowa, black terns nested mainly in marshes of > 20 ha and were found in marshes in the 5-10.9 ha size-class only when these smaller marshes were part of a larger wetland complex (Brown and Dinsmore 1986).

### Wintering and Migration

Habitats where black terns have been observed during the nonbreeding season include marshy reservoirs in Mexico (Williams 1983), coastal areas of the Netherland Antilles (van Halewijn 1973), brackish swamps and estuaries of large rivers in Suriname (Spaans 1978), and coastal areas and a large lake in Panama

(Eisenmann 1951, Zaret and Paine 1973). Observations of black terns on spring and fall migration outnumber the observations of these terns on wintering areas. Only saltwater habitats were used at wintering locations in Suriname (Spaans 1978).

## BIOLOGY

### Reproductive Biology

**General.** - The black tern has been described as a semi-colonial nesting species (Cuthbert 1954, Bergman et al. 1970). Nests may be clumped closely in favorable habitat or more widely scattered in other, perhaps less favorable areas. As is typical of colonial nesting gulls and terns, black terns will join together to defend the nesting area from intruders (Cuthbert 1954). Black terns show a tendency to return to the same general area for nesting, although return rates may vary considerably among specific sites. Stern et al. (1985) found that 67% of recaptured terns nested within the same primary wetlands, while Bailey (1977) and Dunn (1979) reported return rates of 40% and 27% for sites in Wisconsin and Ontario, respectively. These return rates, which are low in comparison with other gulls and terns, may be the result of the relative instability of the habitat preferred by the black tern (McNicholl 1975).

**Arrival and Courtship.** - Most black terns in the northeastern U.S. and Canada return to breeding areas during the first two weeks of May, although birds may arrive in western New York as early as the last week of April (Laughlin and Kibbe 1985, Firstencel 1987, Gerson 1987). Conspicuous aerial courtship displays characterize the courtship period, which begins soon after arrival at the breeding site. In the "high-flight," a group of 2-20 terns ascend together to a great height then split into smaller groups of 2 or 3 and descend in rapid glides (Baggerman et al. 1956). During the "fish-flight," a male tern carries a small fish or large insect in its bill and is closely followed by a female as the two fly about the marsh. At the close of this aerial display the male

follows the female to a perch and feeds her (Baggerman et al. 1956).

**Nesting Period.** - In the northeastern U.S. egg-laying begins in late May, but may be initiated as late as the middle of July. Nests with eggs were observed at one site in western New York from 24 May to 12 July (Firstencel 1987). During a 1989 survey of black tern colony sites throughout New York, nests with eggs were observed as early as 25 May and as late as 18 July (Novak 1990). The black tern is not known to be double-brooded and late nests probably represent renesting attempts. At Rush Lake in Wisconsin, Bailey (1977) observed a trend toward three nesting peaks, one in late May, one in early to mid-June, and one in late July. This pattern was attributed to two initial nesting periods characterized by a high degree of synchrony, followed by a period of renesting (Bailey 1977). Baggerman et al. (1956) also reported highly synchronous nesting activity.

**Nest and Nest Site.** - The nest of the black tern consists of a small gathering of aquatic vegetation with a simple, cuplike bowl (Weller and Spatcher 1965, Bailey 1977). Although the first egg may be laid before the nest takes shape, vegetation gathered at the nest site is added throughout the incubation period (Baggerman et al. 1956, Goodwin 1960, Bailey 1977).

Black tern nests are built upon an underlying base substrate which is often floating. Commonly used nest substrates include floating cattail rootstocks, floating piles of emergent vegetation, inactive muskrat (*Ondatra zibethicus*) lodges, muskrat feeding platforms, and floating logs or boards (Cuthbert 1954, Bergman et al. 1970, Bailey 1977, Dunn 1979, Novak 1990). Floating mats of muck or algae, mud flats, mud mounds and islands, and old nests of other marsh-nesting birds also have been used as nest substrates (Cuthbert 1954, Bailey 1977, Dunn 1979, Connell and Norman 1989, Novak 1990).

The height of eggs above water has often been measured in studies of nesting black terns. Although the height may vary based upon the substrate chosen for nesting, the eggs are rarely located more than a few cms above the water level. The range reported in the literature varies from an average of 2.3 cm for 23 nests on dead,

floating vegetation in Iowa (Bergman et al. 1970), to 20.0 cm for 2 nests located on mud islands in New York (Firstencel 1987). The latter is clearly the exception as 8.6 cm for seven nests on old muskrat houses is the next highest figure reported (Weller and Spatcher 1965).

Black tern nests may be placed in a variety of vegetative situations, from dense stands of emergent vegetation to open water (Bergman et al. 1970, Novak 1990), but moderate or sparse vegetation appears to be preferred (Cuthbert 1954, Weller and Spatcher 1965, Dunn 1979). Nests are typically located in shallow water, close to open water or openings in stands of emergent vegetation. The range of water depths reported varies from a "few inches" (Bent 1921) to 1-2 m (Dunn 1979). Bailey (1977) found that nests were never more than 1-2 m from open water. Dunn (1979) and Cuthbert (1954) reported the average distance to open water as 4 m and 4.6-6.1 m, respectively. One site in New York where nine nests averaged 25.3 m from open water (Novak 1990) appears to be the exception, although Firstencel (1987) also reported nests located "deep within the cattails."

**Eggs and Clutch Size.** - The shape of a black tern egg is ovate with a tendency toward ovate pyriform (Bent 1921). The base color varies from dark-olive to light-buff with markings of dark-brown and gray. The markings vary from small dots and scrawls to very large blotches and are often particularly heavy around the larger end of the egg (Goodwin 1960). The average dimensions for 122 eggs in the U.S. National Museum were 34 x 24 mm (Bent 1921).

One to five eggs may be laid, although the normal clutch is usually two or three (Bent 1921). Clutches with 4 eggs have been reported in only 2 recent studies (Bergman et al. 1970, Mossman 1980), as well as from 3 out of 363 nests in New York in 1990-91 (R. Miller pers. comm.), and are apparently quite rare. Other than Bent (1921), there are no published reports of five-egg clutches. Single-egg clutches may often be replacement nests or nests where one or more eggs have already been lost (Bent 1921, Cuthbert 1954, Firstencel 1987). In a recent study in Wisconsin, the average clutch size for 41 closely monitored nests was 2.9 (Bailey 1977). Four nests

had clutches of two, but no nests contained less than two or more than three eggs. Average clutch sizes reported in other recent studies where nests may not have been monitored as carefully range from 2.25-2.75 (Cuthbert 1954, Goodwin 1960, Bergman et al. 1970, Mossman 1981, Firstencel 1987, Novak 1990, R. Miller pers. comm.).

**Incubation.** - Incubation begins with the laying of the first egg, and eggs require 20-23 days to hatch (Goodwin 1960, Bergman et al. 1970, Bailey 1977). Both sexes incubate (Goodwin 1960).

**Chicks.** - Black tern chicks are able to swim, walk and run by the time they are two days old (Goodwin 1960). The chicks grow rapidly, doubling their weight in less than three days and quadrupling their weight in less than six days (Bailey 1977). The rate of weight gain slows after the eighth day. In some cases, black tern chicks may be relocated from the nest site to "auxillary" nests within a few days after hatching (Cuthbert 1954, Firstencel 1987). If disturbance at the nest is minimal, the young may remain at the original nest site for as long as 14-25 days, although they hide in the vegetation at the sign of danger and may be found swimming as far as 40 ft (36.6 m) from the nest (Cuthbert 1954, Goodwin 1960).

The age at fledging is difficult to determine. Bailey (1977) reported fledging at 18 and 19 days for two chicks of known age and suggested that the majority of chicks are flying at 21 days of age with a mean fledging age possibly less than 20 days. Baggerman et al. (1956) and Goodwin (1960) reported fledging at 21 days.

**Nest Success.** - Estimates of nest success from four nest studies are as follows (expressed as a percentage of nests where at least one egg was hatched successfully): 27% (15 of 55 nests) in Ontario, 29% (56 of 192 nests) in Iowa, 34% (13 of 38 nests) in Wisconsin, and 50% (12 of 24 nests) in New York (Dunn 1979, Bergman et al. 1970, Bailey 1977, Firstencel 1987, respectively). There was no obvious correlation between nest success, height of eggs above water, and number of nests per substrate in the Iowa study (Bergman et al. 1970).

Survival of young to fledging is difficult to measure because of the mobility of chicks. Bailey (1977) attempted to measure chick survival by placing fencing around nests to prevent young from moving away from the nest site. Just 3 of 26 (12%) chicks monitored fledged successfully; 16 chicks were lost to predation and several chicks died in the pen netting. Fledging success at unfenced nests (perhaps a better representation of fledging success) was estimated at 15-20% (Bailey 1977). Recent surveys have presented estimates of reproductive success based on the number of fledglings produced per egg laid in the colony (Rabenold 1987, Novak 1990). Estimates for 3 small (< 10 pairs) colonies in Indiana were 0%, 53%, and 67%, for an overall average of 30% (Rabenold 1987). Estimates also varied widely for 19 sites in New York, from 4-38%, with an overall average of 20% (Novak 1990). In 1990-91, average fledging success was estimated at 17% in New York colonies (R. Miller pers. comm.). Mossman (1980) reported a 25% reproductive success based on the ratio of young to adults observed at one study area in Wisconsin.

The similarity between reproductive or fledging success rates and nest success (hatching success) supports the observation by Dunn (1979) that most losses occur during the egg stage. Wind and wave action and storms were responsible for most nest losses in several studies (Bergman et al. 1970, Bailey 1977, Dunn 1979, Faber and Nosek 1985, Mosher 1987). Nest losses have also been attributed to egg inviability, predation, muskrat activity, and intraspecific interactions (Bergman et al. 1970, Bailey 1977, Dunn 1979, Firstencel 1987).

### Food Habits

On the breeding grounds the black tern is primarily insectivorous, although small crustaceans, spiders and small fish are also regular food items (McAtee and Beal 1912, Bent 1921). The diet may vary depending on habitat and food availability. Fish may be an especially important food item at some sites in the Northeast. In a sample of 376 feedings of young in different nests at North Pond in New York, Goodwin (1960) found that 41% of the items brought by parents were minnows and 59% were insects, including

45% damselflies. Insects comprised 93.6% of 602 feedings to chicks in Michigan while fish accounted for just 4.9% (Cuthbert 1954). Although many of the insects could not be identified, damselflies, dragonflies, and mayflies were important food items. In Ontario, Dunn (1979) was unable to identify the majority of 56 food items brought to young, but 13% were minnows and 6% were dragonflies.

In wetlands, food is captured in the air, at or just below the water surface, and from the surface of emergent vegetation (Goodwin 1960). In the prairies, much of the food is obtained from plowed land and fields of grain (Pittman 1927). Foraging over agricultural land near marshes has also been observed in New York (M. Morrison pers. comm.).

### Migration

In the fall, black terns migrate to the Atlantic and Pacific coasts, then south to wintering grounds from Panama south to Peru and Suriname (American Ornithologists' Union 1983, Carroll 1988). The black tern is strictly a migrant in Mexico where the fall migration extends from July through early November. During the southbound migration, birds pass through the interior highlands as well as along both coasts. The spring migration through Mexico is shorter in duration (30 March-2 June), with greatest numbers in April and May, and is almost entirely coastal or offshore (Williams 1983).

### Winter Biology

There are few published reports regarding the black tern on its wintering grounds. Immature black terns that spend their first winter and summer at Gatun Lake in Panama feed primarily on a small fish called the silverside (*Melaniris chagresi*). Black terns in Suriname used only saltwater habitats (Spaans 1978).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

In North America, the breeding range of the black tern extends from central British Columbia, eastward across the prairie provinces to central Ontario and southern Quebec, and south to central California, northern Utah, Wyoming, Kansas, Iowa, Illinois, Indiana, and Ohio, central and northern New York, and northern New England (American Ornithologists' Union 1983, Figure 1). The species breeds locally in New Brunswick and Nova Scotia. There is a single breeding record for coastal New Jersey (D. Hughes pers. comm.).

The black tern winters along both coasts from Panama to South America.

Gerson (1987) surveyed provincial and state biologists regarding the current status of the black tern in various regions of North America and arrived at the following conclusions: the species (1) is widespread and common throughout much of Canada, (2) has a limited distribution in the northwestern U.S., but may be locally common, (3) is widespread and common in the northcentral U.S. (including North and South Dakota), and (4) becomes local and rare in the northeastern U.S.

In contrast to this subjective, and generally positive assessment of the status of the black tern, quantitative measures indicate the species has been experiencing a major population decline, virtually rangewide, for at least the past 20 years.

The U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes provide a quantitative measure of black tern population trends from 1966-89. During this period the breeding population in North America declined at an annual rate of 5.6% per year, for an overall population decline of 71.8%. The decline has been greater in the U.S. (8.2% per year, 84.8% overall) than in Canada (4.8% per year, 66.1% overall). Of 12 states and provinces with sufficient sample size to determine population trends, only British Columbia showed an increasing trend for black terns. States and provinces with sufficient sample size and

statistically significant population declines include Alberta, Iowa, Michigan, Minnesota, North Dakota, and Ontario. New York is the only state in the Northeast where black terns have been recorded on BBS routes, but sample size is not sufficient to determine a population trend for this state (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).

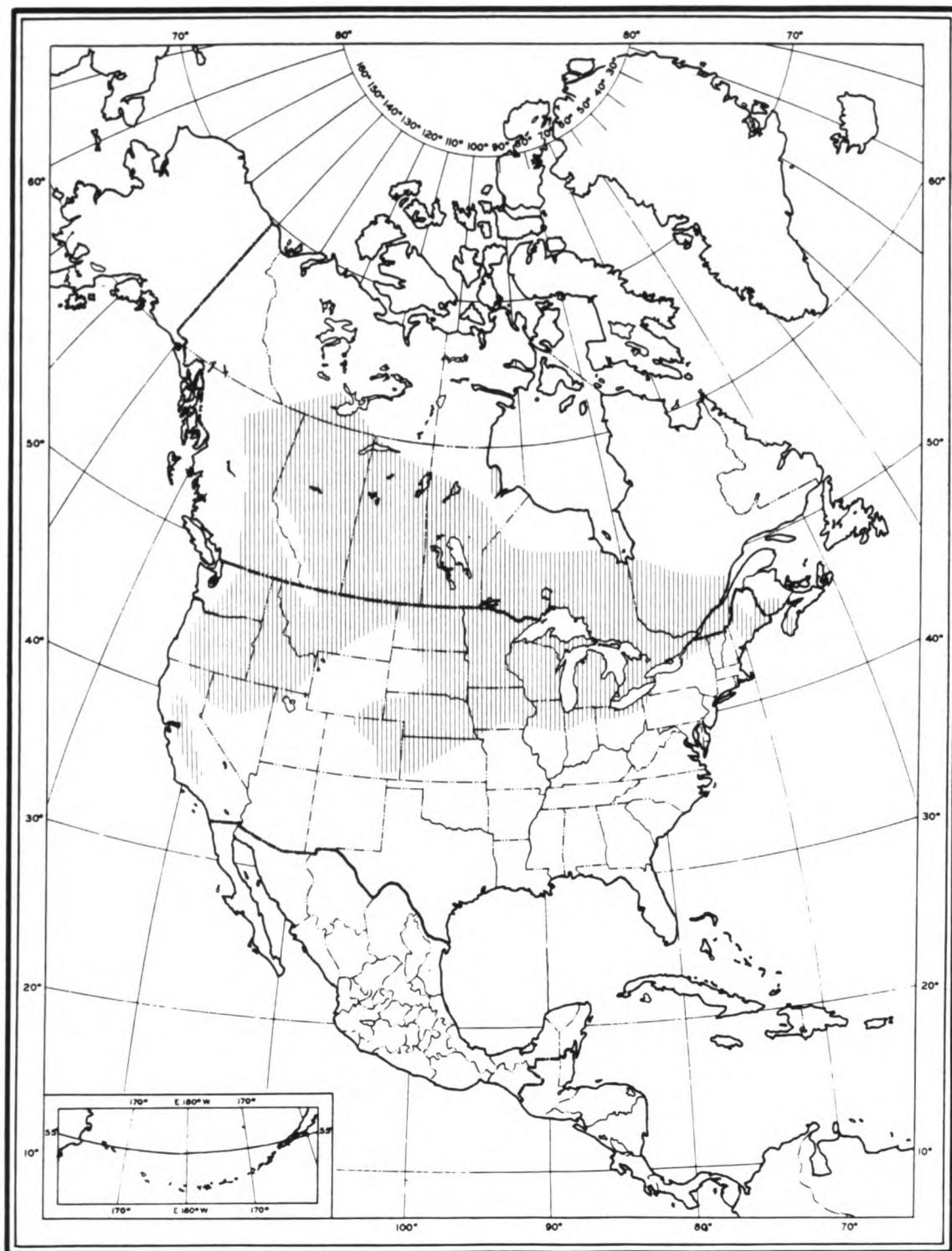
The black tern is considered a *migratory nongame bird of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987), and has most recently been proposed as a *candidate (category 2)* for listing as a federally endangered or threatened species (U.S. Fish and Wildlife Service 1991). The species also has special status as an endangered or threatened species in many of the states within its breeding range. In the northcentral states, Illinois, Indiana, and Ohio list the black tern as an endangered species while Iowa lists it as special concern. In the Northeast, the black tern is listed as *endangered* in Pennsylvania, *special concern* in New York, and is on a *watch list* in Maine (Table 1). New York State is in the process of revising its state lists and it is likely that the status of the black tern will be changed to *endangered* or *threatened*. The black tern was recently proposed for *threatened* listing in Canada (Gerson 1987).

Natural Heritage Programs, typically a joint effort of the Nature Conservancy and an agency of state government, assign state ranks based on the number of known occurrences. They range from *S1* (typically  $\leq 5$  occurrences) to *S5* (demonstrably secure). Unlike the legal status assigned by a state agency, these *S* ranks carry no added legal protection. However, Heritage Programs do use standard criteria among the states. The Heritage Program state ranks for the Northeast also appear in Table 1.

### Distribution and Status in the Northeast Region

The current breeding range of the black tern in the northeastern U.S., based upon state breeding bird atlas (BBA) data, includes Maine, Vermont, New York and Pennsylvania (Figure 2). A single nesting record, albeit unsuccessful, was

Figure 1. The North American range of the black tern (*Chlidonias niger*) (American Ornithologists' Union 1983).



Breeding

**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the black tern (*Chlidonias niger*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	watch list	S2
New Hampshire	-	SN
Vermont	-	S2
Massachusetts	-	SN
Rhode Island	-	SN
Connecticut	-	SN
New York	special concern	S2
Pennsylvania	endangered	S1
New Jersey	-	SN
Delaware	-	SN
Maryland	-	SN
Virginia	-	SN
West Virginia	-	SN

<sup>a</sup>SN = regularly occurring but not breeding

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

documented at Cape May Point in New Jersey during 1984 but no subsequent breeding attempts have been observed at this site or elsewhere in the state (D. Hughes pers. comm.). In all other states within the region the black tern is present on migration but is not known to breed.

**Maine.** - The black tern was first documented as nesting in Maine in 1946 when a 4-6 day-old chick was captured in a marsh at the south end of Lake Messalonskee in Belgrade (Palmer 1949). Nesting was confirmed at seven additional sites between 1959 and 1983 (Pierson 1983). These colony sites were included along with three other marshes in a survey of wetland birds conducted in 1989-90 (Gibbs and Melvin 1990) and 1991 (Maine Department of Inland Fisheries and Wildlife unpubl. data). During these surveys two additional colony sites were located, yet no black terns were observed at three sites where terns had previously nested. Vacant sites included:

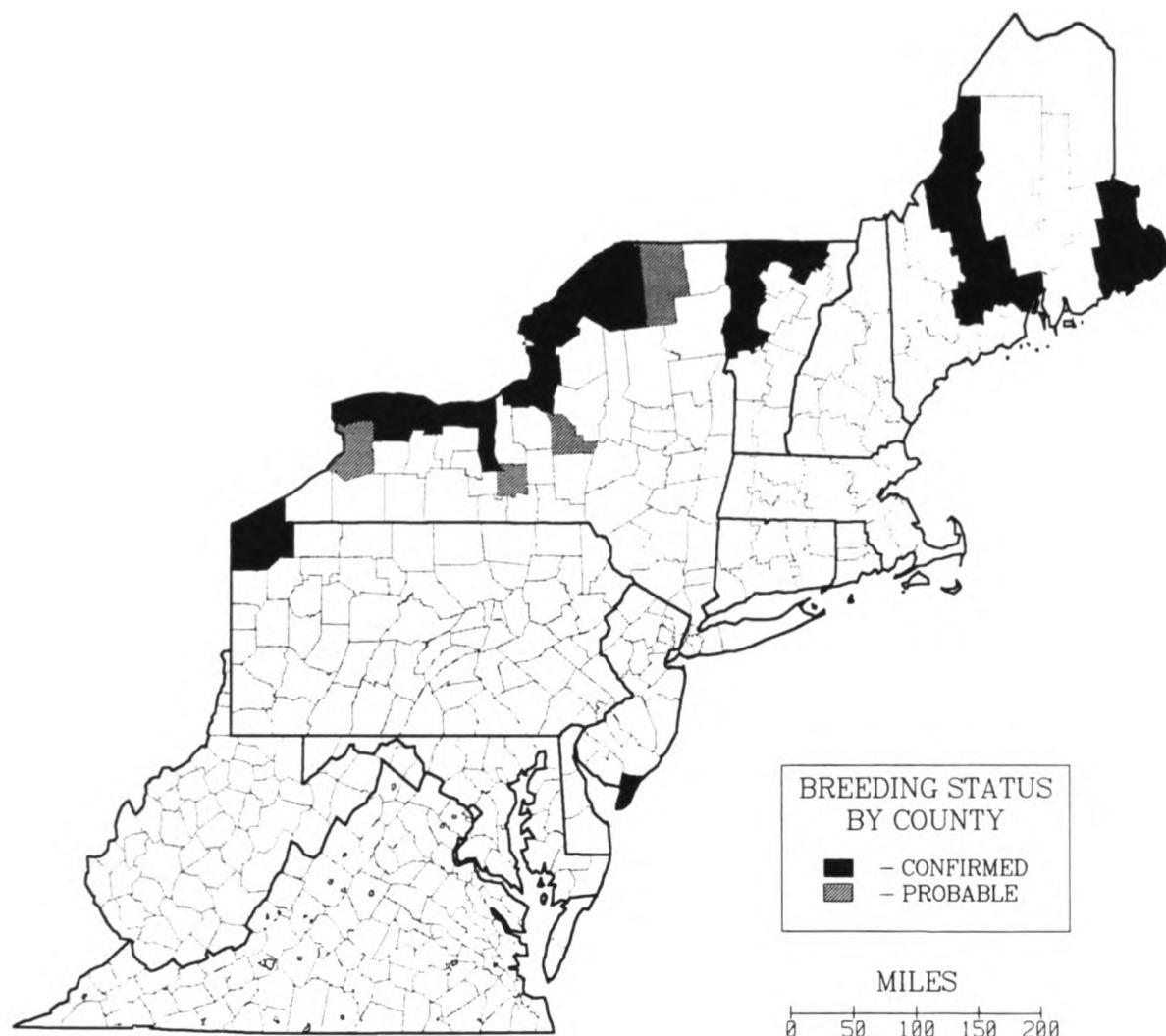
- **Portage Lake, Aroostook County.** - Nesting was observed at this site in 1959, and one to two pairs possibly nested into the 1960s.
- **Corundel Marsh, Penobscot County.** - Seven adults were observed feeding recently fledged young in 1968; no previous or subsequent breeding records are recorded.
- **Upper Mud Lake (East Machias River), Washington County.** - An estimated seven pairs nested successfully from 1973-75.
- **Grand Falls (Huntley Brook) Flowage, Washington County.** - Five to six pairs nested from 1970-82 (Pierson 1983); three pairs nested in 1991.

Sites where black tern nesting was confirmed in 1991 include:

- **Lake Messalonskee (Belgrade Bog), Kennebec County.** - Twenty to 25 adults were observed (Gibbs and Melvin 1990). At least six pairs nested in 1991. This site was first occupied in 1946, with 15-17 pairs each year since and a maximum of 22 adults observed in 1982 (Pierson 1983).
- **Plymouth Pond, Penobscot County.** - Eleven adults were found in 1989 and 1990 (Gibbs and Melvin 1990). Approximately eight pairs nested in 1991.
- **Carlton Pond (Carlton Bog), Waldo County.** - Twelve adults were recorded in 1989 (Gibbs and Melvin 1990) and four pairs nested in 1991. This site was first occupied in 1980 and has supported five to six pairs each year since (Pierson 1983).
- **Mainstream Pond, Somerset County.** - Fourteen adults observed in 1989 (Gibbs and Melvin 1990) and three pairs nested in 1991.
- **Douglas Pond, Somerset County.** - Twenty to 30 adults in 1989 and 1990 (Gibbs and Melvin 1990), but only 3-4 pairs nested in 1991. This site was first used in 1977 and, together with Madawaska Pond, less than one-half mile away, has supported 12-15 pairs each year since the mid-1970s (Pierson 1983). No birds nested in Madawaska in 1991.

In addition to these sites with confirmed nesting, individual birds were observed in June at

Figure 2. Northeast breeding distribution of the black tern (*Chlidonias niger*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



two other sites in Penobscot County. Individual birds are also regularly sited at coastal common tern (*Sterna hirundo*) and arctic tern (*S. paradisaea*) colonies at Petit Manan and Jenny islands and Eastern Egg Rock.

The 1989-91 surveys covered all historically occupied colony sites in Maine. Surveys to be conducted as part of an ongoing wetland bird project may turn up additional colony sites. At this time, the black tern population in Maine appears to include approximately 25-30 pairs at 7 sites. One site in south-central Maine (Lake Messalonskee) has been occupied since 1946 and seems to be stable or declining. A small colony (Huntley Brook Flowage) still persists in eastern Maine, but has declined since the mid-1970s. A colony in northern Maine (Portage Lake) no longer exists. The remaining colonies are clustered in central Maine and seem to be stable or declining. Annual fluctuations of numbers at individual sites most often reflect movements between wetlands associated with renesting, changes in habitat, or sometimes colonization of new areas. Overall, the black tern population in Maine may be described as small and localized, and stable or possibly declining.

**New Hampshire.** - There are no breeding records for black terns in New Hampshire. The best nesting habitat in the state occurs at Lake Umbagog where the black tern has been noted since the turn of the century (C. Foss pers. comm.). Three black terns were observed in this area during June 1987 but did not breed. Other breeding season observations for the state include: Cherry Pond (6 July 1978), Lake Wantastiquet (20 May 1959), Lake Winnipesaukee (10 June 1936), and Lake Umbagog (29-30 June 1936) (Hentcy 1990).

**Vermont.** - The first documented breeding record for the black tern in Vermont is from Lake Champlain in 1937 (Laughlin and Kibbe 1985). A large colony was noted at the north end of Lake Champlain in 1950 and the first report from the south end of the lake, at Dead Creek WMA, is from 1965. The species was first reported from Lake Memphremagog in 1949 and was breeding there in 1963 (Laughlin and Kibbe 1985). The black tern probably expanded its breeding range

into New England and the maritime provinces after the late 1930s although it has been suggested that more intensive field observations may have led to its discovery at this time (Palmer 1949, Laughlin and Kibbe 1985).

Regardless of which situation is true, the breeding range of the black tern in Vermont, as determined by field work for the Vermont BBA, included the northcentral border at Lake Memphremagog, and six or more locations in the marshes of Lake Champlain (Laughlin and Kibbe 1985). The breeding population at the time was estimated at 180-300 pairs, with more than half the pairs at Missisquoi National Wildlife Refuge (NWR) (Grand Isle County) at the north end of Lake Champlain (Laughlin and Kibbe 1985). Other breeding sites included Lake Memphremagog (Orleans County), Dead Creek WMA (Addison County), Mud Creek WMA, (Chittenden County), Little Otter Creek (Addison County), and St. Albans Bay (Franklin County).

Recent surveys of these sites indicate a significant decline in the total population to 59 pairs in 1990 and 71 pairs in 1991 (Shambaugh 1991). Over half the number were at Missisquoi NWR, with the other pairs at Lake Memphremagog, Dead Creek and Mud Creek WMAs, and Little Otter Creek (Kibbe 1989, Shambaugh 1991).

**Massachusetts.** - There are no breeding records for the black tern in Massachusetts where this tern is a rare spring and regular fall migrant (B. Blodget pers. comm.).

**Rhode Island.** - There are no breeding records for the black tern in Rhode Island. This tern is generally recorded each year during fall migration and every other year during spring migration, with single individuals usually reported (R. Enser pers. comm.).

**Connecticut.** - There are no breeding records for the black tern in Connecticut. This tern is a rare migrant along the coast from late July to late September and less numerous in the spring from mid-May to early June. Inland, the black tern is reported as a very rare spring migrant over marshes, lakes, and rivers. Historical information suggests that the black tern was rare before 1920

with an increase in the number of sightings through the 1940s and 1950s, probably coinciding with the extension of the breeding range eastward into Vermont, Maine, and New Brunswick (Bevier et al. in press). There are no recent summer sightings to suggest that the black tern can be expected to nest in Connecticut in the future.

**New York.** - The black tern has nested in New York State since the turn of the century (Carroll 1988). Black terns were recorded in 73 blocks during field work for the New York State BBA (1980-85). Confirmed breeding was documented in 27 blocks while evidence of probable breeding was observed in 17 blocks (Andrie and Carroll 1988). Carroll (1988) reviewed atlas and historical records and identified 56 historical colony locations in New York. The historical range includes marshes along the shores of Lake Ontario from Buffalo to Watertown, marshes along the St. Lawrence River, inland marshes in western, central and northwestern New York, an isolated site in the Adirondack Mountains, and a site near Lake Champlain (Bull 1974).

During an extensive field survey of historical and potential colony sites conducted in 1989, black terns were observed at 37 of 95 sites visited. Evidence of confirmed breeding (nests with eggs or young, adults feeding young, or adults with fledged young) was obtained at 22 sites, with evidence of probable breeding (aerial courtship displays, courtship feeding, copulation, or nest building) observed at 6 additional sites. Overall, the breeding population was estimated at approximately 235 pairs distributed among 37 locations in 8 counties (Novak 1990). Approximately 25% of the population nested at a single site near Cape Vincent in Jefferson County, with close to another 25% at just two other sites (Perch River WMA and Lakeview WMA), also in Jefferson County (Novak 1990). All but a few other sites had less than 10 pairs.

Surveys in 1990-91 confirmed continued low breeding numbers. Only 23 of 69 sites visited in 1990, and 25 of 64 sites visited in 1991 had black terns present (R. Miller pers. comm.). Of these, 15 sites in 1990 and 17 sites in 1991 were used by breeders. The total number of breeding pairs estimated for New York was 215 in 1990 and 284 in 1991.

At Montezuma NWR, where as many as 1,000 pairs of black terns nested in the late 1950s, less than 10 birds were present from 1989-91, and no breeding attempts were observed (Novak 1990, R. Miller pers. comm.). Numbers are substantially reduced at several other colony sites for which population data are available, and some historical colony sites are no longer occupied. The range of the black tern in New York has become more restricted as the population declines. The species is no longer known to breed in central New York in the vicinity of Oneida Lake, in the Adirondacks, or near Buffalo.

**Pennsylvania.** - During migration the black tern may be found throughout Pennsylvania, but the breeding distribution is restricted to Erie and Crawford counties in the northwestern part of the state (Pennsylvania Natural Diversity Inventory: *Vertebrate Characterization Abstract*, Middletown).

In recent years the black tern has nested in at least four sites, with birds observed, but not confirmed as breeding, in several others (Brauning in press). The birds have not been consistent in their nesting, i.e., they have not been present at each site every year (C. Bier pers. comm.). The lone Erie County site, at Presque Isle on Lake Erie, has been used historically and was reestablished in 1987 when a single nest was found (C. Bier pers. comm.). Three distinct sites in Crawford County have been used in the last several years. At one site near Hartstown, six adults were observed and three nests were found in 1988. Up to three birds were observed in Smith's Marsh at Meadville Junction during 1988. Some defense of territory was noted, but no nests were found. The Tern Marsh area at Conneaut Lake has been used by nesting black terns since 1910. Six to 10 pairs were present at this site in 1982 and 1983, and 4-5 pairs in 1985. No black terns were observed at this site in 1989 (C. Bier pers. comm.). Although the black tern has always had a restricted breeding range in Pennsylvania, it is evident that there has been a general decline in numbers in recent years (Genoways and Brenner 1985), accompanied by a lack of consistency in returning to historically occupied colony sites.

**New Jersey.** - Records of black terns in full breeding plumage displaying agitated parental

behaviors were noted at coastal sites in early and mid-July 1921, 1929, and 1932 (Stone 1937). Black terns in breeding plumage were noted at three different locations in early June 1981 (D. Hughes pers. comm.). A record of a confirmed nesting attempt for the black tern in New Jersey was finally obtained in 1984 when an adult was observed sitting on a nest, brooding two eggs in a marsh at Cape May Point, Cape May County, on 10 June. The bird was photographed on 12 June but the nest was deserted on 17 June. Vegetation in the vicinity of the nest was primarily cattail and the nest was composed of a floating mass of vegetation typical of this tern (D. Hughes pers. comm.).

Despite the occasional summer occurrence of adult black terns in New Jersey, the 1984 record remains the only case of confirmed breeding for the state. There is no evidence that a regular breeding site has been established.

Although close monitoring of future summering individuals may produce additional nesting records, the black tern is still considered a transient in New Jersey, rare in the spring and uncommon in the fall. During spring and fall migration this tern occurs both at inland lakes and marshes, and along the coast. The numbers reported during migration appear to be stable (New Jersey Natural Heritage Program, *Vertebrate Characterization Abstract*, Trenton).

**Delaware.** - There are no documented breeding records for the black tern in Delaware, although it is listed as possibly an accidental breeder. Breeding season records of small numbers of black terns were reported from several counties throughout the 1960s and 1970s, but there have been no summer records in the 1980s (Hess et al. in press).

**Maryland and the District of Columbia.** - Although the black tern is reported as having nested in Garrett County in the 1930s (Hall 1983), this report is unsubstantiated and is believed to be an inaccurate reporting of a rare summer record where breeding did not occur (R. Bartgis pers. comm.). No additional summer records are known and the black tern must be regarded as an uncommon migrant in Maryland

with no confirmed breeding records (R. Bartgis pers. comm.).

**Virginia.** - The black tern does not breed in Virginia. This tern is an uncommon to rare spring transient and a common fall transient along the coast. The black tern is a rare summer visitor at Chincoteague NWR but no nesting attempts have been observed (S. Ridd pers. comm.).

**West Virginia.** - The black tern does not breed in West Virginia. There is a single summer record of one individual from Morgan County, 11 June 1957 (Hall 1983). This tern is an uncommon migrant in West Virginia with numbers apparently reduced in recent years (Hall 1983).

## LIMITING FACTORS AND THREATS

### Loss Of Habitat

Loss of breeding habitat has undoubtedly been a major contributing factor in the decline of the black tern throughout its range. Since European settlement, 54% of all wetlands in the U.S. have been lost (Tiner 1984). The loss of 4.75 million acres (1.92 million ha) of palustrine emergent wetlands during the mid-1950s to mid-1970s has probably had an especially significant effect on black tern populations. Similar losses of wetlands have been documented in Canada. A decline of 9-61% of the original wetland area has been documented for portions of some prairie provinces, while 70% of southern Ontario's original wetlands have been converted to other uses, particularly agriculture (Gerson 1987). Along the St. Lawrence River, 42% of the wetlands from Cornwall, Ontario to Matane, Quebec were lost between 1945-75 (Gerson 1987). Wetlands in the U.S. and Canada continue to be threatened despite efforts to curb their loss.

In addition to the outright loss of wetlands, successional change, management practices, and degradation of water quality have altered the character of some wetlands, thus rendering them less suitable to black terns for breeding habitat. Loss of high quality habitat through successional

changes was noted at several historical colony locations during a recent survey in New York (Novak 1990). The quality of some remaining wetlands may be reduced because of pollution and runoff associated with increased development in the vicinity of wetlands (Gerson 1987). Wetlands in Central and South America where black terns winter are also threatened by land-use changes and degradation of water quality (Gerson 1987). In Panama, the introduction of the peacock bass (*Cichla ocellaris*), has resulted in the decline of an important native food fish (Zaret and Paine 1973).

Purple loosestrife (*Lythrum salicaria*), an invasive, wetland exotic which outcompetes native emergent species, has drastically altered the character of wetlands in parts of the northeastern U.S. The changes wrought by this plant, along with changes brought about in the attempt to control it have altered the character of some wetlands. Purple loosestrife may have been a factor in the decline of the black tern at Montezuma NWR in New York where high quality nesting habitat is currently lacking (Novak 1990).

#### Weather

Black tern nests are typically a floating structure located close to open water with the eggs placed just a few centimeters above water level. This arrangement predisposes the species to some natural sources of nest failure. Both Bailey (1977) and Rabenold (1986) attributed major nest failures to the flooding of nests caused by elevated water levels resulting from heavy rains. Goodwin (1960) noted that some nests were so flimsy that storms caused them to submerge. Wind and wave action has also been noted as a major cause of nest failure (Cuthbert 1954, Bergman et al. 1970). Relatively low nesting and fledgling success may be typical of black terns in regions where spring and summer storms occur regularly.

#### Human Disturbance

The impact of various types of human disturbance on nesting black terns is not well

documented. Waves caused by boats may represent a major source of egg and chick mortality at many sites in New York and other parts of the species' range. Boat wakes were observed swamping logs both in the open and within moderately dense cattails at one site in New York, and fledgling success rates were comparatively low at several sites where boat traffic was heavy (Novak 1990). However, a study of boat traffic at four New York colonies in 1990 indicated that traffic volume, as well as size and speed of the boat, did not have a visible behavioral influence on nesting black terns (R. Miller pers. comm.). At one site within 50 m of a marina, terns reacted only when simultaneously flying over the channel with a moving boat. During 66 hours of observation, boat wakes were not observed to flood any tern nests, primarily because the nests were protected by vegetation or shallow water. Small boats, such as canoes, moving directly through colonies present the greatest threat to nesting terns. Although black terns sit tightly when incubating or brooding, they will leave the nest at the close approach of people. Repeated or prolonged disturbance may increase the likelihood of abandonment, predation or exposure to inclement weather. Field training of dogs is reported to be a problem at one site in Pennsylvania (C. Bier pers. comm.).

#### Contaminants

Organochlorines including PCBs, DDT, DDE and Dieldrin were detected in black tern eggs collected during several studies in the Midwest (Faber and Hickey 1973, Mossman 1980, Faber and Nosek 1985). PCBs, DDE, Hexachlorobenzene and Octachlorostyrene (an industrial pollutant) were found in eggs and a chick collected from a site on Lake Ontario in western New York (Firstencel 1987). Significant changes in eggshell thickness were also noted in one study (Faber and Nosek 1985). Accumulated organochlorine contamination has been linked to lowered reproductive success and developmental problems in other tern species (Hays and Risebrough 1972, Fox 1976, Gerson 1987).

These results suggest that an accumulation of environmental contaminants may be contributing

to reduced reproductive success in the black tern throughout various parts of the breeding range. It is suspected that many of the contaminants may be accumulated while the terns are in migration or on their wintering grounds (Faanes 1979, Faber and Nosek 1985, Firstencel 1987).

The use of pesticides can also affect food supplies by depressing prey populations for black terns breeding near agricultural areas.

### Predation

Three of 38 (8%) nests and 16 of 26 (61.5%) penned young were taken by predators at 1 study site in Wisconsin (Bailey 1977), but the predators responsible for these losses were not identified. Mosher (1987) reported the loss of 8.6% of nests to predation. Great blue herons (*Ardea herodias*) and northern harriers (*Circus cyaneus*) have been observed taking black terns (Chapman and Forbes 1984, Maxson 1989). A host of other potential predators have been observed in or near nesting colonies and are suspected of regular or occasional predation on black tern eggs or chicks. Suspected predators include great horned owls (*Bubo virginianus*), black-crowned night herons (*Nycticorax nycticorax*), American crows (*Corvus brachyrhynchos*), water snakes (*Natrix sipedon*), snapping turtles (*Chelydra serpentina*) and ring-billed gulls (*Larus delawarensis*) (Cuthbert 1954, Goodwin 1960, Faber and Nosek 1985, Firstencel 1987, Novak 1990).

### Diseases and Parasites

Ectoparasites of the black tern include feather mites and lice (Peters 1936, Perez and Atyeo 1984). A trematode, *Aporchis larus*, was recorded from the black tern in Russia (Mirzoeva 1980). The effects of these parasites have not been studied.

Black terns are susceptible to avian botulism. A few dead birds have been found in Nevada and Manitoba (Alcorn 1942, Manuwal 1967), but no major die-offs from this disease have been reported. No other reports of diseases of the black tern were located.

### MANAGEMENT POTENTIAL

The relatively low reproductive output of the black tern appears to be due, at least in part, to storms, wind, and waves which destroy nests. Without a better understanding of other causes of low reproductive success which may include contaminant accumulation, predation and human disturbance, and factors affecting the black tern away from the breeding grounds, it is difficult to speculate on the likelihood that this species will recover from the widespread population decline it is currently experiencing. However, there are several reasons to believe that if prompt actions are taken, some degree of recovery is possible.

### Breeding Biology Considerations

Renesting after initial nest failure has been reported by various authors (Bergman et al. 1970, Bailey 1977, Novak 1990) and may partially offset losses suffered during initial nest failures.

Nesting area choice may vary from one year to the next depending upon changing water levels, nest substrate availability, and vegetation density (Weller and Spatcher 1965, Bailey 1977, Dunn 1979). Because of this sensitivity to habitat change the black tern is not tied to traditional nesting areas if the habitat deteriorates, and may colonize newly created, suitable habitat. For example, in Nova Scotia, black terns nested for the first time in the 1970s in new water impoundments (Gerson 1987).

The black tern can use a variety of nest substrates and is not restricted to wetlands with specific vegetation types. This adaptability reduces the chances of habitat limitation from the lack of available nest sites.

Nest success was as high as 50% in one recent study (Firstencel 1987). Although such high nesting success seems to be the exception, it indicates that the black tern is capable of achieving high reproductive output under some conditions. Based on banding records, black terns may live as long as 8 years (Clapp et al. 1982), perhaps longer, and therefore could benefit greatly from several successive years with high reproductive success.

## Wetland Protection Initiatives

The *North American Waterfowl Plan*, the President's "No-Net-Loss of Wetlands" policy, and the "Swampbuster" provision of the Food Security Act of 1985 (P.L. 99-198, commonly known as the 1985 Farm Bill) which prevents farmers who drain wetlands from receiving agricultural subsidies and other economic benefits of the bill, can all help to curtail the destruction of wetlands. Enforcement of state and federal wetlands regulations and a greater public recognition of wetlands values, also would help in the effort to preserve wetland habitat.

## Current Distribution and Population

Despite the fact that the black tern is currently experiencing a widespread population decline, this inland nesting tern has a North American breeding distribution which extends nearly continent-wide. The black tern remains widespread and fairly common in much of the Prairie province region and parts of the northcentral and western U.S. (Gerson 1987, Hands et al. 1989). The black tern has actually increased in New Brunswick and began nesting in Nova Scotia in the 1970s (Gerson 1987). The current distribution and population ensures that in most regions, at least for the near future, black terns should be available for both the recolonization of traditional or restored sites and for colonizing newly created habitats.

## Potential for Beneficial Management

Relative to overall state populations, large numbers of black terns breed on government-managed wetlands in Wisconsin (Mossman 1982), Michigan (Adams 1988, Einsweiler 1988), New York (Novak 1990), Vermont (Laughlin and Kibbe 1985), Maine (Pierson 1983, Gibbs and Melvin 1990), and perhaps other states as well. Management of wetland complexes has been employed successfully with waterfowl for many years (Fredrickson and Taylor 1982). Management of wetlands to benefit black terns should be possible in many cases (see *Management*

*Procedures*) and may be largely compatible with practices currently focusing on waterfowl.

If human disturbance is determined to have adverse effects on nesting black terns, education and restrictions on human access may be employed to eliminate or reduce the impact.

Black terns used 5 of 10 artificial nest platforms at 1 study site in Minnesota (Hands et al. 1989), all 3 platforms placed in New York in 1990 (R. Miller pers. comm.), and nearly 100% of the platforms placed at a managed area in Michigan in 1990 (W. Scharf pers. comm.). Although the value of artificial nesting platforms for black terns has not yet been demonstrated, platforms may provide a safer, more stable substrate than naturally floating objects, may be useful in luring terns to nest in more protected locations, and may provide suitable nest substrates in wetlands where natural substrates are in short supply. In any of the above situations, the result may be a higher reproductive success. However, it is also possible that in some cases platforms may simply lure black terns to nest in unproductive sites.

## LAND PROTECTION AND PRESERVE DESIGN

In states and provinces where the black tern is endangered, threatened or declining rapidly, every effort should be made to protect any colony sites currently in use regardless of the size of the site or the number of black terns present. Historical sites which still have  $\geq 5$  ha of suitable habitat should be protected as well. Preservation of these wetland areas by acquisition, lease, conservation easements, or management agreements should be actively pursued. In regions where black terns are less severely threatened, large ( $> 11$  ha) wetlands and sites which harbor substantial black tern populations should be similarly protected.

Habitat preservation alone will probably not ensure the recovery of the black tern in regions where population declines have been substantial. Successional processes, changes in water levels, invasion by exotic wetland plants, and degradation of water quality, which may alter both the food

web and the vegetative structure of the wetland, have the potential to render wetlands unsuitable for use by black terns. Management of wetlands will be required to maximize their value to black terns. The hemi-marsh stage (Weller and Fredrickson 1973), where open water and emergent vegetation are present in approximately equal amounts, is widely recognized as preferred nesting habitat for black terns. Although management recommendations and capabilities will vary from site to site, maintenance of large areas in the hemi-marsh stage should be a management goal. In large ponds, lakes and river systems where black terns nest in patches of rushes, cattail, or other emergent vegetation (Goodwin 1960, Bailey 1977), maintenance of these "islands" of emergent vegetation is recommended.

Development of agricultural lands surrounding wetlands supporting black terns should be discouraged because the terns may use the fields for foraging. Maintenance of buffer zones to block siltation, pesticide, and fertilizer runoff to the wetlands may also be desirable.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

**Monitoring Procedures.** - Surveys of black tern breeding colonies should be taken 2-3 weeks after terns begin to arrive on the breeding grounds. In northern states and Canadian provinces surveys should be conducted between late May and late June. Surveys should not be made on windy days or days with significant rainfall.

Standardized counts of adults may underestimate the actual number of breeding pairs present at a site, particularly at larger colonies (Tilghman 1979, Novak 1990). However, nest searches result in disturbance to nesting terns and caution should be exercised when conducting nest searches. Prolonged disturbance of nesting pairs should be avoided regardless of the survey method chosen.

The number of eggs or chicks present should always be noted while conducting nest searches. Information on nest substrate, water depth,

dominant plant species, height of eggs above water, distance to open water, and approximate distance to other tern nests may also be useful. If counts of adults, rather than nest searches, are chosen as the census method, a period of observation to determine the general areas being used by terns should precede the actual count. Multiple observers viewing separate portions of large sites will probably increase the accuracy of counts at some sites.

To establish a measure of reproductive success, follow-up visits to breeding colonies can be conducted to count fledged young and chicks. These surveys must be timed to occur soon after most young have fledged, but before dispersal from breeding areas begins. In the Northeast, these visits should generally be conducted between the first week and the end of July.

In regions where black terns are very rare, surveys should focus first on historical colony locations where suitable habitat is present, and secondarily on other sites which appear to provide high quality breeding habitat. In regions where black terns are more common it may not be possible to survey all potential colony sites and a sampling procedure will be necessary. Roadside transects patterned after BBS routes have been used effectively in Wisconsin (Mossman 1981, 1982).

**Monitoring Programs.** - The status of the black tern is currently being monitored primarily in states where fewer than 20 colony sites have been identified or where there is concern over population declines. These states include: Iowa, Idaho, Illinois, Indiana, Maine, Michigan, Nebraska, New York, Ohio, Pennsylvania, Vermont, Washington, Wisconsin, and Wyoming.

Field surveys have been conducted in many of these states in the past several years including Indiana, Wisconsin, New York, Vermont, Pennsylvania, Michigan, and Maine. Monitoring efforts will probably continue in most of these states although surveys may not be conducted on a yearly basis.

## Management Procedures and Programs

**Management Procedures.** - The range of options available for the management of specific sites to benefit black terns will vary with the degree to which water levels can be regulated at the site, the size and nature of the site, and the degree to which factors such as predation and disturbance are a problem at the site. Potential management procedures for a sample of sites of different types are as follows.

In managed inland marsh complexes, managers usually have some ability to regulate water levels in various impoundments or pools. Management procedures will vary from site to site depending upon a variety of factors including size of the area, number of pools in which water levels can be regulated, sources of water for altering water levels, natural precipitation rates, muskrat populations, and other management goals to be considered. In general, management for the black tern should be aimed at maintaining one or more large impoundments in the hemi-marsh stage for as long as possible. Pools in other stages of marsh succession may be used for foraging, but will be less preferred for nesting. Periodic drawdowns and reflooding, and management of the muskrat population through regulated trapping should be the primary management tools at these sites.

At most natural areas of shallow marsh associated with large lakes, ponds, and rivers there may be no practical means of regulating water levels. Because these sites are associated with larger, more open bodies of water, they may be used extensively for boating, fishing, and other forms of water-based recreation. Repeated disturbance and wave action may pose serious threats to black tern reproductive success at these sites. Educational efforts and/or restricting access may be effective at some sites. The degree and type of disturbance may influence the best strategy to be utilized. At colonies with excessive disturbance close to nest sites, restricting entry during the breeding season may be the only option available. In other cases, establishing no-wake zones or posting signs to discourage visitors may be effective. However, signs may also draw attention to colony sites and may be ineffective when enforcement is not possible (Connell and

Norman 1989). In these situations, efforts to educate the public may be the most reasonable method of reducing disturbance. Placement of artificial nest platforms to encourage terns to nest in areas where disturbance is less of a problem may provide a further management tool in some instances.

It is possible that extensive predation may pose a serious threat to nesting success at some colonies. Decisions regarding predator management should be made only after identification of the specific predators involved, documentation of the extent and effect of the predation, and careful consideration of alternatives and the likelihood of success in improving the situation.

**Management Programs.** - To date, the only known management efforts directed specifically toward the black tern include several recent studies on the use of artificial nest platforms (Rabenold 1987, Hands et al. 1989, W. Scharf pers. comm.). Nest platform utilization studies in Michigan are expected to continue.

The black tern is being considered in management planning at Iroquois and Montezuma NWRs in New York State, as well as Carlton Pond at Sunkhaze Meadows NWR in Maine. Specific programs have not yet been finalized.

## RESEARCH NEEDS AND PROGRAMS

### Research and Management Needs

The list of research and management needs for the black tern is extensive. No priority is intended in the listing of these needs.

- Determine the causes of nest failure and mortality in all age classes at nesting colonies in the northeastern and northcentral U.S.
- Evaluate the effectiveness of artificial nest platforms for increasing nesting success or population densities of nesting black terns. Emphasis should be placed on sites where natural nest substrates appear to be limited

or where terns may be encouraged to nest in areas where disturbance may be reduced.

- Incorporate management planning for the black tern in overall plans for all managed wetlands with suitable black tern habitat within the current or historical breeding range in the northeastern and northcentral U.S.
- Determine nest site fidelity of adults and philopatry of young.
- Determine the effects of contaminants on nesting success, chick development and juvenile and adult survival. Assuming significant negative effects are identified, determine how widespread the effects are with respect to the species' range.
- Assess the effects of human disturbance on black terns. In the northeastern U.S., emphasis should be placed on the impact of boating and other water-based recreational activities on nesting colonies.
- Assess the factors affecting renesting after initial nest failure and determine the productivity resulting from renesting attempts in comparison to initial nesting attempts.
- Determine foraging range and habitat use for black terns at important breeding sites in the northeastern and northcentral U.S.
- Develop or improve the capability to regulate water levels and manage habitat for the benefit of breeding black terns at key wetlands in the northeastern and northcentral U.S.
- Determine the movements, mortality rates, causes of mortality, and feeding habits of adult and subadult black terns during the nonbreeding season. Identify migration routes and critical habitats along migration pathways which may be in need of protection. Determine the extent of the winter range. Identify critical overwintering

sites and determine faithfulness to these sites.

### Research Programs

Studies on the utilization of artificial nest substrates were conducted in Michigan during 1989 and 1990 and are expected to continue (W. Scharf pers. comm.). Despite the extensive list of research needs no other ongoing research projects on the black tern were identified at this time.

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# SHORT-EARED OWL

## *Asio flammeus*

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**Short-eared owls have never been abundant as breeders in the Northeast, but fewer are now known from historical nesting sites. Today in the region, short-eared owls nest only in Vermont, New York, Massachusetts, and Pennsylvania. Populations traditionally fluctuate in response to the cycles of microtine rodents, their preferred food base, but loss of suitable breeding habitat to land-use changes appears to be the major limiting factor. Their ground-nesting habits in grasslands, saltmarshes, and on offshore islands renders the birds susceptible to mammalian predators, whose numbers are enhanced by human disturbance. Large tracts of open habitat are required for breeding and hunting and must be maintained in order to enhance the regional population. The use of fragmented habitats is unknown for breeding success. Adequate monitoring techniques are currently lacking to track short-eared owl populations.**

### DESCRIPTION

#### Taxonomy

One subspecies of the short-eared owl, *Asio flammeus flammeus*, is recognized over most of its extensive global range, which includes Europe, North America, and Asia (American Ornithologists' Union 1983, Cramp 1985). Seven (Mikkola 1983) or eight (Dement'ev and Gladkov 1951, Cramp 1985) other subspecies have been recognized from the rest of its range, including six endemic island populations (Hawaii, Galapagos, Falklands, Hispaniola, Puerto Rico, and Ponape in the Caroline Islands), and two resident, South American populations. Other than *A. f. flammeus*, the only subspecies in the U.S. is the

Hawaiian race, *Asio flammeus sandwichensis*. This endemic race is resident on all the islands of the Hawaiian chain from Kauai eastward, and apparently similar in form and coloration to the continental form.

#### Morphology and Plumages

The short-eared owl is a small to medium-sized member of the Family Strigidae. Published lengths range from 37-39 cm (Cramp 1985) to 34-42 cm (Mikkola 1983), with females slightly larger than males and considerably heavier, averaging 411 g compared to 350 g for males (Mikkola 1983). They are excellent flyers with long wings (95-110 cm) (Cramp 1985), and light wing-loading ( $0.333 \text{ g/cm}^2$ ) (Clark 1975). There is little

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difference in wing length between the sexes (Clark and Ward 1974).

Short-eared owls are tawny brown to buff-colored on the back and upper wing surfaces with heavy but indistinct streaking. They are much lighter on the ventral surfaces with brown streaking on the breast. Mature males are bright white on the underwing, while mature females show somewhat more buff coloration (Bent 1938, Village 1987). It is, nonetheless, difficult to sex or age these birds in the field. Females are generally darker than males but young birds are also darker than older ones (Mikkola 1983), thus a young male may be darker than an old female. Both sexes have a distinct, black carpal bar and dark wingtips. Juveniles possess full adult plumage by October of the first year (Bent 1938, Cramp 1985).

The facial disc is circular and whitish with dark areas around the bright yellow eyes. Recently fledged and juvenile owls show much darker coloration overall and a much darker facial disc which whitens with age. The owl gets its common name from the small ear-tufts over the eyes. These tufts are part of the facial disc and are erected when the bird is annoyed or alert. They may possibly aid in making birds more cryptic when in vegetation by breaking the line of the circular facial disc.

### Field Identification

Short-eared owls are probably the most diurnal of owls (Lockie 1955, Clark 1975) and may often be observed from late afternoon until nightfall, or at dawn. A crow-sized owl abroad during daylight in open country will most likely be a short-eared owl. However, they also hunt at night.

In the field the owl is easily recognized by its blunt-headed profile and the fact that it glides with its wings held horizontally. This contrasts with the shallow V-shape of the northern harrier (*Circus cyaneus*) with which the owl often shares habitat and may be confused. Northern harriers may also be distinguished from short-eared owls by their white rump patch. Habitat is useful in separating short-eared owls from long-eared owls (*Asio otis*), the latter being predominantly a

woodland dweller. The long-eared owl is also more slender with much longer ear tufts. The short-eared owl's style of flight is unique and has at times been called mechanical, moth-like, or even "slovenly" (Peterson 1934). To Forbush (1927), in contrast, the owl's flight exemplified "the poetry of motion."

### Vocalizations

The bird is generally silent but does vocalize in courtship (a low, repeated, hooting: "voo, hoo, hoo, hoo," or in conjunction with defensive behavior or annoyance ("yaps" or "barks"). Young owls give a food-begging call ("pssssip") that apparently aids adults in locating young from the time they leave the nest until after fledging. Adult owls may squeal while feigning injury during broken-wing acts to distract intruders from nests or young. Both young and adults will clack their bills when annoyed or in defense.

Apparently, no data exist on the use of broadcasting tape-recorded vocalizations of short-eared owls for detection or monitoring purposes.

### HABITAT

#### Breeding

Short-eared owls require broad expanses of open land with low vegetation for nesting and foraging. Habitat types frequently mentioned as suitable include fresh and saltwater marshes, bogs, dunes, prairies, grassy plains, old fields, tundra, moorlands, river valleys, and heathland (Dement'ev and Gladkov 1951, Clark 1975, Mikkola 1983, Holt and Melvin 1986). In general, any area that is large enough, has low vegetation with some dry upland for nesting, and that supports suitable prey may be considered potential breeding habitat, although many will not have breeding short-eared owls. Dement'ev and Gladkov (1951) assert that "nearby water" is a requirement for nesting habitat. The proximity of many northeastern nesting sites to water is therefore an interesting correlation.

## Wintering

Suitable breeding habitat may also be occupied by wintering birds. Conversely, Clark (1975) noted two occasions when winter territories became breeding territories. Short-eared owls tend to congregate and roost communally in the winter (Banfield 1947, Craighead and Craighead 1956, Clark 1975), often in sheltered sites near hunting areas. Winter roosts have been reported in abandoned dumps, quarries, gravel pits, storage yards, stump piles, old fields, small evergreen groves, bayberry thickets, dunes, and open, abandoned cellars (Clark 1975, Bosakowski 1986). Short-eared owls may also roost directly on the ground in tall grasses, possibly choosing vegetation of a coloration that blends with their plumage (Craighead and Craighead 1956).

In winter the species may abandon its ground roosting habit and roost in trees, possibly in response to deep snow (Banfield 1947, Bosakowski 1986). Smith (1989) noted drastic decreases in numbers of short-eared owls at known winter roosts on Point Peninsula in Jefferson County, New York, after a heavy snowfall created deep cover.

## BIOLOGY

### Reproductive Biology

**Courtship and Territorial Defense.** - Wing-clapping, exaggerated or deep wing-beats, and skirmishing are three displays seen predominantly during the breeding season (Lockie 1955, Clark 1975). They can be agonistic or territorial in nature, and are also used in courtship flights. The courtship flight of the short-eared owl is unique among birds and involves song, a spiraling flight, and wing-clapping by the male (DuBois 1924, Mikkola 1983, Holt 1985).

Short-eared owls wing-clap along territorial boundaries or during flights within a territory, in aggressive displays to other birds or to human observers, and in courtship flight. When wing-clapping, the owl's wings are brought below the

body and clapped together in short, rapid bursts. Both males and females may wing-clap.

Exaggerated wing-beats also occur in the same contexts as wing-claps. In this behavior the owl brings its wings high over its body, prominently displaying the underwing. Owls often patrol territorial boundaries using exaggerated wing-beats. Skirmishing involves other neighboring owls, usually along territorial boundaries, and is aggressive and territorial in nature. Exposure of talons, hovering, and sometimes actual striking of the other bird is involved.

Any of these displays observed during the breeding season may signify a territorial bird. Observation and mapping of these behaviors over a nesting season is the best way to delineate an owl's breeding territory (Lockie 1955, Village 1987, Tate 1991).

Short-eared owls generally begin courtship in mid to late March on Nantucket Island along the coast of Massachusetts (Holt and Melvin 1986; Tate and Melvin 1987, 1988). Courtship has been reported as occurring in mid-March in Montana (Dubois 1924) and as early as late February in Jefferson County, New York (G. Smith pers. comm.). Pitelka et al. (1955a) reported initial courtship activity in the first week of June at Barrow, Alaska. Unpaired males may engage in courtship flights well into the breeding season (Clark 1975, G. Tate pers. obs.). The breeding season is often reported to commence in direct relation to vole abundance with a larger prey population yielding an earlier start to breeding activities (Randall 1925, Snyder and Hope 1938, Lockie 1955, Mikkola 1983).

**Nesting.** - Depending on latitude, nesting activities generally begin in late winter to early spring across the owl's distribution. Timing of nesting may be correlated with latitude and prey abundance (Mikkola 1983, Cramp 1985).

The nesting cycle from nest initiation to fledging of young takes approximately 7-9 weeks in temperate zones lasting from mid-March to mid-September in the Northeast Region. During a 4-year study of short-eared owl breeding ecology on Nantucket Island, egg-laying began in April each year (as early as the first week) and all young were fledged by the first week of September (Holt and Melvin 1986; Tate and Melvin 1987, 1988;

Combs and Melvin 1989). Late nests or renests accounted for young fledging in late August and September (Tate and Melvin 1987, 1988).

Polygyny may result in two nests within one short-eared owl territory. On Nantucket, two broods from different females that overlapped temporally were raised within a territory defended by a single male (Tate 1991).

Unlike most owls that nest in holes or take over the abandoned nests of crows (*Corvus brachyrhynchos*) or other birds, the short-eared owl is unique within its family (Strigidae) in building a nest, albeit a crude one, on the ground. The female makes a small scrape in the ground with her body and lines it with nearby material. Nests may be lined with grass, leaves, twigs or feathers (Bent 1938, Clark 1975). These small nest depressions do not last long after the young have dispersed from the site (G. Tate pers. obs.).

Short-eared owls generally nest on high ground or upland sites (Pitelka et al. 1955; Clark 1975; Holt and Melvin 1986; Tate and Melvin 1987, 1988; Combs and Melvin 1989). Urner (1925) reported nests in a saltmarsh, one of which was subsequently flooded by a high tide, but in general, drier sites are preferred. During 5 years of study on Nantucket and Tuckernuck islands, all 41 nests found were in dry upland areas, though wetter sites were available (Holt and Melvin 1986; Tate and Melvin 1987, 1988; Combs and Melvin 1989; Combs and Griffin 1990). Eight nest sites at Monomoy National Wildlife Refuge, east of Nantucket, found between 1982 and 1985, were also all on dry upland sites (Holt and Melvin 1985).

Using a line-intercept technique (Brower and Zar 1977, Holt and Melvin 1986), vegetation characteristics of 15 nest sites on Nantucket were evaluated in 1986 and 1987. This analysis showed that low dense shrubs, mainly black huckleberry (*Gaylussacia baccata*) and bayberry (*Myrica pensylvanica*), that were < 0.5 m comprised 40.4%, and high dense shrubs (same species, ≥ 0.5 m) comprised 37.14% of the cover within 5 m of the nest (Tate and Melvin 1987, 1988). Other vegetation included low sparse shrubs (11.1%), low dense grass (8.1%), and high dense grass (3.0%, mostly *Andropogon scoparius* and *Ammophila breviligulata*). These data demonstrate

that in choosing nest sites on Nantucket, most short-eared owls seek low, dense shrub cover.

Short-eared owls generally lay between 4-9 eggs and sometimes more (Bent 1938), although Mikkola (1983) reported a range of 2-13 from 121 European records. Murray (1976) reported a mean clutch size of 5.61 from 186 nests in North America. A trend for mean clutch to increase from south to north was also noted in this sample. The largest clutch ever reported in the literature is 16 from Finland (Mikkola 1983). Large clutches of 14 in Scotland (Adair 1892) and 13 from Finland (Mikkola and Sulkava 1969) have also been reported. All exceptionally large clutches were laid in years of peak vole abundance in these areas.

Clark (1975) reported a mean clutch of 8.6 from 5 clutches in 1969 in Manitoba, Canada. Pitelka et al. (1955b) reported a range in clutch size of 4-8 with a mean of 6.3 from 22 nests in Alaska. A 4-year study of nesting owls on Nantucket reported clutch sizes of 5.8 (n = 6), 7.7 (n = 9), 6.8 (n = 8), and 5.2 (n = 8) in 1985-88 respectively, with an inclusive range of 4-9 (Holt and Melvin 1986; Tate and Melvin 1987, 1988; Combs and Melvin 1989).

Two broods are sometimes raised and, if the nest is destroyed or predicated, the female may renest (Lockie 1955, Mikkola 1983). Pitelka et al. (1955a) saw no evidence of renesting by short-eared owls in Alaska; this was apparently tied to the shorter warm season. In 1986 and 1987, single late nests with eggs were found on Nantucket in mid-July (G. Tate unpubl. data). These were suspected to be either second broods or renests.

Witherby et al. (1938) reported an incubation period for short-eared owls of 24-28 days in temperate zones. With data from six eggs in four nests, Pitelka et al. (1955b) reported an incubation period for Barrow, Alaska, that ranges from 26-37 days (mean = 30). He saw no evidence that incubation takes longer there than at lower latitudes. From a Finnish study of four nests, Gronlund and Mikkola (1969) reported an incubation period of 24-29 days (mean = 25.7). In 1986, three eggs, each from separate nests on Nantucket, were documented as having 29-, 30-, and 31-day incubation periods (mean = 30) (Tate 1991).

Normally the female does all of the incubation (Witherby et al. 1938, Dement'ev and Gladkov 1951, Pitelka et al. 1955a, Clark 1975) and lays at approximately 24-hour intervals (Mikkola 1983). She begins incubation with the first egg laid and hatching is therefore asynchronous. According to Mikkola (1983), the first and last eggs laid take the same length of time to incubate. Young owls leave the nest before fledging and wander into the surrounding area at about 2 weeks of age (Lockie 1955, Clark 1975). The young owlets stay concealed but continue to wander and are found and fed by both parents by means of the food-begging call given by the young. Fledging has been reported variously at 24-27 days (Witherby et al. 1938) and 31-36 days (Urner 1923). On Nantucket, young owls dispersed from the nest at 14-17 days and fledged when about 30 days old (Holt and Melvin 1986).

Age of first breeding is reported as one year or less (Mebs 1966, cited by Mikkola 1983; Glutz and Bauer 1980). Field evidence of breeding at one year has been obtained on Nantucket in 1990, when a sitting female that had been banded as a nestling in 1989 was trapped. This female was brooding on a nest only 98 m from her natal nest site (K.P. Combs unpubl. data). Short-eared owls have been known to live as long as 12.5 years (Mebs 1966, cited by Mikkola 1983).

Short-eared owls usually offer little defense of the nest from human intruders. Wing-clapping, circling overhead with deep wing-beats, "barks" or "yaps," and broken-wing acts are employed when any defense is attempted. Adults perform a distraction display that is a dramatic broken-wing act accompanied by vocalization. It is most often used by the male when an observer is at, or near, a nest or dispersed young (Clark 1975, G. Tate pers. obs.). However, often both owls vacate the vicinity of the nest site while an intruder is present. At times the female may desert the area, retreating to another part of the breeding territory, while the male remains nearby. Females may return to the nest by flying low and remaining inconspicuous (G. Tate pers. obs.).

While short-eared owls have been observed diving at house cats (G. Tate pers. obs.), the best defense is their cryptic coloration and the fact that the female sits tightly on the nest. On Nantucket, some females remained on the nest

while observers passed within 2 m, and on Tuckernuck Island, a female on a nest would not budge in spite of repeated attempts from as close as 1 m to flush her (G. Tate pers. obs.). These behaviors make it extremely difficult to find nests of this species.

The general area of a nesting can be ascertained by diligent observation. If one is fortunate enough to observe courtship there is a very good likelihood that the pair will nest in the immediate vicinity (D. Holt pers. comm.). Nests can be found by walking the area in question on foot in an attempt to flush the female. Repeated, systematic coverage is often needed due to the tenacity of the sitting female. During nest visits precautions should be taken against leading ground predators to nests.

Many short-eared owl nests have been found incidental to an ongoing study of waterfowl nesting at Grizzly Island Wildlife Area in California. Surveys of nesting waterfowl were made using a drag rope. It was concluded that this method was also efficient for surveying areas for nesting owls (Larsens 1987).

### Food Habits and Hunting Behavior

Short-eared owls take small mammals and some small birds while unfledged young may also take some insects. Their preferred prey is often reported as microtine rodents (e.g., Goddard 1935, Lockie 1955, Mikkola 1983). Clark (1975) reported meadow voles (*Microtus pennsylvanicus*) in  $\geq 91\%$  of pellets analyzed from a winter roost in New York between 1967 and 1970. Pitelka et al. (1955a) found that short-eared owls breeding

in Alaska, fed solely on brown lemmings (*Lemmus sibiricus*). Mikkola (1983) summarized studies of prey items taken in the breeding season by short-eared owls in Finland, Norway, Germany, and Hungary and found that voles (*Microtus* spp.) made up 78.9%, 65.1%, 94.6%, and 21.1% of the diet respectively. In Hungary, shrews (Soricidae) predominated in 69.9% of the diet while in Finland and Norway, voles (*Clethrionomys* spp.) and shrews accounted for most of the other identifiable prey items. Two voles, *Microtus agrestis* and *M. arvalis*, comprised 98% of all prey

species for short-eared owls in a Finnish study in 1977 (Korpimaki 1984).

On Nantucket Island, 7 species of small mammals were identified from prey remains in 1,992 short-eared owl pellets collected over 2 years (1986-87), predominately during the spring and summer (Tate 1991). Of these, meadow voles (*M. pennsylvanicus*) were the most abundant, accounting for 89% of prey items. Short-tailed shrews (*Blarina brevicauda*) and white-footed mice (*Peromyscus leucopus*) each comprised 3% of the total, with birds and insects making up 2.5% and 1.0%, respectively. Analysis of 1,214 pellets collected on Nantucket in 1985 showed that meadow voles comprised 93.3% of total prey remains (Holt and Melvin 1986).

Short-eared owls are attracted to areas with abundant food resources, and may breed opportunistically and sporadically in such areas. When they do find areas of especially abundant resources they may breed in large numbers (Pitelka et al. 1955a, Beske and Champion 1971, Larsen 1987) and produce super-normal clutches (Adair 1892, Goddard 1935).

Short-eared owls hunt predominantly by flying low over open areas in coursing flights much like those of northern harriers. Upon detecting prey the owl drops or pounces, sometimes briefly hovering beforehand. They may also hunt from a perch and dive directly on prey. At times short-eared owls also hunt using a hovering flight similar to American kestrels (*Falco sparverius*). They hang in the air for protracted periods of time at considerable heights until prey is sighted. As Clark (1975) notes, this protracted hovering has often been reported in the European literature but has been seldom mentioned for North American birds. On Nantucket, short-eared owls were most often observed hunting in this fashion over dunes of grasses (*Annmophila breviligulata*) (G. Tate pers. obs.).

During the breeding season, short-eared owls sometimes cache food (Young et al. 1988). On Nantucket Island three separate owls were observed caching food on four separate occasions. Each cache involved meadow voles that were taken short distances from the site of capture and placed beneath small shrubs. Each owl then resumed hunting (Tate 1991). Short-eared owls were observed caching food in Jefferson County,

New York, during a winter when food resources appeared abundant (G.A. Smith pers. comm.).

### Population Dynamics

Short-eared owls are often reported to fluctuate greatly in abundance between years in the same location. Their appearance and fecundity during peaks in density of microtine vole populations and their disappearance or poor breeding success when vole numbers are low is well documented (e.g., Adair 1892, Goddard 1935, Lockie 1955, Pitelka 1955a, Beske and Champion 1971).

Pitelka's Alaskan study (1955a,b) covered the period 1951-53, but only in 1952 and 1953 was the study site systematically searched for breeding birds; then an increase from 0 to 28 nesting pairs was recorded. Also, his discussions with local people did not reveal a prior familiarity with the species, suggesting little regularity of short-eared owls at that arctic site. Clark (1975) followed breeding by owls on a site in Manitoba for 2 years and recorded an increase from 1 to 6 nesting pairs. In Britain, Lockie (1955) recorded a change from 7 to 2 territories within a single breeding season. Village (1987) also recorded variation in the number of breeding short-eared owls in Scotland during a 3-year study. He presented data relating this to a decline in vole numbers over the course of the study.

Holzinger et al. (1973, cited by Mikkola 1983) studied breeding short-eared owls at a moorland near Ulm in southern Germany for 18 years. Variations in numbers at this site include shifts from 8 pairs in 1964 to 1 pair in 1965, and from 17 pairs in 1967 to 1 pair in 1968. In 1971 there were 14 breeding pairs at this site. Korpimaki (1984, 1985) recorded variations in short-eared owl abundance in an area of western Finland over 7 years. The number of pairs from 1977-83 was 34, 30, 31, 5, 2, 22, and 4, respectively. These fluctuations in numbers correlated significantly with fluctuations in vole (*Microtus* spp.) densities (Korpimaki 1985).

In contrast, little variation in the number of breeding short-eared owls was recorded during a 4-year study on Nantucket (Holt and Melvin 1986; Tate and Melvin 1986, 1987; Combs and Melvin

1989). A range of only 11-14 short-eared owl territories was observed from 1985-88 and long-time island residents have regularly reported breeding season sightings since the early part of this century in all areas where the study documented nesting (J.C. Andrews pers. comm.)

### Migration

Migrational patterns of short-eared owls are not well known. They apparently vacate more northerly parts of their range in the fall and move southward (Craighead and Craighead 1956, Clark 1975). In North America the movement is mostly from Canada and Alaska as far south as the southern U.S. and even Mexico (Cramp 1985). Toward the central part of the species' range (temperate zones) owls are seen year-round. Because breeding birds move south in fall and winter and are replaced by migrants from more northerly areas, Clark (1975) suggests that separate populations may occupy these areas during the different seasons.

It is also possible that the species migrates only in search of food and therefore may remain year-round in an area that provides sufficient resources. Short-eared owls do congregate in areas where prey is plentiful (Bent 1938, Mikkola 1983) and may migrate accordingly. This feature of their biology has earned them a reputation as an "irregular migrant," or as "nomadic," "irruptive," or "vagrant" (Clark 1975, Mikkola 1983).

### Winter Biology

In the winter, short-eared owls congregate at sites that provide good foraging (Craighead and Craighead 1956). Congregations of up to 200 birds have been reported (Bent 1938). Assemblage sites usually provide shelter and are within, or adjacent to, hunting areas (Clark 1975).

In wintering areas in New York where vole densities were high, Clark (1975) saw owls establish and defend hunting territories. Territories were less distinct when vole numbers were low.

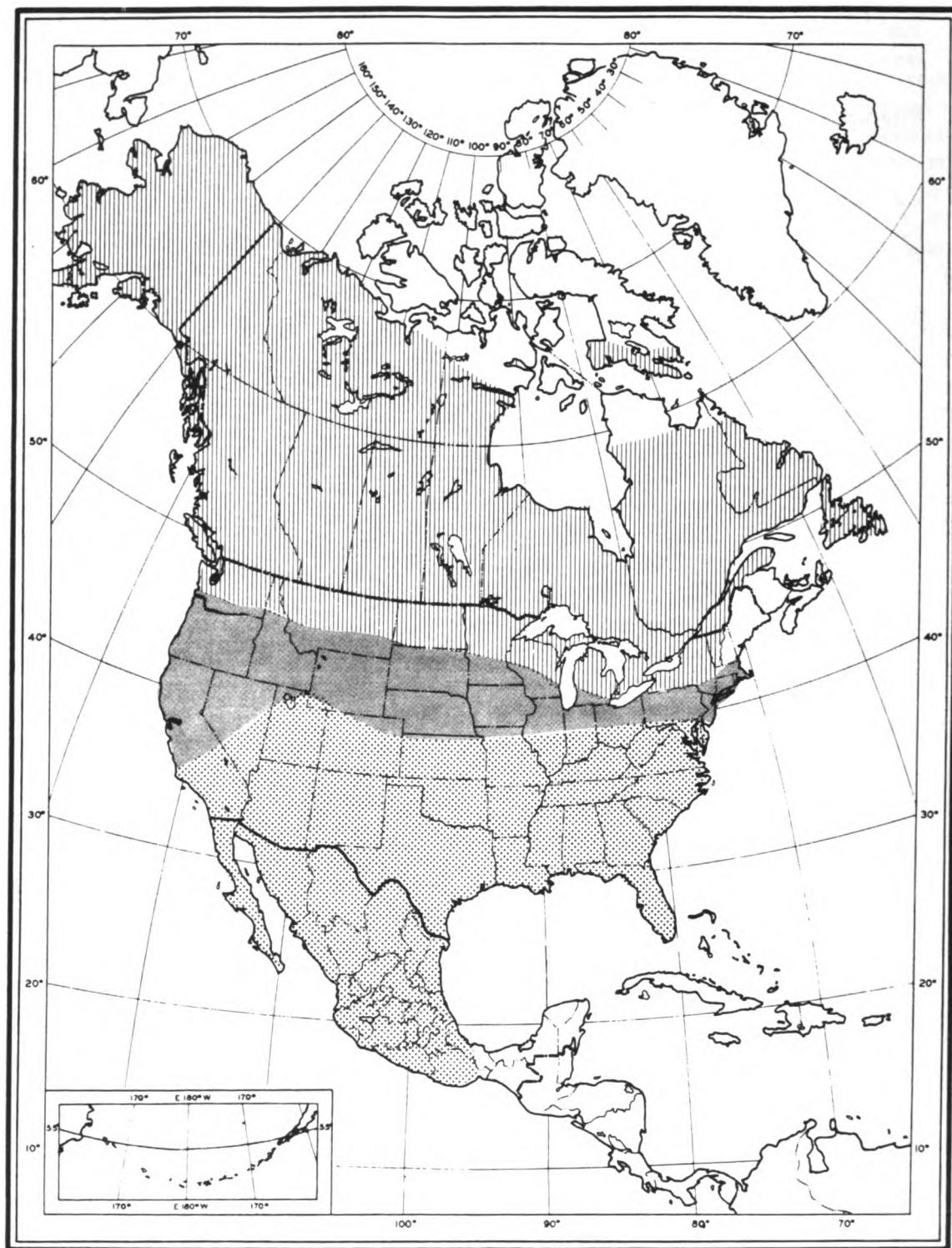
## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution in North America

The short-eared owl is widely distributed in its breeding and wintering range in North America (Figure 1). Nesting records as far south as latitude 30°N and as far north as latitude 70°N exist for North America (Murray 1976). The more northerly breeding areas are vacated in winter when the species is found in southern Canada (below approximate latitude 53°N) west of Ontario and, at times, as far south as southern Texas (Clark 1975). Clark (1975) describes a zone of permanent residency (where members of the species may be found year-round) that includes as its northern boundary southern British Columbia, Alberta, Saskatchewan, Manitoba, and southwestern Ontario; the southern shores of Lakes Superior and Huron; and southeastern Ontario east into the southern halves of Vermont, New Hampshire, and extreme southern Maine. Johnsgard (1988) places northern wintering limits further south into northern Idaho, southern Montana, northern South Dakota, and southern Minnesota, and further eastward including the coasts of Maine, New Brunswick, and Nova Scotia.

The southern limits of Clark's (1975) zone of permanent residency include central California, northern Nevada and extreme northern Utah; much of Colorado excluding the southwestern Rockies; southern Kansas, northwest Missouri and all of Iowa except the southeastern portion; southern Wisconsin and into southern Michigan, extreme northern Ohio, most of New York, and coastal New Jersey. Johnsgard (1988) places southern limits further south into southcentral California, most of Utah, only northern Kansas, and the northern halves of Illinois, Indiana, Ohio, and Pennsylvania. Winter sightings are often recorded in Oklahoma, Texas and New Mexico, and along the Mississippi River into Arkansas and Alabama (Root 1988).

Figure 1. The North American range of the short-eared owl (*Asio flammeus*) (American Ornithologists' Union 1983, Johnsgard 1988).



Breeding

Year-round

Wintering

## Status in North America

Bent (1938) first noted diminishing numbers of short-eared owls, and attributed this to shooting. The species has been included on the National Audubon Society's *Blue List* of declining bird species since 1976. In the most recent *Blue List* (Tate 1986), three of the seven North American regions reporting (Central Southern, Prairie Provinces, and Middle Pacific Coast Region) listed the bird as "greatly down in numbers," while the remaining four (Hudson-Delaware, Ontario, Middle-Western Prairie, and Southern Great Plains) reported it as "down in numbers." In a review of the status of small owls in the Northeast, Melvin et al. (1989) listed the short-eared owl as the "rarest and most threatened species of owl" in the region. Across its entire range, the species' Natural Heritage ranking is G5 (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

Although never an abundant breeder in the Midwest, the short-eared owl is apparently in decline in this region. The species is considered imperiled by the state Natural Heritage Programs in Michigan, Indiana, and Illinois (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia). It is designated as S1 (5 or fewer occurrences in the state) or S2 (6-20 occurrences) by all of the Natural Heritage inventories in the Midwest except in Minnesota where it is listed as S3 (20-100 occurrences) and as a *species of special concern*. Short-eared owls have recently been reclassified from state *extirpated* to state *endangered* by the Missouri Department of Conservation because of the discovery of four nests in prairie habitat during the 1990 breeding season. Rodent populations apparently peaked during this season (B. Jacobs pers. comm.). Various observers relate the decline in breeding short-eared owls in the Midwest to loss of habitat, specifically loss of native prairies (D. Figg and C. Hedge pers. comm.). Reclaimed strip-mining areas may present the best possible new nesting habitat for the species (D. Rice and C. Hedge pers. comm.). Throughout the Midwest, short-eared owls are more commonly seen during migration and in winter.

