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Birds

by
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Overview

Migratory bird populations are an international resource for which there is special federal responsibility. Moreover, birds are valued and highly visible components of natural ecosystems that may be indicators of environmental quality. Consequently, many efforts have been directed toward measuring and monitoring the condition of North America's migratory bird fauna. The task is not an easy one because the more than 700 U.S. species of migratory birds are highly mobile and may occur in the United States during only part of their annual cycle. Some species annually make round-trip migrations spanning thousands of kilometers or miles, others engage in short or irregular migrations of tens or hundreds of kilometers, and even resident species are capable of moving great distances over short intervals. One often cannot tell whether a bird observed at a given moment is a resident, a migrant, a visitor from another locality, or the same individual seen 10 minutes earlier.

Determining status and trends is further complicated by the fact that each of these species has its own patterns of distribution and abundance, and each species has populations that respond to different combinations of environmental factors. Finally, the sheer abundance of birds estimated at 20 billion individuals in North America at its annual late-summer peak (Robbins et al. 1966) may make it difficult to obtain accurate counts of common species, and the absolute abundance of some may mask important changes in their status.

Biologists have developed many different approaches to determining abundance and trends in abundance, and nearly all of the recognized census methods applicable to birds are represented by the articles in this section. Not surprisingly, trends among the large number of populations treated are mixed.

Results from the nationwide Breeding Bird Survey (Peterjohn et al., this section) and a portion of the large-scale Christmas Bird Count (Root and McDaniel, this section) show that some populations are declining, others increasing, and many show what appears to be normal fluctuations around a more or less stable average. Overall, approximately equal numbers of species appear to be increasing and decreasing over the past two to three decades. Groups of species with the most consistent declines are those characteristic of grassland habitats, apparently reflecting conversion of these habitats to other types of vegetative cover.

Waterfowl populations are monitored closely as a basis for regulating annual harvests at levels consistent with maintenance of populations. Goose populations (Rusch et al.,

Hestbeck's "Canada Geese," Hupp et al., all this section) have shown some impressive gains over the past decades, but most gains have been registered by large-bodied geese, with several smaller species and smaller subspecies of the highly variable Canada goose (*Branta canadensis*) having depressed populations.

Censusing and determining the status of natural Canada goose populations are made more difficult by the widespread introduction and establishment of resident goose populations, which breed outside the traditional Arctic nesting areas and mix with migratory populations on the wintering grounds.

Duck surveys address more than 30 species that might be legally hunted. Even though some species are stable or even increasing, many duck populations have declined in the past decade (Caithamer and Smith, this section). Biologists attribute these declines to losses of breeding and wintering habitats and a long period of drought in breeding areas. Among species receiving special emphasis, canvasbacks (*Aythya valisineria*; Hohman et al., this section) showed a complex pattern with regional changes in distribution and abundance, and pintails (*Anas acuta*; Hestbeck's "Decline of Northern Pintails," this section) showed a widespread and nearly consistent pattern of decline.

Results are preliminary, but two new census programs, the MAPS and BBIRD programs (Martin et al., this section), promise to provide much higher quality information on status and trends by measuring not only the presence of bird populations in breeding areas, but also their success. When fully operational, this approach may offer important clues regarding the causes of observed population changes.

Shorebirds are highly migratory, and status and trends of their populations are largely determined from observations made during periods in their life cycles in which birds congregate in limited breeding, staging, or migratory stopover areas. Populations of eastern (Harrington, this section) and western (Gill et al., this section) species show general patterns of decline, although some species, including those using inland areas, are too poorly studied to detect trends. Apparent dependence on critical breeding and staging areas suggests that populations of many species are vulnerable to habitat loss and disturbance.

Seabirds in the Pacific region (Carter et al., Hatch and Piatt, both this section) include many diverse species that respond differently to factors such as human proximity to nesting areas, oil spills, introduction of predators, depletion of fishery stocks, and availability of human refuse as food. Some species, including certain gulls, brown pelicans (*Pelecanus occidentalis*), and double-crested cormorants (*Phalacrocorax auritus*), have responded positively to recent changes in some areas, whereas others, including murrelets and murres (Family Alcidae) and kittiwakes (Genus *Rissa*), have shown declining trends. Populations of other species appear to fluctuate widely, and information for many species is insufficient to determine long-term trends.

Colonial waterbirds of the continental and east coast regions of the United States (Erwin, this section) show trends related to many of the same factors operating in the Pacific region, with some species recovering from past losses from pesticides while some other species that exploit human refuse are increasing dramatically. Populations of other species, especially certain terns, are declining, probably as a result of habitat loss and degradation or other kinds of human disturbance. Special efforts have been made to determine status and trends of the piping plover (*Charadrius melodus*; Haig and Plissner, this section), a species listed as endangered in certain parts of its range and as threatened in others.

Populations of raptors (Fuller et al., this section) are difficult to census, but ospreys (*Pandion haliaetus*), bald eagles (*Haliaeetus leucocephalus*), and peregrine falcons (*Falco peregrinus*) have increased in numbers as they recover from past effects of pesticides. Populations of most vultures, hawks, and owls are either poorly known or believed to be stable. Notable exceptions are California condors (*Gymnogyps californianus*; Pattee and Mesta, this section), the crested caracara (*Caracara plancus*; Layne, this section), and spotted owls (*Strix occidentalis*), all of which enjoy or have been considered for additional protection. Mortality factors of eagles (Franson et al., this section) have been monitored and, although these data do not directly measure population status, they do indicate trends in the kinds of factors that tend to depress population growth.

The wild turkey (*Meleagris gallopavo*; Dickson, this section) has shown dramatic increases in distribution and abundance in recent decades because of translocations, habitat restoration, and harvest control. Mourning doves (*Zenaida macroura*; Dolton, this section) have shown generally stable populations, although recent population declines in the western states are disturbing. Regional increases of ravens (*Corvus corax*) in the southwest (Boarman and Berry, this section) are primarily of concern because of their potential effects as predators on eggs and young of the desert tortoise (*Gopherus agassizii*).

Populations of severely endangered species, like the California condor (Pattee and Mesta, this section), the Mississippi sandhill crane (*Grus canadensis pulla*; Gee and Hereford, this

section), and the Puerto Rican parrot (*Amazona vittata*; Meyers, this section), are reasonably well known. Through censusing these species, biologists have tracked declines, often to a few individuals, and slow recoveries resulting from intensive management activities. Other rare species have populations that are depleted or vulnerable because of recent trends, but which can be censused with far less certainty. For example, willow flycatchers (*Empidonax traillii*; Sogge, this section) breed sparsely in parts of the Grand Canyon where exotic species have displaced natural riparian vegetation; likewise, the status of the red-cockaded woodpecker (*Picoides borealis*) appears closely tied to the decline of the longleaf pine (*Pinus palustris*) ecosystem (Costa and Walker, this section).

Broad-scale programs such as the Breeding Bird Survey, annual waterfowl surveys, and wintering surveys such as the Christmas Bird Count may provide information on status and trends for as many as 75% of U.S. bird species, at least to the extent that they would provide evidence of catastrophic declines. Remaining species may be censused only with difficulty and often with imprecision because they are secretive, rare, highly mobile, or occupy poorly accessible areas. Specialized surveys provide information on some of these groups but, as indicated by the articles in this section, they do so with varying degrees of success. Much work remains to be done on obtaining better information and developing better ways of interpreting available information on difficult-to-census species.

If any overall conclusion is possible on the wide array of information now available on status and trends of bird populations it is this: apparent stability for many species; increases in some species, many of which are generalists adaptable to altered habitats; and decreases in other species, many of which are specialists most vulnerable to habitat loss and degradation.

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Breeding Bird Survey: Population Trends 1966-92

by
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The North American Breeding Bird Survey (BBS) was begun in 1966 to collect standardized data on bird populations along more than 3,400 survey routes across the continental United States and southern Canada. The BBS has been used to document distributions and establish continental, regional, and local population trends for more than 250 species.

We summarize here survey-wide patterns in the 1966-92 population trend estimates for 245 species of birds observed on a minimum of 40 routes with a mean relative abundance of 1.0 bird per route. Survey-wide trend estimates are also summarized for six groupings of birds, providing insight into broad geographical patterns of population trends of North American birds.

Methods

The BBS routes are located along secondary roads and surveyed each year during the peak of the breeding season by observers competent in bird identification. Each route is 39.4 km (24.5 mi) long, with 50 stops placed at 0.8-km (0.5-mi) intervals (Robbins et al. 1986). To estimate population change, we used a procedure called route regression, described in greater detail by Geissler and Sauer (1990).

We examined population change in several ways. First, we estimated overall population change for individual species over the entire survey area. Second, we looked for temporal and geographic patterns in individual bird species (e.g., Sauer and Droege 1990).

Additionally, we analyzed overall patterns of population change for several species of particular management interest. Groups of birds were defined by migration status (nonmigratory, short-distance, and Neotropical migrants) or by breeding habitat (grassland, shrubland, or woodland; *see* also Peterjohn and Sauer 1993). For each group, we determined the percentage of species with positive (> 0) trends. If population change is not consistent within the group, about half (50%) of the species should show positive trends. Clearly, some species will show very significant declines (or increases) over the interval, and these species can be identified in the Appendix. However, the percentage of species with positive population trends is a convenient summary of information from all species within the group to demonstrate overall trend patterns.

Finally, to display regional patterns of population change, we calculated the mean trend for the species in each group for each survey route. We used an Arc/Info geographic information system to summarize and display geographic patterns of population change (Isaaks and Srivastava 1989; ESRI 1992).

Trends

Of the 245 species considered, 130 have negative trend estimates, 57 of which exhibit significant declines. Species with negative trend estimates are found in all families, but they are especially prevalent among the mimids (mockingbirds and thrashers) and sparrows. A total of 115 species exhibits positive trends, 44 of

which are significant increases. Flycatchers and warblers have the largest proportions of species with increasing populations.

The percentage of increasing species within each group of species having shared life-history traits is summarized in the Table. The most consistent declines are by grassland birds; only 18% have increasing population trends. These declines are most widespread in eastern North America, where few grassland species breed (Fig. 1). Declining populations are also prevalent across the Great Plains, which includes the breeding ranges of most grassland birds. The pattern within western North America is mixed, except for regions of declines along the Pacific coast.

A significant proportion of shrubland and old-field bird species also exhibits population declines (Table). As with grassland birds, regions with declines are most prevalent in eastern North America as well as in the southern Great Plains from Kansas and Missouri south to Texas (Fig. 2). Shrubland species appear to be generally increasing in western North America.

A majority of woodland bird populations is increasing across most of the continent (Fig. 3). Decreasing populations prevail in a few regions, such as along the Appalachians from West Virginia to northern Alabama, from Arkansas across central Texas, and along the Pacific coast from Oregon to central California. Woodland birds, however, are increasing in more areas than either grassland or early successional species.

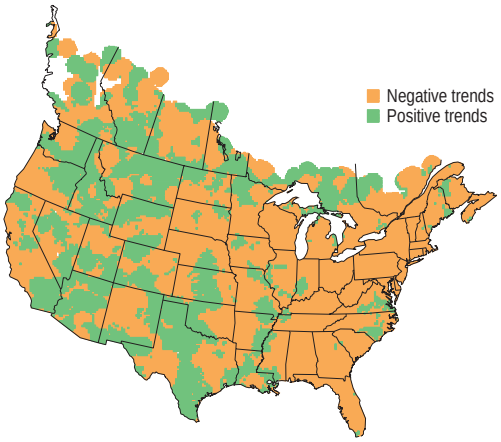

Fig. 1. Geographic patterns in the mean trends for grassland bird species during 1966-92.

Table. Percentage of species with increasing populations for six groups of birds having shared life-history traits. The *P* value indicates the probability that the percentage differs from 50%.

Group	No. of species in each group	Increasing (%)	*P*
Breeding habitats			
Grassland	17	18	0.01
Shrubland	58	34	0.02
Woodland	80	59	0.15
Migration			
Short distance	69	42	0.23
Nonmigratory	41	41	0.35
Neotropical	98	50	0.92
All species	237	47	0.36

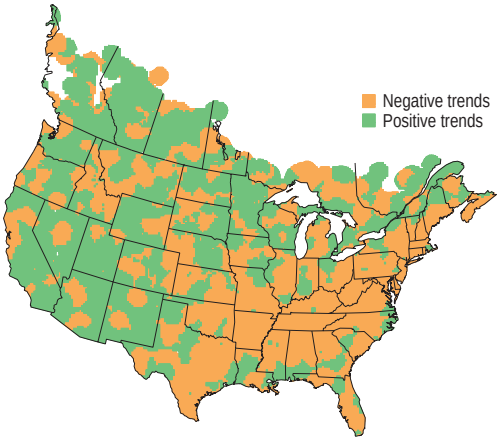

Fig. 2. Geographic patterns in the mean trends for shrubland and old-field bird species during 1966-92.

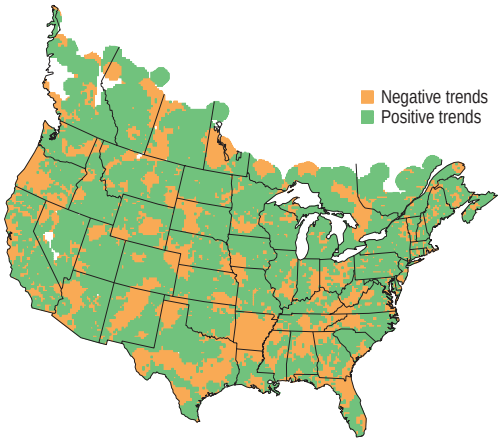

Fig. 3. Geographic patterns in the mean trends for woodland bird species during 1966-92.

Neotropical migrants have received considerable attention in recent years, yet as many species have increased as have decreased during 1966-92 (Table). A region with apparently declining populations extends from the southern Great Plains across the southeastern states and along the Appalachian Mountains to southern New England (Fig. 4). Increasing mean populations prevail across the northern Great Plains and throughout much of western North America. The pattern of population decline in the eastern United States noted by Robbins et al. (1989) occurred after 1978 and is not reflected in these long-term trends.

Short-distance migrants and permanent residents have slightly greater percentages of decreasing species (Table). Both groups have negative mean trends in the southeastern states and from the lower Great Lakes into the Appalachian Mountains, but the patterns elsewhere are mixed (Figs. 5, 6).

These results indicate that grassland and shrubland birds are experiencing the most consistent and widespread declines of any group of species. Whenever possible, appropriate conservation measures should be undertaken to enhance the population trends of these species. While the BBS data indicate the population

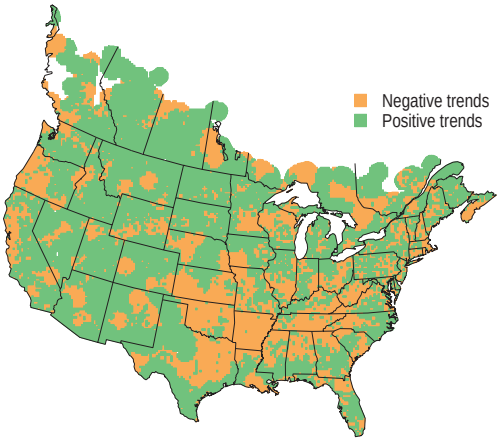

Fig. 4. Geographic patterns in the mean trends for Neotropical migrant bird species during 1966-92.

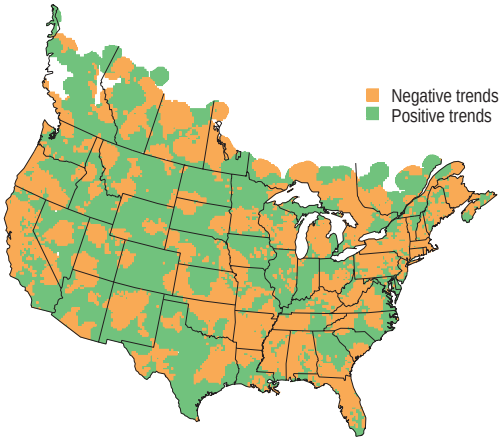

Fig. 5. Geographic patterns in the mean trends for short-distance migrant bird species during 1966-92.

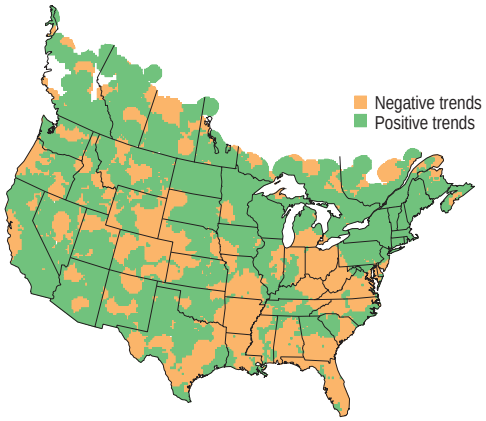

Fig. 6. Geographic patterns in the mean trends for permanent resident bird species during 1966-92.

trends for breeding birds, these data are not designed to identify the factors responsible for these trends. To understand how bird populations are responding to the changing habitat conditions in North America, additional studies are needed that would combine the BBS results with regional data on land-use changes, weather conditions, and other variables.

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Appendix. Population trends of birds from the North American Breeding Bird Survey. To appear in this list, the species must have been seen on > 40 routes at an average count of > 1 bird/route. We present trends (%/year), probability (*P*), and the number of routes on which the species was seen. *See* Peterjohn and Sauer 1993 for group classification.

Species	Scientific name	Trend	*P* *	No. of routes
American white pelican	*Pelecanus erythrorhynchos*	3.60	0.00	152
Great egret	*Casmerodius albus*	1.5	ns	513
Little blue heron	*Egretta caerulea*	-1.45	ns	429
Cattle egret	*Bubulcus ibis*	2.09	ns	475
White ibis	*Eudocimus albus*	3.17	ns	173
White-faced ibis	*Plegadis chihi*	32.27	0.01	61
Canada goose	*Branta canadensis*	7.05	0.00	1,090
Mottled duck	*Anas fulvigula*	-5.27	0.03	64
Mallard	*A. platyrhynchos*	0.98	ns	1,890
Northern pintail	*A. acuta*	-5.65	0.00	502
Blue-winged teal	*A. discors*	-0.92	ns	814
Northern shoveler	*A. clypeata*	0.18	ns	379
Gadwall	*A. strepera*	3.76	0.00	389
Lesser scaup	*Aythya affinis*	2.08	ns	263
Red-breasted merganser	*Mergus serrator*	-9.57	0.02	53
Black vulture	*Coragyps atratus*	1.72	ns	540
Turkey vulture	*Cathartes aura*	0.37	ns	1,691
Ring-necked pheasant	*Phasianus colchicus*	-1.24	0.10	1,263
Northern bobwhite	*Colinus virginianus*	-2.43	0.00	1,338
Scaled quail	*Callipepla squamata*	-3.31	0.00	104
Gambel's quail	*C. gambelii*	0.90	ns	82
California quail	*C. californica*	-0.04	ns	264
Mountain quail	*Oreortyx pictus*	1.37	ns	112
American coot	*Fulica americana*	-0.51	ns	620

Species	Scientific name	Trend	*P* *	No. of routes
Sandhill crane	*Grus canadensis*	4.30	0.00	259
Killdeer	*Charadrius vociferus*	-0.38	ns	2,692
Black-necked stilt	*Himantopus mexicanus*	0.63	ns	119
Willet	*Catoptrophorus semipalmatus*	-0.72	ns	295
Upland sandpiper	*Bartramia longicauda*	3.28	0.00	687
Long-billed curlew	*Numenius americanus*	-1.61	ns	234
Marbled godwit	*Limosa fedoa*	0.71	ns	188
Common snipe	*Gallinago gallinago*	0.14	ns	1,011
Laughing gull	*Larus atricilla*	6.01	0.00	125
Franklin's gull	*L. pipixcan*	-5.95	ns	231
Ring-billed gull	*L. delawarensis*	7.43	0.02	684
California gull	*L. californicus*	-1.27	ns	230
Herring gull	*L. argentatus*	-2.06	0.09	474
Glaucous-winged gull	*L. glaucescens*	3.85	0.09	40
Great black-backed gull	*L. marinus*	-1.47	ns	125
Black tern	*Chlidonias niger*	-4.51	0.00	368
Rock dove	*Columba livia*	1.04	0.06	2,255
Band-tailed pigeon	*C. fasciata*	-3.69	0.00	189
White-winged dove	*Zenaida asiatica*	0.03	ns	78
Mourning dove	*Z. macroura*	0.02	ns	2,726
Common ground dove	*Columbina passerina*	-3.13	0.01	194
Yellow-billed cuckoo	*Coccyzus americanus*	-1.30	0.00	1,637
Lesser nighthawk	*Chordeiles acutipennis*	5.08	0.03	118
Common nighthawk	*C. minor*	-0.34	ns	1,609

Species	Scientific name	Trend	*P* *	No. of routes
Chuck-will's-widow	*Caprimulgus carolinensis*	-0.78	ns	522
Black swift	*Cypseloides niger*	1.61	ns	79
Chimney swift	*Chaetura pelagica*	-0.84	0.08	1,789
White-throated swift	*Aeronautes saxatalis*	-3.38	ns	189
Broad-tailed hummingbird	*Selasphorus platycercus*	0.42	ns	115
Rufous hummingbird	*S. rufus*	-3.38	0.00	188
Red-headed woodpecker	*Melanerpes erythrocephalus*	-1.84	0.00	1,236
Acorn woodpecker	*M. formicivorus*	0.98	ns	138
Golden-fronted woodpecker	*M. aurifrons*	-1.86	ns	56
Red-bellied woodpecker	*M. carolinus*	0.59	ns	1,246
Yellow-bellied sapsucker	*Sphyrapicus varius*	-0.85	ns	605
Nuttall's woodpecker	*Picoides nuttallii*	1.44	ns	95
Downy woodpecker	*P. pubescens*	0.14	ns	2,214
Yellow-shafted flicker	*Colaptes auratus*	-2.75	0.00	2,062
Red-shafted flicker	*C. cafer*	-0.87	ns	689
Olive-sided flycatcher	*Contopus borealis*	-2.52	0.00	736
Western wood-pewee	*C. sordidulus*	-0.39	ns	637
Eastern wood-pewee	*C. virens*	-1.64	0.00	1,719
Yellow-bellied flycatcher	*Empidonax flaviventris*	3.58	0.01	263
Acadian flycatcher	*E. virescens*	0.50	ns	854
Alder flycatcher	*E. alnorum*	1.30	0.04	788
Willow flycatcher	*E. traillii*	-0.62	ns	1,152
Least flycatcher	*E. minimus*	-0.55	ns	1,150
Hammond's flycatcher	*E. hammondii*	1.50	ns	221
Dusky flycatcher	*E. oberholseri*	0.72	ns	265
Pacific-slope flycatcher	*E. difficilis*	1.47	ns	218
Eastern phoebe	*Sayornis phoebe*	0.64	ns	1,650
Ash-throated flycatcher	*Myiarchus cinerascens*	2.38	0.01	370
Great crested flycatcher	*M. crinitus*	0.03	ns	1,804
Brown-crested flycatcher	*M. tyrannulus*	6.15	0.00	47
Cassin's kingbird	*Tyrannus vociferans*	-1.74	ns	138
Western kingbird	*T. verticalis*	1.51	0.01	942
Eastern kingbird	*T. tyrannus*	-0.10	ns	2,267
Scissor-tailed flycatcher	*T. forficatus*	-0.08	ns	244
Horned lark	*Eremophila alpestris*	-0.65	ns	1,750
Purple martin	*Progne subis*	0.71	ns	1,623
Tree swallow	*Tachycineta bicolor*	1.27	0.04	1,707
Violet-green swallow	*T. thalassina*	0.76	ns	511
Northern rough-winged swallow	*Stelgidopteryx serripennis*	0.95	ns	2,119
Bank swallow	*Riparia riparia*	-0.48	ns	1,318
Cliff swallow	*Hirundo pyrrhonota*	0.98	ns	1,737
Barn swallow	*H. rustica*	0.37	ns	2,701
Gray jay	*Perisoreus canadensis*	-1.28	ns	350
Steller's jay	*Cyanocitta stelleri*	0.39	ns	328
Blue jay	*C. cristata*	-1.81	0.00	1,986
Scrub jay	*Aphelocoma coerulescens*	1.27	0.04	272
Pinyon jay	*Gymnorhinus cyanocephalus*	-1.65	ns	132
Black-billed magpie	*Pica pica*	-1.34	0.05	577
American crow	*Corvus brachyrhynchos*	0.85	0.06	2,578
Fish crow	*C. ossifragus*	2.93	0.01	466
Chihuahuan raven	*C. cryptoleucus*	-2.48	ns	87
Common raven	*C. corax*	3.66	0.00	1,202
Black-capped chickadee	*Parus atricapillus*	1.89	0.00	1,433
Carolina chickadee	*P. carolinensis*	-0.67	ns	862
Mountain chickadee	*P. gambeli*	0.07	ns	291
Chestnut-backed chickadee	*P. rufescens*	-1.54	ns	133
Plain titmouse	*P. inornatus*	-2.30	0.01	180
Tufted titmouse	*P. bicolor*	0.64	ns	1,289
Black-crested titmouse	*P.b. atricristatus*	2.06	0.03	64
Verdin	*Auriparus flavicep*	-1.38	ns	97
Bushtit	*Psaltriparus minimus*	-1.13	ns	250
Red-breasted nuthatch	*Sitta canadensis*	2.48	0.00	872
Brown-headed nuthatch	*S. pusilla*	-1.30	ns	290
Cactus wren	*Campylorhynchus brunneicapillus*	-0.89	ns	136
Rock wren	*Salpinctes obsoletus*	-1.68	0.04	509
Carolina wren	*Thryothorus ludovicianus*	1.01	0.03	1,118
Bewick's wren	*Thryomanes bewickii*	-0.35	ns	594
House wren	*Troglodytes aedon*	1.55	0.00	1,924
Winter wren	*T. troglodytes*	2.25	ns	659
Golden-crowned kinglet	*Regulus satrapa*	-0.01	ns	541
Ruby-crowned kinglet	*R. calendula*	-1.31	ns	656

Species	Scientific name	Trend	*P* *	No. of routes
Blue-gray gnatcatcher	*Polioptila caerulea*	1.03	ns	1,233
Black-tailed gnatcatcher	*P. melanura*	-0.22	ns	57
Eastern bluebird	*Sialia sialis*	2.52	0.00	1,633
Mountain bluebird	*S. currucoides*	0.56	ns	422
Veery	*Catharus fuscescens*	-1.06	0.06	964
Gray-cheeked thrush	*C. minimus*	-4.46	ns	43
Swainson's thrush	*C. ustulatus*	0.00	ns	707
Hermit thrush	*C. guttatus*	2.10	0.01	912
Wood thrush	*Hylocichla mustelina*	-1.88	0.00	1,510
American robin	*Turdus migratorius*	1.03	0.01	2,588
Varied thrush	*Ixoreus naevius*	2.16	0.06	148
Wrentit	*Chamaea fasciata*	-1.39	ns	113
Gray catbird	*Dumetella carolinensis*	-0.42	ns	1,941
Northern mockingbird	*Mimus polyglottos*	-0.98	0.03	1,694
Sage thrasher	*Oreoscoptes montanus*	1.16	ns	244
Brown thrasher	*Toxostoma rufum*	-1.19	0.01	1,917
Curve-billed thrasher	*T. curvirostre*	-3.59	0.00	100
California thrasher	*T. redivivum*	-4.06	0.05	83
Sprague's pipit	*Anthus spragueii*	-3.52	0.02	140
Cedar waxwing	*Bombycilla cedrorum*	2.36	0.00	1627
Phainopepla	*Phainopepla nitens*	2.53	0.05	104
Loggerhead shrike	*Lanius ludovicianus*	-3.20	0.00	1364
European starling	*Sturnus vulgaris*	-0.99	0.02	2727
White-eyed vireo	*Vireo griseus*	-0.15	ns	945
Solitary vireo	*V. solitarius*	3.28	0.00	954
Warbling vireo	*V. gilvus*	1.31	0.01	1740
Philadelphia vireo	*V. philadelphicus*	1.50	ns	191
Red-eyed vireo	*V. olivaceus*	1.39	0.01	2020
Tennessee warbler	*Vermivora peregrina*	4.21	ns	341
Orange-crowned warbler	*V. celata*	-0.71	ns	346
Nashville warbler	*V. ruficapilla*	1.35	ns	673
Northern parula	*Parula americana*	0.82	ns	970
Yellow warbler	*Dendroica petechia*	0.94	0.05	2161
Chestnut-sided warbler	*D. pensylvanica*	-0.60	ns	788
Magnolia warbler	*D. magnolia*	2.80	0.00	527
Cape May warbler	*D. tigrina*	2.95	ns	239
Myrtle warbler	*D. coronata*	1.41	0.09	575
Audubon's warbler	*D.c. auduboni*	0.08	ns	386
Black-throated gray warbler	*D. nigrescens*	2.32	0.07	190
Townsend's warbler	*D. townsendi*	1.63	ns	145
Hermit warbler	*D. occidentalis*	0.79	ns	82
Black-throated green warbler	*D. virens*	-0.45	ns	637
Blackburnian warbler	*D. fusca*	0.87	ns	511
Pine warbler	*D. pinus*	2.12	0.00	797
Prairie warbler	*D. discolor*	-2.15	0.00	773
Bay-breasted warbler	*D. castanea*	-0.04	ns	216
Blackpoll warbler	*D. striata*	-0.33	ns	178
Black-and-white warbler	*Mniotilta varia*	0.91	ns	1126
American redstart	*Setophaga ruticilla*	-0.58	ns	1299
Ovenbird	*Seiurus aurocapillus*	0.55	ns	1278
Northern waterthrush	*S. noveboracensis*	0.49	ns	615
Kentucky warbler	*Oporornis formosus*	-0.77	ns	685
Mourning warbler	*O. philadelphia*	0.15	ns	538
MacGillivray's warbler	*O. tolmiei*	-0.58	ns	309
Common yellowthroat	*Geothlypis trichas*	-0.48	ns	2361
Hooded warbler	*Wilsonia citrina*	1.49	ns	608
Wilson's warbler	*W. pusilla*	0.53	ns	525
Canada warbler	*W. canadensis*	-0.73	ns	504
Yellow-breasted chat	*Icteria virens*	-0.43	ns	1273
Summer tanager	*Piranga rubra*	-0.19	ns	761
Scarlet tanager	*P. olivacea*	0.22	ns	1257
Western tanager	*P. ludoviciana*	-0.31	ns	472
Northern cardinal	*Cardinalis cardinalis*	-0.21	ns	1591
Pyrrhuloxia	*C. sinuatus*	-0.73	ns	61
Rose-breasted grosbeak	*Pheucticus ludovicianus*	-0.19	ns	1146
Black-headed grosbeak	*P. melanocephalus*	-0.32	ns	509
Blue grosbeak	*Guiraca caerulea*	1.86	0.00	1014
Lazuli bunting	*Passerina amoena*	0.14	ns	417
Indigo bunting	*P. cyanea*	-0.57	ns	1725
Painted bunting	*P. ciris*	-3.21	0.00	269
Dickcissel	*Spiza americana*	-1.58	0.02	791
Green-tailed towhee	*Pipilo chlorurus*	0.41	ns	212
Rufous-sided towhee	*P. erythrophthalmus*	-1.99	0.00	1951

Species	Scientific name	Trend	*P* *	No. of routes
California towhee	*P. californicus*	-0.22	ns	113
Brown towhee	*P. fuscus*	-2.67	0.00	83
Cassin's sparrow	*Aimophila cassinii*	-2.85	0.00	171
Chipping sparrow	*Spizella passerina*	-0.04	ns	2,300
Clay-colored sparrow	*S. pallida*	-1.31	0.02	444
Brewer's sparrow	*S. breweri*	-3.68	0.00	376
Field sparrow	*S. pusilla*	-3.25	0.00	1,581
Vesper sparrow	*Pooecetes gramineus*	-0.25	ns	1,488
Lark sparrow	*Chondestes grammacus*	-3.42	0.00	935
Black-throated sparrow	*Amphispiza bilineata*	-3.78	0.02	225
Sage sparrow	*A. belli*	-2.43	ns	210
Lark bunting	*Calamospiza melanocorys*	-2.86	0.03	359
Savannah sparrow	*Passerculus sandwichensis*	-0.57	ns	1,461
Baird's sparrow	*Ammodramus bairdii*	-1.52	ns	134
Grasshopper sparrow	*A. savannarum*	-4.48	0.00	1,479
Fox sparrow	*Passerella iliaca*	0.44	ns	224
Song sparrow	*Melospiza melodia*	-0.80	0.09	2,079
Lincoln's sparrow	*M. lincolnii*	3.99	0.02	420
Swamp sparrow	*M. georgiana*	0.50	ns	783
White-throated sparrow	*Zonotrichia albicollis*	-1.44	0.01	635
White-crowned sparrow	*Z. leucophrys*	-1.91	0.01	274
Slate-colored junco	*Junco hyemalis*	-0.47	ns	545
Oregon junco	*J.h. oregonus*	-1.23	0.08	341
Gray-headed junco	*J.h. caniceps*	2.04	ns	50
McCown's longspur	*Calcarius mccownii*	8.32	0.00	68
Chestnut-collared longspur	*C. ornatus*	0.62	ns	153
Bobolink	*Dolichonyx oryzivorus*	-1.33	0.01	1,147
Red-winged blackbird	*Agelaius phoeniceus*	-1.06	0.01	2,760
Tricolored blackbird	*A. tricolor*	4.83	ns	69
Eastern meadowlark	*Sturnella magna*	-2.18	0.00	1,742
Western meadowlark	*S. neglecta*	-0.56	ns	1,334
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	1.53	ns	649
Brewer's blackbird	*Euphagus cyanocephalus*	-1.15	0.06	1,006
Great-tailed grackle	*Quiscalus mexicanus*	7.40	0.00	198
Boat-tailed grackle	*Q. major*	2.52	0.05	118
Common grackle	*Q. quiscula*	-1.44	0.00	2,196
Bronzed cowbird	*Molothrus aeneus*	-1.12	ns	55
Brown-headed cowbird	*M. ater*	-0.88	0.06	2,780
Orchard oriole	*Icterus spurius*	-1.38	0.03	1,313
Baltimore oriole	*I. galbula*	0.26	ns	1,594
Bullock's oriole	*I.g. bullockii*	-0.81	ns	614
Scott's oriole	*I. parisorum*	2.26	ns	113
Pine grosbeak	*Pinicola enucleator*	6.36	0.01	152
Purple finch	*Carpodacus purpureus*	-1.19	0.05	921
Cassin's finch	*C. cassinii*	1.27	ns	235
House finch	*C. mexicanus*	-0.14	ns	1,420
Red crossbill	*Loxia curvirostra*	2.13	ns	438
White-winged crossbill	*L. leucoptera*	-5.52	0.09	155
Pine siskin	*Carduelis pinus*	0.38	ns	778
Lesser goldfinch	*C. psaltria*	-1.22	ns	281
American goldfinch	*C. tristis*	-1.06	0.03	2,165
Evening grosbeak	*Coccothraustes vespertinus*	-0.37	ns	596
House sparrow	*Passer domesticus*	-1.65	0.00	2,557

*ns--not significant.

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Winter Population Trends of Selected Songbirds

by
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Many studies have found significant changes, primarily declines, in populations of breeding birds throughout the United States. Most studies have focused on birds that migrate to the Neotropics for winter. Speculations about causes of observed declines have primarily implicated habitat fragmentation and loss (e.g., deforestation) in Central and South America. The National Audubon Society's Christmas Bird Counts (CBC), begun in the winter of 1900-01, provide the data needed to discern consistent population trends in birds wintering throughout the United States.

For this study we used the CBC data to examine population trends of songbirds with ranges that apparently are limited by lower temperatures in the North. We chose these species to track populations of birds that could be in peril in the future. These birds potentially will be more quickly affected by changing climate than other birds, and we need baseline information on them to document possible consequences of global climatic change. The species that are indeed declining need to be monitored because the possible synergistic effects of declining populations and changing climate could result in local and even regional extinctions.

Methods

We examined 30 years of CBC data (winters of 1959-60 to 1988-89) for 50 songbirds whose northern range edges are associated with January minimum temperatures (Root 1988b). For each songbird species or subspecies at each count site, we calculated the number of individuals seen per counting effort (e.g., hours of observation). Yearly averages for each of the conterminous states were determined from these values for each species. Data were used from all count sites that were censused at least 25 of the 30 years. For details on the method we used to calculate population trends, *see* Geissler and Noon (1981) and contact us. All of our conclusions rest on very conservative analyses.

Trends

Of the 50 songbirds examined, 27 (54%) exhibited a statistically and biologically significant trend in at least one state (Fig. 1). Of these 27 species, 16 (59%) had populations declining in more states than states in which they were increasing; 12 exhibited only declines and 4 had a population increase in at least one state. Ten (37%) of the 27 species had populations increasing in more states than states exhibiting declines, with 7 exhibiting only population increases. One (4%) species had populations increasing and decreasing in the same number of states.

In general, the populations of birds that eat seeds from grasses and forbs (e.g., sparrows and meadowlarks) seem to be declining more frequently than those of birds that eat seeds from shrubs and trees, or berries (e.g., tufted titmouse [*Parus bicolor*] and cedar waxwing [*Bombycilla*

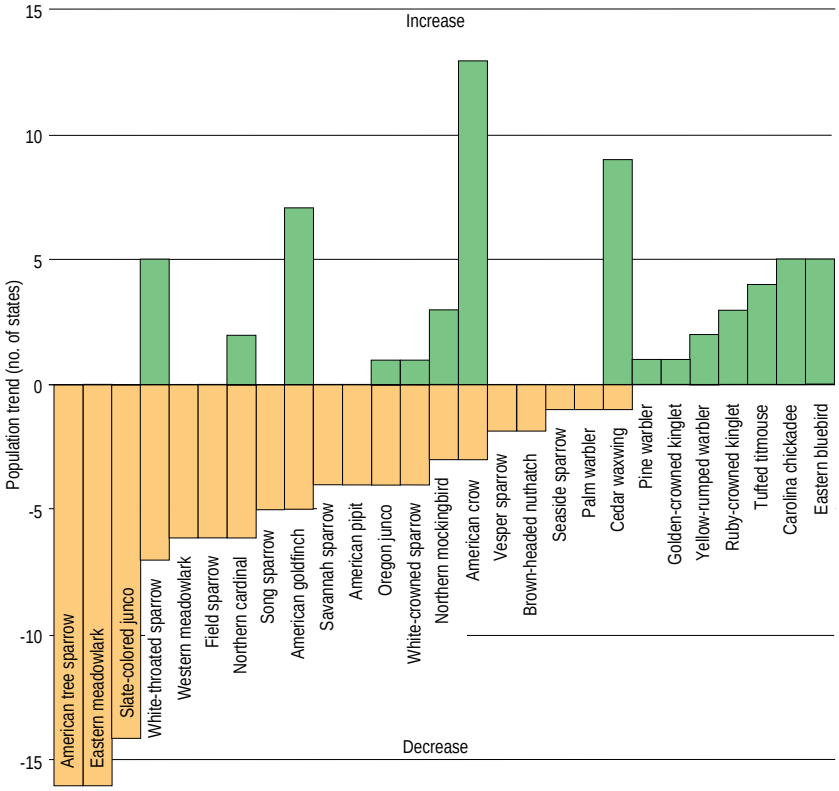

Fig. 1. Number of states with population trends either declining or increasing for 27 songbirds.

cedrorum]) (Fig. 1). This situation may be due to the fact that the grassland and early successional habitats are being modified, while ornamental fruiting bushes, shrubs, and trees planted in urbanized areas may be benefiting the increasing species (Beddall 1963). The explanation, however, is certainly more complex than this, given that some birds do not fit the pattern. For example, the American pipit (*Anthus rubescens*), which eats berries, crustaceans, and mollusks (Ehrlich et al. 1988), is decreasing in four states and increasing in none (Fig. 1).

To evaluate the areas of the conterminous states showing increases or decreases in their bird populations, we counted the number of species showing a population change in each state and then calculated the percentage with respect to the number of the 27 species occurring in each state (Fig. 2). A total of 24 (50%) of the states has greater than 5% of these wintering bird species showing positive population trends, while 32 (67%) show declines of similar magnitudes.

Mapping the percentages (Fig. 3) indicates that the largest increase is in South Carolina, with the far western states, those in the north-central region, and a scattering of states in the eastern portion of the conterminous states showing positive population trends.

The largest decreases (Fig. 3) are in South Carolina, Georgia, Florida, Alabama, Louisiana, and Delaware. The Pacific states, those in the

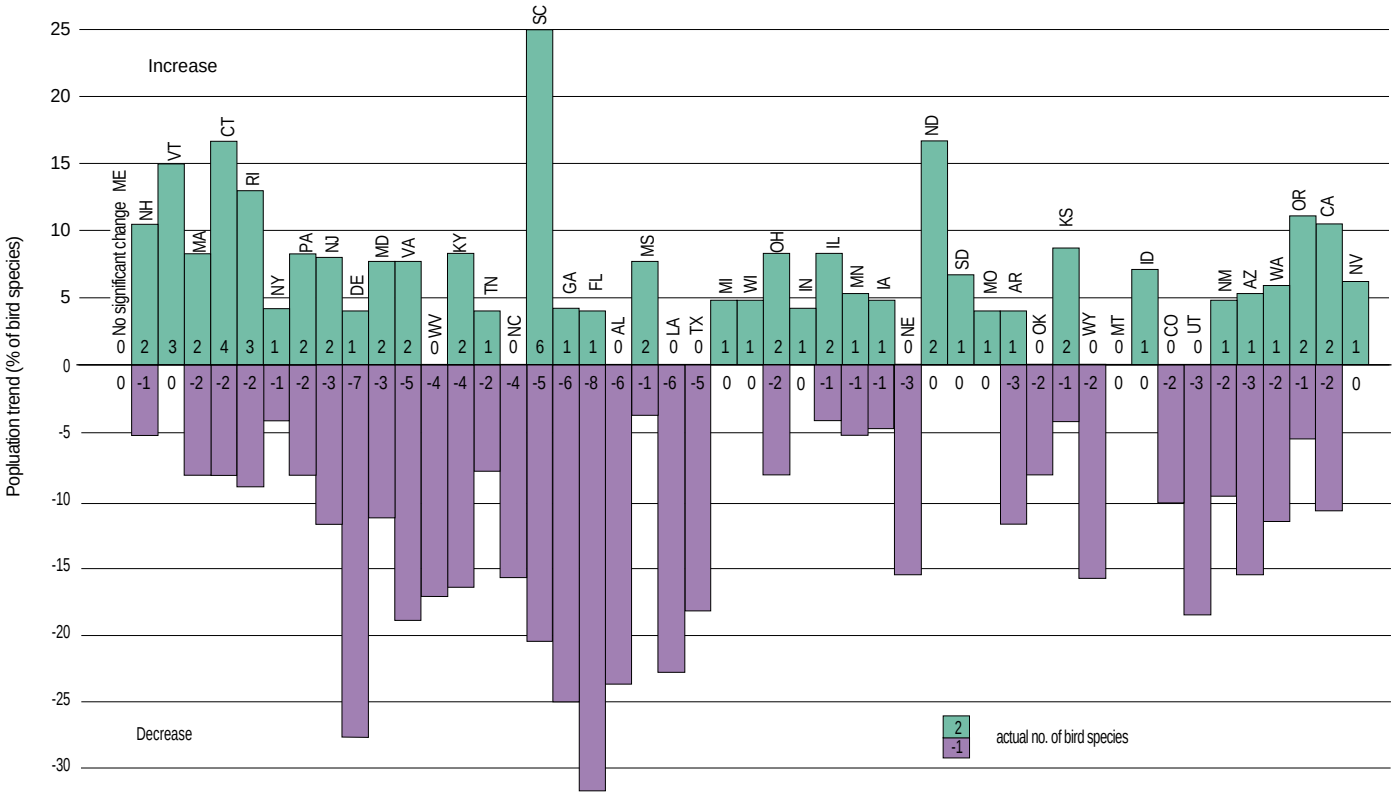

Fig. 2. Number and percentage of 27 birds showing declining and increasing population trends.

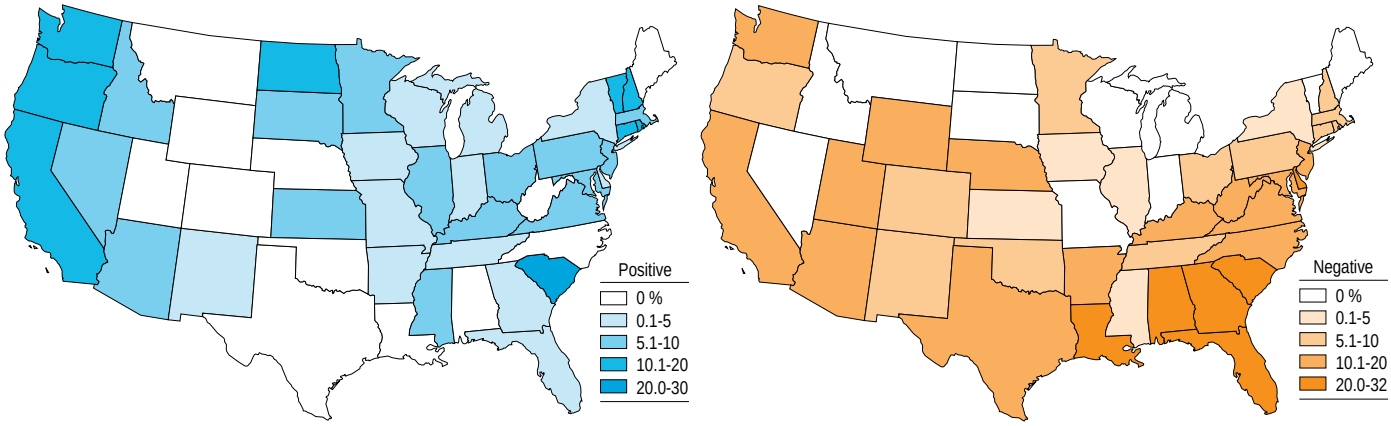

Fig. 3. Percentage of 27 birds showing positive and negative trends.

Great Plains, and the southeastern portion of the conterminous states generally show the greatest declines, though the actual reasons for these population changes will need to be examined in more detail. Certainly, the pattern of extensive declines in most of the southern coastal states is quite alarming.

Additionally, regions of the country that could be particularly influenced by global climatic change are the southern coasts (because of increased storms and degradation of coastal wetlands; IPCC 1990), and the Great Plains (owing to a significant decline in soil moisture; Leatherman 1992). Hence, the populations of birds in these areas need to be closely monitored to ensure preservation actions are taken before the combined effects of population declines and climate change result in extinctions. More studies and monitoring are warranted to understand the possible consequences of these patterns.

The analyses presented here can also be used to investigate population trends of target species across the country. Compare, for instance, the trends by state for the American tree sparrow (*Spizella arborea*; one of the most declining birds examined) and the cedar waxwing (one of the most increasing birds) with maps of their winter range and abundance patterns (Root 1988a). This comparison reveals that significant population trends, whether positive or negative, seem to occur primarily along these species' northern range boundaries and in many coastal states. Such analyses could help target specific regions of the country where population trends of key (e.g., threatened) species need watching.

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Breeding Productivity and Adult Survival in Nongame Birds

Populations of many North American landbirds, including forest-inhabiting species that winter in the Neotropics, seem to be declining (Robbins et al. 1989; Terborgh 1989). These declines have been identified through broad-scale, long-term survey programs that identify changes in abundance of species, but provide little information about causes of changes in abundance or the health of specific populations in different geographic locations.

Population health is a measure of a population's ability to sustain itself over time as determined by the balance between birth and death rates. Indices of population size do not always provide an accurate measure of population health because population size can be maintained in unhealthy populations by immigration of recruits from healthy populations (Pulliam 1988). Poor population health across many populations in a species eventually results in the decline of that species. Early detection of population declines allows managers to correct problems before they are critical and widespread.

Demographic data (breeding productivity and adult survival) provide the kind of early warning signal that allows detection of

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unhealthy populations in terms of productivity or survival problems (Martin and Guepel 1993). In addition, demographic data can help determine whether population declines are the result of low breeding productivity or low survival in migration or winter. Breeding productivity data also can help identify habitat conditions associated with successful and failed breeding attempts. Such information is critical for developing habitat- and land-management practices that will maintain healthy bird populations (Martin 1992). Here, we provide examples of the kinds of information that can be obtained by broad-scale demographic studies.

Demographic Programs

The Monitoring Avian Productivity and Survivorship (MAPS) and Breeding Biology Research and Monitoring Database (BBIRD) programs were developed to gather the demographic data needed to provide early and locality-specific warning signals of population problems. MAPS uses large, stationary mistnets to capture and examine young and adult birds for between-year changes and to determine long-term trends in adult population size, productivity, and adult survival. BBIRD locates and monitors bird nests to study changes in nesting success, determine causes of nesting failure (e.g., weather, habitat, nest predation, or nest parasitism), and identify habitat conditions associated with successful reproduction. Though both programs are new, they are growing rapidly. We present example data to demonstrate initial results and burgeoning potential of these programs for the future.

MAPS

Initiated in 1989 and coordinated by The Institute for Bird Populations, MAPS is a cooperative effort among federal and state agencies, private organizations, and bird banders to operate a standardized continent-wide network of mist-netting and banding stations during the breeding season (DeSante 1992; DeSante et al. 1993a, 1993b). A typical MAPS station involves about ten 12-m (39-ft) mistnets over a 20-ha (49-acre) area. All birds captured throughout the breeding season are identified to species, age, and sex, and are banded with U.S. Fish and Wildlife Service bands.

As of 1992, 170 stations were in operation and more than 94,000 captures of more than 200 bird species were recorded. The number of adult birds captured is used as an index of adult population size while the proportion of young provides an index of postfledgling productivity (Baillie et al. 1993).

BBIRD

The BBIRD program, initiated in 1992, provides detailed information on nesting productivity and habitat needs of nongame birds at a national scale. BBIRD is a cooperative effort among biologists studying nesting productivity at local sites across the country. Participants follow a standard field protocol to obtain raw data on nesting productivity, causes of reproductive failure, vegetation measures at several spatial scales, and point counts (bird counts). Data from each local site are overseen by individual independent investigators who can obtain comparative information from other sites. In addition, overview analyses to identify national and regional trends are conducted at the Montana Cooperative Wildlife Research Unit.

BBIRD study sites are in large forested blocks to minimize fragmentation effects and provide baseline information on productivity in undisturbed habitats as well as in auxiliary sites that have no habitat restrictions (e.g., grazed, fragmented, or logged sites). The BBIRD program now includes 23 sites in 17 states. Over 8,000 nests of more than 150 bird species were monitored during the first 2 years of the program.

Variation in Productivity

The data provided by MAPS and BBIRD suggest that weather may be an important influence on population dynamics at large and even continental scales. Prior data from constant-effort mist-netting in scrub habitat on the west coast have suggested that avian productivity may peak during average weather conditions and may be depressed when weather conditions deviate from average (DeSante and Geupel 1987). These facts are especially important because one of the most important ecological results of global climate change may be a greater annual variability in both local and large-scale weather conditions.

Changes in indices of adult population size and postfledging productivity from the first 4 years of MAPS are presented for all species pooled and for each target species caught at 10 or more stations in 1992 in the Northeast and Northwest regions. These data indicate that productivity varied greatly from year to year, presumably a result of large-scale weather conditions (e.g., precipitation and temperature) just before and during the breeding seasons. Productivity was poor across most of North America, but especially in the eastern third of the continent in 1990. Adult population sizes declined significantly in the East in 1991, presumably a result of the poor productivity in 1990. In 1992 productivity was poor again in

the East but good in the West. These results suggest that productivity in a given year may influence population sizes and population dynamics in subsequent years for many species over a large area.

BBIRD data likewise suggest that weather may substantially affect nesting productivity. Unusually wet weather conditions were reported at 6 of 14 BBIRD sites in 1992 when nest success of several species, including wood thrush (*Hylocichla mustelina*) and red-eyed vireo (*Vireo olivaceus*), was lower in 1992 than in 1993 (Table 1). These same two species also had reduced breeding productivity based on MAPS data. They produced fewer young per successful nest in 1992 than in 1993, a fact which also may be related to weather; some research suggests that clutch size as well as fledging success can be affected by weather conditions and may even provide a particularly sensitive measure of a species' tolerance to changing climatic conditions (e.g., Rotenberry and Wiens 1989). Further research may show that climatic variability is an important influence on the population trends of species.

Table 1. Wood thrush and red-eyed vireo nest success based on Mayfield (1961, 1975) estimates at midwestern BBIRD sites during 1992 and 1993 (numbers of nests are in parentheses).

State	Wood thrush		Red-eyed vireo	
	1992	1993	1992	1993
Ohio	23.0 (52)	33.1 (194)	6.6 (19)	33.7 (83)
Arkansas	45.6 (11)	58.0 (15)	35.3 (35)	42.1 (36)
Minnesota			19.0 (51)	23.0 (25)

Habitat-specific Differences

Forest fragmentation, where large forest blocks are cut and interspersed with open habitat, is believed to be particularly detrimental for breeding nongame birds. For example, BBIRD data show that fragmentation was associated with lower nest success in several species at midwestern BBIRD sites. Ovenbirds (*Seiurus aurocapillus*) were particularly sensitive to fragmentation effects; their reduced nest success resulted primarily from increased predation, although the parasitism rates of brown-headed cowbird (*Molothrus ater*) were also higher in fragments. No clear effect of fragmentation was noted for red-eyed vireos, although nest success differed substantially among unfragmented sites, potentially reflecting more subtle differences in habitat suitability or landscape-level effects (Table 2).

Adult Survival in Two Eastern Thrushes

Analysis of 3 years (1990-92) of MAPS data for veery (*Catharus fuscescens*) and wood thrush indicated low and substantially different

State	Ovenbird		Red-eyed vireo	
	Fragmented	Unfragmented	Fragmented	Unfragmented
Ohio	13.7 (35)	33.1 (45)	30.0 (52)	24.6 (50)
Wisconsin	19.8 (30)	42.6 (51)	26.4 (13)	50.8 (13)
Arkansas		51.9 (41)		38.7 (71)
Minnesota		44.5 (159)		21.0 (76)

Table 2. Ovenbird and red-eyed vireo nest success based on Mayfield (1961, 1975) estimates at fragmented and unfragmented midwestern BBIRD sites during 1992 and 1993 (numbers of nests are in parentheses).

($P < 0.06$) adult survival probabilities from 1990 to 1991. According to Breeding Bird Survey data, veery populations declined by 1.0% per year between 1966 and 1991, while wood thrush populations showed a statistically greater decline of 2.0% per year (Peterjohn and Sauer 1993). This difference in population declines is mirrored by survival indices; MAPS estimates of wood thrush survival are half that of the veery, possibly because of differences in adult survival over winter. This possibility is especially interesting because wood thrushes winter in Mexico and Central America where a greater proportion of the tropical forests have been cleared than in South America where veeries winter. Differences in estimated survival of the two species, however, could simply reflect different life-history traits (e.g., wood thrushes having lower adult survival associated with higher fertility; Martin in press). Estimated survival differences could also result from differences in breeding-site fidelity, which is related to nest success; a variety of evidence shows that birds disperse more in breeding seasons that follow nesting failure, potentially biasing survival estimates. Further nest-monitoring data from North America and survivorship data from both North America and the Neotropics are needed to identify causes of population declines in these and other Neotropical migratory landbirds.

Trends

Preliminary results from the MAPS and BBIRD programs suggest that population trends of nongame landbirds are influenced by

Courtesy T. Martin, NBS

Monitoring of nests, such as this one belonging to a red-faced warbler (*Cardellina rubrifrons*), provides information on breeding productivity.

weather-induced productivity problems, survival problems during migration or winter, and degradation of breeding habitat. These results emphasize the importance of national programs such as MAPS and BBIRD in providing baseline information on both continental and local habitat-specific processes that influence avian population dynamics. Ultimately, these data on breeding productivity and adult survival and their underlying environmental determinants will provide information critical for managing North American landbirds.

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Canada Geese in North America

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Canada geese (*Branta canadensis*) are probably more abundant now than at any time in history. They rank first among wildlife watchers and second among harvests of waterfowl species in North America. Canada geese are also the most widely distributed and phenotypically (visible characteristics of the birds) variable species of bird in North America. Breeding populations now exist in every province and territory of Canada and in 49 of the 50 United States. The size of the 12 recognized subspecies ranges from the 1.4-kg (3-lb) cackling Canada goose (*B.c. minima*) to the 5.0-kg (11-lb) giant Canada goose (*B.c. maxima*; Delacour 1954; Bellrose 1976).