In southern Ontario, short-eared owls are limited in their distribution as a breeder and have declined during this century (Cadman et al. 1987). According to the authors, the short-eared owl is "*the most limited in occurrence of Ontario's owls*." This decline may be related to the loss of wetlands to agriculture and housing developments. Breeding short-eared owls occur along the coasts of Hudson and James Bays and in some southern agricultural areas. Short-eared owls may be more abundant in northern Ontario (Cadman et al. 1987).

Numbers of breeding short-eared owls appear to fluctuate considerably in the Maritime Provinces of Canada. A.J. Erskine (unpubl. data) estimates 100 breeding pairs or fewer in any given year. More than one-half of these pairs breed in New Brunswick, with regular concentrations at the Tantramar Marshes, nearby Shepody Marsh, and on the Acadian Peninsula. Nova Scotia is estimated to have about one-third the total, with a small breeding concentration at Grand Pre, while Prince Edward Island has very few breeding pairs. In 1980, when meadow vole numbers were reportedly high, R.E. Simmons found 33 nests in a 200-ha area of reclaimed dikeland, mostly hayfields, near Joicure, New Brunswick (A.J. Erskine unpubl. data.). Such fluctuations have led to the conclusion that numbers of breeding short-eared owls are no more or less abundant than in the past. In the Great Plains states and into southern Canada and westward, the short-eared owl is still an abundant nester where there is suitable habitat. In most of the Great Plains and western states where it is found, Natural Heritage inventories consider the species demonstrably secure (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

However, in some western areas, local and regional declines have been noted. The short-eared owl is a rare and local breeder in Kansas where it was formerly more common (B. Busby pers. comm.). In Nebraska and California, short-eared owls are listed as S2 (6-20 occurrences statewide) by the state Natural Heritage Programs, and the State of Utah designates it as a *species of special concern* in the northern region (D. Paul pers. comm.). California has designated the short-eared owl a *species of special concern* as a result of its decline as a nesting species. In that

state the species is now absent as a breeder from many former nesting locations, particularly along the southern coast. This decline, as with others, is tied to destruction of habitat; in California, marsh and tall grassland particularly are being lost (Remsen 1978). Remsen (1978) also mentions the species' vulnerability to shooting as a factor in its decline in California.

In more northerly areas, such as northern Canada and Alaska, short-eared owls may occur irregularly in relation to the abundance of prey (Pitelka et al. 1955b).

### Distribution and Status in the Northeast Region

The short-eared owl has never been an abundant breeder in the Northeast. Its numbers are, nonetheless, definitely declining and this decline seems tied to habitat loss (Melvin et al. 1989). Presently, owls are known to nest consistently only in Vermont, New York, Massachusetts, and Pennsylvania. The following is a state-by-state summary of past and present records in the Northeast Region.

**Maine.** - Short-eared owls presently receive no special legal protection from the Maine Department of Inland Fisheries and Wildlife other than that afforded by general migratory bird laws. They are considered historical nesters (*SH*) by the Maine Natural Heritage Program because of reported breeding along the southern coast during the early part of this century (Table 1). Circumstantial evidence of past breeding exists for Aroostook County (Palmer 1949). There are no recent confirmed breeding records (Adamus 1988).

**New Hampshire.** - Short-eared owls are not known to have nested in New Hampshire, but wintering birds are sometimes seen, mostly at coastal areas. Occasional sightings are reported from other locations across the state (E. Hentcy pers. comm.). The species is ranked as *SN* (regularly occurring but not breeding) by the state Natural Heritage Program.

Table 1. Summary of state listing status and Natural Heritage Program state ranks for the short-eared owl (*Asio flammeus*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	SH
New Hampshire	-	SN
Vermont	special concern	S1
Massachusetts	endangered	S1
Rhode Island	-	S?
Connecticut	threatened	SH
New York	special concern	S2
Pennsylvania	endangered	S1
New Jersey	endangered	S1
Delaware	-	SH
Maryland	special concern	SH
Virginia	-	S1
West Virginia	-	SH

<sup>a</sup>SH = historical records only, but suspected extant

SN = regularly occurring but not breeding

S? = status uncertain

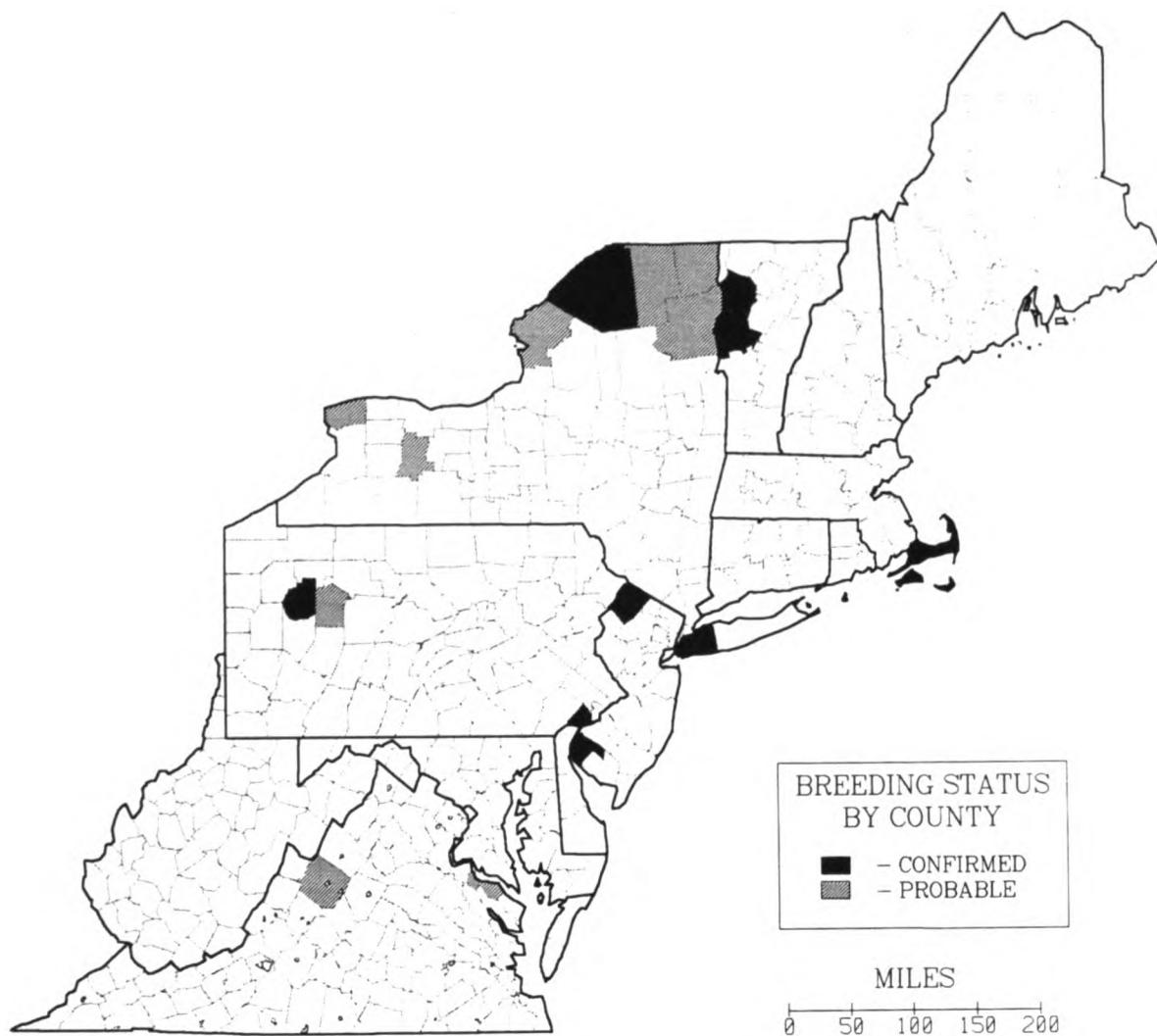
S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

**Vermont.** - Although past and recent nesting records exist, the species has always been rare in Vermont. The state's breeding bird atlas (BBA) (Laughlin and Kibbe 1985) reports two recent (1980 and 1981) confirmed nestings and two other records (in 1976 and 1977) of pairs or single birds observed in the breeding season (Figure 2). These nestings, and most sightings, occurred in the Champlain Lowlands and significant wintering concentrations have been known to occur in Ferrisburg, at Dead Creek Wildlife Management Area headquarters in Addison, and in Vergennes. The owl is listed as a *species of special concern* by the Vermont Department of Fish and Wildlife, and is ranked as *S1* by the Vermont Natural Heritage Program.

**Massachusetts.** - Short-eared owls have declined greatly as a breeding species in

**Figure 2.** Northeast breeding distribution of the short-eared owl (*Asio flammeus*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



Massachusetts (Melvin et al. 1989). Breeding pairs are currently restricted to the islands of Nantucket, Tuckernuck, and Monomoy (Holt and Melvin 1986). These populations may represent the largest concentration of nesting short-eared owls on the east coast of North America, estimated at 20-25 resident breeding pairs in 1985 (Holt and Melvin 1986). In 1987, surveys of breeding short-eared owls located only 18 pairs plus at least 6 unpaired, territorial birds (Tate and Melvin 1988). The Nantucket and Tuckernuck populations are regular breeders with consistent numbers of pairs, and even consistent breeding territory locations (Tate 1991).

Short-eared owls bred regularly on Martha's Vineyard prior to the 1980s. The last known breeding attempt on this island was in 1987; it was unsuccessful (T. Simmons pers. comm.).

The last record of breeding on the Massachusetts mainland was on Cape Cod in 1985 (Holt and Melvin 1986). Earlier, birds were reported as summering in 13 of 14 counties and confirmed as breeding in 5 counties (Forbush 1927). The species is listed as *endangered* by the state of Massachusetts and given a rank of *S1* by the Massachusetts Natural Heritage and Endangered Species Program.

Significant wintering populations often occur in the state. A Massachusetts Audubon Society winter census of short-eared owls in 22 coastal, or near coastal areas in 1990, reported 46 owls from 10 sites (Perkins 1990).

**Rhode Island.** - Although Forbush (1927) reports egg dates for Rhode Island and lists the bird as a "rare summer resident," there are no confirmed reports of short-eared owls nesting in the state. Wintering birds in small numbers are sometimes reported from several mainland coastal sites (e.g., Sachuest Point National Wildlife Refuge) and on Block Island.

**Connecticut.** - Short-eared owls are considered extirpated from Connecticut as a nesting species, apparently in the late 1800s. There are references to possible nesting locations along the Connecticut River north of Hartford, and along the coast before the turn of the century. Wintering populations, mostly at coastal locations, are also less common than in the past (Bevier et

al. in press). The Connecticut Department of Environmental Protection lists the short-eared owl as a *threatened* species, and its state Natural Heritage rank is *SH* (historical records only).

**New York.** - New York apparently has the largest population of breeding short-eared owls in the Northeast Region. The New York BBA (Andrie and Carroll 1988) reports 5 blocks with confirmed breeding pairs, 9 with probable breeding, and 22 blocks with possible breeding. Concentrations of owls are located along the southwest shore of Long Island, in upstate New York along the northeast Lake Ontario Basin, and west of the Finger Lakes region. The short-eared owl is nonetheless an uncommon breeder in New York and is designated a *species of special concern* by the New York State Department of Environmental Conservation and ranked *S2* (6-20 occurrences) by the New York Natural Heritage Program.

**Pennsylvania.** - Pennsylvania lists the short-eared owl as *endangered* and it is ranked *S1* by the Pennsylvania Natural Diversity Inventory. One site with consistent nesting is known from Philadelphia County and two nestings were recently added (1988) from Clarion County in the northwestern part of the state (Brauning in press) in areas of reclaimed strip mines. Scattered winter occurrences of fluctuating numbers occur around the state (D. Brauning pers. comm.).

**New Jersey.** - Short-eared owls nested in New Jersey earlier in this century in marshland near Elizabeth and Newark and further south at Cape May (Urner 1921, 1923; Bent 1938). Two recent nestings were reported during the New Jersey BBA from Sussex and Salem counties (D. Hughes pers. comm., Figure 2). The winter population appears stable and sightings during this season in appropriate habitat are widespread (New Jersey Natural Heritage Program: *Vertebrate Characterization Abstract*, Trenton). Short-eared owls are listed as *endangered* in New Jersey and ranked *S1* (critically imperiled) by the Natural Heritage Program.

**Delaware.** - Short-eared owls bred in Delaware in the 1930s at Bombay Hook National

Wildlife Refuge but no recent nesting records are known. Winter sightings are fairly common at Bombay Hook and Prime Hook National Wildlife Refuges (Hess et al. in press). The species' state Natural Heritage rank is *SH* (historical records only).

**Maryland and the District of Columbia.** - There has been only one confirmed nesting record for the state of Maryland. Five young were found on 13 May 1958, in brackish marsh habitat near the mouth of the Blackwater River in Dorchester County (Maryland Birds 1958). Other breeding season records of one to three birds exist from Dorchester, Howard, and Montgomery counties (Robbins 1969, 1972; Klimkiewicz and Solem 1974). The species is listed as *special concern* by the Maryland Department of Natural Resources and *SH* by the state Natural Heritage Program. Wintering populations are uncommon but may occur in tidal saltmarshes along Chesapeake Bay or in open agricultural land (Stewart and Robbins 1958).

**Virginia.** - Scattered occurrences of wintering short-eared owls are regularly reported in the state from late October until late March at sites that provide suitable habitat. Numbers are generally low but an "irruption" of 50-75 birds in Loudoun County was reported in the winter of 1949-50 (J. Bazuin pers. comm.). Most winter records are from the Atlantic coastal marshes.

Short-eared owls have apparently always been rare in Virginia during the breeding season. During the early part of the century, Bailey (1913) speculated that nesting occurred on the southern tip of the Eastern Shore. Two other nestings have been reported for the state (1950 and 1973), both in Loudoun County (Murray 1952). In 1987 a single short-eared owl was flushed from a marsh in Matthews County in early July (J. Bazuin pers. comm.). The only probable breeding record from the Virginia BBA was from Westmoreland County where a pair of birds was present in the same area for 13 consecutive months (S. Ridd and R. Wadja pers. comm.). The species is ranked as *S1* by the state Natural Heritage Program.

**West Virginia.** - The short-eared owl is not known to nest in West Virginia. Winter sightings

are uncommon to rare with most classified as "accidental" or "casual" (Hall 1983). Little habitat exists, except for extensive bogs. Short-eared owls are ranked *SH* by the state Natural Heritage Program.

## LIMITING FACTORS AND THREATS

### Habitat Loss

It appears that the short-eared owl is declining because of the loss of appropriate breeding habitat. This may be a result of development, changing land-use patterns (e.g., farmlands to woodlands, or to development), changing farming practices (e.g., hay fields to row crops), reforestation, wetland loss, or a combination of these factors.

Prey abundance may be a limiting factor in the short-eared owl's distribution and breeding success (Melvin et al. 1989). This owl's reported reliance on microtine rodents emphasizes this specific habitat component (Lockie 1955, Hagen 1969, Clark 1975).

### Diseases, Parasites, and Predation

Disease is presently not known to limit short-eared owl populations. Harrison (1943) reported a short-eared owl infected with *Mycobacterium tuberculosis avium*. In 1987, four young owls from two widely separated nests on Nantucket Island were discovered to be suffering from a feather disorder of unknown cause (Tate 1991), in which the juvenal plumage was not developing properly and the emerging feathers were twisted or malformed. All of these young had developed open sores, apparently from picking at the skin around these feather shafts with their beaks. This problem is not known to have been reported previously in the literature and no cause was determined.

As a ground-nesting bird, short-eared owl eggs and young may fall prey to various mammalian ground predators such as foxes (*Vulpes* spp.), raccoons (*Procyon lotor*) and mustelids. Predation by the striped skunk (*Mephitis mephitis*) may be one of the reasons for

the extirpation of the short-eared owl as a breeding bird on Martha's Vineyard (Melvin et al. 1989). Domestic cats and dogs have been known to disturb owl nests on Nantucket Island (Tate 1991). The potential for an increase in the threat of predation or disturbance by domestic or feral cats and dogs may be high.

Avian predation on this species has been reported by a number of observers. Predatory species include great horned owls (*Bubo virginianus*), snowy owls (*Nyctea scandiaca*), peregrine falcons (*Falco peregrinus*), and marsh harriers (*Circus aeruginosus*) (Clark 1975). Northern harriers, American crows, and European kestrels (*Falco tinnunculus*) have been known to pirate prey from short-eared owls (Village 1987, Tate 1991).

Short-eared owls have often been subjected to shooting by humans (Bent 1938, Clark 1975, Remsen 1978, J.C. Andrews pers. comm.).

### Other Factors

Two birds with broken wings were recovered in Massachusetts in 1986 from separate localities. These may have been the result of collisions with large, aerial radio antennas or high-tension guy wires. Clark (1975) reported mortality caused by collisions or entanglement with trains, cars, aircraft, farm machinery, and wire fencing.

Loss of open grasslands to later successional stages of community development reduces available hunting and breeding habitat. On Nantucket Island, succession of maritime heathland to scrub oak (*Quercus ilicifolia*) forest may be reducing habitat available to short-eared owls (Tate 1991).

### MANAGEMENT POTENTIAL

Evidence that short-eared owls have begun to breed in recovered strip-mine areas of Pennsylvania and possibly Ohio is encouraging. Although breeding populations in the Northeast and the Midwest are declining, large breeding populations still exist in other parts of the country. Recovery of the Northeast population

depends on restoration of large areas of open habitat and an influx of breeding birds.

### LAND PROTECTION AND PRESERVE DESIGN

Some useful data exist on which to base preserve designs for conservation of this species. Acreage of breeding territories has been estimated in a few studies. These data should serve as guidelines for the minimum acreage necessary for breeding since nocturnal habitat use is seldom documented.

The mean size estimate for 27 short-eared owl breeding territories on Nantucket Island from 1985-88 was 51 ha with a range of means from 40 ha (98 ac,  $n = 4$ ) in 1987 to 67 ha (166 ac,  $n = 5$ ) in 1988 (Holt and Melvin 1986; Tate and Melvin 1987, 1988; Combs and Melvin 1989). The smallest breeding territory size during this period was estimated to be 25 ha, while the largest was 98 ha (Holt and Melvin 1986, Combs and Melvin 1989). These estimates, based on observed territorial displays and daylight foraging, are undoubtedly underestimates since they do not take into account nocturnal foraging, which may be extensive as evidenced by a radio-tagged short-eared owl on Nantucket Island in early September of 1988 (K.P. Combs pers. comm.). Lockie (1955) estimated territory size for breeding owls in Scotland, and observed a change from 7 to 2 territories within a single season (April/May compared to June/July). Seven early-season territories averaged 18 ha, while two late-season territories were 156 and 118 ha. Clark (1975) estimated a mean territory size of 73.9 ha for five pairs on a Manitoba study site in 1969, and 121.4 ha for one nesting pair in 1968. Estimates of small mammal abundance for either of these studies are insufficient to relate to territory size.

Over 3 successive years, Village (1987) estimated territory sizes for 23 short-eared owls in winter (solitary) and 51 in summer (breeding pairs) in southern Scotland. Mean territory size in winter ranged from 42-72 ha and in summer from 62-112 ha. The mean size estimate for summer territories was 85 ha ( $n = 51$ , range = 25-242 ha).

From the available data it is obvious that land preservation efforts must be aimed at protecting large tracts of open habitat with low vegetation. In areas where short-eared owls nest, observation of hunting flights and territorial displays during the breeding season will help to delineate areas used by resident owls. Any adjacent similar habitat not used by the birds should be considered potential habitat. Areas of approximately 50 ha or larger, of low, open grasslands or similar habitat with abundant small mammal populations should also be considered as potential breeding or wintering habitat. Because of the abundance of coastal sightings in the Northeast in winter, protection of large areas of coastal grassland, heaths, and saltmarshes for wintering owls is also recommended.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Combs and Griffin (1990) reported that birds are most visible in June on Nantucket Island. Abundance was estimated by driving standard survey routes within territories and counting birds seen during early mornings and late afternoons. Short-eared owls are often active in early-mornings, at dawn and just after, and late-afternoons, 2-3 hours before sunset. Therefore, counts at these times of day are most beneficial for accurately censusing populations. On Nantucket most young have left the nest in June and are either dispersed nearby or have recently fledged (Holt and Melvin 1986; Tate and Melvin 1987, 1988; Combs and Melvin 1989). Therefore, population counts at this time would presumably include both adult males and females since the female is no longer on the nest. Seasonal timing of monitoring is important in obtaining an accurate count of total number of breeding pairs in an area. Fledged young may also be distinguished at this time by their darker overall coloration.

Village (1987) monitored changes in owl numbers in one year from fall to summer by counting all owls seen while driving along roads within appropriate habitat. He then estimated

abundance by dividing the total number of owls seen by the total length of roads driven.

### Management Procedures and Programs

Management procedures should address known and suspected limiting factors. Melvin et al. (1989) listed limiting factors in the Northeast as (1) availability of suitable habitat, (2) sufficient prey abundance, (3) predation, and (4) human-related disturbance.

Management for suitable habitat includes maintaining large tracts of open grassland, salt or freshwater marshes, or other appropriate habitat. Protection of such habitat is crucial to the persistence of short-eared owls as breeding and wintering birds in the Northeast and elsewhere. Restoration or new establishment of grasslands (e.g., from strip-mined areas) may offer potential habitat.

Active maintenance of open habitat may be beneficial to short-eared owls where succession to woody vegetation occurs. Any management practice used to maintain open habitat and inhibit the growth of woody vegetation, such as mowing or burning, might be used. However, these management practices must be employed outside the nesting season to avoid the destruction of nests, eggs and young. In addition, care must be taken to allow for adequate build-up of the litter layer that provides habitat for microtine rodents. *Microtus* spp. populations require adequate cover for several aspects of their ecology (e.g., see Birney et al. 1976). Maintenance of an adequate prey base for short-eared owls is essential since distribution and abundance seems to be tied to prey density (Adair 1892, Lockie 1955, Clark 1975, Melvin et al. 1989).

In California, short-eared owls have evidently benefitted from management strategies designed to increase waterfowl nesting habitat (Larsen 1987). Fields that were experimentally managed to produce a dense litter layer of dried vegetation contained both the highest density of duck nests and the highest density of owl nests. Prey populations also appeared to increase, although no data were collected on prey density. Such litter layers occurred in fields in the second year after disking and planting in perennials or annuals.

Management might include the control of predators in areas where short-eared owls nest, where populations of potential ground predators are high, and where predators are known to have a significant impact on owl nesting success. However, trapping and removal of predators or elimination of predators by killing is seldom practical. Predator exclosures have apparently never been used at short-eared owl nests. Their design would have to take into account the dispersal of the young on foot and the type, or types, of ground predators to be excluded.

Educational programs on the natural history, ecology, and conservation of short-eared owls, other raptors, and grassland birds might help alleviate human-related disturbances, such as shooting and harassment by domestic pets. Education should address the need to legally protect declining species and the consequences and benefits of legal protection. Education could also lead to an increased willingness to preserve habitat. Although single-species protection is important, education aimed at increasing the awareness of the inclusive benefits of habitat protection is now needed. Preservation of open habitat in the form of grasslands, heathlands, marshes, etc., is a key component in the preservation of a number of declining birds. Such species as grasshopper sparrow (*Ammodramus savannarum*), Henslow's sparrow (*A. henslowii*), upland sandpiper (*Bartramia longicauda*), loggerhead shrike (*Lanius ludovicianus*), and northern harrier would also benefit from an increased effort to preserve or restore open habitat. This knowledge should be an added incentive to proponents for open habitat preservation.

### Research Needs

Adequate monitoring procedures are needed. Many of the present monitoring techniques, such as walking large areas in an attempt to flush sitting females, are labor and time intensive. Furthermore some procedures, such as counting birds seen from roadsides, may underestimate the number of individuals present. Short-eared owls are relatively inconspicuous and easily missed. Development and implementation of a standardized population monitoring procedure

throughout the Northeast should be a high research priority.

Expanded efforts are needed to locate and study local breeding and wintering populations to (1) accurately determine number of breeding or wintering birds, (2) locate regularly occurring populations which would facilitate long-term ecological studies, and (3) determine more precisely the limiting factors and management needs of these populations. The location of regularly breeding and wintering populations would provide the data necessary for land protection efforts. The mapping of short-eared owl territories on Nantucket and Tuckernuck Islands, for example, has allowed the protection of known breeding sites by conservation organizations (Tate and Melvin 1987).

Mapping of short-eared owl breeding territories through the observation of territorial displays during the breeding season allows for good estimation of territory size (Lockie 1955, Tate 1991). More data is needed concerning relationships between territory size and the abundance of small mammals to determine the amount of open habitat and the prey base necessary to support a breeding pair. Such information would enhance the efficacy of land preservation efforts.

Research on the management of open habitat and its effect on prey populations is needed. The effect of such practices as burning, mowing, or plowing on small mammal populations must be taken into account. Any management for the restoration or maintenance of open grassland habitat for short-eared owls must also manage for a sufficient prey base.

The question of habitat fragmentation, or isolation, for short-eared owls may be important. Continuous expanses of open habitat are rare in the Northeast. The high mobility of the species may allow it to utilize disjunct areas of habitat, but whether short-eared owls will occupy an isolated patch of appropriately-sized habitat and breed successfully is not known. Wildlife managers seeking to design preserves would need to know to what degree utilization of fragmented habitat is detrimental to the owl's territorial integrity and breeding success.

Other research needs include (1) the investigation of nocturnal movements, (2) the

study of movement in and out of isolated populations, e.g., Nantucket Island and Monomoy National Wildlife Refuge, and (3) estimates of adult and juvenile annual mortality.

Despite important research needs, there is presently enough information to effectively guide conservation efforts in the Northeast Region. Attention to the conservation needs of this species is needed soon if the short-eared owl is to persist as a breeding bird in the Northeast.

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# SEDGE WREN

## *Cistothorus platensis*

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Sedge wrens nest in tall, dense growths of sedges and grasses in wet meadows and hayfields, upland margins of ponds and marshes, and coastal brackish marshes. In the Northeast, sedge wrens breed at scattered localities from New England and northern New York south to tidewater areas of Chesapeake Bay. Sedge wrens exhibit low breeding and wintering site fidelity and will abandon sites rendered too dry by draining or drought or too wet by flooding. Historical accounts suggest that populations in all states have undergone substantive declines during this century. Breeding Bird Survey data indicate that sedge wrens in the Northeast declined 1.47% annually from 1966-90. Existing surveys may undersample sedge wrens because territory establishment and nesting often do not occur until July. Urbanization, agricultural practices, and natural succession continue to degrade and eliminate habitat. Preservation and maintenance of complexes of breeding habitat, especially grasslands and wet meadows adjacent to natural or impounded wetlands, is the most urgent management need. Tidewater marshes in New Jersey, Delaware, Maryland, and Virginia currently support the largest, most stable populations of sedge wrens in the Northeast and should be priorities for protection. Given the small and transitory nature of sedge wren populations, habitat management across wide geographic areas may be necessary to significantly enhance regional populations.

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### DESCRIPTION

### Taxonomy

Formerly known as the short-billed marsh wren, this bird's common name was changed to

the sedge wren (*Cistothorus platensis*, Latham 1790), to better distinguish it from the marsh wren (*C. palustris*) by emphasizing the habitat differences between species (American Ornithologists' Union 1983). Three disjunct populations occur in North, Central, and South

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America, but probably represent a single superspecies (Traylor 1988).

### Morphology and Plumages

The adult plumage of sedge wrens is warm and pale brown overall, giving the birds a "dull, plain-faced" appearance (Vickery 1983). The deep brown wing coverts and short tail are barred, the flanks are buff-orange, the chin, throat and center belly are dull white, and the undertail coverts are plain buff. The crown is "finely, somewhat obscurely, streaked with tan and dark brown" (Vickery 1983), and the tail is typically cocked over the back. In contrast to marsh wrens, the only other wren occurring in eastern wetlands, sedge wrens lack a distinct white stripe over the eye, are lighter-colored, and have a shorter bill (Vickery 1983).

Sexes have similar plumages, and males are about a gram heavier than females (Taylor et al. 1983). Juveniles are similar to adults except that the streaking on the head and neck is less conspicuous, and the juvenal plumage is darker above and more buffy on the throat and abdomen (Forbush 1929, Walkinshaw 1935).

### Vocalizations

The song of male sedge wrens is a dry, staccato chattering: "*chap chap chap chapper-rrrr*" (Peterson 1980). Among individual sedge wrens, the song's introduction is stereotyped, whereas the trill is highly variable (Kroodsma and Verner 1978). The call note is "*churr churr*," "*chap churr*," or "*chap*" (Bent 1948, Peterson 1980). Sedge wrens frequently sing at night (Vickery 1983).

### HABITAT

Sedge wrens nest among dense, tall growths of sedges and grasses in wet meadows, hayfields, retired croplands, and upland margins of ponds and marshes. Nesting also occurs in coastal, brackish marshes. Scattered shrubs and an absence of standing water are also typical features of nesting habitats. Sedge wrens are highly sensitive to habitat conditions, and abandon sites

rendered too dry by drainage or drought or too wet by flooding. They will also abandon sites if shrubs become too prevalent. Sedge wrens are opportunistic breeders; they may renest at different locations during the same breeding season, and usually do not occupy the same site for more than 1-3 years.

In New England, Bagg and Eliot (1937) suggested that sedge wrens colonized wet meadows early in the nesting season, but, due to summer drying, used permanently wet, tussocky marshland in July for renesting. Nesting has been reported in Maine from a variety of habitat types, including wet fields, bush-covered pastures, sedgy and grassy meadows, marshes, bogs, and thick, grassy areas on barrenlands (Palmer 1949). Preferred habitats in tidewater areas in Maryland consisted of switchgrass (*Panicum virgatum*) meadows along the inner margins of tidal marshes (Stewart and Robbins 1958). In the Allegheny Mountains of Maryland, sedge wrens usually occupied sedge meadows in boreal bogs, whereas orchard grass pastures and hayfields were used at upland sites elsewhere in the state (Stewart and Robbins 1958).

At moist soil impoundments in Missouri, sedge wrens were associated with unflooded areas with rank or dense vegetative cover, and avoided short, sparse, or open vegetative cover, openings, and flooded areas (Fredrickson and Reid 1986). In a large, marsh complex in Wisconsin, Manci and Rusch (1988) observed sedge wrens primarily in habitats dominated by sedges (*Carex* spp.), and secondarily in upland grasslands. Sedge wrens avoided areas of deepwater cattail (*Typha* spp.), shallow-water cattail, dry cattail, and river bulrush (*Scirpus fluviatilis*).

In Minnesota, sedge wrens preferred dense sedge growth intermixed with patchy, 1-2 m high shrubs (Niemi and Hanowski 1984). Average vegetative characteristics of sedge wren territories in Minnesota were 303 sedge stems/m<sup>2</sup>, 16 forb stems/m<sup>2</sup>, 50 shrub stems/m<sup>2</sup>, and a predominant vegetation height of 1.1 m (Niemi 1985). Higher wren densities occurred in areas with medium shrub densities (11-32 stems/0.0025 ha) than in areas with more or less shrub growth (Niemi and Hanowski 1984). At another Minnesota site, major vegetation types in sedge wren territories were stands of great bulrush (*Scirpus acutus*)/

tussock sedge (*Carex stricta*)/water sedge (*C. aquatilis*) and monotypic stands of *C. aquatilis* (Burns 1982).

Sedge wrens at the Delta Marsh in Manitoba occurred in wet, white-top (*Scolochloa festucacea*)/sedge (*Carex atherodes*) meadows (Picman and Picman 1980). Crawford (1977) studied sedge wrens in a glacial marsh complex in Iowa, and found nests in drier sites around marshes in areas dominated by reed canary grass (*Phalaris arundinacea*) (17 nests) and river bulrush (14 nests) stands. Reproductive success was higher, however, for males with territories consisting primarily of bulrush (*Scirpus fluviatilis*) (Crawford 1977). Similarly, Walkinshaw (1935) considered the portion of a marsh intermediate between deepwater sections and the bordering meadow to be preferred habitats of sedge wrens in Michigan. Sedge wrens used areas where sedges and grasses predominated, with clumps of shrubs and herbaceous growth, and with no standing water (Walkinshaw 1935).

In a Nebraska study, sedge wrens avoided high ground and occurred most commonly along moist, ungrazed margins of relict channels (sloughs) where water sedge, common ragweed (*Ambrosia artemisiifolia*), and river bulrush predominated (Lingle and Bedell 1989). Sedge wrens in Lingle and Bedell's (1989) study nested at lower densities in grazed than ungrazed areas due to a reduction in vegetation stature caused by trampling and grazing by cattle. A southward-sloping hillside dominated by a mixture of rank grasses and weeds about 1-m high, interspersed with clumps of smooth sumac (*Rhus glabra*), supported a nesting population of sedge wrens during a particularly wet year in Kansas (Tordoff and Young 1951).

Little information is available on wintering habitats of sedge wrens. Brackish and freshwater sedge meadows and marshes are used, as are old fields and prairies with dense, matted grass or weeds (Howell 1932). Drier portions of salt marshes may be used during migration (Forbush 1929, Palmer 1949).

## BIOLOGY

### Reproduction

**Mobility.** - Sedge wren habitats are characterized by vegetation types and soil moisture regimes that are highly susceptible to drying or flooding caused by annual and seasonal variation in rainfall. Vegetative succession and disturbance by grazing, haying, and planting also impart a highly transitory character to sedge wren nesting habitats. This habitat instability may lead to the high mobility and low site tenacity of sedge wrens (Kroodsma and Verner 1978). The complete regional absence of sedge wrens in some years, however, suggests that some factor in addition to habitat instability is involved in population fluctuations, perhaps regional patterns of rainfall (Cody 1985) or weather conditions during migration. The nature of the sedge wren's communication system as a unique adaptation to high population mobility (Kroodsma and Verner 1978) suggests that opportunistic breeding and low site tenacity by sedge wrens has occurred for a long time, rather than being of recent origin, such as in response to agriculture and habitat loss (Burns 1982). Sedge wrens are "curiously local" and occur sporadically within the Andean Zone of South America, and there is no reason to believe that anthropogenic factors influence the species' distribution there (Fjeldsa 1988).

**Chronology.** - Nesting phenology of sedge wrens may be related to site latitude and often occurs in two "waves" during a given season. Sedge wrens may arrive at nesting areas as early as mid-April (Bent 1915) in some regions or as late as July in others (Lingle and Bedell 1989), and have been observed nesting as late as the end of September (Schwilling 1982). Nesting in many northeastern states is not initiated until July. In more northerly latitudes, nesting may be initiated from May to June, e.g., in North Dakota, Wisconsin, Minnesota, and Michigan, and again in late July through August, whereas in more southerly areas, e.g., Kansas and Arkansas, nesting may not be initiated until the latter period (Lingle and Bedell 1989). Late nesting at certain sites could represent renesting attempts by birds

arriving from elsewhere or could be an adaptive response of local birds to delayed availability of moist, but unflooded, grassland habitats. Peak nesting populations in mixed-grass prairie in North Dakota occurred after unusually wet springs, but nesting activity showed little response to spring temperature (Cody 1985). Occupancy of a given site varies among years, and sites used one year may be abandoned the next (Palmer 1949, Burns 1982).

**Courtship and Breeding Behavior.** - Upon arrival at nesting areas, male sedge wrens establish territories that are used for courtship, nesting, and foraging (Burns 1982). Territory boundaries are fluid throughout the nesting season, and males may shift their activity and defend new areas as the season progresses. Forty weekly estimates of territory size for five males in Minnesota averaged 1,280 m<sup>2</sup> (Burns 1982). Densities (males per 10 ha) of nesting sedge wrens have been reported as 4 in grazed areas and 12 in ungrazed areas in Nebraska (Lingle and Bedell 1989), 86 in Michigan (Walkinshaw 1935), 18 in Wisconsin (Manci and Rusch 1988), 12 in Illinois (Birkenholz 1984), 15 in North Dakota (Higgins et al. 1984), 19 in Minnesota (Hanowski and Niemi 1983), 3 in Kansas (Cink and Sepahi 1983), 7 in Iowa (Wilson 1983), and 3 in Maryland (Stewart and Robbins 1958).

Males defend territories and attract mates by singing vigorously throughout the breeding season, as much as 22 h/day and at rates of up to 12-15 songs/min (Walkinshaw 1935, Kroodsma and Verner 1978). Songs of sedge wrens have stereotyped introductions but have highly variable trills. This combination of song components may permit mixing of highly dispersive populations of sedge wrens (Kroodsma and Verner 1978) and may represent an evolutionary compromise between species identification and sexual selection among individuals. Because local dialects would be swamped in such highly mobile populations, neighboring male sedge wrens do not share song-type repertoires nor do they counter-sing by matching song-types (cf. marsh wrens).

**Nesting.** - Male *Cistothorus* wrens build multiple, domed nests that figure prominently in courtship, and may also serve as wren dormitories

and decoys for predators (Verner 1965, Picman and Picman 1980, Burns 1982). In Minnesota, male sedge wrens built an average of 7.4 complete nests and 0.8 incomplete nests on each territory (Burns 1982). Nests are well-hidden, rounded balls made of woven, fine grasses and culms of sedges, and are often built near the ground (< 0.5 m) and attached to live stems of grasses and sedges (Walkinshaw 1935, Tordoff and Young 1951, Harrison 1978). Nests typically have a single side-entrance. Nests used for incubating eggs are built closer to the ground than dummy nests (Walkinshaw 1935) and have a substantial inner lining of grass, sedge, and feathers added by the female (Burns 1982).

Females begin laying one egg daily about the third day of nest lining (Burns 1982), and initiate incubation before the clutch is complete. Usually 7 eggs are laid per clutch (range = 2-8), although clutches laid later in the season may be smaller than earlier clutches (Bent 1915, Harrison 1978, Burns 1982). Eggs are ovate or pointed-ovate, white and unmarked, and smooth and moderately glossy (Harrison 1978). Measurements of 50 eggs averaged 16 x 12 mm (Bent 1915). Incubation is by the female only and lasts about 14 days (Burns 1982).

Females in some populations of sedge wrens are double-brooded (Walkinshaw 1935, Burns 1982), but are single-brooded in others (Crawford 1977). Males may be serially or simultaneously polygynous and females may be serially polyandrous (Crawford 1977, Burns 1982). Mates of monogamous males had higher reproductive success than both primary and secondary females mated with polygynous males in Burns' (1982) study, whereas polygynous males had higher reproductive success than monogamous males.

Nest success (nests with at least 1 egg hatching) was reported at 68% ( $n = 31$  nests, Crawford 1977) and 69% ( $n = 18$  nests, Burns 1982). Sources of nest loss include predation, infertile clutches, heavy rains, trampling by grazing cattle, and nest destruction by other sedge wrens (Walkinshaw 1935, Crawford 1977, Picman and Picman 1980, Burns 1982). The nestling period lasts 13-14 days (Walkinshaw 1935, Crawford 1977). Fledging success (number of young fledged per number of eggs laid) has been reported at 0.67 in Minnesota (Burns 1982).

Virtually all parental care of nestlings is provided by the female, although males occasionally feed nestlings (Walkinshaw 1935, Burns 1982). Nest-building and continual defense by males of territories with abundant food may compensate females for a lack of male parental care of offspring.

**Fledgling and Migration.** - Fledglings move about in small groups until migration occurs in September or October, depending on the region (Bent 1915, Walkinshaw 1935). Incidental kills of sedge wrens at transmitting television towers in Florida suggest that most fall migration of sedge wrens occurs in October, well after the migration of most warblers, vireos, and other, similarly sized passerines is underway (Taylor et al. 1983). Clustering of multiple wren species at tower-kills indicates that sedge wrens migrate in flocks with other species of wrens (Taylor et al. 1983).

### Feeding

Information on the diet of sedge wrens is limited. Howell (1932) examined 34 stomachs of sedge wrens overwintering in Florida and concluded that the diet consisted wholly of spiders and insects, including ants, bugs, weevils, ladybird beetles, moths, caterpillars, locusts, crickets, and grasshoppers. Seven stomachs of sedge wrens from Pennsylvania contained beetles and spiders (Warren 1890). Walkinshaw (1935) observed sedge wrens feeding their young moths, spiders, mosquitos, flies, grasshoppers, and bugs.

The "mouse-like," inconspicuous foraging behavior of sedge wrens (Howell 1932, Walkinshaw 1935) suggests that they forage mainly at ground level, probably for insects hiding in moist soil and among bases of sedges and grasses. Particular adaptations of sedge wrens to foraging in shrubby grasslands include their small size, relatively long bills, and longer pelvis and legs in comparison to congeners using forested habitats (Niemi 1985). The abundance of insect food in upland habitats frequented by sedge wrens is probably a function of rainfall, soil moisture, and productivity of adjacent marshlands. Aquatic insects characteristically move from wetlands following metamorphosis and seek refuge on

nearby uplands (Orians 1980), where they may serve as an important food source for sedge wrens.

### POPULATION DISTRIBUTION, STATUS, AND TRENDS

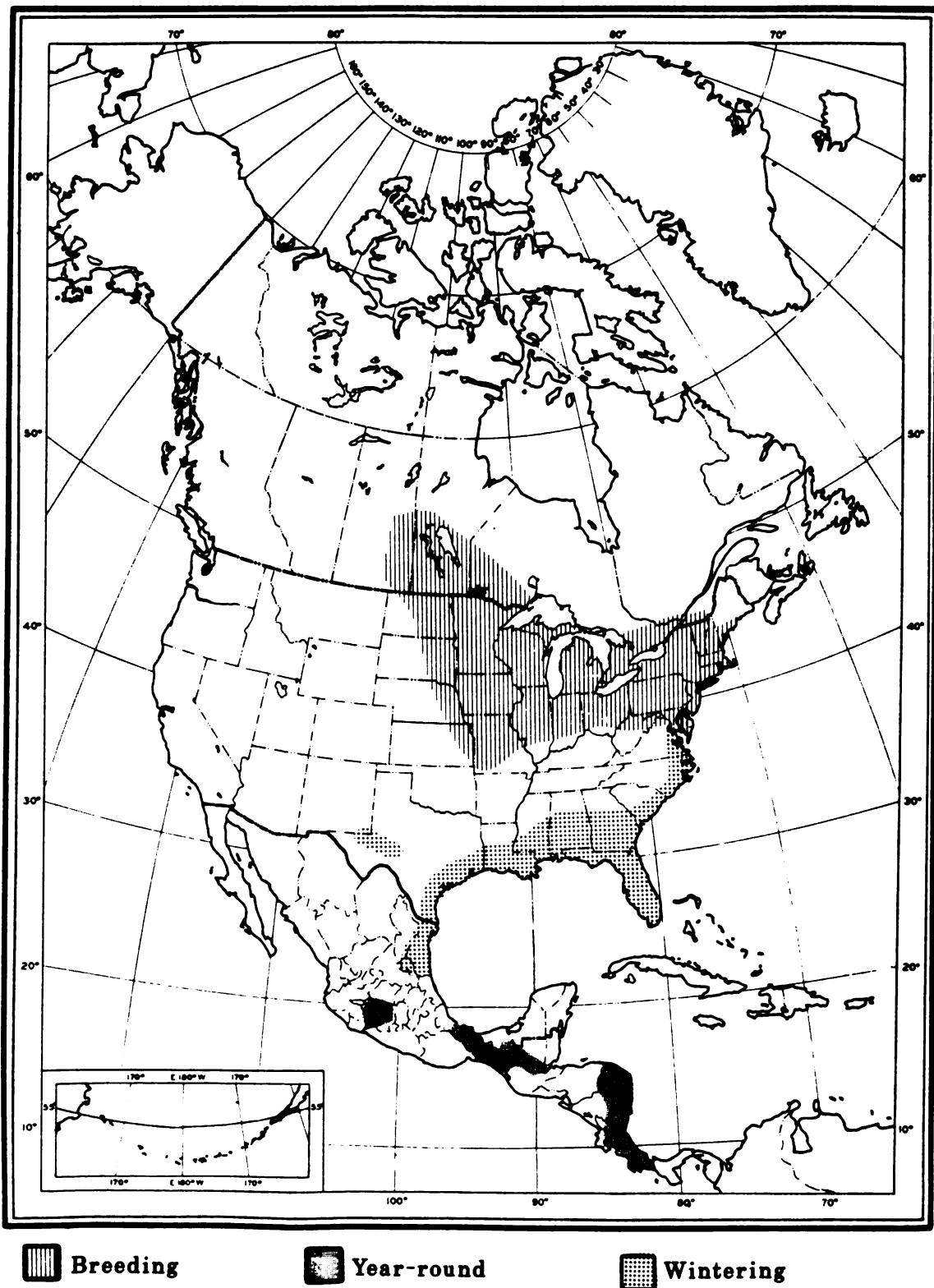
#### Distribution and Status in North America

The breeding range of the sedge wren extends from southeastern Saskatchewan, central Manitoba, southern Ontario, and central Maine to eastern Kansas, central Missouri, central Indiana and Virginia (Figure 1). Throughout its range, the sedge wren is sparsely and locally distributed. In the northeastern U.S., breeding occurs patchily in agricultural areas of western Vermont, northern and western New York, and western Pennsylvania, and in brackish marshes of the Delaware and Chesapeake Bays (Figure 2). The sedge wren has nested in small numbers in New Brunswick (Squires 1976) and possibly in Nova Scotia (Tufts 1986).

Data from U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; Robbins et al. 1986) indicate that sedge wren populations across the U.S. showed no significant trends from 1966-89 ( $n = 282$  routes), although more routes ( $P < 0.01$ ) had decreasing sedge wren populations (0.60) than increasing populations (0.38). In the eastern U.S., populations remained stable during the period 1966-89 ( $n = 224$  useable routes), although a significantly ( $P < 0.01$ ) greater proportion of routes had decreasing sedge wren populations (0.60) than increasing (0.36). In the Northeast Region (U.S. Fish and Wildlife Service Region 5), populations declined -1.47% annually from 1966-90 ( $P < 0.05$ ,  $n = 31$ ). Data were too few, however, to assess populations in any state in the northeastern U.S. (i.e., all states had  $\leq 10$  useable routes), except New York. Populations in the Northeast were quite sparse ( $\leq 0.02$  birds encountered/route in any state vs. a U.S. average of 0.87 birds/route).

The sedge wren was *blue-listed* by the National Audubon Society in 1979 and 1981 and

Figure 1. The North American range of the sedge wren (*Cistothorus platensis*) (American Ornithologists' Union 1983).

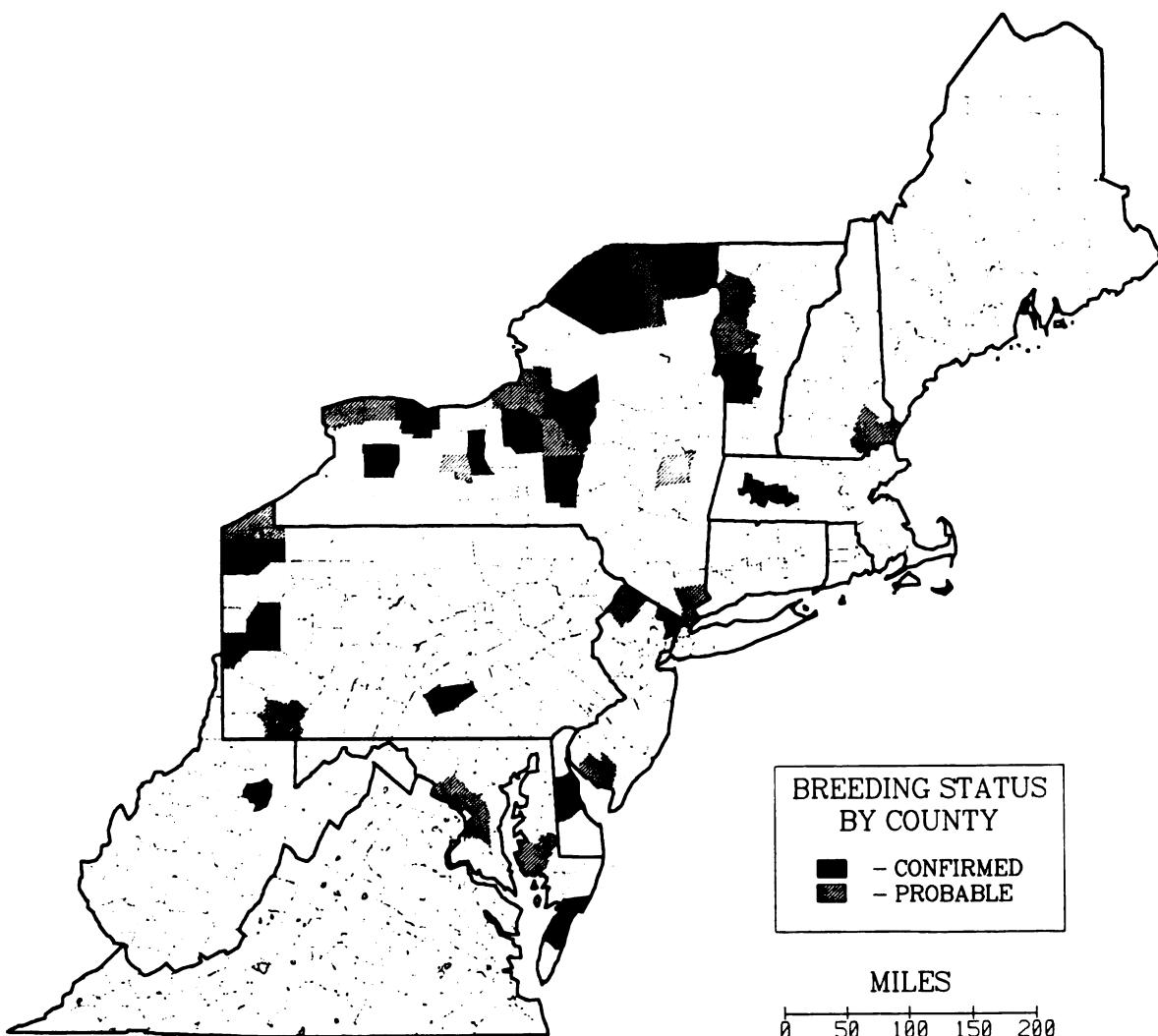


■ Breeding

■ Year-round

■ Wintering

**Figure 2.** Northeast breeding distribution of the sedge wren (*Cistothorus platensis*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andriele and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



considered a *species of special concern* from 1982-86 due to reports of depressed populations in the midwestern prairie region, northeastern Maritimes, Hudson-Delaware region, and Ontario (Tate 1986). Clean-farming of wet fields (i.e., establishment of crop monocultures) was suggested as a primary cause of declines in the midwest region (Tate and Tate 1982). The sedge wren has a Natural Heritage Program global rank of G5, i.e., demonstrably secure globally (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

Like breeding populations, overwintering sedge wrens may occur in disjunct populations that persist for only a year or two (Root 1988). Overwintering birds regularly occur, however, in regions below 150 m elevation with average January temperatures  $> 4^{\circ}$  C, such as along the Gulf coast of Texas and Louisiana and inland regions extending to the northeastern border of Louisiana. Other concentrations of overwintering birds occur along the Pecos River in western Texas, around and east of Pensacola, Florida, and in the Green Swamp and other marshy lowlands of North Carolina (Root 1988, Figure 1). Along the Atlantic coast, sedge wrens overwinter north to Maryland (Bull 1974). Evidence of a higher proportion of female than male sedge wrens at southerly locations among incidental kills of migrants at television transmitter towers (Taylor et al. 1983) suggests that males may occupy more northerly portions of the wintering range than females.

#### Distribution and Status in the Northeast Region

**Maine.** - Knight (1908) reported that sedge wrens were known to breed in Maine at a single location near Bangor. Palmer (1949) reported that since 1933, sedge wrens had been reported in small to fairly large breeding populations at about 21 localities in 10 counties, including up to 50 pairs in Milford in 1934. Sedge wren populations have evidently declined markedly in the state since the 1940s. Bond (1967) considered the sedge wren an occasional summer resident on Mount Desert Island, and stated that populations had declined there since 1940. Sedge wrens were not

encountered on any BBS routes in Maine during the period 1966-89, and no evidence of breeding was reported for several years prior to and during statewide surveys for the Maine Breeding Bird Atlas (BBA) (Adamus 1988). In 1989, however, two males, a female, a nest, and five young were located in a wet, abandoned pasture near Waterville (P. Vickery pers. comm., Maine Department of Inland Fisheries and Wildlife unpubl. data). Declines may be attributable in part to habitat loss: between the late 1960s-70s the availability of fields and pastures along a sample of woodcock (*Scolopax minor*) survey routes in Maine declined by 37% (Dwyer et al. 1983). Sedge wrens have a Natural Heritage Program state rank of SH (historical records only but suspected extant) in Maine, and are listed as *endangered* by the Maine Department of Inland Fisheries and Wildlife (Table 1).

Table 1. Summary of state listing status and Natural Heritage Program state ranks for the sedge wren (*Cistothorus platensis*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	endangered	SH
New Hampshire	endangered	S1
Vermont	threatened	SH
Massachusetts	endangered	SH
Rhode Island	-	SN
Connecticut	endangered	SH
New York	special concern	S2
Pennsylvania	threatened	S1
New Jersey	endangered	SH
Delaware	-	SH
Maryland	special concern	S2
Virginia	-	S2
West Virginia	-	S1

<sup>a</sup>SH = historical records only, but suspected extant

SN = regularly occurring but not breeding

S1 = critically imperiled;  $\leq 5$  occurrences

S2 = imperiled because of rarity; 6-20 occurrences

**New Hampshire.** - The sedge wren was first described as a rare and local summer resident of brackish marshes in southern New Hampshire (Allen 1903). Forbush (1929) considered the sedge wren a rare, local summer resident found chiefly in southern parts of the state. Between 1963-89, 14 breeding season reports were documented within the state, usually only 1-2 sightings/year. Recent, statewide breeding bird atlas surveys (1981-86) reported probable breeding at four locations scattered in the southern portion of the state (C. Foss pers. comm., Figure 2). Once considered numerous in the Concord area, the species now appears more rare and local than before, and its continued presence in New Hampshire is uncertain. Sedge wrens were not encountered on any BBS routes in New Hampshire during the period 1966-89. Population declines may be attributable in part to habitat loss: the availability of abandoned fields along woodcock survey routes in New Hampshire dropped by 65% in the 1970s alone (Dwyer et al. 1983). Sedge wrens have a Natural Heritage Program state rank of *SI* (critically imperiled) in New Hampshire, and are considered an *endangered* species by the New Hampshire Fish and Game Department (Table 1).

**Vermont.** - Forbush (1929) considered the sedge wren a rare, local summer resident in Vermont, occurring chiefly in the southern part of the state. The sedge wren is currently among the rarest breeding birds in Vermont. Surveys for the Vermont BBA found breeding activity at only two priority survey blocks (1% of total), although birds were found at seven additional locations in west-central Vermont during the breeding season (Laughlin and Kibbe 1985, Figure 2). Sedge wrens were not encountered on any BBS routes in Vermont during the period 1966-89.

Considerable, seemingly suitable, but currently unoccupied grassland occurs in the Champlain Lowlands and at other locations, and it is unclear why populations in the state are so sparse (Laughlin and Kibbe 1985). Because of the uncertain status of this small population, the sedge wren is considered a *threatened* species in Vermont by the Vermont Fish and Wildlife Department (Table 1). Sedge wrens have a Natural Heritage Program state rank of *SH* in

Vermont (historic records only, but suspected extant, Table 1).

**Massachusetts.** - The sedge wren in Massachusetts was first described as "not uncommon" in parts of the state by Howe and Allen (1901). Brewster (1906) considered the sedge wren a locally common summer resident in Massachusetts, but declining and absent from many former nesting areas. Forbush (1929) described the sedge wren as a local but "not rare" summer resident in Massachusetts. The sedge wren was considered "not really uncommon" and irregular in the Connecticut Valley but rare in the western hill-country by Bagg and Eliot (1937). In the Concord area, Griscom (1949) described the sedge wren as a "...formerly common summer resident in all sedgy meadows throughout the area," whose population had first declined due to succession of woody growth on meadowlands and had then been extirpated by heavy mortality during the severe winter of 1940. Griscom and Snyder (1955) characterized the sedge wren as a rare and local breeder, having formerly been a common summer resident at lower altitudes. Statewide breeding bird atlas surveys from 1974-79 found sedge wrens present during the breeding season at two locations in west-central Massachusetts (J. Baird pers. comm.). Sedge wrens were not encountered on any BBS routes in Massachusetts during the period 1966-89. Population declines are attributable in part to habitat loss: the availability of abandoned fields along woodcock survey routes in Massachusetts declined by 58% in the 1970s alone (Dwyer et al. 1983). Sedge wrens have a Natural Heritage Program state rank of *SH* in Massachusetts (historical records only but suspected to be extant), and are listed as *endangered* by the Massachusetts Division of Fisheries and Wildlife (Table 1).

**Rhode Island.** - Sedge wrens were considered to be rare summer residents in Rhode Island in the late 1800s (Howe and Sturtevant 1899). Forbush (1929) considered the sedge wren a local but not rare summer resident of the state. Sedge wrens were not encountered on any BBS routes in Rhode Island during the period 1966-89, nor during recent statewide surveys for a breeding bird

atlas in Rhode Island (R. Enser pers. comm.). There are no extant occurrences of breeding birds, and sedge wrens have a Natural Heritage Program state rank of SN in Rhode Island, i.e., a regularly occurring but nonbreeding species for which no effective habitat conservation measures can be taken (Table 1).

**Connecticut.** - In the late 1800s, the sedge wren was considered a rare breeder in Connecticut whose range was restricted to the southern border and the Connecticut Valley (Merriam 1877). Sage et al. (1913) also described the species as a rare summer resident in the state, although common in Litchfield County. Forbush (1929) considered the sedge wren a local, but not rare, summer resident in Connecticut, and Bagg and Eliot (1937) described it as irregular but "not rare" in the state. Population declines by 1900 due to habitat loss were evident to Bagg and Eliot (1937). Saunders (1950) concurred that the species was usually rare in southwestern Connecticut, but noted that "*considerable numbers*" appeared in the area during 1941-42. The marsh supporting sedge wrens was drained shortly after Saunders' (1950) observations were made. In the Guilford area, MacKenzie (1961) knew of no historical records of breeding activity by sedge wrens.

There are now no extant occurrences of sedge wrens in Connecticut (Bevier in press; BBA data 1982-86, Figure 2), although sporadic nesting attempts occurred during the 1970s (Zeranski and Baptist 1990). Sedge wrens have a Natural Heritage Program state rank of SH in Connecticut (historical records only, but suspected extant) and are listed as *endangered* by the Connecticut Department of Environmental Protection (Table 1), although Zeranski and Baptist (1990) consider sedge wrens to be extirpated from Connecticut as a breeding species. Loss of grassy and sedge bogs and wet meadows since the early 1900s is thought to have led to the species' extirpation. In the 1970s alone, availability of abandoned fields along woodcock survey routes in Connecticut declined by 52% (Dwyer et al. 1983). Sedge wrens were encountered on too few ( $n = 3$ ) BBS routes in Connecticut to detect population trends during the period 1966-89.

**New York.** - Probably never common in New York, the sedge wren is now considered a very rare to uncommon, local, and late-season (July-August) breeder in New York (Bull 1974, Andrie and Carroll 1988). Surveys for the New York BBA recorded sedge wrens during the breeding season at 57 survey blocks (1% of total: Andrie and Carroll 1988). Atlas surveys indicated that sedge wrens occurred most commonly below 152 m in elevation and along the Great Lakes and St. Lawrence plains (Andrie and Carroll 1988, Figure 2). Breeding records were "*scattered and sporadic*" and mostly absent from eastern, southeastern, and coastal portions of the state (Andrie and Carroll 1988). There was a notable lack of sedge wren sightings during Swift's (1987) extensive studies of the avifauna of Hudson River tidal marshes.

Sedge wrens were encountered on 17 BBS routes (average of 0.02 birds/route) in New York during the period 1966-89; population trends appeared stable, although a significantly larger proportion of routes had decreasing sedge wren populations (0.71) than increasing (0.29). Despite seemingly stable BBS trends, habitat loss seems to have led to local population declines in parts of the state. Bull (1974) attributed sedge wren declines in the New York City area to burning of brackish, coastal meadows and destruction of grassland breeding habitats for housing developments. Declines were also reported during the 1950s and 1960s in western New York (Beardslee and Mitchell 1965). Substantial, long-term declines in sedge wren populations have occurred in the Cayuga Lake region (Temple and Temple 1976), and it is doubtful that populations could ever recover their former abundance because nesting habitat at sedge marshes and wet meadows has been destroyed. Rates of wetland loss in the agricultural region of central New York, where the analysis of Temple and Temple (1976) was focused, have been perhaps the highest among all North Atlantic states (Jorde et al. 1989). Grazing, crop-planting, and invasion of wet meadows by shrub growth also are thought to have caused habitat loss (Andrie and Carroll 1988). During the 1970s alone, the availability of abandoned fields along woodcock survey routes in New York declined by 24% (Dwyer et al. 1983). Sedge wrens have a Natural Heritage Program state rank of S2 in New York (imperiled because

of rarity), and are listed as a *species of special concern* by the New York Department of Environmental Conservation (Table 1).

**Pennsylvania.** - Warren (1890) considered the sedge wren in Pennsylvania to be a regular but rare summer resident. Breeding populations were reported from Chester, Delaware, Lancaster, and Crawford counties, but were absent from mountainous districts (Warren 1890). Sutton (1928) also described the species as a rare and local summer resident. Todd (1940) reported the sedge wren from only a few localities in western Pennsylvania (Presque Isle, Conneaut Lake, and the Pymatuning Swamp), and believed that populations may then have been increasing in the region. Sedge wrens were considered rare summer residents of very local distribution and erratic occurrence by Poole (1964), who noted that populations in the state were in decline.

Breeding was scattered but widespread in Pennsylvania prior to 1950, but the sedge wren is now considered almost nonexistent as a breeder and is listed as a threatened species (Gill 1985, Table 1). Recent surveys for the Pennsylvania BBA found evidence of breeding at 13 locations, mostly in western portions of the state (Brauning in press, Figure 2). Loss of undisturbed grassland habitats due to changing agricultural practices is thought to have led to declines (Gill 1985). The availability of fields and pastures along woodcock survey routes in Pennsylvania declined by 30% in the 1970s alone (Dwyer et al. 1983). Sedge wrens were encountered on too few ( $n = 2$ ) BBS routes in Pennsylvania to detect population trends during the period 1966-89. Sedge wrens have a Natural Heritage Program state rank of SI in Pennsylvania (critically imperiled, Table 1).

**New Jersey.** - The sedge wren is a very rare breeder in New Jersey, with few, if any, extant breeding locations known (Leck 1984). The sedge wren was formerly a locally common summer resident in the northern half of New Jersey and at Cape May (Stone 1908, 1937). Breeding also occurred in short-grass meadows along the Delaware Bay shore and at wet meadows, marshes, and bogs inland (Leck 1984). A state breeding bird atlas project indicates a significant population decline has evidently occurred within

the state, with breeding activity currently restricted to just three counties in northern and southern parts of the state (D. Hughes pers. comm., Figure 2). Sedge wrens were not encountered on any BBS routes in New Jersey during the period 1966-89. Population declines are attributable in part to habitat loss. The availability of abandoned fields along woodcock survey routes in New Jersey dropped by 22% during the late 1960-70s (Dwyer et al. 1983). Sedge wrens have a Natural Heritage Program state rank of SH in New Jersey (historical records only, but suspected extant), and a state listing of *endangered* by the New Jersey Division of Fish, Game and Wildlife (Table 1).

**Delaware.** - Until the late 1970s, the sedge wren was a rare, irregular breeder in sedge marshes and wet meadows at widely scattered localities in Delaware (Harding and Harding 1980). Nesting was increasingly rare during the 1970s (one colony of 13 was reported in Sussex County in 1975), and surveys for the Delaware BBA (1983-87) found evidence of breeding at only one site (Hess et al. in press). Sedge wrens were encountered on too few ( $n = 2$ ) BBS routes in Delaware to detect populations trends during the period 1966-89. The sedge wren is thought to be nearly extirpated in the state, and presently has a Natural Heritage Program state rank of SH in Delaware (historical records only, Table 1).

**Maryland and the District of Columbia.** - Stewart and Robbins (1958) described the sedge wren as a common breeding bird in tidewater marshes along the southeastern reaches of Chesapeake Bay (Somerset, Wicomico, and Dorchester counties), uncommon in tidewater marshes along the Eastern Shore and Western Shore, uncommon and local in the Allegheny Mountain section, and rare and irregular in the Piedmont section and interior of the Western Shore. Sedge wrens were encountered on too few ( $n = 2$ ) BBS routes in Maryland to detect population trends during the period 1966-89. Recent surveys for the first breeding bird atlas of Maryland and the District of Columbia (G. Therres pers. comm.) found sedge wrens during the breeding season at 11 survey blocks (Figure 2). Sedge wrens have a Natural Heritage Program

state rank of S2 (imperiled because of rarity) in Maryland (Table 1), although much suitable habitat remains unsearched for sedge wrens and much prime breeding habitat already is protected. Because of their uncertain status sedge wrens are listed as a *species of special concern* by the Maryland Forest, Park and Wildlife Service (Table 1). Marsh fires during the breeding season are considered an important threat to sedge wrens in Maryland. Upland nesting populations are threatened by loss of old fields and pasturelands to vegetative succession and urbanization. Availability of abandoned fields along woodcock survey routes in Maryland dropped by 80% in the 1970s alone (Dwyer et al. 1983).

**Virginia.** - Although a common transient and winter resident along the coast, the sedge wren in Virginia is at the southern edge of its breeding range; recent breeding bird atlas data indicate the bird is currently a rare and local breeder of wet meadows and marshes along the coastal plain (S. Ridd and R. Wadja pers. comm.). A small colony was reported in Albemarle County in the Piedmont in the 1950s (Virginia Society of Ornithology 1979). Sedge wrens were not encountered on any BBS routes in Virginia during the period 1966-89. Sedge wrens have a Natural Heritage Program state rank of S2 in Virginia (imperiled because of rarity), but no special listing with the Virginia Department of Game and Inland Fisheries (Table 1). Despite supporting only small nesting populations, tidewater marshes in Virginia evidently represent important wintering areas and stopover sites for sedge wrens during migration.

**West Virginia.** - The sedge wren is considered an uncommon, very local summer resident in wet, grass-sedge meadows and hayfields in West Virginia (Hall 1983). Reports of summer residents have been widespread, originating from at least 10 counties in the state (Hall 1983). Although in the 1950s sedge wrens were common and regular summer residents in the grassy swamps of northern Canaan Valley, no definitive nesting records were reported by Hall (1983). Recent, statewide breeding bird atlas surveys, however, confirmed nesting at only one location and considered breeding possible at another (C.

Stihler pers. comm., Figure 2). Sedge wrens were not encountered on any BBS routes in West Virginia during the period 1966-89. Suitable habitat is limited for sedge wrens in the state, but not to the degree experienced by obligate swamp or marsh nesters. Sedge wrens have a Natural Heritage Program state rank of S1 in West Virginia (critically imperiled) (Table 1).

## Summary

The status of the sedge wren as a breeding species in the northeastern U.S. is precarious (Table 1). Breeding populations in all states are very sparse, sporadic, and local (or extirpated). Populations in all states have evidently undergone substantive declines during this century. Furthermore, the habitat base has deteriorated and continues to decline throughout the region.

Sedge wren populations in the northeastern U.S. may have waxed and waned over a 200-year period in response to changes wrought upon the landscape by humans. Prior to the arrival of Europeans, the East Coast was heavily forested and probably provided scant nesting habitat for sedge wrens, although the species likely bred in tidal marshes, sedge meadows along water courses, old beaver meadows, and sedgy areas of bogs and other wetlands throughout the region. In response to massive deforestation and the creation of fields and pasturelands during the late 1700s to mid-1800s, however, sedge wren populations probably expanded over much of the region. It is notable that sedge wrens were scarcely known north of Massachusetts and southern New Hampshire until the end of 1800s, but thereafter were reported with some regularity in northern Vermont, New Hampshire, Maine, and Quebec (Forbush 1929, Bagg and Eliot 1937, Palmer 1949), an expansion that may have followed an increase in the availability of abandoned farmland in northern regions. Habitat gains for the sedge wren were subsequently lost during this century due to drainage and filling of shallow wetland areas, loss of grassland communities to vegetative succession, and a shift from low-intensity agriculture, which required extensive areas of fields to be left fallow for 2-3 years, to high-

intensity agriculture, which keeps land in continuous cultivation.

Such a pattern of colonization and retreat has been exhibited by the sedge wren in Ontario (Cadman et al. 1987). Sedge wren populations are thought to have expanded into the province following the clearance of forests for agriculture > 150 years ago, and the current distribution of Ontario BBA records closely follows the distribution of abandoned farmlands. Also, breeding records are remarkably few from portions of the state that are intensively cultivated and subject to clean-farming (Cadman et al. 1987).

It is plausible that heavy mortality, of both migrating and overwintering birds, during the severe weather of 1940 reduced already diminished sedge wren populations in the Northeast to such low levels that populations were never able to recover (Griscom 1949, Griscom and Snyder 1955, Bond 1967). Although Griscom (1949) refers mainly to weather conditions in the southern U.S., climatological events in New England during 1940 included widespread "heavy to killing" frosts in late August, two hurricanes in September, an extremely heavy snowstorm in November, and severe cold during the first week of December (Noyes 1941), events that could have killed many late-breeding and migrating wrens in the northeastern states.

Although habitat loss seems to be a major factor in population declines, many authors have noted that sedge wren populations seem well below the level that available habitats could support (Palmer 1949, Laughlin and Kibbe 1985, Andrie and Carroll 1988). Thus, although regional sedge wren populations may now be stationary (as indicated by BBS trends), individuals may be too sparsely distributed to effect any substantive or rapid population recovery.

#### LIMITING FACTORS AND THREATS

Loss of nesting habitat may be the major cause of declines in sedge wren populations. About 4.75 million acres (1.92 million ha) of palustrine emergent wetlands, which include wet meadows important to nesting sedge wrens, were

lost in the U.S. between the mid-1950s and mid-1970s (Tiner 1984). Wetlands preferred by sedge wrens, such as sedge/grass meadows with moist or saturated soils, are the most easily drained and filled and are the wetland type most frequently destroyed by agriculture and urbanization (Tiner 1984). Substantial losses of grassland habitats in the northeastern region to urbanization and vegetative succession are evident. In just 10 years (late 1960s-70s) along woodcock survey routes in all northeastern states combined, the availability of abandoned and active fields declined by 23-25%, whereas the amount of land in young forest increased by 63% and that in urban/industrial uses increased by 33% (Dwyer et al. 1983).

Aside from direct habitat loss, habitat disturbance such as late-summer burning of tidewater marshes in Maryland and livestock grazing (Walkinshaw 1935, Lingle and Bedell 1989) are considered threats to nesting sedge wrens. Regional reductions of the water table due to extensive urbanization in the Northeast may prevent occurrence of water "ponding" in fields, which creates the wet-meadow habitat preferred by sedge wrens, and also may increase rates of vegetative succession.

Wetland loss often leads to drying processes on adjacent upland habitats, and may ruin nesting habitats for sedge wrens, which prefer to nest at moist upland margins of wetland patches. Severe weather conditions during winter could result in heavy mortality in sedge wrens and influence population levels (Griscom 1949). Considerable mortality of migrant sedge wrens has been observed at television towers throughout the eastern U.S. (Taylor et al. 1983). In the Argentine Pampas, overgrazing by cattle, soil compaction, and fires have eliminated tall grass habitats and have led to severe declines in sedge wren populations (Bucher and Nores 1988). No information is available concerning the extent that disease, parasites, predation, competition, or contaminants may limit sedge wren populations.

#### MANAGEMENT POTENTIAL

A lack of information about the reproductive potential and population structure of sedge wrens precludes making any definitive statement

concerning population management potential. The sedge wren's large clutch sizes (up to eight eggs) and potential ability to raise two broods per year (although unlikely in the Northeast), suggests high management potential. Because of their high mobility, sedge wrens can readily colonize new nesting areas. High mobility also confers high rates of genetic mixing upon populations, and thereby offsets some of the genetic problems facing the recovery of a severely diminished population. Sedge wrens will nest in a variety of grassland habitats, including wet meadows, marsh margins, bogs, and restored prairies (Higgins et al. 1984). Given an expanded and more stable habitat base and a long period without severe climatic events (hard winters or droughty summers), an increase in sedge wren populations in the northeastern states seems possible.

### LAND PROTECTION AND PRESERVE DESIGN

Lands preserved as sedge wren nesting habitats should have dense, tall growths of sedges and grasses. Wet meadows and grassy upland margins of ponds, marshes, streams, or rivers, hayfields and retired croplands, and brackish marshes represent high priority areas for protection. Other features of protected lands should include a scattering of shrubs and no standing water. Breeding sites should be protected from both drought conditions and flooding, and from fire, grazing, and human activity during the nesting season (May to August). Protected areas should be of sufficient size (e.g., > 5 ha) to reduce area-related effects on nest predation (Burger 1988). Preserves should be buffered against an influx of pesticides from adjacent agricultural areas. Wetland areas adjacent to nesting areas should be preserved to prevent drying of upland margins and to retain sources of insect prey. The U.S. Fish and Wildlife Service's responsibility for reviewing wildlife habitat attributes of properties foreclosed by the Farmers Home Administration (U.S. Department of Agriculture) presents an opportunity to acquire and preserve grassland habitats for sedge wrens in the northeastern states.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Most conventional survey programs for terrestrial birds are conducted in spring and completed by early summer (e.g., Breeding Bird Surveys). Therefore, sedge wrens may often be missed because, in many areas of the Northeast, nesting is not initiated until late June or July (e.g., Andrie and Carroll 1988). Efforts to survey sedge wren populations must extend throughout the summer months. Although sedge wrens are quite vocal and, if active, can be located using conventional aural census techniques, broadcast of recorded calls is useful for eliciting responses from inactive birds (Manci and Rusch 1988). A familiarity with sedge wren vocalizations is required of observers because, unlike marsh wrens, sedge wrens typically remain in dense cover and are difficult to identify visually. In areas with relatively dense, regularly occurring sedge wren populations (e.g., tidewater marshes of Delaware and Maryland), annual censuses should be conducted to monitor local populations. In other parts of the Northeast, where breeding activity is very scarce and sporadic, casual reports from birdwatchers may be the only feasible means of surveying populations. Birdwatchers should be actively encouraged to search for sedge wrens in old fields and along grassy, wetland margins throughout the summer months. Biologists employed by private consulting firms to inventory wetlands and farmlands associated with potential development sites may represent a valuable source of information on sedge wren occurrences. All reports of breeding activity should be verified and cataloged; this might best be accomplished by having a single agency responsible for overseeing survey efforts.

### Management Procedures and Programs

Preservation of nesting habitat for sedge wrens and maintenance of a complex of potential habitats throughout the Northeast is perhaps the most pressing management need. Particular attention should be directed toward safe-guarding nesting habitats at tidewater marshes in New

Jersey, Delaware, Maryland, and Virginia, which currently support the largest, most regularly occurring populations of sedge wrens in the northeastern U.S. Preservation of wet meadows and grassy margins of marshes, rivers, and other wetlands with historical evidence of sedge wren use is critical elsewhere. The integrity of wetland ecosystems, which serve as sources of insect prey and prevent drying of adjacent nesting areas on uplands, should be maintained. Flooding will destroy sedge wren nesting habitat as readily as drainage, and should be prevented during the nesting season. Overgrazing can easily eliminate the tall, rank cover on moist sites upon which sedge wrens depend.

Habitat preservation for sedge wrens in the Northeast will depend on management programs that focus specifically on the sedge wren, and protection and management of habitat for sedge wren populations on state and other public lands is suggested. Riparian-zone management strategies at state and federal wildlife refuges can be altered to preserve or create grasslands and wet meadows adjacent to waterfowl impoundments and other wetlands (e.g., as at Sunkhaze and Moosehorn National Wildlife Refuges, Maine (D. Pence pers. comm.)). Because of the sporadic and mobile nature of sedge wren nesting, conservation will also depend on voluntary protection of nesting areas by private landowners. Given the small and transitory nature of populations, however, management to protect specific sites may have a low probability of attracting and holding sedge wrens; rather, extensive efforts over wide geographic areas may be necessary to have significant effects on regional populations.

Livestock grazing influences the structure and composition of plant cover and the supply and availability of invertebrate foods for sedge wrens. Grazing animals should be excluded from meadows used by sedge wrens during the nesting season, although controlled grazing at other times of the year may be a means of managing meadowland habitats to benefit sedge wrens. Fire can be used to stop encroachment of woody plants at nesting areas and to regenerate stands of tall grasses and sedges that provide needed cover for sedge wrens. Prescribed burning should be restricted, however, to the nonbreeding season.

Protecting large habitat blocks (e.g., > 5 ha) can reduce area-related effects on predation of grassland-nesting birds (Burger 1988). Habitat blocks should be managed in rotation to ensure annual availability of nesting habitat. For a useful overview of management of grassland and agricultural ecosystems for nongame wildlife such as sedge wrens, see Ryan (1986).

### Research Needs

- Initiate standardized survey programs in areas with established sedge wren nesting populations (tidewater marshes of New Jersey, Delaware, Maryland, and Virginia) and coordinate and encourage birdwatchers and professional ornithologists to search for and record nesting activity elsewhere in the northeastern region throughout the summer season (May-August).
- Conduct detailed studies of the habitat requirements of sedge wrens in the Northeast to determine preferred soil moisture regimes, vegetation height, density, and composition, specific cover requirements for nesting, and minimum effective habitat area.
- Conduct a detailed study of the nesting biology of sedge wrens in the northeastern U.S. (most information currently comes from Midwest populations) to determine nesting phenologies, extent of polygamy, ability to renest, causes of reproductive failure, causes and rates of mortality in juveniles and adults, juvenile dispersal and philopatry, and diet. Such studies will probably be most successful if conducted at sites with relatively long-established, reliable use by breeding sedge wrens.
- Conduct detailed studies of relationships between habitat use by sedge wrens and abundance of invertebrate foods, including relationships between invertebrate populations and soil moisture, rainfall, wetland proximity, and grassland type.

- Identify major stopover sites and overwintering areas of the sedge wren in the northeastern and southeastern U.S.
- Determine effects of disease, parasites, predators, weather, and contaminants on sedge wren reproduction and populations, and monitor contaminant levels in sedge wrens and their eggs in industrialized and agricultural portions of the Northeast.

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# SEASIDE SPARROW

## *Ammodramus maritimus*

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The seaside sparrow occurs in relatively small, localized populations mostly confined to certain coastal saltmarshes within its range in the eastern U.S. The species has attracted the interest of systematists since the end of the nineteenth century, but serious field studies were not undertaken on any population until the mid-twentieth century. In the early 1960s, concern about the status of two east and south Florida populations focused renewed attention on the species. Currently, as wetlands around the nation are destroyed or disrupted by drainage and development activities, and those remaining are increasingly threatened, there is heightened concern for the welfare of all species dependent on these habitats. The seaside sparrow, as a maritime wetland specialist, represents a potentially valuable "indicator" of the continued ecological integrity of certain types of coastal marshes, and has already proven to be sensitive to habitat modification in the southeastern sections of its range. Populations of this sparrow in the Northeast are likely as susceptible to habitat disturbance and restriction as those now threatened or endangered in Florida.

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### DESCRIPTION

### Taxonomy

The seaside sparrow (*Ammodramus maritimus*) is distributed along the Atlantic and Gulf coasts from New Hampshire to Texas. Since its discovery by Wilson in 1811, various populations have been described and named, nine of which are currently accepted as subspecies of a single polytypic species (American Ornithologists' Union 1957, 1983). The taxonomic status of these populations has been controversial and the subject of several revisions and reviews (Chapman 1899;

Ridgway 1901; Griscom and Nichols 1920; Oberholser 1931; Hellmayr 1938; Griscom 1944, 1948; Austin 1983). Many of the populations are poorly differentiated though two at the southeastern extremities of the species' range in Florida (*A. m. nigrescens* and *mirabilis*) are quite distinct (Kale 1983). The latter were accorded full species status for a time (Hellmayr 1938, American Ornithologists' Union 1957).

Genetic (Avise and Nelson 1989) and zoogeographical (Funderburg and Quay 1983) evidence suggests that two phylogenetically distinct groups of races are isolated by the Florida peninsula. The Atlantic coast group, which is

relatively homogeneous genetically but not morphologically, includes the extinct race *A. m. nigrescens*. The genetic position of the endangered race *mirabilis* in south Florida is uncertain (Avise and Nelson 1989).

### Morphology and Plumages

Sexes are similar in this species, although males on average are larger than females (Post 1972). Birds of the northern, nominate race (*A. m. maritimus*) along the northcentral Atlantic coast are olive-gray above, with a stronger olive wash on the mid-back and laterally on the crown, and whitish below with indistinct gray streaks that are most prominent on the breast and sides. A whitish malar (cheek) streak and the white throat segregate a gray submalar stripe. An area in front of the eyes (supraloral) and the bend of the wings (wrist) are bright lemon yellow. Flight feathers are dusky. Primaries are edged with olive while some secondaries, tertials, and greater secondary coverts are fringed with rich russet. In fresh plumage, a cinnamon wash across the breast often produces an indistinct band. The tips of the tail feathers are attenuated, and the bill is elongate conical. Northern juveniles are browner than adults, with extensive blackish streaks dorsally. Ventrally they are whitish, more or less strongly tinged with ochraceous, and streaked with dusky (Dwight 1900, Ridgway 1901, Gruber 1955).