Market hunting and poor stewardship led to record low numbers of geese in the early 1900's, but regulated seasons including closures, refuges, and law enforcement led to restoration of most populations. Winter surveys were begun to study population trends and set responsible harvest regulations for these long-lived and diverse birds. Winter surveys begun in 1936-37 probably represent the oldest continuing index of migratory birds in North America.

Surveys

Sporadic counts of migrating and wintering Canada geese from the ground were supplemented by regular tallies from the air in the early 1950's. Winter surveys began because the subarctic and arctic nesting areas of many subspecies were still unknown and aerial surveys of these remote areas were impractical.

The well-designed spring surveys of Canada geese that began in the 1970's with the Eastern Prairie population have now expanded to include several others (Office of Migratory Bird Management 1993). Spring surveys estimate numbers of each population at the time of year when subspecies are reproductively isolated and geographically separated. The smaller subspecies of Canada geese nest farther north (arctic and subarctic regions of Alaska and Canada), and most winter farther south (gulf states and Mexico) than do the larger subspecies.

Status and Trends

Most aggregations of wintering geese were overharvested in the early 1900's. Those

subspecies that nested in temperate regions closer to humans were most heavily hunted. By 1930 the giant Canada geese, which nested in the northern parts of the deciduous forest and tall-grass prairie, were believed extirpated. Numbers of the large geese that nested in the Great Plains and Great Basin (*B.c. moffitti*) were also severely reduced. Small Canada geese from the remote arctic and subarctic breeding ranges fared somewhat better, possibly because of less exposure to unregulated exploitation, but were also reduced in number.

Although hunting depleted numbers of Canada geese, human activity also created new habitats for these birds. Agriculture led to the clearing of forests and the plowing of prairies, creating the open landscapes preferred by geese. Cereal grains and pastures provided new food sources for geese, and the development of mechanical combines and pickers created an increased supply of waste grain (Hine and Schoenfeld 1968). In addition, uniform hunting regulations and improved wildlife law enforcement curtailed goose harvests after the signing of the Migratory Bird Treaty in 1916, and most goose populations increased over the next several decades (Figure). National wildlife refuges provided key sanctuaries and further assisted recovery of Canada goose numbers.

The giant Canada goose was "rediscovered" by Harold C. Hanson, a biologist of the Illinois Natural History Survey; the publication of his book *The Giant Canada Goose* in 1965 initiated a restoration effort that became one of the great success stories of wildlife management. These large geese were restored to their former range in the Mississippi and Central flyways and now breed in all states east of the Mississippi River.

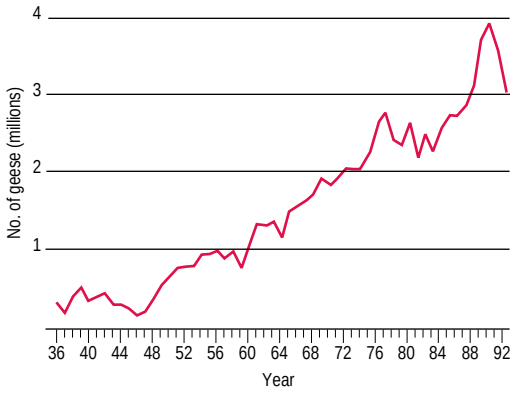

Figure. Total numbers of Canada geese counted on winter surveys, 1936-93.

Research and improved scientific management led to better understanding of diversity, distribution, and population dynamics of Canada geese in the 1970's. Awareness of differences in distribution and migration among the subspecies allowed managers to effectively control goose harvests. Improved management led to stable or increasing numbers of Canada geese in most populations (Table). The Mississippi Flyway Giant, Hi-line, Rocky Mountain, and Western Prairie/Great Plains populations, all composed mainly of large subspecies (*B.c. maxima* and *moffitti*), grew at about twice the rate of other populations that contained mainly smaller subspecies. The population numbers of the large geese that breed in the states of the Atlantic Flyway have also increased dramatically, but this trend was masked by declining numbers of geese in

Table. Canada goose population indices (in 1,000's) based on surveys conducted during fall and winter, 1969-93.

Year	Population*											
	AP	SJBP	MVP	Max(MF)	EPP	WP/GP	TGPP	SGPP	H-LP	RMP	DSKY	CCG
1969-70	775.2	106.9	324.7	50.8	106.6			151.2	44.2	25.8	22.5	
1970-71	675.0	127.3	292.3	64.4	126.3		133.2	148.5	40.5	25.3	19.8	
1971-72	700.2	117.6	293.9	55.8	157.4		160.9	160.9	31.4	36.6	17.9	
1972-73	712.0	101.3	295.9	54.2	181.4		148.4	259.4	35.6	37.1	15.8	
1973-74	760.2	136.0	277.9	57.6	205.8		160.5	153.6	24.5	42.8	18.6	
1974-75	819.3	101.0	304.4	57.0	197.1		133.5	123.7	41.2	46.7	26.5	
1975-76	784.5	115.5	304.9	62.1	204.4		203.7	242.5	55.6	51.6	23.0	
1976-77	923.6	129.8	478.5	58.5	254.2		171.3	210.0	67.6	54.3	24.1	
1977-78	833.2	180.4	575.5	60.1	270.2		215.5	134.0	65.1	59.0	24.0	
1978-79	823.6	142.7	434.5	77.1	207.2		187.6	163.7	33.8	62.7	25.5	
1979-80	780.1	127.0	394.9	86.4	171.8		165.9	213.0	67.3	77.3	22.0	64.1
1980-81	955.0	120.3	367.4	102.9	150.9		257.7	168.2	94.4	93.8	23.0	127.4
1981-82	702.6	118.5	250.9	107.6	145.3	175.0	284.7	156.0	81.9	64.3	17.7	87.1
1982-83	888.7	129.9	303.7	149.9	213.4	242.0	171.8	173.2	75.9	68.2	17.0	54.1
1983-84	822.4	129.9	352.8	103.9	163.1	150.0	279.9	143.5	39.5	55.5	10.1	26.2
1984-85	814.2	129.3	477.2	151.7	168.4	230.0	207.0	179.1	76.4	90.3	7.5	25.8
1985-86	905.4	158.0	618.9	180.1	169.0	115.0	198.2	181.0	69.8	68.3	12.2	32.1
1986-87	754.8	129.8	514.6	231.9	183.4	324.0	163.2	190.9	98.1	71.5		51.4
1987-88	737.9	158.8	564.6	225.9	228.5	272.1	315.8	139.1	66.8	71.4	12.2	54.8
1988-89	660.7	170.2	734.6	252.2	184.5	330.3	224.2	284.8	100.1	73.9	11.8	69.9
1989-90	733.8	159.4	1098.2	284.3	324.9	271.0	159.0	378.1	105.9	102.4	11.7	76.8
1990-91	706.9	142.2	939.7	345.1	218.4	390.0	315.5	508.5	116.6	86.7		110.2
1991-92	654.5	107.2	766.8	234.8	189.4	341.9	280.4	620.2	140.5	115.7	18.0	104.6
1992-93	569.2	104.4	673.4	282.6	146.4	318.0	238.7	328.2	118.5	99.5	16.6	149.3

*Populations are Atlantic (AP), Southern James Bay (SJBP), Mississippi Valley (MVP), Mississippi Flyway Giant (Max[MF]), Eastern Prairie (EPP), Western Prairie/Great Plains (WP/GP), Tall-grass Prairie (TGPP), Short-grass Prairie (SGPP), Hi-line (H-LP), Rocky Mountain (RMP), Dusky (DSKY), and Cackling Canada Goose (CCG).

Canada's eastern subarctic regions.

Although small geese with long migrations have generally not fared as well as large geese with short migrations, some small geese have responded well to intensive management. Introduced Arctic foxes (*Alopex lagopus)* depleted populations of the Aleutian Canada goose (*B.c. leucopareia*), and the subspecies was nearly extinct by 1940. About 300 were rediscovered in the Aleutians on Buldir Island in 1962 (Jones 1963). Subsequent removal of foxes and translocation of wild geese have led to increases to about 750 geese in 1975 and more than 11,000 in 1993.

Heavy hunting caused numbers of cackling Canada geese to plummet to record lows in the early 1980's, but intensive research (Raveling and Zezulak 1992) and harvest control have brought about a sustained recovery (Table).

Recent genetic studies of Canada geese support the existence of two major groups that last shared a common ancestor about 1 million years ago. The large-bodied group (*B.c. canadensis, interior, maxima, moffitti, fulva, occidentalis*) is mainly continental in distribution, while the small-bodied group (*hutchinsii, taverneri, minima, leucopareia*) breeds in coastal Alaska and Arctic Canada (Rusch et al. in press).

The future of these diverse stocks of Canada geese depends upon information adequate to permit simultaneous protection of rare forms, responsible subsistence and recreational hunting of abundant populations, and control of nuisance Canada geese in urban and suburban environments. Delineation of breeding ranges and spring surveys that monitor numbers of pairs and their productivity offer the most realistic approach to population management and the conservation of this remarkable diversity of geese.

Ranges of most populations have been described, and spring surveys are in place for some. Development and continuation of spring surveys for each subspecies of Canada geese are prerequisites for their conservation and management. The species can no doubt be perpetuated without spring surveys, but without continued monitoring, management, and conservation, it is likely that rare forms will disappear, opportunities for subsistence and recreational hunting will diminish, and nuisance problems caused by large geese living near humans will increase.

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Canada Geese in the Atlantic Flyway

by
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Large changes have occurred in the geographic wintering distribution and subspecies composition of the Atlantic Flyway population of Canada geese (*Branta canadensis*) over the last 40 years. The Atlantic Flyway can be thought of as being partitioned into four regions: South, Chesapeake, mid-Atlantic, and New England. Wintering numbers have declined in the southern states (North Carolina, South Carolina, Georgia, Florida), increased then decreased in the Chesapeake region (Delaware, Maryland, Virginia), and increased markedly in the mid-Atlantic region (New York, New Jersey, Pennsylvania, West Virginia) (Serie 1993; Fig. 1). In the New England region (Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, Connecticut), wintering numbers increased from around 6,000 during 1948-50 to between 20,000 and 30,000 today (Serie 1993).

Overall, the total number of wintering geese reached a peak of 955,000 in 1981 and has since declined 40% to 569,000 in 1993. Compounding these distributional changes in wintering numbers, the subspecies composition has also changed. The Canada goose population is composed of migrant geese (primarily *B.c.*

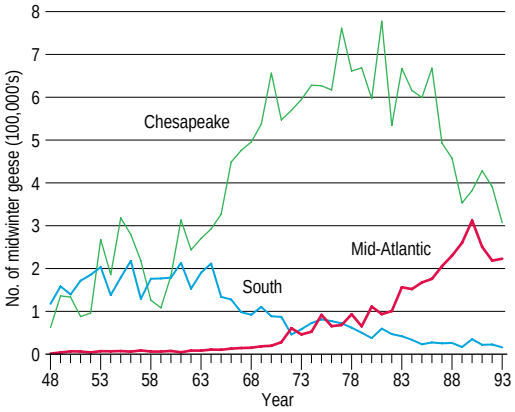

Fig. 1. Midwinter number of Canada geese in mid-Atlantic, Chesapeake, and South regions of the Atlantic Flyway, 1948-93 (Midwinter Survey, U.S. Fish and Wildlife Service, Office of Migratory Bird Management).

canadensis and *B.c. interior*) that breed in the subarctic regions of Canada and resident geese (primarily *B.c. maxima* and *B.c. moffitti*) that breed in southern Canada and the United States (Stotts 1983). The number of resident geese in Maine to Virginia has increased considerably from maybe 50,000 to 100,000 in 1981 (Conover and Chasko 1985) to an average of 560,000 in 1992-93 (H. Heusman, Massachusetts Division of Fisheries and Wildlife, personal communication). This rapid increase in resident geese suggests that the migrant population has declined more than the 40% decline observed in total wintering geese from 1981 to 1993.

Courtesy R. Gehman

Flying neck-banded goose (*Branta canadensis*).

Population Changes

Changes in population numbers result from changes in production, survival, and movement, acting singly or in combination. Consequently, understanding the reason for population changes involves detecting variation in survival, production, and movement over time and relating that variation to changes in wintering numbers. During the 1970's, the decrease of wintering geese in the South and increase in the Chesapeake region appeared to result from increased survival of geese in the Chesapeake and possibly from movement or short-stopping of geese from the South to the Chesapeake (Trost et al. 1986). Short-stopping occurs when migrant geese winter in a more northern location than their traditional, more southern, migration terminus.

During the 1980's, the decrease of wintering geese in the Chesapeake appeared to result from an 11% decrease in average survival from 1963-74 to 1984-88 (Hestbeck 1994a). This decrease in survival corresponded to a 36% increase in average harvest rate for the Atlantic Flyway from 1963-74 to 1984-88 (Fig. 2). Overall, the flyway harvest rate, as a 3-year average, increased from 19% in 1962-64 to 34% in 1982-84, and then slowly declined to 31% by 1990-92. The eastern Canada harvest rate has slowly increased from 4.2% in 1968-70 to 8.1% in 1990-92. The slight decline in the harvest rate in the flyway since 1982-84 has been partially offset by harvest rate increases in eastern Canada.

The decrease in number of geese wintering in the Chesapeake region in the 1980's was not related to changes in production. Production for migrants, measured from the Canadian data, remained constant over the period of population decline in the Chesapeake (Fig. 3). Average production recently declined during 1991-92 for geese harvested in Quebec. I also used harvest age ratios for the mid-Atlantic and Chesapeake regions to test for differences in production between these regions (Hestbeck 1994b). If the changes in wintering number resulted from changes in production, the average annual change in the age ratios would be higher for the mid-Atlantic region than for the Chesapeake region. The average annual changes were not different between these regions, however, indicating that regional production differences were not present.

The decrease in number of geese wintering in the Chesapeake region in the 1980's was not caused by migrant geese short-stopping in the mid-Atlantic instead of returning to the Chesapeake. From neck-band data, the probability of returning or moving to the different regions was estimated and indicated that, although geese traditionally returned to the same wintering area, they also changed wintering areas from year to year (Hestbeck 1994b). In years with harsher winters, geese wintered farther south than during milder winters (Hestbeck et al. 1991). Overall, the probability of returning or moving to the Chesapeake region was higher than the probability of returning or moving to any other region. When population size, survival, and movement were combined to estimate net movement among regions, the estimated net movements among regions were small and did not correspond to the changes in numbers of wintering geese. Taken

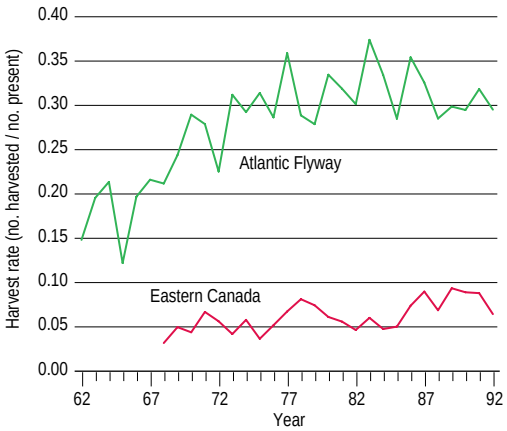

Fig. 2. Harvest rate of Canada geese in the Atlantic Flyway, 1962-92 (Harvest and Midwinter Surveys, U.S. Fish and Wildlife Service, Office of Migratory Bird Management) and eastern Canada, 1968-92 (Harvest Survey, Canadian Wildlife Service, National Wildlife Research Centre).

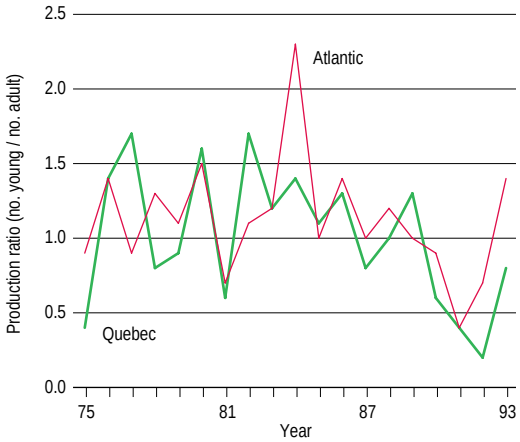

Fig. 3. Production ratio of Canada geese in Quebec and Atlantic regions of eastern Canada, 1975-93 (Waterfowl Parts Collection Survey, Canadian Wildlife Service, Atlantic Region, Sackville, N.B.).

together, these results suggested that the increases in the number of wintering geese in the mid-Atlantic region did not result from short-stopping of geese.

The increase of wintering geese in the mid-Atlantic most likely resulted from expanding resident populations. Resident geese generally have larger body sizes, allowing them to winter farther north than smaller-bodied migrant geese (Lefebvre and Raveling 1967). Resident and migratory-resident geese may selectively remain in the mid-Atlantic region. In addition, the resident population may be increasing faster than the migrant population because survival and production appear higher for residents than for migrants. Residents survive better partly because they are familiar with areas of food and refuge and may avoid hunting areas (Johnson and Castelli 1994). Production may be higher for resident than migrant geese because the climate is less variable and milder with a longer growing season in southern Canada and the United States than in the subarctic. Resident geese may also reach reproductive age earlier than migrant geese because the southerly growing season is longer, providing greater food resources.

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Arctic Nesting Geese: Alaskan Populations

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North American populations of most goose species have remained stable or have increased in recent decades (USFWS and Canadian Wildlife Service 1986). Some populations, however, have declined or historically have had small numbers of individuals, and thus are of special concern. Individual populations of geese should be maintained to ensure that they provide aesthetic, recreational, and ecological benefits to the nation. Monitoring and management efforts for geese should focus on individual populations to ensure that genetic diversity is maintained (Anderson et al. 1992).

Alaska is the only state with viable breeding populations of arctic geese. Five species (11 subspecies) nest in Alaska, and although these species also breed in arctic regions of Canada or Russia, most geese of the Pacific Flyway originate in Alaska or use Alaskan habitats during migration. Alaskan geese are often hunted for subsistence by Alaskan Natives.

While data for some areas are lacking, populations of greater white-fronted geese (*Anser albifrons frontalis*) and medium-sized Canada geese (*Branta canadensis*) in interior and northern Alaska appear stable or have increased (King and Derksen 1986). Although only a small number of lesser snow geese (*Chen caerulescens caerulescens*) nest in Alaska, substantial populations occur in Canada and Russia. Populations of Pacific black brant (*B. bernicla nigricans*), emperor geese (*C. canagica*), greater white-fronted geese, and cackling Canada geese (*B.c. minima*) on the Yukon-Kuskokwim Delta (YKD) of western Alaska have declined from their historical numbers and are the focus of special management efforts (USFWS 1989). In addition, populations of tule white-fronted geese (*A.a. gambeli*), Aleutian Canada geese (*B.c. leucopareia*), Vancouver

Canada geese (*B.c. fulva*), and dusky Canada geese (*B.c. occidentalis*) are of special concern because of their limited geographic distributions and small numbers.

Inventory of Arctic Geese

An annual index of the Pacific black brant population has been obtained since 1964 by the U.S. Fish and Wildlife Service (USFWS) during aerial surveys of wintering areas along the Pacific coast (Bartonek 1994a). Population trends of cackling Canada geese and greater white-fronted geese from 1965 to 1979 were based on surveys conducted by USFWS and state agency biologists on migration areas in the Klamath Basin of Oregon and California. Population trends of those two species from 1980 to 1993 were based on coordinated surveys on wintering areas (Bartonek 1994b).

Emperor geese have been inventoried by USFWS biologists during aerial surveys of spring and fall migration areas on the Alaska Peninsula and the YKD since 1980 (Bartonek 1992). We used the highest count within a year to determine the population trend for emperor geese. Population indices for tule white-fronted geese were obtained from surveys on wintering and migration areas in the Pacific Flyway in intermittent years since 1978. Aleutian Canada geese have been counted on a spring staging area in northern California since 1975. Dusky Canada geese have been inventoried on their wintering areas in the Pacific Flyway since 1953. There are no data on population trends of Vancouver Canada geese; however, the winter population in the northern portion of southeastern Alaska was estimated by USFWS biologists in 1986.

Status of Alaskan Geese

Yukon-Kuskokwim Delta Geese

Most geese on the YKD nest within 30 km (15-20 mi) of the Bering Sea but winter in diverse areas. Pacific black brant primarily winter along the Pacific coast of Mexico while greater white-fronted geese and cackling Canada geese primarily winter in the Central Valley of California. In recent years, increasing numbers of cackling Canada geese have wintered in Oregon. Most emperor geese winter in the Aleutian Islands.

These four species experienced sharp population declines (30%-50%) between the early 1960's and mid-1980's (Fig. 1). The declines were likely due to the combined effects of subsistence harvest of breeding birds and eggs on the YKD, excessive sport harvest on the wintering areas, poor weather during nesting, and fox predation of nests (USFWS 1989). In 1984, the USFWS, Yupik Natives, state wildlife agencies, and sport hunters cooperated to reduce sport and subsistence harvest. Since then populations of cackling Canada geese and greater white-fronted geese have begun to recover while emperor geese and black brant remain near historical lows (Fig. 1). Poor winter survival of juvenile emperor geese may be slowing recovery of that species (Schmutz et al. 1994). Winter survival of cackling Canada geese has improved since the reduction in sport hunting; however, there is no evidence that their survival in summer has improved (Raveling et al. 1992).

Tule White-fronted Geese

The only known nesting area for tule white-fronted geese is in Upper Cook Inlet (Timm et al. 1982) and the adjacent Susitna in south-central Alaska. Tule geese may also occur on the Innoko National Wildlife Refuge in western Alaska. The numbers of tule geese counted on wintering areas in the Central Valley of California in recent years are higher than during the late 1970's (Fig. 2). It is unclear if the increase is due to population growth or because of improved understanding of the winter distribution.

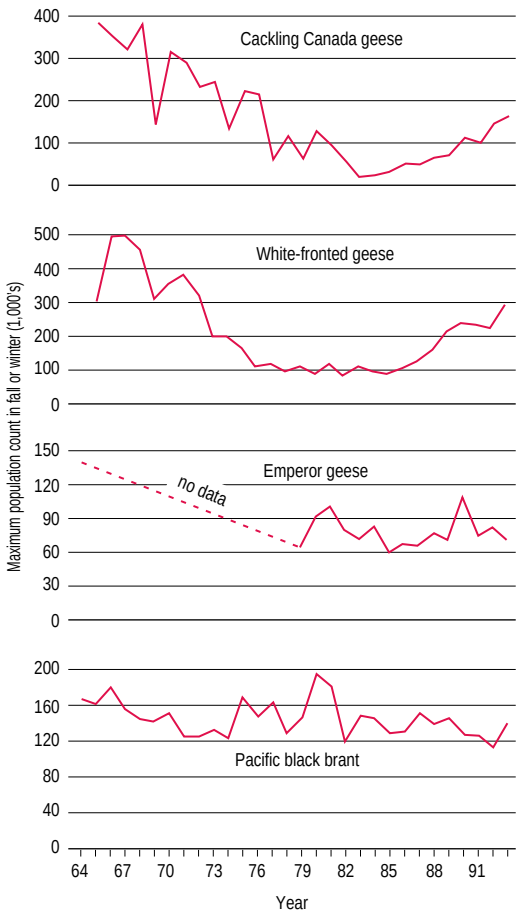

Fig. 1. Population trends of arctic geese that nest on the Yukon-Kuskokwim Delta, Alaska (1964-93).

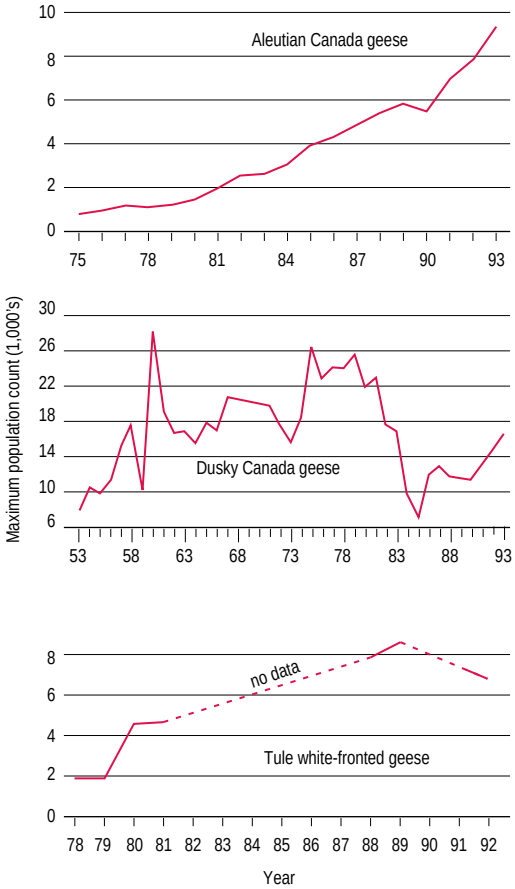

Fig. 2. Population trends of Aleutian Canada geese (1975-93), dusky Canada geese (1953-93), and tule white-fronted geese (1978-89).

Dusky Canada Geese

Dusky Canada geese primarily nest on the Copper River Delta of south-central Alaska, the islands of Prince William Sound, and Middleton Island in the Gulf of Alaska. They winter in the Willamette Valley of Oregon and the lower Columbia River. The population was stable or increased between the 1950's and 1970's. During the early 1980's, however, the population declined, then stabilized at a lower level in the mid-1980's (Fig. 2). The decline was largely due to reduced nesting success as a result of habitat changes on the nesting area following the 1964 Alaska earthquake. Invasion of shrubs and loss of wet meadow habitats resulted in more mammalian predators and greater nest predation (Subcommittee on Dusky Canada Geese 1992).

Aleutian Canada Geese

Although once abundant on the Aleutian, Commander, and Kuril islands, the numbers of Aleutian Canada geese were greatly reduced by foxes and dogs introduced to nesting islands by commercial fur farmers before World War II (Byrd and Woolington 1983). The subspecies was classified as endangered in 1967, and by the mid-1970's fewer than 800 individuals remained (USFWS 1991). Sport harvest on migration and wintering areas in Oregon and California was stopped in 1975, and fox control was initiated on nesting islands. Geese were also transplanted to fox-free islands. The population of Aleutian Canada geese responded to recovery efforts and has grown to more than 9,000 individuals (Fig. 2). The status of the subspecies was changed from endangered to threatened in 1991.

Vancouver Canada Geese

Vancouver Canada geese nest and use brood-rearing areas in southeastern Alaska (Lebeda and Ratti 1983) and winter on coastal wetlands near the breeding areas. Few data on breeding numbers exist because Vancouver Canada geese nest in coastal forests and are difficult to survey. About 10,000 Vancouver Canada geese wintered in the northern portion of southeastern Alaska in 1986 (Hodges and Conant 1986). Wintering sites are scattered among coastal wetlands and have not been consistently surveyed. Consequently, population trends of this subspecies are not known. Population trends are likely influenced by environmental variables because sport and subsistence harvest are minimal (King and Derksen 1986).

Status of Habitats of Special Concern

Yukon-Kuskokwim Delta

The YKD (Fig. 3) is the primary waterfowl nesting area in Alaska (King and Dau 1981); it provides critical nesting and brood-rearing habitat for more than 400,000 geese. In addition, the entire population of Wrangel Island lesser snow geese uses the YKD during fall staging (Ely et al. 1993). While much of the YKD is within the Yukon Delta National Wildlife Refuge, it is also a region where more than 17,000 Yupik Natives live in 40 Native villages. Large private inholdings, primarily Native corporation lands, exist within the refuge and contain important waterfowl nesting habitat. Meeting the subsistence needs of Native people while maintaining or enhancing waterfowl populations on the YKD requires close coordination among the Yupik Natives and federal and state agencies. Management of subsistence waterfowl harvest on the YKD has been difficult because of cultural differences and constraints imposed by the Migratory Bird Treaty Act. Coordinated management efforts will be especially important in the future as Native populations increase.

Izembek Lagoon

Nearly the entire world population of more than 120,000 Pacific black brant uses Izembek

Lagoon (Fig. 3) as a fall staging area for about 2 months. Although Izembek Lagoon is protected as a national wildlife refuge and state game refuge, it is near offshore oil leases in Bristol Bay. Should oil development proceed, increased aircraft activity over Izembek Lagoon could result in a significant increase in disturbance that could prevent brant from accumulating sufficient body fat for their nonstop flight to wintering areas in Mexico. This lack of sufficient body fat could result in increased mortality (Ward et al. 1994).