Variation in base color is marked in some populations and less so in others. Along the Atlantic Coast, northern birds are pale and monochromatic, while northeast Florida birds are monochromatic and quite dark (eg., *A. m. nigrescens* was very melanistic). The mid-Atlantic birds (*macgillivraii*) are polychromatic with pale, intermediate, and dark variants. A similar pattern of geographic variation exists in the Gulf Coast races. The palest birds (Texas) and darkest ones (northwest Florida) belong to monochromatic populations while a polychromatic population (*fisheri*) occupies the mid-Gulf area (Funderburg and Quay 1983). Isolated *mirabilis* is monochromatic and decidedly more olive than any other race.

The natal plumage of nestlings is quickly replaced by a complete prejuvenile molt. Contrary

to Dwight's (1900) assertion that there is a complete first prebasic molt, this molt is incomplete and involves nearly all feathers except the remiges and their coverts, and possibly the retrices (J. Greenlaw unpubl. data). Adults undergo a single, complete prebasic molt each year following breeding. There is no prealternate molt in the spring. Breeding aspect is acquired entirely by wear. By late summer, the body feathers are badly frayed leaving the birds faded and nondescript. In New York, some juveniles and adults begin the prebasic molt as early as late July, but most enter this molt by mid to late August, with completion by early October (J. Greenlaw unpubl. data).

### HABITAT

#### Breeding Season

The seaside sparrow is a habitat specialist that occupies coastal tidal marshes throughout its range (Kale 1983, Robbins 1983). One population (*A. m. mirabilis*) in Florida commonly occurs in freshwater *Muhlenbergia* (*M. filipes*, a tussock grass) prairie (Werner and Woolfenden 1983), and another near Charleston, South Carolina, evidently avoids the outer coastal marshes for breeding and uses brackish, more sheltered marshes away from the coast (Sprunt and Chamberlain 1970).

Northeastern sparrows occupy both high marsh (dominated by salt meadow vegetation including salt-meadow grass (*Spartina patens*), black-grass (*Juncus gerardi*), glasswort (*Salicornia* spp.), and marsh elder (*Iva frutescens*)), and low marsh (mainly various ecological forms of smooth cordgrass (*Spartina alterniflora*)) habitats (Woolfenden 1956; Post 1970a, 1970b, 1974; Reinert et al. 1981; Greenlaw 1983; Marshall and Reinert 1990). Descriptions of sparrow habitat elsewhere in the species' range can be found in Nicholson (1928, 1946), Tomkins (1941), Sprunt and Chamberlain (1970), Werner (1975), Sykes (1980), Post (1981a), Kale (1983), and Post et al. (1983), and Werner and Woolfenden (1983).

Throughout its range, this species exhibits a patchy or discontinuous distribution on local

marshes. The composition and physiognomic characteristics of marsh vegetation occupied by breeding sparrows are varied and reflect a certain behavioral opportunism in using available substrate (Greenlaw 1983, Post et al. 1983). Still, in New York, the general habitat response is predictable, as it appears to be elsewhere (Greenlaw 1983). Two biologically significant habitat characteristics evidently shared by most or all breeding populations are: (1) suitable elevated nest sites that offer protection from periodic tidal and storm-related flooding, and (2) nearby openings in the vegetation, or pool and ditch edges that permit access to the bases of rooted plants and open mud during foraging. Different microhabitats fulfill these divergent requirements for nesting and feeding. In low marshes in New York and New England, nests are commonly in areas of medium-height cordgrass (40-100 cm) growing densely enough to form a turf of partly clumped, semi-erect, persistent stems in the spring. Stands of dwarf cordgrass at or near mean high water level, and tall, open stands in the lower intertidal zone are avoided as nesting substrates. In high marshes, sparrows nest on *Iva*-dominated spoil deposits, or in *Iva*/salt meadow ecotones on the inner marsh, but they shun extensive areas of pure salt meadow grasses. Optimum habitat contains nesting and feeding microhabitats in close proximity, otherwise sparrows commute between a nest-centered territory and more distant undefended (but see DeRagon 1988) feeding areas (Tomkins 1941, Woolfenden 1956, Post 1974, Greenlaw 1983, Post et al. 1983, Marshall 1986, DeRagon 1988, Marshall and Reinert 1990).

#### Nonbreeding Season

Populations along the southeastern Atlantic and Gulf coasts are nonmigratory and continue to occupy their breeding marshes during the nonbreeding season. But, in some of these populations, there may be local or regional dispersal as birds respond to seasonal changes in food. Young sparrows near Charleston, South Carolina, leave the brackish, subcoastal, breeding marshes shortly after they are able to fly and move into the outer coastal marshes (Sprunt and Chamberlain 1970). In New York, post-breeding birds frequent the tall stands of cordgrass along

the bay edges where they harvest the rich supply of seed (J. Greenlaw pers. obs.). Beyond the fact that the birds remain in tidal marshes during the winter, little is known about the characteristics of the wintering habitat of northeastern sparrows.

#### Dispersion on Marshes

The patchy, uneven dispersion of breeding sparrows on marshes, and their absence from apparently suitable microhabitat in some areas, have led authors to describe the bird as "colonial" or "semicolonial" (e.g., Nicholson 1928, 1946; Stimson 1968; Werner 1975; Austin 1983). This is misleading since evidence suggests that clusters of territories simply reflect a common response to widespread patterns of temporal and spatial heterogeneity in saltmarsh vegetation (Post 1974, Greenlaw 1983, Post et al. 1983). The absence of breeding sparrows from seemingly suitable areas can be a simple consequence of low population size and poor recruitment.

### BIOLOGY

#### Social System

**Mating and space-use.** - Seaside sparrows are monogamous, territorial, and altricial. No cases of natural polygyny are known, although males can be induced to accept more than one mate (Greenlaw and Post 1985). Mates remain paired throughout the breeding season (Greenlaw and Post 1985). The territory is nest-centered and usually is enclosed within a larger, undefended home range (activity space) that includes additional feeding sites (Post 1974). Territory size varied in New York, depending on breeding density, from 0.01-0.9 ha, while overall home range size varied from 0.02-1.76 ha (Post 1974). Geographically, the average size of activity spaces ranged from 0.12 ha (New York) to 3.60 ha (Florida) (Werner 1975, Post et al. 1983). Space use varies between populations within and between regions (Post 1974, Werner 1975), but there is no evidence that any population is nonterritorial (Stimson 1968, Werner and Woolfenden 1983). The female alone builds the

nest, incubates eggs, and broods young. On average, males and females provide parental care about equally to dependent young (Post 1974, Post and Greenlaw 1982). Adults are highly philopatric, and some first-year birds in New York return to breed in their natal marshes (J. Greenlaw unpubl. data).

*Display repertoire and song.* - The display repertoire of seaside sparrows has been examined in New Jersey (Woolfenden 1956), Florida (Werner 1975, McDonald 1983, Werner and Woolfenden 1983), and New York (Post and Greenlaw 1975). Repertoire composition appears to be very similar in all these populations. Northern sparrows employ 14 visual displays and 15 vocal displays in their social system (Post and Greenlaw 1975, J. Greenlaw unpubl. data). The male's primary song is short (about 1 second in length) and low-pitched (Borror 1961, Post and Greenlaw 1975). Song structure varies in some details between populations, but not markedly (Hardy 1983, McDonald 1983, Werner and Woolfenden 1983). In the Northeast, song begins with a brief series of sharp notes, and sometimes a short trill phrase, followed by a longer, buzzy trill that is highly frequency modulated. This wheezy, unmusical song and associated behavior commonly receive incidental attention in general accounts of the species (e.g., Howell 1924, 1932; Forbush 1929; Stone 1937; Saunders 1951; Bull 1974; Lowery 1974; Peterson 1980). McDonald's (1989b) experimental study in Florida confirmed that primary song contains information that permits the male to establish and hold a territory (agonistic function) and to attract and retain a mate (sexual function). The male also performs a towering flight display that incorporates a complex song vocalization (Post and Greenlaw 1975).

### Breeding Phenology and Chronology

Median date of first egg-laying varies between years at given localities, depending on earliness of the season. In New York, first eggs usually appeared in nests from 13-16 May (J. Greenlaw unpubl. data), in Rhode Island about 23 May (DeRagon 1984), and in Massachusetts from

25-26 May (Marshall and Reinert 1990). The average date of initiation of first clutches in New York was 19 May, whereas it was 20 July for last clutches (Post et al. 1983). Egg dates ranged seasonally from 17 May-25 July in New York (J. Greenlaw unpubl. data), from 27 May-21 August in Rhode Island (DeRagon 1984), and from 25 May-30 July in Massachusetts (Marshall and Reinert 1990).

Incubation begins with the laying of the last or next to the last egg (J. Greenlaw unpubl. data). Incubation period varies from 11-14 days (mean = 12.2 days in New York, 12.4 days in Massachusetts) (Worth 1972, Marshall and Reinert 1990, J. Greenlaw unpubl. data). Adults continue to feed young out of the nest for an additional 20 days (DeRagon 1988, J. Greenlaw unpubl. data). In New York, nestlings started appearing in the population as early as the end of May and the first few days of June, while nests containing young occurred as late as 14 August (J. Greenlaw unpubl. data). The length of the nesting cycle from start of nest construction to fledging averaged 28.7 days (range = 27-30 days) (Marshall and Reinert 1990).

Two broods are reared successfully by some pairs in New York (J. Greenlaw unpubl. data), but Marshall (1986) felt that only one brood was attempted in Massachusetts. In New York, the interval between renesting and nest failure was 5.5 days (Post et al. 1983), and in Massachusetts, it was 6.0 days (range = 4-8 days) (Marshall and Reinert 1990). The interval in a Florida population was 7.6 days (Post et al. 1983). The time from fledging to the initiation of a new clutch on Long Island was 17.5 days (J. Greenlaw unpubl. data). Breeding season length varied between years from 67-88 days, averaging 76.8 days in New York (Post and Greenlaw 1982). In contrast, total season length averaged 96 days in a Florida population (Post et al. 1983).

### Nests and Nest Sites

Nests are either open cups or partly domed with a lateral entrance. They are typically elevated high enough in suitable vegetation to minimize the problem of normal flooding and low enough to be sheltered from predators and

weather (Woolfenden 1956, Greenlaw 1983, Post et al. 1983, DeRagon 1988, Marshall and Reinert 1990). In New York, mid-summer nests suspended in new-growth cordgrass averaged 19.0 cm above the mud (Post 1974). Early nests are typically placed in clumps of residual cordgrass, but later nests are in the vegetation column between erect, live culms of cordgrass (Post 1974, Marshall and Reinert 1990). In the latter case, the tops of the grasses are often pulled over the nest to form a canopy (J. Greenlaw pers. obs.). Occasionally, nests are placed 1-4 m above the ground in a shrub (usually *Iva* spp. in the Northeast) or small tree (Arnow 1906, Woolfenden 1956, Marshall 1986, J. Greenlaw unpubl. data). In Florida, the activity of predatory rats influences nest site use by sparrows (Post 1981b).

### Clutch Size

Overall, clutch size in northeastern populations varies from 3-6 eggs. Two-egg clutches are very rare and are perhaps incomplete (Post and Greenlaw 1982, Post et al. 1983, Marshall and Reinert 1990). Mean clutch size in populations from New Jersey to Massachusetts was 3.7 eggs (Woolfenden 1956, Post and Greenlaw 1982, Marshall 1986). Average clutch size varies seasonally. Modal clutch size on Long Island in early nests was 4 eggs, while it was 3 eggs in later nests (J. Greenlaw unpubl. data). Clutch size also averages larger in northern than in southern populations; mean clutch size in two Florida populations was 3.2 eggs, about 0.5 eggs smaller than clutches of northeastern sparrows (Post et al. 1983, Werner and Woolfenden 1983).

### Breeding Success and Productivity

Reproductive success in this species has been studied in New York (Post 1972, 1974; Post and Greenlaw 1982; Post et al. 1983), Massachusetts (Marshall 1986, Marshall and Reinert 1990), and Florida (Post et al. 1983). Nest mortality is high in all populations, but especially in Florida. The average (all 2-year averages) sparrow egg had a 34.4% chance to become a fledgling in New York (Post and Greenlaw 1982), a 32.4% chance in

Massachusetts (Marshall and Reinert 1990), and only a 3% chance in Florida (Post et al. 1983). However, there is considerable variation between years in average breeding success within a region. In New York, the extremes were 19.8-47.7%, and in Massachusetts 22.1-42.6%. These differences reflected variation in predation (New York) and flooding risks, which are the main causes of nest failure in the Northeast (Post et al. 1983, Marshall and Reinert 1990).

Occasional storm-driven high tides and heavy rain in New York, and monthly spring high tides in Massachusetts, sometimes caused catastrophic mortality in poorly elevated nests still active at the time of flooding. Sparrows suffering nest loss responded by quickly renesting. The first egg in the replacement nest is laid as early as 3-4 days following destruction of a nest (Marshall and Reinert 1990, J. Greenlaw unpubl. data).

Habitat differences in overall breeding success are evident as well. Post (1972, 1974) found in New York that 47.0% of sparrow nests in a low marsh fledged young, while in a high marsh nearby, the average was 66.1%.

Predation was low (about 11%) in Massachusetts (Marshall and Reinert 1990), but it was an important secondary source of mortality in New York (Post et al. 1983), and the primary cause of nest loss in Florida (Post 1981a,b; Post et al. 1983). The rice rat (*Oryzomys palustris*) took more eggs than young in Florida, but the reverse was true in New York where Norway rats (*Rattus norvegicus*) are significant predators (Post et al. 1983).

In New York, annual productivity averaged 4.41 young/female and ranged between 3.38-5.57 young/female in different years. The only comparative data are for a Florida population for which productivity was 0.58 young/female/year (Post and Greenlaw 1982, Post et al. 1983).

### Breeding Density

Population sizes can vary from one or two territorial males isolated on a marsh to many dozens of males on contiguous or overlapping activity spaces (Post 1974, Greenlaw 1983, Post et al. 1983). Only a few studies supply information on breeding densities in the Northeast. With one exception on Long Island, densities ranged from

64-214 singing males/km<sup>2</sup> on marshes from Maryland to New York (Post 1970b, Robbins 1983). Post (1970a) reported an exceptionally high density of 2,000 males/km<sup>2</sup> on an unaltered low marsh on Long Island. On the same marsh several years later, Greenlaw (1983:107) found 982 males/km<sup>2</sup>. The two latter values are ecological densities (unsuitable habitat and a tidal pool were excluded from calculations), while most or all of the values reported elsewhere (see Robbins 1983 for summary) are crude densities. Apart from this, Post's (1970a) value may be exceptionally high because his survey focused on a relatively small area of the marsh (2.75 ha) that contained a dense cluster of territories (see Post 1974).

Breeding densities in New England varied from 2-114 males/km<sup>2</sup> (Reinert et al. 1981, Marshall 1986, DeRagon 1988, Marshall and Reinert 1990). The mean density in the region for all types was 30.1 males/km<sup>2</sup>. These are all crude densities.

Ditched and unaltered marshes usually support different densities of sparrows, with highest densities in the latter habitat (Post 1970a,b, 1974; Reinert et al. 1981; Greenlaw 1983; DeRagon 1988). In Rhode Island, mean density was 14 males/km<sup>2</sup> and 55 males/km<sup>2</sup> in ditched and unditchered marshes, respectively (DeRagon 1988). On a Gulf Coast marsh in Florida, breeding densities in the race *peninsulæ* varied from 158-260 males/km<sup>2</sup> between 1980-89 (Post 1981a; McDonald 1982, 1983b, 1984, 1989a, 1990).

### Survival and Replacement Rate

Based on cumulative return rates, adult survival was estimated to be 57-60% for a population on an unaltered low marsh in New York (Post et al. 1983). Two Florida populations had minimum return rates for adults of 85.7% (*peninsulæ*) (Post et al. 1983) and 88% (*mirabilis*) (Werner 1975). Post-fledging survival to independence of birds banded as nestlings in New York was 36% (Post and Greenlaw 1982).

The estimated lifetime reproductive output of an average female (replacement rate) of 2.72 (2-year average) in a New York population suggests that the population was increasing (Post et al. 1983). As noted, this population exhibited

an exceptionally high breeding density (Post 1970a). In Florida, the replacement rate for *A. m. peninsulæ* averaged 1.11, indicating that the population was just maintaining itself. These populations were in low marsh habitats; no similar data are available for high marshes.

### Feeding Behavior

Northern seaside sparrows forage mainly in open stands of cordgrass, along bay or marsh edges, on patches of wrack, along the edges of pools and ditches, and in muddy *Salicornia* spp. pannes (Woolfenden 1956, Post 1974, Merriam 1979, Delaney and Mosher 1983, Greenlaw 1983, Post et al. 1983). They obtain their arthropod prey by either walking on the marsh substrate, or by climbing through the matrix of vegetation above the ground. From the ground, they glean insects from vegetation by stretching their neck, lunging, or chasing, and they probe or peck mud and water surfaces. They also wade into shallow water. Only rarely do they hover or flycatch. Above the ground, sparrows peck at vegetation and snap at flying insects (Post et al. 1983, J. Greenlaw pers. obs.). Both the vegetation column and the marsh substrate are significant sources of food for sparrows in northern marshes, but only the vegetation is important to birds in Florida (Post et al. 1983).

### Diet

There is little quantitative information on the diets of adults in this species. Judd (1901) found that about 70% of the food consumed consisted of arthropods, mainly insects and spiders, while the balance was seeds of marsh plants. Martin et al. (1951) reported that 94%, 100%, and 40% of the spring, summer, and fall diets, respectively, were comprised of invertebrates. The following invertebrate taxa have been found in the adult diet: Annelida (marine worms), Gastropoda (small snails), Decapoda (small crabs), Amphipoda (sand fleas), Araneida (spiders), Homoptera (leafhoppers), Hemiptera (true bugs), Diptera (flies, adults and larvae), Lepidoptera (moths), Orthoptera (crickets and grasshoppers), Odonata (dragonflies), and

Hymenoptera (wasps) (Judd 1901, Howell 1924, Obersholser 1938, Martin et al. 1951, Sprunt 1968). In the north, the seeds of *Spartina alterniflora* are used heavily by post-breeding birds before migration (J. Greenlaw pers. obs.).

The diets of nestling seaside sparrows in the Northeast are much better known (Merriam 1979, 1983; Post et al. 1983). In a low marsh in New York, invertebrate from at least 38 taxa were fed to nestlings, while in a neighboring high marsh, invertebrates from 25 groups were provided by the adults. The major taxa represented were Insecta (at least 37 families), Araneida (5 families), Acari, Pseudoscorpionida, Amphipoda, Isopoda, and Mollusca (Merriam 1979). Diptera were the most important food for nestlings in low and high marshes. In the ditched high marsh, the tabanid flies, *Tabanus nigrovittata* and *Chrysops* spp., constituted 71% of the overall diet, while in the unaltered low marsh, tabanids, stratiomyid flies (especially *Odontomyia microstomata*), and noctuid and pyralid moths made up 70% of the diet. Mirids (Hemiptera) also were consumed in large numbers but comprised relatively little bulk (Merriam 1979).

Nestling dietary composition changes seasonally to reflect available stocks of invertebrates (Merriam 1983, Post et al. 1983). Mud-inhabiting prey groups (e.g., stratiomyid and dipteran larvae) were taken in proportion to availability in the mud, while some prey groups in the vegetation (Diptera, Lepidoptera, Araneida) were exploited disproportionately to their availability (Merriam 1979).

### Migration

The northern nominate race is largely migratory, while most southern races are resident. Winter departures occur in populations at least south to North Carolina (Tomkins 1941).

Fall departures in the Northeast span a period from mid-September to at least mid-November, when autumn movement is essentially complete. The fall peak of migration on Long Island occurs in mid-October (Cruickshank 1942; Bull 1964, 1974). A few individuals are usually detected in early winter on Christmas Bird Counts (CBCs) as far north as Cape Cod (Hill 1965), but in most winters lingering birds in New York and

southern New England probably leave later or succumb since marshes generally freeze over during much of January and February. The relative incidence of birds detected in CBC areas along the Atlantic and Gulf coasts was evaluated by Robbins (1983) and Root (1988).

In some years, sparrows in New York return to breeding marshes as early as mid-April, but the usual arrival time for vanguard birds in this region is the last week of April (Bull 1964, J. Greenlaw unpubl. data). First arrival in southern New Jersey is also late April (Stone 1937, Woolfenden 1956). Early birds in Massachusetts appear during the first week of May (Marshall 1986). Males arrive ahead of females, and older males before 1-year-old males (J. Greenlaw unpubl. data). The median dates of spring arrival for all sparrows in a Long Island low marsh was 5 May (early season) and 18 May (typical season) (J. Greenlaw unpubl. data). Based on 30 years of records in Connecticut, Saunders (in Robbins 1983) found the median date of spring arrival to be 18 May.

### Winter Biology

Remarkably little is known about the behavior and ecology of northern sparrows on their wintering grounds. Winter resident birds begin arriving in Georgia and Florida as early as 3-4 October (Robbins 1983). There are no winter recoveries of sparrows banded on breeding areas in the Northeast, so it is unknown where most birds from different northern populations spend their winters. Overall, it seems that Atlantic birds winter mostly from the central South Carolina coast south to northeastern Florida (Robbins 1983).

Burleigh (1958) commented that northern seaside sparrows in Georgia confined their activity to dense saltmarsh grasses where they were quiet and inconspicuous. In the Southeast, they mingle with the resident birds.

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

The last species-wide assessment of the breeding distribution of the seaside sparrow was Robbins (1983). The distribution and status of Florida populations have been examined by Kale (1978, 1983), Kushlan et al. (1982), Kushlan and Bass (1983), and Werner and Woolfenden (1983).

The seaside sparrow breeds from Massachusetts south along the coast to the St. John's River in northcentral Florida, and from Tampa Bay on the Gulf Coast of Florida to Corpus Christi, Texas (American Ornithologists' Union 1983). The northernmost population was known at Plum Island in northeastern Massachusetts (American Ornithologists' Union 1957, Robbins 1983), but recently breeding birds were discovered at Hampton, New Hampshire (C. Foss pers. comm.). It is unlikely that summering populations can become established much farther north since the physiography of tidal marshes changes in southern Maine and habitat becomes unsuitable (Norton 1897, Montagna 1942, J. Greenlaw pers. obs.).

Overall, the seaside sparrow within its North American range is secure at the moment, but populations are few in some regions, especially in parts of Florida and in New England (Figure 1). The Nature Conservancy rates it globally as a G4 species, one level below the most secure designation (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia). Indeed, pooled Breeding Bird Survey (BBS) data from four states (Florida, Delaware, Maryland, and New Jersey) suggested that populations along routes that sample the species have been increasing at an annual (median) rate of about 4% from 1966-89. The rate of increase in the individual three northeastern states ranged between 1.9% (Maryland) to 7.9% (New Jersey) over the same period and were statistically significant (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). Across the Northeast, BBS data indicate an annual increase of 2.14% from 1966-90 ( $P < 0.01$ ), but these data must be treated with some caution, since only a

few seaside sparrow populations are censused by BBS routes ( $n = 13$  routes). The Office of Migratory Bird Management considers the seaside sparrow a *migratory nongame bird of management concern* (U.S. Fish and Wildlife Service 1987).

The vulnerability of this species varies radically in different regions. Several states classify the status of the species or some of its races within their respective borders as either S1 or S2, the two highest classes of vulnerability (Table 1). Two Florida populations (races) are

**Table 1.** Summary of state listing status and Natural Heritage Program state ranks for the seaside sparrow (*Ammodramus maritimus*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	SN
New Hampshire	-	S1
Vermont	-	-
Massachusetts	-	S2
Rhode Island	special concern	S2
Connecticut	special concern <sup>b</sup>	S3
New York	-	S3
Pennsylvania	-	SA
New Jersey	-	S4
Delaware	-	S4
Maryland	-	S5
Virginia	-	S4
West Virginia	-	-

<sup>a</sup>SN = regularly occurring but not breeding

SA = accidental

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

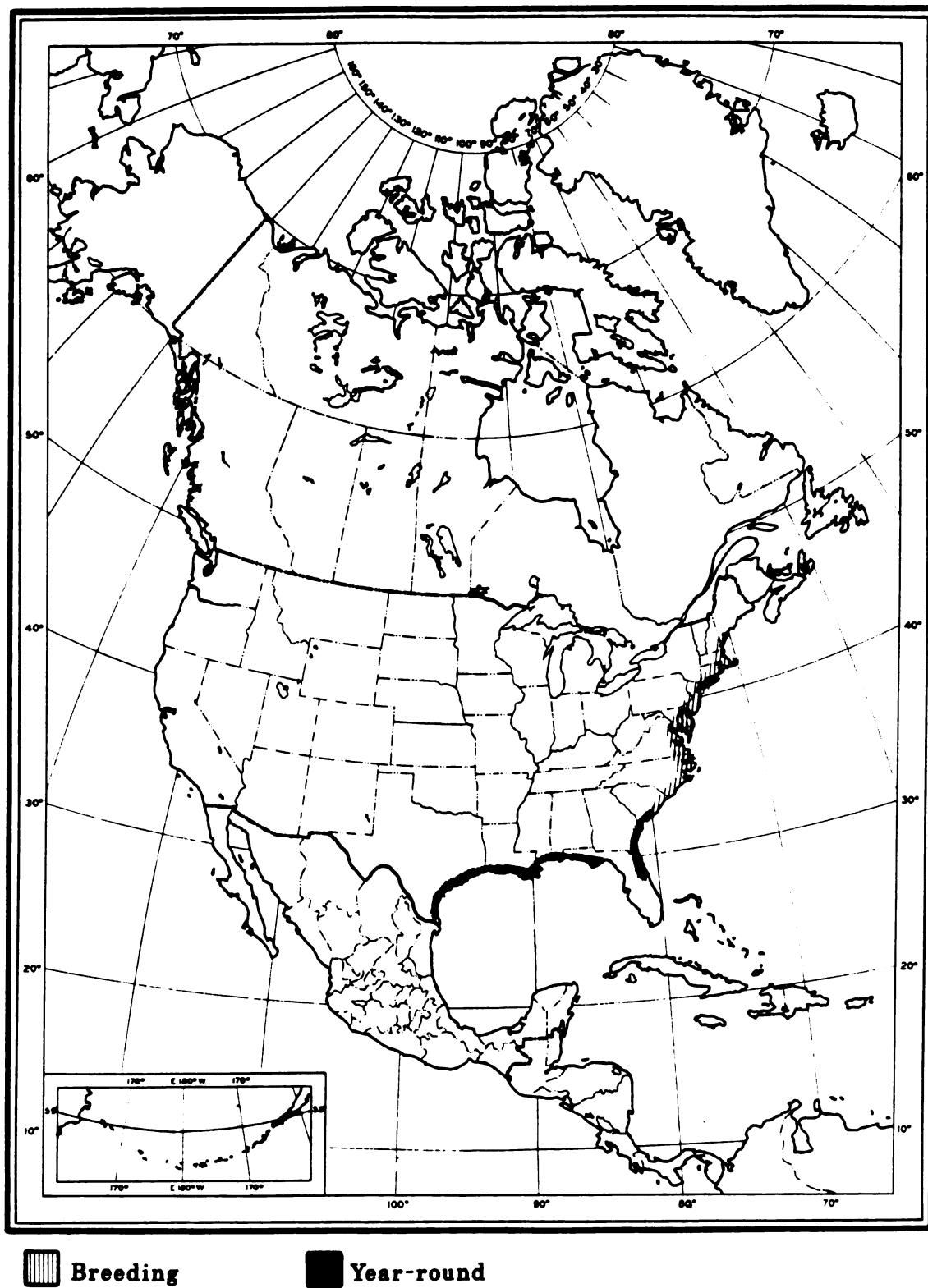
S4 = apparently secure

S5 = demonstrably secure

<sup>b</sup>Proposed

listed as endangered, and one of these (*A. m. nigrescens*) has since been extirpated. Although there is some question about the current status (and even the taxonomic validity) of *pelonotus* in

Figure 1. The North American range of the seaside sparrow (*Ammodramus maritimus*) (American Ornithologists' Union 1983, Robbins 1983).



northeastern Florida, Kale (1983) thought it likely that it too is now extinct. In contrast, except for New England, northeastern states have relatively healthy populations.

### Distribution and Status in the Northeast Region

Seaside sparrows breed from New Hampshire to Virginia (Figure 2). The data, presented by county, were made available from breeding bird atlases (BBAs) in the respective states.

**Maine.** - The species is accidental in Maine (Palmer 1949) and has a state Natural Heritage rank of SN (regularly occurring but not breeding) (Table 1). As noted earlier, suitable breeding habitat evidently does not occur in southern Maine where it otherwise might be expected (J. Greenlaw pers. obs.).

**New Hampshire.** - The seaside sparrow is at the northern edge of its range in this state. A single small population was discovered and breeding confirmed by G. Gavutis in July 1985 at Hampton (Rockingham County). The birds arrive there in late May and leave in September. Territories are contiguous in a wet, poorly drained section of the Hampton marsh (C. Foss pers. comm.).

The species is a relatively recent breeding immigrant in New Hampshire. Coues' (1868) early comment that it occurred abundantly north to New Hampshire was in error (Stearns 1881). Forbush (1929) regarded it as accidental in the state, but it had colonized certain marshes on the nearby coast of northeastern Massachusetts by 1949 (Griscom and Snyder 1955).

The state Natural Heritage Program has given the species its most vulnerable ranking (S1). G. Gavutis considered the status of the Hampton population to be very precarious, consisting of only 6-8 pairs when discovered, and possibly occupying a single marsh in the state.

**Vermont.** - Seaside sparrows are not found in Vermont.

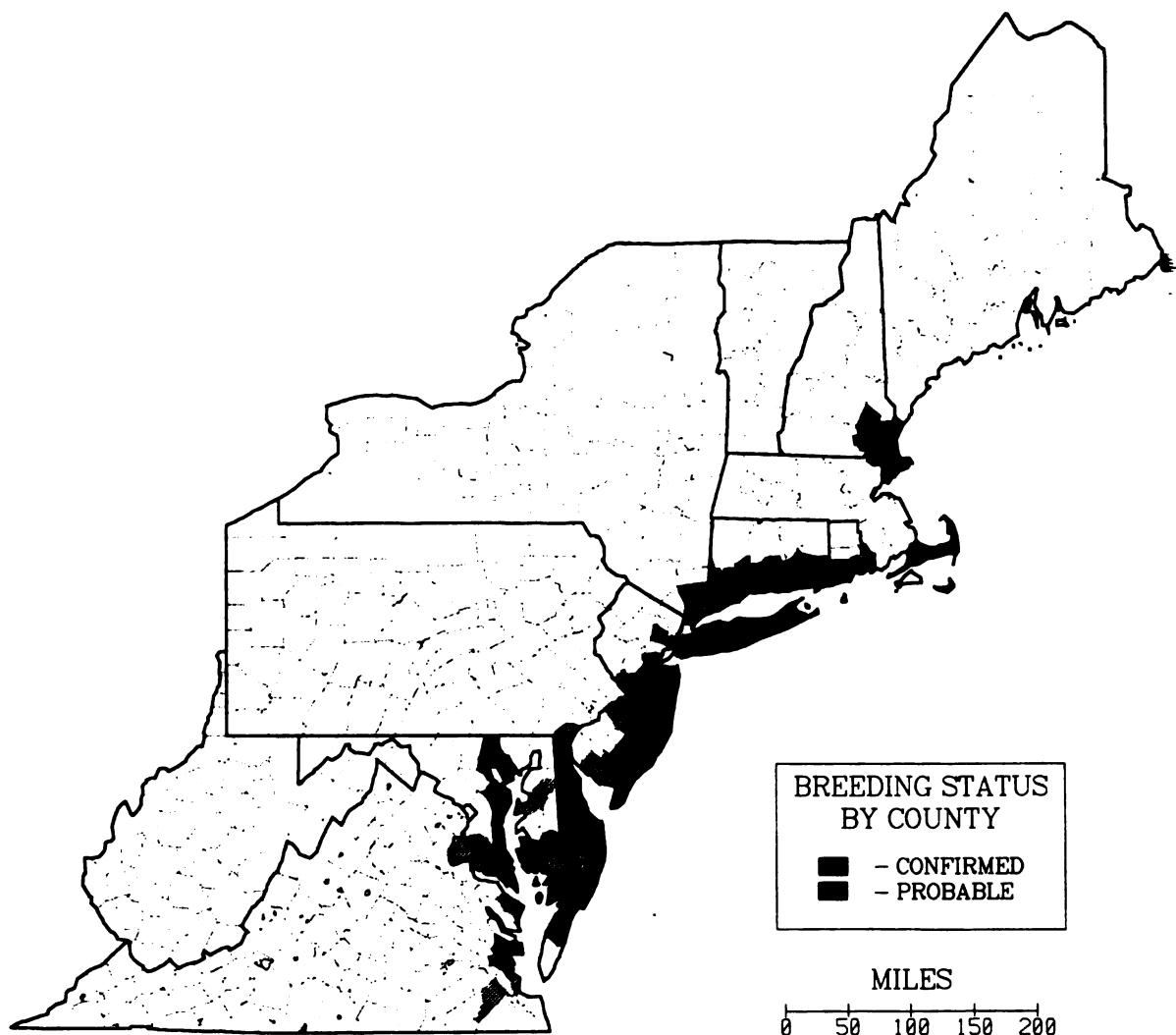
**Massachusetts.** - Seaside sparrows in this state are near the northernmost outpost of their breeding range. Presently, they are an uncommon and very sparsely distributed nester. Although Hill (1965) considered the bird a "permanent resident" on some Cape Cod marshes, it should be judged a summer resident overall in the state. Some individuals attempt to overwinter in the Cape area (Forbush 1929, Hill 1965) as they do elsewhere in the Northeast. Evidently, overwintering was fairly regular in the past at Barnstable (Griscom and Snyder 1955).

In 1981, R. Forster (cited by Stoll and Golet 1983, F. Golet pers. comm.) knew of only five marshes in Massachusetts that had breeding seaside sparrows (Westport Point, South Westport, Chatham-Monomoy, Barnstable, and Newburyport). The state BBA documented the presence of summering birds in two different regions of the state (J. Baird pers. comm.). One occurs along the southeast coast from the Westport-Dartmouth area (Bristol County) east to Monomoy and South Chatham (Barnstable County), and the other is along the northeastern shore in the Newburyport/Plum Island area (Essex County).

Records of sparrows in some Massachusetts marshes cover many years. The first definite record in the state may have been a bird taken at Nahant in August 1877 (Stearns 1881). Sparrows were initially found in the Westport marshes in southern Massachusetts in 1896 (Griscom and Snyder 1955). A breeding population (48-50 pairs) nearby in South Dartmouth was studied recently by Marshall (Marshall 1986, Marshall and Reinert 1990). Forbush (1929) regarded the species as a local summer resident only along the southeastern coast, and remarked that it occurred casually in summer only as a visitor north to Essex County. It was found in summer on the Plum Island marshes in 1949 where it likely bred in 1952 and 1953, though it was absent the next year (Griscom and Snyder 1955). Still, the bird remains established in the area (J. Baird pers. comm.).

The species' Natural Heritage rank (S2) indicates that it is restricted in occurrence and very vulnerable in the state (Table 1).

**Figure 2.** Northeast breeding distribution of the seaside sparrow (*Ammodramus maritimus*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



**Rhode Island.** - This is the only state in which sparrow surveys have been conducted in all tidal wetlands (Reinert et al. 1981; Stoll and Golet 1983; DeRagon 1984, 1988; F. Golet pers. comm.). The species is a relatively uncommon and sparsely distributed summer resident, mainly in larger saltmarshes throughout coastal Rhode Island. In the summer of 1981, the species was known in summer from only seven locations in the state (Stoll and Golet 1983). The state BBA confirmed breeding in 6 of 13 atlas blocks in which the species was recorded from 1982-87; breeding was rated as probable in 4 other blocks (R. Enser pers. comm.). Since 50 atlas blocks in the state contained marshes that varied from about 4-40 ha in total area (DeRagon 1984), sparrow distribution is severely limited. Between 1935 and 1982, pairs of sparrows had been observed at several marshes in the state for a series of years before disappearing. In others they disappeared only to reappear a year or more later (R. Ferren pers. comm., F. Golet pers. comm.). The seaside sparrow is listed as a species of *special concern* and is rated as very vulnerable (S2) by the state Natural Heritage Program (Table 1).

**Connecticut.** - The seaside sparrow is an uncommon summer resident that rarely overwinters in Connecticut. Breeding has recently been confirmed in New Haven, Middlesex, and New London counties, and is probable in Fairfield County (Bevier in press). Preliminary distribution data accounted for the birds at 11 sites in the state, from the Stamford area east to the Stonington and Pawcatuck areas near the Rhode Island border (Bever in press). Thus, considering its limited distribution in Connecticut, the state's current Natural Heritage ranking of S3 for the species may need to be reviewed.

Historically, the seaside sparrow was considered abundant and widespread in Connecticut (Sage et al. 1913). Marsh destruction from development has reduced suitable habitat severely (Bever in press). The state recently has proposed that this bird be accorded the status of *special concern*.

**New York.** - The species is a locally common to abundant summer resident only in southwestern Long Island. It once occurred on Staten Island, but is now extirpated there (Andrie and Carroll 1988). Seaside sparrows occur regularly but sparingly in winter (Bull 1964, 1974), except during mild winters when it is more common (W. Post unpubl. data). Currently and historically, sparrows are concentrated mainly in the extensive marshes on the north side of the barrier islands from Jamaica Bay (Kings and Queens counties) eastward to Shinnecock and Mecox bays (Suffolk County). Within this region, their center of abundance is in the south shore bays of Nassau and western Suffolk counties. Numbers in Jamaica Bay are greatly reduced now (J. Greenlaw pers. obs.). Outlying populations occur to the east on Gardiner's and Fisher's islands. Except for a few birds in small marshes at the heads of Little Neck Bay and Hempstead Bay in Nassau County, the species is absent from the north shore of Long Island. There is one locality that presently supports summering birds in Westchester County (Andrie and Carroll 1988). At one time, the species was more widespread in the New York City area, even occurring in a brackish marsh at Piermont on the Hudson River (Rockland County) (Bull 1964). The state Natural Heritage Program ranks it as an S3 species (Table 1). It currently receives no special protection status by the state.

**Pennsylvania.** - The species is accidental in the state (S4 Natural Heritage ranking). Poole (1964) gave two records, one based on an old specimen taken near Reading, Berks County, and the other at Berwyn, Chester County.

**New Jersey.** - The seaside sparrow is a common breeder in suitable habitat in the state. It winters there in small but seasonally variable numbers (Robbins 1983). Breeding sparrows occur most abundantly along the lower Delaware Bay and Atlantic shores (Cumberland, Cape May, Atlantic, Ocean, and Burlington counties) in southern New Jersey. Along the south Atlantic coast, they inhabit marshes in the bays behind the barrier islands north to Barnegat Bay (Stone 1937, J. Greenlaw pers. obs.). North of this area, they

become more local and sparsely distributed to Raritan Bay in Middlesex County (J. Greenlaw pers. obs.), with outlying birds recently confirmed breeding in a Hudson County marsh (D. Hughes pers. comm.).

Fables (1955) remarked that the species has decreased in New Jersey due to mosquito control activities. However, in some populations in the state, BBS survey data indicate that sparrows increased 7.9% annually (median) from 1966-89, a trend that was only marginally significant (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). The species' status in the state as a breeding bird is rated as secure overall (*S4*) by the Natural Heritage Program (Table 1).

**Delaware.** - Seaside sparrows breed commonly and winter uncommonly in the state (Robbins 1983, Hess et al. in press). Birds can be found nesting along the Atlantic and Delaware Bay shores, and most populations are in the two southern counties (Kent and Sussex) where breeding has been confirmed during recent BBA work. Birds were reported at only two sites in New Castle County along the bay (Hess et al. in press). They estimated that the state's breeding population is large, however, they commented that recent CBC data indicated a decline of more than 80% over the past 20-25 years in the numbers of birds lingering on Delaware marshes in December. They attributed this to habitat changes related to wetland management practices that focus on other species to the detriment of sparrows. BBS survey data from Delaware over the period 1966-89 suggest that breeding populations along 3 of 5 reporting routes may be increasing, a trend that was not statistically significant (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). The Natural Heritage ranking for this species in the state is *S4* (Table 1).

**Maryland and the District of Columbia.** - The species is a widespread and abundant breeder in suitable habitat in the state, but most birds withdraw during the winter (Robbins 1983). It is especially prevalent in the four counties on the Delmarva Peninsula (Dorchester, Wicomico, Somerset, and Worcester) (Robbins 1983). Scattered populations occur along the west shore

of the upper Chesapeake Bay as far north as Gibson Island (Anne Arundel County) and upstream along the Potomac River to Popes Creek (Charles County). On the east shore of the bay, birds are found north to Queenstown (Queen Anne's County) (G. Therres pers. comm.). Overall, Maryland's sparrow populations are large and secure at present and ranked as *S5* by the state Natural Heritage Program (Table 1). Along three BBS routes in the state that sample the species, populations exhibited a small (median = 1.9%) increase over the census period. This trend was significant statistically, and was noted on all routes (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).

**Virginia.** - The seaside sparrow is a common summer and an uncommon winter resident in some coastal areas in the state. Preliminary BBA data identify about 26 populations, most of them (73%) concentrated in two counties (Accomack and Northampton) on the Delmarva Peninsula (S. Ridd and R. Wadja pers. comm., see also Robbins 1983). Scattered populations occur along the west side of Chesapeake Bay in Northumberland, Mathews, Gloucester, York, and Isle of Wight counties. The greatest concentrations of birds are located in bay marshes behind barrier islands along the Atlantic shore. In 1985, the state Natural Heritage Program assessed the bird as reasonably secure and ranked it as an *S4* species (apparently secure) (Table 1).

**West Virginia.** - Seaside sparrows are unrecorded in this state (Hall 1983).

## Summary

Seaside sparrows breed in all the coastal states in the Northeast except Maine (Figure 2). The largest and least threatened regional populations (Natural Heritage statuses *S4* and *S5*) are found along the east shore of the lower Chesapeake Bay in Virginia and Maryland, and along the Atlantic shores of Virginia, Maryland, southern Delaware, and southern New Jersey (including the lower Delaware Bay) north to Ocean County. Fairly large and reasonably secure (*S3*) populations also occur in southwestern Long

Island. The sparrows along the New England coast north to Hampton, New Hampshire, are sparsely distributed in small, vulnerable to highly vulnerable (statuses S1 and S2) populations. Connecticut's populations seem to be more vulnerable than the species' current state Natural Heritage ranking (S3) implies.

## LIMITING FACTORS AND THREATS

### Present Threats to Habitat

The major threat to seaside sparrows is unbridled coastal development and the consequent loss and degradation of habitat through filling, draining, diking, and pollution. Since the mid-1950s, estuarine wetland loss in the U.S. coastal zone has accelerated to about 0.5% annually. Tidal wetland destruction has occurred in all coastal states, but in the East, losses have been greatest in Florida, Louisiana, New Jersey, and Texas (Tiner 1984). Since the 1700s, an estimated 40% of tidal marshes on Long Island have been destroyed (G. Richard pers. comm.). By 1938, about 90% of salt marshes from Virginia to Maine were ditched (Nixon 1982). Significant populations of seaside sparrows are present in all these regions. Most coastal states have enacted special laws to protect estuarine wetlands, however, these vary markedly in the extent of protection provided (Tiner 1984).

Early symptoms of population trouble in this species are the reduced number of extant populations in a region and the smaller sizes of those that remain. Seaside sparrows are sufficiently adaptable to be able to persist as one or two isolated pairs in a remnant marsh (J. Greenlaw pers. obs.). But, small populations resulting from diminished marsh size face the increased likelihood of stochastic extinction. This stage is already in progress nearly everywhere along the Atlantic and Gulf coasts. Local losses are cumulative until regional extirpation and range contraction occur. As noted, the final stage (extirpation) has been achieved along the Atlantic coast of Florida (Kale 1983).

### Diseases, Parasites, and Predation

There is no evidence that disease is an important limiting factor in seaside sparrows, but this may be partly because of a lack of study. A low incidence of pox disease occurs in New York sparrows (J. Greenlaw pers. obs.)

Neither is there reason to believe that parasites represent a serious problem in this species. Apparently healthy sparrows often carry body loads of endoparasites (Trematoda, Cestoda, Nematoda, and Acanthocephala). Acanthocephalans are especially prevalent in the blood of birds in the Carolinas (Hunter and Quay 1953). For most endoparasite groups, immature birds have larger body loads than adults (Hunter and Quay 1953). The blood fluke *Pseudospelotrema ammospizae* (Trematoda) was originally described from the seaside sparrow (Hunter and Vernberg 1953). Ectoparasites (Mallophaga, Diptera: Hippoboscidae, Acarina) are also present (Post and Enders 1970, J. Greenlaw pers. obs.), but infestations tend to be small and occasional in New York birds (J. Greenlaw pers. obs.).

Terrestrial and avian predators are an important secondary source of nest mortality in New York and New England sparrows (Post et al. 1983, Marshall and Reinert 1990). Known predators on seaside sparrows or their nests in the Northeast are Norway rats, northern harriers (*Circus cyaneus*), fish crows (*Corvus ossifragus*), and garter snakes (*Thamnophis sirtalis*) (Post et al. 1983, J. Greenlaw unpubl. data). The American crow (*C. brachyrhynchos*) and common grackle (*Quiscalus quiscula*) in New England (Marshall and Reinert 1990), and ardeids on Long Island (J. Greenlaw pers. obs.) may be problems as well. Austin (1983) mentioned other species, including microtine rodents, that may be important predators within the species' range. On Long Island, the meadow vole (*Microtus pensylvanicus*) sometimes uses abandoned sparrow nests (J. Greenlaw pers. obs.). But, there is no evidence that this rodent is a nest predator or that it actively evicts sparrows from their nests. In one case, W. Post (pers. comm.) witnessed an adult sparrow drive a vole from the vicinity of its nest.

### Other Factors

Tidal and weather-related flooding is a significant mortality factor, especially in northern populations (Post et al. 1983, Marshall 1986, Marshall and Reinert 1990). Sparrows adaptively compensate for this ever-present risk to their nests by quickly renesting following floods, and by elevating their nests in the vegetation (Marshall and Reinert 1990).

Hurricanes represent a substantial risk to coastal species, especially to those in the Southeast. *A. m. mirabilis* was extirpated from its type locality by a hurricane that struck Cape Sable, Florida, in 1935 (Stimson 1968).

Fires are an important factor in some populations of sparrows in Florida, both from a detrimental (Austin 1983) and a beneficial (Werner and Woolfenden 1983) standpoint. Except in certain local areas, fire is not important as an ecological factor in northeastern marshes. Where it does occur, fire tends to be restricted to high marsh environments that support few seaside sparrows or none at all. Fire also occurs as a postbreeding factor rather than during the breeding season. The greatest impact of fire on breeding sparrows might be to destroy clumps of persistent, overwintering marsh grasses that they use as vernal nest sites.

Natural successional changes (primary succession) that converts low marsh to high (Niering and Warren 1980) represents a problem over a period of several hundred years (Redfield 1972). High marshes provide suboptimal or marginal sparrow habitat (Reinert et al. 1981), so long-term changes in sparrow population productivity resulting from succession can be expected even in protected tidal wetlands.

### MANAGEMENT POTENTIAL

The seaside sparrow is an adaptable species that is capable of scaling important aspects of its behavior to spatial and temporal variation in environmental factors, thus minimizing the effects of this variation on reproduction (Post 1974). In low marsh habitats at least, New York and Florida populations were able to achieve growth

or stabilizing replacement rates in the face of unusually high predation or catastrophic nest losses from flooding (Post et al. 1983). Adults renest quickly and synchronously after flood-related nest destruction (Marshall and Reinert 1990), and readily colonize newly available microhabitats on marshes (Post 1974).

Still, there are limits to this adaptability. Although sparrows are able to utilize some human-modified (ditched) marshes, they do so only at reduced densities and are absent on other altered marshes (Stoll and Golet 1983). Since this species tends to occupy an intermediate position along moisture gradients in tidal marshes (Sharp 1969, 1970; J. Greenlaw 1983), any change that creates drier or wetter conditions will tend to affect it adversely.

Thus, the seaside sparrow's biological characteristics as an opportunistic species adapted to an unpredictable and variable environment give it a high management potential, as long as its microhabitat requirements are maintained. ☐

### LAND PROTECTION AND PRESERVE DESIGN

In the Northeast, as in much of its range elsewhere, the seaside sparrow is a specialist on *Spartina alterniflora*. For this reason, not only are these birds sensitive indicators of the health of tidal wetlands, but they are also vulnerable to habitat modification. Saltmarsh protection is paramount for their survival. In many areas, especially in New England and parts of Florida, populations are small and widely scattered, so local losses quickly lead to range contraction (Sykes 1980, Kale 1983).

Simple marsh protection may not always be sufficient to stem local extinctions since small populations are notoriously subject to stochastic processes. Recent studies (Greenlaw 1983, Stoll and Golet 1983, DeRagon 1988, Marshall and Reinert 1990) showed that certain vegetative and physiognomic characteristics associated with small changes in marsh relief were the principal factors affecting sparrow dispersion and density in the Northeast. Optimal sparrow habitat is found in unaltered low marsh that contains expanses of

medium-height cordgrass with medium to high stem density and a turf of clumped, residual stems. Spots that are not subject to regular and extreme flooding from tides and that have pools or open muddy areas are especially suitable (Post 1970a, 1974). Stoll and Golet (1983) discovered that this microhabitat profile occurred at 8 of 9 sites in Rhode Island where seaside sparrows were observed. These characteristics were absent or confined to small areas at 24 other marshes that did not support the species. A similar pattern was found on Long Island (Greenlaw 1983). High marshes that support sparrows fulfill the birds' basic requirements in other ways, but not all high marshes provide compensating conditions.

In general, large marshes are preferred over small areas of remnant marsh. The key requisite is that populations should be as large as possible in each favorable locality. This means that the mix of preferred microhabitats should be as expansive as possible.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Regionwide surveys of all saltmarshes are needed to determine the current abundance and distribution of seaside sparrows. An initial survey of this sort has been conducted in Rhode Island (Stoll and Golet 1983, F. Golet pers. comm.), but most of the information available in other states is spotty in quality and concerns only presence or absence. Once a broad-based survey is completed, then selected "indicator" populations should be monitored on a scheduled basis to detect long-term trends and to guide policy or management decisions. Monitoring should probably proceed on a stratified schedule. A series of small, vulnerable populations (minimum of 5 to 10) should be censused annually to detect changes quickly. However, since stochastic extinction can be a problem in small, local populations, it would be necessary to follow one or two large, apparently healthy populations in the same region as well, perhaps on a 3-year schedule. Changes in the latter populations are likely to portend the

development of serious problems that might threaten the species regionally.

A transect census method (Emlen 1977) is an efficient technique to monitor sparrow population size (J. Greenlaw pers. obs.). A single observer following a compass heading, or walking presurveyed lines through a marsh can detect a high proportion of the resident males under prescribed conditions. A useful transect width is 50-m (25 m to each side of observer); strips 100-m wide are too broad unless the population is sparse or two observers work together. Broadcast of tape-recorded primary song for 1-2 minutes at every 25 or 50 m-stop along the route greatly increases the efficiency of the procedure, since males (and often females) come to the top of the vegetation in response. There is no evidence that flushing causes sparrows to desert nests with eggs or young, or that song broadcasts affect them in any adverse way (J. Greenlaw unpubl. data). Song playback also makes it possible to census during late mornings and at midday, and during stages of nesting when males are less active vocally. If taped song is not employed as an adjunct procedure, censuses should be restricted to the period between sunrise and 08:00 and towards dusk, during the last week of May and the first 2-3 weeks of June. Ancillary data on breeding ecology and behavior should be collected routinely during surveys. Such data, including information on seasonal timing of nest-building, feeding young, presence of independent juveniles, use of different substrates as nest sites, and fates of any nests discovered can be obtained with little extra effort by field personnel.

Currently, no monitoring programs focus on seaside sparrows anywhere in the Northeast. In Florida, only the endangered race *A. m. mirabilis* receives attention (Kushlan et al. 1982), but in recent years monitoring has not been done on an annual schedule (O. Bass, Jr. pers. comm.).

### Management Procedures and Programs

At times, management intervention may be necessary to enhance or restore sparrow habitat. Since seaside sparrows favor poorly-drained sections of tidal wetland where medium-length cordgrass grows, managers should consider

blocking selected ditches on altered marshes to create additional habitat. Intervention that forms a mosaic of habitat patches consisting of favorable nesting substrate and suitable foraging sites should increase local sparrow populations significantly. Predator control may be necessary in some areas. On high marshes, shallow pools constructed near spoil deposits (soon colonized by *Iva* spp.) should encourage sparrows to settle, albeit at relatively low densities.

Techniques for breeding and rearing seaside sparrows in captivity (Post and Antonio 1981, Webber and Post 1983), and for preserving sparrow sperm (Gee and Sexton 1983) were developed during the recovery effort for *A. m. nigrescens*. The overall reproductive success of captive sparrows was about the same as that in a wild population in Florida (Webber and Post 1983). These techniques should be readily transferable to populations elsewhere in the species' range if needed.

### Research Needs

The primary objectives of any management program for seaside sparrows should be to maintain present distribution and abundance in regions where their current vulnerability is low (Natural Heritage Program state ranks of S4 and S5), and to undertake wetland enhancement steps to improve numbers where they are at greater risk (S1 and S2) (see Table 1). Research needs that particularly address these concerns are most appropriate. Following are some pertinent questions.

- What are the present patterns of seaside sparrow distribution and abundance in each region? To answer this, surveys must be conducted for seaside sparrows on specific marshes, such as has been completed in Rhode Island. State BBA surveys do not supply this information because they vary in quality and do not provide data on numbers of birds on specific marshes.
- How variable are annual productivity and survival within and between sparrow populations, and how much recruitment occurs between marshes in a local area? At the moment, only one population in the Northeast (New York) has been studied sufficiently to characterize most of these parameters (Post and Greenlaw 1982, Post et al. 1983, J. Greenlaw unpubl. data), but this population may be atypical since sparrow density was exceptionally high during the years of investigation (Post 1970a). Nest studies and banding programs in selected populations should be undertaken.
- Where do sparrows from the Northeast spend the winter? In general, CBC data show that Atlantic coastal birds are primarily concentrated from the central South Carolina coast (Charleston County) south to northeastern Florida (Nassau and Duval counties) (Robbins 1983, Root 1988). Further information on the wintering ecology and behavior of sparrows in these marshes should be gathered. In conjunction with color-marking programs in selected populations in the Northeast, intensive searches for marked birds employing cooperators in the Southeast should be undertaken in potential wintering marshes. The localized nature of saltmarshes and the habitat specialization of the sparrows make this approach feasible.
- How are seaside sparrows affected by ongoing "open marsh water management" programs? These programs represent a developing strategy to control mosquito populations while minimizing changes in normal marsh hydrology (Lent et al. 1990). Maryland, Delaware, New Jersey, and Massachusetts have been especially active in pursuing such programs on an experimental basis (Nixon 1982, Lent et al. 1990), but the

effects of such programs on seaside sparrow populations are not yet known.

Research efforts of this sort must be long-term enterprises. To reduce uncertainties related to multi-year commitments of time and effort by academic investigators, the task of undertaking and coordinating these efforts should reside with federal (U.S. Fish and Wildlife Service) and state nongame wildlife agencies. The actual field work could be performed by personnel at research-oriented wildlife observatories and field stations (e.g., Manomet Bird Observatory, Manomet, Massachusetts; Seatack Foundation Environmental Program at Seatack National Wildlife Refuge, Islip, New York; Cape May Bird Observatory, Cape May, New Jersey), or by other biologists under contract to appropriate governmental agencies. Earmarked funds for research could be provided by private conservation organizations and state wildlife and Natural Heritage Programs.

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# **Fields**



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# UPLAND SANDPIPER

*Bartramia longicauda*

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**Upland sandpipers breed sparingly in grasslands and fields throughout the Northeast. Their preferred habitat consists of short grass areas for feeding and courtship interspersed with tall grasses for nesting and brood cover. In many northeastern states, airfields currently provide the majority of suitable habitat for the species, although grazed pastures and grassy fields are also used as nesting areas. Changes in farming practices, development, and reforestation are responsible for the steady decline of upland sandpipers in the Northeast, averaging 0.80% annually over the last decade. Enhancement and protection of grassland habitats should result in population growth, although factors during migration and wintering in South America may also limit populations. Annual censuses of breeding areas are necessary to provide information on long-term trends.**

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## DESCRIPTION

### Taxonomy and Morphology

The upland sandpiper (*Bartramia longicauda*), formerly the upland plover, ranges from 27.9-32.5 cm in size and is the most terrestrial of North American shorebirds inhabiting open grasslands of the New World. The sexes are outwardly alike; females average slightly larger than males (Forbush 1925, Prater et al. 1977).

Breeding adults are overall scaly-brown in appearance above with a long slender neck, small rounded head, and relatively long tail. The upper neck is buff-streaked brown with sharply defined V-shaped markings becoming more barred on the lower breast and flanks. The throat and abdomen are white. The eye is large with a dark iris. The

bill is short, slightly decurved and dusky at the tip. The tail feathers are barred, dark brown with outer tertials pale orange-brown basally, tipped with white. Legs and feet are yellow-grey (Forbush 1925, Roberts 1955, Prater et al. 1977). Adults captured at the nest may be sexed by wing chord and tail length. This method of sex determination is estimated to be 88.3% accurate for mated pairs (Peterson 1983).

Downy young are a fine, mixed pattern of black, white and buff yellowish-brown above. A black stripe runs from the base of the bill over the top of the head. There is a band of buff or yellowish-brown across the upper breast. The sides of the head, chin and underparts are generally white (Forbush 1925).

Juveniles resemble adults, but the upperparts are darker and scalier with the buffy color of the neck, breast and wings much deeper and the streaks of the foreneck and breast less distinct.

The wing coverts have clear buffy edges and dark submarginal lines. The scapulars are uniformly dark with narrow, defined buff-white fringes. The tail feathers are notched with pale buff. Following the first prenuptial molt the young become indistinguishable from adults (Forbush 1925, Hayman et al. 1986).

Winter plumage is similar to that of the breeding adult, but paler (Forbush 1925).

### Vocalizations

The upland sandpiper's unique vocalizations include a rapid, liquid "quip-ip-ip-ip" series of alarm notes and a penetrating "whip-whee-ee-you" windy whistle (Johnsgard 1981). Upland sandpipers typically perch on fenceposts, poles and snags to watch for intruders and during courtship. The behavioral habit of momentarily holding its wings straight up when alighting (Forbush 1925) and its distinctive calls are diagnostic (Johnsgard 1981).

### HABITAT

The upland sandpiper is associated with open grassy fields and native grasslands (Forbush 1925). The species is rare along shores and mudflats in migration (American Ornithologists' Union 1983).

### Breeding

Breeding distribution is restricted primarily to extensive, open tracts of short grassland habitat. Upland sandpipers nest in native prairie, dry meadows, pastures, domestic hayfields, along highway rights-of-way and on airfields (Higgins et al. 1969a, American Ornithologists' Union 1983, Osborne and Peterson 1984). Nesting is also known to occur in dry patches of wet meadows (Stewart 1975, Herman et al. 1984) and in blueberry (*Vaccinium* spp.) barrens (J. Albright pers. comm.). A survey of nesting habitats in Wisconsin (White 1983) suggests that upland sandpipers favor a level topography with a minimum of tall vegetation edges and proportionately high acreages of agricultural crops

which duplicate prairie grasslands in terms of structure.

Vegetation height is an important factor in the selection of upland sandpiper nesting sites (Kirsch and Higgins 1976). Nesting studies by Ailes (1980) in Wisconsin recorded 54% of upland sandpiper nests in cover between 25-40 cm in height, not exceeding 70 cm at the time of egg hatching. Upland sandpipers in North Dakota were found nesting in cover between 15.5-30.8 cm in height, and appeared to avoid cover over 61.5 cm (Kirsch and Higgins 1976). White and Melvin (1985) reported that vegetation surrounding six active nests located on a Massachusetts airfield ranged from 8.0-25.0 cm in height.

Agricultural land use patterns and farming practices influence the upland sandpiper's choice of nesting sites. In central Wisconsin, Ailes (1980) found that idle fields and hayfields accounted for the majority of nesting habitats. Nesting studies in North and South Dakota indicated that the majority of upland sandpipers nested in ungrazed grasslands of medium density with abundant ground litter (Higgins et al. 1969a). A 5-year survey (1969-74) of intensively cultivated areas in the prairie pothole region of east-central North Dakota recorded 57% of upland sandpipers nests in untilled habitats, which comprised only 7% of the total study area (Higgins 1975). In Illinois, Buhnerkempe and Westemeir (1988) reported that sandpipers selected stands of grass and forbs for nesting and avoided fields of uniform grass and legumes.

In North Dakota, Kirsch and Higgins (1976) recorded their lowest mean upland sandpiper nest densities in annually tilled croplands and their highest mean nest densities in native grasslands the second season after a prescribed burn. Seeded grass/legume mixtures generally grew too tall and dense for upland sandpipers. Kirsch and Higgins (1976) found that the majority of North Dakota nests were located in either thin, uniform vegetative cover or in scattered clumps of fairly dense cover characterized by standing stubble fields, moderately grazed pastures, mowed areas with heavy regrowth, brush clumps with some understory vegetation, and undisturbed vegetation on poor soils. Residual vegetation from the previous growing seasons accounted for 25% of the cover at 54% of sandpiper nests.

Upland sandpipers use grassy fields of low vegetation height for feeding and brood rearing. Ailes (1980) observed 66% of adults with young in Wisconsin in grazed pastures, 13% in ungrazed pastures, and 11% in hayfields. Ailes (1980) found a large percentage of adults with young in Wisconsin in heavily grazed fields with vegetation ranging from 0-10 cm in height. Buhnerkempe and Westemeir (1988) reported that upland sandpipers in Illinois selected brood habitats of wheat stubble fields, recently hayed legumes, old redtop meadows (*Agrostis* spp.), and moderately grazed pastures. A South Dakota grasslands management study showed upland sandpiper habitat use (nesting was not documented) to be highest in recently burned fields with short, open, new growth and no litter or old growth (Huber and Steuter 1984).

Upland sandpipers accept a variety of native and introduced grasses (Buss and Hawkins 1939). Timothy (*Phleum* spp.), bluegrass (*Poa* spp.), needlegrass (*Stipa* spp.), bluestem (*Andropogon* spp.), quackgrass (*Agropyron* spp.), Junegrass (*Koeleria* spp.), and bromegrass (*Bromus* spp.) are among the grasses associated with upland sandpiper nesting fields (Buss and Hawkins 1939, Meanley 1943, Buss 1951, Higgins et al. 1969a, Kirsch and Higgins 1976, Ailes 1980).

### Wintering

On their wintering grounds in South America, upland sandpipers have been observed in pastures of remote estancias (Wetmore 1927), harvested and burnt-over canefields, football fields, airfields (Haverschmidt 1966), and on sandy beaches where the vegetation is open or recently burned (Spanns 1978). The largest known wintering concentrations (November-February) of this species occur in Argentina, north and west of Buenos Aires, principally in the provinces of Entre Ríos, Santa Fe, Corrientes, and Córdoba, and in Uruguay (White 1988). Sightings of lesser numbers of overwintering sandpipers come from Brazil, Paraguay, Bolivia, and Venezuela (White 1988). Haverschmidt (1966) noted that small flocks of upland sandpipers were regularly present in Suriname during winter months. Recently, a flock of 50 presumed migrants was sighted in

migration in the high Paramo of the Andes Mountains in Ecuador (B. Harrington pers. comm.).

Records indicate an apparent shift in the wintering range of upland sandpipers in Argentina and Uruguay within the past 60 years. Changes in land use and agricultural practices may be critical to the upland sandpiper's limited numbers and suggested change in wintering habitat (White 1988). In the province of Córdoba, where the greatest numbers have been reported to date, native espinal (scrub trees) have been converted to dairy farms planted in pasture and alfalfa. Upland sandpipers possibly prefer the drier climate and planted grasses to historically utilized wetter, native grasslands found farther south and southeast (White 1988).

### BIOLOGY

Upland sandpipers tend to be loosely colonial while breeding (Bowen 1976), often occupying the same nesting fields in successive years (Buss and Hawkins 1939, Ailes 1980). Studies by Bowen (1976) and Ailes (1980) suggest that adults may exhibit some degree of site faithfulness, although Ailes (1980) found that none of the 61 young he banded returned to their natal grounds the following year.

Upland sandpiper nest territories are generally grouped and consist of a nesting site, plus a loafing and feeding area near or adjacent to the nest territory which is shared communally (Buss and Hawkins 1939). In the central portion of the species' range in North Dakota, breeding densities of up to 20 pairs/mi<sup>2</sup> (2.59/km<sup>2</sup>) have been recorded (Stewart and Kantrud 1972).

Upland sandpipers feed almost exclusively on insects, especially grasshoppers and crickets (Orthoptera), weevils (Coleoptera), and other small invertebrates gathered from or close to the ground. Occasional seeds of weeds, grasses and waste grains, including wheat, are also consumed (McAtee and Beal 1912, Forbush 1925).

Courtship is exhibited in spectacular soaring displays while ascending in great circles high into the sky accompanied by a long, drawn-out "whip-whee-ee-you" whistle, and in low over-the-ground flight on stiff, quivering wings (Buss and Hawkins

1939). On the ground, the male will sometimes approach the female, raising his tail and running towards her while giving a short, guttural whistle (Ailes 1976). This pre-copulatory behavior is sometimes followed by mating.

Initial nesting activity, which is thought to be somewhat synchronous (Higgins and Kirsch 1975), begins 2-3 weeks after spring arrival in breeding areas, from mid-April to early May (Forbush 1925, Buss and Hawkins 1939, Ailes 1980). The maximum period between the earliest initiated nests and the latest hatched nests in North Dakota (Higgins and Kirsch 1975), Wisconsin (Ailes 1980), and Massachusetts (White and Melvin 1985) ranged from 8-10 weeks. Some late nesting, or renesting, due to early nest failure may occur (Ailes 1980).

The nest is a shallow depression in the ground approximately 10-13 cm in diameter and 5 cm deep, lined with pieces of dry grass (Bent 1929). Nests are usually well hidden, frequently by vegetation that hangs over the nest hiding it from above (Johnsgard 1981). The eggs are cinnamon to pale olive-buff or greenish-white in color, spotted with brown and underlaying spots of ecru or pale grey. Clutch size is normally four eggs, sometimes three, and rarely five (Bent 1929).

Upland sandpipers incubate their eggs an average of 24 days (Higgins and Kirsch 1975), with extremes of 21-28 days reported by some investigators (Johnsgard 1981).

The chicks are precocial and leave the nest within 24 hours after hatching (Ailes 1980). Broods are tended by one (Ailes 1980) or both adults (Buss 1951) until the young attain adult weight and are capable of flight at 30-34 days (Buss and Hawkins 1939, Ailes 1980). Ailes (1980) reported that adults with young in Wisconsin utilized brood-rearing fields within a short distance of the nesting site for several weeks following hatching. In contrast, Buss (1951) found that adults with young in the Yukon Territory required a large home range, up to 3.2 km in diameter.

Estimates of nesting success for upland sandpipers in Northern Plains states range from 63% (Lindmeier 1960) to 100% (Lokemoen and Duebbert 1974). Using the Mayfield (1961) method to determine seasonal nest success rates, Buhnerkempe and Westemeier (1988) calculated

upland sandpiper nest success in Jasper County, Illinois to be 48%.

The time elapsed between upland sandpiper arrival and departure from breeding areas has been observed to be as brief as 100 days (Higgins and Kirsch 1975) and as long as 165 days (Buss and Hawkins 1939). Higgins and Kirsch (1975) have correlated the average, frost-free period with the dates of first nest initiation and final departure from breeding areas in North Dakota and Wisconsin, and suggest that some breeding ground activities may be directly or indirectly related to temperature at northern latitudes. In a similar analysis, Buss (1951) was able to correlate the timing of fall migration in the Yukon with decreasing numbers of available insects.

Data obtained from marked birds in Kansas suggest that upland sandpipers first breed when they are one year old (D. Bowen pers. comm.). The natural longevity of the species is not known. The longest known survival of a banded bird is 5 years (Clapp et al. 1982).

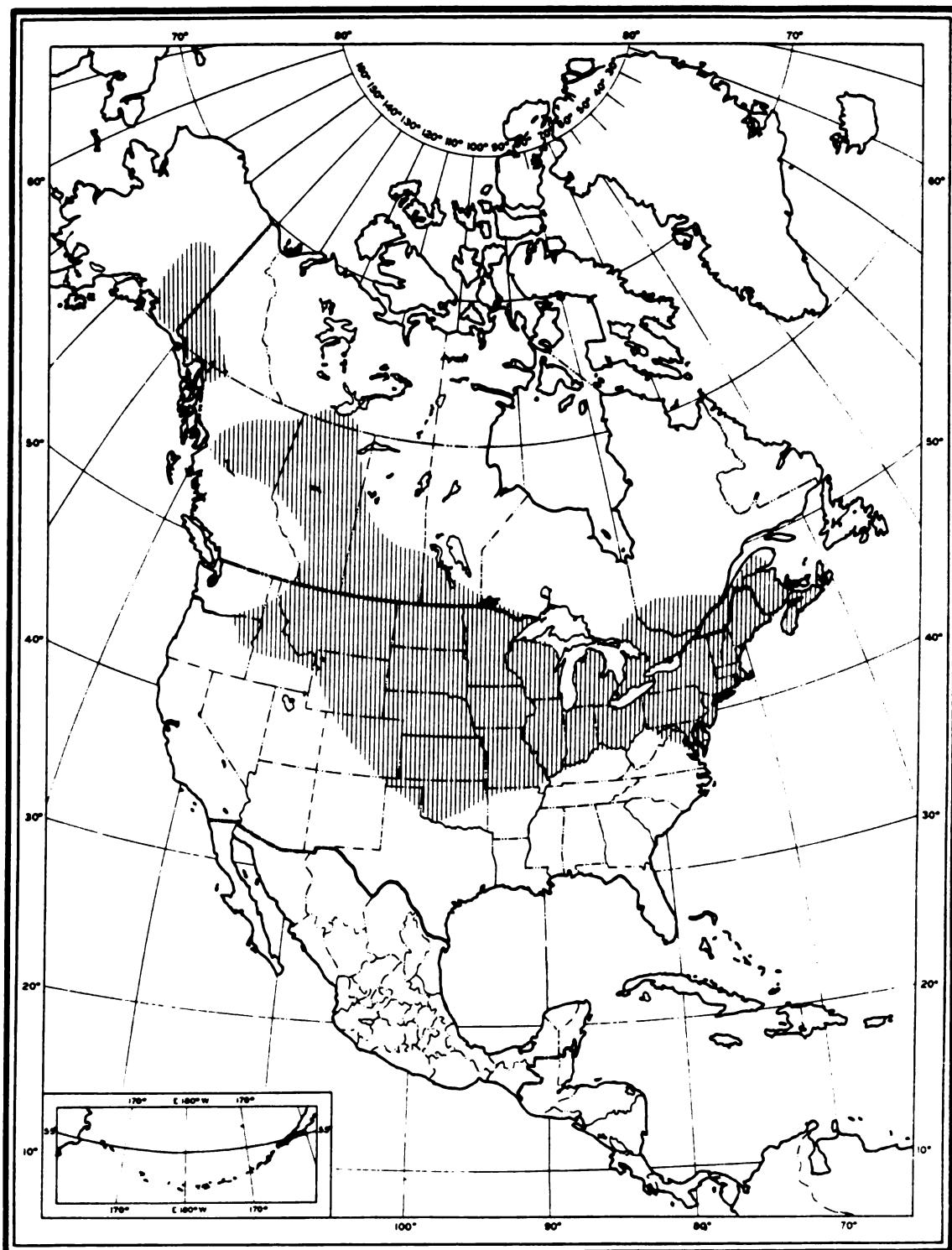
## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

The upland sandpiper breeds locally from north-central Alaska, northern Yukon, northwestern British Columbia, extreme southwestern Mackenzie, northern Alberta, southeast through southern portions of Saskatchewan, Manitoba, Ontario, Quebec to New Brunswick and central Maine; south in the interior to eastern Washington, northeastern Oregon to portions of Idaho, Colorado, Oklahoma, Texas, Missouri, Illinois, Kentucky, Ohio, possibly Tennessee, West Virginia, to Virginia and Maryland (American Ornithologists' Union 1983) (Figure 1). Highest nesting densities occur in the Northern Plains States and Prairie Provinces.

On the basis of the U.S. Fish and Wildlife Service's (USFWS) Breeding Bird Survey (BBS) data gathered from 1980-89, the continental upland sandpiper population has shown a gradual

Figure 1. The North American range of the upland sandpiper (*Bartramia longicauda*) (American Ornithologists' Union 1983).



Breeding

2.4% ( $P \leq 0.10$ ) annual increase over the past decade (Office of Migratory Bird Management upl. data: Laurel, Maryland). In the Northern Plains states of North Dakota and Montana, and in Ontario, the upland sandpiper appears to be maintaining its numbers or increasing (S. Droege pers. comm., Cadman et al. 1987). In Minnesota, Montana, and North and South Dakota, the 1985 Farm Bill has idled millions of acres of croplands through the Agriculture Conservation Reserve Program, potentially creating new habitat for upland sandpipers (H. Kantrud pers. comm.). Upland sandpiper populations in the central portions of the continent have increased an average of 2.9% ( $P \leq 0.05$ ) annually over the last decade. Statistically significant annual increases in the relative abundance of upland sandpipers have been registered in the High Plains Border (8.5%,  $P \leq 0.05$ ) and Missouri Plateau-Glaciated (11.8%,  $P \leq 0.05$ ) strata of the Great Plains Physiographic Region (Office of Migratory Bird Management unpubl. data: Laurel, Maryland). On the western periphery of its range in Washington (J. Hickman pers. comm.) and Oregon (M. Henjum pers. comm.), the species appears to be stable or decreasing. Population trend information is not available for Alaska and much of Canada.

In the Midwest, extensive plowing and cultivation and the cumulative effects of growth have destroyed much of the species' habitat (Osborne and Peterson 1984, DeGraaf and Rudis 1986). BBS data indicate a significant annual 6.8% ( $P \leq 0.01$ ) decline in Ohio's upland sandpiper population from 1980-89 (Office of Migratory Bird Management unpubl. data: Laurel, Maryland). The species is ranked globally as G5 (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

#### Distribution and Status in the Northeast Region

Over much of its former eastern range, where old field habitats are maturing or being replaced, the upland sandpiper is slowly declining, stable at low levels, or absent (Tate and Tate 1982, Tate 1986). Based on 171 routes, BBS data (1980-89) show that the upland sandpiper

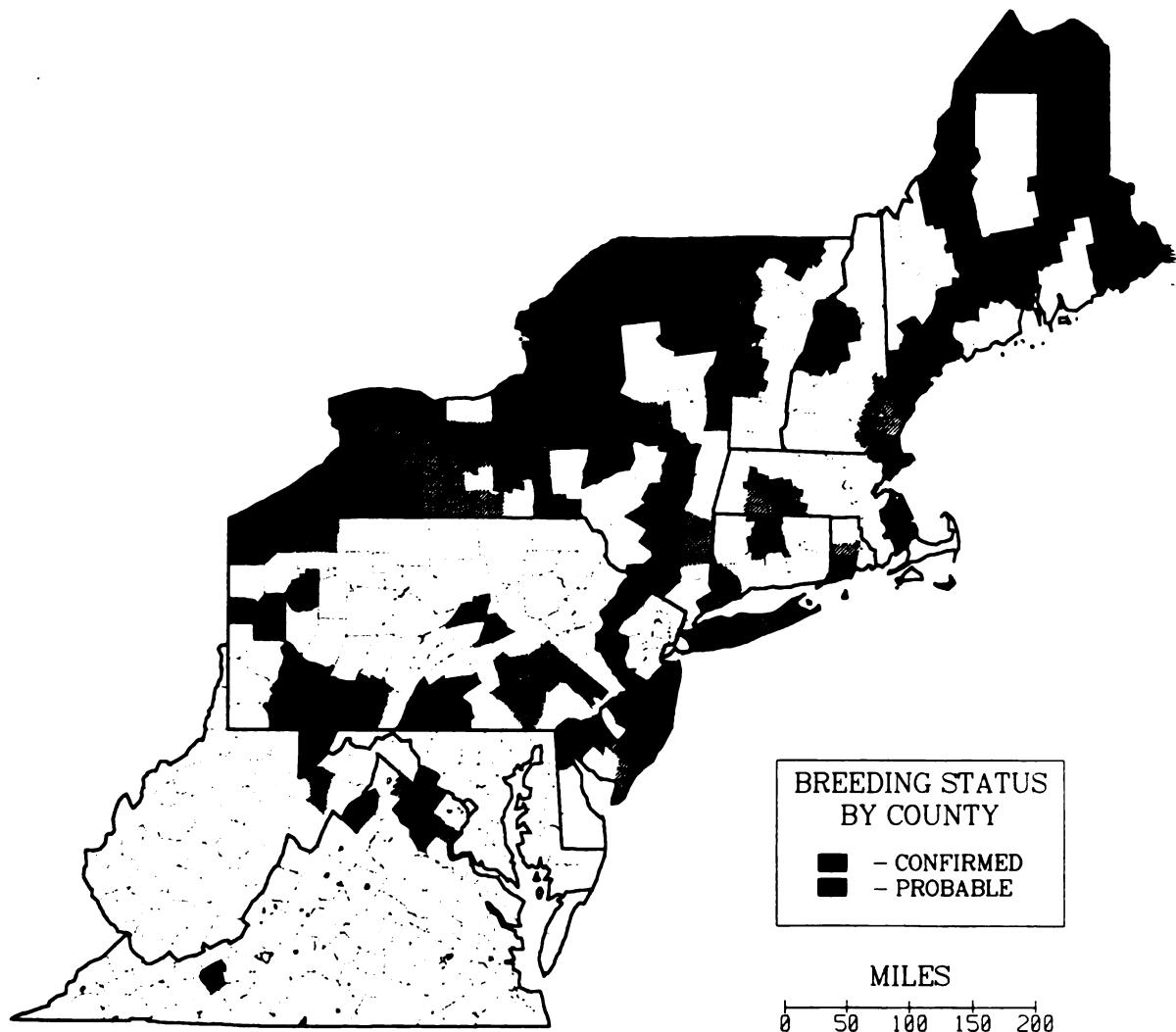
population in the East has decreased an average of 0.80% annually over the past decade. This trend is not statistically significant, however. Highly significant annual population decreases of 2.5% ( $P \leq 0.01$ ) and 18.3% ( $P \leq 0.01$ ), respectively, have been registered in the Northern Piedmont, and Ridge and Valley strata of the Eastern Piedmont Plateau Region. Within the St. Lawrence Plain stratum the species has declined an average of 6.1% ( $P \leq 0.10$ ) per year (Office of Migratory Bird Management unpubl. data: Laurel, Maryland).

**Maine.** - The upland sandpiper formerly occurred commonly in Maine during the migratory season and was not rare as a summer resident in various portions of the state (Knight 1908). Palmer (1949) states there is some slight evidence that the species nested on Maine coastal islands in early times, although its history is too obscure to furnish any details. At the turn of the century the upland sandpiper was listed as a common summer resident in 13 of the state's 16 counties (Palmer 1949). Currently the upland sandpiper is found in approximately 30 locations from York County to Washington County in open lands, including airports and blueberry barrens. Four to six pairs of upland sandpipers were reported at Brunswick Naval Air Station, Sagadahoc County, from 1985-87 (Maine Natural Heritage Program 1988). Nearly all confirmed breeding locations occur in the southern third of the state (Adamus 1988, Figure 2). Blueberry barrens, a primary habitat, are subject to alteration including extensive herbicide application, the effects of which are not known (S. Melvin pers. comm.).

Upland sandpipers are ranked as S3 (rare or uncommon) by the Maine Natural Heritage Program, and are of *indeterminate status* according to the Maine Department of Inland Fisheries and Wildlife (Table 1).

**New Hampshire.** - The peak of New Hampshire's upland sandpiper population was probably between 1860-80, when one or more pairs occupied nearly every field in the Merrimack Valley (Silver 1957). From 1981-86, breeding bird atlas (BBA) workers documented probable or confirmed breeding at only five locations in New

**Figure 2.** Northeast breeding distribution of the upland sandpiper (*Bartramia longicauda*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the upland sandpiper (*Bartramia longicauda*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	indeterminate	S3
New Hampshire	endangered	S2
Vermont	threatened	S3
Massachusetts	endangered	S1
Rhode Island	threatened	S1
Connecticut	endangered	S1
New York	special concern	S4
Pennsylvania	threatened	S2
New Jersey	endangered	S1
Delaware	status pending	S1
Maryland	endangered	S1
Virginia	endangered <sup>b</sup>	S1/S2
West Virginia	scientific interest	S1

<sup>a</sup>S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

<sup>b</sup>Proposed

Hampshire: Pease Air Force Base, Rockingham County; Fabyan Point in Newington, Rockingham County; Salmon Falls Road, Rochester, and a site near the county farm in Dover, both in Strafford County; and a farmland site in North Haverill, Grafton County (C. Foss pers. comm., Figure 2). The latter breeding location is since believed lost (D. DeLuca pers. comm.). Pease Air Force Base hosts the largest known resident population within the state. In June 1990, 15 individuals were present on the base (New Hampshire Natural Heritage Inventory unpubl. data: Concord). Peak counts of as many as 60 upland sandpipers were reported at Pease Air Force Base in August 1989 (Overtree et al. 1989) indicating that this recently closed military installation may be an important post-breeding/migratory site, further affirming the biological value of this area as a proposed National Wildlife Refuge.

In New Hampshire, the species is listed as endangered, and ranked as S2 (imperiled because of rarity) by the Natural Heritage Program.

**Vermont.** - Accelerated conversion of farmlands has precipitated a recent decline in upland sandpiper numbers in Vermont. Currently, the species is restricted to the northern two-thirds of the state, primarily the dairy lands of the Champlain Valley and Lake Memphremagog regions.

A roadside survey of these regions in June 1989 (Rimmer and Fichtel 1989) located 77 adults and 2 juveniles (conservative estimate of 35-40 breeding pairs) at 43 sites. Highest numbers were recorded in Addison County (29), followed by Chittenden (13), Orleans (11), Franklin (9), Grand Isle (8), Rutland (4) and Caledonia (1) counties. All 1989 sightings, with the exception of two, were in hayfields and pastures of actively farmed areas, the majority estimated to consist of more than 100 acres (40.5 ha). Given the probability of undetected individuals and lack of statewide coverage of all suitable habitats, these results undoubtedly underestimate the resident population. Survey coordinators estimate the state's population to exceed 50 pairs with a likely high of 60-70 pairs (Rimmer and Fichtel 1989).

BBS data (1980-89) indicate a 19.9% ( $P \leq 0.01$ ) annual decline in upland sandpiper numbers in Vermont within the past decade. The species is listed as threatened and has a Natural Heritage rank of S3.

**Massachusetts.** - Once a common summer resident of Massachusetts' open fields (Forbush 1925), the state's upland sandpiper population now numbers less than 50 pairs. With few exceptions, populations are restricted to airport grassland habitats. In recent years resident populations have been observed at 3 active sites within the state: Westover Air Force Base in Chicopee, Hampden County, where 20 pairs bred in 1989 (B. Sorrie pers. comm.); Otis Air National Guard Base in Bourne, Barnstable County, which supported 14 nesting pairs in 1985 (White and Melvin 1985); and Hanscom Air Force Base in Bedford, Middlesex County, where 7 adults and 3 chicks were noted in 1988 (J. Carter pers. obs.). In 1990, six pairs of upland sandpipers (including

four pairs with chicks) were seen at Logan Airport, Boston, confirming this suspected location as a fourth breeding site (N. Smith pers. comm.). Additional locations known to support one to three resident pairs within the past five years include: Cumberland Farms in Middleboro, Plymouth County; Newburyport, Essex County; Hadley, Hampshire County; and Fort Devens in Lancaster, Middlesex County (B. Sorrie pers. comm.).

Upland sandpipers are listed as *endangered* in Massachusetts and ranked S1 (critically imperiled) by the Natural Heritage and Endangered Species Program.

**Rhode Island.** - The upland sandpiper was a common summer resident during the 1800s, but shooting in the latter part of the century and the gradual disappearance of agricultural lands, especially hay fields, have sharply reduced their numbers in Rhode Island (R. Ferren pers. comm.). The only remaining confirmed breeding site is on Block Island where two pairs have been resident at the Lewis-Dickens Farm in recent years. One to two pairs are suspected to nest at Quonset Point Airfield in North Kingston, Washington County (C. Raithel pers. comm.). A third, recently active site in West Greenwich, Kent County, is believed lost as a result of industrial development of the area in the late 1980s (Rhode Island Natural Heritage Program unpubl. data: Providence). Away from the above locations, the species is essentially unknown in Rhode Island in summer and is very rarely reported as a transient (R. Ferren pers. comm.). Upland sandpipers are listed as *threatened* and ranked S1 by the Natural Heritage Program.

**Connecticut.** - The upland sandpiper is a locally uncommon migrant and a rare nester in Connecticut. Currently, Brainard and Bradley airports in Hartford County are the only known locations supporting breeding populations within the state. In 1988, one to two, and five pairs, respectively, were observed at these two sites (D. McKay pers. comm.). No nesting was evident at Brainard Airport in 1989 (L. Bevier pers. comm.). A mowing regime and vehicular prohibition management plan compatible with upland sandpiper nesting phenology is being implemented

at Bradley Airport in an effort to increase the resident population (D. McKay pers. comm.). Together with a probable site in Fairfield County and a possible site in Litchfield County, these restricted breeding areas represent the only known extant occurrences of nesting upland sandpipers in Connecticut (L. Bevier pers. comm.). Consequently, the species is state-listed as an *endangered* species and has a Natural Heritage rank of S1.

**New York.** - The upland sandpiper is a widely distributed, but generally uncommon breeder in New York. Numbers appear to have decreased considerably within recent years and fewer than 250 known or suspected breeding sites remain within the state (Ardle and Carroll 1988), primarily in farmlands of the Mohawk Valley, eastern Great Lakes Plains, St. Lawrence Plains and in the Lake Champlain Valley, where populations seem to have expanded (Figure 2). The species also occurs in scattered locations across the Appalachian Plateau Region, with small concentrations in the eastern Finger Lakes and Cattaraugus Highlands farm areas. It is still present in Orange County and several other locations in the Central Hudson Valley, but has disappeared from most of the Taconic Highlands (Ardle and Carroll 1988). Seven birds were present at Blue Chip Farm, Ulster County, in June 1988 (Paxton et al. 1988). It is now only a local summer resident on Long Island (Ardle and Carroll 1988). In downstate New York the best remaining site is J. F. Kennedy International Airport, Queens County, where an estimated 20 pairs summered in 1988 (Paxton et al. 1988). Rapid loss of grassland habitats to residential and industrial development, and agricultural shifts from grain crops to corn may be factors limiting upland sandpiper populations in New York (Tate 1986, Ardle and Carroll 1988). The species is listed as *threatened* and ranked S4 (apparently secure) by the Natural Heritage Program.

**Pennsylvania.** - Recent surveys (1983-89) identified 20-30 active upland sandpiper sites in Pennsylvania, the majority in agricultural lands in the western and south-central portions of the state. Some nesting may also occur in reclaimed coal strip mine areas. An estimated 50-75 upland

sandpiper pairs currently nest in Pennsylvania in small groups of 1-3 pairs, principally in Erie, Lawrence, Westmoreland and Adams counties. In recent years one to two pairs have been reported from scattered locations in Butler, Clarion, Crawford, Cumberland, Fayette, Franklin, Northumberland, Somerset and Union counties in the central and western regions. The largest known colony, located near Gettysburg, Adams County, supports one to four breeding pairs annually (Brauning in press). A site in the vicinity of Gilbertsville, Montgomery County, is the only upland sandpiper breeding site remaining in southeastern Pennsylvania (Boyle et al. 1987).

Reports indicate, however, that the species continues to utilize sites in eastern Pennsylvania during the post-breeding season. In early August 1983, 56 birds were seen feeding in cut alfalfa fields near Intercourse, Lancaster County. Twelve individuals were present in Upper Leacock Township, Lancaster County, in late August 1988 (Brauning in press). Airports in Pennsylvania, including the Greater Pittsburgh Airport and the Philadelphia Airport, should be censused during the breeding and migratory seasons. Historical records indicate that the "upland plover" was regularly present at the Johnstown Airport in Cambria County during summer months (Todd 1940).

BBS data show a 17.6% ( $P \leq 0.05$ ) annual decline in Pennsylvania's breeding population over the past decade. The species is listed as *threatened* in the state, and has a Natural Heritage rank of S2.

**New Jersey.** - Based on surveys of historically known or suspected nesting sites in New Jersey, the upland sandpiper continues to decline as a breeding species. A summary of available records compiled by Wander (1981) and subsequent field observations by Wander and Plage (Plage 1988) documented a total of 26 upland sandpiper breeding sites in New Jersey for the period 1970-87. Of these 26 sites, only 4 were known to be active in 1987 (Plage 1988). Maximum numbers of the state's resident population occur at the Federal Aviation Administration Technical Center, Atlantic County, where 14 individuals were counted in May 1987 (Plage 1988), and at Lakehurst Naval Air Base, Ocean County, where

an estimated 6-8 pairs were noted in 1988. Newark Airport, Essex County, and another site in Somerset County may support additional nesting pairs (K. Clark pers. comm.). Post-breeding congregations of upland sandpipers appear to favor certain agricultural habitats in New Jersey. Seventy-five upland sandpipers were present at the sod farms in Columbus in late August 1980 (Paxton et al. 1988).

In New Jersey, the species is ranked S1 by the Natural Heritage Program and listed as *endangered*.

**Delaware.** - The upland sandpiper is a rare spring migrant in Delaware, but a common summer visitor and fall migrant. The presence of 22 individuals at Greater Wilmington Airport, New Castle County in late June 1987, and reports of two half-grown young in mid-July of the same year, confirm this site as the only known breeding location in Delaware (Hess et al. in press). Numerous July-August sightings of upland sandpipers at Greater Wilmington Airport noted from the early 1960s to present (23 individuals on 15 July 1989) (Boyle et al. 1989), and late summer reports from Dover Air Force Base, Kent County (34 individuals on 15 July 1989) (Boyle et al. 1989) during recent years, suggest these airports are utilized as stopover sites by birds in passage.

The legal status of the species is pending in Delaware. The state Natural Heritage Program ranks upland sandpipers as S1.

**Maryland and the District of Columbia.** - The upland sandpiper was formerly a common summer resident of agricultural habitats in the Frederick Valley and an uncommon local breeder in Baltimore, Montgomery, Washington and Allegheny counties (Stewart and Robbins 1958). Since 1978, only three confirmed and two probable breeding sites in the extreme western portion of Maryland, in the vicinity of Red House, Garrett County, have been reported. Presently, fewer than five pairs are believed to nest in Maryland annually (Maryland Natural Heritage Program unpubl. data: Annapolis). In recent years up to 13 upland sandpipers have been present at the Easton Airport, Talbot County, during mid-August with lesser numbers reported from Frederick Airport, Frederick County,

indicating some fall movement through the state (R. Ringler pers. comm.).

In Maryland, the species is listed as *endangered* and ranked *S1* by the Natural Heritage Program.

**Virginia.** - According to Linzey (1979) the upland sandpiper was formerly a common summer resident in Virginia's Great Valley south to at least Montgomery County. Estimates of the present breeding population number between 15-20 pairs (J. Bazuin pers. comm.), primarily inhabiting agricultural lands. Within the past 2 years breeding populations have been noted in the Luckett's area of Loudoun County (six individuals) (Armistead 1989); near Remington, Fauquier County (two adults with young) (Armistead 1989); in the Roudebush Ponds area, Pulaski County, southwest of Blacksburg (five to six pairs) (J. Bazuin pers. comm.); and in Edinburg, Shenandoah County (T. Kain pers. comm.). Based on May-June sightings from the mid-1970s to the mid-1980s, other probable breeding locations include Dulles Airport, Loudoun County (J. Bazuin pers. comm.); alfalfa fields near Harrisonburg, Rockingham County (Hall 1989); and Nokesville, Prince William County (S. Ridd and R. Wadja unpubl. data). Late summer records from the 1970s and early 1980s indicate autumnal migrant concentrations at Byrd Airport, Richmond; Dulles Airport, Loudoun County; Langley Air Force Base, Hampton; Oceana Naval Air Station, Virginia Beach; Roanoke Airport, Roanoke; and Wallops Naval Air Station, Accomack County (T. Kain pers. comm.). A high count of 46 upland sandpipers was recorded at Wallops National Aeronautics and Space Administration (NASA) facility, August 1971, in marked contrast to August 1989 censuses yielding a maximum of 4 birds at this site (C. Vaughn pers. comm.).

Upland sandpipers are proposed for *endangered* status in Virginia and ranked as *S1/S2* by the Natural Heritage Program.

**West Virginia.** - The upland sandpiper is a rare migrant and uncommon summer resident in West Virginia. There are a few scattered records of nesting starting in the early 1900s from Berkeley, Grant, Greenbrier, Hancock, Jefferson,

Kanawha, Mason, Mineral and Monongalia counties. The upland sandpiper formerly occurred regularly and in some numbers in the Shenandoah Valley and in Jefferson and Berkeley counties (Hall 1983). Only one site (number of individuals unknown), near Petersburg, Grant County, was reported active in 1989 (G. Hall pers. comm.). Incomplete survey coverage may result in the undetected occurrence of upland sandpipers in West Virginia. During the recent BBA period (1984-89), workers were able to cover less than 20% of the blocks within the state. Airfields, including Yeager Airport in Charleston, are not routinely censused during the breeding and migratory seasons (G. Hall pers. comm.).

The state of West Virginia considers the species of *scientific interest*. The Natural Heritage Program ranking for upland sandpipers is *S1*.

### Summary

In the Northeast the upland sandpiper has steadily declined within the past century as a result of shooting, reforestation and decreases in agriculture. Current breeding populations are precipitously low, widely scattered and continue to be threatened by encroaching urbanization and changes in agricultural practices. The species is state *endangered* or proposed for endangered status in six states within USFWS Region 5, and *threatened* or of *special concern* in four additional states (Table 1). As a summer resident the upland sandpiper is nearly extirpated from Rhode Island and Delaware. Upstate New York, where many farmlands remain intact, is the only locality within the Northeast supporting fairly widespread resident populations (Paxton et al. 1988). In Delaware, New Jersey, Connecticut, and Massachusetts the largest breeding densities are restricted to airports.

Little is known of the regional population dynamics of upland sandpipers, or what constitutes a minimum viable population. Given the precarious status of the upland sandpiper [in the Northeast], all possible efforts should be made to maintain and enhance opportunities for success at the sites where the species continues to breed (Plage 1988).

## ASSESSMENT OF HABITAT QUALITY

The quality of a particular habitat is best indicated by the total number of birds present during May and June. Observations by Buss and Hawkins (1939) suggest a delicate distinction between acceptable and unacceptable sites. A slight change in an accepted field may cause it to become unacceptable, i.e., heavy or early grazing, standing water, burning, and manuring may reduce or exclude nesting from fields accepted the previous year. Abandoned fields with invading shrubs and trees may sometimes exclude upland sandpipers (Laughlin and Kibbe 1985), although at Camp Edwards/Otis Air National Guard Base in Massachusetts, upland sandpipers nest in fields with scattered shrubs and 1-2 m tall pine trees (*Pinus spp.*) (White and Melvin 1985).

Airports and airfields offer excellent habitat for breeding colonies, providing level expanses of short grass fields attractive to upland sandpipers. Nesting surveys in the mid-1980s in Ohio (Osborne and Peterson 1984) and in Massachusetts (White and Melvin 1985) showed that airport habitats in these states were utilized over all other habitats with respect to the number of sites and number of individuals per site. Upland sandpipers use the short grassy strips along runways and taxiways for feeding, loafing, nesting, brood-rearing and pre-migratory flocking. Upland sandpipers are believed to pose little threat to aircraft at airports because of their small size, typical behavioral patterns, and tendency to remain mostly on the ground. Flight is usually low and direct (White and Melvin 1985).

## LIMITING FACTORS AND THREATS

Loss and fragmentation of habitat due to increased urbanization, changes in farming practices and natural forest succession pose the most serious threats to upland sandpiper populations.

Upland sandpiper eggs and chicks are vulnerable to mammalian predation by coyotes (*Canis latrans*), badgers (*Taxidea taxus*) (Herman et al. 1984), raccoons (*Procyon lotor*) (Kirsch and Higgins 1976), mink (*Mustela vison*), skunks

(*Mephitis mephitis*) (Buss and Hawkins 1939), domestic dogs, cats, and humans. American crows (*Corvus brachyrhynchos*) (Buss and Hawkins 1939), golden eagles (*Aquila chrysaetos*), northern goshawks (*Accipiter gentilis*) sharp-shinned hawks (*A. striatus*) (Buss 1951), Cooper's hawks (*A. cooperii*), northern harriers (*Circus cyaneus*) (Herman et al. 1984), American kestrels (*Falco sparverius*) (White and Melvin 1985) and snowy owls (*Nyctea scandiaca*) (N. Smith pers. comm.) represent avian threats to adults, eggs and young.

Livestock trampling and mowing cause damage to nests and eggs (Ailes 1980, J. Carter pers. obs.).

## MANAGEMENT POTENTIAL

The upland sandpiper's ability to adapt to certain agricultural crops and other human-made landscapes (White 1983) and its recent continental population trend indicate favorable recovery potential if appropriate habitat is available. Lack of suitable nesting and brood-rearing habitat appear to be the major factors limiting the upland sandpiper population in the Northeast.

## LAND PROTECTION AND PRESERVE DESIGN

Upland sandpipers require a relatively large home range for successful breeding which provides extensive feeding and loafing areas nearby. Many apparently ideal habitats within the upland sandpiper's North American breeding range are too small to be acceptable (Buss 1951). Proposed changes in land use or farming practices in established breeding areas should be reviewed for compatibility with upland sandpiper nesting and brood-rearing requirements. Breeding locations sufficiently large enough to support viable populations of upland sandpipers should be preserved from development. In the Northeast, the approximately 300-acre Lewis-Dickens Farm area on Block Island, off the Rhode Island coast, has historically supported two pairs of upland

sandpipers during the breeding season (C. Raithel pers. comm.).

Optimal upland sandpiper breeding habitat contains a mixture of short grass areas for feeding and courtship, interspersed with taller grasses and forbs for nesting and brood cover (Kirsch and Higgins 1976, Ailes 1980). The height and density of grasses in nesting and feeding areas should permit adults and chicks to move through the vegetation easily. An upland sandpiper summering area is of good quality if it meets the physical and biotic needs described under *Habitat*, is located in idle or very lightly used cover (Higgins et. al. 1969a), and remains undisturbed during the nesting phase of the breeding cycle.

Management for upland sandpipers on public lands can partially compensate for loss and deterioration of habitat on private land (Kirsch and Higgins 1976). Publicly-owned natural prairies should be managed to preserve their original status (Tester and Marshall 1962). Owing to the decline of both native and non-native grasslands, particularly in eastern portions of the species' range, airports, if adequately managed, could play an important role in producing stable densities of upland sandpipers and other grassland avifauna (Osborne and Peterson 1984).

The species' protracted length of travel along migration routes emphasizes the need to preserve not only traditional breeding and wintering sites, but also upland shorebird habitats throughout the hemisphere (White 1988).

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Annual censuses of known or traditional upland sandpiper breeding locations are necessary to provide information on long-term population trends, to promote protection of breeding sites, and to assess the effectiveness of active management programs. Monitoring programs are currently conducted primarily in states where the species has special status, including Maine, Vermont, New Hampshire, Massachusetts, Rhode Island, Pennsylvania, New Jersey, Delaware, Idaho, Oregon and Washington.

Censuses of adult upland sandpiper populations can be taken within one to two weeks of arrival in breeding areas, before the majority of nest territories become established. A field occupied by sandpipers prior to the nesting season usually will have one or more nesting sites nearby.

To confirm nesting, fields can be searched on foot, with two or more individuals walking slowly abreast at a distance of 6-8 ft (1.8-2.4 m), depending on the density of cover. Biologists at the Northern Prairie Wildlife Research Center in North Dakota have effectively used a cable-chain device stretched between two vehicles to systematically search for shorebird and duck nests over extensive croplands, grasslands, and areas of mixed grass/brush vegetation (Higgins et al. 1969b, 1977). Possible disadvantages of this technique are that trails left by vehicles may provide mammalian predators with travel lanes through dense cover (Higgins et al. 1969b), and that vehicles may inflict damage to nests of other species. While flushing does not cause upland sandpipers to desert their nests (Buss and Hawkins 1939), workers should take precautions to reduce the risk of predation induced by human scent at nest sites. Nest searches should not be conducted during cold or wet weather when the incubation of eggs and the brooding of young are most critical.

Searching for nests over a large area is very labor-intensive, and alternate survey techniques can be used. Mobbing, agitated calling, and broken-wing distraction displays can indicators of the presence of nests or chicks. Surveys to determine the number of fledged young can also be used to measure productivity. These should generally be conducted no later than mid-July, before local populations begin to fluctuate with the arrival of transient birds from farther north. In central Illinois, upland sandpipers have been known to begin fall migration as early as 9 July (Buss and Hawkins 1939).

Important census data include date, number of individuals or breeding pairs present, number and density of nests, and number of eggs or hatched young. Observations of habitat use and descriptive data taken at nest sites might include date, number and activity of birds, height of vegetation, dominant plant species, percentage of

dead vegetation or litter, and land use (Kirsch and Higgins 1976, Ailes 1980).

### Management Procedures and Programs

Periodic treatment by fire, light grazing, and mowing to remove cover on designated areas may be desirable for the long-term maintenance of suitable upland sandpiper habitat and to maintain grasslands in the best ecological condition (Kirsch and Higgins 1976, Kirsch et al. 1978). The frequency of treatments needed to maintain high-quality vegetation for upland sandpipers will vary over the geographic range and with differences in the successional stage of the vegetation. Seeded grasses, like native grasses, require treatment to maintain plant vigor and retard succession (Kirsch 1974).

**Management on Native Prairies.** - To maintain native mixed-grasslands in prime condition for upland sandpiper nesting, Kirsch and Higgins (1976) recommend rotational burning at 3-year intervals. If burning is not possible, grasslands should remain undisturbed. Recent grassland management studies by Higgins (1986) in North Dakota suggest that, when averaged over the following three to four growing seasons, fall burns may enhance upland sandpiper nest success more than spring burns.

Kirsch (1974) reported gross increases in insect life, especially grasshoppers (Orthoptera), on burned grasslands in North Dakota. Similarly, Queal (1973) noted a greater variety of small insects on burned grasslands in Kansas.

Grasslands managed by fire need periods of rest to allow vegetation regrowth and some residual cover accumulation. Northern grasslands should not be burned at intervals of less than two years (Higgins 1986).

**Management in Grazed Pastures.** - In grazed pastures cattle should optimally be restricted from nesting fields during the egg-laying and incubation periods, 1 May - 15 July. Alternatively nests can be protected from trampling by constructing a fence around the nest site. Only the upland sandpiper's tolerance to disturbance and reluctance to desert makes this management technique possible. In Wisconsin low stakes or

"tepees" were used to exclude animals but they were found to attract cattle as rubbing posts, thus inviting nest destruction (Buss and Hawkins 1939).

Annual grazing of native grasses may not be a suitable management method to control vegetational succession. In Wisconsin, light to moderate grazing did not control encroachment of woody species in natural grasslands. Heavy, prolonged grazing, on the other hand, can lead to destruction of desirable components of prairie vegetation (Tester and Marshall 1962).

**Management in Fields of Domestic Grasses.** - Grasses in nesting fields should be short at the time of spring arrival, from 15-20 cm in height (K. Higgins pers. comm.). All haying operations in nesting fields should be curtailed until after chicks have hatched in mid-July. Fence posts, used by upland sandpipers for display, can be erected where none are available (K. Higgins pers. comm.). An education campaign can help protect nesting sites and keep human-induced mortality of nests and chicks to an absolute minimum.

**Management at Airfields.** - At airfields grasses should be maintained at a height of 20-30 cm over portions of the airfield not directly adjacent to runways or taxiways. Mowing of these areas should be restricted during the nesting and brood-rearing period, 1 May-31 July, to reduce the potential for nest destruction and mortality of incubating adults or flightless young. Maintenance of taller grass on portions of the airfield not directly adjacent to runways provides nesting habitat for upland sandpipers, discourages large concentrations of social flocking birds, such as blackbirds and gulls, and reduces mowing costs.

Instituting a "watch policy" for grounds maintenance personnel helps to minimize destruction of nests and chicks encountered during mowing operations (White and Melvin 1985).

Grassland management programs preserving upland sandpiper habitat have been implemented at Bradley Airport in Connecticut (D. McKay pers. comm.) and Westover Air Force Base in Massachusetts (S. Melvin pers. comm.), and at the Northern Prairie Wildlife Research Center (A. Kruse pers. comm.) and Lostwood National

Wildlife Refuge (K. Smith pers. comm.) in North Dakota.

### Research Needs and Programs

Detailed information on regional breeding trends, including the Canadian provinces and Alaska, is needed to determine the current status of upland sandpiper populations. Information on regional fledging success and characterization of nesting cover are essential to making local management recommendations. Knowledge of upland sandpiper migration patterns and refueling requirements, as well as identification of South American wintering areas and conditions on wintering grounds, are critical to long-term protection of the species.

Several forthcoming papers are expected to address the management of upland sandpiper habitat. B. Bowen and A. Kruse are co-authors of a paper (in prep.) presenting the results of a 7-year study on the effects of grazing on nesting of upland sandpipers in south-central North Dakota mixed-grass prairie. Data from over 600 North Dakota, South Dakota, and southwestern Manitoba upland sandpiper nests has been incorporated into an upcoming publication by H. Kantrud and K. Higgins on the nest site characteristics of ground-nesting birds of the Northern Great Plains grasslands. K. Smith and M. Green are co-authors of an upcoming paper on grassland bird responses to periodic burn and grazing treatments to reestablish native prairie at Lostwood National Wildlife Refuge in North Dakota. Since the refuge's prairie restoration project was initiated in 1979, upland sandpiper nesting densities have increased from 0 to 5-6 nests per 100 acres (40.5 ha) (K. Smith pers. comm.).

P. Vickery, of the University of Maine at Orono, is examining the effects of herbicide and insecticide use on the nesting ecology of grassland birds of the Kennebunk Plains sandplain grasslands and blueberry barrens of southern Maine (P. Vickery per. comm.).

Data from over 600 North Dakota, South Dakota and southwestern Manitoba upland sandpiper nests are being compiled by K. Higgins and H. Kantrud for an upcoming publication on the nesting biology of shorebirds in the Northern

Great Plains. Among the approximately 30 habitat and nesting variables incorporated into the report will be analyses of cover and land use treatments in nesting areas, dominant plant types, percent dead vegetation at nest sites, and size of selected nesting fields (K. Higgins and H. Kantrud pers. comm.).

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# BARN OWL

*Tyto alba*

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In the Northeast, barn owls inhabit agricultural grasslands and tidewater marshlands where they typically nest in tree cavities or in barns, silos, church steeples, warehouses, and other structures. Although little reliable historical data demonstrates that barn owl populations have changed significantly since 1900, limited data suggests a general decline throughout the Northeast and several states consider it as rare and declining. The loss of dense grass habitats for foraging appears to be the species' most significant limiting factor, but this could be overcome by grassland management programs aimed at preserving large fields near to nesting sites. Long-term monitoring of grassland availability and small mammal abundance, as well as barn owl nest sites, should be implemented regionally to track barn owl population trends. Nest boxes can be placed in areas of good foraging habitat to supplement natural nest sites. Grass habitats can also be managed by light grazing or mowing to maintain the grass sere without altering dense ground cover used by small mammals.

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## DESCRIPTION

### Taxonomy and Morphology

The barn owl (*Tyto alba*) occurs across much of the temperate and tropical world. A single subspecies, *T. a. pratincola*, breeds in North and Central America (American Ornithologists' Union 1957).

The barn owl has a 30-37 cm body length and a 104-120 cm wingspread (Colvin 1984, Marti 1990). The upper plumage is golden-brown with varying amounts of gray. The breast and belly color ranges from white to buff and is sparsely to heavily speckled with small black spots. The species lacks ear tufts and has relatively small dark eyes and a distinctive heart-shaped facial disk which ranges in color from white to buff.

The sexes differ in size and plumage. Females are larger and heavier (569 g vs. 475 g), darker, and more heavily speckled than males (Bloom 1978, Colvin 1984). Variability of these characteristics makes it difficult to objectively determine the sex of some individuals. Linear discriminant functions which use weight and color measurements can be used for determining sex (Colvin 1984).

Juvenile barn owls basically resemble adults. Males younger than 1 year may have buff coloration on the breast (whereas adult males almost always lack such coloration) but are not as heavily speckled as females (Bloom 1978). Molt patterns provide a means of distinguishing adults from juveniles and for accurately aging barn owls through 36 months (Bloom 1978).

### Vocalizations

The barn owl produces a wide variety of calls. Fifteen vocal and two nonvocal sounds were described by Bunn et al. (1982). B. Colvin (pers. comm.) described the five most frequently heard vocalizations: 1) the "contact call" is a drawn-out screech frequently given in flight when approaching a nest site from a distance; 2) the "alarm call" is an intense screech made in response to human or other disturbance which is typically given at a nest site and only after chicks have hatched; 3) "squeaking/ticking calls" are rapid, high-pitched notes which are associated with pair bond maintenance or distress situations; these calls are commonly produced during courtship, incubation, and first evening flights after chicks have hatched; 4) "snoring" is a greatly varying hiss which is repeated persistently by juveniles in and out of the nest; this call is used for food begging and may be heard at nest sites from sunset to sunrise; and 5) the "defensive hiss" is a very loud and prolonged hiss typically produced by nestlings when disturbed.

### HABITAT

#### Foraging Habitat

The barn owl relies chiefly on dense grass fields as foraging habitat, including saltmarsh, wet meadows, lightly grazed pastures, grass hayfields, and recently abandoned agricultural fields (Colvin 1980, 1984, 1985; Rosenburg 1986; Gubanyi 1989). Radiotelemetry studies indicate that these habitats are actively selected (Colvin 1984, Rosenburg 1986, Gubanyi 1989). Furthermore, the quantity and quality of dense grass habitats are significantly correlated with barn owl nest activity (Colvin and Hegdal 1988).

Other habitats occasionally used by the barn owl include alfalfa/grass (Colvin 1984), small grain (Ault 1971, Rosenburg 1986), fencelines, and roadsides (Ault 1971, Byrd 1982). In an intensively farmed area in eastern Virginia where grass availability was very low, barn owls foraged in small grain, a 5-year-old clearcut, barnyards, and a pine (*Pinus* spp.) plantation used as a

blackbird roost (Rosenburg 1986). Cultivated habitats in general are of little importance to the barn owl because of low prey populations and/or dense protective cover (Colvin 1984, Rosenburg 1986).

#### Nesting Habitat

The barn owl is a cavity-nesting bird which uses natural as well as human-created cavities. Tree cavities are the principal nest site used by the barn owl in most areas of the Northeast (Colvin et al. 1984); those most frequently used are silver maple (*Acer saccharinum*), American sycamore (*Platanus occidentalis*), and white oak (*Quercus alba*) (Colvin et al. 1984, Byrd and Rosenburg 1986). Although cut bank burrows and cliff recesses are frequently used in the western U.S. (Otteni et al. 1972, Martin 1973, Rudolph 1978, Millsap and Millsap 1987, Gubanyi 1989), only a few cases of the use of such sites have been reported in the Northeast. R. Ferren (pers. comm.) described barn owl nest holes in the steep bluffs on the north and south ends of Block Island, Rhode Island. Recesses in a clay embankment along the Patuxent River in Maryland supported a breeding pair of barn owls during the late 1980s (S. Smith pers. comm.). Exposed barrels in a cut bank along the Rappahannock River of eastern Virginia supported approximately 15 nesting pairs of barn owls in the late 1970s (S. Doggett pers. comm.).

A wide variety of human-made "cavities" are used as nest sites by barn owls. Large platforms within barns and silos, tunnels dug into silage in roofed or topless silos, cavities among hay bales stored inside barns, barn cupola shelves, wooden water tanks, and offshore duckblinds are frequently used; feed bins, church steeples and belfries, platforms within commercial and industrial buildings (e.g., warehouses, grain elevators, mills, factories), attics of abandoned or occupied houses, ledges within chimneys, platforms beneath bridges, and World War II cement watch towers are occasionally used (Stotts 1958, Scott 1959, Reese 1972, Klaas et al. 1978, Soucy 1979, Bunn et al. 1982, Hegdal and Blaskiewicz 1984, Colvin 1984, Byrd and Rosenburg 1986, Matteson and Petersen 1988,

Parker and Castrale 1990). In addition, nest boxes are readily used by the barn owl (Otteni et al. 1972, Marti et al. 1979, Soucy 1980b, Ziesemer 1980, Colvin et al. 1984, Cook 1985, Schulz 1986, Byrd and Rosenburg 1986, Bendel and Therres 1988, Parker and Castrale 1990).

## BIOLOGY

### Reproductive Biology

The barn owl matures and breeds within its first year (Stewart 1952a, Maestrelli 1973, Marti 1990) and sometimes as early as 7 months of age (B. Colvin pers. comm.). The species is typically monogamous, but Colvin and Hegdal (1989) reported that as many as 10% of adult males in their New Jersey study area may be polygynous. The barn owl usually raises one brood of young per year, and occasionally two when prey is abundant. Nests with eggs or young have been found in the northeastern U.S. during every month of the year (Poole 1930, Bent 1938, Scott 1950, Stewart 1952a, C. Rosenburg unpubl. data), but peak egg laying occurs during mid-April (Colvin 1984, Byrd and Rosenburg 1986). Second clutches are typically laid between June and September (Wallace 1948, Keith 1964, Reese 1972, Soucy 1979).

Clutch size ranges between 1-13 eggs (Bent 1938, Parker and Castrale 1990) with the mean clutch size ranging between 4-6 eggs (Otteni et al. 1972, Reese 1972, Smith et al. 1974). Eggs are usually laid two days apart and hatch asynchronously since incubation starts after the laying of the first egg (Wallace 1948, Smith et al. 1974). The incubation period is 30-31 days (Smith et al. 1974, Marshall et al. 1986). The peak of hatching in the Northeast occurs in mid-May (Colvin 1984, Byrd and Rosenburg 1986, S. Smith pers. comm.). Young fledge at 8-10 weeks of age (Pickwell 1948, Reese 1972, Smith et al. 1974). Peak fledging occurs in mid to late July (Colvin 1984, Byrd and Rosenburg 1986). Juveniles may remain in the vicinity of the nest site for several weeks before dispersing (Otteni et al. 1972, Smith et al. 1974, Marti 1990).

### Dispersal and Migration

Hatching-year barn owls have been recovered great distances from natal areas (commonly > 80 km and as much as 1800 km) (Stewart 1952a; Soucy 1980a, 1985). Although juveniles have been recovered from essentially every compass direction from their natal area, most had traveled in a southerly direction (Stewart 1952a). Stewart suggests that juveniles in the northern U.S. migrate south but return to nest somewhere within 320 km of their natal sites. Marti (1990) reported that most individuals banded as nestlings and later found breeding did so at distances of about 50 km from their natal areas. Cases of dispersal > 320 km have also been documented. Ehresman et al. (1989) reported that an individual banded as a nestling in southwestern Iowa was recovered as a breeding adult 419 km to the east. A nestling banded in central New Jersey was found nesting in Ohio (B. Colvin pers. comm.). Extensive banding of nestlings and capture of adults in southwest New Jersey reveals that only a small percentage of nestlings banded within the study area enter the adult population there: 5% of 181 nestlings banded in 1988 were found in the adult population in 1989 (Colvin and Hegdal 1989). Although barn owls may return to breed relatively close to their natal area, individuals frequently become established great distances away. The barn owl is very successful at colonizing new areas because of this broad dispersal behavior.

Stewart (1952a) described the barn owl as partly migratory in the Northeast and presents extensive evidence of barn owls banded in northern latitudes that were later recovered in southern latitudes. Duffy (1985) reported that large numbers of migrant barn owls are captured during fall night trapping at Cape May, New Jersey. Although most captured owls were juveniles, 20% of 171 captures at Cape May were adults (P. Kerlinger pers. comm.). Most adult barn owls migrate past Cape May early in October while hatching-year birds migrate throughout October and early November (Duffy 1985). Recoveries of barn owls banded in the northeastern U.S. indicate that migrant individuals winter chiefly in the southeastern U.S. and Texas (Stewart 1952a, Soucy 1980a).

Northeastern U.S. Christmas Bird Count data (Stewart 1980, Butcher and Lowe 1990) demonstrate that not all individuals near the northern edge of their range leave for the winter; numerous counts from all northeastern states except Maine, New Hampshire, and Vermont reported barn owls every year. During the winter of 1989-90, 74% of 35 active Virginia barn owl sites visited supported wintering owls (C. Rosenburg unpubl. data). Although there is an observed migration of both juvenile and adult barn owls, many individuals winter within the northeastern U.S. There seem to be no obvious migration patterns (e.g., coastal vs. inland populations, mild winters vs. severe winters) within the northeastern U.S.. However, no investigations have been made into the possible influence of microtine rodent abundance on the percentage of the population which apparently migrates.

#### Food Habits

The barn owl in North America is highly nocturnal (Colvin 1984, Rosenburg 1986). Individuals range over large areas; mean home range size (based on the minimum home range method (Mohr and Stumpf 1966)) has been reported as 355 ha in southern Texas (Byrd 1982), 757 ha and 921 ha in southwestern New Jersey (Colvin 1984, Hegdal and Blaskiewicz 1984), 414 ha in eastern Virginia (Rosenburg 1986), and 198 ha in western Nebraska (Gubanyi 1989). Overlap of individual owl home ranges is common (Smith et al. 1974, Colvin 1984, Rosenburg 1986). The barn owl may travel as much as 5.6 km between its nest site and foraging areas, although distances within 1.6 km are more usual (Colvin 1984, Hegdal and Blaskiewicz 1984, Rosenburg 1986).

The barn owl typically forages with a relatively low quartering flight which includes frequent hovering intervals (Honer 1963, Haverschmidt 1970, Burton 1973, Karalus and Eckert 1974, Marti 1974, Rudolph 1978, Bunn et al. 1982, Mikkola 1983, Rosenburg 1986). Some individuals also hunt rather frequently from a perch, especially along field edges (Byrd 1982, Rosenburg 1986). The barn owl has extremely keen hearing (Payne 1971, Konishi 1973) and

night vision (Dice 1945, Marti 1974). Its ability to capture prey by hearing alone (Payne 1971) is especially advantageous for hunting animals such as voles (*Microtus* spp.) and shrews (Soricidae) which are often concealed from view as they travel in runways beneath grass cover.

Numerous pellet analyses throughout north temperate North America and Europe have identified microtines as the barn owl's primary prey (Ticehurst 1935; Wilson 1938; Pearson and Pearson 1947; Wallace 1948; Phillips 1951; Boyd and Shriner 1954; Glue 1967, 1974; Smith et al. 1972; Marti 1973; Webster 1973; Jackson et al. 1976; Lovari et al. 1976; Dexter 1978; Bethge and Hayo 1979; Colvin 1980, 1984; Hegdal and Blaskiewicz 1984; Cook 1985; Colvin and McLean 1986; Rosenburg 1986; Campbell et al. 1987; Feldhamer et al. 1987; Parker 1987; Hammerson 1988; Marti 1988). The meadow vole (*Microtus pennsylvanicus*) is the most important prey animal in the northeastern U.S. and the short-tailed shrew (*Blarina brevicauda*) is an important secondary prey; by frequency, meadow voles typically comprise 60-90% of the barn owl's diet (Boyd and Shriner 1954, Jackson et al. 1976, Colvin 1984, Cook 1985, Rosenburg 1986, Hammerson 1988). The marsh rice rat (*Oryzomys palustris*) is occasionally an important prey animal in coastal areas (Jemison 1962, Blehm and Pagels 1973, Jackson et al. 1976, Colvin 1984, Feldhamer et al. 1987). In the southern U.S., the cotton rat (*Sigmodon hispidus*) is the primary prey (Baumgartner and Baumgartner 1944, Parmalee 1954, Otteni et al. 1972, Hamilton and Neill 1981, Byrd 1982, Baker 1986, Marra et al. 1989).

The foraging behavior of the barn owl has been closely studied (Colvin 1980, 1984). Colvin (1984) concluded that barn owls seek prey of a particular size (approximately 40-60 g), which provides the most energy efficient diet. The average weights of the meadow vole, marsh rice rat, cotton rat, and Norway rat (*Rattus norvegicus*) juveniles fall into this size range.

The barn owl shows greater diet diversity: 1) in areas with relatively low microtine or cotton rat availability (Ticehurst 1935; Hawbecker 1945; Pearson and Pearson 1947; Glue 1967, 1974; Blehm and Pagels 1973; Marti 1974; Bauer 1983; Colvin 1984; Lenton 1984; Colvin and McLean 1986; Rosenburg 1986; Parker 1987; Campbell et al.

1987); 2) during times of poor microtine availability (Fitch 1947, Wallace 1948, Glue 1967, Otteni et al. 1972, Webster 1973, Marti 1974, Jackson et al. 1976, Bethge and Hayo 1979, Baker 1986); and 3) when nonmicrotine prey are readily available (Evans and Emlen 1947, Sage 1962, Carpenter and Fall 1967, Smith et al. 1972, Klaas et al. 1978, Byrd 1982, Fritzell and Thorne 1984, Rosenberg 1986, Jentsch 1988). These studies identified birds (mostly blackbirds and sparrows), short-tailed shrews, least shrews (*Cryptotis parva*), house mice (*Mus musculus*), and Norway rats as relatively important prey in such situations.

#### POPULATION DISTRIBUTION, STATUS, AND TRENDS

##### Distribution and Status in North America

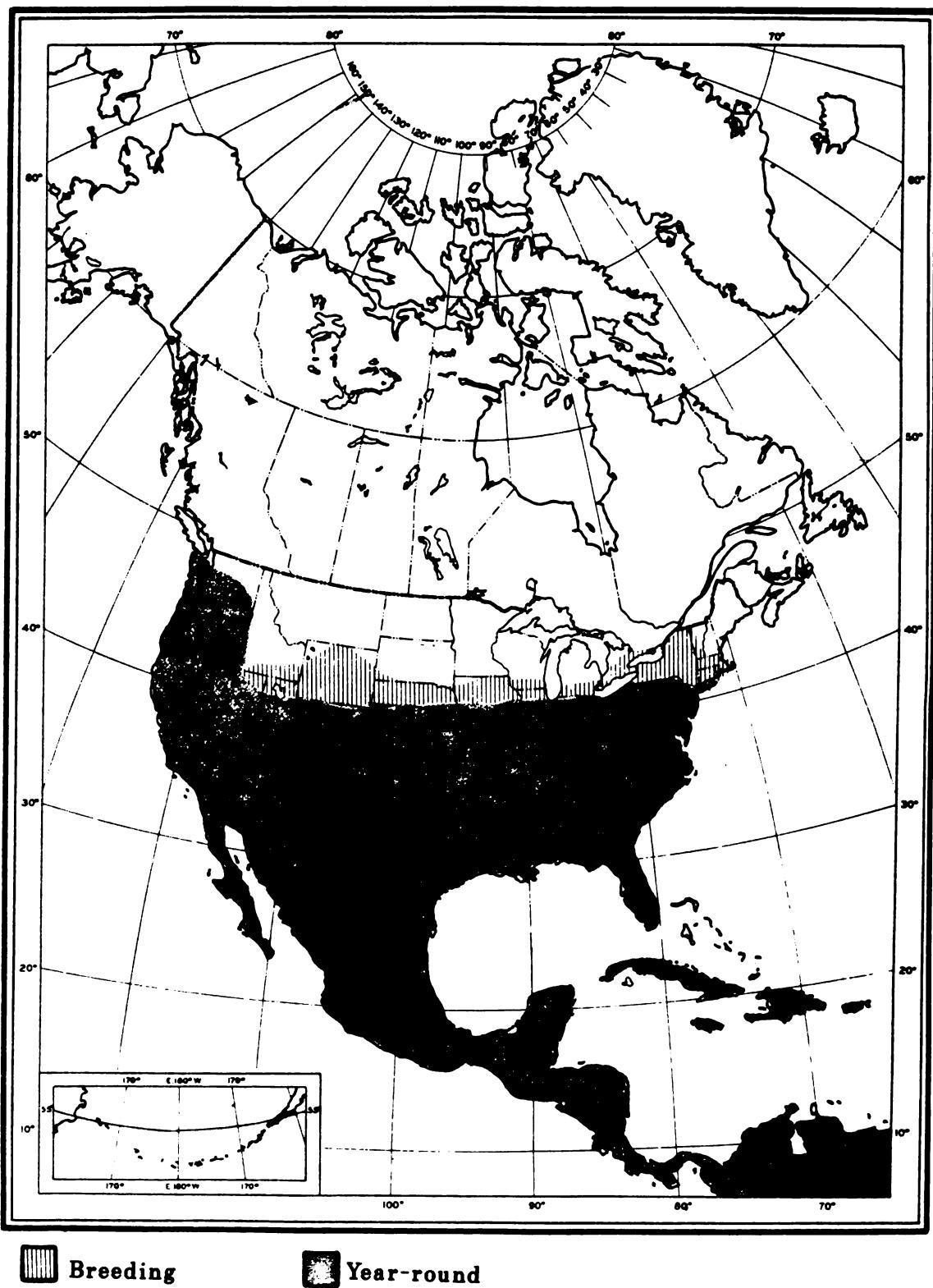
Barn owls are found throughout the tropical and temperate latitudes of the world. The North American subspecies, *T. a. pratincola*, occurs from Central America to approximately 50° north latitude along the west coast and to approximately 44° north latitude throughout the remainder of the continent (Figure 1). The barn owl is considered a *migratory nongame bird of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987). The Nature Conservancy's Natural Heritage Program global ranking is G5 (globally secure) (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia). It was blue-listed from 1972-81 and classified as *special concern* from 1982-86 by the National Audubon Society (Tate 1986). Western U.S. populations appear to be mostly stable (Marti and Marks 1987). Populations have declined severely in the Midwest and the species is classified as *endangered* by seven of the eight states which comprise the Northcentral Region (Hands et al. 1989). Four of the 10 states in the Southeast Region classify the barn owl as *special concern*.

##### Distribution and Status in the Northeast Region

Every state in the Northeast Region has breeding records of barn owls, although very few are from Maine, New Hampshire, and Vermont. Breeding records also occur from southern Ontario (Weir 1987) and southern Quebec (Bull 1974). The barn owl's distribution in the Northeast, based on BBA data (Figure 2), is very similar to a map of its distribution based on 1963-87 Christmas Bird Count (CBC) data (Butcher and Lowe 1990). Throughout the Northeast, barn owl populations are clearly associated with tidewater marshlands and agricultural grasslands, where flourishing microtine populations are found (Blodget 1988). Records from tidewater marshlands extend as far north as northeastern Massachusetts. Interior records extend as far north as the Lake Champlain lowlands of central Vermont and the St. Lawrence River Valley near Montreal. The distribution of interior records is closely associated with valley areas, especially several major river valleys (Merrimack, Connecticut, Hudson, Genesee, Ohio, Monongahela, and Kanawha) and the Great Valley (which includes the Lehigh, Lebanon, Frederick, Hagerstown, and Shenandoah valleys). Harsh climatic conditions are a formidable barrier north of southern New England and in northern interior areas (Blodget 1988); Martha's Vineyard data demonstrate the susceptibility of northeastern populations to extensive winter mortality and poor breeding success in subsequent years (see *Limiting Factors and Threats*).

Christmas Bird Count data were analyzed to determine population trends for the barn owl throughout North America (Butcher and Lowe 1990). The analyses of data from 1979-87 identified a nonsignificant decline ( $P > 0.05$ ) for North America and for the Northeast Region. Significant declines ( $P < 0.05$ ) were identified for Connecticut, Pennsylvania, Maryland, and West Virginia (Table 1). Some problems are associated with using CBC data for monitoring population trends for the barn owl (Butcher and Lowe 1990), but regardless, the results of their analyses suggest barn owl population declines and support

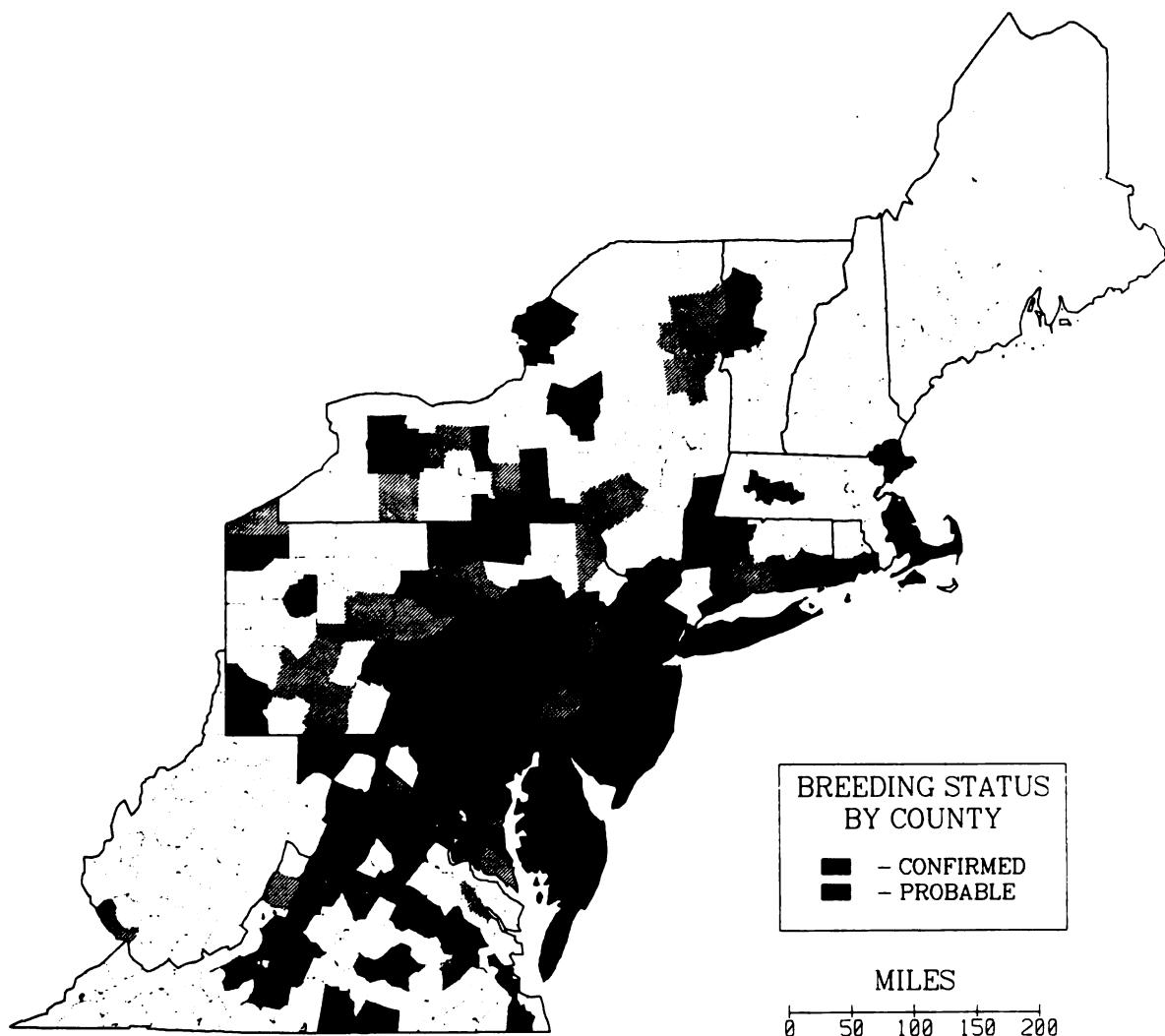
Figure 1. The North American range of the barn owl (*Tyto alba*) (American Ornithologists' Union 1983, National Geographic Society 1987).



Breeding

Year-round

**Figure 2.** Northeast breeding distribution of the barn owl (*Tyto alba*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding, probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



**Table 1.** Barn owl (*Tyto alba*) population trends in the Northeast based on 1979-1987 Christmas Bird Count (CBC) data (Butcher and Lowe 1990).

Area	No. CBCs <sup>a</sup>	Median total trend <sup>b</sup>
<b>State</b>		
Maine	0	-
New Hampshire	0	-
Vermont	0	-
Massachusetts	7	-2
Rhode Island	3	*+444
Connecticut	6	**-46
New York	14	-3
Pennsylvania	22	***-47
New Jersey	19	+15
Delaware	5	***-23
Maryland	17	***-61
Virginia	20	+20
West Virginia	2	***-22
Northeast Region	116	-12
North America	463	-4

<sup>a</sup>Number of CBC locations for which route regressions were calculated.

<sup>b</sup>Total trend estimates as calculated by the route regression method. The median trend is calculated by a bootstrapping procedure. Statistical significance of trend: \*  $P < 0.10$ , \*\*  $P < 0.05$ , \*\*\*  $P < 0.01$ .

continued attention to this bird's population trends.

There is considerable concern for the status of the barn owl in the Northeast; 8 of the 13 states in the region classify the barn owl as *threatened* or *special concern* and 9 states assign it a state Natural Heritage ranking of *S3* (rare) or higher (Table 2). However, Blodget (1988) found little reliable data demonstrating that barn owl populations throughout most of the Northeast have changed very drastically since 1900. The absence of base-line population data indicates a pressing need for a region-wide population monitoring scheme applied consistently over a long period of time (Blodget 1988).

**Table 2.** Summary of state listing status and Natural Heritage Program state ranks for the barn owl (*Tyto alba*) in the Northeast (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	SA
New Hampshire	-	SA
Vermont	special concern	S1
Massachusetts	special concern	S2
Rhode Island	threatened	S1
Connecticut	threatened	S1/S2
New York	special concern	S3
Pennsylvania	special concern	S3
New Jersey	-	S4
Delaware	-	S3?
Maryland	-	S5
Virginia	special concern	S3
West Virginia	special concern	S3

<sup>a</sup>SA = accidental

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

S5 = demonstrably secure

**Maine.** - The barn owl is considered to be an accidental species in Maine. There is only one documented nesting record for the state which occurred in Westbrook (near Portland, close to Maine's southern coast) in 1960 (Werner 1960). No records for the barn owl were reported during the state's breeding bird atlas (BBA) (Adamus 1988). This species has no official state listing. Its state Natural Heritage ranking is *S4* (accidental) (Table 2).

**New Hampshire.** - The barn owl is considered to be an accidental species in New Hampshire. There are several nonbreeding records, mostly from within the Connecticut and Merrimack River valleys, and one documented nesting record from Hollis (near Nashua, within the Merrimack Valley) in 1977 (Audubon Society of New Hampshire unpubl. data, Concord). No records for the barn owl were reported during the state's

BBA (C. Foss pers. comm.). The species has no official state listing and its state Natural Heritage ranking is S4 (accidental) (Table 2).

**Vermont.** - The barn owl is a very rare breeder in Vermont with few nesting records known prior to the Vermont BBA's intensive survey efforts (Laughlin 1985). Nonbreeding records exist from scattered valley areas of the state (Forbush 1927, Laughlin 1985). The barn owl was a confirmed breeder in four (< 1%) and a probable breeder in zero blocks of the state's BBA; all four confirmed blocks are located within Lake Champlain lowlands (Laughlin 1985). The barn owl is state listed as *special concern* (Table 2). Its state Natural Heritage ranking is S1 (critically imperiled). None of the four nesting sites identified during the BBA are currently active (N. Martin pers. comm.).

**Massachusetts.** - The barn owl is a rare breeder in Massachusetts and is limited mostly to the coast (Forbush 1927, Keith 1964, Blodget 1988). It occurs mostly in southeastern Massachusetts northward sporadically to the lower Merrimack Valley of northeastern Massachusetts. In the interior of the state, the species is an extremely rare visitor and nester and is mostly encountered in the Connecticut River Valley (Blodget 1988). The barn owl was a confirmed breeder in 11 and a probable breeder in 2 of the state's BBA blocks (J. Baird pers. comm.); all blocks are coastal except one which is located near Amherst in the Connecticut River Valley. The barn owl is state listed as *special concern* (Table 2). Its state Natural Heritage ranking is S2 (imperiled). Except in southeastern Massachusetts, where the species has achieved notable stability benefiting from relatively mild winters, expansive microtine habitat, and (at Martha's Vineyard) an aggressive nest box program, the species has not successfully established itself over most of the state (Blodget 1988).

**Rhode Island.** - The barn owl has always been considered an uncommon nester in Rhode Island where it is usually found at sites near the coast; it is most stable on Block Island, but also found in low numbers on Aquidneck Island (R. L. Ferren

pers. comm.). It was a confirmed breeder in four (3%) and a probable breeder in zero of the state's BBA blocks; all confirmed blocks are located on Block and Aquidneck Islands (R. Enser pers. comm.). The barn owl is state listed as *threatened* (Table 2). Its state Natural Heritage ranking is S1 (critically imperiled). Its status in Newport County is in jeopardy because of expansive residential development of agricultural lands (R. Ferren pers. comm.).

**Connecticut.** - The barn owl is a rare breeder found principally along the coast and within large river valleys of Connecticut. It was a confirmed breeder in six (1%) and a probable breeder in six (1%) of the state's BBA blocks (Bevier et al. in press). Most of these blocks are located along the coast or near Middletown (Connecticut River Valley) where there is an active monitoring and nest box program (G. Zepko pers. comm.). The barn owl is listed as a *threatened* species in the state (Table 2). Its state Natural Heritage ranking is S1/S2 (critically imperiled/imperiled). The barn owl occurs in low numbers probably because grasslands and farmlands are declining in the state (Connecticut Geological and Natural History Survey unpubl. data, Storrs). Its historic population change in Connecticut, if any, is imperfectly understood because it is a very difficult species to detect (Zeranski and Baptist 1990.)

**New York.** - The barn owl is an uncommon breeder in New York State and it is most often found on Long and Staten Islands, in the Hudson and Genesee River valleys, in the Finger Lakes Region, and in the Lake Erie and Lake Ontario lowlands (Eaton 1914, Beardslee and Mitchell 1965, Bull 1974). It was a confirmed breeder in 46 (1%) and a probable breeder in 18 (< 1%) of the state's BBA blocks. These blocks are distributed throughout lowland areas of New York with the heaviest concentration on Long and Staten Islands, the Hudson and Genesee River valleys, and the Finger Lakes Region (Sibley 1988). The barn owl is state listed as *special concern* (Table 2). Its state Natural Heritage ranking is S3 (rare). Sibley (1988) presents evidence that this species has declined somewhat in the past century in agricultural

regions of northern and western New York: a number of early breeding records were cited by Eaton (1914) and Bull (1974) from Lake Ontario and Lake Erie lowlands where only four recent atlas blocks had "possible" records.

**Pennsylvania.** - The barn owl is a fairly common breeder found primarily in the western counties and the southeastern third of Pennsylvania, with occasional records from the valleys of the Ridge and Valley Province (Todd 1940, Poole 1964, Gill 1985). It was a confirmed breeder in 106 (2%) and a probable breeder in 43 (1%) of the state's BBA blocks. These blocks are concentrated in the Lebanon and Lehigh valleys as well as other broad valleys of the Ridge and Valley Province and the adjacent Piedmont of southeastern Pennsylvania (Brauning in press). The barn owl is state listed as *special concern* and its state Natural Heritage ranking is S3 (rare) (Table 2). The loss of farmland in some areas has limited the population, and the construction of "pigeon-proof" barns may also have had an adverse impact on this species (Gill 1985). The feeling among most observers is that there has been a decline in the commonwealth, but this needs more study (Gill 1985).

**New Jersey.** - The barn owl breeds in every county of the state (Soucy 1983). It is abundant in Somerset and Hunterdon counties (northcentral New Jersey) where a long-term nest monitoring program has been underway since 1970 (L. Soucy pers. comm.). It is also abundant in western Salem and Cumberland counties (southwestern New Jersey) where extensive research programs were begun in 1980 (Hegdal and Blaskiewicz 1984, Colvin 1984). The species is reported to be particularly scarce in the northwestern part of the state (Leck 1984). The barn owl has no state listing status in New Jersey and its state Natural Heritage ranking is S4 (apparently secure) (Table 2).

**Delaware.** - The barn owl is a fairly common permanent resident of Delaware and is primarily restricted to saltmarshes and adjacent farms (Hess et al. in press). It was a confirmed breeder in 28 (13%) and a probable breeder in 4 (2%) of the state's BBA blocks (Hess et al. in press). It has

been assigned no state listing. Its state Natural Heritage ranking is S3? (questionably rare) (Table 2). The barn owl's winter numbers have apparently increased substantially, based on Christmas Bird Count data, but it may have been more common inland earlier in the century (Hess et al. in press).

**Maryland and the District of Columbia.** - The barn owl is a fairly common permanent resident found locally on the Eastern Shore of Maryland (Stewart and Robbins 1958, Reese 1972). It is uncommon in the Western Shore, Upper Chesapeake, Piedmont, and Ridge and Valley sections and rare in the Allegheny Mountain section (Stewart and Robbins 1958). Nest site data from the early 1970s (Klaas et al. 1978) indicate that it was once relatively common within the lower Potomac estuary of the western shore of Chesapeake Bay. The barn owl was a confirmed breeder in 79 (6%) and a probable breeder in 20 (2%) of the state's BBA blocks. These blocks are distributed mostly along the Atlantic Coast, Chesapeake Bay shoreline, and within the Frederick and Hagerstown valleys (G. Therres pers. comm.). The barn owl is assigned no official state status. Its state Natural Heritage ranking is S5 (demonstrably secure) (Table 2). Recent surveys of duck blinds in Dorchester and Worcester counties on the Eastern Shore (Bendel and Therres 1988) documented very little use by barn owls. Reese (1972) had previously documented extensive use of blinds by barn owls in this area.

**Virginia.** - The barn owl is an uncommon to rare permanent resident throughout much of Virginia, although healthy populations exist in various localities of the Great Valley (especially the Shenandoah Valley section), northern Piedmont, and coastal marshlands (Rosenburg in press). Recent nesting attempts are known from 49 (52%) of the state's counties. The barn owl was a confirmed breeder in 89 (5%) and a probable breeder in 9 (< 1%) of the state's BBA blocks (S. Ridd and R. Wajda pers. comm.). The barn owl is proposed for state *special concern* status (Table 2). Its state Natural Heritage ranking is S4 (apparently secure). Only 39% of 111 sites that were known to support nests

between 1976-85 were still active in 1986 (Byrd and Rosenburg 1986). Many of the sites which were inactive in 1986 were lost because the nest site was no longer available or the surrounding habitat was degraded by development or intensified agricultural use. During the 1989 breeding season, 76 nest sites (46 of which were in nest boxes) were known to be active (Rosenburg 1990a).

**West Virginia.** - The barn owl is an uncommon permanent resident which probably occurs throughout the state except at higher elevations (Hall 1983). The location of ten counties from which nests have been reported (Hall 1983) indicate an association with the Ohio, Monongahela, and Kanawha River valleys and the Great Valley of extreme eastern West Virginia. The barn owl was a confirmed breeder in three (1%) and a probable breeder in one (< 1%) of the state's BBA blocks. These four blocks are located in the eastern panhandle (Great Valley) area of the state (C. Stihler pers. comm.). It is state listed as *special concern* and ranked S3 (rare) by the West Virginia Natural Heritage Program (Table 2). Little is known about the actual status of the barn owl in West Virginia, although populations seem to have declined slightly in recent years. The concern for this owl is primarily a result of its low numbers and its vulnerable position as a top predator (West Virginia Department of Natural Resources 1987).

## LIMITING FACTORS AND THREATS

### Habitat Availability

Foraging habitat availability appears to limit barn owl numbers most significantly (Colvin et al. 1984, Colvin 1985, Rosenburg 1986, Ehresman et al. 1989, Gubanyi 1989). Loss of farmland to development and the intensification of agricultural practices on remaining farmlands have substantially reduced the quantity and quality of dense grass habitats in agricultural areas (Honer 1963, Shrub 1970, Colvin 1985, Rosenburg 1986). The availability of pasture and grass hayfield has been reduced substantially. Colvin (1985)

reported an approximately 53% decline in hayfield acreage in Ohio between 1921-80 and identified a significant correlation between a barn owl population decline in Ohio and the replacement of grass-associated agriculture by row crops. In Virginia, the acreage of pasture, grass hayfield, and idle areas was reduced 55% between 1945-78 and the quality of pasture declined because of increased grazing pressure (Rosenburg 1986). In many intensively-farmed areas, dense grass habitats are present only in small fields that are patchily distributed; such fields would apparently provide a limited foraging resource (Rosenburg 1986).

The reduced quality and quantity of dense grass habitats has substantially reduced prey availability in some areas. Prey availability has been shown to be closely associated with barn owl productivity (Ault 1971, Otteni et al. 1972, Colvin 1984, Gubanyi 1989). This association is so close that one year of poor meadow vole abundance can result in a rapid population decline while one year of substantial meadow vole abundance can result in rapid population recovery (B. Colvin pers. comm.).

### Nest Site Availability

The availability of secure nest sites is an important limiting factor in some areas (Marti et al. 1979, Schulz and Yasuda 1985, Byrd and Rosenburg 1986, Gubanyi 1989). Tree cavity sites may be limited in availability (Rosenburg 1986), plus they are ephemeral (Colvin et al. 1984, Byrd and Rosenburg 1986), and relatively insecure (Colvin et al. 1984). Competition for this resource may also be a factor; Colvin (1984) documented usurpation of barn owl nest cavities by wood ducks (*Aix sponsa*) and raccoons (*Procyon lotor*). Secure nest sites within human-made structures are limited in availability (Schulz 1986, C. Rosenburg unpubl. data). In addition, the gradual deterioration and disappearance of old-style barns, silos, and water tanks plus the screening of entrances to prevent rock dove (*Columba livia*) access has eliminated many previously productive nest sites (Honer 1963, Heintzelman 1966, Schulz and Yasuda 1985, Byrd and Rosenburg 1986, Parker and Castrale 1990).

Areas that support abundant foraging habitat may lack an adequate supply of secure and stable nest sites in close proximity to foraging habitat.

### Diseases, Parasites, and Predation

Disease, parasites, and predation are natural factors that may in part limit barn owl populations. The barn owl appears to be resistant to many diseases that infect other raptors (Schulz 1986). In California, diseases documented for the barn owl include tuberculosis, aspergillosis, and trichomoniasis (Schulz 1986). Toxoplasmosis and eastern equine encephalitis have been detected in New Jersey barn owls, although no impact to the birds was apparent (Colvin and Hegdal 1986b, 1987). Salmonellosis has been recorded in Pennsylvania (Locke and Newman 1970) and New Jersey (Kirkpatrick and Colvin 1986). Kirkpatrick and Colvin (1986) found *Salmonella*-positive nestlings at 5 of the 25 New Jersey nest sites examined, and reported that all infected young apparently fledged. They suggested that salmonellosis may limit barn owl survival and reproduction at times when stresses, such as severe weather and poor prey availability, are acting. This stress dependent impact may be true for other diseases and parasites as well (Honer 1963, Kirkpatrick and Colvin 1989).

Dipteran ectoparasites and lice have been found on the barn owl (Schulz 1986, Kirkpatrick and Colvin 1989). The endoparasites *Trypanosoma*, *Capillaria*, and *Porrocaecum* have been identified from the feces of New Jersey barn owls (Colvin and Hegdal 1986b).

Predation may limit barn owl numbers in some areas. Raccoons and black rat snakes (*Elaphe obsoleta*) prey on eggs and nestlings (Ehresman 1984, B. Colvin pers. comm., C. Rosenburg unpubl. data), and great horned owls (*Bubo virginianus*) prey on juveniles and adults (Rudolph 1978, Knight and Jackman 1984, Lerg 1984, Rosenburg 1986, Millsap and Millsap 1987, Ehresman et al. 1989) and may inhibit barn owl activity because of their dominance (Rudolph 1978). Information concerning predation rates on eggs, nestlings, juveniles, and adults is limited and further investigation is warranted (Hands et al. 1989).

### Climate

Weather is the most important factor influencing annual productivity of the barn owl in southwestern New Jersey (B. Colvin pers. comm.). Moist weather conditions enhance dense grass habitats and thereby enhance vole populations, which results in higher barn owl productivity. Exceedingly dry conditions have a negative impact on vole populations and thereby result in poor barn owl productivity (Colvin and Hegdal 1986b, 1987, 1988, 1989). Starvation of chicks, the single most important barn owl mortality factor (B. Colvin pers. comm.), is widespread during exceedingly dry conditions.

Throughout the Northeast, the barn owl is susceptible to starvation and exposure during extended periods of extreme cold and deep snow cover. Winter weather mortality has been documented in Wisconsin (Errington 1931), Illinois (Speirs 1940), Ohio (Stewart 1952b), Massachusetts (Keith 1964), Utah (Marti and Wagner 1985), and Virginia (C. Rosenburg unpubl. data). Marti and Wagner (1985) reported that some winter weather mortality appears to occur every winter in northern Utah and that such mortality is widespread and consequential in some years. They found 77 dead barn owls after severe weather during the winter of 1981-82, and a 40% decline in breeding attempts occurred the following summer. Keith (1964) documented winter weather mortality of more than 10 owls during the winter of 1960-61 and the subsequent lack of nesting during the following 3 breeding seasons at Martha's Vineyard, Massachusetts. Barn owls did not nest at Martha's Vineyard again until 1973 (Brett 1987).

### Pesticides

Secondary poisoning from rodenticides has been considered to be a potential hazard to the barn owl because of the importance of rodents in the barn owl's diet and the fairly widespread use of rodenticides in agricultural areas. Laboratory studies of barn owls have demonstrated that the consumption of rats or mice poisoned with bromadiolone or brodifacoum rodenticides can cause lethal hemorrhaging and that consumption

of rats poisoned with difencoum rodenticide can cause sublethal hemorrhaging (Mendenhall and Pank 1980, Newton et al. in press). These studies demonstrated that the barn owl is especially sensitive to the anticoagulant brodifacoum. Secondary poisoning of wild barn owls from rodenticides has been documented in the U.S. (Schulz 1986, L. Soucy pers. comm.) and Great Britain (Newton et al. in press). The potential for poisoning appears to be greatest in marginal habitat areas such as intensively farmed sites (Rosenburg 1986). However, there appears to be no appreciable impact to barn owl populations from rodenticide poisoning (Colvin 1984, Hegdal and Blaskiewicz 1984, Newton et al. in press).

Organophosphate insecticides have been shown to be potentially hazardous to the barn owl. Laboratory experiments conducted by Hill and Mendenhall (1980) demonstrated that barn owls which consumed famphur-poisoned prey exhibit secondary poisoning in the form of significant cholinesterase inhibition. Mass mortality of wild raptors, including 22 barn owls, occurred after azodrin was improperly used to kill voles in Israel (Mendelsohn and Paz 1977). Rodents contaminated with organophosphate and carbamate insecticides are present in agricultural fields (Montz 1988). These rodents are potentially hazardous to raptors because birds in general are extremely sensitive to anti-cholinesterase compounds (Brealey et al. 1980). It is unlikely that organophosphate or carbamate insecticides have impacted barn owl populations since these pesticides are not targeted for barn owl foraging habitats or prey species (B. Colvin pers. comm.). However, no field studies have examined cholinesterase levels of barn owls in agricultural areas where these pesticides are widely used. Further investigation may be warranted (L. Brewer pers. comm.).

The barn owl is sensitive to contamination from organochlorine insecticides. Laboratory studies by Mendenhall et al. (1983) found that the barn owl is very sensitive to eggshell thinning by DDE and that dieldrin can cause adult barn owl mortality. These insecticides were found in concentrations that may have been detrimental to barn owl reproduction in 15% of the barn owls in the lower Potomac River, Maryland in the early 1970s (Klaas et al. 1978). Extensive feeding on

passerine birds by this portion of the population is believed to have caused the elevated organochlorine levels; the majority of the population preyed chiefly on mammals and remained relatively uncontaminated. In Great Britain, poisoning from organochlorine pesticides was an important cause of mortality during the years 1963-77 when these chemicals were used extensively (Newton and Wyllie in press). Organochlorine insecticide residues have been found in barn owls or barn owl eggs collected in Florida (Johnston 1978), Oregon (Henny et al. 1984), and Virginia (Gwynn 1987). Acute effects to the barn owl from organochlorine insecticides are unlikely, though, since raptors which feed chiefly upon small mammals are not highly susceptible to organochlorine insecticide poisoning (Henny 1972) and since few potentially harmful organochlorine insecticides are in use in the U.S. today (Newton 1979, S. Wiemeyer pers. comm.).

#### Other Factors

A number of other barn owl mortality factors have been identified. Collision with vehicles has been reported as an important mortality factor (Glue 1971, Smith and Marti 1976, Keran 1981, Schulz 1986, Newton and Wyllie in press). Drowning of young as they attempt to fledge from offshore duck blinds appears to be acting as a population sink in coastal Maryland and possibly other areas where substantial numbers of barn owls nest in these structures (G. Therres pers. comm.). Other mortality factors include electrocution, entrapment in buildings, shootings, and entanglement in farm or industrial machinery (Glue 1971, Smith et al. 1974, Smith and Marti 1976, Keran 1981, Colvin 1984, Hegdal and Blaskiewicz 1984, Lerg 1984, Schulz and Yasuda 1985, Schulz 1986, Ehresman et al. 1989, Newton and Wyllie in press, C. Rosenburg unpubl. data). The degree to which these latter four factors limit barn owl numbers appears to be low.

## MANAGEMENT POTENTIAL

Barn owls have a high recovery and management potential. The species has a potentially high reproductive output because of its 1) large clutch size, 2) occasional second broods, 3) sexual maturity at one year, 4) lack of strict territoriality, and 5) occasional polygyny. These characteristics provide mechanisms for rapid population expansion during times of prey availability (Wallace 1948, Stewart 1952a, Henny 1969, Colvin 1984). This potentially high reproductive output, coupled with the barn owl's colonizing abilities (see *Biology*), indicates that the barn owl can establish or expand populations in areas where stable habitats exist. This ability was demonstrated during the first part of this century when the barn owl rapidly expanded its range into the Midwest in response to the clearing of forests and the establishment of agriculture which supported an abundance of dense grass habitats (B. Colvin pers. comm.).

In addition, the major biological and ecological limiting factors to the barn owl are fairly well understood and management techniques have been developed. Therefore, intelligent recovery programs that effectively increase barn owl populations are possible.

## LAND PROTECTION AND PRESERVE DESIGN

The most important land protection tool for maintaining or expanding barn owl populations is the preservation of dense grass foraging habitats (Lerg 1984, Rosenburg 1986, Gubanyi 1989, B. Colvin pers. comm.). Acquisition of fee title or the use of conservation easements, management agreements, property registration, or tax incentives (Millsap et al. 1987) are means of preserving existing grasslands or acquiring additional land, such as cropland, which can be converted to grassland. Large-scale habitat acquisition programs are very costly. Programs which utilize other preservation strategies, such as conservation easements and tax incentives, and which focus on all of the grassland dependent

species of management concern may be cost effective.

When preserving grassland areas for the barn owl, large areas of quality habitat should be established. Habitat use studies have not adequately identified the area of dense grass habitats necessary to support a breeding pair of barn owls in coastal and agricultural areas of the Northeast, but evidence indicates that an area between 60-260 ha of dense grass habitats is necessary (Colvin 1984, Rosenburg 1986, Gubanyi 1989). Further quantification of this habitat requirement is needed. The necessary area is difficult to quantify, of course, since prey abundance varies between years, between grass habitat types, and even between different fields of a particular grass type. However, extensive comparisons of dense grass acreage surrounding successful barn owl nest sites will provide a useful estimate of the acreage needed (see *Research Needs and Programs*).

Areas large enough to support more than one nesting pair of barn owls should be preserved when possible since such areas would accommodate a more stable population. An area of 10 km<sup>2</sup> with abundant foraging habitat and nest sites is capable of supporting 3-6 nesting pairs (see *Management Procedures and Programs*). Grassland preserves should consist of large fields in close proximity to each other that are near secure nest sites, thus maximizing foraging efficiency. The barn owl typically hunts within 1.6 km of nest sites (Colvin 1984, Rosenburg 1986). Therefore, adequate areas of grassland should be present within 1.6 km of a secure nest site.

No large-scale grassland preservation programs currently exist in the Northeast other than roadside right-of-way delayed mowing agreements with state highway departments. Roadside right-of-way grassland preservation projects are not recommended for preserving dense grass habitats because 1) the barn owl requires a very large area of habitat in close proximity to secure nest sites, and 2) the barn owl is quite vulnerable to collisions with vehicles.

## MONITORING PROCEDURES AND PROGRAMS

### Populations

Few reliable data are available concerning barn owl population size and trends in the Northeast (see *Population Distribution, Status, and Trends*). It is essential that baseline population data be gathered and that populations are thereafter monitored closely so that sound decisions can be made concerning status assignments and management objectives. Blodget (1988) suggests that Northeast states institute a long-term, systematic monitoring scheme using a fixed number of "sentinel nest box" monitoring stations (Colvin et al. 1984, Henry 1987). Blodget (1988) and Hands et al. (1989) emphasized the need for consistency and coordination of such efforts among state wildlife agencies. This may best be accomplished by a regional agency such as the U.S. Fish and Wildlife Service. Two individuals in each state (Connecticut and Rhode Island, and Maine and New Hampshire could be combined) should be contracted to establish and monitor sentinel nest boxes and to monitor grassland availability and meadow vole population cycles as well. Long-term monitoring of these resources is essential before barn owl population trends in the Northeast can truly be understood.

In each state, an extensive system for identifying barn owl nest sites should be initiated or, for those states that have already initiated efforts, these efforts should be expanded. Two years of diligent efforts should be made to gather specific site data. All states should identify (using remote sensing, U.S. Geological Survey Land Use and Land Cover, or state geographic information system (GIS) data) areas in which large amounts of dense grass habitats are found. Survey efforts should then be concentrated in these areas. Effective methods of identifying barn owl sites include 1) advertising in farm magazines, state naturalist magazines, county newspapers, and agricultural and naturalist newsletters; 2) distributing "wanted posters" to feed stores, Soil Conservation Service (SCS), Agricultural Stabilization and Conservation Service (ASCS), Cooperative Extension Service (CES), and Farm

Bureau offices; 3) using radio station public service message or advertising services; 4) contacting state wildlife biologists and game wardens as well as local bird club members, livestock veterinarians, SCS, ASCS, and CES representatives; 5) reviewing *American Birds*, state ornithological journals, and bird club records (where available) for Christmas Bird Count and other reports of barn owls; 6) requesting site details from people who reported barn owls during state BBAs; 7) searching appropriate structures and, as much as possible, tree cavities in areas supporting quality habitat; and 8) listening by ear or with bioacoustics equipment to locate food-begging nestlings or fledglings (Petersen 1980, Colvin 1984, Colvin et al. 1984, Hegdal and Blaskiewicz 1984, Lerg 1984, Byrd and Rosenburg 1986, A. Parker pers. comm.).

Sites identified using the above techniques should be inspected for barn owl nesting activity. Nest boxes should be erected at the sites where there are no apparent nest sites or the nest site is insecure or short-lived (e.g., tree cavity, silage, hay bale, or small platform nests). Nest boxes should also be erected at sites located near quality foraging habitat, even if there is no sign of barn owl activity, because barn owls are readily drawn to such nest boxes (Colvin et al. 1984, C. Rosenburg unpubl. data). See *Management Procedures and Programs* for specific suggestions for erecting nest boxes. Blodget (1988) recommends establishing 200 sentinel nest boxes throughout the Northeast; 500 or more could be established and monitored efficiently if coordinated efforts are made in each state. Efforts should be made to establish a roughly uniform distribution of nest boxes within available grassland areas throughout the Northeast. The establishment of sentinel nest boxes will provide relatively secure and stable nesting sites which can be monitored over a long period of time (Colvin et al. 1984, Henry 1987, Blodget 1988).

After the 2-year site establishment phase, a long-term population monitoring program can be initiated. Long-term monitoring is essential for a species like the barn owl which is highly dependent upon a cyclic prey base such as microtine mammals (Blodget 1988, Taylor et al. 1988, B. Colvin pers. comm.). Contractors should identify the number of active sites, the number of

nests that are successful, and the number of young fledged for each state. To identify the number of young fledged, a minimum of two visits is necessary. At least one must be made during the nestling stage to count and band young and another made after young fledge to identify the number of banded young that died prior to fledging (Colvin et al. 1984).

It is important to avoid disturbing nests during incubation and when young near fledging age are present so that nest abandonment and premature fledging is avoided (Colvin et al. 1984, Grier and Fyfe 1987). Nests should not be disturbed until young are at least 2 weeks old (Colvin et al. 1984). It is best to band young between 3-6 weeks of age. The variability of egg-laying dates makes it difficult to totally avoid disturbing nests at critical times, but problems can be minimized by scheduling nest visits in June and early July. Since there is potential for raccoons to follow human scent into barn owl nest sites, naphthalene or paradichlorobenzene crystals should be scattered on the ground at entrances of vulnerable sites (e.g., silos) to discourage subsequent predation (Ray 1968).

#### Prey

Long-term meadow vole population monitoring should be conducted in each state in conjunction with barn owl population monitoring. Such surveys are needed to track the relative abundance of this cyclic prey species and to provide a means of testing for a correlation between their relative abundance and the productivity of barn owls. Such data may reveal that short-term barn owl population fluctuations are caused by prey cycles as opposed to environmental degradation.

Locations, times, and trapping methods for meadow vole surveys should be standardized. The best locations for such surveys are within areas where barn owl nests are being monitored and where there is an abundance of quality grass habitat. Surveys should be made during the peak of barn owl egg-laying (i.e., mid-April) and again near the peak of barn owl fledging (i.e., mid-July) in order to evaluate the relative abundance of the meadow vole during the period that barn owl productivity is being evaluated. The suggested

trapping method is the use of snap traps placed along 3 parallel transects consisting of 10 trap stations each. The suggested spacing between trap stations is 10 m. Five such trap grids placed in separate grass fields and trapped for 2 consecutive nights would result in 300 trap-nights per monitoring period or a total of 600 trap-nights per year for each state.

Meadow vole abundance is strongly influenced by weather, in particular moisture conditions (Colvin and Hegdal 1986b, 1987, 1988, 1989). Therefore, moisture conditions should be monitored annually to gauge effects on vole abundance. A moisture index based on evapotranspiration data (Colvin and Hegdal 1988) could be developed for use as an index of meadow vole abundance in the Northeast.

#### Habitat

A long-term grassland monitoring system should be established throughout the Northeast in order to monitor trends in the availability of this essential resource. Productive grasslands are a declining resource and the availability of quality grass habitats is the single greatest factor limiting barn owl productivity (see *Limiting Factors and Threats*). The use of remote sensing or state GIS data is the ideal means for monitoring this resource. The U.S. Department of Commerce, Census of Agriculture (e.g., U.S. Department of Commerce 1981) also provides grassland area data within its state land-use summaries which are conducted at approximately 5-year intervals. These summaries can be used when more reliable sources are not available. Conscientious monitoring programs may identify a need for grassland acquisition programs or expanded conservation programs.