Bristol Bay Lowlands

Estuaries on the north side of the Alaska Peninsula (Fig. 3) provide critical migration habitat for cackling Canada geese, Taverner's Canada geese (*B. c. taverneri*), and emperor geese, and nesting habitat for a unique group of greater white-fronted geese. Part of this area is protected in State Critical Habitat Areas managed by the Alaska Department of Fish and Game. At least 5,265 ha (13,000 acres) of important habitat, however, is state land that may be subject to resource development.

Teshekpuk Lake Special Areas

Up to 32,000 Pacific black brant (25% of the world population) and 30,000 individuals of other goose species molt annually on Teshekpuk Lake Special Area (TLSA) (Fig. 3) on the National Petroleum Reserve in Alaska (Derksen 1978; King 1984). The area is managed by the Bureau of Land Management, and special regulations govern resource development on the TLSA to minimize adverse impacts to wildlife. Energy development in adjacent areas, though, may result in increased aircraft activity that could disturb molting geese and reduce their ability to secure forage needed for feather replacement (Jensen 1990).

Interior Wetlands

Greater white-fronted and Canada goose-nesting and brood-rearing habitats occur in interior wetlands near the Yukon, Tanana, Kuskokwim, Koyukuk, Susitna, and Innoko rivers (King and Lensink 1971). National wildlife refuges encompass much of the important habitat, although some areas are managed by the state of Alaska, private landowners, and the Bureau of Land Management. At present, there is relatively little human-related disturbance in these areas, although placer mining, oil exploration and development, timber harvest, and military training could affect some areas.

Upper Cook Inlet

About 100,000 geese and swans use Upper Cook Inlet (Fig. 3) as spring migration habitat.

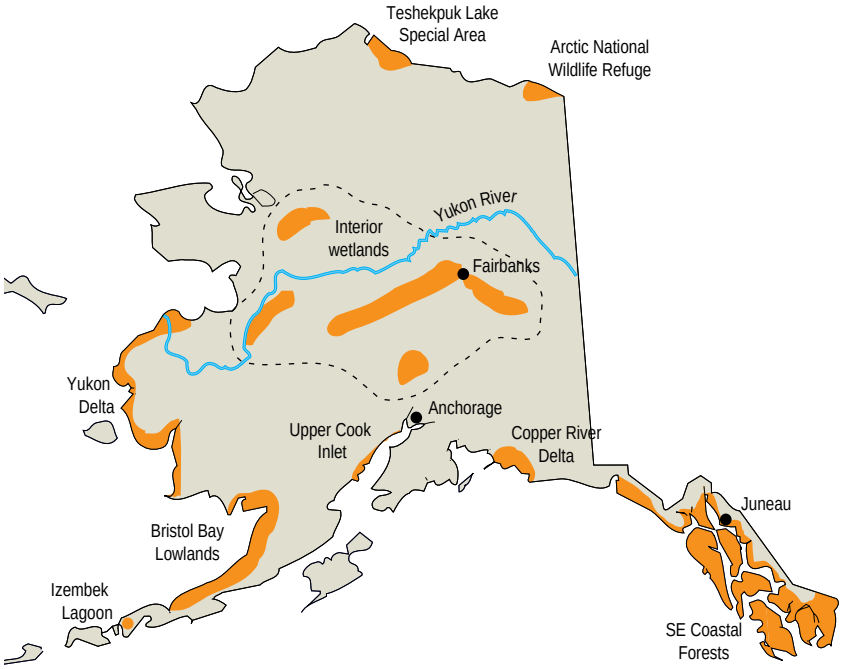

Fig. 3. Alaskan habitats of special importance to geese.

In addition, this inlet is one of two nesting areas of tule white-fronted geese. Development of oil and gas, coal, timber, and mineral deposits has either been proposed or is ongoing in Upper Cook Inlet and may affect coastal wetlands used by migratory waterfowl. Most of the important waterfowl habitats in this area are state game refuges or Critical Habitat Areas managed by the Alaska Department of Fish and Game.

Alaska Coastal Forests

Some nesting and brood-rearing areas of Vancouver Canada geese (Fig. 3) occur in areas of commercially harvestable timber (Lebeda and Ratti 1983). Logging activities on U.S. Forest Service land on the Tongass National Forest could affect these habitats. In addition, timber harvest on Native corporation lands may restrict opportunities to transplant Vancouver Canada geese into areas of suitable habitat or may limit natural expansion of the subspecies range (King and Derksen 1986). Use of tidal areas to store harvested timber before shipping can affect wintering habitat of Vancouver Canada geese and migration habitats of other waterbirds.

Arctic National Wildlife Refuge

As many as 300,000 lesser snow geese and an unknown number of greater white-fronted geese stage on the Arctic National Wildlife Refuge (Fig. 3) before fall migration. During staging, geese feed intensively and build fat reserves for migration. Proposed petroleum leases on the refuge would result in increased aircraft activity that could disrupt feeding behavior of geese, displace birds from feeding

habitats, and reduce their ability to accumulate body fat before migration (Brackney et al. 1987). Diminished fat reserves could reduce survival during migration.

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North American Ducks

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Increased predation and habitat degradation and destruction coupled with drought, especially on breeding grounds, have caused the declines of some duck populations. More than 30 species of ducks breed in North America, in areas as diverse as the arctic tundra and the subtropics of Florida and Mexico. For many of these species, however, the Prairie Pothole region of the north-central United States and south-central Canada is the most important breeding area (Fig. 1), although migratory behavior and the life histories of different species lead them to use many wetland habitats.

Numerous sources of information are available on the status of duck populations in North America. The two most comprehensive and reliable sources are the Breeding Population and Habitat Survey, conducted since 1955 and encompassing the Prairie Pothole region, boreal forests, and tundra habitats from South Dakota to Alaska (Caithamer et al. 1993; Fig. 1), and the Midwinter Survey, encompassing the United States and portions of Canada and

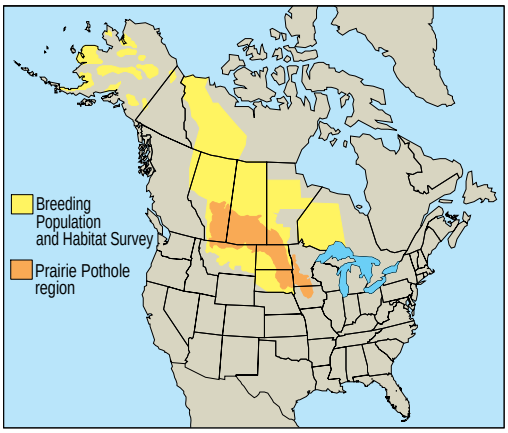

Fig. 1. The Prairie Pothole region and areas sampled in the Breeding Population and Habitat Survey.

Mexico at regular intervals. Results from these surveys are the basis for this article.

The Breeding Population and Habitat Survey is conducted during May and June when most species occupy their breeding ranges. Pilot-biologists and observers in airplanes identify and count ducks on a sample of transects. Not all ducks are visible from the air, so some transects are resurveyed more thoroughly with a helicopter or from the ground to obtain complete counts. These data are used to correct the air counts and obtain unbiased estimates of duck densities in these areas. Estimates of number of pairs of ducks are expanded to provide population estimates for the entire surveyed area. This survey, conducted by the Canadian Wildlife Service and the U.S. Fish and Wildlife Service (USFWS), is among the most extensive and comprehensive surveys conducted annually for any group of animals anywhere in the world. Survey estimates are the major determinant governing the regulation-setting process for the sport harvesting of ducks by both Canadian and United States provincial, state, and federal governments.

The Breeding Population and Habitat Survey is most reliable for mallards (*Anas platyrhynchos*), gadwall (*A. strepera*), American wigeon (*A. americana*), green-winged teal (*A. crecca*), blue-winged teal (*A. discors*), northern shoveler (*A. clypeata*), redhead (*Aythya americana*), canvasback (*A. valisineria*), and scaup (*A. affinis* and *A. marila*). Researchers and managers are trying to expand the geographic range of this survey in the Pacific Flyway, eastern Canada, and the northwestern United States.

The breeding survey, however, poorly monitors species such as whistling ducks (*Dendrocygna* spp.), mottled ducks (*Anas fulvigula*), American black ducks (*A. rubripes*), most sea ducks and mergansers (*Lophodytes cucullatus*, *Mergus merganser*, *M. serrator*), and wood ducks (*Aix sponsa*).

The Midwinter Survey has been conducted annually in early January since the mid-1940's. It is not as reliable as the breeding survey because of methodological shortcomings and because winter is a poor time to survey population abundance (Eggeman and Johnson 1989). Despite its limitations, this survey does provide useful information on such species as the black duck that are not well surveyed by the breeding survey (Conroy et al. 1988).

Status and Trends

Population estimates of all ducks from the breeding survey have varied from 26.5 to 42.8 million since 1955 (Fig. 2). Generally, breeding populations were high in the 1950's and 70's and low in the 60's, 80's, and 90's. The 1993 estimate of 28.0 million was 20% below the 1955-92 average.

Estimates of ducks from the Midwinter Survey also have varied since 1955 (Fig. 2). The 1993 estimate of 10.3 million ducks was the lowest recorded, and 44% below the 1955-92 average.

The Breeding Population and Habitat Survey provides reliable estimates for seven species of dabbling ducks, while the Midwinter Survey provides estimates for eight. The breeding population of total dabbling ducks in 1993 was 20% below the 1955-92 average. Compared with the 1955-92 average, 1993 breeding population estimates suggest population declines for mallards, American wigeon, blue-winged teal, and northern pintail. Population estimates were unchanged for green-winged teal and increased for gadwall and northern shoveler (Figs. 3-5). During the most recent 10-year period, the breeding population of northern pintail decreased, gadwall populations increased, and populations of six other species were stable (Table). Midwinter estimates of all species of dabbling ducks were stable or increased during 1984-93 (Table).

Midwinter estimates are the only long-term data available for black ducks. Apparent differences in population trends between the breeding and midwinter surveys (Table) are a function of differences in the quality of the surveys and in the populations monitored by the surveys. For example, breeding mallards have increased in recent years in the Atlantic Flyway, which is outside the breeding survey area. The breeding survey indicates a stable trend for mallards while the winter survey indicates an increasing trend; the two surveys monitor different portions of the total continental population.

Five species of diving ducks are monitored by breeding and winter surveys. Because lesser scaup are not distinguished from greater scaup in the surveys, these species have been combined. Breeding populations of diving ducks in 1993 were 18% below the 1955-92 average. Redhead and scaup breeding populations were lower than average, whereas the canvasback population was near average, and the ring-necked duck (*Aythya collaris*) population was above average (Figs. 4, 6). From 1984 to 1993, the breeding population of scaup declined while the breeding population of ring-necked ducks increased (Table). The Midwinter Survey also indicated an increasing population of ring-necks during this period (Table).

Fourteen species of sea ducks, mergansers, and their allies were monitored by the breeding survey. These 14 species plus the harlequin duck (*Histrionicus histrionicus*) were monitored during the Midwinter Survey. Because

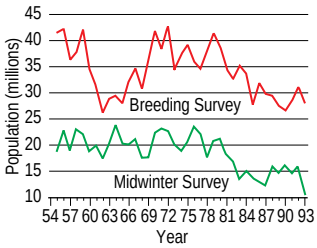

Fig. 2. Duck populations in North America, 1955-93, from the Breeding Population and Habitat Survey and the Midwinter Survey.

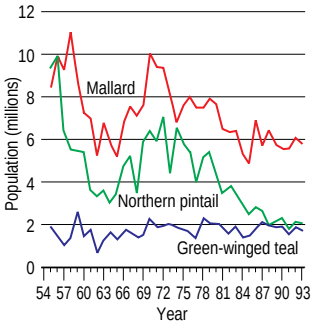

Fig. 3. Mallard, northern pintail, and green-winged teal breeding population estimates, 1955-93.

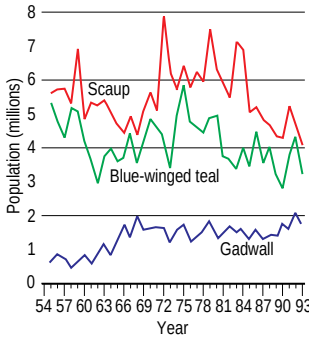

Fig. 4. Scaup, blue-winged teal, and gadwall breeding population estimates, 1955-93.

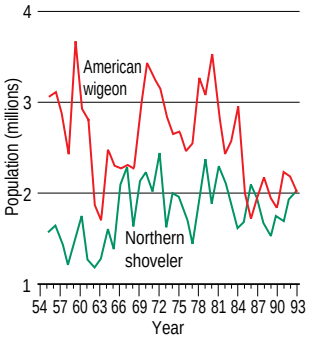

Fig. 5. American wigeon and northern shoveler breeding population estimates, 1955-93.

Table. Estimated annual numbers (in thousands) and recent trends (1984-93) of ducks based on the survey areas monitored by breeding and midwinter surveys.

Tribe and species	Breeding		Wintering	
	No.	Trend	No.	Trend
Perching ducks				
Wood duck			33	Stable
Dabbling ducks				
American black duck			278	Stable
American wigeon	2,053	Stable	1,088	Stable
Blue-winged teal	3,192	Stable		
Blue-winged and cinnamon teal			166	Increasing
Gadwall	1,755	Increasing	1,168	Increasing
Green-winged teal	1,694	Stable	2,086	Increasing
Mallard	5,708	Stable	4,994	Increasing
Mottled duck			129	Increasing
Northern pintail	2,053	Decreasing	2,241	Stable
Northern shoveler	2,046	Stable	638	Stable
Diving ducks				
Canvasback	472	Stable	298	Stable
Greater and lesser scaup	4,080	Decreasing	1,070	Stable
Redhead	485	Stable	336	Stable
Ring-necked duck	868	Increasing	421	Increasing
Sea ducks and mergansers				
Bufflehead	869	Increasing	126	Increasing
Eiders[a]	8	Decreasing	132	Stable
Goldeneye[b]	592	Stable	122	Stable
Harlequin duck			<1	Stable
Mergansers[c]	528	Stable	264	Increasing
Oldsquaw	174	Decreasing	10	Decreasing
Scoters[d]	1,006	Decreasing	160	Stable
Stifftails				
Ruddy duck	387	Stable	110	Decreasing

[a]Eiders include common eider (*Somateria mollissima*), king eider (*S. spectabilis*), spectacled eider (*S. fischeri*), and Steller's eider (*Polysticta stelleri*).
[b]Goldeneye include Barrow's goldeneye (*Bucephala islandica*) and common goldeneye (*B. clangula*).
[c]Mergansers include hooded merganser (*Laphodytes cucallatus*), red-breasted merganser (*Mergus serrata*), and common merganser (*M. merganser*).
[d]Scoters include black scoter (*Melanitta nigra*), surf scoter (*M. perspcillata*), and white-winged scoter (*M. fusca*).

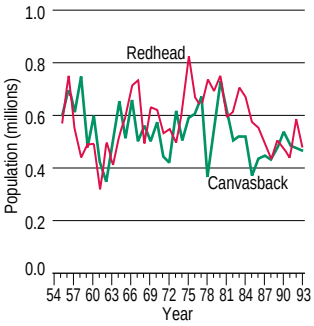

Fig. 6. Redhead and canvasback breeding population estimates, 1955-93

some of these species are difficult to identify during aerial surveys, or are encountered rarely, they are combined with related species (*see* Table).

Collectively, breeding populations of mergansers and their allies were 9% lower in 1993 compared to the 1955-92 average. Merganser, oldsquaw (*Clangula hyemalis*), eider, and scoter breeding populations in 1993 were all lower than their 1955-92 averages (*see* Table for species). The breeding population of goldeneye in 1993 was similar to the 1955-92 average, whereas the bufflehead (*Bucephala albeola*) breeding population was higher than the long-term average. During the last 10 years, breeding populations of eiders, oldsquaw, and scoters decreased, bufflehead increased, and goldeneye and mergansers were stable (Table). Winter population estimates during 1983-92 decreased for oldsquaw, increased for bufflehead and mergansers, and were stable for other species in the sea duck tribe (Table).

In the United States and Canada, wood ducks are the only representative of the tribe Cairinina and ruddy ducks (*Oxyura jamaicensis*) are the only representative of the Oxyurini tribe. Wood ducks are hard to survey because they inhabit forested wetlands where it is difficult to obtain reliable counts. Their current population, however, is greater than in the early 1900's (Bellrose 1980). Midwinter counts of wood ducks during 1983-92 indicated a stable population (Table). Ruddy duck breeding populations in 1993 were similar to the 1955-92 average.

Factors Affecting Population Status

Duck population changes occur on breeding, staging, and wintering habitats, with the changes on breeding habitats having the greatest effect on populations. Degradation and destruction of wetlands over the last 200 years have diminished duck populations; wetland alteration and degradation continue. The rate of wetland loss has been greatest in prime agricultural areas such as the Prairie Pothole region (Fig. 1), and lowest in northern boreal forests and tundra. Thus, species such as dabbling ducks that mostly nest in the severely altered Prairie Potholes have been harmed more than species like sea ducks and mergansers that nest farther north (Bellrose 1980; Johnson and Grier 1988).

Because most dabbling ducks need grassy cover for nesting (Kaminski and Weller 1992), conversion of native grasslands to agricultural production, including pastures, has reduced available nesting cover and contributed to a reduced nesting success for dabblers. This condition is especially true in the Prairie Pothole region of the United States and Canada (Fig. 1). In addition, highly variable precipitation in the Prairie Potholes has changed the number of wetlands available for nesting. For example, in 1979 there were 6.3 million wetlands in the surveyed portion of the Prairie Pothole region, but by the next spring, wetlands in the same area had decreased 55% to 2.9 million. Two years later they increased more than 100% to 4.2 million. These annual changes can temporarily mask the long-term declining trend in wetland abundance across the Prairie Pothole region.

The changing availability of wetland habitats in the Prairie Potholes region causes substantial fluctuations in some duck populations. During periods of high precipitation, larger wetland basins are full or overflowing, and shallow wetlands are abundant. Species such as the northern pintail, which tend to use shallow or ephemeral wetlands for feeding, produce more young when wetland numbers increase (Smith 1970; Hochbaum and Bossenmaier 1972). Consequently, population numbers increase as they did during the 1970's.

During the driest periods, however, such as those in the 1980's, only the deepest and most permanent wetlands retain water, causing population declines in species such as pintails that rely primarily on shallow wetlands. Population numbers are more stable for species like the canvasback, which rely on deeper marshes, and are therefore less affected by annual changes in wetland numbers because deeper marshes consistently retain water, providing ample habitat in most years (Stewart and Kantrud 1973).

Nest success in the Prairie Pothole region has declined in recent years largely because of increased nest predation caused by the range expansion of some predators and by reduced nesting habitat (Sargeant and Raveling 1992). Fewer and smaller areas of nesting habitat concentrate duck nests, enhancing the ability of predators to find nests. Predators such as raccoons (*Procyon lotor*) have expanded their range northward, probably because they can den in buildings, rock piles, and other human-made sites during winter.

Although wetland drainage, urbanization, and other human-caused changes have resulted in wintering habitat losses, these losses have been offset, at least for dabbling ducks, by increased fall and winter food from waste grain left in stubble fields. In addition, the national wildlife refuge system has protected and managed many staging and wintering areas for the benefit of waterfowl.

Modern duck-hunting regulations are believed to keep recreational harvest at levels compatible with the long-term welfare of duck populations. The proportion of ducks harvested varies regionally and by species, age, and sex. In 1992, 2%-12% of the adult mallards from the Prairie Pothole region were killed by hunters. Harvest rates of other species were generally lower. These conservative harvest rates are unlikely to cause population declines (Blohm 1989).

Conclusions

Changes in duck populations reflect changes in quality and quantity of waterfowl habitats. Long-term declines in populations have been caused by extensive habitat alterations. By contrast, short-term changes primarily reflect weather and resultant availability of wetland habitats. Maintenance of the current monitoring system and initiatives to improve our monitoring capability are essential for effective duck management.

Maintaining or increasing the quality and quantity of waterfowl habitat is needed to stabilize or increase duck populations. Agricultural policies and practices can profoundly affect habitat availability in Canada and the United States. For example, the Conservation Reserve Program, in which certain agricultural areas were set aside and planted in grasses, has added much-needed dabbling duck nesting habitat and therefore has improved their productivity in the U.S. portion of the Prairie Pothole region (R.E. Reynolds, USFWS, personal communication). The North American Waterfowl Management Plan, through its regional joint ventures, is striving to increase the habitat available for waterfowl and to improve monitoring of some populations.

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Decline of Northern Pintails

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The size of the continental breeding population of northern pintail (*Anas acuta*) has greatly varied since 1955, with numbers in surveyed areas ranging from a high of 9.9 million in 1956 to a low of 1.8 million in 1991. This variation results primarily from differences in the numbers of breeding pintails in the prairie region of Canada and the United States (Fig. 1); these numbers ranged from 8.6 million in 1956 to 0.5 million in 1991; numbers in the northern regions from Alaska to northern Alberta and northern Manitoba varied primarily between 1 and 2 million.

Breeding pintails prefer seasonal shallow-water habitats without tall emergent aquatic vegetation (Smith 1968). The proportions and distribution of breeding pintails on the prairies vary annually depending on the amount of annual precipitation and the resulting increase or decrease in the availability of suitable breeding habitat (Smith 1970; Johnson and Grier 1988).

Changes in the size of the continental pintail population result from changes in production, survival, or both. Consequently, understanding population changes involves detecting variation in survival and production over time and relating that variation to changes in population size. Once the cause of the decline is determined, appropriate management strategies can be developed to reverse it.

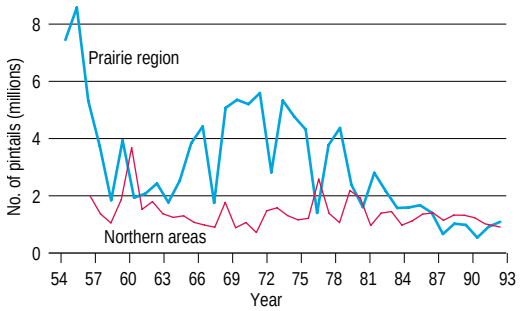

Fig. 1. Number of pintails in northern areas from Alaska to northern Alberta and northern Manitoba and in the prairie region from southern Alberta and central Montana to southern Manitoba and the Dakotas from 1955 to 1993 (Breeding Population and Habitat Survey, U.S. Fish and Wildlife Service, Office of Migratory Bird Management).

Status and Trends

I arbitrarily partitioned the population data into periods of relative growth, stability, and decline to help explain changes in the continental breeding population, which declined from 1955 to 1962, increased from 1963 to 1970, remained at a high stable level from 1971 to 1979, and declined from 1980 to 1992. I also partitioned the continental population into flyways based on data from recoveries of winter-banded pintails. This data indicated that pintails exhibit a high fidelity to the winter-banding region and flyway (Hestbeck 1993). Data from recoveries of summer-banded pintails were used to associate birds between breeding and wintering areas.

Courtesy J.B. Hestback

Pintails (*Anas acuta*).

Data on the pintail population were obtained through various surveys conducted by the United States and Canada. The Breeding Population and Habitat Survey provided estimates for the number of breeding pintails and for the total number of ponds. The total number of ponds was used as an index of breeding-habitat availability where the availability increased as the number of ponds increased. Annual survival rates were estimated from legband recoveries of summer-banded pintails.

I estimated average survival rates for the previously listed time periods for all areas with banding data. As an index of production, I used the number of young females divided by the number of adult females (i.e., age-ratio) harvested annually in each flyway reported in the Waterfowl Parts Collection Survey (U.S. Fish and Wildlife Service, Office of Migratory Bird Management). Because of possible harvest differences among flyways and large variation in annual ratios, I estimated the average age-ratio for each flyway for the above time periods.

Changes in the continental population can be addressed by studying changes in flyway populations because pintails from different summer breeding areas were associated with certain wintering areas. Generally, pintails wintering in the Pacific Flyway were associated with breeding areas in the western states and provinces from Alaska to Saskatchewan and central Montana. Pintails in the Central Flyway were primarily associated with breeding areas in Saskatchewan, eastern Montana, Manitoba, and the Dakotas. Pintails in the Mississippi Flyway were primarily associated with breeding areas from Saskatchewan and Minnesota to James Bay. Pintails in the Atlantic Flyway were primarily associated with breeding areas from James Bay to the Canadian Maritimes.

If 1980-92 population declines were caused by poor reproduction, production would be lower. Production, however, remained relatively constant over periods of population growth (1963-70), stability (1971-79), and decline

(1980-92) for the Atlantic, Mississippi, and Central flyways (Fig. 2). Production in the Pacific Flyway exhibited a substantial decline from 2.40 in 1963-70, to 1.78 in 1971-79, and to 1.60 in 1980-92.

Likewise, survival would be lower during 1980-92 if population declines were caused by declines in survival. Comparisons of average survival rates between 1980-92 and earlier periods were possible for only a limited number of areas because few pintails were banded in many regions. In the area encompassing northern Alberta, northeastern British Columbia, and southwestern Northwest Territories, average survival during 1980-92 was higher than the average for earlier periods for adult males (80% versus 68%), young males (68% versus 53%), and adult females (69% versus 64%). In southern Alberta, average survival during 1980-92 was higher than the average for earlier periods for adult males (74% versus 70%) and young females (86% versus 55%). Survival remained constant between 1980-92 and earlier periods for all age-classes of pintails banded in southern Saskatchewan and southern Manitoba. In the Dakotas, average survival during 1980-92 was higher for only adult males (77% versus 66%).

These data reveal that possible declines in pintail survival did not cause the population declines observed during the 1980's. Overall, survival was higher during 1980-92 than during earlier periods for adult males that winter in the Pacific, Central, and Mississippi flyways and for young females that winter in the Pacific Flyway. Survival remained constant between time periods for adult females and young males in the Pacific, Central, and Mississippi flyways.

Given the small changes in production and survival, pintail numbers should stabilize in the Central and Mississippi flyways and possibly the Atlantic Flyway. In the Pacific Flyway, however, the survival increases of young females has not compensated for the overall decrease in production.

During the 1980's the Canadian prairies on the average received less precipitation, resulting in reduced availability of pintail breeding habitat. Hopes for increased pintail population size have been based, in part, on the expectation that increased precipitation in the western Canadian prairies would result in increased breeding habitat and production. Female-based age-ratio data suggest, though, that increased production is unlikely to occur even with increased precipitation because pintail production remained low even when water was plentiful. Average age-ratios for the Pacific Flyway when water in the western Canadian prairies was above average (total May ponds for southern Alberta and southern Saskatchewan exceeding 2.68 million) steadily declined from 3.11 in the 1960's, to 2.03 in the 1970's, and 1.86 in the 1980's.

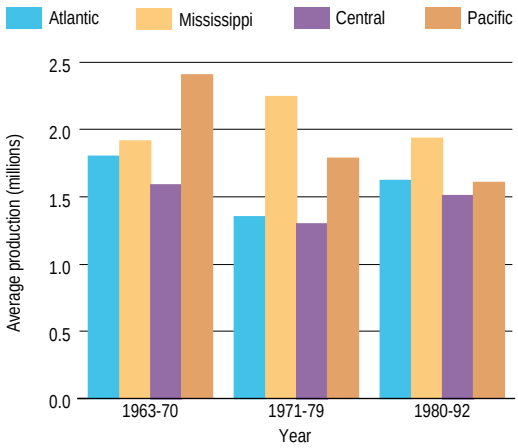

Fig. 2. Average production of pintails in Atlantic, Mississippi, Central, and Pacific flyways for 1963-70, 1971-79, and 1980-92 (Waterfowl Parts Collection Survey, U.S. Fish and Wildlife Service, Office of Migratory Bird Management).

Consequently, a fundamental change appears to have occurred in pintail productivity on western Canadian prairies, meaning that we cannot base pintail management on the hope that increased precipitation will result in a return to the higher levels of production experienced in the 1960's.