#### Current Programs

**Massachusetts.** - The Massachusetts Division of Fisheries and Wildlife compiled barn owl site information from ornithological records (1960-85) and is currently compiling state records from independent cooperators monitoring nest boxes and other sites throughout the state (B. Blodget pers. comm.).

**Connecticut.** - Approximately 50 nest boxes and other sites in the state are being monitored by various individuals. Currently, there is a lack of a coordinated effort and therefore no comprehensive population information has been compiled (G. Zepko pers. comm.).

**Pennsylvania** - The Pennsylvania Game Commission conducted a barn owl survey in southeastern Pennsylvania. They also monitor their own nest boxes, compile records from individuals conducting independent nest box programs, and are proposing to mail a questionnaire to farmer cooperators to identify additional sites (J. Hassinger pers. comm.).

**New Jersey.** - The Raptor Trust has been monitoring approximately 100 nest sites in Somerset and Hunterdon counties (northcentral New Jersey) since 1970 (L. Soucy pers. comm.). The Barn owl Research Program has maintained records of 519 nestings at 143 sites in western Salem and Cumberland counties (southwestern New Jersey) since 1980 (B. Colvin pers. comm.). Barn owl migration past Cape May has also been monitored since 1969 (Duffy 1985).

**Maryland.** - Maryland Forest, Park, and Wildlife Service personnel monitored duck blind and nest box use in coastal areas from 1988-91. They are also compiling data from independent cooperators monitoring nest boxes and are soliciting site information from the farming community through SCS, ASCS, CES, and Farm Bureau personnel. The Service plans to make full use of GIS capabilities to identify major areas of grassland and to monitor trends in this resource (G. Therres and S. Smith pers. comm.).

**Virginia.** - The Virginia Department of Game and Inland Fisheries conducted a statewide survey from 1985-86 to identify historic and recent barn owl sites throughout the state. All sites were visited during the 1986 breeding season to determine their current status (Byrd and Rosenburg 1986). All Department nest box sites were monitored from 1987-89 (Cross 1988, Rosenburg 1990a) and records from additional independent nest box programs were compiled. In addition, all nests in natural sites which were

active in 1986 were visited again in 1989. A total of 147 confirmed nest sites active between 1986-90 were identified (Rosenburg 1990a).

## MANAGEMENT PROCEDURES AND PROGRAMS

### Captive Release Programs

Captive release programs have been conducted in the midwestern states where barn owl populations are very low. Extensive release programs have been completed in 4 states; a total of approximately 1,200 barn owls (including juveniles and breeding pairs) were released during the 1980s in Iowa, Kansas, Missouri, and Wisconsin (Matteson and Petersen 1988, Ehresman et al. 1989, Marti 1989). Missouri is the only state that has verified nesting by captive-released barn owls (W. Crawford pers. comm.). Nebraska and Iowa reported probable nesting by released birds (Ehresman et al. 1989).

Problems associated with captive release programs include high mortality of released birds (Jungemann 1986, Ehresman et al. 1989) and rapid dispersal from release areas (Daniels 1984, Jungemann 1986). Captive release programs are not recommended for the Northeast since recruitment does not seem to be a severe problem, as demonstrated by the presence of existing populations throughout much of the Northeast. Also, the barn owl is not limited by colonization ability, reproductive potential, or maturation rate as is the case with many other raptor species (B. Colvin pers. comm.).

### Nest Box Programs

Nest box programs can be effective for maintaining or increasing barn owl populations (Marti et al. 1979, Soucy 1980a, Ziesemer 1980, Juillard and Beuret 1983, Colvin et al. 1984, Lenton 1984, Schulz and Yasuda 1985). Nest box programs are beneficial in many areas because the availability of secure and stable nesting sites in close proximity to quality foraging habitat is often limited (see *Limiting Factors and Threats*). Barn owl nests within nest boxes have been shown to

have higher productivity than nests placed in other locations (Colvin et al. 1984, Byrd and Rosenburg 1986, Gubanyi 1989). Nest box programs can expand barn owl populations in areas that support an abundance of quality foraging habitat but have few natural or human-made nest sites (e.g., extensive cordgrass (*Spartina* spp.) marsh). Densities between 3-6 breeding barn owl pairs per 10 km<sup>2</sup> have been reported from areas with concentrated nest box programs and abundant foraging habitat in New Jersey (Hegdal and Blaskiewicz 1984), New York (Cook 1985), Utah (Marti and Marks 1987), and Virginia (K. Bass and M. Causey unpubl. data).

Nest box programs are effective only when they are concentrated in areas which support an abundance of dense grass foraging habitat (see *Land Protection and Preserve Design*). Therefore, the availability of quality foraging habitat should be the prime consideration in the selection of sites for nest boxes (Marti et al. 1979, Colvin et al. 1984, Rosenburg 1986, Marti and Marks 1987). Documented use of a particular area or structure by barn owls should not be considered mandatory when selecting sites for nest boxes; boxes near appropriate habitat have often been used even when obvious sign of previous owl use was lacking (Colvin et al. 1984, C. Rosenburg unpubl. data). Although occupancy of nest boxes by rock doves may preclude use by barn owls (Colvin et al. 1984), several nest boxes erected in Virginia structures, which once harbored many rock doves, now support successfully breeding barn owls and few or no rock doves (C. Rosenburg unpubl. data).

There are four basic types of barn owl nest boxes: 1) nest tray attached inside roofed silos and barns (Soucy 1980b, Colvin 1983, Rosenburg 1990b), 2) enclosed nest box attached inside topless silos and barns or the outside of structures (Soucy 1980b, Rosenburg 1990b), 3) flush-mount (Ohio style) nest box attached inside barns with the owl entrance hole made through the barn wall directly into the nest box (Colvin 1983, Colvin and Hegdal 1986a), and 4) enclosed nest box mounted on posts or utility poles (Soucy 1980b, Cook 1985, Bendel and Therres 1988). Nest trays mounted inside inactive roofed silos have been very successful in Virginia (Rosenburg 1990a; K. Bass and M. Causey unpubl. data). Flush-mount

nest boxes have been widely used in New Jersey, Ohio, and Indiana (Colvin et al. 1984, A. Parker pers. comm.). These boxes provide ideal predator protection and are especially valuable for sites where the landowner prefers to exclude owl and rock dove access to the inside of the barn (Colvin 1983, Colvin and Hegdal 1986a). Enclosed nest boxes mounted on posts in or adjacent to *Spartina* marsh have been very successful in Maryland (Bendel and Therres 1988) and New York (Cook 1985). Predator guards are essential for these boxes (Cook 1985, Bendel and Therres 1988) and a slanted roof discourages osprey (*Pandion haliaetus*) nest building on the roof of the box (Bendel and Therres 1988).

### Habitat Manipulation

Since the meadow vole is closely associated with dense grass cover (Eadie 1953, Getz 1961, Zimmerman 1965, Birney et al. 1976, Klatt and Getz 1987), habitat manipulation should focus on maintaining the grass sere without removing the dense ground cover essential for meadow voles. Grass fields should be lightly grazed or mowed once a year. Ideally, litter from mowing should be left in the field to maintain high meadow vole populations. However, one annual hay cutting appears to have little long-term impact on vole carrying capacity and should therefore be considered compatible with grassland management objectives. Many other grassland bird species cannot tolerate hay cutting in early summer (Ryan 1986), so this certainly is not an option in areas managed for additional species. Prescribed burning may be a potential management practice in some areas. Tidewater marshes of Maryland are burned extensively during winter months resulting in rapid vegetation growth without removing litter (S. Smith pers. comm.).

Other factors which affect meadow vole abundance and which may therefore deserve management attention are vegetational nitrogen content (Hunly and Inouye 1987) and the availability of preferred forage (Zimmerman 1965). Important forage plants for the meadow vole include *Agropyron*, *Ambrosia*, *Bromus*, *Carex*, *Fragaria*, *Medicago*, *Muhlenbergia*, *Panicum*, *Poa*, *Taraxacum*, and *Trifolium* spp. (Zimmerman 1965,

Lindroth and Batzli 1984, Marquis and Batzli 1989, Bucyanayandi and Bergeron 1990).

Little detailed information regarding specific management programs that increase meadow vole populations has been published. Therefore, there is a need for further investigation into the results of various types and degrees of grassland manipulation on meadow vole populations in the Northeast before reliable habitat management practices can be suggested confidently.

### Education

Efforts towards educating farmers and other landowners about the barn owl's unique characteristics, rodent-catching abilities, and reliance upon dense grass habitats should be made. Educational efforts will generate a stronger interest in the barn owl and may stimulate conservation efforts that will benefit the barn owl. Educational efforts are also a good public relations tool that may result in a better acceptance of state and regional efforts. They often lead to reports of barn owls and may help identify individuals who are willing to conduct local barn owl nest box programs (A. Parker pers. comm.). Potential educational efforts include natural history and nest box pamphlets, articles in conservation and agricultural magazines, and slide presentations to ornithological groups. The Indiana Department of Natural Resources produced a barn owl slide program which has been very popular and has stimulated local nest box programs (A. Parker pers. comm.).

### Current Programs

**Massachusetts.** - A nest box program is maintained at Felix Neck Wildlife Sanctuary on Martha's Vineyard by G. Ben David (Brett 1987). Individuals in western Massachusetts also maintain a limited number of nest boxes (B. Colvin pers. comm.). To date, none of the nest boxes in the western part of the state have been active (B. Blodget pers. comm.).

**Connecticut.** - A total of approximately 50 nest boxes are being maintained statewide by various individuals (G. Zepko pers. comm.).

**New York.** - Jamaica Bay National Wildlife Refuge personnel and volunteers maintain 20 nest boxes on the refuge (Cook 1985, D. Riepe pers. comm.).

**Pennsylvania.** - A total of approximately 100 nest boxes have been erected by Pennsylvania Game Commission personnel, Volunteers for Wildlife, and other interested parties throughout the state (J. Hassinger pers. comm.).

**New Jersey.** - The Raptor Trust maintains nest boxes and repairs deteriorating structures, such as wooden water tanks, to prolong the life of these important nest sites (L. Soucy pers. comm.). A total of 67 nest boxes are currently maintained by B. Colvin and P. Hegdal in western Salem and Cumberland counties (B. Colvin pers. comm.).

**Maryland.** - Maryland Forest, Park, and Wildlife Service personnel established 40 marsh and 40 offshore nest boxes in Fishing Bay and Monie Bay to test for an acceptable box design in marsh ecosystems (Bendel and Therres 1988, G. Therres pers. comm.). The Service is also working closely with cooperators conducting nest box programs statewide. Nest box efforts in coastal marshlands and agricultural areas were expanded in 1991 (G. Therres pers. comm.). A brochure which encourages the provision of nest boxes in appropriate areas has been circulated among the farming community.

**Virginia.** - The Virginia Department of Game and Inland Fisheries erected 82 nest boxes throughout much of the state and maintains the 65 that remain (Rosenburg 1990a). K. Bass and M. Causey maintain 50 nest boxes in Prince William County. D. Ober and the Montpelier Naturalists maintain 40 nest boxes in Orange County. Several other individuals maintain smaller nest box programs elsewhere in the state. A leaflet describing the construction and placement of nest boxes (Rosenburg 1990b) has been printed by the department for distribution to interested persons.

## RESEARCH NEEDS AND PROGRAMS

Although many aspects of the barn owl's life history have been thoroughly investigated, some of the factors which may potentially limit barn owl populations deserve further study. Relatively little is known about the precise habitat characteristics (e.g., quantity and quality of grass habitats) needed to support a productive barn owl nesting pair. Therefore, there is a need to identify and quantify the foraging habitat variables associated with successful barn owl nest sites in coastal and inland areas. Such information would be useful for selecting locations where nest boxes could be placed and for locating barn owls in areas where population density is low. Little detailed information regarding the effects of specific land use practices upon meadow vole populations has been published. Further investigation into this subject is necessary before reliable habitat management practices can be suggested confidently. Also, there is a need to obtain a better understanding of the degree to which barn owls are contaminated with anticoagulant rodenticides (Rosenburg 1986, Schulz 1986, Marti and Marks 1987) and organophosphate insecticides (W. Brewer pers. comm.), especially in marginal habitat areas such as intensively farmed regions (Rosenburg 1986). Comprehensive contaminant analyses may identify direct and indirect barn owl mortality factors.

The following are three suggested research objectives that warrant attention:

- Identify important habitat variables associated with successful barn owl nest sites in the Northeast. Compare habitat variables (e.g., percentage of open vs. wooded vegetation, acreage of grass habitats supporting *Microtus* spp., and kilometers of edge) within 1.6 km of successful nest sites versus random points. Separate comparisons should be made for agricultural and coastal marsh sites. Use canonical discriminate analyses to identify the relative importance of each variable for supporting successful barn owl nests. Identify the minimum area of grassland with *Microtus* spp. within 1.6 km necessary to support successful breeding pairs of barn owls in the Northeast.

- Determine the results of habitat manipulation upon meadow vole populations in order to evaluate the effects of existing land use practices and potential management practices upon this important prey species. Use standard live or snap trapping methods to compare meadow vole population indices between grass fields exposed to varying degrees of mowing, burning, grazing, and haying.
- Identify the levels of anticoagulant rodenticide and organophosphate insecticide residues present in free-ranging barn owls. An extensive effort to collect barn owl carcasses for pesticide analyses, such as the effort made in Great Britain (Newton and Wyllie in press, Newton et al. in press), should be made in the Northeast. Liver, blood, and brain tissue samples should be collected from barn owls euthanized in wildlife rehabilitation centers or those found shortly after death (S. Porter pers. comm.). Liver samples can also be taken from owls which were not found shortly after death. Liver tissue samples should be analyzed for residues of difencoum and brodifacoum (Newton et al. in press). Blood and brain tissue samples should be analyzed for significant cholinesterase inhibition (Hill and Fleming 1982, Hill 1988, L. Brewer pers. comm.). Fresh pellet and fecal samples should be collected at various intensively farmed areas of the Northeast. Analyses of these pellet and fecal samples can identify organophosphate insecticide groups that may be present (L. Brewer pers. comm.).

### Current Programs

**New Jersey.** - The Barn owl Research Program has investigated secondary poisoning hazards of rodenticides, foraging behavior, habitat use, nest site use, nesting behavior, vocalizations, movements, population dynamics, sexual dimorphism, nest and mate fidelity, diseases and parasites, and capture and management techniques within their southwestern New Jersey study area (Brett 1987). B. Colvin (pers. comm.) reported that their current efforts are focusing on

the population dynamics of this dense and well-marked population that has over 1,400 nestlings and 200 adults banded since 1980.

**Maryland.** - Maryland Forest, Park and Wildlife Service personnel have been using radio-telemetry to evaluate fledging success of young from offshore duck blinds, offshore nest boxes, and marsh nest boxes. The Service has also contracted University of Maryland researchers to characterize nesting habitat in agricultural areas based on known barn owl nest sites, both current and historic (G. Therres pers. comm). Comparisons between successful and unsuccessful nest sites are also being made.

**Virginia.** - S. Erdle and J. Pagels of Virginia Commonwealth University and C. Rosenberg are analyzing approximately 5,000 barn owl pellets collected at approximately 75 sites throughout Virginia during the summers of 1986 and 1989, and the winter of 1989-90.

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# LOGGERHEAD SHRIKE

## *Lanius ludovicianus*

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The loggerhead shrike, a bird of short grassland habitats, has undergone a dramatic 87% decline in the Northeast in the past 25 years. Shrikes are small avian predators that hunt from perches and impale their prey on sharp objects such as thorns and barbed-wire fences in pastures. The species once bred throughout much of the Northeast, but now remains as a breeder only in a small population in western New York, and a larger, though rapidly declining, resident population in Virginia, Maryland, and West Virginia. Wintering populations from the north that once used coastal areas are no longer found. Remaining breeders experience relatively high reproductive success, and pesticide contamination does not appear to be a problem. Habitat loss from farmland abandonment, development, and changes in agricultural practices may be causing widespread declines, although seemingly suitable unoccupied habitat remains. Mortality from collisions with vehicles is also implicated. Many questions remain as to the cause of this sweeping population decline, though the maintenance of active pastures is believed essential to the continued survival of the species.

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### DESCRIPTION

#### Taxonomy

Loggerhead shrikes (*Lanius ludovicianus*) belong to the shrike family, Laniidae, in the order of perching birds, Passeriformes. The two subspecies in the northeastern states (*L. l. ludovicianus* and *L. l. migrans*) are distinguished by relatively subtle characters as described in Miller (1931) and Palmer (1898). For adults, the respective differences between *L. l. ludovicianus* and *L. l. migrans* include: anterior forehead neutral gray with restricted light gray area versus generally pale gray to white near borders, and

hind neck and back neutral to deep gray versus neutral to light gray. Also, *L. l. ludovicianus* has slightly shorter wings, a longer tail, and larger bill than *L. l. migrans* (Miller 1931). According to Miller (1931), the wings are shorter than the tail in *L. l. ludovicianus*, but are as long or longer than the tail in *L. l. migrans*.

#### Morphology and Plumages

Slightly smaller than an American robin (*Turdus migratorius*), the loggerhead shrike is a predaceous perching bird of open country. It frequently impales its prey on thorns and other sharp objects and flies with very rapid wingbeats. Adult males and females are similar in

appearance, with gray backs, light breasts, black wings and tail, light wing patches and tail bars, a black facial mask, large head, and dark, heavy, hooked bill (Fraser and Luukkonen 1986). Breeding adults have pale gray unbarred underparts with white wing patches; the juvenal plumage has darker, lightly barred underparts, buffy wing patches, and a somewhat buff to reddish-brown cast to the upperparts. First-year birds resemble the adults, but may have some buff coloration to the wing tips (Miller 1931, Kridelbaugh 1982). If necessary, sex and age can be segregated somewhat accurately by morphometric analysis (Blumton 1989). Loggerhead shrikes have a complete annual molt in the fall and a partial prenuptial molt in late winter and early spring (see summary in Kridelbaugh 1982).

## HABITAT

### Breeding Habitat

Year-round, shrikes generally concentrate their activity in grassland habitats. They typically nest in areas with an abundance of short grass, with active pastures being the most heavily used (Luukkonen 1987, Novak 1989, Gawlik and Bildstein 1990). In Missouri, pastureland surrounded 67% of 60 nests (Kridelbaugh 1982). In New York, occupied nest sites were in pasture areas with less than 20% woody cover (Novak 1989). Idle pastures and lawns are also selected for nesting sites (Luukkonen 1987, Gawlik and Bildstein 1990) and historically, orchards were seemingly used with some frequency (Novak 1989). Brooks (1988) found that nestling growth rate, nesting success, and fledgling success in shrikes were positively correlated with the percentage of home range coverage in grassland. In Virginia, shrike pairs nesting in active pastures produced twice as many young as those in other habitats (Luukkonen 1987). Luukkonen also reported that active shrike nests were often near water, but pointed out that active pasture usually had water for livestock so that the relationship between shrikes and a water source is unclear. Blumton (1989) determined that reproductive success in Virginia was positively correlated with

bare soil areas, which may provide increased prey visibility and accessibility.

For nesting, shrikes appear to select those trees which provide protection and concealment from predators (Luukkonen 1987, Gawlik and Bildstein 1990). Based on availability, shrikes select red cedars (*Juniperus virginiana*) and hawthorns (*Crataegus* spp.) much more frequently than expected, with hawthorns used more frequently farther to the northeast (Luukkonen 1987, Brooks 1988, Novak 1989, Gawlik and Bildstein 1990). Other woody plants are selected on occasion, especially osage orange (*Macharia pomifera*) and multiflora rose (*Rosa multiflora*) (Kridelbaugh 1982, Luukkonen 1987, Bartgis 1989, Hershberger 1989). Regardless of the species selected, trees with a thick covering of vines such as Japanese honeysuckle (*Lonicera japonica*) or grapes (*Vitis* spp.) are often preferred (Luukkonen 1987, Novak 1989).

However, nesting success in Virginia was not correlated with either the tree species selected or concealment (Luukkonen 1987). Nests in multiflora rose are less successful than those in trees, perhaps because the nests are lower and poorly supported (Kridelbaugh 1982). Birds in more northern areas may breed in spruce (*Picea* spp.) and fir (*Abies* spp.) trees (Bent 1950, Brooks 1988). Shrikes most frequently nest in isolated trees, but it is not unusual for them to nest in hedgerows (Luukkonen 1987, Brooks 1988). Shrikes also tend to nest in areas with several potential suitable nesting trees. Brooks (1988) reported that at least seven potential nesting sites occurred within 0.25 mi (0.40 km) of all occupied nest sites in Minnesota. Shrikes also appear to select trees without low branches, since they frequently swoop up to the nest from below when there is a cleared approach (R. S. Dean pers. comm.).

Pastures used by shrikes for hunting typically have many potential perches (e.g., utility wires, fencerows, mullen (*Verbascum* spp.), rock outcrops, etc.) and areas for impaling food items. Barbed-wire fence is frequently used for impaling prey items, although any thorny shrub or tree may be used (Fraser and Luukkonen 1986). Taller perches tend to be used later in the summer as vegetation grows taller (Novak 1989).

## Wintering Habitat

In the Northeast, winter foraging habitat does not seem to differ strikingly from summer habitat, although hayfields and idle pastures are used heavily. Winter-dieback in vegetation may make these areas easier to hunt. Blumton (1989) reported that during the winter many shrikes in Virginia move from pastures to shrub and open forest habitats during periods of cold, wet weather.

## BIOLOGY

### Reproduction

**Courtship and Nesting.** - Male shrikes select territories in late winter and very early spring and announce their presence by flashing their white wing and tail markings and by singing from a conspicuous perch (Miller 1931). Intruding males are chased, and if not repulsed, a "flutter" display follows (Kridelbaugh 1982). The male performs an erratic, zigzagging nuptial flight and occasionally chases the female during courtship (Miller 1931, Bent 1950). During the courtship period, the male feeds the female as she crouches and flutters her wings during a begging display (Kridelbaugh 1982).

In Maryland and the Virginias, where some shrikes are resident year-round, it is not unusual for shrikes to nest at a site used the previous year. Occasionally they will nest again in the same tree (Luukkonen 1987, Bartgis 1989). Shrikes begin to lay eggs in Maryland and the Virginias in early to mid-April (Luukkonen 1987, L. MacIvor and R. Dean pers. comm.), but do not return to nesting sites in New York until mid to late April (Novak 1989). In New York, Novak (1989) reported that nests are typically placed 1.5-2.5 m high in 4- to 5-m-tall trees, and are typically set more than a meter in from the outside of the tree. In Virginia, Luukkonen reported that first nests were placed, on average, 2.6 m high in trees averaging 6.8 m in height. Both sexes participate in selecting the nest site and in nest construction (Kridelbaugh 1982). The nests are usually placed in a crotch, or sometimes on top of an older nest;

most are made from course twigs and lined with plant material and animal hair (Fraser and Luukkonen 1986). A clutch of 4-6 eggs is typical for Maryland, New York, Virginia, and West Virginia (Luukkonen 1987, Novak 1989, R. Dean and L. MacIvor pers. comm.). One egg a day is laid, with incubation probably starting with the next to last egg. Egg laying may be delayed in wet years (Blumton 1989). The male feeds the female during incubation (Miller 1931). In Virginia, eggs hatch asynchronously, with incubation lasting 16-18 days and nestlings fledging 17-20 days later (Luukkonen 1987).

Rangewide, there is geographic variation in the number of broods raised (Kridelbaugh 1982). If the first nesting attempt fails, second attempts are frequent in the Virginias and Maryland (R. Bartgis pers. obs., Luukkonen 1987, Davidson 1988), but second attempts may be less common in New York (Novak 1989). In Virginia, many pairs with successful first nests initiated second nests, with nesting activity peaking in late May and early June (Luukkonen 1987). In Maryland, second attempts have been documented in June and early July (L. MacIvor pers. comm.). Clutch size is somewhat smaller in subsequent nestings. If a second nest fails, a third nesting attempt often follows, but most are unsuccessful (Luukkonen 1987). Second and third nests are placed spatially close to the first (mean = 90 m) whether the first was successful or not, and later nests are typically higher in a tree than are first nests (mean = 5.5 m above ground in Virginia) (Luukkonen 1987). At a Maryland site, a second brood was raised in the same nest as the first successful brood (L. MacIvor pers. comm.).

**Fledgling.** - During the first few days out of the nest, young shrikes stay concealed in dense tree growth, but begin to try to follow foraging adults within a week. About 2 weeks after leaving the nest, fledglings begin to capture food for themselves; they continue to be fed by adults for an additional 2 weeks (Luukkonen 1987, Novak 1989). By this time, adults and young begin foraging in pastures away from the nesting territory (Novak 1989). In New York, family groups begin to break up and disperse in August (Novak 1989). In Virginia, juvenile shrikes moved an average of 5.5 km from the parents' territories

to fall territories at about 10-13 weeks of age (Blumton 1989). Fall home ranges are established from late July to mid-September in Virginia (Blumton 1989). Observed fledgling disappearance in summer may represent movement and not mortality.

Some females have been observed to desert their mates during the fledging period and raise a second brood in another location with another male (Novak 1989). In Missouri and New York, females disappeared during the fledging period, leaving the male to feed a brood by itself (Kridelbaugh 1982, Novak 1989). The male may also draw the young shrikes in close to a prey item, leaving it for them to capture. Kridelbaugh (1982) believed this to be training behavior.

**Nesting Success.** - Nesting success for shrikes (defined as the percentage of nesting attempts fledging at least one young) is fairly high, reaching 87% in Maryland (1991), 74% in Minnesota, 50% in New York, 55-62% in Virginia and generally greater than 60% rangewide (Luukkonen 1987, Brooks 1988, Blumton 1989, Novak 1989, L. MacIvor pers. comm.). Rangewide, shrikes typically fledge 3-4 young per successful nest (Brooks 1988). In New York, Novak (1989) reported an average of 3.5 young fledged per successful nest. In Virginia, Luukkonen (1987) reported an average of 4.0 young fledged per pair in 1985 and 1986, while Blumton (1989) reported an average of 2.6 young fledged per pair in 1987. In 1991, 3.3 young fledged per pair in Maryland (L. MacIvor pers. comm.). Nest failures have been attributed to cold, wet weather and predation in New York (Novak 1989) and to predation, abandonment, and inadequate support of nests in Virginia (Luukkonen 1987). Nesting success is significantly better in dry, warm years than in cool, wet years (Kridelbaugh 1982). Suspected nest and fledgling predators in the Northeast include black rat snakes (*Elaphe obsoleta*), blue jays (*Cyanocitta cristata*), sharp-shinned hawks (*Accipiter striatus*), house cats, and even house wrens (*Troglodytes aedon*) (Luukkonen 1987, Novak 1989). Nest predation by brown-headed cowbirds (*Molothrus ater*) is rare (DeGeus and Best 1991). While predation is the leading cause of nest failure, the overall nesting success

rate is high and predation is not described as limiting the population.

**Site Fidelity.** - Early observers felt that shrikes were very faithful to their nesting territory from previous breeding seasons, occupying territories for up to 6 years (Bent 1950, see also Kridelbaugh 1982). However, these observations were based on unmarked individuals. Males are most likely to reoccupy previous breeding territories (Kridelbaugh 1982, Luukkonen 1987, Brooks 1988). Currently, there is a low return rate from year to year for adult shrikes (particularly females) and a very low return rate for shrikes banded as nestlings. In Virginia, only 2% of color-banded nestlings were located in subsequent years (Blumton 1989). In Minnesota and Virginia, respectively, 50% and 30% of occupied breeding territories were not occupied the following year (Luukkonen 1987, Brooks 1988). In Virginia, areas of active pasture were more likely to be reoccupied in succeeding years than areas where pasture was allowed to grow tall (Luukkonen 1987). The number of abandoned nesting territories is thought to reflect a high nonbreeding season mortality, as well as selection for new territories (Luukkonen 1987, Brooks 1988, see also Novak 1989).

#### Wintering Biology

In winter, shrikes are solitary (Kridelbaugh 1982). Individuals that breed in southern Canada, New York and New England typically winter south of their breeding range. Shrikes that breed in the Virginias and Maryland are frequently resident, but there appears to be some southward movement. In Virginia, winter home ranges averaged about 52 ha, with open grassy habitats most heavily used. In inclement weather, shrikes utilize shrub-forest habitats, which provide cover and food (Blumton 1989). In Virginia, the heaviest mortality occurred in January, the month with the coldest temperatures, and hawks and owls accounted for 57% of the observed fall and winter mortality (Blumton 1989).

## Food Habits

Most hunting during the breeding season occurs in active pastureland (Novak 1989). Pastures provide a more stable, reliable food supply in the summer than row crops (Kridelbaugh 1982). Perhaps more importantly, sparse, short cover also improves foraging success by shrikes. Shrikes hunt almost exclusively on perches from which they swoop down upon their prey, but they sometimes hover like an American kestrel (*Falco sparverius*) or walk (Bent 1950, Luukkonen 1987). When actively hunting (usually in early morning and late evening), shrikes perch close to the ground; when foraging passively they perch higher above the ground (Miller 1931). Shrikes kill vertebrates by repeatedly pecking and biting the back of the prey's neck until the spinal cord is severed (Miller 1931). In Minnesota, loggerhead shrikes feed up to 0.25 mi (0.40 km) away from nest sites during the nesting season (Brooks 1988). In New York, Novak found that successful nesting pairs foraged over an area of 5.7-9.3 ha.

Shrikes eat a variety of prey items, including small mammals and birds, lizards, snakes, frogs, and insects, especially grasshoppers, crickets and beetles (Miller 1931). Vertebrates usually predominate as prey items in the winter; invertebrates predominate in the summer (Miller 1931). Shrikes have been observed eating carrion and scavenging meat scraps left by hawks (Fraser and Luukkonen 1986). In Virginia, cattle dung provided habitat for scarab beetles, which were frequently impaled (Luukkonen 1987). In New York, Novak (1989) reported that scarab, dung, and ground beetles, bumblebees and small mammal (*Microtus*, *Sorex*, and *Peromyscus* spp.) remains were most frequently recorded in nesting season pellets, while grasshoppers, caterpillars, beetles, and ants were prey items most frequently observed being caught or fed to young. Novak also reported song sparrows (*Melospiza melodia*), American goldfinches (*Carduelis tristis*), meadow voles (*Microtus pennsylvanicus*), a frog, an upland sandpiper (*Bartramia longicauda*), and earthworms impaled in his study. Other food sources are exploited as they become available, including earthworms on roadways, dung beetles on spread

manure, and insects in recently mowed hayfields (Hershberger 1989, Novak 1989).

It has been suggested that shrikes impale their prey as an aid in handling food and as caches for later feeding (Kridelbaugh 1982, Fraser and Luukkonen 1986). Because shrikes lack the strong feet that allow other predaceous birds to hold their prey while tearing off small pieces, shrikes may impale food on a solid object to hold it. Watson (1910) reported loggerhead shrikes feeding in winter on dried lizards impaled the previous summer. An incubating female has been observed visiting a fencerow and feeding on insects impaled by the male; both adults also fed nestlings items that were previously impaled (Applegate 1977). It has also been suggested that male loggerhead shrikes cache food as visual displays for both attracting mates and indicating territories to other males (Sloane 1991).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

Loggerhead shrikes historically bred from southern British Columbia east to southern Quebec, south to Baja California, northern Mexico, the Gulf Coast and central Florida (American Ornithologists' Union 1957). Two recognized subspecies breed in the northeastern states. The migrant loggerhead shrike (*L. ludovicianus migrans*, as defined by Miller 1931) historically bred from Quebec and Minnesota south to Virginia and Arkansas. The nominate race breeds from Virginia to Louisiana and Florida. Miller (1931) reported a relatively broad zone of intergradation. However, recent morphometric data from Maryland and the Virginias indicate those birds are *L. L. ludovicianus* (Blumton 1989, R. Dean and L. MacIvor pers. comm.).

The loggerhead shrike has declined significantly rangewide according to the U.S. Fish and Wildlife Service's (USFWS) Breeding Bird Survey (BBS) (Robbins et al. 1986). For the period 1966-89, its population has declined 55% nationally, 47% in the central states, and 59% in

the western states (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). Christmas Bird Count (CBC) data also show a major nationwide decline for the period 1963-87 (Lowe and Butcher 1990). The loggerhead shrike is listed as a *migratory nongame bird of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987), and is ranked G4 globally (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

One subspecies, the San Clemente loggerhead shrike (*L. ludovicianus mearnsi*) of California, is listed by the USFWS as an *endangered species* (U.S. Fish and Wildlife Service 1980). The species as a whole is proposed as a *candidate (Category 2)* under review for possible federal listing (U.S. Fish and Wildlife Service 1991). Outside of the Northeast, the shrike is listed as *endangered* in Indiana, Michigan, Ohio, and Wisconsin, as *threatened* in Illinois and Minnesota, and as *special concern* in North Carolina.

### Distribution and Status in the Northeast Region

Many have suggested that shrikes entered the Northeast from the Midwest with the clearing of forestland for agriculture (Palmer 1898, Bent 1950, Novak 1989). This hypothesis does not fully agree with historical and current distributional data from Milburn (1981), Robbins et al. (1986), and the breeding bird atlas (BBA) projects, which indicate that two historically distinct breeding populations occurred in the Northeast (Figure 1). One was part of a breeding population in the Midwest that extended into extreme western Pennsylvania, eastward through New York and southern Ontario, and farther east through New England. A second breeding population extended from the Southern Appalachians, Piedmont and Coastal Plain northward to extreme south-central Pennsylvania and Cape May, New Jersey. It seems unlikely that the latter population originated in the Midwest.

Milburn (1981) has summarized literature accounts of the species' former abundance in the Northeast. Reports of shrikes in the Northeast

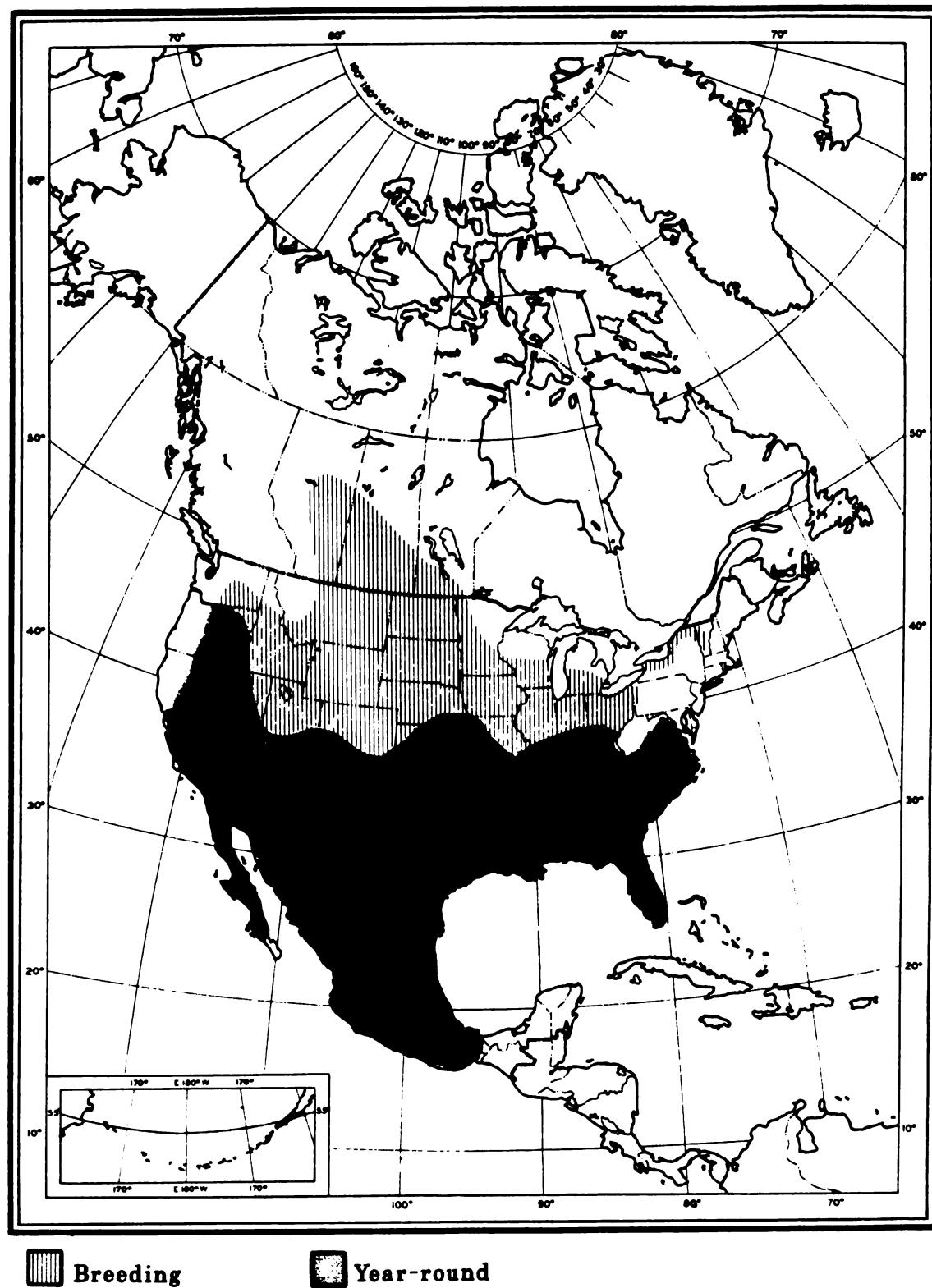
were few until birdwatching became popular among amateur naturalists in the late nineteenth century. By the 1880s and 1890s, shrikes were reported as common to abundant in a number of northeastern states (Milburn 1981). In the early part of the twentieth century the species had been described as "common" or "abundant" in Maine, Maryland, New York, Pennsylvania, and both Virginias (Milburn 1981). Most states also experienced a peak in breeding season reports in the 1950s or 1960s. There is no evidence to suggest that this represents anything other than the increase in birdwatching and birdwatcher mobility that followed World War II (Milburn 1981).

A decline in loggerhead shrikes in the Northeast was first noticed in the mid-1960s with data from the BBS, although it is generally thought to have begun well before the beginning of the survey in 1966. While the survey does not, therefore, accurately indicate the full degree of the decline, it does show that the decline is continuing. From 1966-90, the loggerhead shrike declined an average of 8.7% annually in the Northeast ( $P < 0.01$ ,  $n = 38$  routes) (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). For most northeastern states, the population of breeding shrikes is now too small to make trend analyses for individual states.

Because sightings of migrating shrikes are sporadic, trends are not easily determined (Milburn 1981). Shrikes which formerly bred in New England apparently followed a predominantly coastal migration route (Milburn 1981). In Maryland and the Virginias, where shrikes are resident year-round, it has not been possible to distinguish local birds from migrants. However, very few breeding shrikes remain north of this area, as indicated by BBA data (Figure 2).

A few loggerhead shrikes have been reported in winter from Pennsylvania, New York, and New Jersey. No trends can be determined from CBCs because so few have been observed (Milburn 1981). In the mid-Atlantic states, CBCs do, however, indicate a decline. The loggerhead shrike is currently listed as *endangered*, *threatened* or of *special concern* in many eastern states (Table 1).

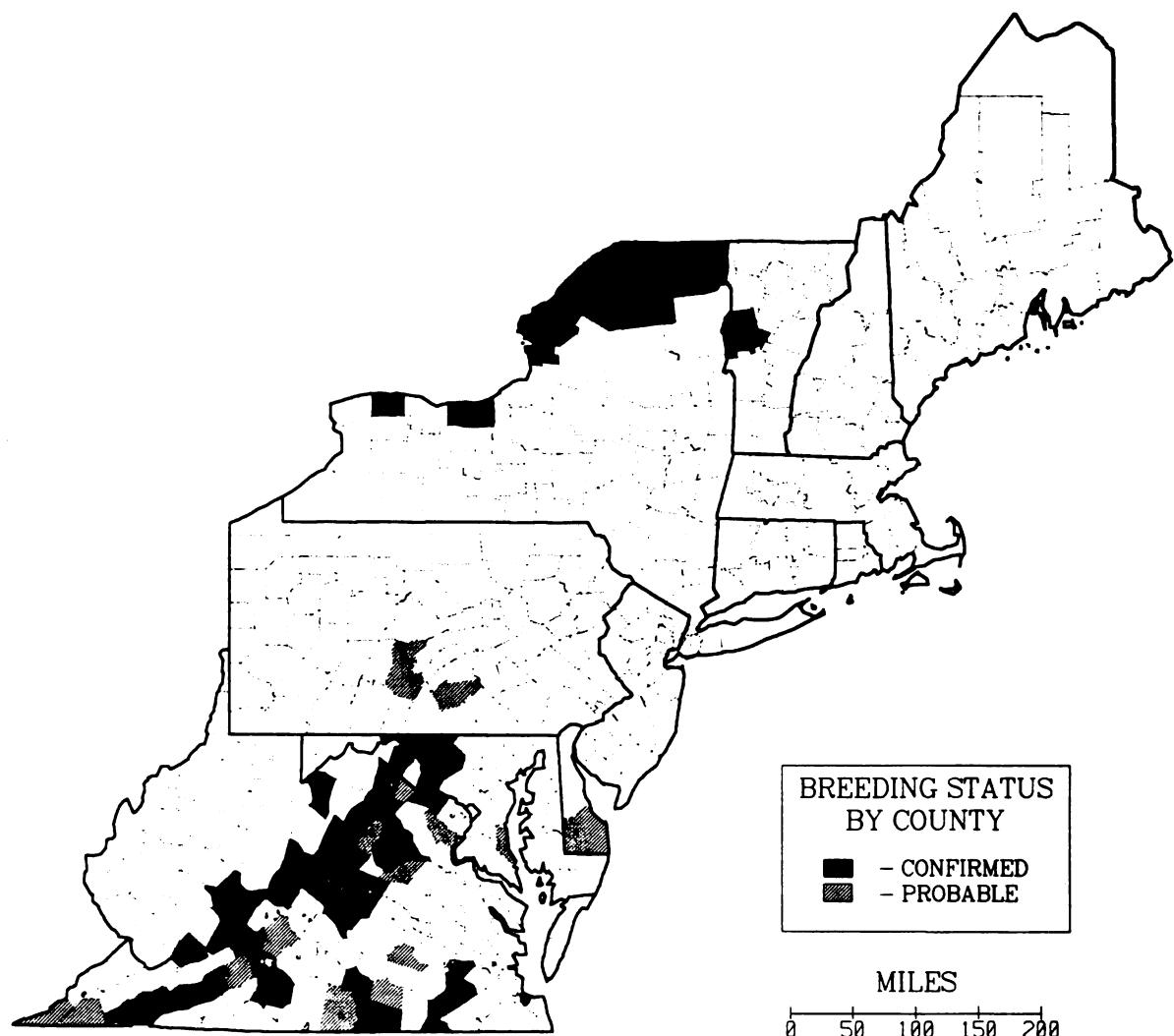
Figure 1. The North American range of the loggerhead shrike (*Lanius ludovicianus*) (American Ornithologists' Union 1983, National Geographic Society 1987).



■ Breeding

■ Year-round

**Figure 2.** Northeast breeding distribution of the loggerhead shrike (*Lanius ludovicianus*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the loggerhead shrike (*Lanius ludovicianus*) in the Northeast (The Nature Conservancy: Central Scientific Database, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	extirpated	SX
New Hampshire	endangered	SX
Vermont	endangered	S1
Massachusetts	endangered	SX
Rhode Island	-	SA
Connecticut	-	SX
New York	endangered	S1
Pennsylvania	extirpated	SH
New Jersey	endangered	SN
Delaware	-	SX
Maryland	endangered	S1
Virginia	endangered	S1
West Virginia	-	S1

<sup>a</sup>SX = apparently extirpated

SH = historical records only, but suspected extant

SA = accidental

SN = regularly occurring but not breeding

S1 = critically imperiled; ≤ 5 occurrences

**Maine.** - In Maine, Milburn (1981) reported verified and potential nesting records from thirteen counties, with the last active shrike nest reported in 1963.

**New Hampshire.** - In New Hampshire, breeding has been reported historically from five counties, mostly in the northern part of the state (Milburn 1981). No nests have been reported in the state since 1910 (E. Hentcy pers. comm.).

**Vermont.** - In Vermont, Milburn (1981) reported confirmed and potential breeding records from nine counties, mostly in the western Champlain Lowlands. During the Vermont BBA, breeding was confirmed only twice, with three other possible breeding records (Kibbe 1985). All atlas reports came from the Champlain Lowlands (Figure 2). No shrikes were found during intensive searches in the 1980s and the species is

no longer thought to breed in the state (Kibbe 1985, Fichtel 1988).

**Massachusetts.** - In Massachusetts, Milburn (1981) reported verified and potential breeding records from ten widely scattered counties. No nests have been found in Massachusetts since 1971 (Milburn 1981).

**Rhode Island.** - There are no reports of nesting shrikes from Rhode Island. Two reports of potential breeders were most likely late-season migrants (Milburn 1981).

**Connecticut.** - The only verified breeding record from Connecticut occurred in 1893. Seven other breeding season reports of shrikes are known from Connecticut, with five of these before 1910 (Milburn 1981).

**New York.** - Confirmed and potential breeding records have been reported from over 40 widely-distributed New York counties (Novak 1989). By the 1890s the species was frequently found in western and central counties. The decline in shrike numbers had become noticeable by 1930 and has since continued steadily. Ten confirmed breeding records were reported during the New York BBA, 1980-85 (Figure 2). Only six nesting pairs were found in New York during 1986-88, all from far northern counties (Novak 1989). Seven additional breeding season records for the period came from western and northern counties (Novak 1989). No nests have been confirmed in New York since 1988 (P. Novak pers. comm.).

**Pennsylvania.** - In Pennsylvania, confirmed and potential breeding records exist from 20 counties, mostly in the extreme western part of the state. However, there have been no confirmed breeding records since 1950 (Milburn 1981). During the Pennsylvania BBA (1984-87) no confirmed breeding records were reported and probable records came from only two counties (Figure 2).

**New Jersey.** - Two shrike nests were confirmed in the Cape May area in the 1890s, but no nesting activity has been reported since from New Jersey (Milburn 1981). After 1900, only six

birds which Milburn considered to be migrants were reported during the breeding season.

**Delaware.** - Only two shrike nests have been reported in Delaware, both in 1862. Milburn (1981) reported only two other breeding season records. Christmas Bird Counts suggest that loggerhead shrikes regularly wintered in Delaware through the early 1970s. From 1966-79, 89 loggerhead shrikes were reported from CBCs. From 1980-89, only seven shrikes were reported in Delaware and none were reported after 1985 (R. West pers. comm.).

**Maryland and the District of Columbia.** - In the Northeast, most historical and nearly all current breeding season reports of shrikes come from Maryland and the Virginias. Milburn (1981) reported confirmed and potential breeding records from 13 Maryland counties and the District of Columbia. The shrike population in Maryland had "decreased appreciably" by the 1950s (Stewart and Robbins 1958) and the shrike is no longer known to breed in the District of Columbia. During the Maryland and District of Columbia BBA (1983-87), only 3 confirmed and 10 possible or probable nesting records were reported. The confirmed records were from the Great Valley in Washington County and the Frederick Valley in Frederick County. Other records are from scattered Piedmont and Coastal Plain localities. In 1991, Maryland had six confirmed and two probable breeding pairs, all in the Great and Frederick valleys (L. MacIvor pers. comm.).

In Maryland, there was also a marked decline since the early 1970s in the number of shrikes reported during Christmas Bird Counts. By 1979 only 41% of Maryland's traditional shrike routes were still reporting shrikes (Milburn 1981). Declines have been most severe in coastal areas. Currently, most winter reports in Maryland of loggerhead shrikes come from the Great and Frederick Valleys and nearby areas of the Piedmont.

**Virginia.** - Although the Breeding Bird Survey indicates a tremendous decline in the breeding shrike population in Virginia, this state remains the stronghold for the species in the Northeast. In Virginia, where there are enough

BBS routes (26) reporting shrikes to enable statistical analyses at the state level, the decline between 1968 and 1989 has been 88%. About 100 confirmed and possible or potential breeding records were reported from about 40 counties and independent cities during the first 4 years (1984-88) of the Virginia BBA (Figure 2). Most records (especially confirmed ones) came from the Great Valley, over two-thirds of the reports were from Appalachian counties, relatively few records were from the Piedmont, and only three were from the Coastal Plain (Virginia Society of Ornithology 1989). The almost complete loss of the formerly significant breeding shrike population in eastern Virginia may reflect widespread habitat loss to rowcrops (S. Ridd pers. comm.). Luukkonen (1987) reported that population models he developed indicate that the decline should continue in Virginia at a high rate comparable to that indicated by the BBS.

Virginia has the largest wintering shrike population, which has also declined since the early 1970s (Milburn 1981). Luukkonen (1987) reported winter records from 55 counties and independent cities. There were no recent winter records from 56% of these counties and cities.

**West Virginia.** - West Virginia also still has a small number of breeding shrikes. Milburn (1981) reports confirmed or potential breeding records from nine counties in the state, all along or near the eastern border. Fourteen confirmed breeding pairs were found in or near the Great Valley in Berkeley and Jefferson counties in 1991 (R. Dean pers. comm.). During the 1980s, most other breeding season reports for shrikes came from the South Branch Valley (Grant and Pendleton counties) and southeastern counties (Greenbrier, Mercer, and Monroe counties; Figure 2). A juvenile shrike was observed in Canaan Valley at 914 m in elevation on the Allegheny Plateau, 1 August 1987 (R. Bartgis, pers. obs.).

Christmas Bird Counts are relatively few and sporadic in West Virginia, especially within the range of shrikes. Although there have been occasional winter reports of shrikes during the 1980s from the extreme southwestern counties, only routes from the extreme eastern counties have reported shrikes during the CBC. Presently, only those routes in the Great Valley regularly

report shrikes and these have averaged less than one shrike per count area since 1987.

### Summary

Loggerhead shrikes formerly bred in Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Pennsylvania, New Jersey, Delaware, and the District of Columbia, but are no longer known to breed in these states. In the Northeast, confirmed breeding populations are restricted to two areas. A tiny and tenuous population of perhaps less than 10 breeding pairs may remain in extreme northern New York, where it is part of a population along the St. Lawrence Lowlands that also extends into southern Ontario. A larger population centered in and around the Great Valley occurs from western Maryland, through extreme eastern West Virginia, and across western Virginia. This population has declined dramatically but still may support several dozen pairs.

The wintering population along the coast has almost completely disappeared, apparently because of the decline in the migratory breeding population of the northern states. The wintering population in the Appalachians and adjacent Piedmont of the mid-Atlantic states persists, although in considerably smaller numbers than 20 years ago.

### LIMITING FACTORS AND THREATS

Because shrikes are high on the food chain, pesticides have been implicated as a potential cause of the population decline (Fraser and Luukkonen 1986). DDE, a metabolite of DDT, has been detected in shrike eggs from Illinois and Virginia (Anderson and Duzan 1978), but crushed eggs resulting from eggshell thinning have not been reported. While there is evidence of some eggshell thinning in Illinois, there is no apparent eggshell thinning in California and Florida (see Hands et al. 1989). Researchers agree that the relatively high nesting and fledgling success rate of shrikes in the areas of marked decline indicate that pesticides have not reduced reproduction (Kridelbaugh 1982, Luukkonen 1987, Brooks

1988, Novak 1989, Gawlik and Bildstein 1990). Indeed, in Virginia Luukkonen (1987) reported a 91% hatching rate for eggs. Young shrikes exposed to dieldrin have been shown to attack and kill prey more slowly than unexposed birds (Busbee 1977). Widescale use of organochlorides was curtailed in the 1970s, yet the decline in the loggerhead shrike population continues. Blumton (1989) reported that necropsies on shrikes from Virginia found no abnormalities that indicated PCB or pesticide contamination. Tissue samples from the necropsied shrikes and from 30 eggs had pesticides present. All had DDE contamination, with one clutch of eggs having substantially high levels. Blumton (1989) drew no conclusions on the relationship between contamination and mortality.

Although the decline of loggerhead shrikes has seemingly been less severe in the southern U.S., climate is not an apparent cause. Loggerhead shrikes continue to breed in northern New York, Alberta, Manitoba, Ontario, and Saskatchewan (Robbins et al. 1986). In West Virginia, immature shrikes have been observed several times in areas above 900 m in elevation.

Shrikes are not particularly alarmed by proximity to human activity. Brooks (1988) noted that nests near buildings had a similar success rate to those farther away from buildings. In Virginia, a shrike continued to incubate a nest in a tree after the top was trimmed off (Luukkonen 1987), although in Maryland, a nest in a tree was abandoned after a multiflora rose concealing it was killed with herbicide (R. Dean pers. comm.). Some shrikes have nested less than 3 m from a road, but were not flushed by passing vehicles (Luukkonen 1987, Bartgis 1989).

In Virginia, Luukkonen (1987) reported several nests with malnourished nestlings and developmental variability within a brood, both evidence of limited food availability at some sites. However, on a broader scale, early breeding, brood reduction, and multiple broods should offset any local problems from food shortages, at least for the nesting season (Luukkonen 1987).

Limited evidence in most of the Northeast suggests that lack of suitable breeding habitat limits the abundance of loggerhead shrikes. Habitat loss has been caused by farmland abandonment, development, and widespread

changes in farming practices (Novak 1989). Although the acreage in pasture has decreased dramatically since World War II in New York and Virginia, there apparently is a substantial amount of unoccupied habitat remaining in both states (Luukkonen 1987, Novak 1989). Considerable unoccupied breeding habitat also occurs in Maryland and West Virginia (R. Bartgis pers. obs.). In Missouri, declining shrike populations coincided with regions with the highest proportion of lands being converted from pasture to rowcrops (Kridelbaugh 1982). At least on a local level in Virginia, habitat loss may be the primary problem (S. Ridd pers. comm.). Luukkonen (1987) expressed concerns about the fragmentation of potential shrike habitat into islands in the Ridge and Valley of Virginia. Occasional use of marginal habitat by shrikes may be partially to blame for the decline in Virginia. However, it seems unlikely that habitat loss on the whole can fully explain the downward population trend in shrikes.

Because of relatively high reproductive success for shrikes in southern Minnesota, Brooks (1988) concluded that the 20% mean annual rate of decline in the shrike population in the region was "...probably due to factors on their nonbreeding range" to the south. Brooks (1988) further summarizes concerns that the decline in Minnesota is possibly caused by decreases in winter habitat in the birds' wintering range. She believes that "...if resident (southern) shrike populations are being limited by habitat availability, migrant shrikes wintering in the same area are almost certainly being forced to occupy marginal habitats that are not being held by territorial residents." However, a banding study in Missouri indicated that the winter and summer populations in that state are completely separate (Kridelbaugh 1982).

Mortality from vehicle collisions is another concern. A high incidence of automobile-caused mortality among shrikes was noted early by Miller (1931). Shrikes typically fly low to the ground, sometimes across roadways, and often feed on roads. Inexperienced juveniles have been observed following adults across highways and learning from adults to feed on highways (Hershberger 1989, Novak 1989). In many areas, hedgerows, barbed-wire fences, and other habitat

features utilized by shrikes are concentrated along roadways. Fledglings and other juveniles are frequently killed by automobiles. Automobile collisions killed all three fledglings produced at an Ontario nest and four of seven young fledged over 3 years at a New York site (Novak 1989). Juveniles killed by vehicle collisions in the summer have also been observed in both Virginia and West Virginia (Luukkonen 1987, Bartgis 1989). In Maryland, three shrikes were known to have been killed by automobiles in the summer of 1991 (L. MacIvor pers. comm.). Blumton (1989) reported that automobiles accounted for 29% of the observed fall and winter mortality among shrikes in Virginia, where it was second only to predation as a cause of shrike mortality.

## MANAGEMENT POTENTIAL

Since reproductive rates are relatively high for shrikes, an excellent potential exists for the species to significantly expand its current population if the cause or causes of the decline can be identified and eliminated. Although suitable habitat has been lost substantially in the Northeast, there appears to be enough remaining unoccupied suitable habitat to accommodate a larger shrike population. However, trends in habitat loss are likely to continue and may ultimately limit any long-term expansion of the shrike population.

## LAND PROTECTION AND PRESERVE DESIGN

In general, greater amounts of good quality habitat (i.e., short grasslands) are associated with smaller nesting territories of loggerhead shrikes (Kridelbaugh 1982). Miller (1931) reported nesting territories of 4.4-16 ha; Kridelbaugh (1982) reported an average territory size of 4.6 ha in Missouri, with territory size increasing significantly after fledging. In New York, Novak (1989) found the smallest area of active pasture in the territory of nesting shrikes to be about 5.5 ha. In Virginia, juvenile shrikes established 2-36 ha (mean = 19 ha) fall and winter territories,

although use of woody habitats in inclement weather significantly enlarged the home range (Blumton 1989). Inclusion of appropriate site-specific landscape features within a preserve design is discussed under *Management and Research*.

The most effective land protection for shrikes will probably require regional land use planning tools, such as zoning, special agricultural districts, and agricultural easements, which will help maintain large areas of suitable habitat. Sites with both regular summer and winter use by shrikes should be priorities for protection in the Northeast. In many circumstances, acquisition may be unfeasible because of agricultural land values, particularly in the Great Valley and adjacent Piedmont of Virginia and Maryland, and the need to keep the land in production in order to maintain it as shrike habitat. Acquisition may also be of limited value until the cause of the decline can be determined. In New York, Maryland and the Virginias, landowner contact programs will probably be more effective. These programs can appeal to a farmowner's sense of pride in the land while offering guidance to assure that key landscape features, such as suitable nest trees, are maintained. Because of the limited opportunities for habitat acquisition, existing federal facilities within primary shrike range, such as Antietam National Battlefield, Maryland, and the Leetown National Fisheries Center, West Virginia, should be encouraged to manage for shrike habitat when appropriate.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

In Maryland, New York, and West Virginia, where there are few shrikes, nesting territories should be located and monitored every year. This may alert biologists to potentially avoidable losses in habitat quality at each occupied territory, although in Virginia, annual monitoring of each territory may not be practical. Hands et al. (1989) suggest monitoring territories in the northcentral states every 3 years.

Monitoring should be delayed until after courtship and copulation, since birds are more difficult to find during this period (Novak 1989). After incubation begins, though, males are typically found hunting and carrying food to the female, making it relatively easy to locate nests at this time. Announcements in ornithological society newsletters and posters placed in agricultural districts can provide leads, and volunteers familiar with local areas have been effectively used to locate nests (Luukkonen 1987, Novak 1989). Because farmland in the Northeast is crossed with many roads, surveys from vehicles are often productive and effective.

Because nesting success appears to be high, close monitoring of nesting effort may be of little value unless it is done in context of a more comprehensive demographic study. Nevertheless, certain site-specific data for a nest site can be used for preserve design purposes and during discussions with landowners. These items include mapping and describing the adult foraging area, fledgling use area, nest tree selection, and perch selection. Kridelbaugh (1982) found that 2-3 hours of observation were sufficient to determine nesting territory size. Miller (1931) thought that a half-hour or less spent observing the pair was sufficient to adequately map a shrike's territory. Changes in gross habitat characteristics, such as the abandonment or tilling of active pasture or the cutting of perches, should be tracked annually.

If an intensive monitoring program is needed, fledglings should be located to determine nesting success. The site should also be revisited through the summer to determine if any additional nesting attempts occur and if the shrikes shift foraging or nesting territory areas. Monitoring reproductive success does not typically require handling the birds. Spotting scopes are usually suitable for observing younger fledglings while more mobile adults and older fledglings are more readily observed with binoculars (Novak 1989). Development of nestlings can be observed with permanently placed mirrors observed with a spotting scope, as long as significant disturbance to the nest cover does not occur (Lohrer 1974). To minimize such disturbance, workers should check nests with mirrors on poles once or twice during each part of the nesting process (Novak 1989) or about once a week (Luukkonen 1987).

To prevent premature fledging, mirrors should not be placed near the nest when nestlings are 17-18 days old (Luukkonen 1987).

For more intensive monitoring utilizing marking techniques, Bal-Chatri traps and Potter traps are usually used to catch shrikes (Brooks 1988, Blumton 1989, Novak 1989), although Lohrer (1974) found that two-cell wire box traps may stress the birds less. Typical bait includes field and white lab mice and zebra finches (*Taeniopygia castanotis*) (Lohrer 1974, Brooks 1988, Novak 1989). At a distance, wing tags are more easily observed than colored leg bands. However, tagged shrikes have been observed entangled in vegetation with injury and mortality occurring as a result (Lohrer 1974). Brooks (1981) and Luukkonen (1987) alternatively banded nestlings before they fledged. None of these researchers or Kridelbaugh (1982) reported any observed injuries from using aluminum USFWS or colored leg bands. Seams of colored leg bands should be sealed with acetone to prevent the shrike from picking the band off (Blumton 1989).

Mortality rates and causes and seasonal dispersal can be determined by using color-banding and radio telemetry (Hands et al. 1989). Color-banding has been done in Maryland (L. MacIvor pers. comm.), Virginia (Blumton 1989), and West Virginia (R. Dean pers. comm.). However, in New York there may be too few shrikes left for such efforts to be meaningful. Some radio telemetry has been done in Virginia (Blumton 1989). Battery-powered transmitters were attached to shrikes with a backpack harness. The entire rig weighed 1.8-2.5 g, about 3.5-5.0% of shrike body weight. However, a few harnessed birds died within 24 hours of capture (Blumton 1989).

Winter habitat remains poorly studied. Foraging area within repeatedly used wintering territories should probably be determined.

Virginia intensively monitored portions of the Shenandoah Valley population from 1985-87 (Luukkonen 1987, Blumton 1989). Monitoring continues, but less intensively, in the same area (K. Finnegan pers. comm.). The Maryland population has been monitored since 1989, with color-banding beginning in 1991 (Hershberger 1989, 1990; L. MacIvor pers. comm.). Color-

banding began in the West Virginia portion of the Shenandoah Valley in 1991 (R. Dean pers. comm.). New York intensively monitored breeding shrikes from 1986-88, but no subsequent nesting has been confirmed (P. Novak pers. comm.). These efforts have been undertaken with funding from the U.S. Fish and Wildlife Service, state Natural Heritage programs and state fish and wildlife agencies. Unfortunately, long-term funding of these efforts remain in doubt.

#### Management Procedures and Programs

Until the cause of the decline can be determined, maintaining suitable nesting and wintering habitat in areas of regular shrike activity is essential. This will require keeping suitable landscape features in active pastures. Suitable nest tree criteria are described under *Habitat*. Kridelbaugh (1982) recommended discouraging multiflora rose (unsupported bushes, not those which climb up trees) in a breeding area since they provide poor nest support. Potential perch sites are also important, as in Virginia, where pastures with many potential perches had higher productivity (Luukkonen 1987). In New York, Novak (1989) reported tall isolated shrubs were typically present near nest sites at a density of three or more shrubs per ha. Removal of fences, hedgerows, and woody vegetation should be avoided, as it eliminates potential perches and lowers habitat quality (Luukkonen 1987).

The timing of habitat selection could cause shrikes to occupy marginal habitat. For example, an inactive hayfield chosen in early spring could appear to a shrike to be similar to an active pasture, but it would be quite different once vegetation begins to grow (Luukkonen 1987). However, in states where shrikes are resident, birds have the opportunity to continually switch habitats. Changes in location, timing and intensity of grazing could still decrease the predictability of habitat suitability (Luukkonen 1987). Consequently, active grazing regimes should be maintained at the nest site, or if stock use is rotated between fields, the fields should be adjacent so that shrike foraging activity will require minimal changes.

Herbaceous vegetation that is allowed to grow too tall or woody vegetation that becomes

too dense eliminates the area as potential foraging habitat for shrikes. Thorny shrubs, barbed-wire fences, and other objects suitable for impaling prey are also significant features of shrike habitat that should be maintained. For specific sites in the Northeast where it is important to maintain a given shrike territory, landscape features that could attract shrike activity to roadways, such as perches and potential nesting trees near highways, might be eliminated if similar features occur away from roadways. Such management could reduce mortality resulting from collisions with automobiles. (In contrast, in the northcentral states, Hands et al. (1989) recommended maintaining perches along a road since shrikes frequently use roadsides.)

Hands et al. (1989) also recommended restricting pesticide use in shrike habitat in order to avoid reducing the abundance of potential prey items. Also, for local situations where predation is identified as a problem, predator management may be necessary once nest and fledgling predators are identified (Hands et al. 1989).

At present, there are no large-scale habitat management programs for the loggerhead shrike in the Northeast. The proposed land management plan for Antietam National Battlefield includes a comprehensive program for shrike habitat management, including pasture creation and maintenance, nest tree planting, and roadway design (National Park Service 1991). An experimental habitat management program was begun in 1991 at the Presque Isle National Wildlife Refuge, Chesterfield County, Virginia, where several hawthorns were planted in an old field (D. Pence pers. comm.).

#### Research Needs and Programs

Because the loggerhead shrike population has become so small, particularly outside Virginia, that the loss of any breeding pair may be significant, tracking nesting locations and nesting attempts in all states is probably worthwhile. This may identify site-specific problems that can be ameliorated and help contribute to determining the causes of the decline.

Determining the cause(s) of the decline remains the major emphasis of most research activities. In the northeastern and northcentral

states, studies of reproductive success have not provided an answer. However, studying the causes of mortality could prove valuable. For example, is automobile mortality highest for migratory birds, which encounter a greater number of areas with which they are unfamiliar? Is mortality from automobiles higher for inexperienced juveniles than for adults? Are BBS trends correlated with geographic trends in road density?

Although pesticides have not been linked to lowered reproductive success, additional studies of other impacts from pesticides on shrikes should also be conducted. Although the decline in potential breeding habitat acreage in the Northeast is well documented, the impacts of habitat fragmentation remain to be addressed. One approach would involve comparing size and degree of isolation for blocks of suitable habitat among areas which have a stable, declining or extirpated shrike population. Several researchers expressed concern that declines in winter habitat quality or quantity may be affecting migrant populations. Tracking foraging success and overall vigor of resident versus migrant birds in the winter may provide some answers.

For a description of research programs in the Northeast, see *Monitoring Procedures and Programs*.

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# BACHMAN'S SPARROW

*Aimophila aestivalis*

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Bachman's sparrows historically resided in the southern pinelands and expanded their range northward as far as Canada in the early 1900s. In the Northeast the species breeds typically in old fields or recent clearcuts with dense grasses and scattered shrubs or saplings. Following a recent population decline in the Northeast, Bachman's sparrows now breed only in southern Virginia, and possibly in West Virginia and the western part of Virginia. The species may also be declining in the South. Loss of early-successional habitats is largely responsible for the recent declines. Management techniques that repress succession should be used to create suitable habitat, particularly near areas that are currently occupied by the sparrows. Controlled burns alone or in combination with clearcutting are effective techniques to reduce overstory cover and to encourage dense grass and forb growth, and should be repeated at least every 10 years to reduce sapling growth.

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## DESCRIPTION

### Taxonomy

Bachman's sparrow (*Aimophila aestivalis*), formerly known as the pinewoods sparrow, is 1 of 13 species in the genus *Aimophila*, a group of rather secretive and generally non-descript sparrows found in North, Central, and South America. The species was first seen in 1832 by Dr. John Bachman, a colleague of John James Audubon, on the Edisto River in South Carolina

(Sprunt and Chamberlain 1970). Its taxonomy has undergone several revisions since the species was described as *Fringilla bachmani*. The genus was later changed to *Peucaea*, derived from the Greek for pine, and the species name was changed to *aestivalis*, a reference to "summer." At that time *bachmani* became the subspecies name (Sprunt and Chamberlain 1970). In 1931 a revision of the American Ornithologists' Union check-list put the bird in the genus *Aimophila* ("thicket loving") (American Ornithologists' Union 1931). Three subspecies are recognized (American

Ornithologists' Union 1957). Bachman's sparrow is somewhat similar in appearance to Botteri's sparrow (*Aimophila botterii*), but they do not occur together.

### Morphology and Plumages

Among adults the sexes are similar, measuring 14.0-15.9 cm in length and weighing 18.4-22.6 g (Wolf 1977). The top of the head, hindneck, and back are russet or chestnut brown, heavily streaked with light gray. The back is spotted with brownish-black, while the underparts are an unstreaked buffy-gray (Oberholser 1974). The tail measures 6.0-6.6 cm long and is olive-brown or dark-brown, edged with gray or olive-brown. Wings range from 5.5-6.6 cm in length, with dark-brown primaries and secondaries, and olive-brown to fuscous tertials, tipped with hazel or pale-buff (Wolf 1977). The wing's edge at the wrist is lemon-yellow. The head has a broad, grayish-buff superciliary stripe and dull russet postocular streak, but the rest of the head and neck is washed with gray, and the sides of the neck are streaked with russet. The upper mandible is dark-brown; the lower mandible yellowish-brown. The iris is brown, and the legs and feet are flesh-colored or pink (Drilling 1985).

In the field adults are generally recognized by their song, but their rather large bill, buffy breast, dark eye line, and reddish-brown streaked upperparts separate them from other somewhat similar species, such as the field sparrow (*Spizella pusilla*), chipping sparrow (*Spizella passerina*), and grasshopper sparrow (*Ammodramus savannarum*).

Fledglings have heavily-streaked breasts. Young birds molt all their pennaceous body feathers twice in their first 6 months, an unusual pattern that may replace plumage that is severely worn by the bird's abrasive habitat (Willoughby 1986). The birds undergo a partial molt soon after fledging, replacing nearly all the body feathers, and producing a plumage with adult body feathers and juvenal wing and tail feathers. The breast is spotted and intermediate in appearance between the streaked juvenal and unmarked adult pattern (Willoughby 1986).

In the first fall (September through early December) the bird completes a full molt, replacing all the body feathers again as well as the

remiges and retrices. After this second molt, hatching year birds become essentially indistinguishable from adults. Adult Bachman's sparrows undergo an extended body molt during the spring and summer and a complete molt in the fall.

### Vocalizations

The song of the Bachman's sparrow consists of two parts--a long, clear introductory note, followed by a trill or series of identical notes on a different pitch from the introductory note. The series may be higher or lower in pitch than the introductory note, and consecutive songs are usually varied between higher and lower series. The song is frequently described as "see-slip-slip-slip-slip," though the series often consists of six to eight notes. Otherwise, the species is rather silent, giving a high-pitched "seet" or "sip" call note only infrequently, and seldom in the winter season.

### HABITAT

#### Breeding Season

Bachman's sparrows have been reported to nest in a wide variety of open habitats within their moderate-sized range, including abandoned pastures, agricultural fields, and clearcuts (Ganier 1921, Brooks 1938, Clayton 1969, Dorsey 1976). In the southern states, it typically nests in open longleaf pine (*Pinus palustris*) forests with an abundant ground cover of wiregrass (*Aristida stricta*) or saw palmettos (*Serenoa repens*) (Brooks 1938, Wolf 1977, Stoddard 1978, Haggerty 1986). However, the sparrows have also adapted to tung-oil groves in Georgia and Florida (Stoddard 1978). Farther to the north (outside of the range of longleaf pine), typical habitats are old fields with dense grasses and scattered shrubs or saplings (Brooks 1938). In the western portion of its range, the species is also found in naturally-occurring glades and open stands of hardwoods such as oak (*Quercus* spp.) savannas, where dense grassy ground cover is present (Hardin 1977).

The fact that this species appears to be adaptable to a wide variety of readily available habitats, yet is relatively scarce, led to an important study of habitat selection by Bachman's sparrows in South Carolina. Dunning and Watts (1990) quantified the habitat characteristics of clearcut, young, middle-aged, and mature stands of longleaf and loblolly pine (*Pinus taeda*) forests and compared these to the density of breeding Bachman's sparrows. The preferred vegetation characteristics included a high volume of vegetation, especially grasses and forbs, in the first meter above the ground, and low volumes of understory vegetation in the second through fourth meters above the ground (Dunning and Watts 1990). Mature stands were generally chosen over younger ones because they more often met the preferred characteristics. Some clearcuts meeting the requisite vegetation characteristics supported few birds. Here site preparation techniques that reduced surface vegetation and debris resulted in the loss of standing dead timber from the understory. Also, the loss of song perches may be significant to the birds. The results of this study and those of other authors (Haggerty 1986, Wan A. Kadir 1987) suggest the structure and vegetational characteristics preferred by Bachman's sparrows may be found in a variety of ephemeral habitats.

In the Northeast, the breeding habitat is mainly old, abandoned fields or recent clearcuts. Essentially all sites have a dense ground cover of grasses or forbs, with scattered shrubs or saplings (pines or hardwoods) usually present for singing perches. Plant species such as broomsedge (*Andropogon virginicus*), poverty grass (*Danthonia spicata*), goldenrods (*Solidago* spp.), asters (*Aster* spp.), fleabane (*Erigeron* spp.), blackberries (*Rubus* spp.), greenbrier (*Smilax* spp.), and sumac (*Rhus* spp.) are typically present (Weston 1968). In parts of the species' range in the Northeast, such as West Virginia, habitat is often on fairly steep slopes, and many nesting sites contain an erosion gully (Hall 1983). These sites may be on abandoned farmland or in clearcuts. In extreme southern Virginia, the breeding habitat is usually recent clearcuts, with dense grasses and forbs and a scattering of shrubs and saplings, especially loblolly pines. These clearcuts are usually quite extensive ( $\geq 20$  ha) and are usually on flat or

gently sloping ground (Kain 1987). Throughout the breeding range in the Northeast, the habitat is commonly shared with the field sparrow.

### Nonbreeding Season

Bachman's sparrow does not winter in the Northeast (Hardin and Probasco 1983, Root 1988). The species is a permanent resident in open longleaf pine forests of the South (Sprunt and Chamberlain 1970, Oberholser 1974, Imhof 1976, Johnsgard 1979, Potter et al. 1980). It winters from North Carolina southward, primarily in dense grassy cover (Potter et al. 1980, Root 1988), departing from at least some habitats in the more northern parts of its range as late as October (Gainer 1921, Mengel 1965). The habitat is almost always open, though frequently sheltered by a canopy of scattered longleaf pines. The birds also winter in broomsedge fields but they generally avoid fields dominated by forbs such as asters and goldenrods (Weston 1968, Sprunt and Chamberlain 1970). Weston (1968) found wintering birds in riparian habitats and sometimes along the saltwater shores of coastal woodlands.

## BIOLOGY

### Courtship

The onset of the breeding season among Bachman's sparrows varies with latitude and local weather conditions (Sprunt and Chamberlain 1970). In the Deep South, where the males are more likely to be resident, singing may begin as early as February, and by March the song period is in full swing (Stoddard unpubl. ms., cited by Burleigh 1958, Meanley 1959, Sprunt and Chamberlain 1970). In the Northeast singing presumably begins immediately upon the arrival of the males in April (Weston 1968). Song perches are usually fairly conspicuous, generally in the top of a sapling or shrub in clearcuts or old fields. Bachman's sparrows are also believed to sing on the ground (Brooks 1938, Meanley 1959). In the open pinewoods of the South, song perches are commonly the dead lower branches or stubs of living pine trees, from 1.5-3 m (Meanley 1959) to

3-6 m above the ground (H. Legrand pers. obs.). Weston (1968) reported that some males have favorite song perches from which they sing while looking down at their mate on the nest. However, this statement was questioned by other observers.

Singing occurs at all hours of the day, peaking in the early morning and evening and sometimes continuing into moonlit nights (Brooks 1938). Meanley (1959) reported 1 bird in Louisiana that sang 52 songs from the same perch in 15 minutes. Singing continues well into midsummer, usually into July and often into August or September (Stoddard unpubl. ms., cited by Burleigh 1958). Sprunt and Chamberlain (1970) report singing in December.

#### Nesting and Care of the Young

In southern Missouri, Hardin et al. (1982) recorded the average size of 13 nesting territories measured during 1 nesting cycle as 0.62 ha. Haggerty's (1986) 2-year study of 26 territories in Arkansas found a mean size of 2.49 ha, however his measurements were made over the entire breeding season.

The nest of Bachman's sparrow is usually placed on the ground against a grass tussock or under a clump of saw palmetto (Weston 1968, Wolf 1977, Haggerty 1988). It is constructed of various grasses and weed stems and lined with fine grass and/or horsehair (Ganier 1921). Blades of grass from the tussock are frequently woven into the nest (Haggerty 1988). The female builds the nest without the assistance of the male although he may accompany her and sing during the process (Weston 1968). Some nests are apparently dome-shaped, with the entrance on the side (Bendire 1888, Ganier 1921, Reed 1965, Weston 1968), whereas others are simple cups like those of the majority of songbird species (Weston 1968, Haggerty 1988). Brooks (1938) summarized the debate about nest structure in this species. It seems likely that nest shape shows a continuous distribution from a simple cup to a misshapen cup, somewhat higher on one side, to one that is fully domed.

Clutch size ranges from three to five eggs, and is typically four, with one egg laid each day probably at dawn (Weston 1968, McNair 1986,

Haggerty 1988). The eggs are ovate, slightly glossy-white and unspotted, measuring approximately 19 x 15.5 mm (Weston 1968, Sprunt and Chamberlain 1970, Haggerty 1988). Incubation begins after the penultimate egg is laid and is done entirely by the female (Haggerty 1988). The eggs hatch after 12-14 days (Meanley 1959, Sprunt and Chamberlain 1970, Haggerty 1988). Haggerty (1988) found that 16 of 58 clutches showed some evidence of asynchronous hatching.

Incubating females are notoriously difficult to flush. Brooks (1938) and Ganier (1921) both reported approaching to within a few feet of the nest before the disturbed adult flew to a perch approximately 1 m away, then dropped to the ground and continued to run through the grass. After fledging, however, the protective behavior of the parents appears to change. Ganier (1921) and Brooks (1938) describe a distraction display during which the adult bird feigns injury by dragging a wing and fluttering along the ground like a killdeer (*Charadrius vociferous*).

The young are fed by both parents and fledge after approximately 9-10 days in the nest (Haggerty 1988). When the adults are feeding nestlings they do not fly directly to the nest. Rather they land on a distant perch, drop to the ground and approach the nest through the vegetation (Brooks 1938). Meanley (1959) reported that the young are fed primarily grasshoppers (Orthoptera) and beetle larvae (Coleoptera).

The fledgling birds continue to be fed by the parents for approximately 25 days. During this time the female often initiates a second brood and under these circumstances the male assumes the majority of the feeding responsibilities (Haggerty 1988).

#### Food Habits

The food habits of this species are rather poorly known because of the bird's secretive nature. Based on examination of the stomach contents of 10 birds, Howell (1924) found that in Alabama the food consisted of 58% animal matter and 42% vegetable matter. Animal items included leaf beetles, other beetles, bugs, grasshoppers, and crickets. Vegetable items were

mainly seeds of grasses and sedges. Allaire and Fisher (1975) found that Bachman's sparrows were strictly ground feeders taking only seeds that had fallen to the ground. During the breeding season feeding peaked during the first 5 hours after sunrise and the last 2 hours before sunset. Winter feeding was more irregular. Data based on stomach analyses from this Texas study showed that Bachman's sparrows ate mostly grass (*Panicum* spp.) seeds through the summer, fall and winter.

Sprunt and Chamberlain (1970) described Bachman's sparrow as more insectivorous than many other sparrows but provided no quantitative data. One might expect that, during the winter, the diet shifts more toward seeds and other vegetable items, whereas in the warmer months, when more insects are available for consumption, animal matter likely predominates. Meanley (1959) found that the stomachs of five Louisiana birds taken in winter contained primarily grass seeds of *Panicum* and *Sporobolus* spp., beetles and pine seeds. Brooks (1934) described a pair of Bachman's sparrows that came to a window feeding shelf in May and seemed to prefer shelled black walnuts.

### Migration

The species is partially migratory over its range. Birds inhabiting longleaf pines and other habitats from southeastern North Carolina to eastern Texas are present year-round and are likely non-migratory. However, birds that breed north of this region are absent from the breeding sites in winter and spend that time within the range of the southern resident birds. Departure dates are very poorly known because the species is difficult to locate except when males are singing. Departure dates from the Northeast appear to be mainly in late August and September (Weston 1968, Wolf 1977).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

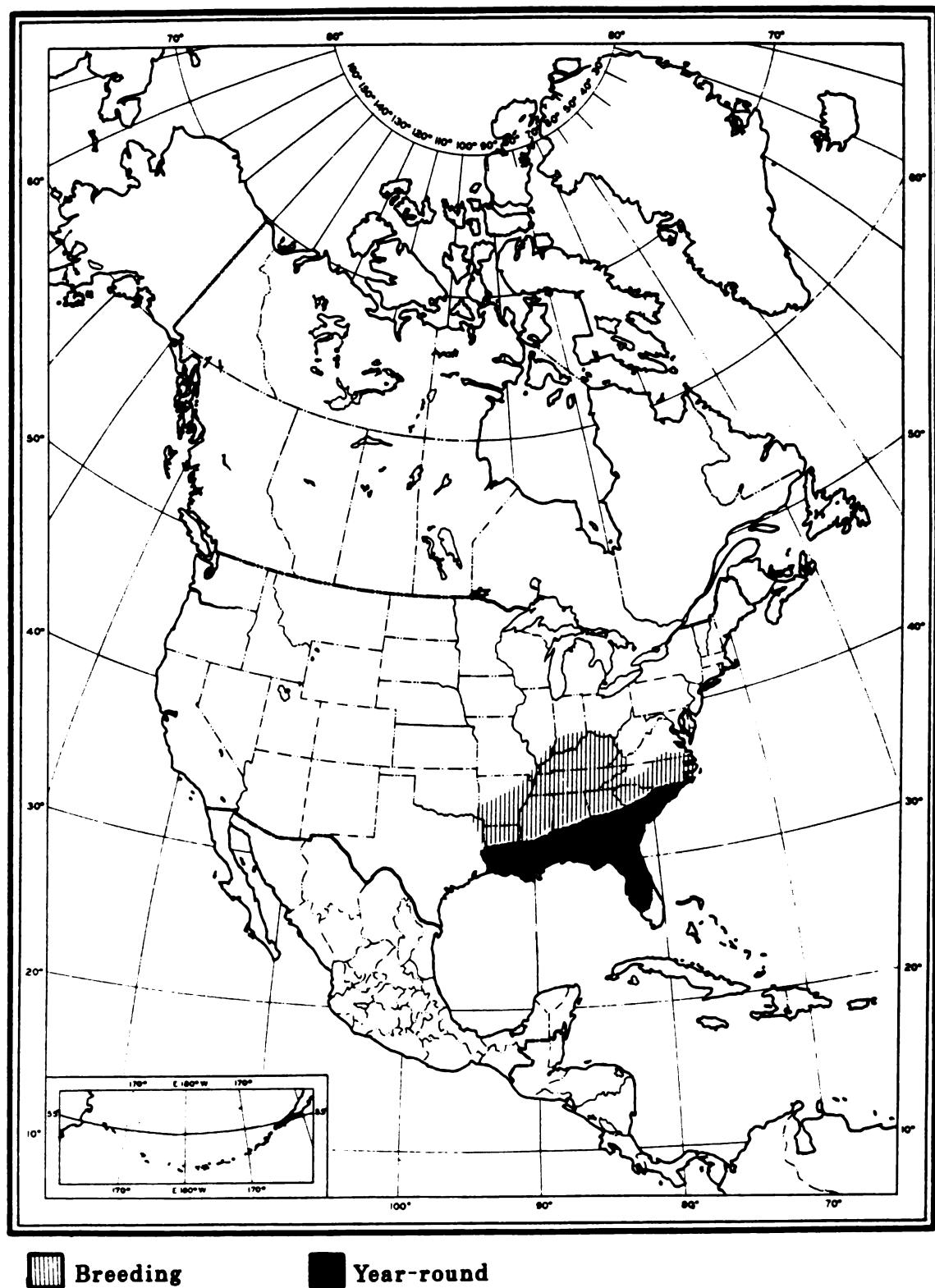
Bachman's sparrow "...breeds (at least formerly) from south-central Missouri, central and northeastern Illinois, central Indiana, central Ohio, southwestern Pennsylvania and central Maryland south to eastern Oklahoma, eastern Texas, the Gulf coast and south-central Florida; [and is] now generally absent (or very local) as a breeding bird in the northeastern portion of the breeding range north of southern Kentucky and North Carolina" (American Ornithologists' Union 1983, Figure 1).

The species winters "...from eastern Texas, the Gulf states and Atlantic coast (from southeastern North Carolina) south through the remainder of the breeding range (casually elsewhere in the northern parts of the breeding range), and to southern Florida" (American Ornithologists' Union 1983).

Within this moderately large range, the species varies widely in abundance. Nowhere is the sparrow truly common; however, it is fairly common to locally common (at least when singing in the breeding season) in the Coastal Plain from southern Louisiana east to northern Florida, and north to South Carolina and the Sandhills of North Carolina. Natural Heritage Programs in Arkansas, Florida, Georgia, Louisiana, Mississippi, North Carolina, and South Carolina all rank the species S3 (21 to 100 occurrences in the state) or S4 (apparently secure) (The Nature Conservancy Central Scientific Databases, Arlington, Virginia). Generally speaking, it is fairly common in areas dominated by longleaf pine, especially in savannas and other open stands with little or no hardwood understory. The species is uncommon to very rare away from longleaf pine and occurs mainly in old fields, glades, and clearcuts; away from pines, it appears to be most numerous in Arkansas. Bachman's sparrow is now essentially absent in the northeastern corner of its range--Ohio, West Virginia, Pennsylvania, and Maryland--where it was at times numerous in the early and middle decades of this century (Brooks 1938).

The species is very secretive in winter and is essentially non-vocal at that season. Most

Figure 1. The North American range of Bachman's sparrow (*Aimophila aestivalis*) (American Ornithologists' Union 1983).



■ Breeding

■ Year-round

accounts from states in the winter range indicate that the species is uncommon (Root 1988). Of course, since the species withdraws from the northern half of the breeding range to winter in the Coastal Plain, numbers in winter in this latter region must be somewhat greater than in summer.

Historically a bird of the southern pinelands, Bachman's sparrow expanded its range into Ohio, West Virginia, Pennsylvania, Illinois and even Canada in the early 1900s (Eifrig 1915, Brooks 1938). This expansion appeared to peak from 1915-22 and concern mounted as the birds disappeared from formerly occupied habitats. Since the 1970s the population decline has become even more evident. In addition to a contraction of the species range at its northern edge, data now suggest that populations in the southern part of the species range have also disappeared (Jackson 1985, Peterjohn 1987). Bachman's sparrow has been continuously listed on the National Audubon Society's *Blue List* since 1972 (Tate 1986) and The Nature Conservancy gives it a global rank of G3, indicating that it is rare and local throughout its range (The Nature Conservancy *Central Scientific Databases*, Arlington, Virginia). The Office of Migratory Bird Management considers the bird a *migratory nongame bird of management concern* (U.S. Fish and Wildlife Service 1987), while the Office of Endangered Species proposed the species as a *candidate (category 2)* for listing, indicating that listing may be warranted but that additional data are needed to evaluate the species' status accurately (U.S. Fish and Wildlife Service 1991).

### Distribution and Status in the Northeast

At this time the Bachman's sparrow is nearly extirpated in the Northeast. In the last decade, however, field work has revealed a handful of breeding season records from extreme southern Virginia (Figure 2). The species was much more numerous and widespread in the Region, especially in West Virginia, during the first half of this century. There are no records from Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island or Delaware. The status of Bachman's sparrow in the remaining northeastern states is discussed below.

**New York.** - Bachman's sparrow is considered accidental in New York State. In 1930 a specimen was taken from Greenport, Long Island (Bull 1964). In 1940 it was recorded from Mendon Ponds Park near Rochester (Drennan 1981) and a singing male was reported on two successive days in Prospect Park in New York City in April of 1948 (Carleton 1958). The species is ranked SA by the state Natural Heritage Program (Table 1).

**Table 1.** Summary of state listing status and Natural Heritage Program state ranks for Bachman's sparrow, (*Aimophila aestivalis*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	-
New Hampshire	-	-
Vermont	-	-
Massachusetts	-	-
Rhode Island	-	-
Connecticut	-	-
New York	-	SA
Pennsylvania	-	SX
New Jersey	-	-
Delaware	-	-
Maryland	-	SH/SX
Virginia	-	S1
West Virginia	special concern	SH

<sup>a</sup>SH = historical records only, but suspected extant

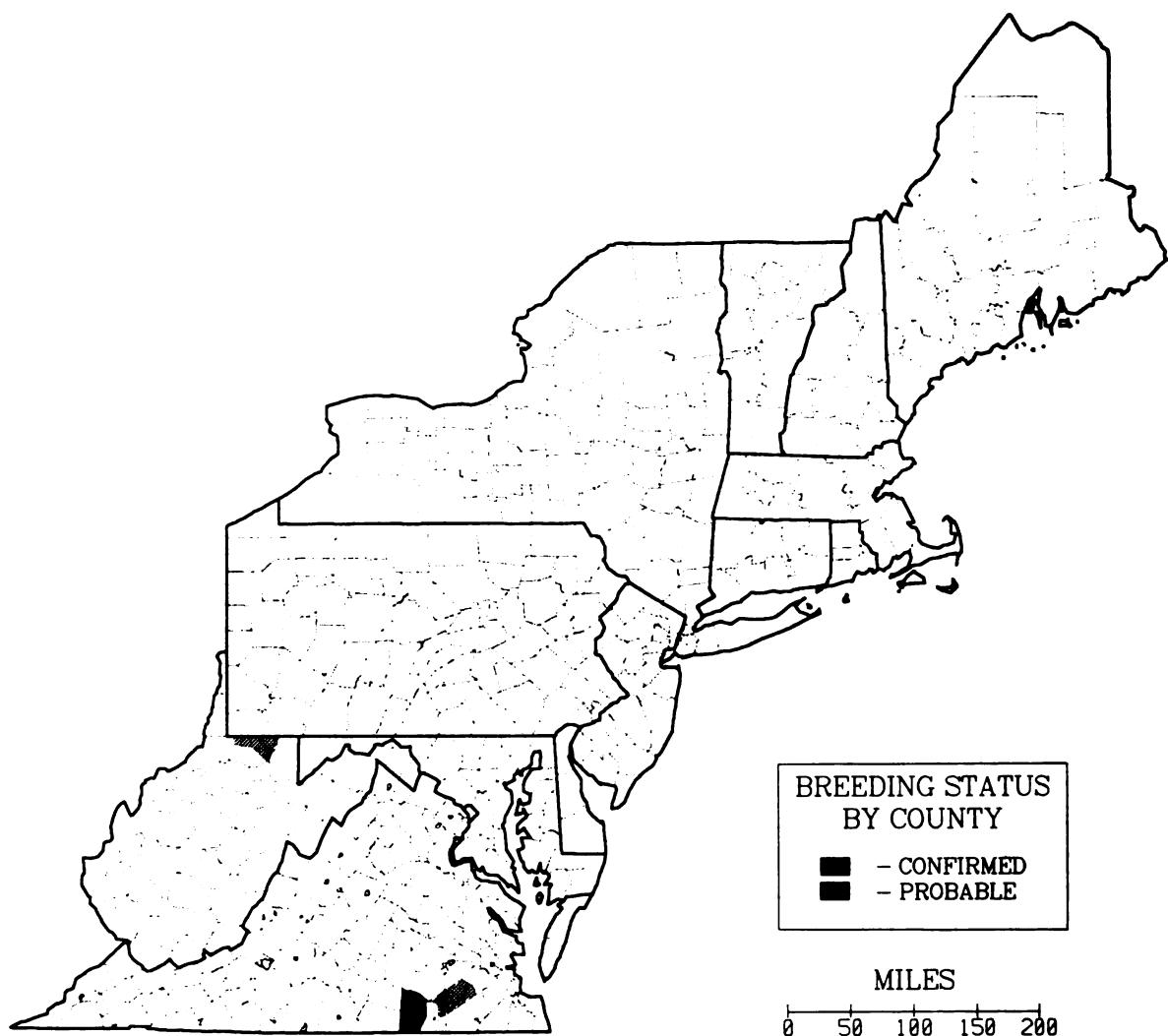
SX = apparently extirpated

SA = accidental

S1 = critically imperiled; ≤ 5 occurrences

**Pennsylvania.** - Historical breeding records of Bachman's sparrow are known from the southwestern corner of Pennsylvania. Historically it was reported from Greene, Allegheny, Beaver and Washington counties (Todd 1940), but there have been no documented breeding records in Pennsylvania in recent years. Today the species is believed to be extirpated from Pennsylvania and

**Figure 2.** Northeast breeding distribution of Bachman's sparrow (*Aimophila aestivalis*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



is ranked as such, SX, by the state Natural Heritage Program (Table 1).

**New Jersey.** - Bachman's sparrow is considered accidental in New Jersey. One singing male, deposited at the U.S. National Museum, Washington, D.C., was taken at Fort Lee, New Jersey in May of 1918 (Bull 1964).

**Maryland and the District of Columbia.** - Historically Bachman's sparrow occurred over most of the state of Maryland but it was most abundant west of the Fall Line (Stewart and Robbins 1958). The last confirmed breeding of Bachman's sparrow in Maryland occurred in 1964 (Maryland Natural Heritage Program 1985) and the state Natural Heritage Program lists it as extirpated from the state (SH/SX) (Table 1).

**Virginia.** - In Virginia, four or five breeding sites have been found since 1984, all in the southeastern corner of the Piedmont in Southampton, Sussex, and Brunswick counties (Figure 2). Several sites occur in Brunswick County, including confirmed nesting (Kain 1987). In the Coastal Plain, Bachman's sparrow is described as "...formerly a rare spring and summer visitor in eastern Fairfax County, but last recorded there in 1964" (Kain 1987). In the Piedmont, it was "...formerly a locally uncommon transient and summer resident...said to have been common in the late 1900's" (Kain 1987). In the Mountains and Valleys, Kain (1987) describes it as a "...formerly locally uncommon to rare transient and summer resident." The Virginia Breeding Bird Atlas (BBA) produced confirmed records from Brunswick County in the Piedmont and a probable record from the Coastal Plain of the state (S. Ridd and R. Wadja, unpubl. data). The species might conceivably still occur somewhere in extreme western Virginia. The Virginia Division of Natural Heritage ranks the Bachman's sparrow as S1, indicating that there are fewer than five occurrences in the state (Table 1). The species has been recommended for state endangered status.

**West Virginia.** - Hall (1983) considers Bachman's sparrow a rare migrant and rare local summer resident in West Virginia. It was first

recorded from Wood County in 1903 (Brooks 1912) thereafter becoming common and widespread in much of central West Virginia, reaching a maximum abundance between 1915-22 (Hall 1983). The last confirmed breeding record was from 1972. The bird is listed as a *species of special concern* by the West Virginia Nongame Wildlife Program (West Virginia Department of Natural Resources 1987) and SH (historical records only) by the state Natural Heritage Program (Table 1). One probable breeding record was noted during the West Virginia BBA (C. Stihler and A. Buckelew, Jr. unpubl. data, Figure 2).

#### Summary

The Bachman's sparrow was somewhat widespread in the early half of the century in portions of Virginia, West Virginia, Maryland, and Pennsylvania. As of 1990, the species bred with certainty only in southern Virginia but may still occur in West Virginia and the western part of Virginia. Outside of the Northeast, the sparrow has declined sharply in the northern states away from the longleaf pine belt. In the South, the decline has been much less severe. In fact, > 20 years of data from the North American Breeding Bird Survey do not show any clear-cut regional declines for the species in the South (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).

#### LIMITING FACTORS AND THREATS

##### Habitat Loss

Over much of the range of Bachman's sparrow, the reasons for its decline are not obvious. However, recent research by Dunning and Watts (1990) has shed considerable light on the problem. Bachman's sparrow expanded its range northward at the turn of the century and populations appear to have peaked by the 1920s and declined in the late 1930s (Brooks 1938, Haggerty 1986). This expansion, followed by a pronounced decline, was almost certainly a response to successional changes on abandoned

farmland (Haggerty 1986). Today there is little abandoned farmland compared to the acreage left idle in the early decades of this century (Odum 1987). Such farmland becomes unsuitable for Bachman's sparrows after approximately 10 years; today most of it is probably woodland.

Within the overall range of the species, habitat loss or modification is a major concern. Weston (1968) discusses the expansive virgin pine forests of the middle 1890s that existed before nearly all the land was timbered (Platt et al. 1988). This fire-adapted community underwent periodic fires that created nearly ideal nesting habitat for Bachman's sparrows. The conversion of longleaf pine forests to agriculture, and especially to fast growing plantations of loblolly pine or slash pine (*Pinus elliottii*), have had major effects on populations of these birds, which use such converted pine forests until the canopy closes 5-10 years after planting (Dunning and Watts 1990).

On the other hand, the birds can exist, conceivably over long periods of time, in naturally occurring, open longleaf pine forests that are maintained by frequent fire. Unfortunately, suppression of natural fire is all too frequent in recent decades; as the human population spreads into more wooded areas, wildfires are more easily detected and more quickly extinguished. Decades ago, such wildfires could burn for days and cover thousands of acres of land, keeping the shrub and understory layers under control while at the same time promoting a dense grass and forb stage. Management of forests by fire (controlled burning) is becoming increasingly difficult because of the presence of nearby homes and the dangers of smoke near highways and human habitation. Suppression of natural fire is also causing a loss of suitable habitat in the western part of the range, where the birds inhabit open stands of hardwoods such as oak savannas.

Bachman's sparrows apparently are able to breed in a variety of open habitat types that have a dense ground layer and an open understory (Dunning and Watts 1990). These include abandoned farmlands, pinelands, oak savannas, glades, barrens, and clearcuts. In the absence of continued active management or natural fires, these habitats quickly become overgrown and unsuitable for Bachman's sparrow. This species'

reproductive capacity is clearly dependent on its ability to colonize new patches of appropriate habitat and adapt to a shifting mosaic of breeding sites. As appropriate breeding sites become more scarce and isolated, the birds may be unable to adapt to a new scale of shifting habitat types, resulting in local extinctions (Dunning and Watts 1990).

### Diseases, Parasites, and Predation

Few data suggest that disease has played a role in the decline of the Bachman's sparrow. Ticks, mites, and lice have been reported on birds over portions of their range (Peters 1936), but it seems unlikely that parasites have contributed significantly to the decline, though this aspect of their biology certainly needs further study.

Predation of eggs and nestlings is a potential problem for most bird species, especially those that nest on the ground like the Bachman's sparrow. Haggerty (1988) found breeding failures to be a major source of mortality among Bachman's sparrows in Arkansas. Of 164 eggs, 6 (2%) were thought to have been removed by female cowbirds (*Molothrus ater*) and replaced with eggs of their own. Only 2 of 66 nests failed because of cowbird parasitism, however. Predation was the major cause of nest failure, accounting for 95% of the nest loss (Haggerty 1986). Of 35 eggs that were lost, 20% were removed by cowbirds and 80% were depredated. Of 115 nestlings, 78% were killed by unknown predators, 11% starved or died of unknown causes, 6% were killed by snakes, and 4% were killed by unidentified mammals (Haggerty 1986). In Arkansas, the yearly production of young was approximately three fledglings per pair (Haggerty 1986), but post-fledging mortality remains essentially unknown. Haggerty (1986) concludes, however, that in central Arkansas annual fecundity and the probability of egg success do not appear to be abnormally low.

### Other Factors

Interspecific competition must always be addressed when assessing causes of declines. One somewhat similar species, the field sparrow, has a

niche approaching that of the Bachman's sparrow. It is generally found with the latter in the northern portions of the range, and almost always outnumbers Bachman's in clearcuts and old fields. However, observers studying Bachman's sparrows have not detected any interactions between these species (Hardin et al. 1982, Haggerty 1986). The field sparrow often feeds in shrubs and saplings and less often on the ground, whereas the Bachman's sparrow feeds almost exclusively on the ground (Allaire and Fisher 1975). In addition, the field sparrow has also undergone a noticeable and significant decline in the last 10-20 years over much of its breeding range, as have most other birds of old field habitat (Office of Migratory Bird Management unpubl. data: BBS data). This decline is apparently due to the loss of abandoned farmland and habitat destruction for development.

Another potential threat, or cause of decline, is severe weather on the wintering grounds. Unusually cold weather, especially if snow or ice covers the ground for extended periods of time, is known to produce heavy mortality of ground-dwelling birds. Whether or not populations of Bachman's sparrows have been depleted by winter mortality is completely unknown.

## MANAGEMENT POTENTIAL

Bachman's sparrow has a moderate management potential. Probably little can be done to encourage farmers or landowners to abandon farmland for the bird's benefit. On the other hand, it seems likely that active management through clearcutting and controlled burns could easily create suitable habitat for these birds, although the proximity of existing populations to manageable sites is also an important consideration. At first, restoration of the birds to suitable habitats may require reintroduction. When the dispersal ability of Bachman's sparrows is more clearly understood, it may be possible to design a network of managed areas that could be occupied through natural dispersal and colonization.

## LAND PROTECTION AND PRESERVE DESIGN

Because the species in the Northeast inhabits early successional habitats that are human-modified and created, there is a philosophical question of whether land protection through acquisition is a suitable or feasible method of protection for the sparrow. Normally, agencies are involved in the acquisition of naturally occurring, climax habitats and natural communities, such as pine savannas, prairies, cedar glades, and mature hardwood forests. The acquisition of sites that require ongoing management to ensure the continued existence of rare species involves a long-term financial commitment that many agencies are unwilling or unable to make. Therefore, the identification and protection of Bachman's sparrows on sites that can be maintained through natural processes should be the first priority. However, in the absence of fire, it seems likely that few, if any such sites exist in the Northeast or perhaps even throughout the species entire range.

The second priority should be the protection of Bachman's sparrows where existing management practices can be easily modified to accommodate the birds' needs for nesting habitat. Some timber lands and agricultural settings may provide management opportunities that could benefit the birds and remain cost-effective. Clearcuts < 8 ha are seldom occupied (H. LeGrand pers. obs.), therefore sites managed in this way should provide a minimum of 75 ha of suitable habitat in any one breeding season. Size and shape of the habitat are also important. Areas managed for Bachman's sparrows should be somewhat square, rectangular, or circular in shape, rather than long and narrow because the birds do not seem to find powerline clearings or other narrow clearings suitable.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Bachman's sparrows are easily detected by song if observers listen at clearcuts and other

early successional habitats. Surveys should be done in the morning hours, from late April into June. Singing declines after late June, though surveys in July or August might not be fruitless. A taped-recording of the song played at dusk sometimes elicits a response at suitable habitat where no birds are initially heard calling (T. Haggerty pers. comm.).

Attempting to find the birds in fields by walking to flush silent individuals is both labor-intensive and inefficient. On the other hand, once singing birds are located, it is worthwhile to search for nests to monitor the fate of the nest and nestlings. Nests are most easily located by watching the behavior of adult birds during nest building or feeding of the young (Haggerty 1988). Occupied sites should be monitored annually to determine the number of singing males and estimate annual productivity.

Most existing monitoring programs, such as the Breeding Bird Survey and National Audubon Society's Christmas Bird Counts, are inadequate to provide statistically meaningful data on a species like Bachman's sparrow that has declined to very low numbers in the Northeast. Projects such as breeding bird atlases are especially useful in locating new sites for early-successional species. However, Bachman's sparrow is so uncommon in the Northeast that many observers are not familiar with its song. This situation increases the likelihood that it may be overlooked.

### Management Procedures and Programs

Bachman's sparrows can be managed through the use of controlled burns. Timber management practices that produce suitable habitat for red-cockaded woodpeckers (*Picoides borealis*) also provide habitat for Bachman's sparrows (Dunning and Watts 1990). At Francis Marion National Forest in South Carolina, where red-cockaded woodpeckers were common prior to Hurricane Hugo in September 1989, forest compartments are burned on a 3-5 year rotation schedule. This burning schedule produces a dense ground layer of bracken fern (*Pteridium aquilinum*), grasses, and blueberries (*Vaccinium* spp.), and an open understory.

Where management of fire-adapted forests through the use of controlled burns is not

possible, it may be feasible to manage the habitat of Bachman's sparrows through the use of clearcutting. Management for the species should encourage clearcutting of forests instead of selective cutting when a decision is made by the landowner or land manager to remove timber from a tract. However, a clearcut, without burning of the tract following timber removal, might not provide the suitable grass cover that is needed by the birds for foraging and nest placement. Thus, burning of a tract should follow cutting to ensure a dense grass cover. At present, many or most clearcuts are apparently not burned following timber harvest, and the cleared areas are left with a stubble of shrubs and other woody vegetation. Burning of clearcuts is a favorable management tool because fire kills or retards such woody and hardwood "stubble." Most importantly, burning stimulates the growth of grasses and forbs, features the sparrow requires in its habitat. Such habitat can remain suitable for Bachman's sparrows for 5-10 years, though it may take 1-2 years after clearing (with or without fire) for a dense cover of grasses to become established. After about 10 years, saplings may become too dense in the field for the sparrows to inhabit it (Dunning and Watts 1990).

In the absence of a short burning rotation, both mature stands and clearcuts quickly become unsuitable for nesting by Bachman's sparrows. Therefore, until we have a better understanding of the dispersal abilities of Bachman's sparrows, it is important to cultivate suitable habitats near those that undergo succession and become too overgrown to support the birds.

Dunning and Watts (1990) found evidence that a site preparation technique called drumchopping reduces the suitability of clearcuts for Bachman's sparrows. Drumchopping is used by foresters to reduce the amount of above-ground vegetation and debris before planting. In their study drumchopping resulted in low, dense shrubs. Clearcuts that were not drumchopped, on the other hand, had tall shrubs and standing dead timber that created exposed song perches.

It may be possible, though not necessarily practical, to manage habitat for Bachman's sparrows without burning. In Dunning and Watts (1990) study, an infrequently burned mature pine stand was occupied by Bachman's sparrows when

an open understory was maintained by the cutting of saplings and girdling of older deciduous trees.

Grazing, however, is probably not a useful management tool for Bachman's sparrows because it results in the loss or reduction of a thick grass cover.

### Management and Research Needs

Although there is adequate information to provide a basic understanding of the breeding biology of Bachman's sparrow, certain aspects of the species life history remain poorly understood, as follows.

- Populations of Bachman's sparrows should be continuously monitored and the relationship between the number of singing males and succession should be clearly defined. Research should identify if areas can be managed to create a mosaic of nearby sites that are continuously occupied.
- Investigations of the dispersal behavior of Bachman's sparrow might effect more informed management of its habitat. The question of how new nesting areas are discovered and how far they are from source populations could be answered by a study using individually color-banded birds.
- A better understanding of the winter habitat of Bachman's sparrow is needed. Winter observations of these birds are few and if Weston's (1968) reports of winter-use of riparian habitats are correct, their winter range may be severely threatened.
- Foresters and managers of public lands need to be informed of the detrimental effects of drumchopping clearcuts and the beneficial effects of burning.
- There is an important need to educate the public on the values of clearcutting. This timber practice is often frowned upon by the general public, in part because recent clearcuts are unsightly, and in part because erosion is enhanced by cutting on sloping ground. However, this method of timber

removal is often the most cost-efficient from the forester's standpoint. Clearcuts also provide old-field habitat for a few years following timber harvest, which is critical, since this type of habitat is no longer being provided adequately through farmland abandonment.

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# HENSLOW'S SPARROW

## *Ammodramus henslowii*

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Henslow's sparrows breed in a variety of grassland habitats with tall, dense grass and herbaceous vegetation. In the Northeast, the species uses hayfields, pastures, wet meadows, dry saltmarsh areas, and old grassy fields. Nests are typically constructed on or near to the ground, and are comprised of woven grasses. Populations have declined throughout the species' range, but remain most abundant in the western portion of the Great Lakes Plain and in Minnesota. In the Northeast, Henslow's sparrows are sparse and localized in the mid-Atlantic and southern states. Population declines have been attributed to the loss of grassland breeding habitats, either from encroaching urbanization or succession to shrublands and forests. Intensive production of row crops also reduces or eliminates the use of hay fields and grazing land. Fragmentation of grasslands to areas less than 30 ha in size may also preclude use by Henslow's sparrows. Breeding populations should be monitored annually in localized areas where the species is known to occur. Management activities that enhance grassland productivity such as mowing, burning, and grazing should be encouraged on conservation lands.

### DESCRIPTION

#### Taxonomy

Two subspecies of Henslow's sparrow are recognized by the American Ornithologists' Union: a western form, *Ammodramus* (= *Passerherbulus*) *henslowii*, and an eastern form, *A. h. susurrans* (American Ornithologists' Union 1957). A third subspecies, *A. h. houstonensis*, has been proposed by Arnold (1983), but is not yet officially recognized by the Check-list Committee of the American Ornithologists' Union.

#### Morphology and Plumages

The Henslow's sparrow is characterized by its large flat head, large gray bill, and short tail. The head, nape, and most of the central crown stripe are olive-colored, with the wings extensively dark chestnut. The breast is finely streaked. The striped, olive-colored head and reddish wings together are diagnostic. When flushed, the bird flies low and jerkily, with a twisting motion of the tail. Otherwise, it is shy and secretive, with its presence most often revealed by its song (Peterson 1980, National Geographic Society 1987).

Among adult Henslow's sparrows the sexes are alike. Body lengths range from 12.1-13.3 cm

and the average weight is 13.0 g. Pyle et al. (1987) report that cloacal protuberance (male) and brood patch (female) are reliable indicators of sex for living birds in the hand during the period from May through September.

Juveniles are clay-colored above and streaked on the head and back with black. Below the plumage is a faint yellow with tinges of buff on the chin and throat. The sides of the throat are typically unstreaked although occasional streaking may occur (Roberts 1949).

### Vocalizations

The Henslow's sparrow's song is distinctive and diagnostic--a short, quiet "see-lick," accented on the second syllable (Peterson 1980, National Geographic Society 1987). The bird sometimes sings on quiet nights, but observers may have exaggerated the frequency of this behavior by Henslow's sparrows, since many species of songbirds occasionally sing at night (pers. obs.). Given the secretive nature of this species, an ability to identify its song is essential for reliable census and survey work.

## HABITAT

### Breeding Habitat

Henslow's sparrows are found in a variety of habitats that contain tall, dense grass and herbaceous vegetation (Smith 1968). Hyde (1939) describes a variety of habitats where Henslow's sparrows commonly occur: upland weedy hayfields or pastures without shrubs, wet meadows, drier areas of saltmarshes, grassy fields, and sedgy hillsides with recently planted pine seedlings (*Pinus* spp.). Gruber (1968) found that their habitat was usually quite dense from 30-61 cm off the ground and added that the species "...has adapted to living in unmowed hayfields." In New York, Peterson (1983) found them in large, ungrazed fields, often on hilltops, with a variety of moisture regimes and no woody invasion. Henslow's sparrows are not typically associated with grazed areas (Peterson 1983, Zimmerman 1988), although they can survive quite well in

pastures that are only lightly (Skinner 1975) or moderately (Smith and Smith 1990) grazed.

Wiens (1969), in his 3-year study, found four Henslow's sparrow territories in the first and third years and none in the second year. He found that Henslow's sparrow territories had a low percent cover of forbs, dense vegetation, a high effective vegetation height, little bare ground or low vegetation, and no trees, posts, or fence lines.

In the eastern part of their range, Henslow's sparrows have been reported in "...moist upland meadows not under the plow, grown up to clumps of ferns, tall meadow rue (*Thalictrum* spp.), and scattered shrubbery" in Vermont (Kibbe and Laughlin 1985); "...in small swales, meadows or other moist grassy lands...in fields of heavy timothy or clover" in Massachusetts (Forbush 1929); and "...in grassy fields and meadows with scattered bushes and herbaceous plants, both in wet and dry situations" in New York (Bull 1974). In addition to the above types of habitats, Henslow's sparrows have been reported in wet meadows vegetated by sedges (*Carex* spp.), rushes (*Scirpus* spp.) and grasses, in fields of seedling pines, and in drier upland portions of saltmarshes in the Northeast (Craig 1979).

Along the Atlantic Coast, the birds probably nested on the edges of saltmarshes before the arrival of Europeans (Hyde 1939, Craig 1979). In West Virginia, Hall (1983) reports that Henslow's sparrows "...prefer fields with growths of orchard grass [*Dactylis glomerata*] or a rank weedy growth." Birds in Maryland have occurred chiefly in broomsedge (*Carex scoparia*) fields and weedy sedge meadows, and occasionally in hayfields (Stewart and Robbins 1958). In Kentucky, Mengel (1965) reported that Henslow's sparrows occurred around Louisville "...exclusively in or near fields largely or wholly composed of the orchard grass much raised locally as a crop, and has preferred the crop fields to patches of untended grass." Mengel noted that Henslow's sparrows in Kentucky often occurred in association with sedge wrens (*Cistothorus platensis*) and comments further that the dry, upland sites in Kentucky are in marked contrast to the marshy, meadow habitats considered typical habitat by Hyde (1939).

In the Midwest and Great Plains regions, Henslow's sparrows formerly bred in tallgrass

prairie interspersed with forbs and shrubs. Where tallgrass prairie persists, J. Zimmerman (pers. comm.) reports that Henslow's sparrows still use it for nesting habitat in Kansas and Missouri. Presently, typical breeding habitat includes neglected grassy fields, pastures and meadows with scattered shrubby vegetation, or hayfields with dense cover, usually in damp or low-lying areas (Whitney et al. 1978, Johnsgard 1979, American Ornithologists' Union 1983).

In a brome grass/alfalfa/red clover hayfield in southwestern Michigan, Henslow's sparrows bred in areas with an intermediate moisture range, a continuous cover of grasses and sedges about 0.8 m high, occasional shrubs, mostly less than 0.9 m high, and accumulated litter (no data on litter depth or percent coverage) (Robins 1971a).

Fall and Eliason (1982) located a nest in Hennepin County, Minnesota, at the top of a knoll in an old field. Timothy comprised about 80% of the biomass of the 0.5 m-high vegetation. Except for a few shrubs approximately 0.5 m-high, there was no woody vegetation within 100 m, the vegetation covered about 75% of the soil surface, and there was a complete litter layer up to 5 cm deep.

Wiens (1969) compared the vegetation structure at Henslow's sparrow nests with the vegetation in unoccupied areas in Fitchburg, Wisconsin. Occupied areas had significantly lower coverage and density of forbs, especially broad-leaved types, and the forb height was significantly greater (occupied average height = 27 cm). In occupied areas, the vertical vegetation density was higher and the litter was deeper and covered a higher percentage of the soil surface (average depth = 4.3 cm, 93% coverage).

Skinner (1975) compared the number of Henslow's sparrows in seven types of grasslands in northern and western Missouri and concluded that there was a direct relationship between vegetation height and Henslow's sparrow numbers. However, Drilling (1985) pointed out that Skinner's conclusion was not entirely supported by his data, since more birds were found in 20.3-30.4 cm high grazed grasslands than in 45.7 cm-high vegetation produced by equipment used to combine grassland. This contradiction may exist because of differences in vegetation density which he did not measure or differences in timing and intensity of

these procedures which he did not document (Drilling 1985).

### Wintering Habitat

Root (1988), in her analysis of Christmas Bird Counts from the U.S. and southern Canada, reported that the most regular sightings of Henslow's sparrows occurred on the eastern coast of Texas, along the Florida panhandle, and around Cocoa Beach, Florida. The highest abundance from these areas was from Galveston Bay, Texas, with 0.01 birds per party-hour. Since there is considerable emphasis on sighting and reporting rarities during Christmas Bird Counts, such information probably presents a reasonable picture of the very limited winter range of Henslow's sparrows, given the rarity of reports of its occurrence in winter. No detailed descriptions or studies of the habitat requirements of Henslow's sparrows on their winter range are available.

## BIOLOGY

### Migration

Henslow's sparrows begin their spring migration from the wintering grounds in the mid-Atlantic and Gulf coast states in early March (Hyde 1939). By the second or third week of April, the birds have reached Kansas, southern Michigan, West Virginia, and New Jersey (Hyde 1939, Hall 1983, Robins 1971a) and, by the middle of May, they have arrived at the northern limits of their range in New England, southern Ontario, and eastern South Dakota (Hyde 1939, Whitney et al. 1978, Johnsgard 1979).

### Reproduction

Henslow's sparrows have been reported to nest in loose "colonies" with contiguous territories (Hyde 1939, Wiens 1969, Johnsgard 1979), although Robins (1971a) found that most of the territories in his southwestern Michigan study area were separated by buffer zones where no breeding sparrows occurred. The average size of

a territory ranges from 0.3 ha in southwestern Michigan (Robins 1971a) to 0.6 ha near Madison, Wisconsin (Wiens 1969). Territory size has been reported to increase through the summer (Robins 1971a, Johnsgard 1979), although this may reflect movements of adults in response to the wanderings of recently fledged young that still require parental care. Robins (1971a) found that the average territory size was smallest and the population density highest in areas with the tallest and densest vegetation.

Conclusions regarding territory size and management strategies based on published information about territory size need to be interpreted with caution, however, since it is well known that territory size in many songbirds is closely related to the availability of food. When food is easily available, territory size tends to be smaller than when food is scarce. Likewise, although it is widely reported in the literature that Henslow's sparrows are colonial, it does not appear that Henslow's sparrows are more colonial than other sparrows. The fact that larger numbers of Henslow's sparrows tend to occur in more suitable habitat does not necessarily imply that the species is colonial in the true sense of the word as it usually is applied to herons, gulls, terns, or colonially nesting swallows. The "clumping" of Henslow's sparrows may be a secondary effect of the clumped nature of suitable habitat for this species in most situations.

No specific data are available on site fidelity but several authors have commented that local populations tend to be unstable from year to year (Hyde 1939, Wiens 1969, Robins 1971a). On the other hand, Henslow's sparrows are reported to have bred consistently in some undisturbed, protected areas, like Hayden Prairie in Iowa (Ennis 1959, Drilling 1985) and Goose Lake Prairie in Illinois (Birkenholz 1983).

Henslow's sparrows usually raise two broods of young per breeding season (Hyde 1939), and perhaps three (Robins 1971a). Nests can be either open or domed and they are located from 0-50 cm above the litter (Hyde 1939, Robins 1971a, Flanigan 1975). Those nests that are off the ground are attached to grass or forb stalks. Hyde (1939) describes a "typical" domed nest in southern Michigan as being located at the base of a clump of grass with dead grass from the clump

forming an arched roof over the nest. The single entrance is located at an oblique angle on the side of the nest. Occasionally a nest is placed in a depression in the ground (Johnsgard 1979), but most are at least 2 cm above the substrate. The nest is loosely woven with dead grass and lined with finer grasses and hair. The female does most or all of the nest-building, taking 5-6 days to complete the process.

Clutch size is from 3-5 eggs. The eggs are approximately 18.3 x 14.4 mm in size (Graber 1968) and are white with spots or blotches of brown, mostly at the larger end. First clutches are normally completed by 20-30 May in the central part of the species' range (Hyde 1939, Graber 1968). Second nests are initiated in July and August with some extending into September (Robins 1971a). Only the female incubates the eggs and broods the young. The incubation period lasts about 11 days and the young stay in the nest 9-10 days. Females make most of the feeding trips during the first 4 or 5 days of the nestling period and about 50% of the trips during the latter half (Robins 1971b).

In birds from southern Michigan, 36% of the adult diet was crickets and short-horned grasshoppers, beetles about 19%, vegetable matter another 18%, and spiders, butterfly larvae and bees the remainder (Hyde 1939). About 80% of the nestling diet in two Michigan studies consisted of grasshopper and butterfly larvae (Hyde 1939, Robins 1971b).

Of the 11 nests found in a southern Michigan brome hayfield, 6 (54.5%) produced at least one young (Robins 1971a). Only one of the 11 nests (9.1%) successfully raised all of the young. Seventeen young were produced from a total of 46 eggs (37.0%).

#### Predators

Potentially important predators of Henslow's sparrows include mammals, snakes, and birds of prey. A thirteen-lined ground squirrel (*Citellus tridecemlineatus*) was observed taking a young sparrow from its nest in southern Michigan (Robins 1971a). Hyde (1939) evicted a blue racer (*Coluber constrictor*) from a nest in Michigan. The remains of Henslow's sparrows have been found in the stomachs of northern harriers (*Circus*

*cyanus*) and sharp-shinned hawks (*Accipiter striatus*) (Hyde 1939, Gruber 1968). Skunks (*Mephitis spp.*), weasels (*Mustela spp.*), and raccoons (*Procyon spp.*) also may prey on nests, especially those occurring in isolated fragments of suitable habitat.

There have been very few reports of Henslow's sparrow nests being parasitized by the brown-headed cowbird (*Molothrus ater*) (Hyde 1939, Robins 1971a). Friedmann (1963) considered Henslow's sparrows an infrequent host for the brown-headed cowbird and reported a total of 11 instances of brood parasitism from Michigan, Illinois, Ohio, Rhode Island, and Maryland. Since nests of Henslow's sparrow are very difficult to find, the available data probably are insufficient to support any final conclusions regarding the frequency or intensity of cowbird brood parasitism and its potential effects on populations of Henslow's sparrows.

#### POPULATION DISTRIBUTION, STATUS, AND TRENDS

##### Distribution in North America

The general geographic distribution for Henslow's sparrows is described as follows (American Ornithologists' Union 1983):

*Breeds from eastern South Dakota, central Minnesota, central Wisconsin, central Michigan, southern Ontario, northern New York, southern Vermont, southern New Hampshire and northeastern Massachusetts south to central Kansas, southwestern and central Missouri, southern Illinois, northern Kentucky, central West Virginia, eastern Virginia, and east-central North Carolina; also locally in eastern Texas (Harris County). Winters in coastal states from South Carolina south to southern Florida, and west to eastern and (rarely) southern Texas, casually north to Illinois, Indiana, New England and Nova Scotia (Figure 1).*

During the breeding season, the eastern subspecies, *A. h. susurrans*, occurs

*...locally, from central New York (Jefferson County, intergrading with *A. h. henslowii*), southern Vermont (Bennington), southern New Hampshire (Wonalancet), and northeastern Massachusetts (West Newbury) south to extreme western West Virginia (Morgan and Berkeley counties), eastern Virginia (Lynchburg; Princess Anne County), and east-central North Carolina (Chapel Hill) (American Ornithologists' Union 1957).*

The winter range of the eastern subspecies is

*...chiefly along the Atlantic Coastal Plain, from South Carolina (Horry County) and Georgia (Sapelo Island, Athens) to northern and central Florida (Glades County) (American Ornithologists' Union 1957).*

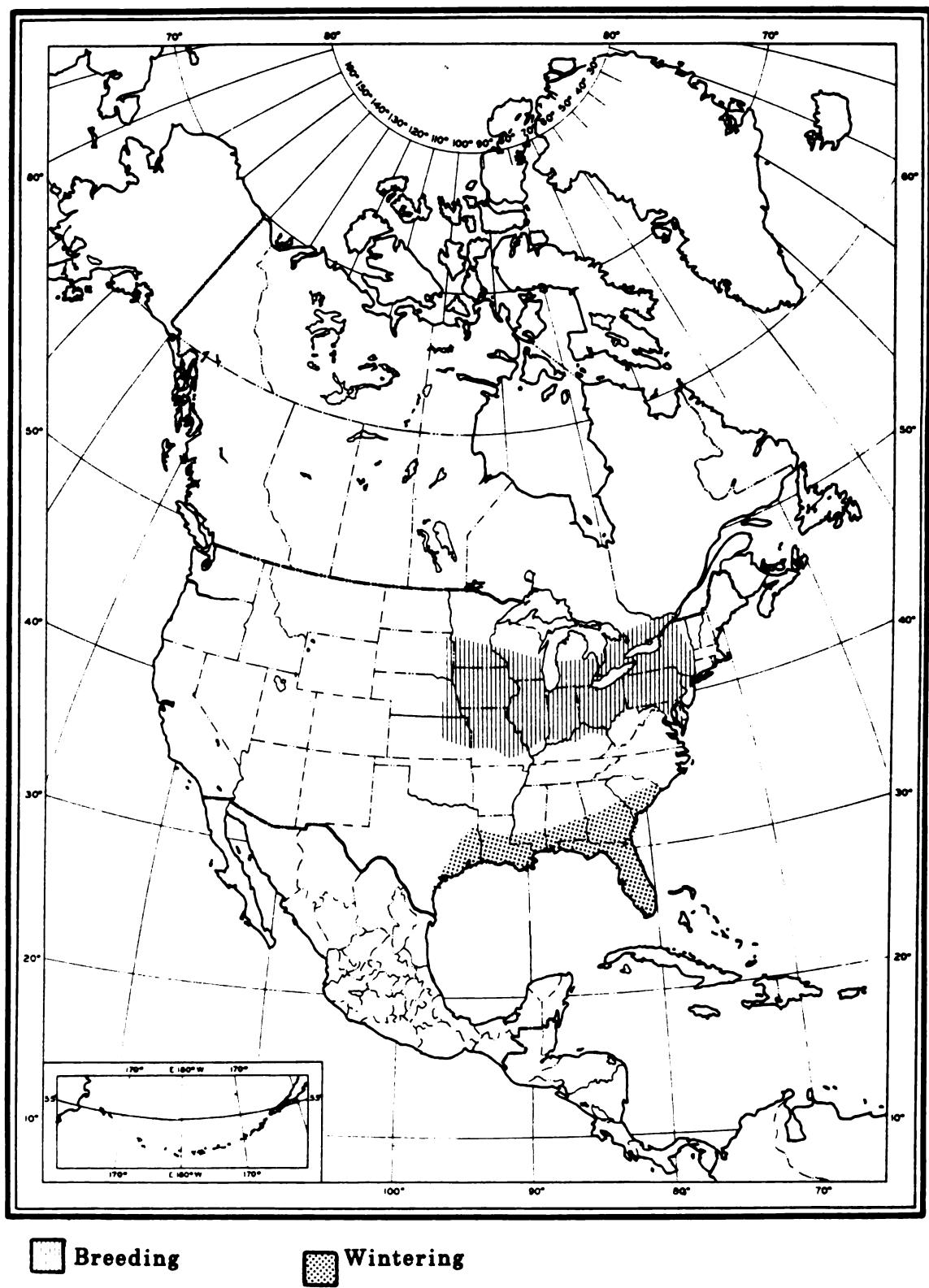
During the breeding season, the western subspecies, *A. h. henslowii*, occurs

*...locally from eastern South Dakota (Moody County), central Minnesota (Grant and Isanti counties), central Wisconsin (Dane and Oconto counties), casually north to central Michigan (Mackinaw City) and southern Ontario (Barrie; Carlsbad Springs) south to central Kansas (Cloud County), northeastern Texas, central Missouri (Hickory and St. Louis counties), southern Illinois (Richland County), northern Kentucky (Jefferson County), and central southern West Virginia (Monroe County).*

The winter range of the western subspecies is

*...from north-central Texas, Louisiana (Monroe, New Orleans), and southern Mississippi (Saucier) to western and northern Florida (Eau Gallie), southeastern Georgia (Tifton, Sapelo Island), and South Carolina (Chester, Aiken, and Charleston counties); casually in southern Illinois and southern Indiana (Jackson County) (American Ornithologists' Union 1957).*

Figure 1. The North American range of the Henslow's sparrow (*Ammodramus henslowii*) (American Ornithologists' Union 1983).



The newly described subspecies, *A. h. houstonensis*, is known only from southcentral Houston, Harris County, Texas, from a privately owned 105-ha hayfield which was under active industrial development in 1981 (Arnold 1983).

### Status in North America

The Henslow's sparrow has been identified by the National Audubon Society as either a *blue list* or a *blue list special concern* species since 1974 (Arbib 1979; Tate 1981, 1986; Tate and Tate 1982) because of apparent population declines across most of its range. In the Midwest (Iowa, Illinois, Missouri, and Indiana) populations have declined precipitously in the past 25 years. The Illinois Natural History Survey (1983) censused grassland bird populations in northern and central Illinois in 1957-58 and 1978-79. During this 20-year span, 65-75% of grassland habitat disappeared and the density of Henslow's sparrows decreased from 0.90 birds/100 acres in 1958 to 0.22 birds/100 acres in 1979. Combining both the habitat and density losses, the Survey estimates that Henslow's sparrow populations in Illinois have declined by 94% since 1958 (Drilling 1985).

In their summary of 15 years of Breeding Bird Survey (BBS) information, Robbins et al. (1986) reported that Henslow's sparrows were represented by a total of 64 birds on 29 of 1,832 BBS routes surveyed in 1977. For the period 1965-79, they report occurrences of Henslow's sparrows from only the following states and provinces (numbers in parentheses are mean number of birds per route; a (+) indicates a value of less than 0.1 birds per route): Delaware (+), Illinois (+), Indiana (0.2), Kentucky (+), Maine (+), Maryland (+), Michigan (0.6), New York (0.3), Ohio (0.3), Ontario (+), Pennsylvania (0.1), Vermont (+), Virginia (+), West Virginia (0.1), and Wisconsin (0.3). Central states and provinces reporting Henslow's sparrows on BBS routes between 1965-79 included Iowa (+), Kansas (+), Missouri (0.1), and South Dakota (+). The species was not reported from any western states or provinces. Robbins et al. (1986) conclude that Henslow's sparrows "...could not be called common anywhere" and that the greatest

densities for the species were in the western portion of the Great Lakes Plain and in Minnesota.

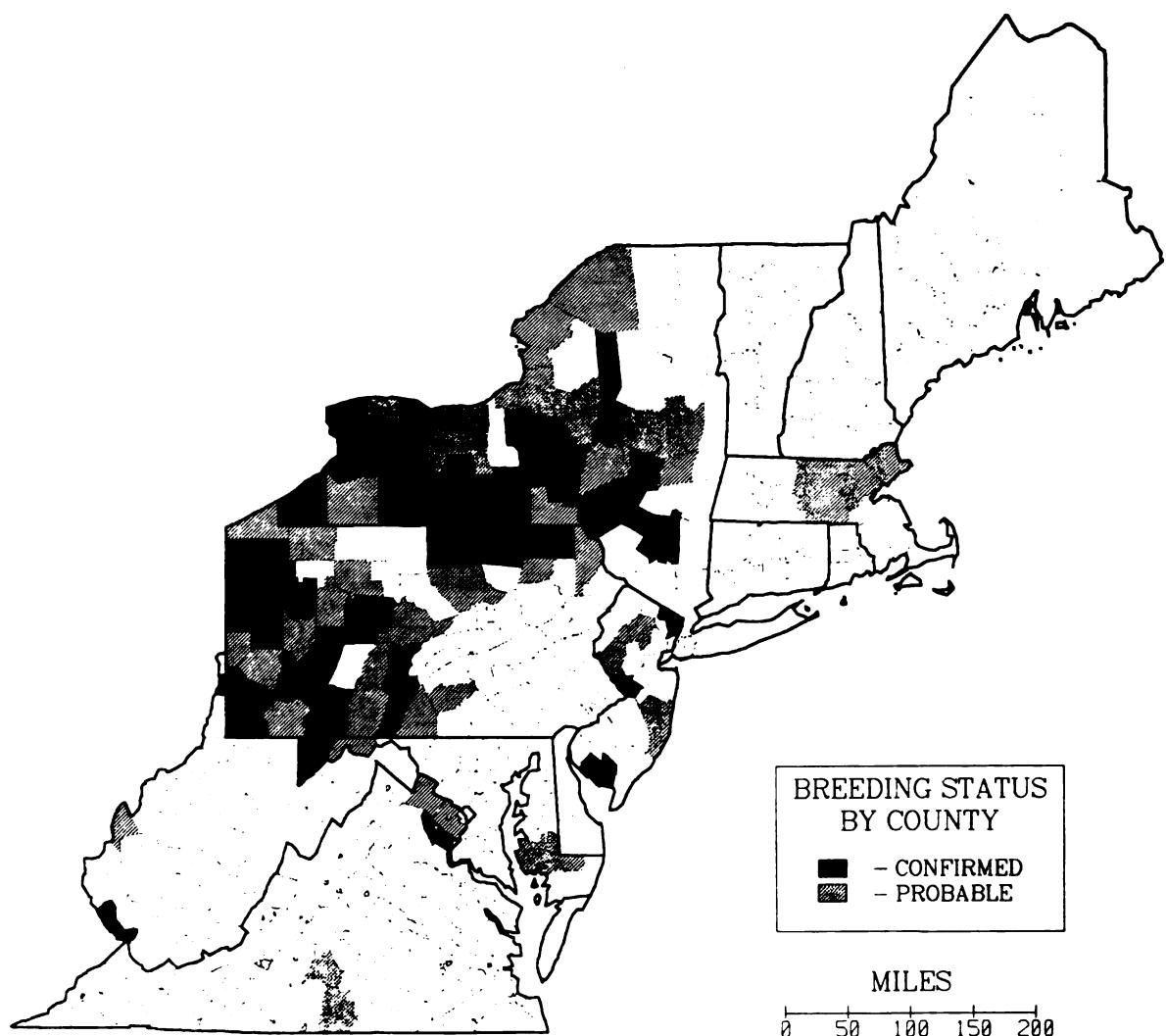
### Status in the Northeast Region

In the Northeast the species was considered locally common but declining in more urbanized areas through the early 1970s (e.g., New York: Bull 1974, 1976). In 1987, the U.S. Fish and Wildlife Service (USFWS) identified the Henslow's sparrow as a *migratory nongame bird of management concern* for Region 3 (IA, IL, IN, MI, MO, MN, OH, and WI) and Region 5 (CT, DE, MA, MD, ME, NH, NJ, NY, PA, RI, VA, VT, and WV) (U.S. Fish and Wildlife Service 1987); and more recently was petitioned to add the species as a candidate (category 2) for listing as a federally endangered or threatened species (U.S. Fish and Wildlife Service 1991). A recent summary of BBS data from 1966-90 shows a highly statistically significant ( $P < 0.01$ ) annual decline of -3.8% for Henslow's sparrows in the Northeast Region (5) of the USFWS (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). Additional analyses covering survey data from 1966-89 show similarly significant declines for the U.S. as a whole ( $P < 0.01$ ), and the East ( $P < 0.01$ ). The species is presumed to have declined significantly in these regions due to loss of suitable nesting habitat.

In the state summaries that follow an attempt is made to evaluate the status of Henslow's sparrows based on a comparison of historic accounts to four types of contemporary data. These include BBS analyses (Office of Migratory Bird Management unpubl. data, Laurel, Maryland), data from state breeding bird atlases (BBAs) (Figure 2), and state agency legal statuses and Natural Heritage Program state ranks (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia; Table 1).

The USFWS cautions against the use of state BBS trend data that are based on fewer than 10 routes. Given these constraints, only two states--New York and Pennsylvania--have adequate data to analyze population trends in Henslow's sparrows.

Figure 2. Northeast breeding distribution of the Henslow's sparrow (*Ammodramus henslowii*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



**Table 1.** Summary of state listing status and Natural Heritage Program state ranks for the Henslow's sparrow (*Ammodramus henslowii*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	-
New Hampshire	endangered	SH
Vermont	endangered	SH
Massachusetts	endangered	SH
Rhode Island	-	SX
Connecticut	threatened	SH
New York	special concern	S4
Pennsylvania	-	S3
New Jersey	endangered	S1
Delaware	-	SH
Maryland	special concern	S1/S2
Virginia	-	S1
West Virginia	special concern	S1

<sup>a</sup>SH = historical records only, but suspected extant

SX = apparently extirpated

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

Natural Heritage Programs, typically a joint effort of the Nature Conservancy and an agency of state government, assign state ranks based on the number of known occurrences of a species. They range from S1 (typically ≤ 5 occurrences) to S5 (demonstrably secure). A more complete explanation of the Heritage ranking system appears in Table 1. Unlike the legal status assigned by a state agency, these S ranks carry no added legal protection. However, the Heritage Programs do use standard ranking criteria from state to state.

**Maine.** - Historical reports of nesting in Maine do not appear to be documented (Knight 1908, Palmer 1949) and the species was not reported in the Maine BBA (Adamus 1988), although it has been reported to occur on Maine BBS routes (Robbins et al. 1986).

**New Hampshire.** - Historically, Henslow's sparrows bred in New Hampshire in small numbers but no occurrences were reported during the state's BBA (E. Hentcy, C. Foss, S. Sutcliffe pers. comm.). The species is considered historic (SH) by the New Hampshire Natural Heritage Program and listed as *endangered* by the New Hampshire Department of Fish and Game.

**Vermont.** - Henslow's sparrows were recorded only twice as a possible breeder during the Vermont BBA although they had been recorded breeding historically from at least seven locations in the state (Laughlin and Kibbe 1985). The species is listed as *endangered* by the State of Vermont and as historic (SH) by the Natural Heritage Program.

**Massachusetts.** - The Henslow's sparrow is listed as *endangered* in the State of Massachusetts. Forbush (1929) considered the bird "*a rare to common local resident*" in the state in the early part of this century. The Massachusetts BBA documented three probable breeding records between 1974 and 1979 but there have been no confirmed breeding records in the state since 1974 (J. Baird pers. comm., Komar 1983).

**Rhode Island.** - The Henslow's sparrow is considered extirpated from Rhode Island although it was apparently an uncommon but regular nester in the southern part of the state for at least 40 years. It reached its highest population in the late 1930s or early 1940s when land-use practices shifted away from agriculture and the abandoned fields that were the most widely used breeding habitat (R. Ferren pers. comm.). Henslow's sparrows were not recorded during the Rhode Island BBA and breeding was last documented in the state in 1960 (R. Enser pers. comm.).

**Connecticut.** - A rare breeder in the state early in the century, Henslow's sparrows were reported to breed regularly in the northwestern part of Connecticut. The species was not recorded during the Connecticut BBA and many considered the species extirpated from the state (Bevier in press), although the state Natural Heritage Program ranks it SH (historically known but not seen in recent years), indicating that the

species might be rediscovered. The Connecticut Department of Environmental Protection lists the species as *threatened*. Zeranski and Baptist (1990) consider the species extirpated from the state as a nesting species, but report observations of "nonbreeding singing males" in 1968 and 1985.

**New York.** - In New York (Andrie and Carroll 1988), Henslow's sparrows were reported from 348 survey blocks (7% of the total blocks in the state). New York was one of the few northeastern states with an adequate number of reports from the BBS to show a steady, dramatic and statistically significant ( $P < .01$ ) declining trend during the period from 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; Smith 1989). The Henslow's sparrow is listed as a *species of special concern* by the New York Department of Environmental Conservation, while the state Natural Heritage Program gives it a state rank of S4 (apparently secure) based on the number of records from the BBA (Figure 2).

**Pennsylvania.** - Pennsylvania BBS data from 1966-89 show a decreasing population trend among Henslow's sparrows ( $P < 0.10$ ), however, no trend is evident on analyses of data from 1980-89. The bird was recorded as a probable or confirmed breeder from 229 BBA blocks between 1984 and 1989 (Brauning in press). Formerly a legally *threatened* species, it is now protected solely by migratory bird laws by the Pennsylvania Game and Fish Commission (D. Brauning pers. comm.).

**New Jersey.** - The state of New Jersey lists Henslow's sparrows as *endangered*. During the New Jersey BBA it was confirmed from five blocks and recorded as a probable breeder from four more (D. Hughes pers. comm.).

**Delaware.** - The Henslow's sparrow was a regular but uncommon breeder in Delaware but it is now considered extirpated from the state (R. West pers. comm.). No breeding records were documented by the Delaware BBA and the last breeding occurred in 1981 (Hess et al. in press).

**Maryland and the District of Columbia.** - Maryland lists the Henslow's sparrow as a *species*

*in need of conservation*, a status that affords no added legal protection. During the state BBA, the species was confirmed from only one block but probable breeding was documented from nine more (G. Therres pers. comm.).

**Virginia.** - Currently, the Henslow's sparrow is considered a local or rare breeder throughout the Piedmont and Coastal Plain of Virginia. Historically it was apparently common from Virginia Beach south to Sandbridge (Murray 1952). It was recorded as a probable or confirmed breeder from only five blocks during the Virginia BBA (S. Ridd and R. Wadja pers. comm.).

**West Virginia.** - Hall (1983) considers the Henslow's sparrow an uncommon summer resident in West Virginia. The species is reported to have declined greatly since the 1970s, having reached its peak in the northern part of the state during the 1950s and 1960s (Hall 1983). During the West Virginia BBA it was recorded as a probable or confirmed breeder from five atlas blocks (C. Stihler and J. Buckelew, Jr. pers. comm.). The species is listed as *special concern* by the state of West Virginia, and ranked as S1 (critically imperiled) by the State Natural Heritage Program.

## LIMITING FACTORS AND THREATS

The major threat to Henslow's sparrows is probably loss of breeding habitat as agricultural grasslands are developed or abandoned and revert to shrublands and forest. Robbins et al. (1986) showed a widespread declining trend for the Henslow's sparrow in their analysis of BBS data and concluded that,

*Henslow's sparrows deserve special attention because of their preferred nesting habitat: fields that have been idle for several years and are growing up in broomsedge (Andropogon spp.), often with a sprinkling of young, woody growth. This habitat is not only transitory but it is becoming less frequent because comparatively*

*few fields are permitted to lie idle for a sufficient number of years."*

In the Midwest, a switch in agriculture methods from hay production and grazing to intensive production of specialized crops such as corn, soybeans, and alfalfa has been a major contributor to habitat loss (Illinois Natural History Survey 1983, Drilling 1985). In the eastern half of its range, increased urbanization and encroachment of woody species into breeding habitat are problems. Another threat is fragmentation of suitable habitat into small, widely scattered plots (Drilling 1985). The effects of normal annual population fluctuations can be more dramatic on smaller preserves, reducing local populations to levels where random events could lead to their extirpation. If grassland areas are small (< 30 ha) and there are no connecting corridors of suitable habitat between nesting areas or no larger "source areas" of suitable habitat nearby to replenish local losses of adults, recruitment from immigration may be insufficient to offset normal annual population fluctuations. These effects have been documented in isolated and fragmented forest habitats (Robbins 1979, Robbins et al. 1989, Askins et al. 1990) and may apply to grassland and prairie habitats as well.

Conflicts also may occur between timing of cutting of hay and nesting of Henslow's sparrows (Bollinger 1988). Highly productive hayfields may attract Henslow's sparrows and other grassland birds to establish territories and begin nesting activities early in the breeding season. When these hayfields are cut, the losses of nests, eggs, and nestlings may lead to a decline in local productivity, creating the "sink" effect described for birds in agricultural landscapes (Best 1986, Temple 1990).

#### MANAGEMENT POTENTIAL

In the Northeast, urbanization and successional change, with encroachment of woody species into suitable nesting habitats, are reducing the amount of available breeding habitat and seriously affecting Henslow's sparrow populations. It may be difficult for these populations to recover or to be maintained. In the rest of its

range, Henslow's sparrow populations probably could be maintained if efforts to preserve the largest and most viable populations and to prevent further fragmentation of suitable habitat are begun immediately. Other grassland species would benefit from aggressive acquisition and management of grassland habitats, as well (Smith and Smith 1990).

#### LAND PROTECTION AND PRESERVE DESIGN

Efforts should be made to identify and prevent fragmentation of breeding habitats. In New York, pastures consisting of at least 30 ha of grassland contained Henslow's sparrow (Smith and Smith 1990). This result corresponds to the casual observations of Zimmerman (1988) in Kansas, who recommends that management to encourage Henslow's sparrows should be carried out on plots of at least 30 ha. These values also fall within Samson's (1980) estimation of 10-100 ha as the minimum area required to support a viable breeding population of Henslow's sparrows, though Samson does not elaborate upon the basis for his conclusions. Peterson's (1983) study in Broome County, New York, found that Henslow's sparrow occurrence was related to distance from the horizon, a measure strongly correlated with grassland area. These apparent minimum area requirements are far from proven and may not hold for all regions where Henslow's sparrows occur. Older accounts reported about 12 pairs living in 4 ha of dense grass in Pymatuning Swamp in northwestern Pennsylvania and 4 pairs in a field of only 3.6 ha (Graber 1968). The indications from recent work, that size is important in habitat choice by Henslow's sparrows, may be confounded by the fact that the species is declining. During periods of decline, a species is less likely to saturate the available habitats and may occupy only the highest quality sites (O'Connor 1981), giving an inaccurate impression of the range of habitats it potentially may occupy at higher population densities.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Known populations of Henslow's sparrows need to be censused annually to monitor population changes. Nesting success data such as clutch size, number of young produced and causes of nest failure may be collected. However, extreme caution should be applied when visiting nests to assure that predators are not led to nests by following the scent trails left behind by human observers.

Populations of Henslow's sparrows on preserves should be censused every year. At the very least, one complete census during the breeding season should be conducted to collect baseline information and known populations should be checked for breeding activity in subsequent years.

The best times to census are during peak nesting periods and peak singing periods during the day. The peak singing period is 1 hour before and after sunrise (Hyde 1939, Robins 1971a). Nests can be found by dragging a very heavy rope between two people through the territory of a singing male and having one or two additional people behind the rope to find the nests. Again, caution should be applied in searching for nests and the risk of introducing additional predation should be weighed carefully against the need for information about nesting activity.

The only monitoring effort providing reliable information about the annual status of Henslow's sparrow breeding populations is the Breeding Bird Survey of the USFWS (Robbins et al. 1986). However, as pointed out earlier, in some states Henslow's sparrows have become so rare that this program does not provide adequate data for monitoring population trends at the state level. Information about status and distribution in winter can be obtained from the Christmas Bird Counts of the National Audubon Society (Root 1988).

### Management Procedures and Programs

**Grazing.** - Skinner (1975) gave no data on the intensity or frequency of grazing on his study

area, but he did find Henslow's sparrows on grazed lands with 20.3-30.4 cm-high vegetation. Few birds were found on grazed areas with shorter vegetation heights. Hyde (1939) reports one instance where a cow trampled a Henslow's sparrow nest; this form of nest destruction may be an important consideration. In New York, Smith and Smith (1990) found Henslow's sparrows using pastures that were grazed by cattle from mid-May through mid-October, and from which approximately 60% of each year's annual productivity of herbaceous biomass was removed by grazing. Pastures used by Henslow's sparrows had an average grass height of 61.3 cm. The use of pastures by Henslow's sparrows is encouraging, since the likelihood of destruction of nests, eggs, and young by mowing is greatly reduced in such areas.

**Mowing.** - Hyde (1939) and the Illinois Natural History Survey (1983) imply that Henslow's sparrows will nest in hayfields that are mowed every year. Smith (1963) notes that Henslow's sparrows nesting in Jefferson County, Pennsylvania, abandon hayfields that are mowed in mid to late June, and two Henslow's sparrow nests were destroyed in Vermont during haying operations in August (Kibbe and Laughlin 1983). In New York, mowing of a hayfield in June led to abandonment of the habitat by five territorial males. Smith and Smith (1990) found no relationships between frequency of mowing and occurrence of Henslow's sparrows, as long as mowing activities were undertaken well after the nesting season (no earlier than mid-August in upstate New York). Periodic mowing may be a viable option for maintaining grasslands suitable for use by Henslow's sparrows, provided that mowing is done well after the breeding season is concluded and young and adults have dispersed.

Zimmerman (1988) noted the apparent preference of Henslow's sparrows for areas with large amounts of standing dead vegetation and proposed two possible explanations for his observation: first, that the standing dead vegetation discouraged new growth and provided more ground area for foraging, and second, that the standing dead vegetation served to protect the nest from predation, parasitism by brown-headed cowbirds, and microclimate extremes. Studies by

Smith and Smith (1990) in New York show that Henslow's sparrows prefer high productivity areas with a higher annual growth of new vegetation, suggesting that Zimmerman's first hypothesis may not be applicable to eastern sites.

In Kansas, Zimmerman (1988) found that Henslow's sparrows did not breed in areas that were moderately grazed the preceding summer. Based on this observation, he concluded that any practice that reduced the standing dead vegetation in a field would eliminate the Henslow's sparrow. On Finger Lakes National Forest in New York, the occurrence of Henslow's sparrows in grazed pastures that were mowed the previous year does not support Zimmerman's conclusion (Smith and Smith 1990); however, the grazing intensities observed by Zimmerman may have been higher than those in New York, accounting for the absence of Henslow's sparrows from his sites. The mowing discrepancy may be explained by one of two hypotheses. First, since Zimmerman's work was done during June and the work of Smith and Smith (1990) during July and August, this later seasonal work gave the vegetation a chance to regrow and allowed the birds to move into these newly regrown areas after losing or raising their first brood. This could be the case since Robins (1971a) found that most Henslow's sparrows in Michigan raise two or three broods, defend territories for as long as 2 months, and frequently change the location of their territories during the breeding season. Alternatively, mowing during late July and August and removing cattle in mid-October, as practiced on Finger Lakes National Forest, allows time for vegetation to regrow partially before winter, possibly providing enough residual cover in spring to attract Henslow's sparrows.

**Burning.** - In Kansas, Zimmerman (1988) found that Henslow's sparrows did not breed in areas that were burned the preceding spring. There is evidence that the species prefers some litter (Wiens 1969, Zimmerman 1988), and burning reduces litter depth. If burning stimulated herbaceous growth, it could be beneficial to Henslow's sparrow populations because they appear to require tall, dense vegetation for nesting. On the other hand, spring burning could prevent or delay nesting until

herbaceous cover is reestablished (Bowles 1981, Drilling 1985).

**Removal of woody species.** - If allowed to progress to a shrubland seral stage, encroaching woody species will eventually eliminate habitat for Henslow's sparrows. From the review of the literature summarized in this report, it appears that grasslands, hayfields, and moderately grazed pastures are optimal habitats for Henslow's sparrows, and that they will tolerate some invasion by woody species. Henslow's sparrows are believed to be disappearing from New England because of conversion of old fields to forest. A reversal of this process may help the species (Drilling 1985).

**Summary.** - Henslow's sparrows should not be disturbed from mid-May through August. Management regimes that produce dense and moderately tall grassy vegetation (> 30 cm) from mid-May through mid-August should be considered. Woody species should be removed. At this time, it is not clear what management objectives should be with respect to litter depth. A minimum area of 30 ha or more of contiguous grassland habitat should be preserved for Henslow's sparrows based upon the observations of Smith and Smith (1990) and Zimmerman (1988).

Significant differences appear between nesting habitat preferences of Henslow's sparrows in moister, eastern sites when compared to more mesic, midwestern locations. These differences could correspond to differences in habitat selection by the two recognized subspecies of the Henslow's sparrow. Therefore, different management recommendations should be considered, particularly with respect to the roles played by fire and grazing, and the requirements for standing dead vegetation and litter depth in the two regions. Also, the role of litter depth in habitat selection by Henslow's sparrows has not been sufficiently investigated in eastern locations where Henslow's sparrows have been studied.

### Research and Management Needs

The following research and management needs are listed in order of decreasing priority:

- Document occurrences of Henslow's sparrows on all existing managed areas, both public and private.
- Initiate annual monitoring of all populations occurring on preserves, counting all singing males present during each breeding season.
- Identify and characterize habitats on wintering grounds in the southeastern U.S., giving special attention to occurrences on any private preserves or public lands.
- Initiate studies to determine the degree of site and mate fidelity, and annual mortality and reproductive success rates, taking special care not to introduce additional sources of predation into studied populations.
- Document the effects of preserve size and habitat fragmentation on populations.
- Initiate studies of effects of various frequencies and timings of burning, mowing, and grazing on existing, stable populations.

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# **Forests**



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# RED-SHOULDERED HAWK

*Buteo lineatus*

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**Red-shouldered hawks breed in mature hardwood forests across most of the Northeast, often in bottomlands or near other water. Limited data on population trends have led to inconclusive information on the status of this species in the region. The reversion of abandoned farmlands to forests across much of the Northeast is gradually creating more extensive habitat for red-shouldered hawks, although the silvicultural treatments of these forests largely influences use by the species. Selective cutting that creates small openings in large forest stands (minimum size 250 ha) may be the best habitat management treatment for red-shouldered hawks. More population information should be collected by using standardized point-counts with broadcast raptor vocalizations during the breeding season in forests throughout the region. A woodland hawk management program should also be organized in the Northeast.**

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## DESCRIPTION

### Taxonomy

Five subspecies of the red-shouldered hawk, *Buteo lineatus* (Gmelin) have been recognized. The nominate, *B. l. lineatus* (Gmelin), or "northern red-shouldered hawk," formerly *Falco lineatus* Gmelin, 1788, is based on the "barred-breasted buzzard" of Latham and the "red-shouldered falcon" of Pennant (American Ornithologists' Union 1983). The other four subspecies are as follows; the "Florida red-

shouldered hawk," *B. l. alleni*; the "insular red-shouldered hawk," *B. l. extimus*; the "Texas red-shouldered hawk," *B. l. texanus*; and the "red-bellied hawk," *B. l. elegans*. Palmer (1988) felt: "The red-shouldered hawk fits better morphologically in *Asturina* than in *Buteo*," and placed it in the former as *Asturina lineata*.

The red-shouldered hawk and Ridgway's hawk, *B. ridgwayi*, a resident on Hispaniola and surrounding small islands (Beata, Gonave, Isle-a-Vache, Alto Velo, Grand Cayemite and Petite Cayemite) might constitute a superspecies (American Ornithologists' Union 1983).

### Morphology and Plumages

The red-shouldered hawk is a medium-sized, long-tailed, slender buteo, larger than the broad-winged hawk, *Buteo platypterus*, and smaller than the red-tailed hawk, *Buteo jamaicensis*. Its long legs and feet are yellow, with less than half the tarsus feathered (vs. half in red-tailed hawks). Wing-tips do not reach the tail tip on perched birds. Sexes are similar, except that the female is larger, with considerable size overlap. In basic feathering, the upperparts are dark but somewhat blotchy, and the lesser upper wing coverts are rusty-reddish or rufous and form the distinctive red-shouldered hawk patch. The flight feathers are boldly barred with black and white above, not as boldly barred below, with the "window"--a white crescent-shaped panel--near the black outer primary tips. The underwing appears two-toned, with rufous coverts darker than the flight feathers. The wings are proportionately long and narrow, without the bulges of the red-tailed hawk. The leading edge is straight, while the trailing edge curves gently or not at all:

*Seen from below, the wing of a red-shouldered hawk suggests a long, rectangular plank. The entire wing juts forward when the bird is in a full soar, as if it were reaching out, arms wide, to embrace something (Dunne et al. 1989).*

The birds typically soar on wings with a slight downward droop. The underparts have transverse rusty to rufescent barring. The tail has several wide and very dark bars; the intervening narrow stripes and the tip of the tail are white. There is geographical and some sexual variation in the number of these tail bars, among both adults and juveniles (Clark and Wheeler 1987, Palmer 1988). The iris is dark-brown and the cere is bright yellow.

Young nestlings are thickly covered with long, soft, silky down, longest on the head, yellowish-white above, tinged with "vinaceous-buff" on the back and wings, and whiter below. Somewhat older nestlings are covered with short, thick, woolly down, thickest and pure white on the belly, and grayish-white above. At about 2 weeks the wing quills sprout, followed by the scapulars,

wing coverts, and contour plumage (Kennard 1894, Bent 1937).

Juvenile plumage is well developed by fledging and held through the first winter into spring when gradually molted, although it may not be completely molted until the following fall. The head is medium-brown, usually with a buffy superciliary line and dark-brown malar stripes. The iris is light to medium gray-brown and the cere is greenish-yellow. The back is dark-brown with some tawny mottling. The upperwing coverts are dark-brown with some tawny and whitish mottling and often a hint of the red shoulder. Primaries are dark-brown with a crescent-shaped tawny area on the upper surface next to the black tips. The white underparts are marked longitudinally with dark-brown blobs. The underwing is uniform white to cream and shows the distinctive crescent panel or "window" when backlit. Leg feathers and undertail coverts are white and spotted with dark-brown. The tail is brown above with many fine lighter brown bands. A few albinistic and partially albinistic birds have been recorded (Clark and Wheeler 1987).

### Vocalizations

As Bent (1937) observed: "*During the breeding season the red-shouldered are the noisiest of our hawks.*" According to Palmer (1988): "*Most common call may be written "Kee-aah," accent on first syllable, and second extended and with falling inflection--as though the bird were tired.*" Another vocalization includes a single or repeated "kip," which the male gives when fetching prey and nearing the nest, and the female responds similarly, while nestlings have a chirping call. The blue jay (*Cyanocitta cristata*) is notorious for mimicking the cry of the red-shouldered hawk. Bent (1937) and Palmer (1988) make note of a variety of other vocalizations attributed to this hawk, mostly variants of the cry or the nest call.

## HABITAT

### Breeding

The breeding habitat used by red-shouldered hawks varies from bottomland hardwoods and riparian areas (Stewart 1949, Henny et al. 1973, Bednarz and Dinsmore 1981, Kimmel and Fredrickson 1981, Woodrey 1986, Preston et al. 1989) to upland deciduous or mixed deciduous-conifer forest (Titus and Mosher 1981, Armstrong and Euler 1983, Morris and Lemon 1983, Crocoll and Parker 1989). Their nesting areas are almost always found near some form of water, such as a swamp, river or pond, and the habitat is usually well forested (Portnoy and Dodge 1979, Kimmel and Fredrickson 1981, Titus and Mosher 1981, Morris and Lemon 1983). Further, the forests surrounding the nests are characterized as having mature to overmature canopy trees and variable amounts of understory vegetation (Postupalsky 1980, Titus and Mosher 1981, Armstrong and Euler 1983, Morris and Lemon 1983, Titus 1984, Preston et al. 1989). Red-shouldered hawks typically fly below the canopy and choose sites with sparse subcanopy vegetation (Bednarz and Dinsmore 1982).

### Wintering

In winter, this species is more widely distributed, yet favors lowland areas near water, either standing or running, including "...river valleys, swamps, marshes, the Everglades, and perhaps canyon bottoms" (Palmer 1988). W. Brewster saw them often in winter in the marshes and along the rivers near Cambridge, Massachusetts, at the turn of the century: "At this season, when they are nearly or quite silent, they are given to haunting level, open country sprinkled with large, isolated trees" (Bent 1937). Bohall and Collopy (1984), conducting winter roadside censuses in Florida, found that red-shouldered hawks were most often observed in open areas such as pastures and fallow fields.

## BIOLOGY

### Reproductive Biology

Migrant red-shouldered hawks begin to reappear from their wintering grounds in mid-February in Maryland to mid-March in Massachusetts (Palmer 1988). Courtship, territory establishment, and nest building (or refurbishing) occur shortly after arrival of the species. In New York, Crocoll and Parker (1989) recorded red-shouldered hawks back on territories and relining nests during the second and third weeks of March. Portnoy and Dodge (1979) in Massachusetts observed courtship flights during March and nest relining during the last week of March and first week of April. Morris et al. (1982) in southwestern Quebec observed territorial hawks soaring from early March to mid-April.

**Nesting.** - Although a few nest a 1 year of age (Apanius 1977), most red-shouldered hawks first breed when at least 2 years old (Palmer 1988). Polyandry has been evidenced and copulation and trio bonding at the nest has been recorded (Palmer 1988).

Aerial nuptial displays are impressive and include "high-circling" and "sky-dancing," both extremely vocal performances. In the sky-dance, a red-shouldered hawk (presumably the male) rides an upward thermal, crying as it circles, then drops with folded wings into a steep dive, pulling up and then shooting upward again. Neighboring pairs will often join in, with as many as ten birds involved. The sky-dance can be immediately followed by copulation, which "occurs repeatedly and over considerable time" (Palmer 1988).

Nesting territories can be used for many years, even in the face of logging and (formerly) egg collecting. Bent (1937) reports an unbroken record of 26 years for a territory that was occupied for at least 42 years, until the woods were nearly eliminated by cutting. The longest period of occupancy Bent recorded was 47 years, but he knew of a tract that was occupied by red-shouldered hawks for over at least a half-century, from 1872-1923. Breeding densities of 1 pair/48.7 ha in Central Maryland have been reported (Stewart 1949), 1 pair/171 ha in western New

York (Crocoll and Parker 1989), and 1 pair/645 ha in Michigan (Craighead and Craighead 1956). Densities of breeding pairs have ranged in other studies from 0.17-0.22 pairs/km<sup>2</sup> in Indiana (Parker 1990); 0.10 pairs/km<sup>2</sup> in Wisconsin (J. Mosher unpubl. data, cited by Sargent 1990); 0.24 pairs/km<sup>2</sup> in Massachusetts (Portnoy and Dodge 1979); and 0.49 pairs/km<sup>2</sup> in Maryland (Henny 1972). In Indiana, densities were highest in areas of > 75% forest cover (Parker 1990). Stewart (1949) found nests a mean distance of 1,072 ± 352 m apart in the wide upper Patuxent River drainage in Maryland, while Crocoll and Parker (1989) found nests a mean distance of 1,271 ± 640 m in the Canadaway Creek Wildlife Management Area of western New York.

The nest is usually built in the main crotch of a large, living tree in mature forest, although in Florida, palmettos (*Serenoa repens*) may be used. At least 43 species of mainly deciduous trees have been chosen; consequently size and shape seem more important than the actual species (Bednarz 1979, Apfelbaum and Seelbach 1983, Titus and Mosher 1987, Palmer 1988). The bulky twig structure, rather flat on top, is typically placed approximately halfway up the tree in the lower portion of the canopy (Morris et al. 1982, Titus and Mosher 1987). The typical height is between 11-15 m but can range from 1.5-33.5 m (Peck and James 1983). The nest is lined with stems, leaves, lichen, and bark. Active nests are decorated with greenery and other materials. Hemlock and other conifer sprigs are often mentioned as nest greenery, as are deciduous sprigs once they have leafed out, and Bent (1937) mentions such plants as flowering violets (*Viola spp.*) and nightshade (*Solanum spp.*). Other materials have included cornstalks, ears, and husks, dried tent caterpillar webs, tissue paper, twine, and nests of eastern wood-peewee (*Contopus virens*), red-eyed vireo (*Vireo olivaceus*), and northern oriole (*Icterus galbula*) (Palmer 1988).

**Clutch and Incubation.** - Clutch size varies from 1-6 (Palmer 1988) with 2-4 eggs being the most common sizes in the Northeast. The eggs are smooth and slightly glossy. The base color is dull white or with a faint buff wash, overlaid with variable blotches, spots, or specks of reddish-brown or dark brown, and rarely pale-lilac, with

the larger markings concentrated toward the larger end.

Incubation is by both sexes, but mainly by the female, who is fed by the male, and commences with the laying of the first egg. Egg dates in New York State are 25 March-26 May (Bull 1974). The incubation period is around 33 days and the young hatch asynchronously and thus vary in size, as with many raptors (Newton 1979). The semi-altricial young are inactive at first, becoming active at about 10 days. Feathering begins in about 2 weeks. The nestling period lasts from 5-6 weeks (Harrison 1978, Crocoll and Parker 1989). Fledging occurs from 13-20 June in Maryland (Janik and Mosher 1982), 6 June-11 July in New York (Bull 1974, Crocoll and Parker 1989) and 25 June-5 July in Massachusetts (Portnoy and Dodge 1979).

**Nest Success and Productivity.** - Nest success (measured as the percentage of nests that fledge at least one young) has been reported to vary from 52.9% in Maryland to 100% in Missouri with an average of 68.7% over nine North American studies (see Crocoll and Parker 1989: Table 12). The average number of young fledged/nest over the same nine studies varied from 1.11 young to 2.9 young/nest. Henny et al. (1973) expressed the opinion that in a stable red-shouldered hawk population each pair should fledge an average of 1.95 young. Four of the 9 above-mentioned studies had fledging numbers below the Henny et al. (1973) standard; 3 of these are from the Northeast: New York (1.11 young fledged: Crocoll and Parker 1989), western Maryland (1.8 young fledged: Janik and Mosher 1982), and central Maryland (1.58 young fledged: Henny et al. 1973).