Researchers suspect that the production decline may be related to the fact that the shallow-water breeding habitat favored by pintails is most susceptible to agricultural drainage. By 1989, 78% of the pothole margins (the transition zone where potholes meet farmland) and 22% of wet basins were degraded by agricultural activity in prairie Canada (F.D. Caswell and A. Didiuk, Canadian Wildlife Service, personal communication). Increased intensification of agriculture may also contribute to lower production on the prairies through increased grazing and cropping, increased nest destruction, and increased use of agricultural chemicals (Ducks Unlimited 1990). Further research on the western Canadian prairies is necessary to determine specific causes of production declines in pintails and to determine methods to increase production.

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Canvasback Ducks

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Canvasbacks (*Aythya valisineria*) are unique to North America and are one of our most widely recognized waterfowl species. Unlike other ducks that nest and feed in uplands, diving ducks such as canvasbacks are totally dependent on aquatic habitats throughout their life cycle. Canvasbacks nest in prairie, parkland, subarctic, and Great Basin wetlands; stage during spring and fall on prairie marshes, northern lakes, and rivers; and winter in Atlantic, Pacific, and Gulf of Mexico bays, estuaries, and some inland lakes. They feed on plant and animal foods in wetland sediments. Availability of preferred foods, especially energy-rich subterranean plant parts, is probably the most important factor influencing geographic distribution and habitat use by canvasbacks.

In spite of management efforts that have included restrictive harvest regulations and frequent hunting closures in all or some of the flyways (Anderson 1989), canvasback numbers declined from 1955 to 1993 and remain below the population goal (540,000) of the North American Waterfowl Management Plan (USFWS and Canadian Wildlife Service 1994). Causes for this apparent decline are not well understood, but habitat loss and degradation, low rates of recruitment, a highly skewed sex ratio favoring males, and reduced survival of canvasbacks during their first year are considered important constraints on population growth.

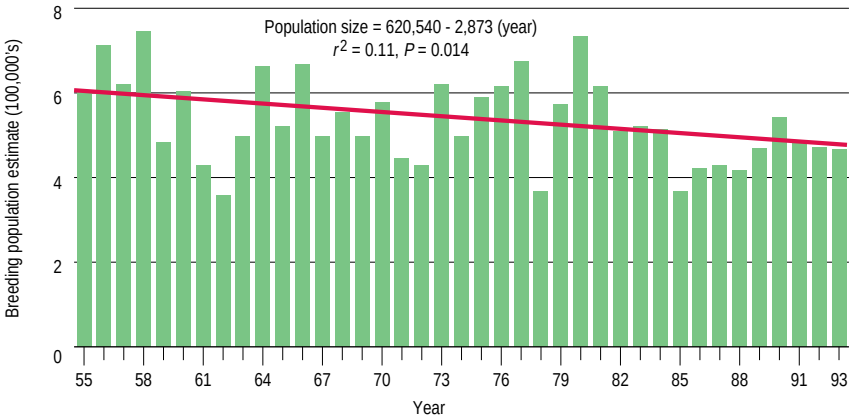

Figure. Estimated breeding population of canvasbacks, 1955-93 (data from the U.S. Fish and Wildlife Service, Office of Migratory Bird Management).

Status and Trends

Canvasback population trends are monitored by means of annual Breeding Waterfowl and Habitat Surveys and Midwinter Waterfowl Inventories (MWI). Readers should refer to cited literature for additional information regarding methods.

Canvasback Numbers and Distribution

Between 1955 and 1993 population indices for canvasbacks fluctuated between 353,700 and 742,400 and averaged 534,000 ducks (Figure). The population showed a general rate of decline of 0.6% per year during the period; however, because population estimates are imprecise, annual differences are difficult to detect. For example, a population change of more than 30% would be needed to detect a significant difference between years with 90% confidence.

The winter distribution of canvasbacks has changed since the 1950's, when most canvasbacks (79%) were found wintering in the Atlantic or Pacific flyways. The proportion of the continental population wintering in the Central and Mississippi flyways increased from 21% in 1955-69 to 44% in 1987-92 as a result of declines in canvasback numbers at Chesapeake Bay and San Francisco Bay and increases in the Gulf of Mexico region. Only about 23,000 canvasbacks winter in Mexico, but numbers may be increasing (Office of Migratory Bird Management, unpublished data). Shifts in winter distribution probably reflect regional differences in habitat availability, but may also indicate differences in survival and recruitment.

Sex Ratios

Canvasbacks have a highly skewed sex ratio favoring males. Sex ratios of wintering canvasbacks in Louisiana (1.6-1.8 males:female; Woolington 1993) and San Francisco Bay (2.2 males:female; J. Takekawa, unpublished data) are lower than those observed in the Atlantic Flyway (2.9-3.2 males:female), but sex ratios apparently decreased in two mid-Atlantic states between 1981 and 1987 (Haramis et al. 1985, 1994). Based on recent (1987-92) MWI and sex ratio data, we calculated that the continental sex ratio for canvasbacks likely lies between 2.0 and 2.5 males:female.

Survival

Annual survival rates of female canvasbacks (56%-69%) are lower than those of males (70%-82%; Nichols and Haramis 1980). Survival rates also vary geographically (survival is greater in the Pacific Flyway than in the Atlantic; Nichols and Haramis 1980) and are positively related to body mass in early winter (Haramis et al. 1986). Survival of females in their first year probably is reduced relative to that of adults. Assuming that all surviving females return to their natal areas to breed, return rates for female canvasbacks breeding in southwestern Manitoba suggest that only 21% of hens survive their first year compared to 69% annual survival of older hens (Serie et al. 1992).

Nichols and Haramis (1980) found no association between canvasback harvest regulations and survival. However, an analysis of return

rates for female canvasbacks in southwestern Manitoba indicated that survival of immatures was significantly related to harvest (M.G. Anderson, Ducks Unlimited-Canada, unpublished data). The canvasback season was closed in the Atlantic, Central, and Mississippi flyways during 1986-93, but about 8,000 birds were harvested annually in Canada and 10,000 in the Pacific Flyway. There is also a substantial illegal harvest of canvasbacks at some sites (Haramis et al. 1993; Korschgen et al. 1993; W.L. Hohman, unpublished data). However, the current level of hunting-related mortality is probably not limiting population growth. Rather, annual variation in recruitment and degradation and loss of breeding, migrational, and wintering habitats are more likely influencing population size.

Courtesy W.L. Hohman, NBS

Canvasback (*Aythya valisineria*).

Time-specific Survival Rates and Sources of Mortality

Survival rates for adults in spring and summer are unknown. In spite of a nationwide ban on the use of lead shot by waterfowl hunters, ingestion of spent lead shotgun pellets by waterfowl is common and likely will remain so for many years. More than 50% of spring-migrating canvasbacks captured at a major staging area on the Mississippi River had elevated blood lead levels (Havera et al. 1992). Lead-exposed birds have reduced body mass, fat, and protein (Hohman et al. 1990), so their subsequent survival and ability to reproduce and perform activities such as courtship, migration, or molt, may be compromised.

Nest success (i.e., embryonic survival) of canvasbacks is highly variable, especially for birds nesting on the prairies. For example, nest success in southwestern Manitoba in wet years was 54%-60%, but in dry years averaged only 17% (Serie et al. 1992). In spite of habitat loss and degradation, ranges in nest success observed in southwestern Manitoba were similar in 1961-72 (21%-62%; Stoudt 1982) and 1974-80 (17%-60%; Serie et al. 1992). Mammalian predation, especially by mink (*Mustela vison*) and raccoon (*Procyon lotor*), is an important factor affecting the nest success of prairie-nesting canvasbacks.

Mortality of prefledged ducklings is high, especially during the first 10 days (C.E. Korschgen, unpublished data). In northwestern Minnesota, estimated survival rates for ducklings up to 10 days old ranged from near zero to 70%, but differed between sexes during the first 25 days of life (male > female; C.E. Korschgen, unpublished data). Predation and weather were the primary sources of duckling mortality. Survival of young between fledging and fall migration is unknown; however, production estimates calculated from harvest information (0.16-1.07 young:adult) suggest that recruitment rates for canvasbacks generally are low compared to other ducks.

Survival rates for fall-migrating canvasbacks have not been studied, but survival rates have been estimated at several major wintering sites. Adult and immature females had high winter survival at Chesapeake Bay (83%-100%; Haramis et al. 1993) and coastal Louisiana (≥ 95%; Hohman et al. 1993). Winter survival was lower at Catahoula Lake, Louisiana (57%-92%), where canvasbacks were not only shot illegally but where substantial numbers of birds were also exposed to lead (W.L. Hohman, unpublished data).

Habitat Trends

Historically, climate, grazing, and fire were major factors affecting habitats of prairie-nesting waterfowl. Since settlement, however, human activities, especially those related to agriculture, have had a major impact on the quantity and quality of breeding habitats. Nationwide, over 53% of original wetlands have been lost. Wetland losses in states where canvasbacks historically nested range from less than 1% (Alaska) to 89% (Iowa); however, deeper wetlands preferred by nesting canvasbacks probably have been drained to a lesser extent than shallower wetlands.

Northern lakes used by canvasbacks for molting and staging before fall migration probably have been least affected by human and natural perturbations. Nonetheless, disturbances related to commercial and recreational activities, nutrient enrichment of lakes resulting from sewage discharges and agricultural runoff, introductions of herbivorous fish, and alteration of lake levels for generation of hydroelectric power have reduced the suitability and use of some traditional staging areas in the southern boreal forest region.

Most of the traditional stopover habitats used by migrating canvasbacks no longer provide suitable feeding and resting opportunities (Kahl 1991). For example, of the more than 40 former migration stopover areas in the upper portion of the Mississippi Flyway, only Lake Christina in west-central Minnesota, two pools on the Upper Mississippi River, and two areas on the Great Lakes have peak populations of more than 5,000 canvasbacks (Korschgen 1989). Restoration efforts begun in 1987 at Lake Christina were successful in reestablishing submersed aquatic vegetation and canvasback use. Habitat on the Upper Mississippi River increased in extent from the mid-1960's to the late 1980's. However, record drought in 1988-89 and extensive flooding in 1993 in the Upper Mississippi River basin have caused major declines in habitat quality and abundance.

In the Great Lakes region, increased bird use of Lake St. Clair and Long Point on Lake Erie coincided with improved water quality and increased production of submersed aquatic plants, especially wildcelery (*Vallisneria americana*). These improvements are attributed to regulation of water discharges into the Great Lakes and perhaps the proliferation of zebra mussels (*Dreissena polymorpha*).

In the Pacific Flyway, coastal habitats used by migrating canvasbacks have not changed greatly since the 1950's, although development has increased in some areas (e.g., Puget Sound). Whereas use of some inland sites (e.g., Great Salt Lake, Utah; Malheur National Wildlife Refuge (NWR), Oregon; and Stillwater NWR, Nevada) declined during the 1970's or 1980's, canvasback use of Klamath Basin NWR, Oregon-California, and Pyramid Lake, Nevada, has increased.

Degradation of water quality in the Chesapeake Bay caused by nutrient enrichment, turbidity, and sedimentation reduced the abundance of aquatic plant and animal foods most important to canvasbacks in winter (Haramis 1991). Declining availability of plant foods caused canvasbacks to shift to mostly animal foods. Canvasback numbers declined in response to loss of aquatic plants in the Chesapeake Bay, but increased in North Carolina and Virginia where preferred plant foods were still abundant (Lovvorn 1989). Aquatic plants are now declining in the coastal areas of North Carolina and other wintering areas throughout the Atlantic Flyway. Unless the widespread decline of aquatic plant foods is reversed, the number of canvasbacks wintering in the Atlantic Flyway is not likely to increase.

San Francisco Bay is the most important wintering area for canvasbacks in the Pacific Flyway. Urban development there has greatly reduced available habitat. In remaining habitats, canvasbacks are exposed to high levels of environmental contaminants (Miles and Ohlendorf 1993). Canvasbacks make extensive use of salt evaporation ponds in northern San Francisco Bay (Accurso 1992). These ponds recently came under public ownership, but their management as tidal salt marshes will probably reduce their use by canvasbacks. Increasing numbers of canvasbacks have been observed recently on wetland easements and sewage lagoons in the northern San Joaquin Valley.

Increased numbers of canvasbacks are wintering in the Gulf of Mexico region, especially at Catahoula Lake, where, since 1985, peak numbers (up to 78,000 birds) have equaled or exceeded counts on traditional wintering areas such as Chesapeake Bay and San Francisco Bay. Birds appear to be attracted to Catahoula Lake because of its abundant plant foods and stable flooding regime (Woolington and Emfinger 1989). These birds are at risk of lead poisoning, however, because of the high density of spent lead shot contained in lake sediments.

Information Gaps

Information needs for improved management of canvasbacks include banding or radio-telemetry data sufficient to provide habitat information and estimates of region-specific rates of survival, band recovery, and recruitment; survival rates of immature birds between hatch and arrival on wintering areas; and cross-seasonal effects of winter nutrition and contaminant exposure on reproduction.

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Breeding Seabirds in California, Oregon, and Washington

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More than two million seabirds of 29 species nest along the west coasts of California, Oregon, and Washington, including three species listed on the federal list of threatened and endangered species: the brown pelican (*Pelecanus occidentalis*), least tern (*Sterna antillarum*), and marbled murrelet (*Brachyramphus marmoratus*). The size and diversity of the breeding seabird community in this region reflect excellent nearshore prey conditions; subtropical waters within the southern California Bight area; complex tidal waters of Strait of Juan de Fuca and Puget Sound in Washington; large estuaries at San Francisco Bay, Columbia River, and Grays Harbor-Willapa bays; and the variety of nesting habitats used by seabirds throughout the region, including islands, mainland cliffs, old-growth forests, and artificial structures.

Breeding seabird populations along the west coast have declined since European settlement began in the late 1700's because of human occupation of, commercial use of, and introduction of mammalian predators to seabird nesting islands. In the 1900's, further declines occurred in association with rapid human population growth and intensive commercial use of natural resources in the Pacific region. In particular, severe adverse impacts have occurred from partial or complete nesting habitat destruction on islands or the mainland, human disturbance of nesting islands or areas, marine pollution, fisheries, and logging of old-growth forests (Ainley and Lewis 1974; Bartonek and Nettleship 1979; Hunt et al. 1979; Sowls et al. 1980; Nettleship et al. 1984; Speich and Wahl 1989; Ainley and Boekelheide 1990; Sealy 1990; Ainley and Hunt 1991; Carter and Morrison 1992; Carter et al. 1992; Vermeer et al. 1993).

Methods

Population status of breeding seabirds on the west coast has been measured primarily through the determination of and trends in population size, based on counts of birds and nests at nesting colonies (e.g., Sowls et al. 1980). At-sea surveys also have been used to approximate population sizes for breeding and nonbreeding populations and species as well as their foraging distribution alongshore and offshore (e.g., Briggs et al. 1987). Rather than just monitoring small plots of nests on a few accessible islands to determine status and trends, relatively accurate and standardized censuses of entire coastal seabird breeding populations (except for certain nesting areas of difficult-to-census species) have been conducted annually or periodically to determine the overall status of many species breeding on the west coast (Figs. 1-4). However, we have considered census accuracy, natural variability, trends at well-studied colonies (e.g., Farallon National Wildlife Refuge) and many other factors in assessing population status and trends.

Status and Trends

Storm-petrels (Hydrobatidae)

Increasing numbers of Leach's storm-petrels (*Oceanodroma leucorhoa*) have been documented recently in Oregon (R.W. Lowe, USFWS, unpublished data), although this

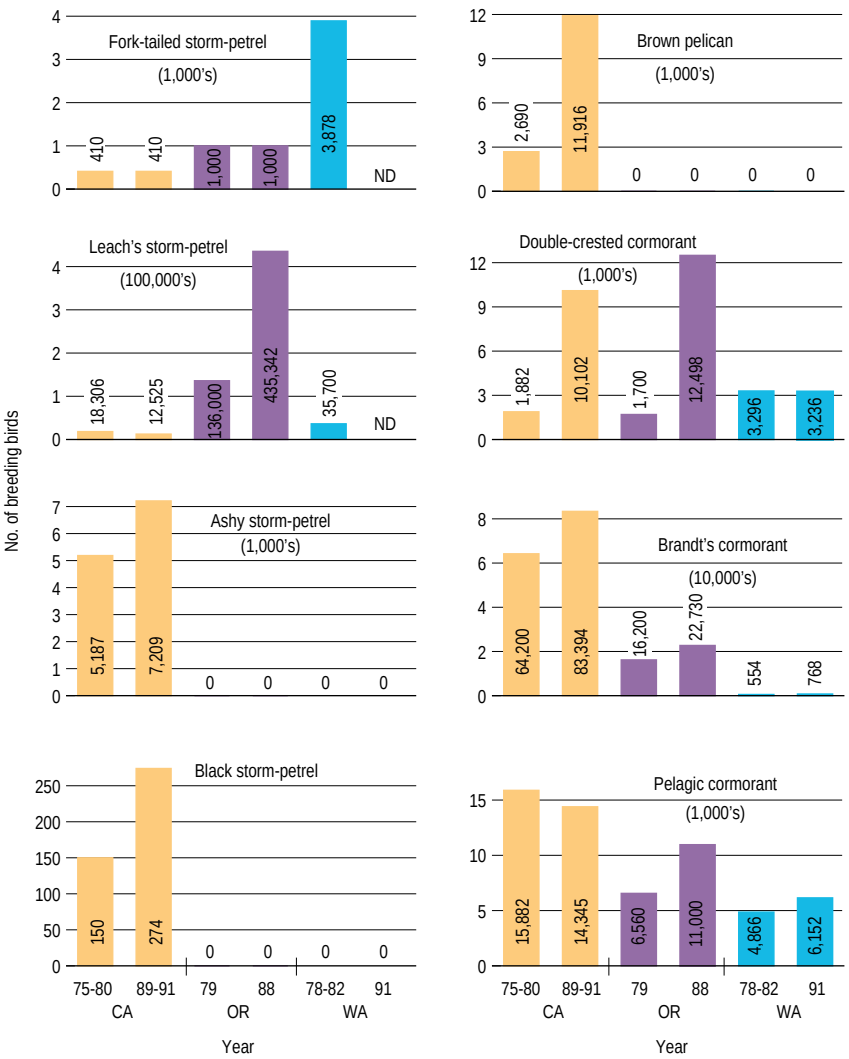

Fig. 1. Status and trends of breeding populations of storm-petrels, pelicans, and cormorants on the west coasts of California, Oregon, and Washington. Data for small inland populations of white pelicans and double-crested cormorants are not included. ND — no data available; 0 — no coastal nesting. Sources: CA (Hunt et al. 1979; Sowls et al. 1980; Carter et al. 1992); OR (Varoujean and Pitman 1979; R.W. Lowe, unpublished data); and WA (Speich and Wahl 1989; U.W. Wilson, unpublished data). Also *see* Carter et al. (in press) for double-crested cormorant.

increase probably represents greater survey effort (Fig. 1). They have declined in northern California because of the loss of burrow-nesting habitats due to soil erosion and defoilation by nesting cormorants (Carter et al. 1992). Ashy storm-petrels (*O. homochroa*) have declined recently at the world's largest known colony at the South Farallon Islands, possibly because of high gull predation (W.J. Sydeman, Point Reyes Bird Observatory, unpublished data). This decline is of concern because the small world population of this species (fewer than 10,000 breeding birds) nests entirely in California. Greater numbers of ashy and black storm-petrels (*O. melania*) have been documented recently in southern California, although this probably reflects greater survey efforts (Carter et al. 1992). In Fig. 1, similar numbers of fork-tailed storm-petrels (*O. furcata*) are indicated over the past decade in Oregon and California because survey efforts confirmed very small numbers. Declines in California are suspected (Carter et al. 1992), but further work is required to establish trends.

Pelicans (Pelecanidae)

Brown pelicans have increased recently at the only two remaining colonies (West Anacapa and Santa Barbara islands) in the Channel Islands in southern California (Fig. 1), following severe pre-1975 declines primarily due to eggshell thinning from marine pollutants (Anderson et al. 1975; Anderson and Gress 1983; Carter et al. 1992; F. Gress and D.W. Anderson, University of California-Davis, personal communication). Breeding success is still low and limited recovery may involve immigration of birds out of Mexico. Concern exists for adverse effects of continuing low levels of marine pollutants, commercial fisheries, and the 1990 *American Trader* oil spill. Although the brown pelican has shown recent population increases, white pelicans have been extirpated from parts of interior California and have declined at inland colonies in northern California because of low reproduction related to water developments and drought (Carter et al. 1992; P. Moreno and D.W. Anderson, University of California-Davis, personal communication). Small colonies still exist at Sheepy Lake and Clear Lake in the Klamath Basin area. These conditions also exist at other inland areas in Oregon, Washington, and Nevada, but problems seem fewer farther east.

Cormorants (Phalacrocoracidae)

Double-crested cormorants (*Phalacrocorax auritus*) have increased dramatically in coastal regions of California and Oregon (Fig. 1) because of reduced human disturbance, reduced levels of marine pollutants in southern California, and recent use of artificial nesting areas in San Francisco Bay and Columbia River estuaries (Gress et al. 1973; Carter et al. 1992). They have not increased in Puget Sound because of high human disturbance and predation by bald eagles (*Haliaeetus leucocephalus*), which has caused colony abandonments (Henny et al. 1989; Speich and Wahl 1989; Carter et al. in press; U.W. Wilson, unpublished data). Declines have been reported at interior colonies in California, Oregon, and Washington due to water developments, human disturbance at colonies, and large-scale shooting of birds at hatcheries (during smolt releases) and at aquacultural facilities (Carter et al. in press; R.W. Lowe, unpublished data; R. Bayer, personal communication; P. Moreno, unpublished data). Brandt's and pelagic cormorant (*P. penicillatus* and *P. pelagicus*) populations have fluctuated in response to El Niño conditions (Ainley and Boekelheide 1990; Ainley et al. 1994). At the South Farallon Islands, these cormorants appear very sensitive to El Niño conditions, which result in quite poor reproduction and mortality

of subadult and adult birds (Boekelheide and Ainley 1989; Ainley and Boekelheide 1990). Overall, numbers have remained stable or increased in most areas in the region (e.g., Carter et al. 1992), whereas these birds now occur at lower abundance than previously at the South Farallon Islands (Ainley et al. 1994). Numbers have increased in southern California, but the birds have suffered from gill-net and oil-spill mortality as well as human disturbance at colonies (H.R. Carter, unpublished data).

Gulls, Terns, and Skimmers (Laridae and Rynchopidae)

The predominant nesting gull on the west coast is the western gull (*Larus occidentalis*). Numbers have increased, especially in California (Fig. 2), probably because of the bird's use of human and fishing refuse and reduced human disturbance. Numbers have reached saturation at the world's largest colony at the South Farallon Islands (Ainley et al. 1994); however, expansion is occurring at other major colonies in central and southern California (Carter et al. 1992). Glaucous-winged gulls (*L. glaucescens*) have remained stable or increased in Puget Sound (U.W. Wilson, unpublished data).

California gulls (*L. californicus*) have recently expanded from interior colonies to nest in San Francisco Bay (Fig. 2; Carter et al. 1992; P. Woodin, San Francisco Bay Bird Observatory, unpublished data). They face serious threats at inland colonies in interior California because of water developments. At the world's largest colony at Mono Lake, low water levels have resulted in the formation of land bridges to nesting islands, allowing access by coyotes (*Canis latrans*) in certain years (Jones and Stokes Associates 1993). Similar problems exist at other northern California colonies for many seabird and colonial waterbird species (W.D. Shuford, Point Reyes Bird Observatory, unpublished data).

The status of California gulls at inland colonies in Oregon and Washington is not well known. Status and trends of inland colonies of ring-billed gulls (*L. delawarensis*) in California, Oregon, and Washington are not known, although problems related to low water levels may occur at many colonies. Many thousands have nested recently in northern California (W.D. Shuford, unpublished data). Small numbers (< 500 breeding birds) also nest along the Washington coast (Speich and Wahl 1989). Small numbers (< 10 breeding birds) of Heermann's gulls (*L. heermanni*) nested in the early 1980's along the central California coast but none are known to do so now. Franklin's gulls (*L. pipixcan*) recently nested in small numbers (< 100 breeding birds) at Lower Klamath Lake, California, but their status in the region is not known.

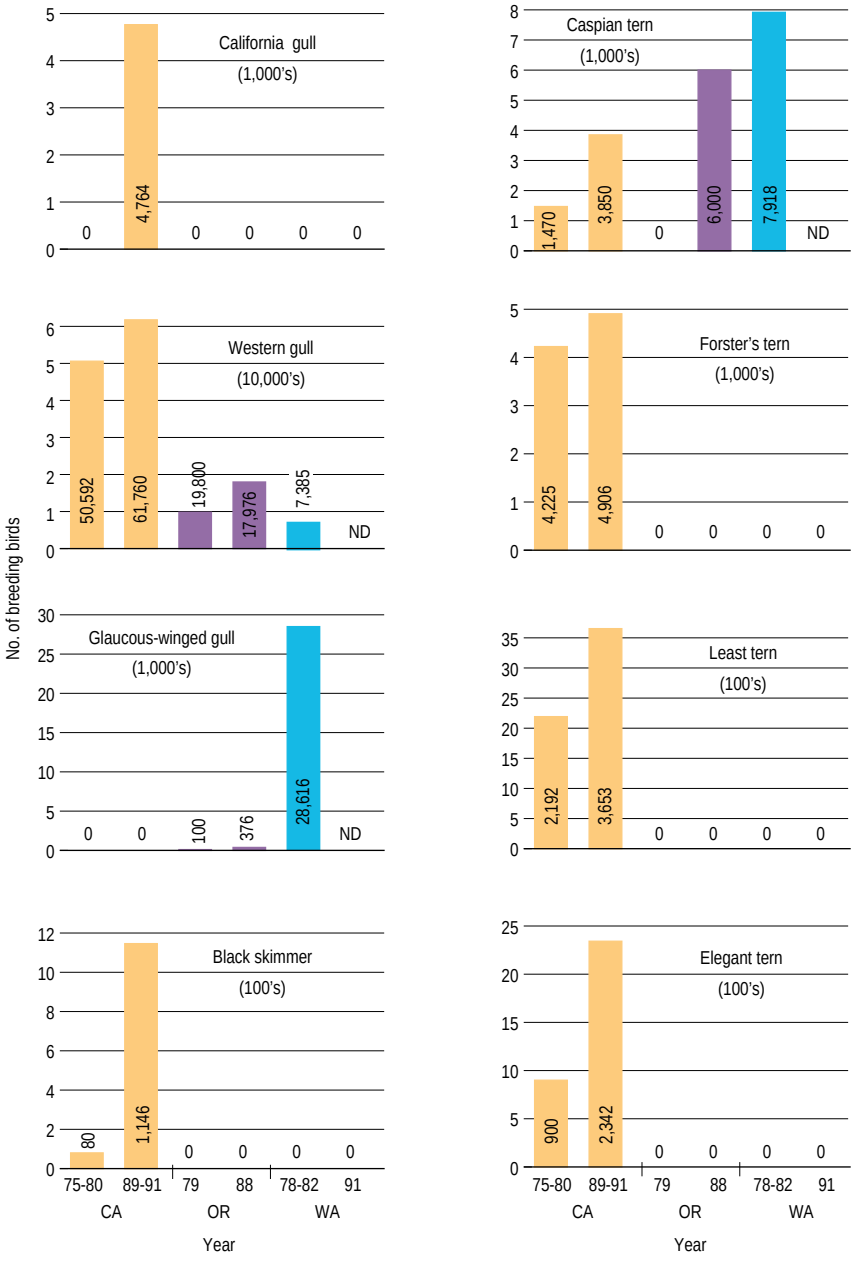

Fig. 2. Status and trends of breeding populations of gulls, terns, and skimmers. Small coastal populations of gulls (Heermann's and ring-billed) and royal terns, as well as large or small inland populations of gulls (ring-billed and California), terns (black, gull-billed, Caspian, and Forster's), and black skimmers are not included. ND — no data available; 0 — no coastal nesting. Sources: CA (Hunt et al. 1979; Sowls et al. 1980; Carter et al. 1992); OR (Varoujean and Pitman 1979; R.W. Lowe, unpublished data); and WA (Speich and Wahl 1989; U.W. Wilson, unpublished data). Also *see* Carter et al. (in press) for double-crested cormorant.

Low thousands of Caspian, Forster's, least, and elegant terns (*Sterna caspia*, *S. forsteri*, *S. antillarum*, *S. elegans*) and black skimmers (*Rynchops niger*) now occur in the region through increases (especially along the southern California coast) due to colony protection and use of artificial nesting sites (Speich and Wahl 1989; Carter et al. 1992). Certain tern colonies have been eliminated or shifted (especially in San Francisco Bay) because of human disturbance and red fox (*Vulpes vulpes*) or other mammalian predation (P. Woodin, unpublished data). Overall, least tern colonies in California appear somewhat stable because of extensive management. They undoubtedly occur at lower

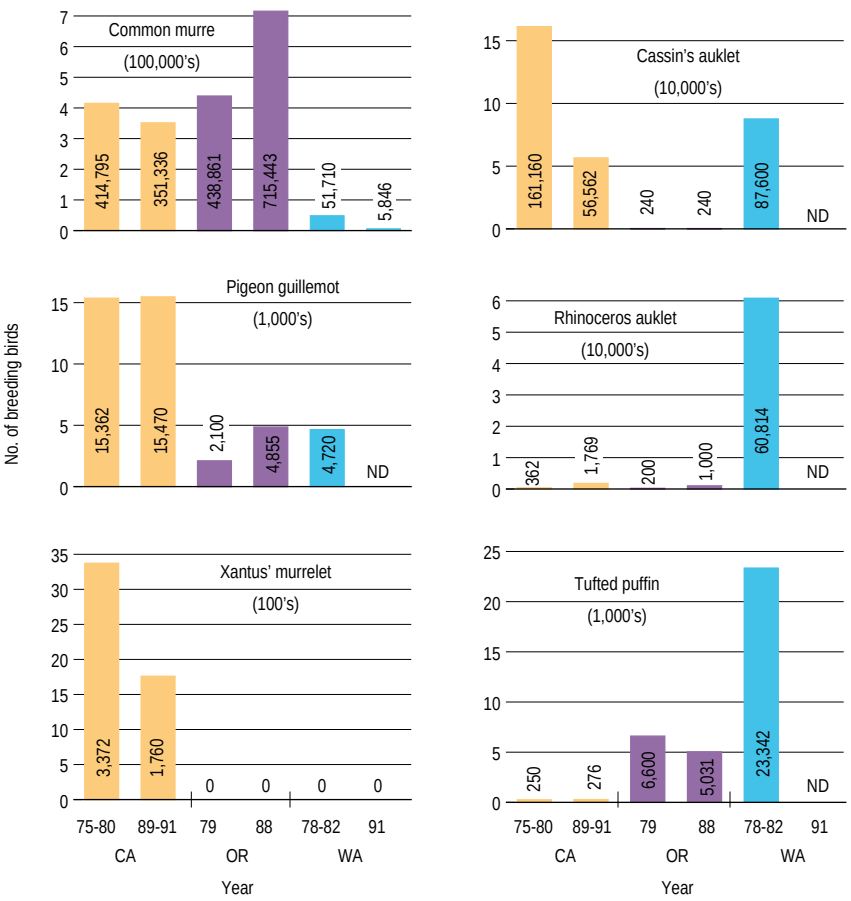

Fig. 3. Status and trends of breeding populations of several alcids in California, Oregon, and Washington. Data for marbled murrelets and historical nesting by ancient murrelets are not included. ND — no data available; 0 — no coastal nesting. Sources: CA (Hunt et al. 1979; Sowls et al. 1980; Carter et al. 1992); OR (Varoujean and Pitman 1979; R.W. Lowe, unpublished data); and WA (Speich and Wahl 1989; U.W. Wilson, unpublished data). Also *see* Carter et al. (in press) for double-crested cormorant.

than historical levels because of loss of nesting habitat, which continues to be threatened (Carter et al. 1992; R. Jurek, California Department of Fish and Game, personal communication). Low numbers (< 100 breeding birds) of arctic terns (*S. paradisaea*) have nested in coastal Washington in the past but not now (Speich and Wahl 1989). Small numbers (< 100 breeding birds) of gull-billed and royal terns (*S. nilotica* and *S. maxima*) recently colonized the southern California coast, although gull-billed terns have nested inland at the Salton Sea for a few decades. The status of black terns (*Chlidonias niger*) is not known.

Alcids (Alcidae)

Pigeon guillemot (*Cepphus columba*) populations have remained stable overall (Fig. 3), but major fluctuations have occurred in response to El Niño events at the South Farallon Islands and on the Oregon coast (Hodder and Graybill 1985; Ainley and Boekelheide 1990). A significant population and new nesting areas have been found recently in southern California, although higher numbers reflect both better survey techniques and population increases (Carter et al. 1992). Ancient murrelets (*Synthliboramphus antiquus*) nested on the Washington coast in the early 1900's but no longer do (Speich and Wahl 1989). Cassin's auklets (*Ptychoramphus aleuticus*) have declined at the largest known colony in the region at the South Farallon Islands, probably because of high gull predation and loss of burrow-nesting habitat from soil erosion (Carter et al. 1992; W.J. Sydeman, unpublished data). However, lower numbers also were found at Prince Island in southern California where numbers of nesting gulls are lower. Differences in survey techniques probably account for part of the lower numbers found recently, but other data on soil conditions, densities of nesting gulls, and gull predation support a decline at the South Farallon Islands (W.J. Sydeman, unpublished data). Hundreds also were killed in the 1984 *Puerto Rican* and 1986 *Apex Houston* oil spills (Ford et al. 1987; Page et al. 1990).

Rhinoceros auklets (*Cerorhinca monocerata*) have increased throughout the region. Largest numbers occur at Protection and Destruction islands, but burrow occupancy has fluctuated widely between years (Wilson and Manuwal 1986; U.W. Wilson, unpublished data). The South Farallon Islands were recolonized after a 100-year absence in the early 1970's (Ainley and Lewis 1974) and reached saturation levels by the late 1980's (Carter et al. 1992; Ainley et al. 1994). Nesting has recently extended to the Channel Islands (Carter et al. 1992). Thousands of rhinoceros auklets were killed in the 1986 *Apex Houston* oil spill (Page et al. 1990).

The largest tufted puffin (*Fratercula cirrhata*) populations occur along the west coast of the Olympic Peninsula (Speich and Wahl 1989), but their status there is not well known. In Puget Sound, this species has declined substantially (U.W. Wilson, unpublished data). At small colonies in Oregon and California, their numbers appear stable (Carter et al. 1992; Fig. 3), despite impacts due to El Niño at the South Farallon Islands (Ainley and Boekelheide 1990; Ainley et al. 1994). They have recently recolonized southern California where they have not nested since the early 1900's (Carter et al. 1992).

Common murres (*Uria aalge*) are the dominant member of the breeding seabird community on the west coast but they have declined substantially in central California and Washington (Figs. 3, 4) because of the combined effects of high mortality from gill-net fishing and oil spills plus poor reproduction during intense El Niño events. In central California, large historical declines in the late 1800's and early 1900's almost led to the extinction of this population (Ainley and Lewis 1974). Population growth occurred, however, between the 1950's and the 1970's, producing about 230,000 breeding birds by 1980-82 (Takekawa et al. 1990). Over 70,000 murres were estimated to have been

killed in gill nets in central California between 1979 and 1987, before heavy fishing restrictions were imposed in 1987 to stop mortality (Takekawa et al. 1990). Additional mortality (10,000+ murres) occurred during the 1984 *Puerto Rican* and 1986 *Apex Houston* oil spills (Ford et al. 1987; Page et al. 1990). At the South Farallon Islands, reproductive success was almost nil during intense El Niño events in 1983 and 1992 (Ainley and Boekelheide 1990; W.J. Sydeman, unpublished data). Because of these and other factors, the central California population declined by over 60% from 1982 to 1989 and has not recovered (Fig. 4; Takekawa et al. 1990; Carter et al. 1992; Ainley et al. 1994; H.R. Carter, unpublished data).

In Washington, murre numbers crashed during the 1982-83 El Niño (Wilson 1991), although there was heavy mortality from gill nets at this time; mortality from gill nets still continues in Puget Sound. In addition, certain colonies have been disturbed by low-flying aircraft, especially near military bases. Numbers of breeding murres in Washington are lower than indicated in Figs. 3 and 4 because many birds counted in colonies in recent years (and used to derive estimates) do not appear to be breeding (U.W. Wilson, unpublished data). Significant mortality occurred during the 1984 *Arco Anchorage*, 1988 *Nestucca*, and 1991 *Tenyo Maru* oil spills. In the *Nestucca* spill alone, about 30,000 murres were estimated to have died (Ford et al. 1991). The Washington population of murres has been almost extirpated over the last decade and has not recovered.

In contrast, murre populations in Oregon and northern California have been stable or increasing to date, despite human disturbance at several colonies (Takekawa et al. 1990; R.W. Lowe, unpublished data) and some losses of Oregon birds from oil spills and the use of gill nets in Washington. In addition, these areas were known to experience lower productivity through colony abandonment during intense El Niño conditions in 1993 (Fig. 4; H.R. Carter, unpublished data; J.E. Takekawa and R.W. Lowe, unpublished data). Thus, it appears clear that decline and lack of recovery of populations in central California and Washington have resulted primarily from human causes, especially gill nets and oil spills.

Marbled murrelets probably have declined substantially throughout the region largely because of the direct loss of most (90%-95%) of their old-growth forest nesting habitat to large-scale logging since the mid-1800's (Carter and Morrison 1992; FEMAT 1994). About 10,000-20,000 birds remain. In addition, hundreds of murrelets have been killed in gill nets and oil spills in central California, Puget Sound, and off the Olympic Peninsula (Carter and Morrison 1992; H.R. Carter, unpublished data). Murrelets appear to have very low reproductive rates (based on nests examined and at-sea counts of juveniles), probably because of high avian nest predation in fragmented forests and possibly lower breeding success during intense El Niño events. This species was listed as threatened in California, Oregon, and Washington in 1992, and is being considered carefully with regard to the future of old-growth forests and the timber industry in this region. Small populations in California, Oregon, and southwestern Washington are isolated and susceptible to extinction from various potential disturbances in the future.

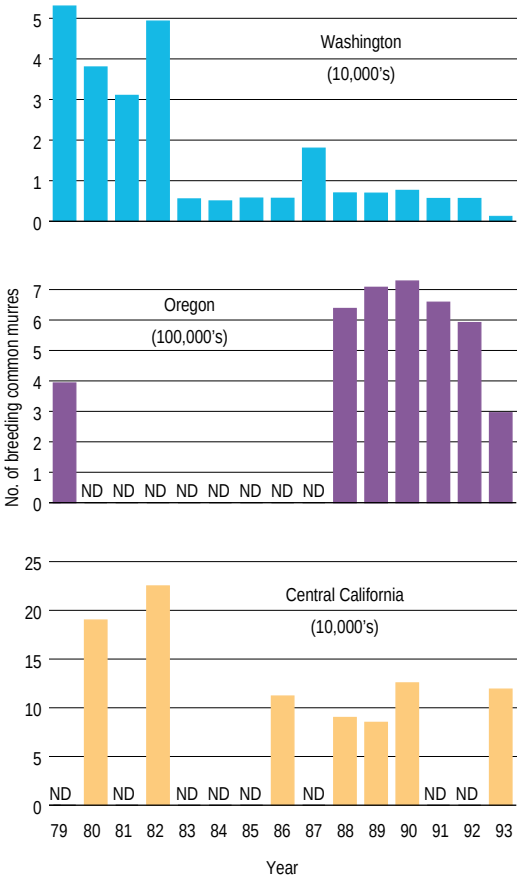

Fig. 4. Status and trends of breeding populations of common murres in Washington, Oregon, and central California. ND — no data available. Sources: WA (Wilson 1991; U.W. Wilson, unpublished data); OR (Varoujean and Pitman 1979; R.W. Lowe, unpublished data); and Central CA (Takekawa et al. 1990; Carter et al. 1992; H.R. Carter and J.E. Takekawa, unpublished data).

The Xantus' murrelet (*Synthliboramphus hypoleucus*) persists in very low numbers (2,000-5,000 breeding birds) only in southern California. Numbers breeding at the largest colony at Santa Barbara Island probably have declined between the mid-1970's and 1991 (Fig. 3; Carter et al. 1992). The decline may have occurred because of many factors, including census differences. Poor reproduction, however, has occurred because of high levels of avian and mammalian predation and has probably led to this decline. Other smaller colonies may disappear because of mortality from oil spills from offshore platforms in Santa Barbara Channel and oil tanker traffic into Los Angeles

harbor and other factors. Larger numbers of nesting birds are now suspected in southern California (H.R. Carter, unpublished data). A significant portion of the small world population of this species nests in southern California while the remainder nests on the northwest coast of Baja California, Mexico. This candidate species may be considered for federal and state listing in the near future.

Future Efforts

Because of the continuing decline of and threats to seabirds on broad regional and local levels along the west coast, efforts to determine status and trends of seabirds must be extended beyond current levels. Long-term efforts must be shared among many federal and state agencies, universities, and private groups, including (1) the development of a coordinated long-term monitoring and research program, including data-base development and maintenance; (2) extending monitoring to all coastal and inland areas and species; (3) developing new methodologies for surveying nocturnal species of murrelets, auklets, and storm-petrels; (4) conducting studies of specific conservation problems such as loss of nesting habitats (e.g., old-growth forests), gill-net mortality (e.g., Puget Sound), oil-spill mortality, human disturbance, water developments, and agricultural practices; (5) restoring lost or depleted seabird colonies and habitats; and (6) examining the possible long-term effects of human fisheries and global climate change on seabird prey resources and nesting habitats.

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Seabirds in Alaska

by
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About 100 million seabirds reside in marine waters of Alaska during some part of the year. Perhaps half this population is composed of 50 species of nonbreeding residents, visitors, and breeding species that use marine habitats only seasonally (Gould et al. 1982). Another 30 species include 40-60 million individuals that breed in Alaska and spend most of their lives in U.S. territorial waters (Sowls et al. 1978). Alaskan populations account for more than 95% of the breeding seabirds in the continental United States, and eight species nest nowhere else in North America (USFWS 1992).

Seabird nest sites include rock ledges, open ground, underground burrows, and crevices in cliffs or talus. Seabirds take a variety of prey from the ocean, including krill, small fish, and squid. Suitable nest sites and oceanic prey are the most important factors controlling the natural distribution and abundance of seabirds.

The impetus for seabird monitoring is based partly on public concern for the welfare of these birds, which are affected by a variety of human activities like oil pollution and commercial fishing. Equally important is the role seabirds serve as indicators of ecological change in the marine environment. Seabirds are long-lived and slow to mature, so parameters such as breeding success, diet, or survival rates often give earlier signals of changing environmental conditions than population size itself. Seabird survival data are of interest because they reflect conditions affecting seabirds in the nonbreeding season, when most annual mortality occurs (Hatch et al. 1993b).

Techniques for monitoring seabird populations vary according to habitat types and the breeding behavior of individual species (Hatch and Hatch 1978, 1989; Byrd et al. 1983). An affordable monitoring program can include but a few of the 1,300 seabird colonies identified in Alaska, and since the mid-1970's, monitoring efforts have emphasized a small selection of surface-feeding and diving species, primarily kittiwakes (*Rissa* spp.) and murres (*Uria* spp.). Little or no information on trends is available for other seabirds (Hatch 1993a). The existing monitoring program occurs largely on sites within the Alaska Maritime National Wildlife Refuge, which was established primarily for the conservation of marine birds. Data are collected by refuge staff, other state and federal agencies, private organizations, university faculty, and students.

Status of Monitored Birds

Kittiwakes

Kittiwakes are small, pelagic (open sea) gulls that range widely at sea and feed on a variety of small fish and plankton, which they capture at the sea surface. Black-legged kittiwakes (*Rissa tridactyla*) have been studied intensively because they are widely distributed and easy to observe. Among 10 locations for which population trend data are available, 3 show significant declines since the mid-1970's, 3 show increases, and 4 show no consistent trends (Fig. 1). The overall trend is unknown, although widespread declines are anticipated because of a downward

Courtesy S. Hatch, NBS

Dense colonies of common murres (*Uria aalge*) breed on bare cliff ledges—here on the Semidi Islands, western Gulf of Alaska.

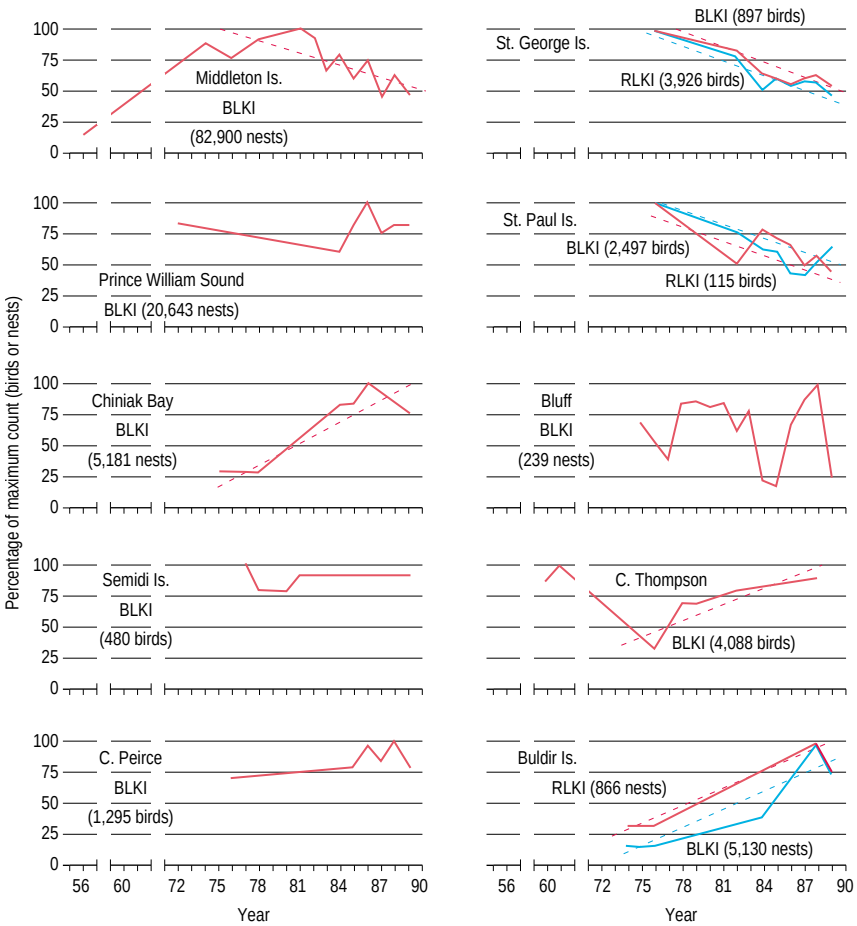

Fig. 1. Population trends of black-legged kittiwakes (BLKI) and red-legged kittiwakes (RLKI) at selected colonies in Alaska. The maximum count of birds or nests is indicated for each location. Dashed lines indicate significant regressions ($P < 0.05$) of data collected since 1970 (P is a measure of the confidence that the decline or increase is statistically reliable. $P < 0.05$ indicates a high probability that the population trend depicted actually occurred). *See* Hatch et al. 1993a and references cited therein for data sources.

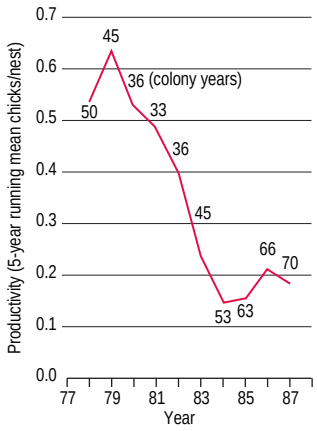

Fig. 2. Productivity (chicks fledged per nest built) of black-legged kittiwakes in Alaskan colonies, 1976-89. The number of colony-years included in each mean is indicated. *See* Hatch et al. 1993a for raw data.

trend in the production of offspring (Fig. 2); some large colonies fail chronically. On Middleton Island, for example, breeding has been a total or near-total failure in 10 of the last 12 years (1983-94; Hatch et al. 1993a; Hatch, unpublished data). The colony is declining at an average rate of 7% per year (equal to adult mortality), suggesting there is no recruitment (Hatch et al. 1993b). If survival estimates obtained on Middleton apply generally, the near-term future of kittiwakes is unfavorable because average productivity of 0.2 chicks per pair (Fig. 2) is inadequate to maintain populations.

Where red-legged kittiwakes (*R. brevirostris*) have been monitored, they show population trends similar to black-legged kittiwakes (Fig. 1). In 1989 their population was down by 50% in the Pribilof Islands, but they were more numerous at Buldir Island than in the mid-1970's (Byrd and Williams 1993). Because most of the world population of red-legged kittiwakes breeds in the Pribilofs (75% on St. George Island), their decline at that location is cause for concern.

Murres

Murres are large-bodied, abundant, and wide-ranging seabirds that feed mostly on schools of fish they pursue by diving underwater, sometimes to depths of 100-200 m (330-650 ft). Repeated counts of one or both murre species (common murre, *Uria aalge*, and thick-billed murre, *U. lomvia*) are available for 12 locations in Alaska (Fig. 3). Since 1970 common murres have declined significantly at two colonies, and thick-billed murres have declined at one. Murres (species not distinguished) increased at two colonies over the same period. Between the 1950's and the 1970's, murres increased at one location (Middleton Island) and declined at another (Cape Thompson), but they have since been relatively stable at both colonies. In 1989 the *Exxon Valdez* oil spill killed substantial numbers of common murres at several colonies in the Gulf of Alaska (Piatt et al. 1990a).

Available data are insufficient to identify overall trends. Murres are relatively consistent producers of young, averaging 0.5-0.6 chicks per pair annually in both species (Byrd et al. 1993).

Threatened and Endangered Species

No breeding seabirds are currently listed as threatened or endangered in Alaska. The short-tailed albatross (*Diomedea albatrus*), with fewer than 1,000 individuals surviving, breeds in Japan but visits Alaskan waters during most months of the year. The species is vulnerable to incidental take by commercial fishing gear, especially gill nets and longlines (Sherburne 1993).

Three species that breed in Alaska were recently listed as category 2 (possibly qualifying for threatened or endangered status, but more information is needed for determination): the red-legged kittiwake, marbled murrelet (*Brachyramphus marmoratus*), and Kittlitz's murrelet (*B. brevirostris*). As noted previously, red-legged kittiwakes have declined substantially on the Pribilof Islands (Fig. 1). Marine bird surveys conducted in Prince William Sound in 1972-73 and 1989-93 suggest a significant decline of marbled murrelets in that area (Klosiewski and Laing 1994). This finding is corroborated by Audubon Christmas Bird Counts from coastal sites in Alaska, which reveal a downward trend since 1972 (Piatt, unpublished data). Kittlitz's murrelet also showed a decline in the Prince William Sound surveys (Klosiewski and Laing 1994). With an estimated population of fewer than 20,000 birds range-wide (van Vliet 1993), this species is one of the rarest of auks (Family Alcidae). Both murrelets were adversely affected by the *Exxon Valdez* oil spill (Piatt et al. 1990a).

Other Species

Scant information is available to assess numerical changes for most seabird species in Alaska. We know that some species were seriously reduced or locally extirpated by foxes introduced to islands in the 1800's and early 1900's. About 450 islands from southeastern Alaska to the western Aleutians were used as release sites for arctic (*Alopex lagopus*) and red foxes (*Vulpes vulpes*) (Bailey 1993). The species most affected included open-ground nesters such as gulls (*Larus* spp.), terns (*Sterna* spp.), and fulmars (*Fulmarus glacialis*), and burrowing birds like ancient murrelets (*Synthliboramphus antiquus*), Cassin's auklets (*Ptychoramphus aleuticus*), tufted puffins (*Fratercula cirrhata*), and storm-petrels (*Oceanodroma* spp.). In spite of natural die-offs and eradication efforts, foxes remain on about 50 islands to which they were introduced (Bailey 1993).

Recent counts suggest that fulmars are increasing at two of their major colonies (Semidi Islands and Pribilof Islands), and several small colonies have been established since the mid-1970's (Hatch 1993b). Counts of least and crested auklets (*Aethia pusilla* and *A. cristatella*) also indicate possible increases at two colonies in the Bering Sea (Piatt et al. 1990b; Springer et al. 1993).

Red-faced cormorants (*Phalacrocorax urile*) declined about 50% on the Semidi Islands between 1978 and 1993, while pelagic cormorants (*P. pelagicus*) increased on Middleton Island between 1956 and the mid-1970's (Hatch, unpublished data). Glaucous-winged gulls (*Larus glaucescens*) increased on Middleton from none breeding in 1956 to more than 20,000 birds in 1993 (Hatch, unpublished data); this species has also shown marked increases following removal of introduced foxes at several sites in the Aleutian Islands (Byrd et al. 1994). Marine bird surveys in Prince William Sound (Klosiewski and Laing 1994) suggest that arctic terns (*Sterna paradisaea*), glaucous-winged gulls, pelagic cormorants, horned puffins (*Fratercula corniculata*), and pigeon guillemots (*Cepphus columba*) have all declined in that area. Terns and guillemots have recently increased on several Aleutian Islands following fox removal (Byrd et al. 1994).

Factors Affecting Seabirds

Alaskan seabirds are killed incidentally in drift gill nets used in high seas (DeGange et al. 1993), and oil pollution poses a significant threat, as demonstrated by the *Exxon Valdez* spill. There is little doubt, however, that the introduction of exotic animals, especially foxes, but also rats, voles, ground squirrels, and rabbits has been the most damaging source of direct mortality associated with human activity (Bailey 1993). Unlike one-time catastrophes, introduced predators exert a continuous negative effect on seabird populations.

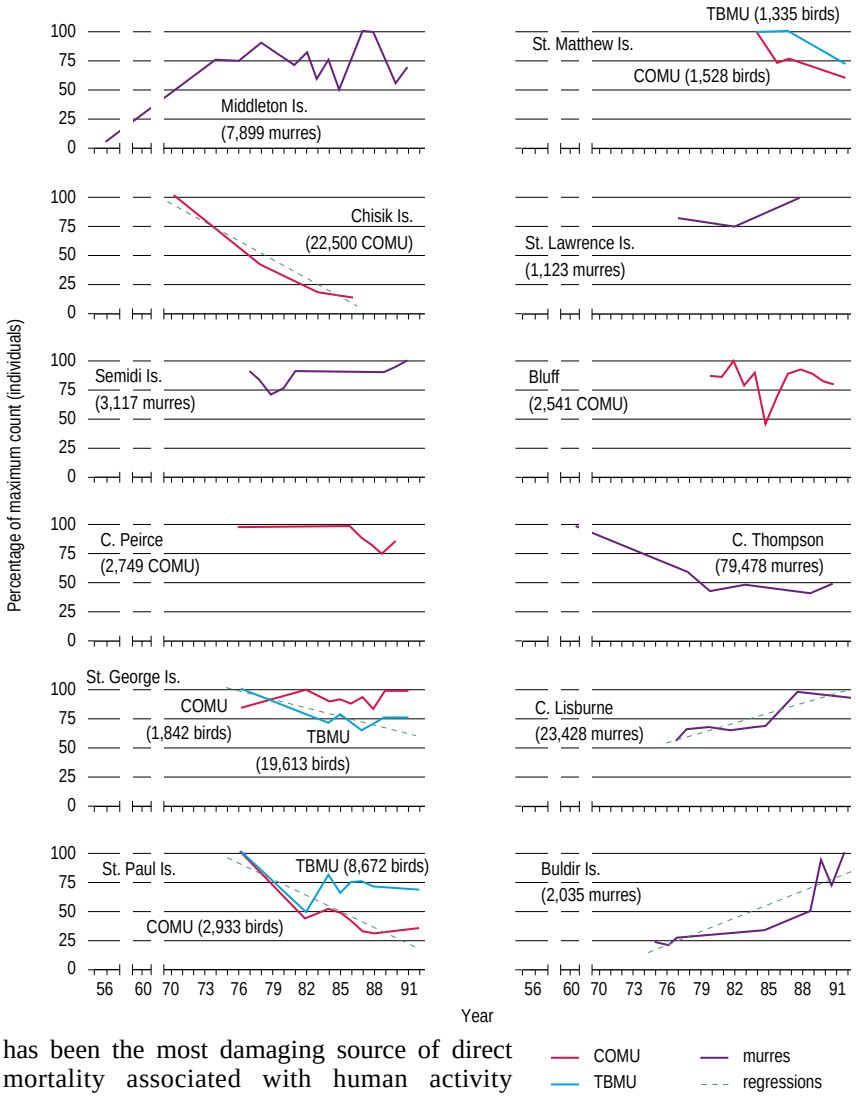

Fig. 3. Population trends of common murres (COMU) and thick-billed murres (TBMU) at selected colonies in Alaska. Counts of "murres" included unspecified numbers of common and thick-billed murres. The maximum count of individuals is indicated for each location. Dashed lines indicate significant regressions ($P < 0.05$) of data collected since 1970. *See* Hatch 1993a for data sources.