### Mortality

Mortality has been reported to occur during the incubation, nestling and fledgling stages of the breeding season (Craighead and Craighead 1956, Janik and Mosher 1982, Bosakowski and Speiser 1986, Crocoll and Parker 1989). Adults and juveniles have also been reported to suffer mortality (McCrary and Bloom 1984, Crocoll and Parker 1989). Mortality has taken the form of

wind-destroyed nests (Wiley 1975, Portnoy and Dodge 1979, Dijk et al. 1990); addled eggs (Janik and Mosher 1982, Crocoll and Parker 1989); starvation of nestlings (Crocoll and Parker 1989); human disturbance at or near the nest-site (Craighead and Craighead 1956, Wiley 1975); and predation of eggs, nestlings or adults (Craighead and Craighead 1956, Wiley 1975, Portnoy and Dodge 1979, Bosakowski and Speiser 1986, Crocoll and Parker 1989). The most commonly mentioned predators are raccoons (*Procyon lotor*) and great horned owls (*Bubo virginianus*).

### Food Habits

Palmer (1988) summarizes:

*Much prey consists of snakes of moderate size, amphibians up to bullfrog size, mammals mostly from shrew to chipmunk size, small lizards and young turtles, birds to grackle size and evidently not in numbers, a few small fishes, a few crayfishes, insects in considerable numbers usually of cricket and large grasshopper size, and a miscellany including centipedes, earthworms, and snails.*

Researchers in Massachusetts, northcentral New York, and southwestern Quebec Province have found that the time of hatching generally coincides with the emergence of juvenile eastern chipmunks (*Tamias striatus*) from their natal burrows, and these constitute the principal prey of red-shouldered hawks at that season (Portnoy and Dodge 1979, Morris 1980, Johnson 1989). However, Bednarz and Dinsmore (1985) in Iowa observed that the proportion of prey types changed dramatically between the 2 years of their study, with mammals dominating one year and amphibians and arthropods the other year. Apparently the change in prey type had no effect on productivity between the 2 years. Some food items of the red-shouldered hawk were too large to transport and might represent injured prey or carrion (Palmer 1988).

Hunting is done beneath the forest canopy, as well as in more open adjacent wetland areas, often by the perch-and-wait method. Hunting from flight is done by flying low, "...rather like a

*harrier [Circus cyaneus], surprising its prey at close range" (Palmer 1988).*

### Migration

The red-shouldered hawk appears to be migratory mainly in the Northeast, although there are flights in the Midwest and Southeast, and a light fall movement in California.

Spring migration is early; birds move by 15 February in Maryland and the District of Columbia (Palmer 1988). Also, in the spring of 1989, the first arrivals on the Lake Plain of New York, where red-shouldered hawks are reluctant to cross the open expanse of Lake Ontario and follow the shoreline, were at Derby Hill by 10 March and Braddock Bay, Hamburg, and Ransomville, New York on 14 March (G. Smith pers. comm.).

Fall migration begins in early September in the Northeast, extending into November and even late December for a few tardy individuals.

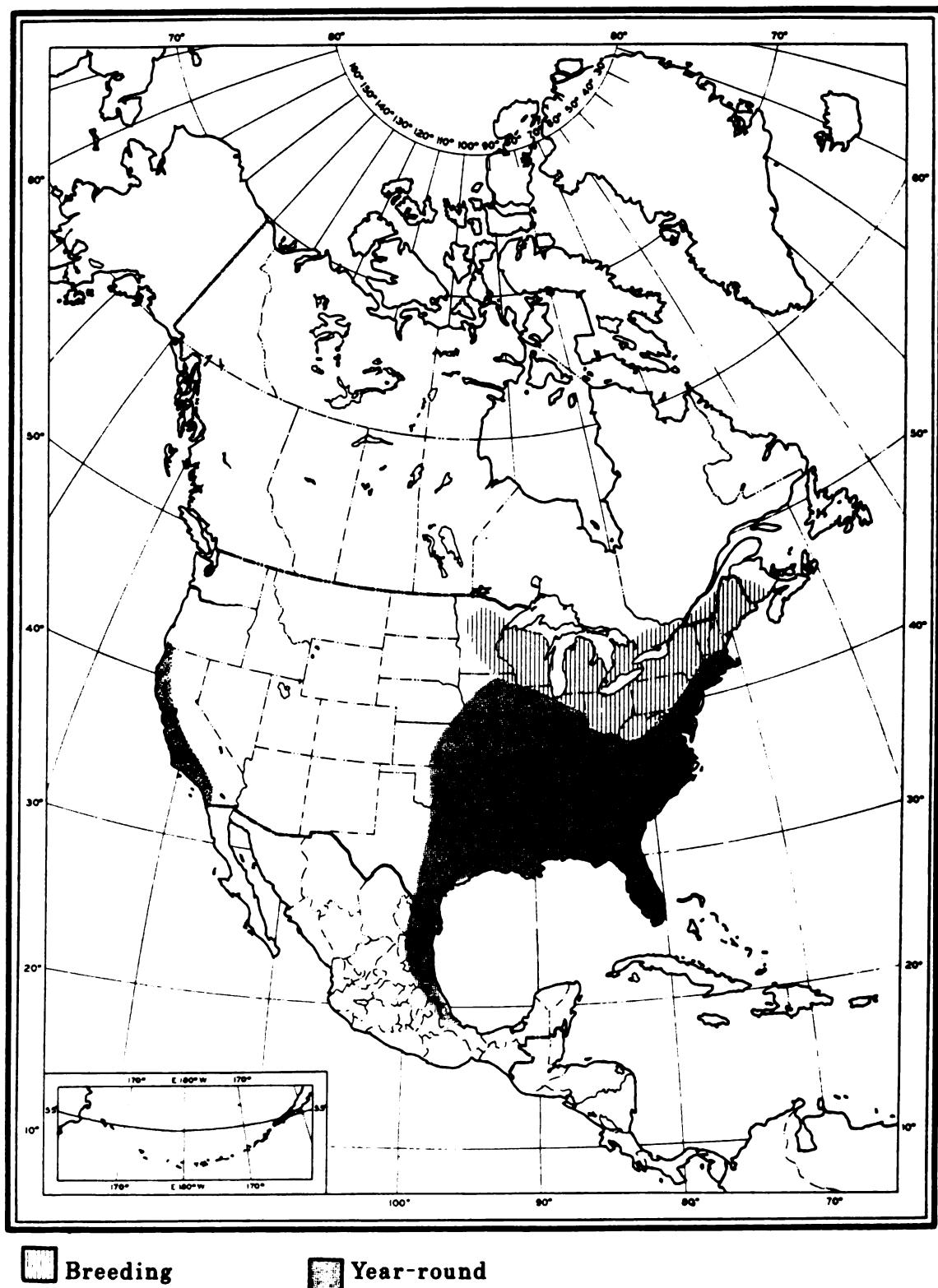
Recent literature is inconclusive on whether there has been a real decline in migrating red-shouldered hawks and further whether any migratory decline is really reflective of a declining population (Bednarz et al. 1990, Titus and Fuller 1990). Nevertheless, both Panko (1990) and Titus and Fuller (1990) have recommended several methods that can help overcome some problems of using hawk-watch count information.

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

The red-shouldered hawk breeds from northern California south, west of the Sierran Divide, to northern Baja California (Figure 1); and from eastern Nebraska, Iowa, central Minnesota, northern Wisconsin, northern Michigan, southern Ontario, southwestern Quebec and southern New Brunswick south to Veracruz, Tamaulipas, central and southern Texas, the Gulf coast and Florida (to the Florida Keys); also locally in the valley of Mexico (recorded in

Figure 1. The North American range of the red-shouldered hawk (*Buteo lineatus*) (American Ornithologists' Union 1983).



Zacatecas and Distrito Federal) (American Ornithologists's Union 1983).

This *buteo* winters, at least sporadically, throughout the breeding range, but in eastern North America primarily from eastern Kansas, central Missouri, the Ohio Valley, northwestern Pennsylvania, southern New York rarely, and southern New England southward where it is found mostly in lowland areas near water (river valleys, swamps, marshes, the Everglades, and perhaps canyon bottoms). In Maryland, they leave the mountains and remain in the Piedmont and coastal areas, while those in Southern parts are resident, with almost none absent from about the latitude of Virginia southward (Palmer 1988).

The red-shouldered hawk has appeared on the *blue list*, compiled and published by *American Birds*, from 1982 through the latest in 1986 (Tate 1986). Contributors reported the species "greatly down" in the Northeast Maritime and Ontario regions; "greatly down-down" in the Hudson-Delaware and Appalachian regions; and "down" in the Southern Atlantic Coast and Middlewestern Prairie regions. However, caution should be observed in using these results to generalize for the entire range of the red-shouldered hawk since not all areas reported a decline.

The species is of *special concern* or officially listed as *threatened* or *endangered* by 3 of the 13 Northeast state agencies (Table 1). At the national level, red-shouldered hawks are considered a *migratory nongame bird of management concern* (U.S. Fish and Wildlife Service 1987). Globally, the species is ranked G5 (globally secure) (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

A recent study (Titus et al. 1989b), however, found no overall trends in analyses of 1966-87 Breeding Bird Survey (BBS) data in either more or less developed regions of the Northeast. Negative trends were noted in less developed regions of Pennsylvania and northern New England and positive trends in more developed regions of Maryland and the northern Piedmont. More recent analyses of BBS data in the Northeast (1966-90) also indicate no significant trends (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). Titus et al. 1989b also indicated no trend in counts of red-

**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the red-shouldered hawk (*Buteo lineatus*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	watch list	S3
New Hampshire	-	S4
Vermont	-	S4
Massachusetts	watch list	S4
Rhode Island	-	S3
Connecticut	special concern	S3
New York	threatened	S4
Pennsylvania	-	S3
New Jersey	endangered <sup>b</sup>	S3
Delaware	-	S3
Maryland	-	S5
Virginia	-	S4
West Virginia	-	S5

<sup>a</sup>S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

S5 = demonstrably secure

<sup>b</sup>Breeding population only; nonbreeding population is threatened

shouldered hawks in its analyses of 1962-83 Christmas Bird Count (CBC) data.

#### Distribution and Status in the Northeast Region

**Maine.** - Shortly after the turn of the century, Knight (1908) remarked:

*The red-shouldered hawk is the most generally seen of our larger Hawks, exclusive of the Osprey [Haliaetus leucocephalus] which is of course more local...They have been found in the State from late March to late October, but there seem to be no records of their wintering though they might be expected to remain through the winter in southern Maine at least.*

At the middle of the century, Palmer (1949) observed:

*The Red-shouldered hawk has maintained its ascendancy in numbers at Umbagog and elsewhere in Maine, undoubtedly because of removal of the original forest and its replacement by mainly deciduous woodland.*

Between 1978-83, Maine's breeding bird atlas (BBA) recorded the red-shouldered hawk in a total of 71 blocks, with confirmed breeding in 20 of those, mostly located in the southern half of the state (Figure 2). Detectability of the species was ranked 3 on a scale of 0-5, and it was noted that this hawk "approaches [the] northern limit of its breeding range" in Maine (Adamus 1988).

**New Hampshire.** - The New Hampshire BBA (1980-84) reported red-shouldered hawks from all priority blocks that had some elevations under 610 m (C. Foss pers. comm.). Approximately 120 active sites were reported, with about 31 confirmed blocks statewide, concentrated mostly toward the south and east (Figure 2). This buteo occurs in New Hampshire from late March to November, and it appears to be recovering in the Northeast after declining during the 1960s and 1970s, apparently from pesticides. Its status in New Hampshire was changed in 1986 from *threatened* to a *species of special concern*. During the 1980s a nesting density of 1 pair per square mile was reported from near Kensington in the mixed forest of the coastal plain, where there were approximately 25 nests from the mid-1970s to mid-1980s. Red-shouldered hawks are reported as greatly reduced in the Squam Lake region, but a few pairs still nest around beaver ponds. The species is more common south of the White Mountains, but is documented as occurring as far north as Pittsburgh in Coos County. BBS data suggest a nonsignificant decline of 0.7% annually from 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland).

**Vermont.** - Vermont is near the northern limit of the species' breeding range, and although the winter range does reach as far north as southern New England, most of these hawks winter farther south. Norse and Fichtel (1985) note three records from December, one from January, and three from February.

During the Vermont BBA (1976-81), the red-shouldered hawk was reported from all seven physiographic regions, but was not confirmed in priority blocks in the Northeast Highlands, North Central, or East Central regions:

*Bottomland deciduous forest habitat is limited in northcentral and northeastern Vermont. Lower densities are expected toward the limits of a species' range, and the scarcity of records for these regions may simply reflect Vermont's proximity to the fringe of the red-shouldered's range (Norse and Fichtel 1985).*

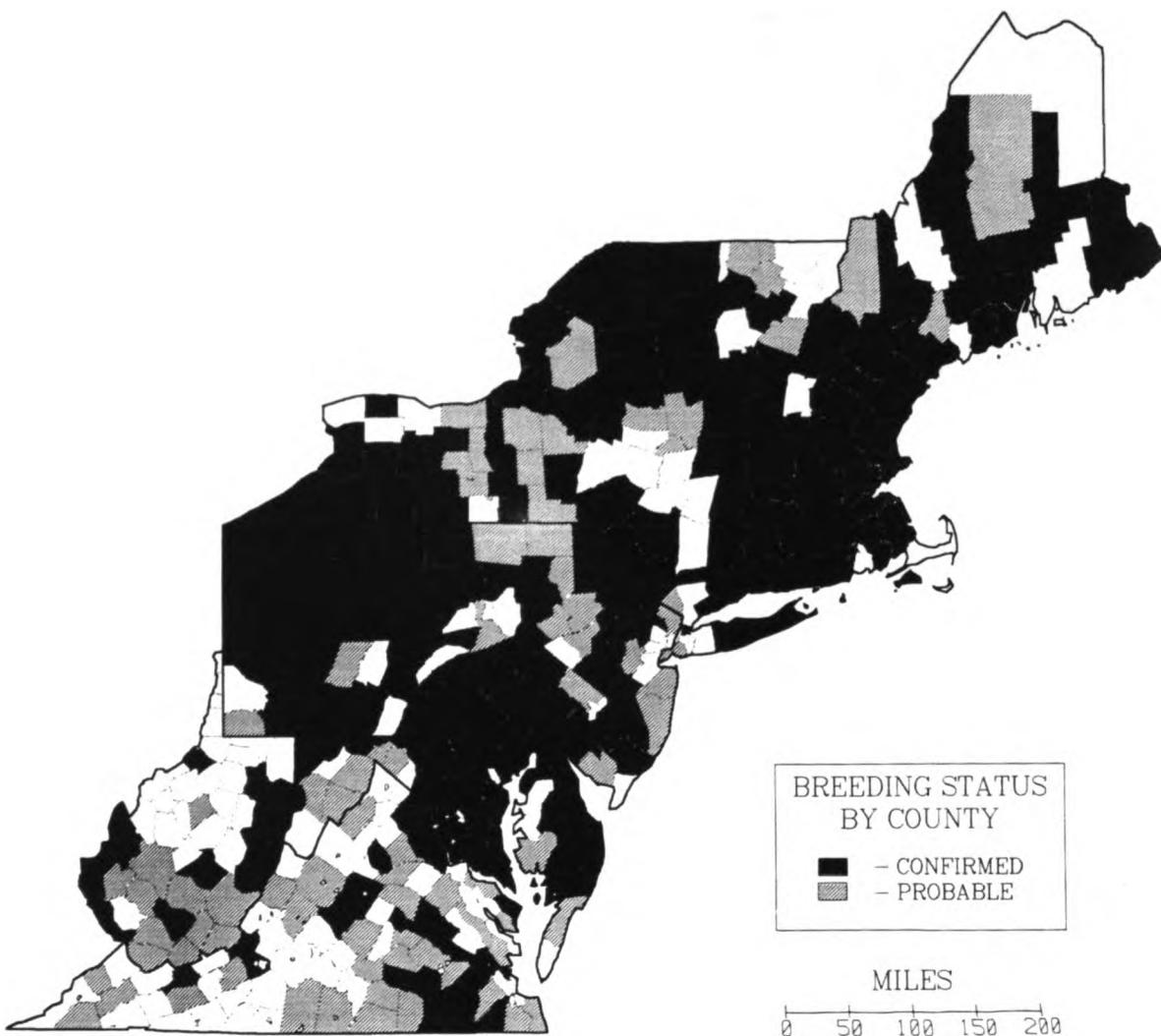
The species was confirmed as breeding in 22 Vermont blocks, and was recorded in a total of 62 "priority" blocks, or 35% of such blocks (Figure 2). Although this buteo is listed as *threatened* in New Hampshire and New York, "...it is not, however, currently being considered for such listing in Vermont" (Norse and Fichtel 1985).

**Massachusetts.** - Red-shouldered hawks are widely distributed across the Commonwealth as a nesting species, except on Cape Cod and the outer islands, and are concentrated in the more forested western half of Massachusetts and also to the southeast, south of Boston. The species was confirmed in about 67 BBA blocks (1974-79) (J. Baird pers. comm., Figure 2).

**Rhode Island.** - Although the red-shouldered hawk probably declined significantly in Rhode Island during the agricultural era of the 18th and 19th centuries, it gradually increased again with reforestation. Records were scattered during the 1950s and 1960s, but concerted efforts during the state's BBA (1982-87) uncovered higher numbers than previously suspected, primarily in the more heavily forested western sections of the state. Recorded in 26 atlas blocks, this buteo was confirmed as breeding in 5 of those (R. Enser pers. comm., Figure 2).

**Connecticut.** - Historically, the red-shouldered hawk was a common resident of Connecticut during the 19th century, but was more frequently seen in winter, according to Merriam (1877). A century ago, during the early 1890s, it nested more frequently than the red-tailed hawk.

**Figure 2.** Northeast breeding distribution of the red-shouldered hawk (*Buteo lineatus*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



Experienced observers noted a nesting decline during the late 1920s-30s, although it remained more abundant than the red-tailed hawk on the western coastal plain. A dramatic decline began in the 1940s with the drainage of wetlands, and continued into the pesticide era. By the mid-1950s it was considered rare, and was nearly extirpated as a breeding bird by the 1960s. Since the 1970s, however, the nesting population has increased (Zeranski and Baptist 1990).

Recent field work for the Connecticut BBA between 1982-86 shows its current status as "...one of only slightly less abundance" than its formerly common breeding numbers, with preliminary results showing it as a breeder in 169 blocks, and confirmed in 51 of those (Bevier et al. in press, Figure 2). The red-shouldered hawk is now widely distributed and breeds in all counties, but seems to be more common in western Connecticut. "An uncommon breeder in the state now, its status may be underestimated because of its secretive habits in breeding season" (Bever et al. in press). The species is listed as *special concern* by the Connecticut Department of Environmental Protection, based upon 51 extant (confirmed) occurrences and the declines from historical numbers.

**New York.** - Eaton (1914) considered the red-shouldered hawk to be the most common large hawk in the central and western portions of the state. Bull (1974) reported that the population had declined in recent years; in particular it was unreported in the wooded swamps of the Lake Ontario Plain.

The red-shouldered hawk was recorded in 702 (13%) of the blocks in New York during the 1980-86 BBA and confirmed in 102 blocks (Andrie and Carroll 1988, Figure 2). This hawk was sporadically distributed, with confirmed and probable breeding reported most frequently from heavily forested sectors of the Appalachian Plateau, Catskill Peaks, the Delaware, Mongaup, and Rensselaer Hills, the Tug Hill Plateau, and the Lake Champlain Valley. It was also widely scattered throughout much of the Adirondacks, particularly near river systems. There were few records of red-shouldered hawks from the lowland, lightly forested areas of the Great Lakes Plain and Long Island. In central and western

counties the red-tailed hawk is now the common large hawk. The lack of records in some areas, including the Western Adirondack Transition and St. Lawrence Transition, were thought to reflect inadequate coverage, rather than an absence of breeding red-shouldered hawks (Smith 1988). Smith (1988) also found that a comparison of atlas data and the historical record strongly suggests that the breeding distribution of the red-shouldered hawk has changed substantially in New York State:

*Whether this change reflects an overall population decline statewide or a shift of breeding areas is difficult to determine. The virtual absence of the red-shouldered hawk from red maple [Acer rubrum]-hardwood swamps along Lake Ontario and the scarcity of records in riparian situations on the Great Lakes Plain is striking. Equally striking is the large number of records from the Adirondack and Tug Hill Plateau forests. This is in sharp contrast to distribution described earlier in this century. The future of New York breeding populations of this hawk now seems closely tied to the heavily forested higher elevations of the state. Reforestation should provide additional nesting habitat for this species elsewhere as well.*

Data from 45 BBS routes show a nonsignificant statewide increase of about 0.5% annually between 1966-89 in New York.

**Pennsylvania.** - Data from the Pennsylvania BBA show that the red-shouldered hawk was recorded in a total of 745 blocks in the Commonwealth, and confirmed as breeding in 134 of those (Brauning in press, Figure 2). Breeding occurs in suitable habitat statewide, though the species appears to be declining in Pennsylvania, and "...migrations (sic) counts on east coast suggest no recovery from low pops. of early 1970s" (Pennsylvania Natural Diversity Inventory, *Vertebrate Characterization Abstract*, Middletown). Bednarz et al. (1990) also suggest a long-term nonsignificant decline in migrating red-shouldered hawks at Hawk Mountain. BBS data suggest a nonsignificant decline of about 2.0% annually.

**New Jersey.** - Formerly common, the red-shouldered hawk is reported as a breeder in moist forests in northern New Jersey, with the highest numbers in Sussex and Somerset counties (D. Hughes pers. comm.: BBA data, Figure 2). In southern New Jersey, breeding is reported in Cumberland and Cape counties and remains a possibility for the Pine Barrens. Although considered an uncommon resident and breeder, it is a common migrant, with peaks in March and late-October. Bull (1964) noted that it is most numerous in the lowlands of New Jersey southeast of the Piedmont, but usually avoids the entire coastal plain, where it is very rare.

**Delaware.** - This buteo occurs all year in Delaware although it is uncommon. The present population is "...estimated at about 50 pairs, but considerable allowance should be made for error" (Hess et al. in press). The state BBA (1983-87) found red-shouldered hawks in 39 of 122 blocks (31%), and confirmed as breeders in 10 of those. By county, they were recorded in 5 of 67 Kent County blocks (1 confirmed), 12 of 55 New Castle County blocks (3 confirmed), and 22 of 100 Sussex County blocks (6 confirmed) (Figure 2). It is unknown to what extent the breeding population migrates. "Tax ditches, channelization of streams, and lack of protection of fresh wetland and streamside forests have led to habitat loss. Mature forests, particularly in wetland areas have decreased" (Hess et al. in press).

**Maryland and the District of Columbia.** - Preliminary data from the Maryland and District of Columbia BBA show the red-shouldered hawk as confirmed in 198 blocks (G. Therres pers. comm., Figure 2). The BBS trend for Maryland shows a significant increase of 2.0% from 1966-89.

**Virginia.** - The red-shouldered hawk is a common coastal plain breeder of wet mixed forests and swamps in Virginia (S. Ridd and R. Wadja pers. comm.: BBA data). Although considered the most generally distributed breeding hawk of eastern Virginia in the 1950s, this species is declining in areas where intense development is occurring on the eastern side of the state. The species has shown a statistically significant annual

increase of 4.9% on BBS routes in Virginia from 1966-89.

**West Virginia.** - Red-shouldered hawks return to West Virginia in late February or early March, and begin to build nests as early as February. In summer, according to Hall (1983), the species is distributed throughout the state in moderate numbers:

*The red-shouldered hawk is most common in the eastern part of the state and in the mountains, but it also occurs in the lowlands. The red-shouldered hawk is almost entirely a forest bird and is seldom seen away from extensive wooded areas, although it is most commonly seen soaring over such areas. At present, the red-shouldered is probably more common than the red-tailed at most places.*

Between 1984-87, the state BBA located this hawk in 131 blocks, and confirmed breeding in 24 of those (C. Stihler and A. Buckelew pers. comm., Figure 2). It is not as common in the Northern Panhandle as elsewhere, according to Hall (1983). The species showed a statistically nonsignificant annual decline of about 0.7% on 13 BBS routes from 1966-89.

## Summary

Even though quantitative information is largely lacking, the red-shouldered hawk appears to be less abundant today than it was a century ago. Within the Northeast, there are few places where this buteo is still more common than the red-tailed hawk. Although BBS data suggest recent losses in New Hampshire and West Virginia, most of the decline in the Northeast has probably come in more agricultural and urbanized states, especially near the coast between Boston and Washington, D.C., and the Lake Ontario Plain of New York. However, there may have been concurrent gains in forested areas, such as the Adirondack Mountains, where the red-shouldered hawk was formerly thought to be scarce. If there has been an overall decline in numbers, which remains to be proven (Titus et al. 1989b), there have also been signs of a recovery in

places and perhaps even a geographical shift of local populations.

## LIMITING FACTORS AND THREATS

### Loss of Habitat

Titus et al. (1989b) suggested that research and management needs should focus on such species as the red-shouldered hawk. Yet the authors clearly stated: "We were unable to detect declines in counts of red-shouldered hawks despite the concerns of many agencies that this species is declining."

Bryant (1986) studied aerial photographs and nest records (from 1953-78) to determine the influence of selective logging on red-shouldered hawks in Ontario. Incursions by red-tailed hawks were strongly associated with reductions in mean tree density and tree-crown diameter, suggesting that selective cutting in woodlots may result in the replacement of red-shouldered hawks by red-tailed hawks. Failure to maintain uncut buffer zones around traditional red-shouldered hawk nest sites might result in the local extirpation of the species (Bryant 1986).

The closed canopy forest favored by this hawk has certainly been reduced, modified or destroyed over the years. Since the European settlement of the Northeast beginning in the 17th century, but especially since the 19th century, the forests have been cut for logging, agriculture, urban, and suburban development, and wetlands have been drained.

Habitat, though, is increasing in some areas as farms are abandoned and reverting to forest (Crocoll and Parker 1988). For example in New York in 1900, 75% of the land was opened or cleared for farming. Today, more than 61% of New York's land area is forested; the state's total forested acreage increased by 1.4 million ha (+23%) between 1950 and 1980 (Considine and Frieswyk 1982). This increase in forest land is true of other Northeastern states as well, although not all of this forested land may be appropriate habitat for red-shouldered hawks.

### Competitors

Bent (1937), Stewart (1949), and Devereux and Mosher (1984) have noted the close proximity of barred owls (*Strix varia*) to red-shouldered hawks during the breeding season and have commented that the two species' habitat requirements are similar. In fact, barred owls have been documented to use old red-shouldered hawk nests (Bent 1937, Crocoll and Parker 1986). Cooper's hawks, *Accipiter cooperii* (Peck and James 1983), and great horned owls (Bent 1938) have also been observed to use red-shouldered hawk nests.

Both broad-winged hawks (Armstrong and Euler 1983, Crocoll and Parker 1989) and particularly red-tailed hawks (Titus and Mosher 1981, Bednarz and Dinsmore 1982, Bryant 1986) are mentioned as potential competitors with red-shouldered hawks for nest sites. In Ontario, broad-wings were found to use dense, deciduous dominated, mixed forests with higher ground cover in comparison to red-shouldered hawks, which nested in mature, open deciduous forests with low ground cover (Armstrong and Euler 1983). Furthermore, broad-wings are often found near partially open forests, whereas red-shouldered hawks usually are restricted to closed forests (Titus and Mosher 1981, Armstrong and Euler 1983, Crocoll and Parker 1989).

Observations of competition for nest sites between red-shouldered and red-tailed hawks have been well documented (Bent 1937, Craighead and Craighead 1956). In contrast to red-shouldered hawks, red-tails showed a preference for forests with less canopy cover and smaller woodlot size (Bednarz and Dinsmore 1982), decreased tree densities and crown diameters (Bryant 1986), and tended to nest closer to the tops of trees with greater nest openness, particularly those on slopes (Titus and Mosher 1981, Bednarz and Dinsmore 1982).

### Diseases and Parasites

Hands et al. (1989) summarize:

*Leucocytozoa sp.*, a hematozoan, was detected in the blood of 1 or 2 red-shouldered hawks

tested in Oklahoma (Kocan et al. 1977). Two lice species *Colpocephalum flavescentes* and *Philopterus taurocephalus* and 1 bird fly species *Lynchia americana* have been found on red-shouldered hawks (Peters 1936). In New York, the ears of nestling red-shouldered hawks commonly were full of maggots *Protocalliphora splendida* (Sargent 1938). These maggot infestations seemingly did not cause deafness or hinder survival.

It should be added that tree diseases, especially of favored nesting trees (beech (*Fagus spp.*), chestnut (*Castanea spp.*), maple (*Acer spp.*), etc.) may have a locally or regionally significant impact. So, too, may the defoliation of vast expanses of Northeast forest by gypsy moths, pear thrips, or other insect species (Crocoll in press).

### Human Disturbance

Bosakowski and Smith (1989) recently found that increasing human disturbance (e.g., offroad vehicles, suburban activities, horseback riders, joggers, turkey hunters, party gangs, and unauthorized campers) is pushing this sensitive species to the deepest, wildest areas left in the Pequannock watershed of northern New Jersey. Although some red-shouldered hawks are seemingly unaffected by human presence (G. Johnson pers. comm.), most are apparently secretive and avoid areas of human use. Shooting from sites such as Hawk Mountain, Pennsylvania, used to take a major toll, but the changing legal and social climate has provided this "Chicken Hawk" (Knight 1908) far more protection than it enjoyed formerly.

### Contaminants

A number of contaminants have been found in the eggs and tissues of red-shouldered hawks: DDE, DDD, DDT, dieldrin, heptachlor epoxide, hexachlorobenzene, PCBs, mercury, chlordane, dieldrin, Furadan 10 (10% carbofuran, a carbamate), organochlorine, and polychlorinated biphenyls (Hands et al. 1989, Havera and Duzan 1986). Decreases in eggshell thickness were detected in the early 1970s, but were apparently

less severe than in accipiters (*Accipiter spp.*), falcons (*Falco spp.*), ospreys, and bald eagles (*Haliaeetus leucocephalus*) and probably had little detrimental effect on reproductive performance in the red-shouldered hawk (Henny et al. 1973).

### MANAGEMENT POTENTIAL

The potential for management of red-shouldered hawks in areas of the Northeast now devoted to agriculture or those now subject to urban/suburban sprawl is obviously low. Where suitable forest tracts remain, an effort should be made to retain sufficient habitat to support breeding pairs.

The potential for recovery of the population in forested and reforesting parts of the Northeast is good (Crocoll and Parker 1988), but will depend upon the cooperation of private landowners, as well as managers of public lands in each of the states. Even so, active management of land for red-shouldered hawks and preservation of large stands of mixed deciduous and coniferous trees are warranted to maintain viable populations of this species as well as others of this habitat type.

### LAND PROTECTION AND PRESERVE DESIGN

Much of the literature has indicated the need for large stands of forest for maintenance of breeding red-shouldered hawks. Bednarz and Dinsmore (1981) believed that red-shouldered hawks needed a minimum of 250 ha of forest area for breeding in floodplain habitats. Large blocks of both upland and wetland forest in many states, in both state and federal ownership, are most likely to provide habitat for red-shouldered hawks, away from the private lands in suburban and urban areas which are under severe development pressure. Reversion of abandoned farmland to forest offers potential future sites for reestablishment or expansion of present red-shouldered hawk populations, although to the detriment of other species of management concern such as northern harriers, loggerhead

shrikes (*Lanius ludovicianus*), and Henslow's sparrows (*Ammodramus henslowii*).

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Current monitoring programs for birds, including the BBS, CBCs, and hawk migration counts, are not suitable for detecting changes in populations of secretive, woodland hawks (Titus et al. 1989b). Many Breeding Bird Survey routes and Christmas Bird Counts record red-shouldered hawks, but too few are counted to reliably detect population trends. Likewise, fall migration counts are unable to discern the location of source populations, and only yield estimates of regional trends. To overcome these problems, specialized surveys should be used to monitor populations of red-shouldered hawks. A number of methods have been devised and described for monitoring woodland hawks, such as searching for birds and/or nests along roads and forest transects and from the air, with or without broadcasting raptor vocalizations (Fuller and Mosher 1987). More recently, Bosakowski and Smith (1989) and Johnson and Chambers (1990) have expanded upon the survey technique of broadcasting tape-recorded calls along a standardized roadside route during the breeding season. Mosher et al. (1990), in a study across five areas of the eastern U.S., collected data indicating that broadcasting conspecific raptor vocalizations, especially great horned owls, increases contact rates with resident raptors along transect routes.

The large home range sizes and sparse population densities of raptors, though, limits the number of points along a route from which a species is detected. In order to produce statistically reliable estimates of relative abundance and distribution, Iverson and Fuller (1989) developed a monitoring technique in which data are used to create an index to the proportion of an area occupied by a species based upon a correction factor called the probability of detection. This parameter was created for survey situations in which individuals are indistinguishable, distance measurements can not

be obtained, and detection rates are low. The probability of detecting red-shouldered hawks during woodland censuses ranged from 13% in Maine (Devaul 1990), 11.8-33.3% in Illinois (Malmborg and Vanderah 1991), and 30-45% in Indiana (Parker 1990). In Indiana, red-shouldered hawks had the highest rate of detection of all raptor species, and responded consistently from great distances and were easily heard. Low rates of detection (< 20%), however, require numerous replications of routes (10 or more) and yield results with large confidence intervals (Devaul 1990). Thus the technique is useful only where rates of detection are high and substantial effort can be committed to undertaking replications. Currently, no known surveys for red-shouldered hawks and other woodland hawks are regularly conducted in the Northeast. Future efforts should focus on comprehensive surveys for woodland raptors (see Devaul (1989) for further information on survey techniques).

### Management Procedures and Programs

Forest area is the primary consideration in managing for red-shouldered hawks and other woodland raptors with large breeding territories. Silvicultural practices within these areas also have considerable impact on red-shouldered hawk populations. Group selection or standard selection cutting may be the best treatment for red-shouldered hawk habitat since it creates small openings scattered throughout a canopy of large overstory hardwoods (Nelson and Titus 1989) with an approximately 70% crown closure (Bushman and Therres 1988). In fact, Bryant (1986) theorizes that managing for a crown closure of > 70% in bottomland forest should prevent red-tailed hawks from displacing red-shouldered hawks. Timber cuts in bottomlands of < 4 ha and comprising < 15% of the forest area were recommended by Bednarz and Dinsmore (1981). Timber harvest in riparian areas should be minimized (Parker 1990).

More specific habitat management information is referenced by Nelson and Titus (1989) in the *Allegheny National Forest Land and Resource Management Plan* which contains guidelines for protecting raptor nests. These

include minimizing disturbances near nest sites, reducing habitat change, and closing roads to public use during the breeding season (see Nelson and Titus (1989) for more details). These management procedures and programs would work best as part of a comprehensive plan aimed at management of forest ecosystems, and could work through a combination of public relations and education, agency rules and regulations, and environmental laws.

### **Monitoring, Management and Research Needs**

The list of needs for the red-shouldered hawk is extensive. No priority is intended in listing these needs.

- Surveys should be set up and maintained at several permanent locations in the Northeast during the breeding season using standardized point-counts with broadcast raptor vocalizations (see *Monitoring Procedures and Programs*.)
- The Northeast states should each establish a woodland hawk management program, based upon state and regional needs, to survey, and manage for red-shouldered hawks.
- Information should be provided to landowners and foresters to assure that group selection or standard selection cutting is used to maintain red-shouldered hawk habitat during silvicultural activities.
- Migratory routes and wintering areas in particular should be better identified.
- Regular monitoring of contaminant levels in red-shouldered hawks should be conducted on eggs, young, and adults. Also, the effect of these contaminants should be evaluated.
- Research should be conducted to monitor the impact of human intrusion on breeding red-shouldered hawks. Subsequently, disturbance buffers should be developed and tested.
- Specific habitat requirements should be better documented so that we can better manage for viable populations through land protection and silvicultural practices. This is particularly important in the Northeast where red-shouldered hawks are found in both bottomland, floodplain, and upland forests.
- Determine the minimum size of a forest stand necessary to maintain a breeding pair or a viable population of red-shouldered hawks.
- Monitor the impact of red-tailed hawks on red-shouldered hawks and study the effect of various silvicultural activities on adjacent pairs of each species.

- Evaluate the importance of breeding and wintering habitat to the survival of red-shouldered hawks and determine which is more critical.

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# OLIVE-SIDED FLYCATCHER

*Contopus borealis*

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Olive-sided flycatchers nest along the edges of lakes, rivers, and beaver meadows and in open forest sites that have been cleared or burned. In northern New England and the mountains of Virginia and North Carolina, the species uses coniferous sites of spruce, fir, and hemlock, mixed with deciduous trees, while in southern New England, olive-sided flycatchers nest in pine habitats. Snags are an important habitat component used for singing and feeding perches, from which they forage for insects. Breeding Bird Survey routes since 1966 have indicated a significant annual decline of 2.6% in the eastern U.S., and approaching 6.1% across its entire North American range. Reforestation and suppression of forest fires in northern New England and loss of habitat to suburban sprawl in the Northeast are possible causes for the decline of this species, although habitat disruption along migration routes and on their wintering grounds in the mountains of Central and South America could contribute significantly to population declines. Further information is needed on the status of wintering populations and habitat. Known breeding areas in the Northeast should be managed by selective patch cutting or burning and also by retaining standing dead trees.

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## DESCRIPTION

## Taxonomy

The type specimen of the olive-sided flycatcher, collected by J. Richardson near Carlton House on the banks of the Saskatchewan River, Canada, was described by W. Swainson as

*Tyrannus borealis* (Swainson and Richardson 1831, American Ornithologists' Union 1983). More recently, the species name has been changed from *Nuttallornis borealis* (American Ornithologists' Union 1957) to the present *Contopus borealis* (American Ornithologists' Union 1983).

In a recent study of evolutionary genetics of flycatchers, Zink and Johnson (1984) found that

the three species of *Contopus* studied were, indeed, closely related: olive-sided flycatcher (*C. borealis*), western wood-pewee (*C. sordidulus*), and eastern wood-pewee (*C. virens*).

### Morphology and Plumages

The olive-sided flycatcher is a rather large (18-20 cm) flycatcher, large-headed, with a proportionately short tail. Plumage is brownish-olive above (browner on juveniles) with a dull white to yellowish throat, breast, and belly. The streaked or mottled chest patches are darker. In the hand, the species can be separated from the wood-pewees by the longer (96-117 mm) wing, and from Coues' flycatcher (*Contopus pertinax*) by plumage, tail length (63-74 mm), and by the horn-colored to blackish lower mandible. Sexes are similar, but may be separable in the hand by wing chord length, 96-109 mm (females) and 103-117 mm (males) (Pyle et al. 1987).

The juvenal plumage, "...essentially like the adult in color pattern, but darker above and brighter below" (Bent 1942), is acquired by a complete postnatal molt. Banders recognize a combination juvenal/hatching year plumage from June to November with "upperparts dark brown; wing bars distinct, brownish-buffy or brownish-white; [and] flight feathers relatively fresh" (Pyle et al. 1987).

An after-hatching-year plumage between January and October consists of grayish-olive upperparts, indistinct, pale grayish-olive wing bars, and worn flight feathers. From April to August, breeding males have a cloacal protuberance, and breeding females have a brood patch (Pyle et al. 1987).

### Vocalizations

The song of the olive-sided flycatcher is a loud, ringing whistle, rather easily imitated, delivered by the male generally from the top of an upright dead snag or living tree (balsam fir (*Abies balsamea*), spruce (*Picea* spp.), tamarack (*Larix laricina*), or other pointed tree), often the tallest in the area. Most observers use the mnemonic "quick-three-beers" of a popular field guide (Peterson 1980) to recall the song. The song is often heard as a foreshortened "free beer."

The call is a trebled "pip-pip-pip" (Peterson 1980). Oologist V. Burtch told Bent (1942) that these notes "...seemed to be made mostly by the female in the vicinity of the nest and that all the nests he found were located after hearing these notes."

Another vocalization is described as being

*more like a twitter, and was uttered during excitement, chiefly when the young were learning to fly. It sounded like 'why, why, why,' repeated very rapidly a number of times. Sometimes this note was given as a prelude to the real song* (Head 1903).

Bent (1942) also describes a two-syllabled song, similar to the eastern wood-pewee's in quality: *"The notes may be whistled as 'too-wee' in ascending pitch, slightly suggesting the [rufous-sided] towhee's [Pipilo erythrophthalmus] notes."*

### HABITAT

#### Nesting Habitat

Olive-sided flycatchers breed in forest and woodland, especially in burned-over areas with standing dead trees, in taiga, subalpine coniferous forest and mixed coniferous-deciduous forest (American Ornithologists' Union 1983). In Ontario, olive-sided flycatchers nest in spruce or tamarack bogs, along the forested edges of beaver (*Castor canadensis*) ponds and rivers, and in burned-over forests (Cheskey 1987). In New York, however, where forest fires have been suppressed for much of this century, olive-sided flycatchers favor small bog ponds and quaking bogs, swampy edges of lakes, marshy streams, backwaters of rivers, and beaver meadows. Most nesting sites contain dead standing trees, which are used as singing and feeding perches, and are bordered by forest (Peterson 1988). Birds also use small mountaintop ponds. Forests surrounding these sites are usually coniferous or mixed with deciduous trees. Black spruce (*Picea mariana*) is frequently mentioned as occurring at northern sites, and red spruce (*Picea rubens*) at sites farther south, along with balsam fir,

tamarack, and eastern hemlock (*Tsuga canadensis*) (Peterson 1988).

Forbush (1927) and Griscom and Snyder (1955) mentioned that in southern New England, olive-sided flycatchers seem to prefer pitch pine (*Pinus rigida*) habitats, including pine barrens, for nesting. Nesting occurs in swamps and open woods or small clearings where fire, flooding or timber harvesting have left standing dead trees (Forbush 1927). High elevation spruce-fir forests are used in the mountains of Virginia and North Carolina (Bailey 1913, Potter et al. 1980).

### Wintering Habitat

The olive-sided flycatcher winters in the mountains of Central and South America. Wintering habitat includes a variety of forest, woodland and open situations with scattered trees, especially where tall dead snags are present (American Ornithologists' Union 1983).

## BIOLOGY

### Reproductive Biology

Courtship of the olive-sided flycatcher includes territorial fights between males, and males pursuing females across the canopies of coniferous forests. Courtship continues for at least 2 weeks until nest sites are chosen and pairing is completed (Bent 1942).

The nest is a loosely formed cup of twigs and grasses, sometimes comprised of lichens of the genus *Usnea*, and lined with finer plant materials and hair. It is placed most often in a conifer (Harrison 1978), on horizontal limbs from 4-15 m from the ground (Peck and James 1987). In Ontario, nests were found in black and white spruce (*Picea glauca*) (14 nests), jack pine (*Pinus banksiana*) (2 nests), and balsam fir (1 nest) (Peck and James 1987). Adirondack nests were built on an outer branch from 7.6-13.7 m high in balsam fir or spruce (Peterson 1988). Even though the nest is bulky, it is well concealed and rather difficult to find. Nests were found in just 21 breeding bird atlas (BBA) blocks in Ontario

(Cheskey 1987) and only 6 nests were found in New York atlas blocks (Peterson 1988).

Clutches of three (rarely four) creamy white, buff, or pink eggs are laid, lightly wreathed on the broad end with brown or gray blotches (Harrison 1978). Egg dates are 6-24 June in Ontario (Peck and James 1987), 9-27 June in New York, and 14 June in Vermont (Fichtel 1985). The species is single-brooded and has an incubation period of 14-17 days in New York. Nestlings leave the nest at 15-19 days, with New York State records of unfledged juveniles on 22 June and fledglings from 10-24 July.

The species is a rare host to the brown-headed cowbird (*Molothrus ater*), with just three records of cowbird parasitism (Friedmann 1963, Terres 1980). Although young rarely fledge before the end of July, olive-sided flycatchers can become very difficult to find as singing begins to slow during late June (Peterson 1988).

### Dispersal and Migration

Olive-sided flycatchers migrate regularly through most of the western U.S. and Middle America, less commonly in the eastern U.S., casually along the southern Atlantic coast and in peninsular Florida (American Ornithologists' Union 1983), although Duncan (1988) found it to be a rare, but regular, fall migrant in extreme northwest Florida. Possibly because of their dependence upon flying insects as prey, olive-sided flycatchers arrive rather late on their breeding grounds from South America. Spring arrivals appear during the second and third weeks of May in Vermont (Fichtel 1985) and even into June in neighboring New York (Bull 1974, Peterson 1988). Olive-sided flycatchers are early fall migrants, usually rare after mid-September in New York.

### Food Habits

Flycatchers forage primarily by hovering or sallying forth, concentrating on prey available via aerial attack. The olive-sided flycatcher generally launches these aerial attacks from a high, exposed perch atop a tree or snag. Like others in the flycatching guild, this species is a passive searcher,

looking for easy to find prey, but is also an active pursuer, attacking prey difficult to capture (Eckhardt 1979).

The olive-sided flycatcher's diet is made up almost entirely of flying insects, and it has a special fondness for wild honeybees and other Hymenoptera (Beal 1912, Forbush 1927, Bent 1942, Terres 1980). Beal (1912) examined the contents of 69 stomachs, collected in 12 states and 3 provinces, and found that Hymenoptera comprised a substantial portion of the diet each month. The fewest Hymenoptera were taken in May, when they amounted to 74.5% of the stomach contents. The average Hymenoptera consumption for the season from April-September was 82.5%. Of all birds examined during that survey, not one species subsisted exclusively on one order of insects. Winged ants were found in 10 stomachs and entirely filled 2 of these. A few parasitic species were identified. Of special interest were 63 honeybees, (*Apis mellifera*), found in 16 stomachs (25% of the entire sample). Other insects included beetles, Diptera (flies), Hemiptera (bugs), grasshoppers, moths, and dragonflies. In a study of the arthropod food of 25 species of forest-inhabiting birds in California from 1962-68, Otvos and Stark (1985) found that Coleoptera (beetles) were the most common food of the olive-sided flycatcher.

### Winter Biology

Marshall (1988) indicated that olive-sided flycatchers may display strong year-to-year site fidelity on the wintering grounds.

In a study of 16 insectivorous, aerially-foraging neotropical flycatcher species (Tyrannidae), including the olive-sided flycatcher, that are seasonally sympatric in the humid Caribbean lowlands of Costa Rica, Sherry (1984) found that migrants are more opportunistic while wintering than the syntopic year-round residents.

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

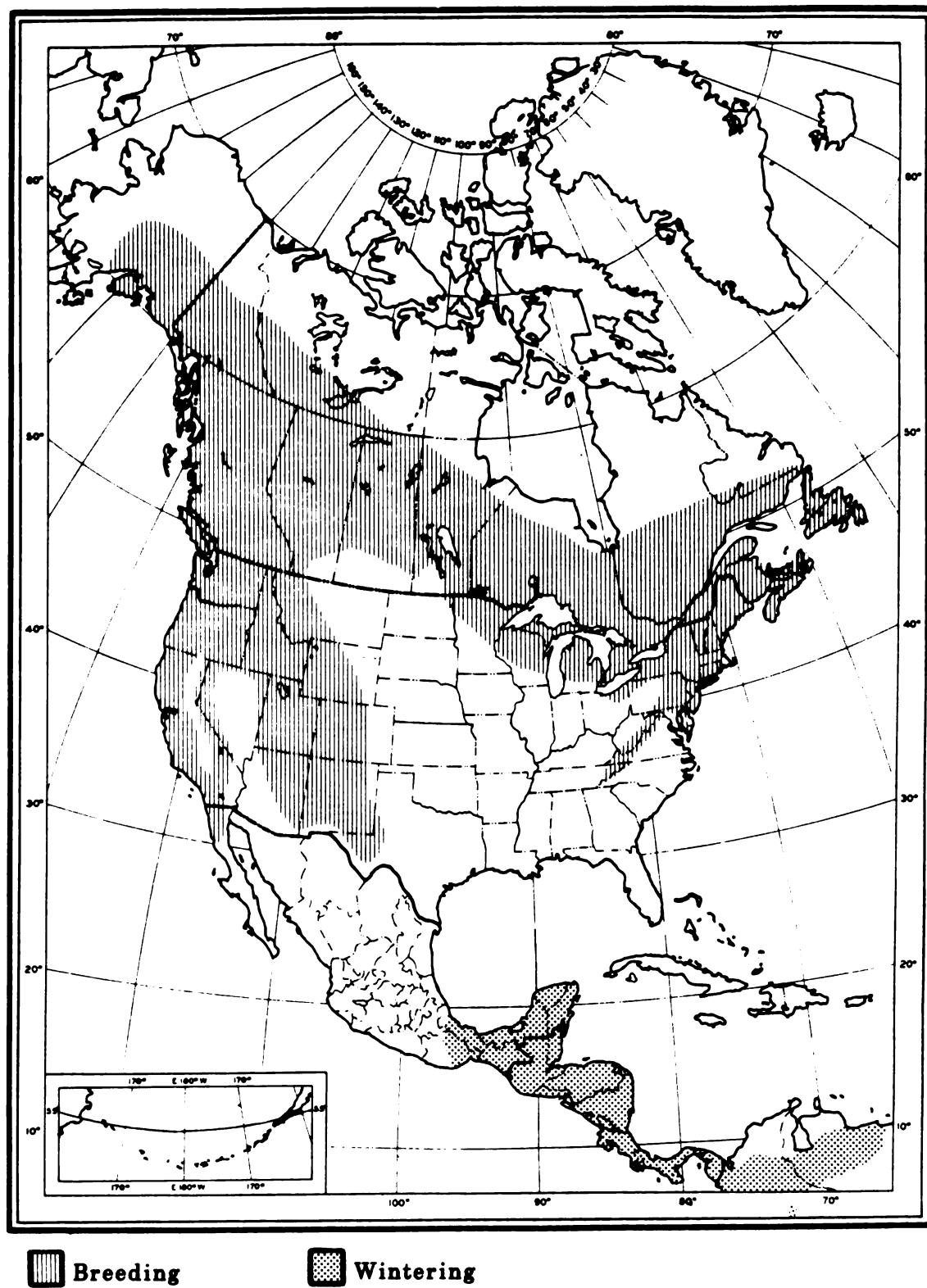
### Distribution and Status in North America

The olive-sided flycatcher breeds from western and central Alaska, central Yukon, westcentral and southern Mackenzie, northern Saskatchewan, northcentral Manitoba, northern Ontario, southcentral Quebec, southern Labrador and central Newfoundland south to northern Baja California, southern California, southern Nevada, central Arizona, southern New Mexico and western Texas, and, east of the Rocky Mountains, to central Saskatchewan, southern Manitoba, northeastern North Dakota, central Minnesota, northern Wisconsin, northern Michigan, southern Ontario, northeastern Ohio and Massachusetts, also locally in the Appalachians south through New York, Pennsylvania, eastern West Virginia and southwestern Virginia to eastern Tennessee and western North Carolina (American Ornithologists' Union 1983, Figure 1).

The species winters in the mountains of South America from Colombia and Venezuela (also Trinidad) south through Ecuador to southeastern Peru, irregularly in Middle America as far north as Oaxaca, Mexico and Belize, and casually in southern California (American Ornithologists' Union 1983). There is a recent record from Nuevo Leon, Mexico (Contreras-Balderas 1988).

Until 1986, when 20-year analyses of Breeding Bird Survey (BBS) data became available (Office of Migratory Bird Management unpubl. data, Laurel, Maryland), there appeared to be no detectable decline of this species in North America. The lack of concern expressed by experienced field observers and the regional editors of *American Birds*, may have resulted from the difficulty of detecting population trends for a species that is locally or patchily distributed even within the center of its range, and absent from vast expanses of apparently ideal habitat. Between 1966-87 a statistically significant ( $P < 0.01$ ) annual decline of 3.5% was recorded on BBS routes in North America (U.S. 3.4%,  $P < 0.01$ ; Canada 3.8%,  $P < 0.01$ ). The greatest

Figure 1. The North American range of the olive-sided flycatcher (*Contopus borealis*) (American Ornithologists' Union 1983).



annual declines were noted in the eastern (2.6%,  $P < 0.05$ ) and western regions (3.8%,  $P < 0.1$ ). Specifically in the Northeast states, a statistically significant annual decline of 3.7% ( $P < 0.01$ ) is suggested by BBS data from 1966-90.

In Ontario, Cheskey (1987) found little evidence to suggest that the distribution or abundance of the olive-sided flycatcher had changed significantly within the past century. However, rangewide BBS analyses from 1966-87 show a statistically significant annual decline of 6.1% ( $P < 0.01$ ). Abundance estimates from Ontario suggest that this species is thinly distributed throughout its range in the province, although the olive-sided flycatcher can reach densities of 100 pairs/km<sup>2</sup> in some areas (Cheskey 1987).

After assessing these BBS trends, the U.S. Fish and Wildlife Service recommended in 1987 that the olive-sided flycatcher be included in the list of *migratory nongame birds of management concern* (U.S. Fish and Wildlife Service 1987).

### Distribution and Status in the Northeast Region

The olive-sided flycatcher's breeding range in the Northeast includes Maine, New Hampshire, Vermont, New York and Massachusetts, and locally in the Appalachians south through Pennsylvania, eastern West Virginia and southwestern Virginia. Farther south, the range continues to eastern Tennessee and western North Carolina (American Ornithologists' Union 1983).

**Maine.** - Forbush (1927) described the olive-sided flycatcher as locally common during the nesting season in Maine. Palmer (1949) mentioned that the species was common as a summer resident throughout coniferous forests of Maine including the offshore islands. In the vicinity of Lake Umbagog, olive-sided flycatchers were abundant in coniferous forests and along the lake shore (Brewster 1924).

Adamus (1988) showed the olive-sided flycatcher to be a widespread breeder over much of the state, found in 240 (40%) of 706 BBA blocks, though it is locally distributed in the southeastern portion of the state (Adamus 1988,

Figure 2). BBS data show no statistically significant population trends for this state. Comparison of historical accounts and recent atlas data suggest that the olive-sided flycatcher population in Maine may be stable. The state Natural Heritage Program considers the species demonstrably secure and gives it a state rank of S5 (Table 1).

**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the olive-sided flycatcher (*Contopus borealis*) in the Northeast (The Nature Conservancy:Central Scientific Databases, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	S5
New Hampshire	-	S5
Vermont	-	S4
Massachusetts	watch list	S1
Rhode Island	-	SN
Connecticut	special concern	S2
New York	-	S5
Pennsylvania	-	SN
New Jersey	-	SN
Delaware	-	SN
Maryland	endangered	SH
Virginia	-	SH
West Virginia	-	S1

<sup>a</sup>S1 = critically imperiled;  $\leq 5$  occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S4 = apparently secure

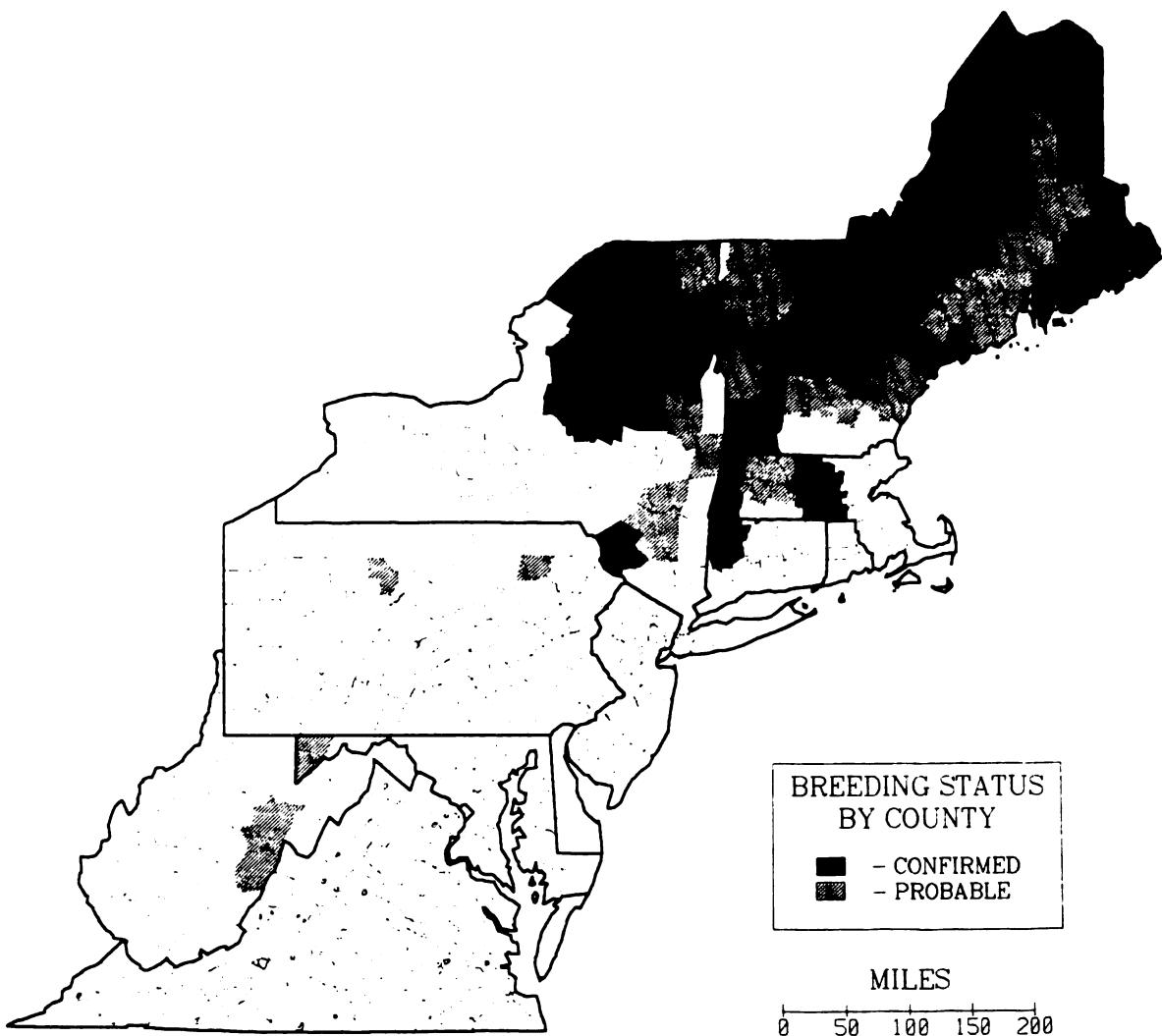
S5 = demonstrably secure

SN = regularly occurring but not breeding

SH = historical records only, but suspected extant

**New Hampshire.** - Data from the New Hampshire BBA reveal the olive-sided flycatcher to be widespread in the state, although absent or sparse in the more populous, less forested southern parts, as might be expected (C. Foss and S. Sutcliffe pers. comm., Figure 2). Breeding was confirmed in 14 atlas blocks. In New Hampshire olive-sided flycatchers are found to nest at elevations of 457-615 m, but historically they may

**Figure 2.** Northeast breeding distribution of the olive-sided flycatcher (*Contopus borealis*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



also have bred in the Coastal Lowlands (Forbush 1927). Fewer clearcuts and the loss of farms are suspected reasons for the decline of this species in New Hampshire (Hentcy 1990). It is considered secure (S5) by the state Natural Heritage Program (Table 1). While BBS data show no statistically significant trends between 1966-89, or 1980-89, non-significant declining trends are apparent during both time periods.

**Vermont.** - The Vermont BBA demonstrated that olive-sided flycatchers are not abundant in Vermont, but do occur regularly during the breeding season in suitable habitats. The species was recorded in 86 (48%) of the priority blocks in the state. Its statewide distribution closely parallels the distribution of coniferous wetlands and intermediate to high elevation forests such as those found in northeastern Vermont and the Green Mountains (Fichtel 1985, Figure 2). Summer habitats include mountain ponds (Ross 1906), high elevation wetlands (Davenport 1907), slashed and burned forests in the mountains (Eaton and Curry 1926), coniferous forests and bogs at elevations between 457-762 m (Spear 1976), and boreal forest in the Northeast Highlands (Vermont Institute of Natural Science 1975).

Historical reports indicate that the olive-sided flycatcher may have been a common breeder in the Green Mountains (Perkins and Howe 1901, Ross 1906, Allen 1909). There are no indications of historic abundance in the boreal Northeastern Highlands region of Vermont, although many of the atlas breeding confirmations were from that region.

Fichtel (1985) concluded that logging activity had probably increased the amount of habitat for olive-sided flycatchers by creating openings in forested areas. In Vermont the bird has a state Natural Heritage rank of S4 indicating that it is apparently secure (Table 1). BBS data suggest a population decline of 3.3%/year from 1966-89, and a statistically significant annual decline of 18% from 1980-89, although sample size was marginal ( $N = 9$ ).

**Massachusetts.** - Historically, the olive-sided flycatcher nested locally in eastern and southeastern Massachusetts, including Cape Cod,

but it has disappeared as a breeder from the eastern section of the state (Griscom and Snyder 1955, J. Baird pers. comm.: BBA data, Figure 2). Prior to 1900 it was apparently common in the eastern Massachusetts pine barrens. Forbush (1927) described the olive-sided flycatcher as a rather common summer resident in western Massachusetts, but rare and local elsewhere in the state.

Despite the lack of meaningful BBS trends due to small sample sizes, it is clear that the olive-sided flycatcher has declined in range and abundance in Massachusetts since the early part of this century. The Massachusetts BBA collected 17 records, 4 of which were confirmed nesting in northwestern and northcentral Massachusetts. The state Natural Heritage Program ranks the species S1, severely imperiled (Table 1).

**Rhode Island.** - No breeding records of the olive-sided flycatcher are known from Rhode Island. The olive-sided flycatcher was unrecorded during the Rhode Island BBA (R. Enser pers. comm.), not unexpectedly, perhaps, given the distance of this state from the nearest breeding populations in adjacent Connecticut and Massachusetts. The species is considered a rare migrant in Rhode Island (Howe and Sturtevant 1899, Forbush 1927).

**Connecticut.** - Historically, the olive-sided flycatcher nested at Stamford in 1897 and Danbury in 1903. Sometime before 1920 the species reportedly disappeared as a breeder in southern New England (Zeranski and Baptist 1990). Since the late 1960s, however, it has been a rare nester in the northwestern hills of the state, with five records, including one confirmed nesting at Norfolk and a suspected nesting at Barkhamsted, during the Connecticut BBA (Bevier in press, Figure 2).

The species was apparently not recorded on Breeding Bird Survey routes between 1966-87. The olive-sided flycatcher has a state Natural Heritage rank of S2 (Table 1) and is listed as *special concern* by the Connecticut Department of Environmental Protection.

**New York.** - The olive-sided flycatcher is a fairly common breeder in the Adirondack region,

although its distribution is often thin or quite local. In the Catskill Peaks and in the Central Tug Hill regions this is a rather uncommon, local flycatcher. It is rare elsewhere in the state as a nester. Except for a small, isolated population found in the Rensselaer Hills during the New York State BBA, no significant changes in distribution have been recorded over the past century (Peterson 1988). The atlas map for the state (Figure 2) is strikingly similar to that presented by Bull (1974), except for additional records in Fulton and Rensselaer counties.

In the Adirondacks, atlas observers reported olive-sided flycatchers at elevations between 168-842 m, lower than the sites mentioned by Eaton (1914) for the Adirondacks 70 years earlier. Early in this century he found that the burned lands of the Adirondacks, slashings, and the borders of flowed lands were major habitats. Such areas of human disturbance are now far less frequent, and many observers in the 1980s found that beavers, largely extirpated from New York in the last century, were responsible for a large proportion of olive-sided flycatcher breeding sites. The cool, deep ravines of the Catskills described by Hough (Bull 1974) seem much less favored, with most Catskill records above 457 m. In Rensselaer County the species was found mainly in bogs at higher elevations and in extensive swamps with dead trees (Peterson 1988). This flycatcher was recorded in 479 (9%) of the atlas blocks in the state, and confirmed in 33 of those. The state Natural Heritage Program considers the olive-sided flycatcher secure in the state and ranks it S5 (Table 1). However, data from BBS routes indicate a significant annual decline of 5.1% from 1966-89.

**Pennsylvania.** - Historically, the olive-sided flycatcher was described as generally rare and local as a summer resident in the northern counties, known to breed in Warren County and at DuBois, and seen often in Pymatuning Swamp (Todd 1940). Warren (1890) mentioned breeding records from Lancaster County (southeastern Pennsylvania) and Lycoming County. Todd (1940) speculated that olive-sided flycatchers bred in the mountains in Cambria and Somerset counties. Wood (1973) believed the species may

have nested occasionally in the northern portion of the state.

Data from the Pennsylvania BBA show six records, two probable and four possible nestings, but none confirmed. Possible breeding in the northeastern part of the commonwealth in Pike and Monroe counties has been suggested (Pennsylvania Natural Diversity Inventory: *Vertebrate Characterization Abstract*, Middletown, Pennsylvania), but there are no verified records in recent years. The state Natural Heritage rank is SN (regularly occurring but not breeding, Table 1) and the sample size from BBS routes is too small to provide meaningful information on trends in the state.

**New Jersey.** - Occasional summering olive-sided flycatchers have been reported from the northern portions of the state, but no verified breeding records have been reported (Leck 1984). The bird has never been reported on BBS routes for the state and it carries a state Natural Heritage rank of SN (regularly occurring but not breeding, Table 1).

**Delaware.** - The olive-sided flycatcher is not known to breed in Delaware. There are 20 records of migrants of this species from the 1970s, and 3 records during the 1980s, the last in 1983 (Hess et al. in press). There are no BBS records and it has a state Natural Heritage rank of SN (regularly occurring but not breeding, Table 1).

**Maryland and the District of Columbia.** - The olive-sided flycatcher is not known to have bred in the District of Columbia, but it nested historically at Cranesville Swamp in Garrett County, western Maryland, until about 1937 (Stewart and Robbins 1958). During the Maryland BBA there were no confirmed nesting records of the olive-sided flycatcher (G. Therres pers. comm.). In Maryland the species has a state Natural Heritage rank of SH (historical records only, Table 1).

**Virginia.** - Bailey (1913) mentioned that the olive-sided flycatcher was a rare breeding bird in the higher mountains of Virginia, and even then it was locally distributed. The Virginia BBA did not confirm breeding in the state. The species has a state Natural Heritage rank of SH (Table 1)

and was not recorded on BBS routes between 1966-89.

**West Virginia.** - The olive-sided flycatcher is a species of special concern in West Virginia, and it has a state Natural Heritage rank of S1 (imperiled, Table 1). Today this bird is a casual to rare local summer resident of West Virginia, but it was formerly more common.

At one time, the olive-sided flycatcher was widely distributed in some numbers in the mountain counties, with breeding records for Preston, Tucker, Grant, Randolph, Pocahontas, Pendleton, and Webster counties. Hall (1983) observed that since 1950 the olive-sided flycatcher has declined in West Virginia and was only rarely reported as a breeder. There are recent summer records from Canaan Mountain, Tucker County; Gaudineer Knob, Pocahontas/Randolph counties; and Cranberry Glades. The reasons for the decline are unclear (Hall 1983).

The preferred habitat of the olive-sided flycatcher in West Virginia is an opening in the spruce forest that contains a few trees for perches (Hall 1983).

Data from the West Virginia BBA show just two probable breeding records, of a pair seen and a bird holding territory in the mountainous eastern portion of the state (C. Stihler and A. Bucklew, Jr., Figure 2). There are no BBS records from West Virginia.

### Summary

The olive-sided flycatcher is still fairly common as a breeding species in northern New England (Maine, New Hampshire and Vermont) and the higher mountains of New York, although BBS data (1966-89) indicate that significant population declines are under way in New York, Vermont and New Hampshire. In southern New England it breeds sparingly, and in states to the south this bird is a rare breeder at best. In southern New England and the middle Atlantic states, the olive-sided flycatcher is noticeably less common than in historic times. In Massachusetts the species is now virtually restricted to the mountainous western portion of the state. In Pennsylvania there are no recent confirmations of

nesting, although historic records indicate that the olive-sided flycatcher may have nested across the northern tier of counties. In West Virginia there were only two probable nesting occurrences in recent years, while historically the olive-sided flycatcher was widely distributed in the eastern mountains of the state. The olive-sided flycatcher has apparently disappeared as a breeder in Maryland and Virginia where it was always rare.

## LIMITING FACTORS AND THREATS

### Nesting Habitat Availability

Prior to European settlement of North America, when much of the Northeast was forested, the olive-sided flycatcher was undoubtedly an opportunistic species that took advantage of openings with dead standing trees, much as it does today. These areas may have been found near water (mountain tarns, backwaters of lakes and rivers, beaver flows), burns (both natural and those set by Native Americans), and blowdowns. Nothing is known of its abundance or density then, but it may well have occupied a much greater area of the Northeast.

What happened to this flycatcher and other species after the Northeast was cleared by European settlers in the 17th and 18th centuries is unclear. A suggestion may be found in the opening of the Adirondack Mountains of New York, the last great wilderness of the Northeast, during the 19th century. There, by the early years of the 20th century, Eaton (1914) mentioned finding the olive-sided flycatcher in habitats that were largely the result of massive human disturbance: burned lands, slashings, and flowed lands. When the Adirondack Forest Preserve was created in 1885, vast tracts of the Adirondacks were waste lands, left barren of trees by lumbering and extensive forest fires.

Such areas of human disturbance are now far less frequent, and much of the area that was clearcut, burned and flooded is now covered with dense, maturing second-growth. Forest fires are vigorously suppressed, logging is more selective in many areas, and with the abandonment of hillside

farms many rural areas of the Northeast are becoming more forested than at any time in more than a century. Indeed, Hentcy (1990) points to these changes as a cause of habitat decline for the olive-sided flycatcher in New Hampshire.

Meanwhile, the beaver was largely extirpated from the Northeast by the end of the 19th century. By 1894, for example, the Forest Commission in New York estimated that only 10 were left in the entire state, including the Adirondacks. Donaldson (1921) documented the subsequent protection and restocking efforts in the early years of this century. By the 1980s, atlas observers found that beavers were largely responsible for many olive-sided flycatcher breeding sites (Peterson 1988).

During the same decade, Fichtel (1985) felt that continued logging in Vermont probably created habitat for this species by creating openings in the forest. Hall (1983) observed that recently lumbered and burned areas supported olive-sided flycatchers in West Virginia.

Thus, despite enormous habitat changes in the Northeast, this opportunistic flycatcher seems to have adapted and has found suitable niches for breeding. However, it may be under dual pressures on the Northeast breeding grounds. Around metropolitan areas it is displaced by urban sprawl, forest fragmentation, and loss of wetlands. Elsewhere it must cope with the regrowth of abandoned farms, selective logging practices, and suppression of forest fires.

Fire suppression throughout the olive-sided breeding range undoubtedly limits the acreage of available habitat. This factor is probably more limiting in the northeastern states where extensive forest fires rarely occur as compared to the western and Great Lakes states. In Minnesota, Apfelbaum and Haney (1981) found that olive-sided flycatchers had established breeding territories following a wildfire in northern Minnesota in an area which had not supported the species prior to the fire.

LeGrand and Hall (1989) suggest that a likely threat to olive-sided flycatcher habitat in the southern Appalachians is acid precipitation and insect damage. A forest dominated by dead trees would not support these flycatchers.

### Wintering Habitat Availability

As a neotropical migrant that may spend only 3-4 months of the year on its North American breeding grounds, the olive-sided flycatcher is also at risk from deforestation on its wintering grounds in Central and South America. Forest land in Middle America is being cleared at such a rate that it may be gone by the end of this decade (Terborgh 1980).

In California, Marshall (1988) found that some forest bird species breeding on Redwood Mountain in Tulare County in the 1930s were no longer present in the 1980s. Although portions of virgin sequoia (*Sequoia sempervirens*) forest within Kings Canyon National Park remained, the olive-sided flycatcher had disappeared. Marshall (1988) speculated that the disappearance of the olive-sided flycatcher from suitable, unchanged habitat was caused by the destruction of corresponding forests in Central America, where these birds maintain their winter territories.

### Diseases, Parasites, and Predation

As noted previously, the olive-sided flycatcher is only rarely a host to the brown-headed cowbird. The flycatcher presents a fairly obvious target for accipiters, but should be relatively secure from mammalian predation. Diseases and parasites of this species are unreported.

### Other Factors

Pesticide applications to control blackflies, mosquitos, or injurious forest insects could have a severe local impact upon the prey base of this flycatcher, both in North America and on its wintering grounds, but this hypothesis lacks documentation.

### MANAGEMENT POTENTIAL

The potential for restoration of peripheral populations of the olive-sided flycatcher in urban, coastal and southern portions of its range is unknown, but may be worthy of investigation.

Restoration efforts might include habitat manipulation, such as selective use of small patch clearcuts or wildfire, beaver restoration or flooding or girdling of trees where there were once historic populations or where vestigial populations remain. Retention of snags should be a consideration during timber harvesting operations in areas known to support olive-sided flycatchers.

Recovery of beaver populations in the Northeast has provided breeding habitat for olive-sided flycatchers via the creation of forest openings and greater amounts of pond shore habitat with tall, standing dead trees. Provided that beaver populations do not plummet as they once did, these habitats should persist.

The potential for recovery of the population in the Northeast, assuming that declines are real, is less certain, since the real problem of breeding numbers may lie in the deforestation of the American tropics. Without addressing the problems on the wintering grounds, any recovery efforts directed solely at the breeding grounds may be too narrow to succeed.

## LAND PROTECTION AND PRESERVE DESIGN

Large blocks of montane spruce-fir or lowland boreal forest habitat will be necessary to ensure breeding habitat for this species. LeGrand and Hall (1989) state that at least 20 ha may be necessary to sustain a single territorial pair. Natural processes (e.g., fire and creation of beaver impoundments) and management (e.g., small patch clearcuts) are important for maintenance of proper breeding habitat for olive-sided flycatchers. Public ownership of these lands, or conservation easements or management agreements to protect suitable forest habitats will be necessary.

## MANAGEMENT AND RESEARCH

### Monitoring Needs and Programs

The U.S. Fish and Wildlife Service Breeding Bird Survey is apparently the only ongoing

monitoring program that begins to adequately address the monitoring needs for the olive-sided flycatcher. In some states olive-sided flycatchers occur on too few BBS routes to allow a meaningful assessment of state population trends. Established BBS routes not presently being run within the range of the olive-sided flycatcher should be reactivated to ensure continuity in collection of population trend data.

A study might also be made of olive-sided flycatchers on Breeding Bird Census plots where the bird is known to occur. Such an effort might allow a better understanding of any changes detected, much as Hall (1984) did with other neotropical migrants in West Virginia.

### Management Procedures and Programs

There are currently no active management programs for this species. Federal and state forest management programs, both on public and private lands, should be encouraged to incorporate management practices such as patch clearcutting and snag retention, which will benefit the olive-sided flycatcher in regions where the species currently breeds.

### Research Needs

Even in Canada, where the olive-sided flycatcher is considered widespread or fairly common, its distribution can be quite thin or widely scattered. The reason for the species' sparse occurrence in the far north is unknown, but lack of preferred prey species, climate, or ancestral ranges may be factors. Whatever the cause, the limiting factor for the presence of olive-sided flycatchers in the far north does not appear to be lack, loss, or destruction of habitat, although hydroelectric projects pose a possible danger. A better understanding of both the ultimate and proximate factors affecting habitat selection is needed.

Site specific studies involving banding/color banding of individual birds are needed to gain information on movements, longevity, causes of mortality, and other aspects of the natural history of this species.

A standardized wintering bird census network, which will cover a variety of habitats, elevations and disturbance regimes throughout Central and South America and the Caribbean, is urgently needed. Although the olive-sided flycatcher is known to winter over a wide area, census work could clarify whether the majority of the species' wintering population is concentrated in a particular region. Banding studies are necessary to reveal where specific breeding populations winter.

Studies of the winter habitat preferences of the olive-sided flycatcher should be undertaken to identify what management approaches are necessary to sustain the species on its wintering grounds.

Future state BBAs should be coordinated to take place simultaneously in all states and provinces in the same time period, using a common block size and mapping system, and universal codes for breeding criteria. This effort should attempt to survey all blocks in each state or province. A scale of abundance for each species within every block should be employed, as was done in Ontario (Cadman et al. 1988).

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# GOLDEN-WINGED WARBLER

*Vermivora chrysoptera*

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**Golden-winged warblers nest in a variety of early-successional forest sites, including abandoned farmland, powerline right-of-ways, recently logged sites, bogs and swamps, and forest openings.** Nests are placed on the ground and are well concealed by vegetation. The species breeds throughout the central and southern states of the Northeast and is sparse in northern New England. Populations have declined in many northeastern states, particularly in Pennsylvania, New Jersey, Virginia, and West Virginia. Succession of forests and farmlands is a major factor in the decline of this species, coupled with the expansion of the closely-related blue-winged warbler into secondary-successional habitats. Hybrid phenotypes of the two species, known as the Brewster's and Lawrence's warblers, occur in the zones of overlap. Competition between the two species may reduce golden-winged warbler nesting success, although observations also to the contrary do not support this hypothesis. Nest parasitism by brown-headed cowbirds may additionally contribute to the decline. Habitat for the species can be created through logging, burning, and intermittent farming. Moderately-sized sites of 10-15 ha can support several pairs of golden-winged warblers, and are preferred over both smaller and larger areas.

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## DESCRIPTION

### Taxonomy and Morphology

Golden-winged warblers (*Vermivora chrysoptera*, L.; Family Emberizidae) have a typical warbler size, shape, and bright coloration. The male has a slate-gray back and nape and a white chest and abdomen. The crown and forehead are bright yellow and contrast with a black cheek and throat patch. Thin white lines separate the crown from the cheek patch, and the cheek patch from the throat patch. The slate-gray wings have a yellow patch. In females the cheek and throat

patch are gray and the back and chest may have a hint of yellow or olive.

Hybridization with the blue-winged warbler (*Vermivora pinus*, L.) produces two color patterns once thought to be separate species: the Brewster's and Lawrence's warblers. The Brewster's warbler has a white throat and, like the golden-winged warbler, usually has a white breast and gray back, but has a thin eye line and separate wing bars typical of the blue-winged warbler. The Lawrence's warbler has the black eye and black throat patch of a golden-winged warbler but has the yellow undersides and yellow-olive back of a blue-winged warbler. Intermediate forms occur and may result from matings between

two hybrids and between hybrids and parental phenotypes. Intermediates show all degrees of variation in color hue and pattern between the parental types. Short (1962) and Gill (1980) derived a hybrid index to help distinguish among the intermediate and parental phenotypes.

### Vocalizations

Golden-winged warblers have two song types, which have been extensively studied (e.g., Ficken and Ficken 1967, Gill and Murray 1972, Highsmith 1989). One song, which is common early in the breeding season, appears to advertise the territorial boundary and the male's presence and seems to have primarily an intersexual function. This Type 1 song is usually given intermittently during the morning for a few hours after sunrise, and sometimes shortly before dusk. The song is quite unmusical with one rasping note followed by two to eight buzzy notes on a constant, lower pitch. Phonetically this is sometimes represented as "bee-buzz-buzz-buzz." A Type 2 song is given during male to male encounters. This song is a series of staccato, rapid notes followed by a lower pitched, raspy trill. The blue-winged warbler has a song that is very similar in behavioral context and sound. Males of both species also sing the Type 2 song almost constantly for about a half hour just before sunrise from the beginning of the nesting season up to the time of fledging. Almost all males can be detected, barring inclement weather, in May and early June by their singing in this pre-dawn period. Females are usually inconspicuous and both sexes are usually secretive around the nest. However, as young fledge the adults become very noisy, frequently giving a distinctive "zzzzp" call as they seem to lead the young from the nest or bring food to them.

### HABITAT

Golden-winged warblers nest in dry uplands and in marshes and bogs. Territories usually have patches of herbs and shrubs, sparse tree cover, and a wooded perimeter. This species nested abundantly in the chestnut-sprout forests

(*Castanea dentata*) of West Virginia following the spread of the chestnut blight (Hall 1983). It commonly nests in upland sites on abandoned farmland in early stages of succession (e.g., Confer and Knapp 1981), or occasionally in logged areas (e.g., Will 1986). In the Canadian shield in Ontario, golden-winged warblers nest "...in alder bogs [*Alnus* spp.], especially when a few taller species [of trees] are present" (Mills 1987). Several observers have mentioned that golden-winged warblers nest in powerline right-of-ways. In southern Michigan golden-winged warblers nested in and around the edges of thickly wooded portions of tamarack (*Larix laricina*) swamps as well as in small, brushy clearings (Will 1986). In northern Michigan, Will described their habitat as including dry fields overgrown with shrubs, and woodland clearings, as well as very wet areas that were recently logged and covered with felled trees and a homogenous cover of new saplings. Will suggested that, overall, golden-winged warblers "...appeared to require proximal access to mature or second-growth woodlands as well as open areas in which there has been considerable invasion by brush, shrubs, and sapling trees."

Vegetative characteristics of golden-winged warbler territories have been quantified for southern, central, and northern New York (Confer and Knapp 1981, Frech and Confer 1987). In southern New York and contiguous New Jersey, golden-winged warblers nest in the Ramapo Mountains (Confer and Knapp 1981, Skully in press). In this rugged topography, territories occurred in marshes between rock outcrops often with a perimeter of alder surrounded by forest. In central and northern New York, territories usually were located on dry, upland sites of abandoned farmland but occasionally in wet sites. All territories had areas with dense herb growth without shrubs or trees. Herb growth of at least moderate density covered 60% or more of the ground, including the growth under woody plants. All territories had patches of dense shrubs which covered about half of each territory. Tree canopy covered less than 15% of the northern and central territories but up to 40% of the southern territories. Central and northern territories usually extended no more than 20 m into a forest, while southern territories frequently extended considerably further. In wetter sites

sedges (*Carex* spp.) were the dominant herb and alders were the dominant shrub. In upland sites a wide variety of herbs occurred while *Viburnum* spp., narrow-leaved meadowsweet (*Spirea alba*), and dogwood (*Cornus* spp.) were the dominant shrubs.