Changes in food supply, whether natural or related to human activity, are another important influence on seabird populations. The postwar period from 1950 to the 1990's has seen explosive growth and constant change in commercial fisheries of the northeastern Pacific (Alverson 1992). Driving these changes, or in some cases possibly driven by them, are major shifts in the composition of marine fish stocks. In the Gulf of Alaska, for example, a shift occurred in the late 1970's and early 1980's toward greater abundance of groundfish (cod, *Gadus macrocephalus*; various flatfishes; and especially walleye pollock, *Theragra chalcogramma*), possibly at the expense of small forage species such as herring (*Clupea harengus*), sandlance (*Ammodytes hexapterus*), and capelin (*Mallotus villosus*; Alverson 1992) (Fig. 4). Coincident with these changes, diets of a variety of seabirds such as murres, murrelets, and kittiwakes have shifted from being predominantly capelin-based

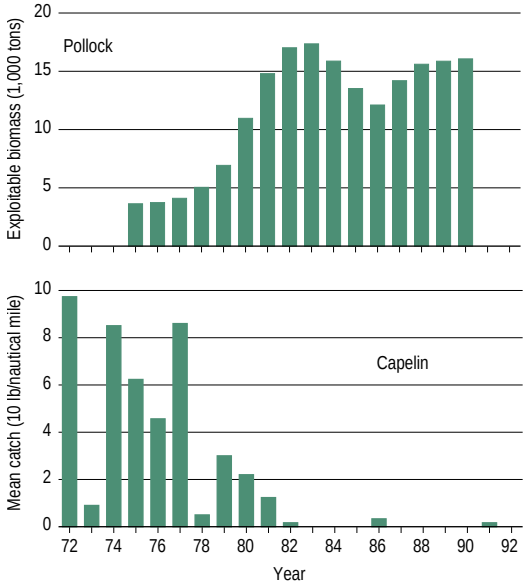

Fig. 4. Temporal changes in marine fish stocks of the Gulf of Alaska: total pollock biomass (age 2+) from stock assessment surveys by the National Marine Fisheries Service, 1975-90 (above; Marasco and Aron 1991), and catch per unit effort of capelin in midwater trawls in Pavlov Bay, western Gulf of Alaska, 1972-92 (below; P. Anderson, NMFS, unpublished data).

to pollock-based (Piatt, unpublished data). Seabird declines and breeding failures correspond to the shift, as do drastic declines in harbor seals (*Phoca vitulina*) and northern sea lions (*Eumetopias jubatus*) in the Gulf of Alaska (Merrick et al. 1987; Pitcher 1990).

The wholesale removal of large quantities of fish biomass from the ocean is likely to have major, if poorly known, effects on the marine ecosystem. An emerging issue is whether fish harvests are altering marine ecosystems to the detriment of seabirds and other consumers like pinnipeds and whales.

The relative role of fishing and natural environmental variation in regulating these systems is another matter for long-term research. In any case, seabird monitoring will continue to provide valuable insights into marine food webs, especially changes that affect the ocean's top-level consumers, including humans.

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Colonial Waterbirds

by
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Colonial waterbirds, that is, seabirds (gulls, terns, cormorants, pelicans) and wading birds (herons, egrets, ibises), have attracted the attention of scientists, conservationists, and the public since the turn of the century when plume hunters nearly drove many species to extinction. The first national wildlife refuge at Pelican Island, Florida, was founded to conserve a large nesting colony of the brown pelican (*Pelecanus occidentalis*). The National Audubon Society also established a game warden system to monitor and protect important waterbird colonies. These efforts helped establish federal laws to protect migratory birds and their nesting habitats in North America.

Although the populations of many species rebounded in the early part of the 20th century, major losses and alteration of coastal wetlands still threaten the long-term sustainability of many colonial waterbirds. A national, coordinated monitoring program is needed to monitor population status and trends in colonial waterbirds (Erwin et al. 1993). The Canadian Wildlife Service has recently established a national seabird monitoring program (D. Nettleship, CWS, personal communication). In addition, better coordination and cooperation for monitoring waterbirds are needed on both their breeding grounds in North America and their wintering grounds in Latin America where wetland loss is also a critical problem (Erwin et al. 1993). This article summarizes the status and trends of selected waterbird species in North America, but excludes Alaska, Hawaii, and the Pacific coast, which are described elsewhere.

Population Surveys

Data on the population status of colonial waterbirds come from many sources. The Breeding Bird Survey (Peterjohn and Sauer 1993) is useful as a visual index for the more widely distributed species that occur along coasts and across the interior of the United States and Canada (e.g., great blue herons [*Ardea herodias*] and herring gulls [*Larus argentatus*]), but it is not effective for many waterbird species that nest in wetlands.

Recently, Christmas Bird Count (CBC) data have been analyzed, providing an index to numbers of wintering birds (J.R. Sauer, National Biological Service, personal communication). For waterbirds, these counts must be used with caution since water conditions can have a major effect on the feeding distribution of waterbirds during the count period in December. Thus, trends in CBC counts may indicate more about trends in wetland conditions than trends in populations of any particular waterbird species.

More precise estimates of species' populations at colony sites have been conducted over the years by state, federal, and private organizations. Although a few states (e.g., Florida, Illinois, Massachusetts, Texas, and Virginia) have conducted annual surveys over a long period for at least some species, there is little consistency among their methods and the frequency of surveys (Erwin et al. 1985). Many data on breeding populations are kept at the state level, but these data seldom predate 1980, precluding assessment of long-term trends in many of these long-lived species.

Even though more than 50 species of colonial waterbirds breed in the United States, Canada, and Mexico, this article focuses on the 22 species for which sufficient data are available to indicate population changes, at least at a regional level.

Courtesy D. Smith

Common tern (*Sterna hirundo*).

Pelecaniformes

Pelicans and their allies (cormorants, anhingas) suffered from DDT use, and their numbers plummeted to the point where the eastern and California brown pelicans became endangered. The eastern subspecies, however, was recently removed from the threatened list because of its rapid numerical and range increases (Table).

The American white pelican *(Pelecanus erythrorhynchos*) has shown similar sharp increases in the western regions of Canada and the United States (Evans and Knopf 1993). Double-

Table. Regional, national, and continental population status and trends of selected colonial waterbirds in the United States[a] as reported by the Breeding Bird Survey, Christmas Bird Counts, and other sources.

Species	Region	Population status: Early period	Population status: Recent period	BBS/CBC trend[b]: % change	BBS/CBC trend[b]: % +/-routes	Years	References[c]
Pelecaniformes							
American white pelican	Continent			+5.3**		1966-91	BBS
	U.S.	17,872 nests (1964)	22,299 nests (1980-81)				1
	Canada	14,103 (1967-69)	53,345 (1985-86)				1
	Mexico	Sporadic (100 nests)					1
	U.S.			+3.8***		1966-89	CBC (winter)
Eastern brown pelican	U.S.	7,800-8,300 (1970-76)	26,461 (1989)	NA			2
Double-crested cormorant	Continent			+6.5***	+0.61***	1966-91	BBS
	U.S.			+2.3**	+0.61***		
	Canada			+11.5***	+0.64*		
	U.S.			+8.2***		1966-89	CBC (winter)
Ciconiiformes							
Great blue heron	Continent			+1.5**	0.60***	1966-91	BBS
	U.S.			+1.9	0.61***		
	Canada			+0.7 ns	0.54*		
Great blue heron	U.S.			+2.2***		1966-89	CBC
Snowy egret	U.S.			+2.0**		1966-89	CBC
Reddish egret	U.S. Gulf coast	1,700-2,200 pr. (1976-78)	1,370-1,900 pr. (1989-90)				3,4,5,6
Black-crowned night-heron	U.S.			+2.8		1966-89	CBC
White ibis	Southeast U.S.	40,000-80,000 pr.(1967-71)	22,000-50,000 pr. (1987-93)				7
	U.S.			+5.0**		1966-89	CBC
White-faced ibis	Western U.S.	4,500-5,500 pr. (1967-75)	13,000-13,500 pr. (1985)				8
	U.S.			+7.6**		1966-89	CBC
Wood stork	Southeast U.S.	2,500-5,200 pr. (1976-82)	6,729 pr. (1993)				9,10
	U.S.			+1.3 ns		1966-89	CBC
Charadriiformes							
Razorbill	N. Gulf St. Lawrence, Can.	16,200 birds (1960)	2,380 birds (1982)				11
Atlantic puffin	Canada (Witless Bay)	300,000-340,000 pr.(1973)	225,000 pr.(1978-80)				11
	U.S.	125 pr. (1977)	135 pr. (1993)				12
Great black-backed gull	U.S.			+3.6**		1966-89	CBC
Herring gull	Atlantic coast U.S.	110,000 pr.(1978-82)	< 100,000 pr. (1988-92)				13
	U.S.			+0.5 ns		1966-89	CBC
Ring-billed gull	Continent			+7.9**	+0.60***	1966-91	BBS
	U.S.			+16.5**	+0.58***		
	Canada			+5.7*	+0.62***		
	U.S.			+4.2***		1966-89	CBC
Franklin's gull	Continent			-6.0 ns		1966-91	BBS
	U.S.			-19.2***			
	Canada			-1.2 ns			
Gull-billed tern	Mid-Atlantic U.S. (VA-SC)	1,100-1,600 pr. (1977)	1,125-1625 pr. (1993)				14,15,16
	Gulf coast U.S. (TX-AL)	1,200-2,100 pr. (1977)	3,000 pr. (1990)				3,4,5
	U.S.			-1.5*		1966-89	CBC
Forster's tern	Continent			-2.4 ns	-0.58*		BBS
	U.S.			-3.2 ns	-0.60**		BBS
	Canada			Insuff. data			
	U.S.			+4.3***			CBC
Common tern	Great Lakes U.S.	1,691 nests (1977)	1,916 nests (1989)				17, 18
Roseate tern	N. Atlantic U.S.	2,855-3,285 pr. (1976-80)	3,200 pr. (1993)				19, 20
	U.S. Caribbean	Uncertain pre-1975	1,900-2,500 pr. (1975-80)				13,18
Least tern (interior ssp.)	Mississippi River	4,100-4,700 birds (1986-87)	6,833 birds (1991)			1986-91	21
Black tern	Continent			-3.9**	-0.59***	1966-92	BBS
	U.S.			-5.6***	-0.64***		
	Canada			-3.4	-0.52 ns		

[a]Excluding Alaska, Hawaii, and the Pacific coast states.
[b]Breeding Bird Survey trends statistically test for an annual (% change) trend (H_0: trend = 0) and % of increasing (+) or decreasing (-) routes (H_0: no. routes + = no. routes -). Probability levels: *$P < 0.10$;** $P < 0.05$; *** $P < 0.01$. A lower P value means there is more confidence that the trend is real. A population trend change at the $P < 0.10$ level is considered statistically significant; ns = not significant. Christmas Bird Count trends are conducted similar to annual BBS trend (J.R. Sauer, NBS, unpublished data).
[c]Sources: numbers refer to literature reference number; BBS = Breeding Bird Survey results (J.R. Sauer and B. Peterjohn, NBS, personal communication); CBC = Christmas Bird Count trend results (J.R. Sauer, personal communication).
1—Evans and Knopf 1993; 2—P. Wilkerson, South Carolina Wildlife and Marine Resources Department; 3—Lange, in press; 4—Portnoy 1978; 5—Martin and Lester 1990; 6—Runde 1991; 7—P. Frederick, University of Florida, unpublished data; 8—D. Manry, unpublished data; 9—Ogden et al. 1987; 10—J. Ogden and M. Coulter, National Park Service, unpublished data; 11—Nettleship and Birkhead 1985; 12—B. Allen, Maine Department of Inland Fisheries and Wildlife, unpublished data; 13—Nisbet, in press; 14—Spendelow and Patton 1988; 15—Erwin 1979; 16—J. Parnell and P. Wilkerson, University of North Carolina and South Carolina Wildlife and Marine Resources Department, unpublished data; 17—Scharf et al. 1992; 18—Blokpoel and Tessier 1993; 19—Gochfeld 1983; 20—J.A. Spendelow, NBS, unpublished data; 21—E. Kirsch and J. Sidle, NBS, unpublished data.

crested cormorant (*Phalacrocorax auritus*) populations also declined during the l940-70 period, probably because of DDT and other pesticides; however, this species has increased dramatically across Canada and the northern United States (Table). In the Great Lakes and elsewhere, this species' increases have attracted considerable attention because of the negative effects on fisheries and on the aquaculture industry (Blokpoel and Scharf 1991; Blokpoel and Tessier 1991; Nettleship and Duffy, in press).

Ciconiiformes

Heron, egret, and ibis nesting colonies were reduced along much of the U.S. coastline in the early 1900's as a result of the millinery trade; however, the species have all recovered their former ranges. Great blue herons are the most abundant and ubiquitous of the wading birds in North America; all indications suggest that their populations have increased, especially in the United States (Butler 1992; Table). One reason for this trend may be that winter survival has increased as herons feed heavily at aquaculture facilities in the southern United States.

The reddish egret (*Egretta rufescens*) is listed as a species of management concern to the USFWS (OMBM 1994). It nests in small numbers along the gulf coast and in southern Florida (Table). Reddish egrets seem to have declined some in Texas (Lange, in press) and Louisiana (Portnoy 1978; Martin and Lester 1990; Figure), but the data are not adequate in Florida to assess trends.

Snowy egrets (*E. thula*) were prized by plume traders at the turn of the century, and the species suffered dramatic population declines; however, by the l970's these egrets had recovered their former range. More recently, their populations declined in some Atlantic regions such as Virginia (Williams et al. 1990) and southern Florida (Robertson and Kushlan 1974; Ogden 1978; Table).

The black-crowned night-heron (*Nycticorax nycticorax*), which occurs across all of North America, may be declining in parts of Canada, south to Texas (Davis 1993) and perhaps Virginia (Williams et al. 1990; Table).

Ibises are more nomadic in their breeding distribution than are other wading birds. White ibis (*Eudocimus albus*) have declined markedly in southern Florida as a result of hydrologic changes in the Everglades (Robertson and Kushlan 1974; Ogden 1978). Their breeding distribution has shifted northward, and large colonies exist in Georgia and the Carolinas (Ogden 1978; Bildstein 1993). Over the entire southeastern United States the species may not have undergone major changes, although state estimates have been erratic (twofold changes in 2-3 years; Table).

The white-faced ibis (*Plegadis chihi*) was formerly (1987) on the USFWS management concern list, but is not on the l994 national list (OMBM 1994). Population data for the central and western populations (noncoastal) indicate a marked increase in the numbers of these ibis from the early l970's to l985 (D. Manry, personal communication; Table).

Wood storks (*Mycteria americana*), which have been federally listed as endangered since l984, nest from Florida north to South Carolina in the United States, in Cuba, and in enormous numbers in the river deltas of eastern Mexico, especially the Usumacinta-Grijalva Delta. Stork colonies have shifted north from the Everglades to central and northern Florida, Georgia, and South Carolina since the l970's (Robertson and Kushlan 1974; Ogden 1978; Ogden et al. 1987). Recent inventories of nesting populations in the United States indicate a modest increase in numbers over the past 10-15 years (Table; Figure).

Because of the mobility of wood storks and ibis, monitoring them requires a regional approach to ensure standardization in survey timing and methods. Individual state inventories are inadequate to address many highly mobile species.

Charadriiformes

This order of colonial-nesting waterbirds includes the alcids (murres, puffins, auks), shorebirds, gulls, terns, and black skimmers (*Rynchops niger*). Although some species of alcids and terns were nearly extirpated by hunters or millinery traders during the early 1900's, they rebounded well in many areas.

Alcid populations are rare in the eastern United States. In maritime Canada, however, alcid numbers are substantial (Nettleship and Birkhead 1985; Erskine 1992), though there is concern over Canada's razorbill (*Alca torda*) populations, which declined by more than 75% from 1960 to 1982 (Nettleship and Birkhead 1985). These declines may be the result of conflicts with commercial fisheries.

Canadian populations of Atlantic puffins (*Fratercula arctica*) have declined a great deal in some areas. The largest Atlantic puffin colony in North America is at Witless Bay, Newfoundland (61% of continental breeding total); this colony has declined by 25%-35% from 1973 to 1980 (Nettleship and Birkhead 1985). Again, competition between birds and commercial fisheries (capelin) may be causing much of the decline. In Maine, a successful transplant program has been in effect for more than a decade to reintroduce nesting Atlantic puffins onto several coastal islands (Kress and Nettleship 1988); numbers remain small, however (Table).

Gull populations have increased substantially from the middle part of the century to the present (Buckley and Buckley 1984; Nisbet, in press). Great black-backed gulls (*Larus marinus*) have increased in some mid-Atlantic states but have probably declined in Maine (Nisbet, in press; Table). Herring gull populations probably peaked around 1980 at about 110,000 pairs along the northeastern U.S. coastline, but populations may have declined during the l980's

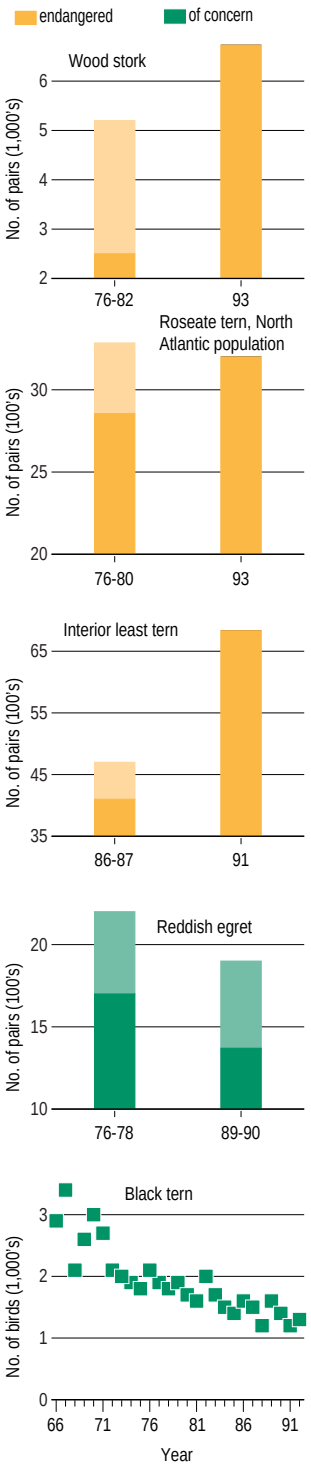

Figure. Trends of selected colonial waterbirds either endangered or on the U.S. Fish and Wildlife Service's list of species of management concern in the lower 48 states (excluding the Pacific coast). Black tern trends are count indices from the Breeding Bird Survey (mean or average number of birds per route). Lighter color shows range of variation in estimates.

(Nisbet, in press); BBS and CBC data do not show any change (Table). Changes in landfill practices that have reduced food supplies along the northeastern coast may have reduced winter survival and slowed the population growth of this species. In the Great Lakes, however, herring gulls have shown a dramatic increase since the late 1970's.

Ring-billed gulls (*L. delawarenis*) continue to increase across the northern tier of states, Canada, and the Great Lakes (Blokpoel and Scharf 1991; Blokpoel and Tessier 1991; Table). The BBS and CBC data suggest significant increases in the United States and Canada (Table). Refuge and resource managers are concerned over the reported decline in the Franklin's gull (*L. pipixcan*), an interior, marsh-nesting species that may be vulnerable to agricultural pesticides (White and Kolbe 1985). The BBS trends indicate that the numbers of this species significantly declined in the United States from l966 to 1991. However, adding 1992 and 1993 data indicates a nonsignificant decline in the United States, which raises the question of the value of BBS data for this flock-feeding species (J.R. Sauer, personal communication).

Gull-billed terns (*Sterna nilotica*) are a species of special concern to many coastal states and were on the former (1987) USFWS management list. Recent population figures from Texas (Lange, in press), Louisiana (Martin and Lester 1990), and the mid-Atlantic region (Virginia to South Carolina) suggest that the population is reasonably stable over the long term but erratic from year to year (Table).

The Forster's tern (*S. forsteri*) nests both along coasts and across the interior of the northern tier of states and Canadian provinces. State surveys do not suggest declines in most states from New Jersey (C.D. Jenkins, New Jersey Division of Fish, Game and Wildlife, personal communication) to Virginia (Erwin 1979). Data are insufficient in the Great Lakes to assess trends. The trends from the BBS and CBC are contradictory, with breeding trends indicating declines and wintering trends a significant increase. This species is erratic in its nesting and probably not sampled well by either of these surveys.

Common terns (*S. hirundo*), while abundant and increasing along the U.S. northeastern coast (Buckley and Buckley 1984), are considered endangered, threatened, or a species of special concern in six Great Lakes states and Ontario (Blokpoel and Scharf 1991; Scharf et al. 1992). Even though tern numbers increased from 1977 to 1989 in the U.S. Great Lakes (Table), the number of their colony sites has declined from 31 to 23. Competition with the ring-billed gull is a major factor in this decline (Scharf et al. 1992).

The roseate tern (*S. dougallii*) is an endangered species (since l987) and breeds in two populations in the western Atlantic. The western North Atlantic population includes the maritime provinces south to Long Island, New York (with a few possibly from New Jersey to Georgia); the U.S. Neotropical population is confined to Puerto Rico, the Virgin Islands, and southern Florida. In the northern population, the number of breeding pairs ranged from 2,855 to 3,285 pairs during the 1976-80 period (Gochfeld 1983) to 3,200 estimated pairs in l993 (J. Spendelow, National Biological Service, personal communication; Table; Figure). In the southern U.S. population, pair estimates from the 1976-79 period range from about 1,900 (Gochfeld 1983) to about 2,600 pairs in the Florida Keys, Puerto Rico, and the Virgin Islands (Blokpoel and Tessier 1993; Table). Earlier records are sparse in this region, making trends difficult to determine.

The least tern (*S. antillarum*) is divided into three subspecies in the United States and Canada; the interior (*S.a. athalassos*) and California (*S.a. browni*) subspecies are listed as endangered. In the Mississippi River drainages, the interior least tern seems to have increased from the l986-87 period to l991 (E. Kirsch and J. Sidle, NBS, unpublished data; Table; Figure). The 1993 floods probably prevented recent nesting in many river stretches.

The black tern (*Chlidonias niger*) is listed as either endangered or a species of concern in many northern states, including New York, Iowa, Illinois, Wisconsin, Ohio, and Indiana. Its population has decreased at the BBS continental and U.S. levels from 1966 to 1992 (Table; Figure). From 1982 to 1991, BBS data indicate a significant increase in Canada with continued decrease in the United States. This suggests a species' displacement to the north, possibly a result of changes in wetland conditions in the northern tier of the United States. A confounding factor may also be that the Canadian surveys have been more intensive for this species in recent years.

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Shorebirds: East of the 105th Meridian

The North American group of shorebirds includes 48 kinds of sandpipers, plovers, and their allies, many of which live for most of the year in coastal marine habitats; others live principally in nonmarine habitats including grasslands, freshwater wetlands, and even second-growth woodlands. Most North American shorebirds are highly migratory, while others are weakly migratory, or even nonmigratory in some parts of their range. Here we discuss shorebirds east of the 105th meridian (roughly east of the Rocky Mountains). Historically, populations of many North American species were dramatically reduced by excessive gunning (Forbush 1912). Most populations recovered after the passage of the Migratory Bird Treaty Act of 1918, although some species never recovered and others have declined again.

High proportions of entire populations of shorebirds migrate by visiting one or a small number of "staging sites," areas where the birds accumulate fat to provide fuel before continuing

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with their long-distance, nonstop flights to the next site (Morrison and Harrington 1979; Senner and Howe 1984; Harrington et al. 1991). Growing evidence (Schneider and Harrington 1981) indicates that staging areas are unusually productive sites with highly predictable but seasonally ephemeral "blooms" of invertebrates, which shorebirds use for fattening. In some cases, especially for "obligate" coastal species, specific sites are traditionally used; even other species sites may shift between years. Because of this, conservationists believe some species are at risk through loss of strategic migration sites (Myers et al. 1987). Other species are threatened by the loss of breeding and wintering habitats (Page et al. 1991; Haig and Plissner 1993; B. Leachman and B. Osmundson, U.S. Fish and Wildlife Service, unpublished data).

The predicted consequences of global warming, such as sea-level change, will also strongly affect the intertidal marine habitats, which many species of shorebirds depend upon. Some of the strongest warming effects will be at high latitudes, including those where many shorebirds migrate to breed, as well as south temperate latitudes, where many of them winter.

Population Trend Data

Information on population trends in North American shorebirds comes largely from studies designed for other purposes, except in the case of a few species that breed within latitudes covered by the Breeding Bird Survey (BBS) and one game species, the American woodcock (*Scolopax minor*). We divide these studies into two types, those based on surveys during breeding and nonbreeding seasons.

Population trend data from breeding seasons come mostly from studies of declining or threatened species such as piping plovers (*Charadrius melodus*; Haig and Plissner 1993), mountain plovers (*C. montanus*; Graul and Webster 1976; F.L. Knopf, U.S. Fish and Wildlife Service, unpublished data), and snowy plovers (*C. alexandrinus*; Page et al. 1991). Additional data come from the BBS and from special survey efforts on game species such as American woodcock (Sauer and Bortner 1991).

Nonbreeding season data come mostly from aerial surveys of migrants on Delaware Bay during spring (Clark et al. 1993), of migrants by the International Shorebird Surveys (ISS) during spring and fall (Harrington et al. 1989), and by the Maritimes Shorebird Surveys (MSS) in eastern Canada during fall (Morrison et al. 1994). Although none of these projects was designed principally to gather data for population trend monitoring, they are the only data bases on migrant species that have been systematically compiled through a period of years. The Christmas Bird Counts are an exception; they are conducted when most shorebirds are south of the United States.

Largely voluntary efforts of the ISS of Manomet Observatory, the MSS of the Canadian Wildlife Service, the BBS of the National Biological Service, and surveys on Delaware Bay (DELBAY) coordinated by New Jersey and Delaware state wildlife agencies have produced rough data useful for trend analysis. Because the BBS is conducted during the breeding season and is based on roadside surveys, its value is greatest in analyzing population change of broadly distributed shorebirds common in temperate latitudes where survey effort is greatest. The ISS, MSS, and DELBAY projects have focused on migration season counts and, therefore, are the best (though not ideal) available resources for monitoring northern-breeding shorebirds, which include most species in North America.

Plovers

Three of the eight species of plover that regularly occur east of the 105th meridian (snowy plover, piping plover, and mountain plover) are species of concern (endangered, threatened, or candidate species); killdeer (*C. vociferus*) and perhaps black-bellied plover (*Pluvialis squatarola*) are in decline (Table). In North America, all of these except the black-bellied plover are distributed principally in temperate latitudes; snowy, piping, and mountain plovers breed in special, localized habitats (principally sandy beaches, salt lakes, and salt flats for snowy and piping plovers, short-grass prairie for mountain plovers). There has been no evaluation of trends for Wilson's plover (*Charadrius wilsonia*), typically a beach-nesting species in southern North America. There are no statistically significant population changes in American golden- (*P. dominica*) and semipalmated plovers (*C. semipalmatus*).

Oystercatchers, Avocets, and Stilts

No significant population changes have been detected in the three species of these groups east of the 105th meridian (Table).

Sandpipers

This is the largest family of shorebirds. Five species of this family listed in the Table—willet (*Catoptrophorus semipalmatus*), upland sandpiper (*Bartramia longicauda*), long-billed curlew (*Numenius americanus*), marbled godwit (*Limosa fedoa*), and American woodcock—commonly breed in the contiguous 48 United States. Two others, the long-billed curlew, which nest principally in short-grass prairie, and the American woodcock found in second-

growth woodland, show significant population declines. Upland sandpipers (tall-grass habitats, including croplands) show a significant increase. The remaining sandpiper species breed principally north of the contiguous 48 states. Six of these—ruddy turnstone (*Arenaria interpres*), red knot (*Calidris canutus*), sanderling (*C. alba*), white-rumped sandpiper (*C. fuscicollis*), Baird's sandpiper (*C. bairdii*), and buff-breasted sandpiper (*Tryngites subruficollis*)—are principally high-latitude breeders; two (red knot and sanderling) of the three species for which trend analysis data are available are in decline (Table). The remaining species can be grouped as taiga or middle Arctic breeders; seven of these have not been evaluated for population trend change; five species—whimbrel (*Numenius phaeopus*), semipalmated sandpiper (*Calidris pusilla*), least sandpiper (*C. minutilla*), short-billed dowitcher (*Limnodromus griseus*), and common snipe (*Gallinago gallinago*)—were in significant decline (Table), and four species—greater and lesser yellowlegs (*Tringa melanoleuca* and *T. flavipes*), spotted sandpiper (*Actitis macularia*), and dunlin (*C. alpina*)—showed no significant change (Table). No species showed significantly increased population trends.