All New York territories had a similar vegetative pattern with patches of herbs and shrubs, a few trees scattered throughout, and a tree row or forest edge forming most of the perimeter (Confer and Knapp 1981, Frech and Confer 1987). In New York, abandoned farmland undergoing secondary succession has this distinctive pattern of vegetation for only about 10-20 years. Thus, golden-winged warblers at upland sites are restricted to a specific and brief stage of succession. Because of this restriction, Confer and Knapp (1981) suggested that the warbler was in some sense a habitat specialist. However, a species that can nest in chestnut-sprout forests in Virginia, tamarack bogs in Michigan, and alder swamps in Ontario clearly tolerates a wide range of conditions. It would be valuable to determine if nesting golden-winged warblers require a specific plant profile but tolerate a wide range of plant species, or tolerate a wide range of both plant profile and plant species.

## BIOLOGY

Golden-winged warblers are migrants and usually arrive in the mid-latitude of their breeding range in central New York during the first week of May. In May and early June the birds sing frequently, especially before and shortly after dawn. In July and August, after the cessation of singing and feeding of young, the birds become very inconspicuous. Extreme dates for coastal New York are 26 April (arrival) and 10 October (departure) (Bull 1974). These birds commonly nest in loose colonies of 2-6 pairs. Territories consist of about 1-2 ha (Ficken and Ficken 1968, Murray and Gill 1976, Confer and Knapp 1977). Monogamy is the norm, but polygamy occurs (J. Confer pers. obs.).

## Nesting

Nests are constructed on the ground, usually at the ecotone of a forest with a field or marsh, or in a small opening in a forest. Most nests are well concealed, frequently at the base of a clump of ferns, briars, or goldenrod (*Solidago* spp.). Often the clump includes a taller stem used for descent to the nest. Nests are coarse with an outer wrapping of bark and straw with leaf petioles protruding outward. All of about 20 nests I have found in central New York contained many large strands of reddish viburnum bark in the bottom and outer portions of the cup. The nest is lined with a criss-cross of fine, reddish-brown material. The cup is only 2-3 cm deep. This species is not known to renest after a successful clutch, but can renest after an initial nest failure. For New York, the earliest egg date is 18 May (Andrie and Carroll 1988) and the latest nestling date is 22 July (J. Confer pers. obs.). The latter is almost certainly a renesting attempt. During 3 years of study in central Michigan, the first egg dates were 15, 18, and 25 May (Will 1986). Usually 4-6 eggs are laid per nest. The eggs are pale cream to very pale blue with a scattering of fine reddish-brown dots concentrated at the blunt end, occasionally with a variety of other dark markings. Incubation by the female lasts 10-11 days. The nestling stage lasts 8-10 days (Andrie and Carroll 1988) with feeding by both sexes. Thirteen nests in central Michigan fledged an average of 3.4 young (Will 1986), while 5 nests in northern New York fledged an average of 2.6 young (J. Confer pers. obs.).

## Hybridization

Hybridization between golden-winged and blue-winged warblers has received considerable attention. Three collections from zones of overlap for these two species consisted of 11%, 17%, and 22% hybrids (Short 1962, Gill 1980). Field observations, which might not detect all hybrids, indicated that hybrids comprised 3%, 10%, and 14% of three sample sites (Frech and Confer 1987). Despite some hybridization, the parental phenotypes always appear to be more common than the hybrids, which supports the

contention that these are valid, but very closely-related species. Historically, the blue-winged warbler nested to the southwest of the golden-winged warbler (Gill 1980). Apparently, the abandonment of farmland and resultant areas of secondary succession provided routes for expansion of the blue-winged warbler into the golden-winged warbler range.

Hybrid phenotypes give the firsthand impression that the plumage differences between the two species are caused by two pairs of genes. The first generation hybrid (Brewster's warbler) gives the appearance of being heterozygous for both gene pairs. A much rarer form (Lawrence's warbler) gives the appearance of being homozygous recessive for both gene pairs. However, all gradations of color and all mixtures of color pattern between the two species have been noted (Short 1963, Gill 1980). This variation suggests that the inheritance of color is more complicated involving, perhaps, multiple but tightly-linked loci, incomplete dominance, or modifying genes.

#### POPULATION DISTRIBUTION, STATUS, AND TRENDS

##### Distribution and Status in North America

The pre-colonial distribution of the golden-winged warbler is poorly known. Expansion into southern, coastal New England was recorded more than a century ago (Gill 1980). Recent expansion has been to the north and west and still continues. On the other hand, there has been widespread decline and even local extinction of the golden-winged warbler from areas first colonized about a century ago.

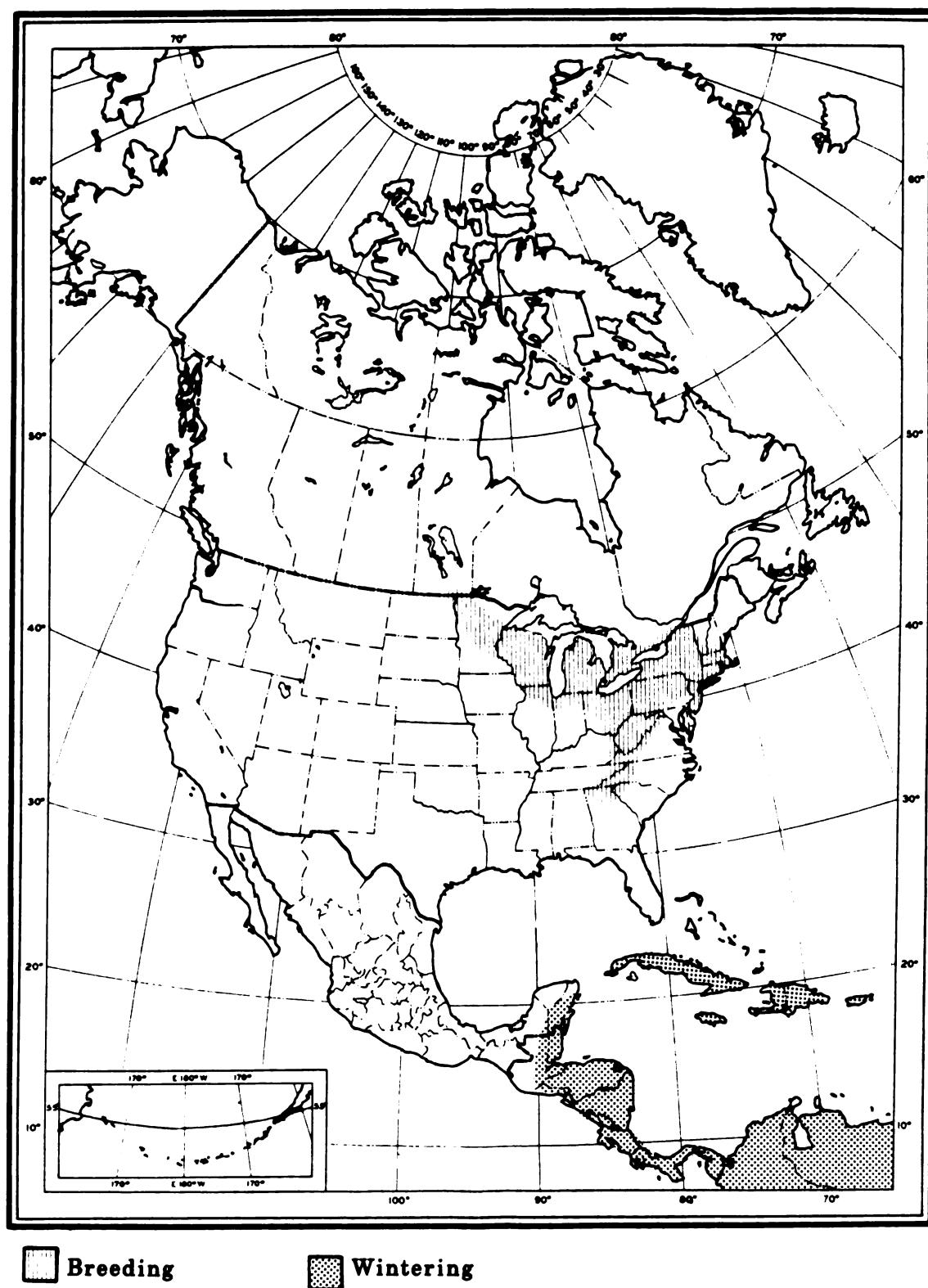
The current breeding range of the golden-winged warbler (Figure 1) includes the higher elevations of Georgia, Maryland, and Virginia where it is rare, and West Virginia where it is locally common but declining; north to central Minnesota, northern Michigan and northern Wisconsin, rarely to northeastern North Dakota and southeastern Manitoba, and rarely in intervening areas such as northern Ohio, northwestern Indiana, southeastern Iowa,

northeastern Illinois, eastern Kentucky, and eastern Tennessee. The range continues east to Ontario with recently established sites as far north as Sudbury and the Ottawa River with concentrations from southeastern Georgian Bay to the Rideau lakes, and southwestern Quebec; southeast with sparse populations in Massachusetts, coastal New Hampshire, and northeastern Connecticut with a few scattered reports in Vermont. Populations occur in the Ramapo Mountains of New York and New Jersey, and occur in moderate abundance in both the higher elevations of Pennsylvania and in New York in the southeastern Lake Ontario Plain.

As described in the *Check-list of North American birds* (American Ornithologists' Union 1983) this species "...winters from the Yucatan Peninsula and Guatemala south through Middle America (mostly on the Caribbean drainage north of Panama) to northern and eastern Colombia and northern Venezuela, and rarely in the Greater Antilles (east to Puerto Rico)."

The golden-winged warbler is listed as a *migratory nongame bird of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987). Changes in the abundance of the golden-winged warbler are indicated by Breeding Bird Survey (BBS) routes conducted annually for the last three decades under the auspices of the U.S. Fish and Wildlife Service. Results suggest a continental decline of golden-winged warblers (-2.1% annual change,  $P < 0.01$ ; Table 1). No golden-winged warblers were detected along routes in Maine, New Hampshire, Rhode Island, Connecticut, or Delaware (Office of Migratory Bird Management unpubl. data, Laurel, Maryland). The Fish and Wildlife Service suggests that population trends are not statistically reliable for states with less than 10 routes or for species in Canada that occurred on less than 25 routes. The general decline noted in Table 1 is tempered by continuing expansion in northern New York (Confer et al. in press) and much of southern Ontario (Mills 1987) and extreme southwestern Quebec. The areas of expansion in Canada may be under-represented in BBS routes, and the regional trend for the eastern U.S. may partially reflect a bias in the distribution of routes. The

Figure 1. The North American range of the golden-winged warbler (*Vermivora chrysopera*) (American Ornithologists' Union 1983).



**Table 1. Mean number of golden-winged warblers (*Vermivora chrysoptera*) per Breeding Bird Survey route and median percent annual change, 1966-89 (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; see Robbins et al. 1986 for details).**

Region	No. routes <sup>a</sup>	$\bar{x}$ birds/route	% annual change <sup>b</sup>
<b>State</b>			
Maine	-	-	-
New Hampshire	-	-	-
Vermont	5	0.13	3.5
Massachusetts	8	0.06	-1.2
Rhode Island	-	-	-
Connecticut	-	-	-
New York	47	0.21	-0.7
Pennsylvania	44	0.33	-4.0**
New Jersey	4	0.11	-5.7
Delaware	-	-	-
Maryland	5	0.59	1.6
Virginia	8	0.13	-3.8
West Virginia	29	0.92	-4.8**
Canada	21	0.05	2.0
North America	303	0.29	-2.1**

<sup>a</sup>Number of routes on which golden-winged warblers were detected (population trends may be invalid for states with < 10 routes).

<sup>b</sup>Statistical significance of trend: \*  $P < 0.05$ , \*\*  $P < 0.01$

golden-winged warbler has a Natural Heritage Program global ranking of G4 (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

#### Status in the Northeast Region

**Maine.** - The golden-winged warbler is not reported to have nested in Maine historically (Forbush 1929, Palmer 1949) and it was not recorded as a probable or confirmed breeder during the Maine Breeding Bird Atlas (BBA) (Adamus 1988, Figure 2). The species has a Natural Heritage Program state ranking of SN (regularly occurring but not breeding) (Table 2).

**New Hampshire.** - Golden-winged warblers were once fairly common in New Hampshire (Forbush 1929). According to preliminary data from the New Hampshire BBA (1981-86), they now occur rarely in the coastal plain of southeastern New Hampshire (E. Hentcy, C. Foss, S. Sutcliffe pers. comm.). The state Natural Heritage Program ranking is S2 (imperiled because of rarity).

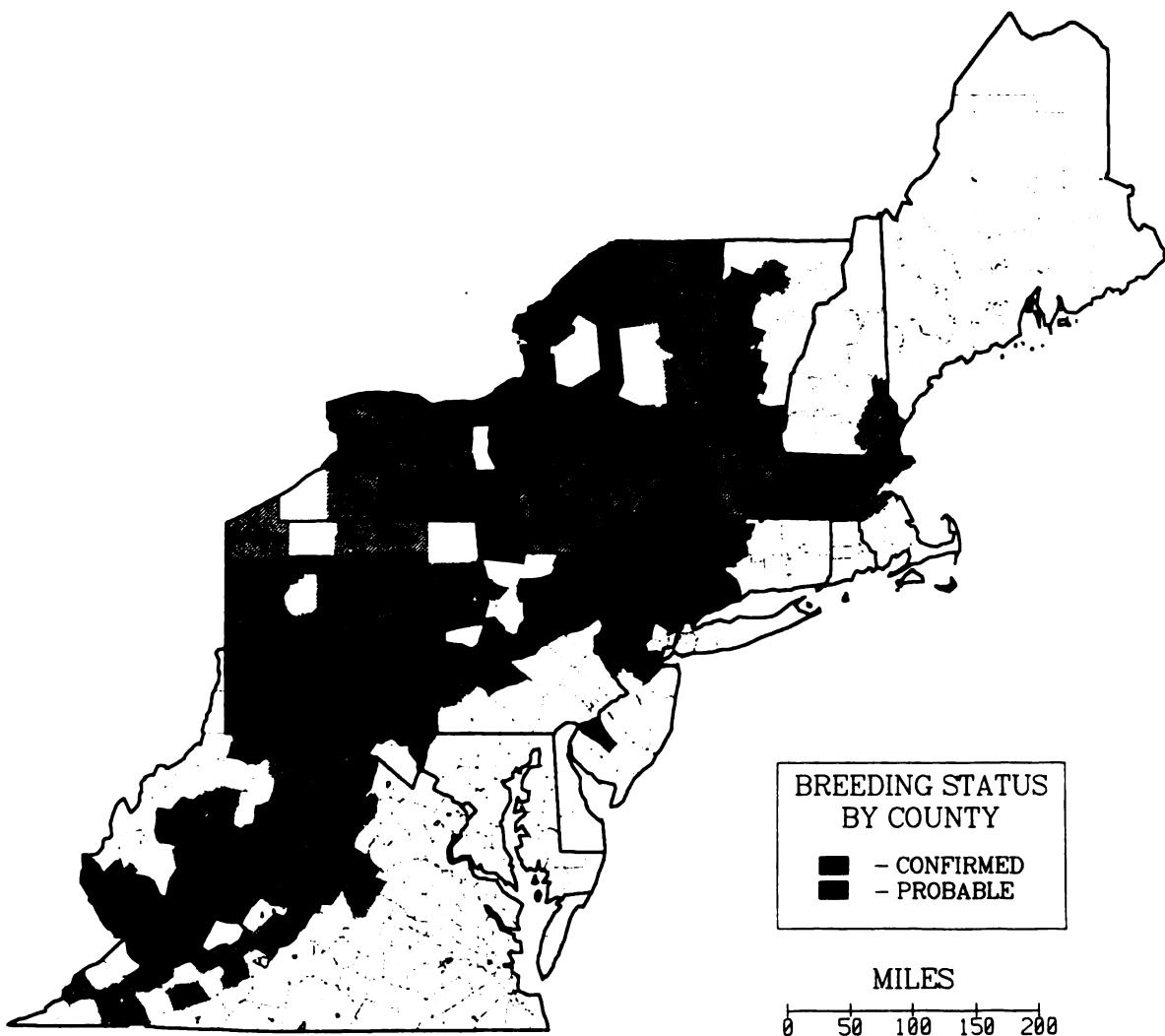
**Vermont.** - The golden-winged warbler was reported as confirmed or probable from nine (5%) of the Vermont BBA blocks (Clark 1985). The sparse reports were widely scattered but suggest a northward expansion. Golden-winged warblers have a state Natural Heritage Program ranking of S3 (rare or uncommon).

**Massachusetts.** - The golden-winged warbler was described as an uncommon migrant and a locally common to uncommon summer resident chiefly in eastern Massachusetts (Forbush 1929). The decline in this state has been documented by Gill (1980). The Massachusetts Division of Fisheries and Wildlife lists the golden-winged warbler as legally *threatened*; the state Natural Heritage Program rank is S1 (critically imperiled).

**Rhode Island.** - The golden-winged warbler was never abundant in this state; the species has only three known nesting records from 1930-56 (R. Ferren pers. comm.). It is now considered *extirpated* from the state and ranked SX by the Natural Heritage Program.

**Connecticut.** - Expansion of the golden-winged warbler into southern Connecticut and eastern Massachusetts during the late 19th century has been summarized by Gill (1980). In Connecticut the species was considered a rare migrant before 1885, but became a locally common nester early in this century (Forbush 1929). It is now considered a rare nester in northeastern Connecticut and more common in the western part of the state, being observed as confirmed or probable in 24 (4%) of the BBA blocks between 1982-86 (Bevier in press, Figure 2). Golden-winged warblers are ranked as S2 (imperiled because of rarity) by the Connecticut

**Figure 2.** Northeast breeding distribution of the golden-winged warbler (*Vermivora chrysoptera*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Laughlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Buckelew, Jr. unpubl. data).



**Table 2.** Summary of state listing status and Natural Heritage Program state ranks for the golden-winged warbler (*Vermivora chrysoptera*) in the Northeast (The Nature Conservancy: Central Scientific Databases, Arlington, Virginia).

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	SN
New Hampshire	-	S2
Vermont	-	S3
Massachusetts	threatened	S1
Rhode Island	extirpated	SX
Connecticut	special concern	S2
New York	-	S4
Pennsylvania	-	S4
New Jersey	-	S3
Delaware	-	SN
Maryland	-	S5
Virginia	-	S3
West Virginia	special concern	S4

<sup>a</sup>SN = regularly occurring but not breeding

SX = apparently extirpated

S1 = critically imperiled; ≤ 5 occurrences

S2 = imperiled because of rarity; 6-20 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

S5 = demonstrably secure

In 1988-89 it appeared that golden-wings were limited to one or two sites in Tompkins County (J. Confer pers. obs.). As the bird was declining in Tompkins County and in most of New York in general (as indicated from BBS routes) it was still expanding in much of northern New York. It is now common in the southeastern Lake Ontario Plain and expanding into the St. Lawrence Valley (Confer et al. in press). In the Ramapo Mountains of southern New York, the golden-winged warbler has maintained a seemingly stable population for nearly a century and has done so in the continual presence of blue-winged warblers. The golden-winged warbler was observed as confirmed or probable in 341 (16%) of the BBA blocks for New York (Andrie and Carroll 1988, Figure 2). The New York Natural Heritage Program has ranked the species as S4 (apparently secure).

**Pennsylvania.** - Within Pennsylvania, the golden-winged warbler is widely distributed although rare in the northern tier of counties and excluded from the southeastern corner. It was reported as probable or confirmed from 323 blocks during the Pennsylvania BBA, 1984-88 (Brauning in press, Figure 2). The golden-winged warbler is ranked S4 (apparently secure) by the state Natural Heritage Program.

**New Jersey.** - The golden-winged warbler breeds in the northeastern portions of New Jersey. The ecology of golden-winged warblers nesting in the hilly topography of Sussex County has been described by Skully (in press). This species was reported as a probable or confirmed in 55 blocks during the New Jersey BBA, mostly in the northern parts of the state (D. Hughes pers. comm., Figure 2). The state Natural Heritage Program ranks golden-winged warblers as S3 (rare or uncommon).

**Delaware.** - The golden-winged warbler is a rare but fairly regular migrant in Delaware, but was not reported as a nesting species during the state's BBA (Hess et al. in press). The species is ranked as SN (regularly occurring but not breeding) by the Delaware Natural Heritage Program.

Natural Diversity Database and a *species of special concern* by the Connecticut Department of Environmental Protection.

**New York.** - Numerous studies of the golden-winged warbler in New York allow a detailed description of what is probably a representative pattern. The golden-winged warbler occurred in only southern New York at the beginning of this century, expanded to the southern Lake Ontario Plain by 1970, and has recently expanded to the lower elevations of northern New York (Eaton 1914, Bull 1974, Confer 1987). In central New York (Tompkins County), the golden-winged warbler arrived in the second decade of the century, and became common within 30 years (Temple and Temple 1976). By the 1980s, 60-70 years after its arrival, the bird had greatly declined (Confer and Knapp 1981, Frech and Confer 1987).

**Maryland and the District of Columbia.** - The golden-winged warbler is rare as a nesting species in this state. It was reported from only five BBS routes since 1966 and confirmed in only three counties during this state's BBA from 1983-87 (G. Therres pers. comm., Figure 2). The species is ranked as S5 (demonstrably secure) by the state Natural Heritage Program.

**Virginia.** - Preliminary data from the Virginia BBA (1984-89) show that the golden-winged warbler is an uncommon breeder restricted to higher altitudes in the northwestern portion of the state (S. Ridd pers. comm., Figure 2). The Virginia Division of Natural Heritage ranks the species as S3 (rare or uncommon) in the state.

**West Virginia.** - The golden-winged warbler was reported as either probable or confirmed in 104 of 502 blocks (21%) of the West Virginia BBA (C. Stihler and A. R. Buckelew, Jr. pers. comm.). It currently breeds in the central and eastern counties and is most numerous in oak (*Quercus spp.*) /chestnut forests above 450 m. It occurs up to 1,150 m, but not into spruce (*Picea spp.*) forests. The species was formerly abundant in chestnut-sprout forests which are now gone (Hall 1983). It has disappeared from many previous breeding areas (Gill 1980), and is continually retreating to higher elevations (G. Hall pers. comm.). Golden-winged warblers are ranked as S4 (apparently secure) by the West Virginia Natural Heritage Program.

### Summary

During the last 150 years the golden-winged warbler has frequently expanded into new areas, increased to moderate abundance, and then declined or even disappeared. This sequence occurred in central New York (Temple and Temple 1976, Confer and Knapp 1981, Confer et al. in press), southern Connecticut and eastern Massachusetts (Gill 1980), southern Michigan (Berger 1958, Will 1986), northeastern Ohio (Campbell 1940, 1974), and southern Ontario (Mills 1987). In West Virginia it has disappeared from former chestnut-sprout forests (Hall 1983) and lower elevation areas (Gill 1980). These

changes as well as the BBS data suggest an overall decline for the golden-winged warbler in the Northeast. However, with its recent advances into northern Minnesota, Michigan and New York and southern Ontario, and persistent populations in Pennsylvania, Virginia, and West Virginia the entire population may be as widespread now as at any time in this century. Still, the golden-winged warbler has declined and disappeared from parts of its range, and the causes of local extirpation may come to apply to the entire population. This raises a concern for the long-term survival for the species (Gill 1980).

### LIMITING FACTORS AND THREATS

The decline of the golden-winged warbler has been correlated with two factors--the arrival of the blue-winged warbler and the loss of habitat as succession proceeds to secondary forests. Speculation of the importance of these factors is much more common than the documentation, and evidence for the relative importance of these two factors is sometimes contradictory.

Loss of winter habitat may become a threat for this species. However, the current range expansion belies any effect of this loss to date.

Speculatively, the decline of the golden-winged warbler and replacement with the blue-winged warbler has been attributed to a greater susceptibility of the first species to nest parasitism by the brown-headed cowbird (*Molothrus ater*). However, a survey of nest records showed that about 30% of the nests of both species were parasitized. This frequency of parasitism may well contribute to a regional decline, but does not support the hypothesis that the golden-winged warbler is more susceptible than the blue-winged warbler (Coker and Confer 1990).

Understanding of the population decline may be enhanced by considering the reasons for range expansion. During the latter part of the 19th century, the golden-winged warbler expanded into Massachusetts and Connecticut following abandonment of farmland in the mid-19th century (Gill 1980). In Tompkins County, New York, which may exemplify the common temporal sequence, farming peaked at the turn of the

century. Within two decades the golden-winged warbler arrived, as farmland was abandoned (Confer and Knapp 1981). Presently, golden-winged warblers are expanding in northern New York where a decline in farming is taking place. Two current study sites with 4-6 pairs each are about 30 mi (48 km) north of Syracuse on Canning Factory Road. Abandonment of this namesake factory about 30 years ago and the reciprocal decline in farming led to an early successional habitat in which golden-winged warblers now nest (J. Confer pers. obs.). Thus, in several areas the historical expansion of the golden-winged warbler occurred as secondary succession on abandoned farmland provided appropriate habitat. Other habitats used for expansion include chestnut-sprout forests in West Virginia, alder bogs of the Canadian shield, powerline right-of-ways, and some logged areas.

Clearly, farmland abandonment contributed to the expansion of the golden-winged warbler in the Northeast. However, it is not clear whether reforestation caused the subsequent population decline or whether the decline preceded the extensive reforestation. Confer and Knapp (1981) noted that the decline in Tompkins County occurred while there was still some apparently suitable but unoccupied habitat. Perhaps some minimal amount of suitable habitat is needed below which a regional population will disappear due either to stochastic events (Lande 1987) or a failure of recruits to locate the suitable habitat. Will (1986) noted that the golden-winged warbler nested in a variety of habitats in Michigan. He suggested that a population decline in southern Michigan was unlikely to be due to a loss of suitable habitat for such an adaptable species. However, a population may use a variety of habitats and appear to be adaptable only because habitat loss forces it to use submarginal habitats. With the uncertain influence of habitat loss, Confer and Knapp (1981) speculated, and Will (1986) strongly argued, that the arrival of the blue-winged warbler into a region with golden-winged warblers caused the decline of the latter species.

Gill (1980) noted that the regional decline of the golden-winged warbler correlated with the arrival of the blue-winged warbler. He documented that in several areas extirpation

occurred within 50 years or less of the arrival of the congener. Assimilation of the golden-winged warbler phenotype by the blue-winged warbler is possible since hybridization between the two species is fairly common. Competition with the blue-winged warbler could lower golden-winged warbler nesting success. Will (1986) observed that the blue-winged warbler usually dominated encounters with the golden-winged warbler. In two instances I have observed an apparently territorial male golden-winged warbler which seemed to be replaced by a newly arrived, male blue-winged warbler at the next visit. These observations support the hypothesis that behavioral interactions with the blue-winged warbler limit the golden-winged warbler.

Other observations suggest that competition with the blue-winged warbler is not the cause of the golden-winged warbler decline. The proximity of blue-winged warbler nests to golden-winged warbler nests did not affect the number of young fledged (Will 1986). An effect would be expected if competition between the species were significant. In southern New York both species have coexisted for nearly a century (Frech and Confer 1987). Perhaps persistence of the golden-winged warbler requires only maintenance of suitable habitat and the presence of the blue-winged warbler may be irrelevant. Alternatively some factor or adaptation may apply uniquely to these two populations.

The golden-winged warbler is slightly larger than the blue-winged warbler (Short 1963, Will 1986), and larger species usually dominate intrageneric interactions (Alatola et al. 1985). Golden-winged warblers dominated the majority of observed encounters with blue-winged warblers in central and northern New York during 1988-89, but the sample size was too small to detect a statistically significant difference (J. Confer pers. obs.).

In summary, the present documentation of factors limiting the golden-winged warbler is weak. Regional declines of the golden-winged warbler are probably affected by both habitat loss and interactions with the blue-winged warbler, although habitat is the essential prerequisite. The effect of nest parasitism and loss of winter habitat is not known. The relative significance of these factors may vary with location and time.

Regardless of other factors, the golden-winged warbler has always been rare or absent in regions dominated by forests, suburban expansion, and/or active farms. Much of the northeastern U.S., where golden-winged warblers were once common, is now covered by secondary forests and no longer has the habitat necessary to sustain a breeding population.

## MANAGEMENT POTENTIAL

The golden-winged warbler appears to be a fugitive species. Major shifts in location and abundance show that the species has the potential to colonize appropriate habitat throughout northeastern North America. However, the ability of the golden-winged warbler to colonize or recolonize areas where blue-winged warblers have subsequently arrived is not known. This capability is a major concern for management of golden-winged warblers in those many areas where the blue-winged warbler is now abundant. Also, the possible future loss of winter habitat and its effect is not known.

## LAND PROTECTION AND PRESERVE DESIGN

A realistic management program must provide sufficient habitat for a self-sustaining population. The population would need to be sufficiently large to reduce inbreeding and prevent the accumulation of deleterious genes in the homozygous condition. Optimally the management area should be somewhat dispersed to minimize the effect of a calamitous, local weather event. Existing information allows only a first approximation of this minimum amount of habitat.

One example of a fairly isolated population which has sustained itself for a prolonged time occurs in the Ramapo Mountains. From the known nesting sites (Confer and Frech 1987, Skully in press) and extrapolating for the entire area of similar habitat, this region might support 50-100 pairs. For comparison, there is great concern for the survival of the Kirtland's warbler

(*Dendroica kirtlandii*) with over 200 pairs. As a first approximation, a management program designed to support a semi-isolated but self-sustaining population should provide habitat for a minimum of 100, but preferably 200 pairs. This may require as much as 800 ha of managed land.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

The annual data from BBS routes combined with a repetition of statewide atlases at approximately 30-year intervals would provide an overview of the population status in large areas. Large changes in local abundance occurring within 10-20 years have been documented (Gill 1980, Confer et al. in press). Consequently, populations that are suspected to be changing should be monitored every several years. Managed areas and research sites should be monitored annually to assess the status and help determine the causes of shifts in golden-winged warbler abundance.

### Management Procedures and Programs

Any management program designed to benefit the golden-winged warbler needs to address four concerns. First and foremost, the program must maintain or create sufficient amounts of appropriate habitat. Second, the management program needs to assess the impact of blue-winged warblers and may need to institute control measures. Third, a management program needs to assess the effect of nest parasitism by cowbirds and may need to institute control measures. Fourth, the effect of loss of winter habitat needs to be assessed and corrective efforts need to be considered.

For upland sites, golden-winged warbler habitat can be created through succession following farming or fires, and sometimes logging. In New York, clearcutting is often followed by a dense and uniform growth of saplings without openings for patches of herbs. Such openings are rarely if ever used by golden-winged warblers in New York. Brushhogging, i.e., cutting woody stems of shrubs at their base, has not been

followed by nesting golden-winged warbler occupancy in the few sites I have seen. Perhaps cutting shrubs stimulates regeneration of a dense growth of woody stems without the requisite herbaceous growth. Golden-winged warblers sometimes nest under powerline right-of-ways if maintenance produces the appropriate patches of shrubs. Frequent application of herbicides may prevent the development of the requisite shrubiness (G. Hall pers. comm.).

The optimal management practice may be a rotation of burning or intermittent farming. A cycle of about 40 years with about 25% of the managed area burned once each decade could produce the following successional sequence. Golden-winged warbler habitat would begin to appear perhaps within 10 years and last about 10-20 years, although these times are approximations and would be influenced by factors such as soil quality, the size and intensity of the burn, and proximity to seed sources. Allowing succession to continue for approximately 40 years would provide the forest edge that is used in almost all territories.

Golden-winged warblers often nest in loose clusters of 2-6 or more pairs in a contiguous site. Speculatively, the birds may attract each other so that one site sufficient to support several pairs might be more productive than several small sites that are each large enough to support only one pair. On the other hand, in my experience the golden-winged warbler does not occur in high densities in large, uniform fields of over 25 ha. Perhaps the middle of such fields do not provide the forest edge which is usually part of a golden-winged warbler territory. Sites of 10-15 ha can support six pairs (Pfannmuller 1979, J. Confer pers. obs.), and this size may be close to the optimum.

At a density of 5 pairs per 10 ha, 100-200 pairs would require 200-400 ha of suitable habitat. With a 40-year cycle of burning, only about half of the managed habitat would be suitable at any one time, requiring a total of 400-800 ha of managed habitat. It is worth noting that many other species would use this habitat, including several species whose abundance is a matter of concern. For the first 10 years after a burn, the successional habitat would favor field species including perhaps Henslow's sparrow

(*Ammodramus henslowii*), American woodcock (*Scolopax minor*), and possibly the upland sandpiper (*Bartramia longicauda*). As shrubs invaded and a site became suitable for golden-winged warblers, such habitat would provide resources for other species, including winter browse for deer (*Odocoileus virginiana*). Allowing aspen (*Populus spp.*) to develop would support many other species including ruffed grouse (*Bonasa umbellus*) which use aspen buds as a major winter food source.

The necessity for control of blue-winged warblers and brown-headed cowbirds should be assessed by research as discussed below.

### Research Needs and Programs

A research program for the management and regional recovery of golden-winged warblers should determine the effect of habitat loss and blue-winged warbler competition. Previous studies have shown that the decline of the golden-winged warbler correlates with both the loss of habitat and expansion of the blue-winged warbler. However, habitat loss and blue-winged warbler expansion are themselves correlated, which makes it difficult to separate the relative significance of these events.

The current and historical relationship between golden-winged warbler abundance and habitat availability should be determined. Habitat abundance may be determined by high resolution aerial photographs, such as those available from the Agricultural Stabilization and Conservation Service. Satellite imagery might be useful but it is not certain that the variety of suitable habitats such as swamps, ungrazed shrubby fields, and logged areas could be detected.

The potential influence of blue-winged warblers could be measured in several ways. Observations of interactions between blue-winged and golden-winged warblers could document behavioral dominance, which would provide inference about a potential effect on reproductive success. Measurements of golden-winged warbler nesting success in the presence of blue-winged warblers could detect a reduction in fledging rate perhaps caused by resource competition. A comparative analysis of habitat selection by

golden-winged warblers in the presence and absence of blue-winged warblers could detect habitat displacement perhaps due to interference competition. An analysis of the recruitment of new, breeding males to a suitable patch could show behavioral exclusion of one species by the other.

Analyses of the effect of blue-winged warblers on recruitment, habitat selection, and nesting success by the golden-winged warbler requires comparisons between a control site with both species and an experimental site without blue-winged warblers. This experimental design may require the removal of blue-winged warblers by intensive mist-netting or perhaps shooting. Experimental sites should be close enough to each other to have nearly identical habitat and climate.

Cowbird nest parasitism is one more stress which may cumulatively reduce reproductive success below maintenance level. The influence of cowbird parasitism on nest success should be monitored at several sites. Earlier surveys suggested some parasitism at about 30% of the nests (Coker and Confer 1990), but the loss of potential fledglings was not estimated. If parasitism caused a 30% reduction in nesting success, a cowbird control program might be justified.

The distribution, use and abundance of golden-winged warbler winter habitat should be monitored.

Loss of breeding habitat, loss of winter habitat, interactions with blue-winged warblers, and nest parasitism may all contribute to the decline of the golden-winged warbler. These are new stresses that occur directly or indirectly because of human activities. Passive acquiescence to the negative effect of human activities, which reduces wildlife abundance and diversity, should not be confused with the slow evolution of species that are better adapted to natural selective forces. Wildlife management, involving habitat manipulation and localized control of some species, may partially redress the inadvertent imbalance resulting from our actions. For much of the northeastern U.S., present trends show that maintenance of regional populations of golden-winged warblers will require deliberate management efforts.

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# CERULEAN WARBLER

## *Dendroica cerulea*

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**Cerulean warblers inhabit mature deciduous forests on both their breeding grounds in North America and their wintering range in the Peruvian Andes. Breeding areas in the Northeast are often in floodplains or other mesic conditions, and are typified by large, mature trees and closed or semi-open forest canopies. Nests are located on the lateral limbs of a tree at considerable distances from both the ground and the bole. Common vegetation characteristics of nest sites are unknown, and use of openings and edge requires further study.** Cerulean warblers have declined in population size across their range in the eastern U.S., although the species has expanded its range, particularly in the Northeast, perhaps in response to large-scale forest maturation. More detailed population information is needed to accurately determine trends, causal factors, and distribution of the species at the landscape scale. Forests occupied by cerulean warblers should be surveyed specifically for the species as well as other neotropical migrants. Large tracts of mature forest (at least 4,000 ha) should be managed for cerulean warblers by regulating timber harvest and allowing immature stands to reach maturity.

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### DESCRIPTION

#### Taxonomy

The cerulean warbler, (*Dendroica cerulea*) was originally described as *Sylvia cerulea* (Wilson 1810; American Ornithology, volume 2, p. 141, plate 17, figure 5 (Pennsylvania = Philadelphia)). Current taxonomy is Order Passeriformes, Family Emberizidae, Subfamily Parulinae, Genus and species *Dendroica cerulea*. No subspecies are recognized (American Ornithologists' Union 1983).

#### Morphology and Plumages

The cerulean warbler is a small, arboreal, typical *Dendroica* warbler. The mean weight of 36 individuals including male and female adults and young was 9.1 gm (s.d. = 0.60 gm, range = 8.1-10.2 gm) (Clench and Leberman 1978). Wing length ranged from 62-70 mm in males ( $n = 30$ ), and 58-64 mm in females ( $n = 26$ ); tail length ranged from 38-43 mm (Pyle et al. 1987). Exposed culmen length was 10 mm (Chapman 1917).

The plumage of the adult male is bright blue above, and white below with a dark blue or black line across the throat. On the wing are two white wingbars. The feathers of the back have distinct black centers, and the flanks are streaked with black. Flight feathers are black edged with blue,

and the alula and primary coverts are black (Pyle et al. 1987).

The plumage of the adult female is greenish above, and washed with blue especially on the rump; the underparts are washed with yellow, the flanks have indistinct black streaking, the flight feathers are blackish edged with bluish-green, and the alula and primary coverts are blackish.

Juvenal plumage is brownish-gray above, with a pale median crown stripe, and entirely white underparts. Young females (hatching year/second year) have brownish flight feathers edged with greenish-yellow, and brownish alula and primary coverts. Young males have brownish-black flight feathers edged grayish-green, with brownish-black alula and primary coverts.

Full adult plumage is acquired at second prebasic molt. Prebasic molt, partial in first summer, and complete in subsequent years, takes place on the breeding grounds. Partial prealternate molt apparently takes place on the wintering grounds (Pyle et al. 1987).

### Vocalizations

The buzzy, frequently two-parted song of this species is similar to that of the northern parula (*Parula americana*). The initial portion of the song is measured, with three or four notes, followed by a second part on a higher pitch of a similar number of notes but sung twice as fast, viz., "ZHEE ZHEE ZHEE ZHEE-zhizhizhizhi." The song is distinguished from that of the northern parula in that cerulean warblers do not terminate their song with a rising trill, or with the "tipping over" note with which the northern parulas typically end their songs. However, songs of northern parulas that terminate in neither of these features are very similar to those of cerulean warblers and considerable practice in the field is necessary to distinguish the two quickly and confidently.

Saunders (cited in Bent 1953) describes the song as similar to that of the blackburnian warbler (*Dendroica fusca*), "...but the loudness, different quality, and lower pitch distinguish it." Allen (cited in Bent 1953) renders the song "WEE WEE WEE bzzz" [emphases mine]. Others note the similarity of the pattern of the song of

this species to that of the black-throated blue warbler (*Dendroica caerulescens*).

### HABITAT

#### Breeding

The breeding habitat of the cerulean warbler is frequently described as mature deciduous forest, particularly in floodplains or other mesic conditions. Robbins et al. (in press), in a detailed examination of the territories of a number of individuals in several populations in Tennessee and Maryland, give considerable substance to this premise, as well as the results of Kahl et al. (1985) from Missouri.

Cerulean warbler territories in central and western Tennessee are found in forest stands with numerous large trees. Within the stands, territories are located in the areas with more large trees than is typical for the stand and are absent from the portions with small trees (Hamel 1988, Robbins et al. in press). Observations of the utilization of forest vegetation by the birds found them in the canopy of the trees, almost always above the midpoint of the tree, at an average height of 17 m in a 22-m-tall tree. These results somewhat contradict the statements of Morse (1989), (citing Anderson and Shugart (1974)), that the species utilizes habitat components in proportion to their availability.

Robbins et al. (1989) studied 469 forest sites in Maryland and adjacent states and found the occurrence of cerulean warblers to be associated with large tracts of mature, semiopen deciduous forest. Distribution of breeding cerulean warblers was positively correlated with the natural logarithm of the area of the forest ( $P < 0.01$ ), and the square root of the basal area of the trees ( $P < 0.05$ ), and negatively correlated with the arcsine of percent canopy cover by coniferous trees ( $P < 0.05$ ).

In Missouri, Kahl et al. (1985), (cited in Robbins et al. (in press)), found that habitat around song perches of cerulean warblers was most consistently characterized by a large number of live stems  $> 30$  cm dbh (range = 50-150/ha), and a high (always  $> 18$  m), closed canopy ( $>$

85%, never < 65%). Other important features included an intermediate number of woody stems < 2.5 cm dbh (1,030-2,800/ha, never < 1,030), and few dead stems 2.5-9.9 cm dbh (always < 175/ha).

In North Carolina, a disjunct population occurs in the old-growth, mature floodplain forest communities of well-drained natural levees within 330 m of the Roanoke River (Lynch 1981, cited in Robbins et al. in press). The dominant canopy species are sycamore (*Platanus occidentalis*), green ash (*Fraxinus pennsylvanica*), and sugarberry (*Celtis laevigata*). These communities are characterized by a closed canopy ranging in height from 24-30 m, a distinct shrub layer, and complete ground cover.

In the Cumberland Mountains of eastern Tennessee, a dense population of cerulean warblers occurs at Frozen Head State Natural Area (C.P. Nicholson pers. comm., pers. obs.). This area's second-growth forest is in advanced growth, with a high, closed canopy of large trees similar to those described above, in mesic cove and slope topography. Forest composition is of a diverse assemblage of hardwood trees. In North Carolina, cerulean warblers are also found in mature cove hardwood forests on relatively steep slopes with little understory (H. LeGrand pers. comm.).

A tentative description of the characteristics of breeding habitat for cerulean warblers is a structurally mature hardwood forest in a mesic or wetter situation, with a closed canopy. I believe that the size of the trees is of paramount importance, and their species secondary to their size. The relationship between the size of the trees, the composition of the stand, and the extent of the stand is discussed below under *Limiting Factors and Threats*. Landscape situation and context has a strong bearing on whether otherwise suitable breeding habitat will actually contain cerulean warblers.

#### Wintering

The winter habitat of the cerulean warbler is concentrated on the eastern slopes of the Andes Mountains in western South America. E elevational range is limited to the lower slopes between 500-1,500 m, in precisely the elevation at which human habitat encroachment is proceeding

most rapidly (Skutch, cited in Bent 1953; J. Fitzpatrick and S. Robinson pers. comm.). The winter habitat is mature deciduous forest, also with large trees, although quantitative measurements are so far lacking. This recent information is different than that quoted by Bent (1953, referring to Taczanowski) that the birds range between 10,000-13,000 ft (3,000-4,000 m) in the Peruvian Andes. Terborgh (1989) associates cerulean warblers with montane forests of middle elevations in the Northern Andes. He further indicates that the species does not accept disturbed habitats.

## BIOLOGY

Cerulean warblers are apparently a monogamous, single brooded, forest dwelling *Dendroica* species. They arrive in North America in April and leave in September for wintering grounds in northern South America. Their primary breeding range is in the deciduous forests of the upper Mississippi and Ohio valleys. Besides the summary by Bent (1953), no life history account has been written on the cerulean warbler. The recent works by Morse (1989) and Terborgh (1989) include little information about this species.

#### Feeding

Cerulean warblers are insectivores, foraging in and about the foliage of deciduous trees for small arthropods which they capture by gleaning and by sallying. A sample of four stomachs taken in Alabama in April 1912 (Howell 1924) contained Hymenoptera (42%); Coleoptera, including weevils (23%); and Lepidoptera (35%). These are the only quantitative data on the diet of the species. Warren (1890) mentions a stomach (presumably from a bird taken in Pennsylvania) that contained fragments of spiders and small beetles. In Nebraska, S. Aughey in June 1865 watched an adult cerulean warbler bring locusts to its young (C. Robbins pers. comm.). No plant materials have yet been reported in the diet. Current information is insufficient to establish a distinguishing characteristic of the diet.

### Nesting

The compact nest of the cerulean warbler is built by the female on the lateral limbs of a tree. The nest is made of brown bark covered with gray plant material such as lichens and mosses, and is lined with mosses (Bent 1953, Harrison 1984). The nest is placed at a considerable distance (e.g., 2-7 m; Harrison 1984) from the bole of the tree, usually saddled on a large, lateral branch, attached perhaps to a small protruding twig. The nest is rather shallow for a warbler (Bent 1953). Variation in site selection is considerable, particularly with respect to the distance from the bole. All authors agree, however, that cerulean warblers do not build their nests near the ground. Heights from 5-20 m are reported by Bent (1953), Harrison (1984), and Hands et al. (1989), and the typical height is probably above the middle of this range.

Placement of the nest in the habitat, however, has been described differently by various authors. Previous summaries of breeding habitat from studies in the 1980s indicate that the forest in which the birds breed is one with a closed canopy. Bent (1953), on the other hand, quotes Burtch that "...the nest is usually placed...over an opening [sic]." A nest discovered in central Tennessee in 1950 was built in an elm at the edge of an opening beside a farmhouse (K. A. Goodpasture pers. comm.). Bent (1953) further describes a nest site discovered by Kirkwood that was 5 m out on a limb of a tulip-tree (*Liriodendron tulipifera*), and 14.8 m up "...with no other limb between it and the ground." Harrison (1984) presents a photograph of breeding habitat which shows a discontinuous canopy edge at a forest road.

Several important questions arise from these conflicting indications of nesting habitat. Do cerulean warblers usually nest in continuous, unbroken forest? Do cerulean warblers build their nests in the canopy of the forest where little other vegetation occurs between the nest and the groundcover? Do cerulean warblers typically build their nests in association with canopy gaps in otherwise unbroken forest? Are cerulean warblers indifferent to nest situation, using closed canopy forest, canopy gaps, and the edge between forest and other land uses in proportion to their

availability? The literature is insufficiently clear to distinguish these alternatives. Observer bias is certainly possible in reporting nest locations when nest searches were haphazard, as was almost certainly the case in the existing literature. In such a case, nests built in locations that are easy to find, i.e., over edges and openings, would appear with greater relative frequency in the literature than in the field. One further possibility is that current observations may not reflect the preferences of the birds as they existed in the past. Instead, the current situation may be only a remnant of the actual capability of the birds; a remnant produced by the action of the limiting factors discussed below.

The female cerulean warbler lays 3-5, though usually 4 eggs. Incubation is believed to be about 12 days (Harrison 1975), and nest life of the young is 9-10 days ( $n = 1$ ; Southern 1962). These values are all typical for a small warbler. The young are fed by both parents, and subsequent to fledging, they move first to lower-level vegetation where their parents feed them, and then gradually farther afield as they approach independence. The birds leave the breeding grounds early, perhaps in early July. Records exist of arrival in Ecuador as early as mid-October (Bent 1953).

### Wintering

Little work has been done on the winter ecology of cerulean warblers. J. Fitzpatrick and S. Robinson (pers. comm.) have observed that the birds participate in canopy-foraging flocks of forest insectivores, and only one or two cerulean warblers occur within a particular flock. This observation implies that the foraging flock is a resource that is actively defended by the individual cerulean warbler against conspecifics. Though empirical data to support this hypothesis are lacking, this behavioral tendency could be of critical importance for the future of the species. Single flock membership implies a finite limit to the population of the species; a limit determined by available space. However, these recent findings are in stark contrast to the earlier reports of movements of large flocks of cerulean warblers (Bent 1953).

## POPULATION DISTRIBUTION, STATUS, AND TRENDS

### Distribution and Status in North America

Cerulean warblers are widely distributed in the deciduous forest region of eastern and central North America. Although their former range was primarily in the Mississippi Valley, over the past 100 years the birds have extended their breeding range to the eastern side of the Allegheny Mountains and into New England and southern Canada (Figure 1).

Specifically, the cerulean warbler:

*Breeds from southeastern Nebraska, northern Iowa, central and southeastern Minnesota, southern Wisconsin, southern Michigan, southern Ontario, southwestern Quebec, western and southeastern New York, northwestern Vermont and central Connecticut south to eastern Oklahoma, northcentral Texas (to Dallas area), southern Arkansas, southeastern Louisiana (probably), central Mississippi, central Alabama and central Georgia, and east to northern New Jersey, northern Delaware, eastern Maryland, central Virginia and central North Carolina. The bird winters from Colombia and Venezuela south, mostly east of the Andes, to eastern Peru and northern Bolivia (American Ornithologists' Union 1983).*

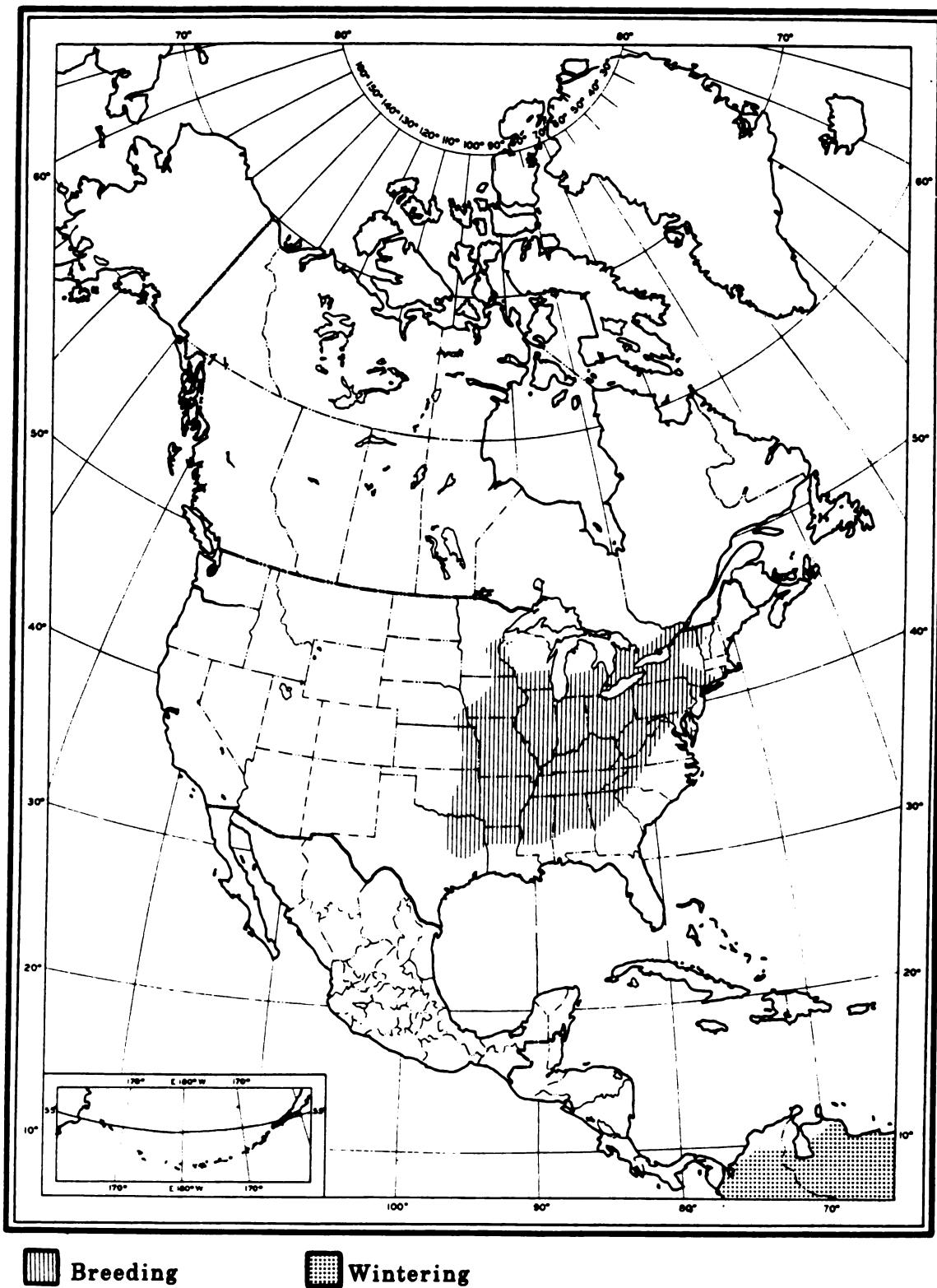
Within this range, particularly the breeding range, the birds have been discontinuously distributed since they became known to science. This is an important feature of their story. Hands et al. (1989) indicate that prior to the 1950s the birds nested in floodplain forests throughout the Midwest west to Minnesota. The current distribution in the northcentral states is primarily east of the Mississippi River.

Currently, two trends appear to be operating in the population of the species. First, a decline in the total population is indicated across its range from 1966-89 by Breeding Bird Survey (BBS) data (Robbins et al. in press). The annual declines in populations of this species as measured by the BBS, 3.4%/year, is the most

precipitous for any warbler species, and only five other species in the North American breeding avifauna exhibited greater declines (Office of Migratory Bird Management unpubl. data, Laurel, Maryland; Robbins et al. in press). The decline is most well-defined in Ohio, Kentucky, and Tennessee. This trend is especially alarming because these states are in the center of the species' range. Across the Northeast, an annual decline of 1.8% is suggested by BBS data from 1966-90 ( $n = 133$ ; not statistically significant), and specific states with high total numbers of cerulean warblers counted, including Pennsylvania, Indiana, Maryland, and Virginia, and particularly West Virginia, recorded populations that were possibly declining as well. In these latter cases, the trends were not significantly different from a constant population, however. Possible increases (also not statistically significantly different from a constant population) were recorded in Wisconsin and New York. Cerulean warblers were recorded on BBS routes in too few numbers in other states to assess trends. In no state was a significant increase in cerulean warbler numbers observed. In 1987, the species was listed as a *migratory nongame bird of management concern* by the Office of Migratory Bird Management (U.S. Fish and Wildlife Service 1987), and more recently as a proposed candidate (category 2) for listing as a federally endangered or threatened species. Cerulean warblers are ranked globally as G4 (The Nature Conservancy, *Central Scientific Databases*, Arlington, Virginia).

Concomitant with the steady rangewide decline in numbers of cerulean warblers is the steady increase in the range of the species to the east and north, particularly in the Northeast. Data from breeding bird atlases (BBAs), summarized below, and other observations indicate that the geographic extent of the range of cerulean warblers east of the Allegheny Mountains is greater than was believed to be the case in the time of Bent (1953). The explanation for this seeming paradox is complex, speculative, and cannot be made satisfactorily. I believe that it lies in land use changes in areas that were logged heavily in the past century and earlier, that are now achieving forests of sufficient stature to attract cerulean warblers. Brooks (1940) discusses

Figure 1. The North American range of the cerulean warbler (*Dendroica cerulea*) (American Ornithologists' Union 1983).



Breeding

Wintering

responses to land use changes in West Virginia by cerulean warblers in the early years of this century. The current range may further be a result of a more thorough documentation of distribution by observers. It is possible that this range "expansion" is actually a "reoccupation."

### Distribution and Status in the Northeast Region

In the state discussions that follow an attempt is made to evaluate the status of the cerulean warbler based on a comparison of historic accounts to three types of contemporary data. These include BBS analyses, the Natural Heritage Program state ranks (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia; Table 1) and data from the

**Table 1. Summary of state listing status and Natural Heritage Program state ranks for the cerulean warbler (*Dendroica cerulea*) in the Northeast (The Nature Conservancy: *Central Scientific Databases*, Arlington, Virginia).**

State	Listing category	Natural Heritage state rank <sup>a</sup>
Maine	-	S?
New Hampshire	-	-
Vermont	-	S1
Massachusetts	-	S?
Rhode Island	-	S1
Connecticut	-	S3
New York	-	S5
Pennsylvania	-	S4
New Jersey	-	S4
Delaware	-	S3?
Maryland	-	S5
Virginia	-	S4
West Virginia	-	S5

<sup>a</sup>S? = status uncertain

S1 = critically imperiled; ≤ 5 occurrences

S3 = rare or uncommon; 21-100 occurrences

S4 = apparently secure

S5 = demonstrably secure

state breeding bird atlases (Figure 2). Natural Heritage Programs, typically a joint effort of The Nature Conservancy and a division of state government, assign state ranks based on the number of known occurrences. They range from S1 (typically ≤ 5 occurrences) to S5 (demonstrably secure). A more complete explanation of the Heritage ranking system appears in Table 1. While these *S* ranks carry no legal protection, the Heritage Programs do use standard ranking criteria from state to state.

**Maine.** - Cerulean warblers have not been recorded in Maine as a breeding species according to the Maine BBA (Adamus 1988). The species is ranked as S? (status uncertain) by the Maine Natural Heritage Program.

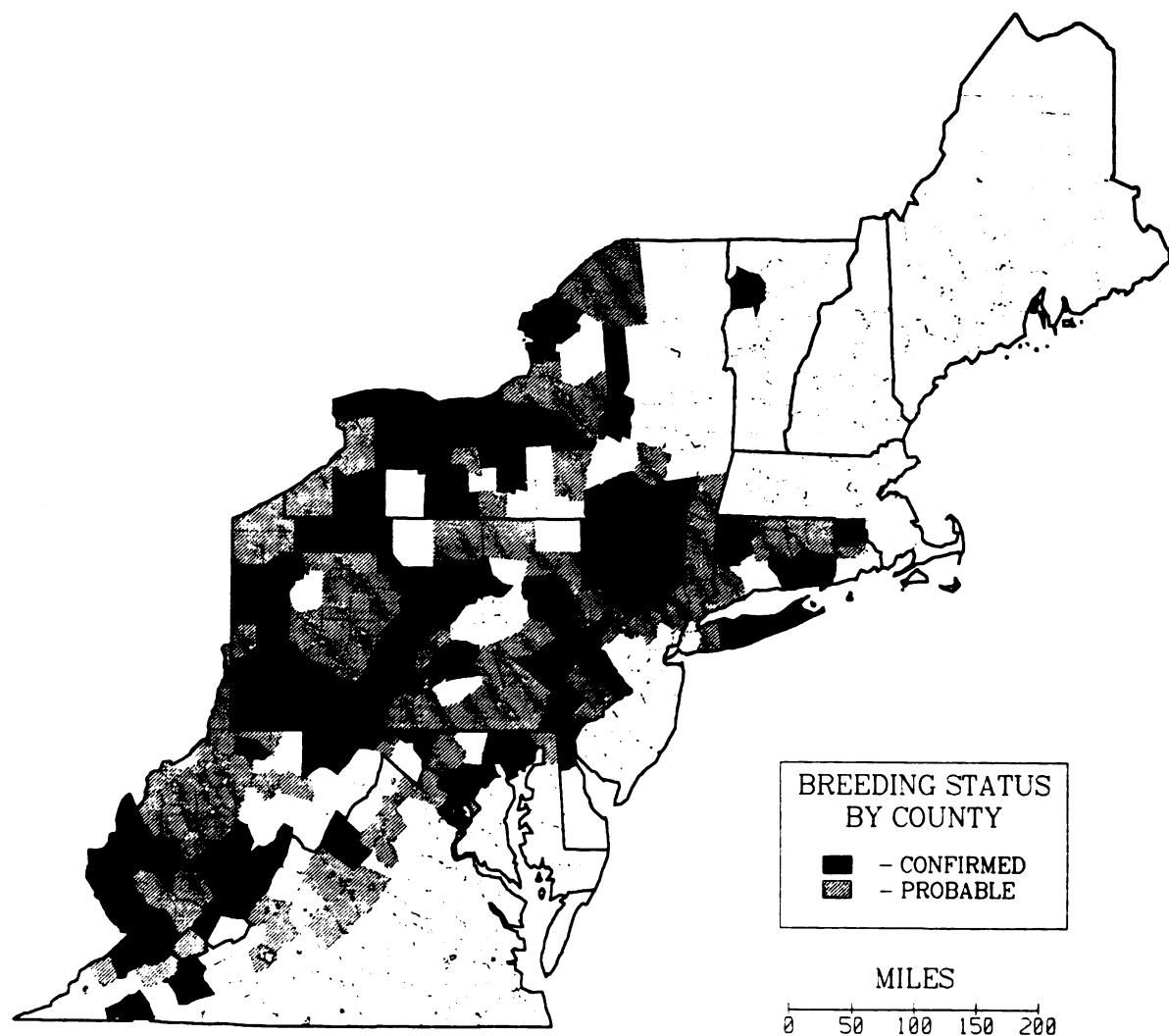
**New Hampshire.** - The species has never been observed breeding in New Hampshire and no occurrences were recorded in the 1981-86 statewide BBA (E. Hentcy, C. Foss, S. Sutcliffe pers. comm.).

**Vermont.** - The cerulean warbler was first confirmed as a breeding species in Vermont in 1977 during the Vermont BBA (Laughlin and Kibbe 1985). The state's population varies from four to six pairs annually at its one known locality along Lake Champlain. The Vermont Fish & Wildlife Department lists cerulean warblers as a *species of special concern* (no extra protective status). The species is ranked as S1 (critically imperiled) by the Vermont Natural Heritage Program.

**Massachusetts.** - Cerulean warblers are not known to breed in Massachusetts (J. Baird pers. comm.). The species is listed as S? (status uncertain) by the Massachusetts Natural Heritage Program.

**Rhode Island.** - Cerulean warblers were discovered in three blocks during the statewide BBA (1982-87) (R. Enser pers. comm.). The species is believed to be a recent invader of the state. The cerulean warbler is ranked as S1 (critically imperiled) by the state's Natural Heritage Program.

**Figure 2.** Northeast breeding distribution of the cerulean warbler (*Dendroica cerulea*) by county based upon probable and confirmed breeding records during state Breeding Bird Atlas projects. Confirmed is defined as positive evidence of breeding; probable means suggestive but inconclusive evidence of breeding. Years of atlas field work and sources for each state are as follows: Maine 1978-83 (Adamus 1988); New Hampshire 1981-86 (C. Foss, E. Hentcy and S. Sutcliffe unpubl. data); Vermont 1976-81 (Langlin and Kibbe 1985); Massachusetts 1974-79 (J. Baird unpubl. data); Rhode Island 1982-87 (R. Enser unpubl. data); Connecticut 1982-89 (Bevier in press); New York 1980-85 (Andrie and Carroll 1988); Pennsylvania 1983-89 (Brauning in press); New Jersey 1981-85 (D. Hughes unpubl. data); Delaware 1983-87 (Hess et al. in press); Maryland 1983-87 (G. Therres unpubl. data); Virginia 1984-89 (S. Ridd and R. Wadja unpubl. data); and West Virginia 1984-89 (C. Stihler and A. R. Bucklew, Jr. unpubl. data).



**Connecticut.** - Small numbers of cerulean warblers breed in the hills of the western part of the state. The Connecticut BBA (1982-86) recorded cerulean warblers on 6% of the blocks representing each county of the state. First nesting was suspected in the 1930s, and first confirmed in 1972 (Bevier in press). The Connecticut Natural Diversity Database ranks the cerulean warbler as S3 (rare or uncommon) in the state.

**New York.** - Rare in many counties of the state, the species is locally common in others. Populations have been known in the Great Lakes Plain and Finger Lakes Highlands since the 1870s (Bull 1974), and east of the Hudson River since 1922. The species only recently reached Long Island (Salzman 1983). Cerulean warblers were recorded on 209 (5%) of BBA blocks in New York, and are widely distributed throughout much of the state exclusive of the Adirondack Mountains (Andrle and Carroll 1988). BBS data indicate less than a 1% annual change in recorded numbers on 15 routes from 1966-87. The number of routes with increasing trends (60%) was not significantly more than the number with decreasing trends (33%). In Ontario adjacent to New York, the birds are local and uncommon in distribution, and "...undoubtedly affected by the significant decline of forest cover" (Cadman et al. 1987). As in New York, the population in Ontario and Quebec has expanded in area in the past century (Robbins et al. in press). The New York Natural Heritage Program currently ranks the cerulean warbler as S5 (demonstrably secure).

**Pennsylvania.** - Todd (1940) reported that cerulean warblers were found primarily in western Pennsylvania west of the Allegheny Mountains where they were formerly fairly common, but had decreased in abundance by 1940. Cerulean warblers also occurred east of the Alleghenies but in lower numbers than in the Ohio Valley. Wood (1973) noted the birds as breeding residents across southern Pennsylvania north to Centre County. BBS data indicate less than a 1% annual change in recorded numbers on 42 routes from 1966-87. The numbers of routes with increasing trends (43%) were approximately the same as numbers with decreasing trends (55%). The

cerulean warbler is ranked as S4 (apparently secure) by the Pennsylvania Natural Diversity Inventory.

**New Jersey.** - Cerulean warblers were first confirmed as a breeding species in New Jersey in 1947. The species is currently recorded breeding locally in the northern third of the state and occasionally in the pine barrens (New Jersey Natural Heritage Program: *Vertebrate Characterization Abstracts*, Trenton). BBS data from three routes, 1966-87, were insufficient to determine any trend information for the population of cerulean warblers in this state. The species is ranked as S4 (apparently secure) by the state Natural Heritage Program.

**Delaware.** - Fewer than ten pairs breed in the state at present in New Castle County. A population in Kent County was extirpated 50 years ago (Hess et al. in press). Cerulean warblers are ranked as S3 (rare or uncommon) by the Delaware Natural Heritage Program.

**Maryland and the District of Columbia.** - Cerulean warblers were observed on 147 recent BBA blocks on 75 topographic maps (G. Therres pers. comm.). Robbins et al. (1989) examined 469 forest tracts in Maryland and adjacent counties of Pennsylvania, West Virginia, and Virginia and discovered cerulean warblers on 15 of these tracts. BBS data indicate less than a 1% annual change in recorded numbers on 15 routes from 1966-87. The number of routes with an increasing trend (40%) was approximately the same as the number with decreasing trends (53%). The Maryland Natural Heritage Program ranks the species as S5 (demonstrably secure).

**Virginia.** - Cerulean warblers were recorded from 59 BBA blocks on 50 different topographic quadrangles, with the majority along the Blue Ridge. The species is considered rare in the piedmont and coastal plain, and uncommon in the mountains, with declining populations (S. Ridd and R. Wadja pers. comm.). BBS data indicate a significant annual decline of 2% ( $P < 0.01$ ) in recorded numbers on 12 routes from 1966-87. The number of routes with an increasing trend (25%) was significantly less ( $P < 0.05$ ) than the

number with a decreasing trend (67%). Cerulean warblers are ranked as S4 (apparently secure) by the Virginia Division of Natural Heritage.

**West Virginia.** - West Virginia is the one bright spot in the Northeast and throughout the breeding range of the species. In this state the birds are not associated solely with any one type of forest community (Hall 1983), and an index of singing males ranged from 57 males per 100 ha in northern hardwoods in Brooke County to 207 males per 100 ha in mature oak (*Quercus* spp.)/hickory (*Carya* spp.) forest in Wetzel County. Their presence was detected on 209 BBA blocks in the state, perhaps as many as 50% of the topographic quadrangles, and confirmed on 31 of them (C. Stihler and A. Buckalew, Jr. pers. comm.). The cerulean warbler is ranked as S5 by the state's Natural Heritage Program.

I suspect that the percentage of forested land in West Virginia holds a key to the bird's abundance in the state, and that the pace of development of that land will determine the future of the species. The fact that the state has lost human population over the past several decades is a hopeful indicator for cerulean warblers in West Virginia. BBS data indicate less than a 1-2% annual decline (not statistically different from constant numbers) in recorded numbers on 33 routes from 1966-87. Numbers of routes with increasing trends (30%) were significantly less ( $P < 0.01$ ) than the numbers with decreasing trends (70%).

### Summary

In the Northeast, cerulean warblers are most abundant in the central and southern states of Pennsylvania, Maryland, Virginia, and West Virginia, where populations are stable or possibly declining. Populations have expanded into Rhode Island, Connecticut, and Vermont in small numbers, and into previously unoccupied areas of New York. Across its range, however, the species is declining at a high annual rate (3.4%/year).

### LIMITING FACTORS AND THREATS

The fact that the rangewide decline in numbers of cerulean warblers is most severe in the center of their range, where the highest numbers of individuals are recorded on the BBS, is a cause for concern.

The threats and limiting factors most frequently mentioned concerning cerulean warblers are destruction of both breeding and wintering habitat. In each case, human agency is associated with the destruction as the primary threat. The specific traits of the species as a single brooded, forest nesting neotropical migrant are believed to be the features that put the species at risk for population decline resulting from habitat destruction.

Habitat destruction on the breeding and wintering grounds may operate at three different scales by three different mechanisms. First, as potential breeding habitats are destroyed, the gross area of potential habitat is reduced, and the actual carrying capacity of the breeding or wintering range is reduced in proportion to the reduction in area. While this threat is straightforward and undeniable, it has not proceeded to the point that it alone is threatening the population with extinction rangewide. Second, the manner in which the reduction in gross area occurs can affect the actual carrying capacity of the remaining habitat. Evidently, patches of habitat below a certain size are simply not capable of supporting breeding cerulean warblers (Robbins et al. 1989). Whether this is also true in the wintering range is unknown.

Third, some evidence indicates that certain tree species of apparent importance to the birds may be experiencing rangewide declines in vigor as a result of as yet undetermined factors (Robbins et al. in press).

Hands et al. (1989) list contaminants, predation, competition, diseases/parasites, weather, and human disturbance as additional potential limiting factors. Of these, they comment only that red squirrels (*Tamiasciurus hudsonius*) may predate the nests, and note populations of red-eyed vireos (*Vireo olivaceus*), northern parulas, and yellow-throated warblers (*Dendroica dominica*) as possible competitors.

Robbins et al. (in press) indicate that nest parasitism by brown-headed cowbirds (*Molothrus ater*) is a likely factor in the decline of the species. It is my opinion that nest parasitism by cowbirds is at least part of the mechanism by which the forest fragmentation effect is manifested.

No specific causal connection of the decline of the species to any particular factor has been investigated or demonstrated. I believe that several factors are involved in synergy, and that protection efforts will be required before the complex of causes can be proved beyond a doubt. I further believe that fragmentation is not the cause of the reduction in populations of the birds in small tracts, but the precursor that permits the operation of that causal factor.

### MANAGEMENT POTENTIAL

The management potential of cerulean warbler populations is unknown. Populations in large tracts in good habitat are apparently stable, suggesting that factors responsible for the decline are not operating uniformly everywhere. The persistent increase in the known range of the species, particularly in the Northeast, suggests that the management potential of the species is good.

Restoration of cerulean warblers to areas from which old forest habitat has been removed will require substantial periods of time. Reverting farmland and early-successional forests can be regenerated to suitable mature hardwood forest within 80 years. Recovery of habitats for this species will require an unavoidably long-term commitment.

### LAND PROTECTION AND PRESERVE DESIGN

Regardless of the mechanisms operating, breeding populations of cerulean warblers in small forest tracts throughout the range of the species are declining rapidly to extirpation. Populations in Wisconsin showed increasing size dependency in a study in which the largest habitat "island" was 40 ha (Gustafson 1985). Robbins et al. (1989) indicate that the probability of occurrence of

cerulean warblers in their Maryland study sites reached 50% of its greatest value when tract sizes were 700 ha or greater. In a study in western Tennessee bottomland hardwood forests, cerulean warblers were not found in tracts less than 1,600 ha in extent (Robbins et al. in press).

The implications of these results for land protection and preserve design are clear: protection of land for cerulean warblers will only be possible in large tracts. I suggest that these tracts must be at least 4,000 ha in extent, and that they be arranged in such a way that a minimal perimeter distance occurs per unit area. No proof exists that the provision of such tracts, composed of suitable breeding habitats, will provide a secure future for cerulean warblers as a breeding species in the North American avifauna. However, there is ample evidence that failing to provide such tracts will result in a decidedly insecure future for the species.

The location of breeding and wintering habitats of individual populations of the species is unknown. Consequently, protection of breeding habitat for a particular population may afford no long-term security if its wintering ground is also not secure. Similarly, protection of a particular winter location may not afford security for the birds that winter there unless their breeding grounds are also secure. Establishment of a network of large preserves with extensive tracts of old forest, representing the breadth of both the breeding and wintering grounds, will most likely provide a potentially secure future for these birds.

Reserves for cerulean warblers are not necessarily incompatible with a variety of other low intensity land uses, including forest management, as long as the openings created by forest harvest activities are small. The definition of small in this instance is speculative, but I believe that it is of the order of magnitude of a treefall gap rather than of an 8-ha clearcut patch. Thoughtful guidance for preserve design considerations is provided by Harris (1984) and Maser (1988).

Following are summaries of the recommendations for preserve design on the breeding grounds:

- Provide a network of large (at least 4,000 ha) compactly shaped reserves, each capable of providing habitat for 1,500 breeding pairs.
- Distribute these reserves in such a way that they represent the breadth of the species' range in the middle Mississippi Valley, including particularly Ohio, Pennsylvania, West Virginia, Kentucky, Tennessee, Arkansas, Missouri, and Indiana.
- Provide habitat in these and in other reserves such that compact, continuous, centrally located tracts of old forest are permitted to become established and persist on good soils in these and in other forest tracts.

On the wintering grounds, preserve design should include the following recommendations:

- Establish a similar network of preserves in primary forest. Such preserves will also likely be of considerable size, a size as yet undeterminable.
- Distribute the preserves so that they represent the breadth of the winter range.
- Investigate the species on the winter grounds.

## MANAGEMENT AND RESEARCH

### Monitoring Procedures and Programs

Hands et al. (1989) suggest:

*Ideally, surveys should be conducted annually at selected sites throughout the northcentral U.S. (i.e., the same sites should be sampled each year). Annual population surveys could be justified if surveys are targeted for cerulean warblers and several species that utilize the same habitat. If annual surveys are not possible, it may be best to survey the same sites for several consecutive years (perhaps*

*three). The frequency of future surveys should then be conducted once every three years.*

*If surveys detect a population decline, studies of breeding success and productivity should be initiated. These studies should be conducted at selected sites throughout the northcentral U.S. Productivity studies should be repeated annually at the same sites until the reasons for population declines are better understood.*

*Coordination of state or regional surveys is desirable so that funds and personnel are used in the most efficient manner possible. One option would be for the U.S. Fish and Wildlife Service Office of Migratory Bird Management to coordinate nongame bird population surveys being done at state or regional levels. By acting as a clearinghouse for survey efforts, the Service could prevent duplicative surveys and standardize methodology when possible.*

Current, wide-scale BBS monitoring of populations of cerulean warblers on the breeding grounds provides useful information. However, this monitoring is too imprecise to document trends accurately and to relate trends to causes. A scheme such as that proposed by Hands et al. (1989) is required to monitor changes. A part of that scheme can be accomplished utilizing the requirement of the National Forest Management Act of 1976 that National Forest lands be monitored for populations of native vertebrates. The cerulean warbler is an excellent candidate for an indicator species for mature hardwood forests, particularly on floodplain sites, but also in mesic upland types as well.

Further monitoring of breeding populations will also be required, i.e., monitoring that can document the actual distribution of cerulean warblers at the landscape scale in regions that are primarily forested and in those that are not. This species is an excellent one to use as the vehicle for a thorough documentation of the operation of the forest fragmentation effect.

Monitoring of winter habitat can be done utilizing remote sensing technology for large-scale examinations of trends. Later, more detailed monitoring of the numbers and distribution of the birds in winter will be required.

## Management Procedures and Programs

Hands et al. (1989) and Robbins et al. (in press) provide the following specific suggestions for management and conservation of the species:

### *Habitat Preservation.* -

Because habitat availability apparently is the major factor affecting cerulean warbler populations, large (possibly > 1,730 ac [700 ha]) tracts of mature, deciduous forests, primarily in lowland areas, should be preserved. However, better information regarding the minimum size of forest tracts necessary to support stable breeding populations of cerulean warblers is needed. In addition, younger hardwood stands adjacent to mature stands supporting cerulean warblers should be protected from timber harvest to ensure that habitat for these warblers is available in the future (Lynch 1981, quoted by Robbins et al. in press). Habitat supporting cerulean warblers can be preserved with conservation easements on, and purchases of, large forest tracts. Enforcement of existing wetland-protection regulations also should help protect lowland hardwood forests for cerulean warblers (Hands et al. 1989).

### *Habitat Management.* -

When preservation of cerulean warblers is one of the management priorities for a forest tract, habitat management consists mainly of protecting the site from timber harvest, preventing chemical contamination, and maintaining the natural hydrology. Planting of trees and protection of young trees on large, lowland tracts should provide habitat for the future (Hands et al. 1989).

### *Population Management.* -

More data on population trends, nest success, productivity, and mortality rates are needed to better assess the population status of the cerulean warbler. As many sites as possible supporting cerulean warblers should be surveyed throughout the northcentral U.S. during the breeding season. Ideally, surveys

should be conducted annually at selected sites

... Estimating mortality rates will be more difficult. Recaptures and resightings of banded birds may be the only method possible. Capture of these canopy-dwelling warblers will require the use of mist nets that can be moved up to, and down from, the canopy with pulleys (see Greenlaw and Swinebroad 1967, Humphrey et al. 1968) (Hands et al. 1989).

### *Predator Management.* -

Recommendations concerning predator management cannot be made until information regarding predators and the frequency of predation is available (Hands et al. 1989).

### *People Management.* -

Humans probably disturb cerulean warblers primarily by destroying and degrading mature forests. Thus, management plans for cerulean warblers should include programs to educate landowners and land managers of the habitat needs of cerulean warblers. These educational programs should stress the need for protection of large tracts of mature, deciduous, primarily lowland forests for cerulean warblers. (Hands et al. 1989).

### *Conservation Implications.* -

The continuing loss, fragmentation, and degradation of breeding and wintering habitat for the cerulean warbler emphasizes the importance of taking steps now to curb this decline. It is not likely that a massive conservation program specifically to benefit this warbler will be undertaken in the near future, but there is much to be done to promote habitat conservation that will benefit the cerulean warbler directly. By concentrating on protecting habitat that the cerulean warbler requires, we shall be benefitting many other neotropical migrants as well.

Because both Hamel's Tennessee study and studies of Robbins and colleagues in the Middle Atlantic states have shown the dependence of the cerulean warbler on large tracts of mature deciduous forest, and because these studies also showed that the great majority of summer records of this species in

*those states are on government-owned lands, we recommend (1) that government agencies at all levels be informed of the need to protect extensive tracts of mature deciduous forest under their control. We also recommend (2) that each state and province identify extensive tracts of mature forest not under government control, and explore ways of preserving the most important of these by purchase, easement, or other means; (3) that where appropriate, younger or smaller tracts be identified and included in long-term management plans so they will ultimately develop into extensive mature tracts; (4) that more be done to educate the public (at all age levels) on the importance of habitat protection; (5) that habitat protection legislation, especially as it relates to floodplain habitats, be improved and better enforced; and (6) that more research on the life history and habitat requirements of the cerulean warbler be encouraged. This research should include finding ways to supplement even-age management with retention of enough old-age trees to make managed timber lands attractive to cerulean warblers (Robbins et al. in press).*

The fact that many known populations of the species are already restricted to public land indicates that public land managers at the state and federal levels are the probable primary protectors of the future of this species. I believe that the tract size proposed by Hands et al. (1989) of 1,730 ac (700 ha) is a minimal estimate. Different studies in different areas have uniformly indicated that forest fragmentation is a significant issue in the protection of the species. That the minimal tract size has varied from region to region in the range of the species indicates that the land use context in different regions has a strong bearing on the operation of the forest fragmentation phenomenon as it affects cerulean warblers.

The current major land management activities that can be carried out on behalf of this species include: (1) the provision of large tracts of old forest, in rich situations rather than in marginal soil types, at several locations throughout the range of the species, and (2) forest management activities that are sensitive to the

fragmentation of existing tracts. Forest management that mimics the gap phase succession of eastern deciduous forests will more likely provide a continuous supply of habitat for this species than will even-aged management in large blocks.

## RESEARCH NEEDS

Few specific, quantitative data exist on the vegetation parameters associated with cerulean warbler breeding and wintering habitats. Few quantitative data also exist on the population structure, demography, productivity, and habitat utilization by the birds. Research needs are thus a substantial portion of the management and monitoring recommendations outlined above.

Hands et al. (1989) present a list of information needs that identifies the fact that at each stage of the annual cycle, and each life stage of cerulean warblers, information is lacking on the biology of cerulean warblers. Robbins et al. (in press) point out that because the species is a canopy dweller whose nests are difficult for humans to access, a life history study of the species has not been done to produce summary values of biological traits. An obvious recommendation is that such a study be supported, and that it include aspects of the monitoring and land management information needs outlined above, as well as traditional data on the life history of the species.

Cerulean warblers represent a kind of rarity that may become much more common in the future in North America--landscape-scale rarity in both the breeding and winter grounds. The species thus has considerable value as a model for the study of landscape scale change on the increase, decline, and redistribution of habitat and population. A number of studies have addressed various aspects of the fragmentation of forests. Study of this species in forest tracts under different landscape contexts, from primarily agricultural to primarily forested ones, will help elucidate the different features operative in the forest fragmentation effect, from the behavioral predisposition of the species, through the operation of edge effects on prey populations,

predator populations, populations of potential competitors, and the effects of nest parasitism.

Research on the behavior, biology, and control of brown-headed cowbirds will be an important part of the understanding of the distribution of breeding populations of cerulean warblers, and of many other host species as well. It is a reasonable hypothesis that the operation of forest fragmentation is a reflection of the penetration of cowbirds into tracts and the extinction of host populations resulting from the reduction of host reproduction below maintenance levels by such parasitism. Research on this topic will be a very useful benefit of study of cerulean warbler populations.

Regardless of monitoring, research, and management on the breeding grounds of the species, the future of cerulean warblers will not be secure until clear understanding of the winter biology, population distribution, and habitat utilization of the species has been determined, and appropriate protection strategies developed and implemented. In this way cerulean warblers are also a model species for the need for international cooperation in conservation.

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