Phalaropes

Only one (Wilson's phalarope; *Phalaropus tricolor*) of the three species of North American phalaropes has been evaluated for population change, and it showed significant declines (Table).

Summary and Recommendations

Population trend evaluation has been conducted for 27 of 41 shorebird species common in the United States east of the 105th meridian. Of the 27 species for which trend data are available, 12 show no change, 1 increased, and 14 decreased (Table). There were no clear correlations with habitat.

It is important that shorebird populations are monitored nationally, yet most species are hard to monitor because they inhabit regions that are difficult to access for much of the year. Migration seasons appear to be the most practical time for monitoring most species. Unfortunately, sampling for population monitoring during nonbreeding seasons presents a group of unresolved analytical challenges. Additional work on existing data can help identify how or whether broad, voluntary, or professional networks can collect data that will better meet requirements for monitoring population change.

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Table. Species, major habitats, and population change in North American breeding shorebirds in the United States east of the 105th meridian.*

Scientific name	Common name	Habitat	Reference and status	Significance
Pluvialis squatarola	Black-bellied plover	Coastal	a- d+	$P < 0.10$(a) ns(d)
P. dominica	American golden-plover	Upland	d-	ns
Charadrius alexandrinus	Snowy plover	Coastal	g threatened	
C. wilsonia	Wilson's plover	Coastal	unknown	
C. semipalmatus	Semipalmated plover	Mixed	a- d+	ns(a) ns(d)
C. melodus	Piping plover	Coastal	c threatened	
C. vociferus	Killdeer	Upland	b-	$P < 0.05$
C. montanus	Mountain plover	Upland	b+	ns
Haematopus palliatus	American oystercatcher	Coastal	unknown	
Himantopus mexicanus	Black-necked stilt	Fresh water	b-	ns
Recurvirostra americana	American avocet	Fresh water	b-	ns
Tringa melanoleuca	Greater yellowlegs	Mixed	a-	ns
T. flavipes	Lesser yellowlegs	Mixed	a+	ns
T. solitaria	Solitary sandpiper	Fresh water	unknown	
Catoptrophorus semipalmatus	Willet	Coastal	a± b+ d-	ns(a) ns(b) ns(d)
Actitis macularia	Spotted sandpiper	Fresh water	b+	ns
Bartramia longicauda	Upland sandpiper	Upland	b+	$P < 0.05$
Numenius phaeopus	Whimbrel	Coastal	a- d+	$P < 0.01$(a) ns(d)
N. americanus	Long-billed curlew	Upland	b-	$P < 0.05$
Limosa haemastica	Hudsonian godwit	Coastal	unknown	
L. fedoa	Marbled godwit	Mixed	b+	ns
Arenaria interpres	Ruddy turnstone	Coastal	a- d+ e-	ns(a) ns(d) ns(e)
Calidris canutus	Red knot	Coastal	a- d- e-	ns(a) $P < 0.10$(d) ns(e)
C. alba	Sanderling	Coastal	a- d- e-	$P < 0.01$(a) ns(d) $P < 0.01$(e)
C. pusilla	Semipalmated sandpiper	Mixed	a- d- e-	ns(a) $P < 0.02$(d) $P < 0.05$(e)
C. mauri	Western sandpiper	Mixed	unknown	
C. minutilla	Least sandpiper	Mixed	a+ d-	ns(a) $P < 0.05$(d)
C. fuscicollis	White-rumped sandpiper	Mixed	unknown	
C. bairdii	Baird's sandpiper	Fresh water	unknown	
C. melanotos	Pectoral sandpiper	Fresh water	unknown	
C. maritima	Purple sandpiper	Coastal	unknown	
C. alpina	Dunlin	Mixed	d- e±	ns(d) ns(e)
C. himantopus	Stilt sandpiper	Fresh water	unknown	
Tryngites subruficollis	Buff-breasted sandpiper	Upland	unknown	
Limnodromus griseus	Short-billed dowitcher	Coastal	a- d- e+	$P < 0.05$(a) $P < 0.08$(d) $P = 0.12$(e)
L. scolopaceus	Long-billed dowitcher	Fresh water	unknown	
Gallinago gallinago	Common snipe	Fresh water	b-	$P < 0.05$
Scolopax minor	American woodcock	Special	b- f-	$P < 0.05$(b) $P < 0.05$(f)
Phalaropus tricolor	Wilson's phalarope	Fresh water	b-	$P < 0.05$
P. lobatus	Red-necked phalarope	Special	unknown	
P. fulicaria	Red phalarope	Special	unknown	

* In the "reference and status" column and the "significance" column, "a" through "g" refer to a reference in footnote **. The reference footnotes also give the years the survey was conducted. If "+" follows the letter in the "reference and status" column, the population is increasing. If "-" follows the letter, the population is declining. In the "significance" column, "ns" means population increase or decrease is not significant. "*P*" is a measure of the confidence that the decline or increase is actually significant. A lower *P* value means there is more confidence that the trend is real. A population trend change at the $P < 0.10$ level is considered statistically significant.

** a — Howe et al. (1989) for 1972-83.
b — B.G. Peterjohn, NBS, unpublished analysis, National Biological Service, Breeding Bird Survey, 1982-91.
c — Haig and Plissner 1993.
d — Morrison et al., in press 1974-91.
e — Clark et al. 1993 for 1986-92.
f — Sauer and Bortner 1991.
g — U. S. Fish and Wildlife Service, Office of Endangered Species, unpublished data.

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Western North American Shorebirds

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Shorebirds are a diverse group that includes oystercatchers, stilts, avocets, plovers, and sandpipers. They are familiar birds of seashores, mudflats, tundra, and other wetlands, but they also occur in deserts, high mountains, forests, and agricultural fields. Widespread loss and alteration of these habitats, especially wetlands and grasslands during the past 150 years, coupled with unregulated shooting at the turn of the century, resulted in population declines and range contractions of several species throughout North America. In the western portion of the continent, efforts to monitor the status and trends of shorebirds have been in effect for only the past 15-25 years and for only a few species. Methods exist to monitor population trends for most shorebirds, but only broadscale, international efforts, relying largely on volunteer help, will accomplish this.

In this article we address shorebirds primarily in western North America, the region west of the Continental Divide from northern Alaska to southern Mexico. The 12 states, a Canadian province and territory, and the western portion of Mexico within this region represent about 25% of the North American landmass (Fig. 1). Western North America includes portions of three broad ecological domains: the Polar Domain, encompassing the tundra and boreal forests that cover most of Greenland, Canada, and Alaska; the Humid Temperate Domain, including the humid midlatitude forests and shrublands within the United States, southern portions of the Canadian prairie provinces, and along the west coast of North America; and the Dry Domain, encompassing the short-grass prairies, sagebrush provinces, and deserts (Fig. 1; Bailey 1978, 1989).

Sources of Data

We derived seasonal distribution of shorebirds within these ecological domains from numerous sources, mostly range maps in field guides, books, and our familiarity with the birds within the region (AOU 1983; Robbins et al. 1983; Hayman et al. 1986; Godfrey 1987; National Geographic Society 1987; Paulson 1993).

No continent-wide protocol exists for monitoring the status and trends of North American shorebirds. Current information has largely been acquired through independent programs sponsored by a combination of federal, state, and private conservation agencies. Efforts have mostly been regional, including broadscale monitoring directed primarily at birds during the nonbreeding season (Howe et al. 1989; Gill and Handel 1990; Page et al. 1992; Skagen and Knopf 1993; Morrison et al. 1994) or have focused on individual species (Handel and Dau 1988; Gill et al. 1991; Page et al. 1991; Haig 1992; Handel and Gill 1992a; Knopf 1994; F.L. Knopf, USFWS, unpublished report). We have relied primarily on this information and that of our ongoing studies to summarize the status and trends of shorebirds in western North America.

Shorebirds of the Region

Breeding

Among the 51 species that regularly breed in North America, 47 (92%) do so within western North America (Table). Within this region, the Polar Domain supports the greatest number of breeding species (37), including 5 that breed nowhere else on the continent. The Humid

Temperate Domain provides breeding areas for 20 species while only 12 breed in the Dry Domain (Fig. 1). The number of species breeding within the domains in the West generally exceeds those breeding east of the Continental Divide, even though the eastern area is much larger.

Western North American shorebirds nest in a variety of habitats, although most species (53%) are restricted to either coastal or interior wetlands (Page and Gill 1994). About a third of the species nest primarily on uplands, especially Arctic and subarctic tundra and dry temperate grasslands.

Wintering

Thirty-six (70%) of the continent's breeding species winter in western North America, including seven that are restricted to the region (Table). The continental distribution of species shifts southward in winter, but numbers are still higher in the West than in the East (Fig. 1). Only 4 of the 37 species breeding in the Polar Domain of western North America remain there during winter. About 30 species spend the winter in the Humid Temperate and Dry domains. Populations of 12 (25%) of western North America's breeding species spend the winter entirely on other continents or throughout Oceania (*see* glossary; Table).

Most shorebirds use a much broader range of habitats during winter than during the breeding period. All species use one or more coastal habitats in winter and two-thirds of the species also use interior habitats (Page and Gill 1994). Wetlands, the single-most important habitat both along the coast and in the interior of western North America, are used by about 80% of all species. Sandy and rocky shorelines along the Pacific coast are also important habitats and are used by about a quarter of the species (Page and Gill 1994).

Migrating

All species of North American shorebirds are migratory to some degree, with the possible exception of both species of oystercatchers and Wilson's plover; they are not migratory in the true sense but do make short, local movements. Shorebirds migrate in spring and fall over three broadly defined corridors encompassing the western, central, and eastern portions of the continent to wintering areas in North, Central, and South America (Morrison and Myers 1989). Other migratory corridors funnel Arctic breeders from western North America across the Pacific Ocean to wintering areas in Asia, Australasia, and Oceania (*see* glossary; Gill and Handel 1981; Handel and Gill 1992b; Page and Gill 1994). The distances traveled between breeding and wintering grounds vary greatly within and among species, often exceeding 8,000 km (5,000 mi) for such species as Hudsonian and bar-tailed godwits.

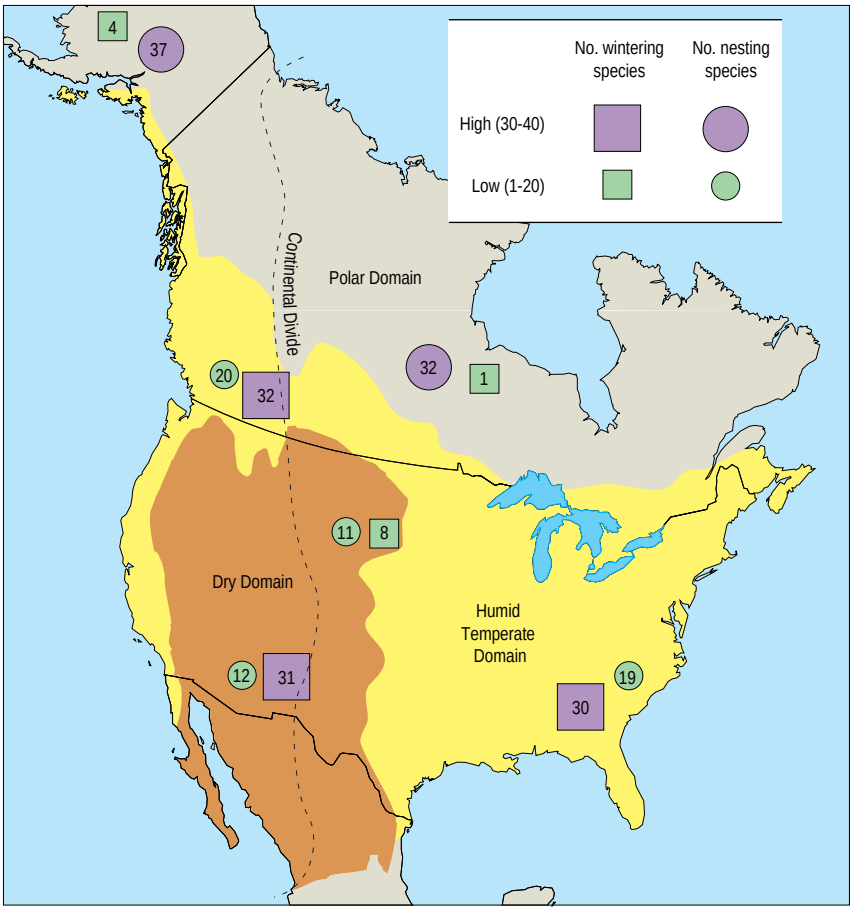

Fig. 1. Number of shorebird species regularly breeding and wintering within three broad ecological domains of North America west and east of the Continental Divide (dashed line).

Wetlands are the most important habitat used by shorebirds during spring and fall migrations. Throughout western North America about 140 discrete wetlands and several additional wetland complexes (e.g., Central Valley of California) have been identified as being important to shorebirds during these periods (Fig. 2). Most staging areas (85%) host populations of 1,000-10,000 birds, but 18 sites support 100,000-1 million shorebirds during the peak of migration (Fig. 2). Because shorebirds use different migration pathways and strategies during spring and fall, the locations of critical staging areas shift between the two seasons (Fig. 2).

Status and Trends

Size of Populations

Population estimates exist for only about a quarter of the species that breed or winter in western North America (Table), and even these few vary widely in terms of statistical rigor and precision. These estimates range between

Table. Seasonal occurrence and status and trends of populations of shorebirds in North America west and east of the Continental Divide.

Species	Occurrence		Population*	
	Breeding	Wintering	Size	Trend
Black-bellied plover (*Pluvialis squatarola*)	Both regions	Both regions	Unknown	Unknown
Pacific golden-plover (*P. fulva*)	Western	Western	Unknown	Unknown
American golden-plover (*P. dominica*)	Both regions	Absent	Unknown	Unknown
Snowy plover (*Charadrius alexandrinus*)	Both regions	Both regions	18,500 b	Population decline and range contraction past century; significant decline in western region past 25 yr (Page et al. 1991; Page and Gill 1994)
Wilson's plover (*C. wilsonia*)	Both regions	Both regions	Unknown	Unknown
Common ringed plover (*C. hiaticula*)	Eastern	Absent	Unknown	Unknown
Semipalmated plover (*C. semipalmatus*)	Both regions	Both regions	Unknown	Unknown
Piping plover (*C. melodus*)	Eastern	Mostly eastern	4,700 b	Declining (Haig 1992)
Killdeer (*C. vociferus*)	Both regions	Both regions	Unknown	Decline of 2.0% per yr in western region past 25 yr; 5.3% decline per yr last 10 yr (J. Sauer and S. Droege, NBS, unpublished data)
Mountain plover (*C. montanus*)	Both regions	Mostly western	5,000-15,000 b	Population decline and range contraction past century; continental decline 3.6% per yr past 25 yr (Knopf 1994; F.L. Knopf, USFWS, unpublished report)
American oystercatcher (*Haematopus palliatus*)	Both regions	Both regions	Unknown	Unknown
Black oystercatcher (*H. bachmani*)	Western	Western	7,600 b	Unknown (Page and Gill 1994)
Black-necked stilt (*Himantopus mexicanus*)	Both regions	Both regions	25,000 w	Population decline and range contraction past century; no significant change in population size past 25 yr (Page and Gill 1994; J. Sauer and S. Droege, NBS, unpublished data)
American avocet (*Recurvirostra americana*)	Both regions	Both regions	100,000+ w	Population decline past century; decline of 3.6% per yr in western region past 10 yr (Page and Gill 1994; G. Page, unpublished data; J. Sauer and S. Droege, NBS, unpublished data)
Greater yellowlegs (*Tringa melanoleuca*)	Both regions	Both regions	Unknown	Unknown
Lesser yellowlegs (*T. flavipes*)	Both regions	Both regions	Unknown	Unknown
Solitary sandpiper (*T. solitaria*)	Both regions	Both regions	Unknown	Unknown
Willet (*Catoptrophorus semipalmatus*)	Both regions	Both regions	Unknown	Population decline and range contraction past century; population increase of 2.8% per yr in United States past 25 yr; 0.5% increase in West in past 10 yr (Page and Gill 1994; J. Sauer and S. Droege, NBS, unpublished data)
Wandering tattler (*Heteroscelus incanus*)	Western	Western	Unknown	Unknown
Spotted sandpiper (*Actitis macularia)*	Both regions	Both regions	Unknown	Population stable over continent and western region past 25 yr (J. Sauer and S. Droege, NBS, unpublished data)
Upland sandpiper (*Bartramia longicauda*)	Both regions	Absent	Unknown	Population decline and range contraction past century; 3.6% annual increase on continental basis past 25 yr; no significant trend in western region (Page and Gill 1994; J. Sauer and S. Droege, NBS, unpublished data)
Eskimo curlew (*Numenius borealis*)	Both regions	Absent	25-50 b	Almost extirpated over past century; may be extinct (Gollop et al. 1986; Alexander et al. 1991)
Whimbrel (*N. phaeopus*)	Both regions	Both regions	Unknown	Unknown
Bristle-thighed curlew (*N. tahitiensis*)	Western	Absent	7,000 b	Unknown (Gill and Redmond 1992; C. Handel, unpublished data)
Long-billed curlew (*N. americanus*)	Both regions	Both regions	Unknown	Population decline and range contraction over past century; annual decrease of 3.0% on continental basis past 10 yr (J. Sauer and S. Droege, NBS, unpublished data)
Hudsonian godwit (*Limosa haemastica*)	Both regions	Absent	Unknown	Unknown
Bar-tailed godwit (*L. lapponica*)	Western	Absent	25,000-40,000 b	Unknown (Page and Gill 1994; R. Gill, unpublished data)
Marbled godwit (*L. fedoa*)	Both regions	Both regions	100,000+ w	Population decline and range contraction over past century; no significant trend throughout continent or western region past 25 yr (Page and Gill 1994; G. Page, unpublished data; J. Sauer and S. Droege, NBS, unpublished data)
Ruddy turnstone (*Arenaria interpres*)	Both regions	Both regions	Unknown	Unknown
Black turnstone (*A. melanocephala*)	Western	Western	61,000-99,000 b	Unknown (Handel and Gill 1992a)
Surfbird (*Aphriza virgata*)	Western	Western	50,000-70,000 s	Unknown (Page and Gill 1994)
Red knot (*Calidris canutus*)	Both regions	Both regions	Unknown	Unknown
Sanderling (*C. alba*)	Both regions	Both regions	Unknown	Unknown
Semipalmated sandpiper (*C. pusilla*)	Both regions	Absent	Unknown	Unknown
Western sandpiper (*C. mauri*)	Western	Both regions	Unknown	Unknown
Least sandpiper (*C. minutilla*)	Both regions	Both regions	Unknown	Unknown
White-rumped sandpiper (*C. fuscicollis*)	Both regions	Absent	Unknown	Unknown
Baird's sandpiper (*C. bairdii*)	Both regions	Absent	Unknown	Unknown
Pectoral sandpiper (*C. melanotos*)	Both regions	Absent	Unknown	Unknown
Purple sandpiper (*C. maritima*)	Eastern	Eastern	Unknown	Unknown
Rock sandpiper (*C. ptilocnemis*)	Western	Western	Unknown	Unknown
Dunlin (*C. alpina*)	Both regions	Both regions	450,000-600,000 w	Unknown (Page and Gill 1994)
Stilt sandpiper (*C. himantopus*)	Both regions	Both regions	Unknown	Unknown
Buff-breasted sandpiper (*Tryngites subruficollis*)	Both regions	Absent	Unknown	Unknown
Short-billed dowitcher (*Limnodromus griseus*)	Both regions	Both regions	Unknown	Unknown
Long-billed dowitcher (*L. scolopaceus*)	Both regions	Both regions	Unknown	Unknown
Common snipe (*Gallinago gallinago*)	Both regions	Both regions	Unknown	Population stable past 25 yr but 3.9% decline per yr on continental basis during past 10 yr; decline in western region not significant past 10 yr (J. Sauer and S. Droege, NBS,unpublished data)
American woodcock (*Scolopax minor*)	Eastern	Eastern	Unknown	Unknown
Wilson's phalarope (*Phalaropus tricolor*)	Both regions	Both regions	1,500,000 f	Population decline past century; expansion of range past 50 yr; annual population declines of 7.5% throughout United States and 8.1% in central region during past 10 yr (Jehl 1988; Page and Gill 1994; J. Sauer and S. Droege, NBS, unpublished data)
Red-necked phalarope (*P. lobatus*)	Both regions	Absent	Unknown	Unknown
Red phalarope (*P. fulicaria*)	Both regions	Western	Unknown	Unknown

*Population estimates for Western North America unless otherwise stated (*see* Fig. 1). Sources for estimates of population size given with trend information. Population size — estimated number of individual birds for b — breeding season, f — fall, w — winter, and s — spring. Geographic regions under population trend are defined in Robbins et al. (1986).

10,000 and 100,000 individuals for most populations, but number from as few as 25 birds for the endangered Eskimo curlew to about 500,000 for the Pacific race of the dunlin (*Calidris alpina pacifica*). A few other species for which some data are available, such as western sandpiper and Wilson's phalarope, have populations that exceed a million (Page and Gill 1994).

Population Trends

For most species, reliable quantitative data on population trends are either not available or too recent to assess trends. Assessment of long-term population trends is based largely on historical accounts of relative abundance and distribution and knowledge of habitat alteration within breeding and wintering ranges. Nonetheless, populations of several species of western North American shorebirds have declined significantly over the past 150 years (Page and Gill 1994). One Arctic breeder, the Eskimo curlew, is on the verge of extinction (Gollop et al. 1986; Alexander et al. 1991). Conversion of native grasslands for agriculture, loss of wetlands, and market hunting before the turn of the century have been attributed as factors primarily responsible for these declines. No species is known to have increased in overall population size over this period.

Information on more recent population trends comes primarily from the North American Breeding Bird Survey (BBS), a system of roadside surveys designed primarily to monitor populations of breeding landbirds. The BBS does not sample most western shorebird breeding populations very well because of its sporadic coverage, poor sampling of wetland habitats, and lack of coverage of the most important shorebird breeding grounds in the Arctic, which are roadless. Despite these limitations, BBS does provide valuable trend information, particularly for grassland species in the temperate zone. Additional information on population trends can also be obtained from surveys that target species of concern, such as the snowy plover (Page et al. 1991), or particular habitats of concern, such as the Arctic Coastal Plain of Alaska (D. Troy Ecological Assoc. and British Petroleum Exploration, unpublished report; Andres 1994).

Recent survey data show a mixture of declining, increasing, and apparently stable population trends (Table). Over the past 25 years, western populations of willet and upland sandpiper appear to have been rebounding (J. M. Sauer and S. Droege, unpublished data). Numbers of several other species, such as the black-necked stilt, marbled godwit, and spotted sandpiper, appear to have stabilized (J.M. Sauer and S. Droege, unpublished data). Western populations of several other species, however, have significantly declined over the past 25 years, including the snowy plover, killdeer, mountain plover, American avocet, long-billed curlew, common snipe, and Wilson's phalarope (Table). Such relatively short-term trends among wetland species are difficult to interpret, however, as they may reflect changes in distribution in response to drought conditions rather than absolute declines in population size (Page and Gill 1994).

Most changes in populations appear linked to habitat alteration. For example, since 1970 the snowy plover, heavily dependent on coastal habitats, has disappeared as a breeding species from over 60% of its historic California nesting sites (Page and Stenzel 1981). Introducing plants to stabilize sand dunes, increasing recreational use of beaches, and heavy nest predation by feral foxes threaten to reduce coastal populations further (Page and Gill 1994). Fluctuating water levels in interior wetlands result in unpredictable changes in availability of nesting habitat away from the coast (Page et al. 1991). The breeding range of the mountain plover has contracted markedly in several western states and the continental population has declined significantly during the past 25 years, probably because of habitat degradation on wintering grounds in central and southern California (Knopf 1994; F.L. Knopf, NBS, unpublished

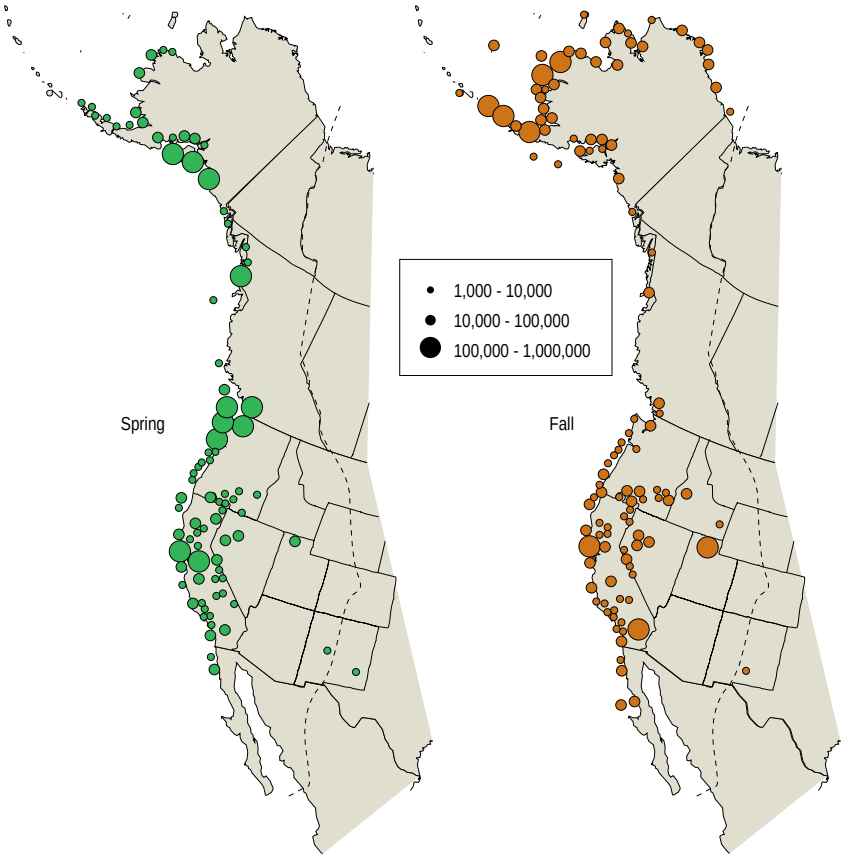

Fig. 2. Location of important staging areas in western North America used by shorebirds during spring and fall migration. Size of dot indicates the estimated peak number of shorebirds at each site.

report). Given the substantial loss of wetlands throughout all western states except Alaska (median loss of 37%; Page and Gill 1994) and a similar loss of native grasslands (Knopf 1994), it is likely that other species of temperate-breeding shorebirds for which we have no trend data have also suffered population declines.

Shorebirds breeding throughout the remote and sparsely populated Polar Domain have been least affected by loss of breeding habitats. Most of these species, however, are dependent on wetlands and other greatly altered habitats outside this region during winter and migration. Information from long-term studies in Europe suggests that populations of Arctic-breeding shorebirds can be affected by conditions on the wintering grounds as well as by those on the breeding grounds (Goss-Custard and Moser 1988; Moser 1988). Arctic breeders such as the buff-breasted sandpiper, upland sandpiper, and American golden-plover winter primarily in grassland habitats of the pampas in South America. These habitats have been virtually eliminated by agricultural development (Bucher and Nores 1988; Blanco et al. 1993). The bristle-thighed curlew, unique among shorebirds because of its flightlessness during molt (Marks 1993), is threatened by problems associated with increasing human populations on wintering grounds in Oceania, including the introduction of mammalian predators (Marks et al. 1990; Gill and Redmond 1992).

Courtesy R.E. Gill, Jr., NBS

Surfbirds (*Aphiza virgata*) and black turnstones (*Arenaria melanocephala*).

In long-term studies of shorebirds nesting at Prudhoe Bay on Alaska's North Slope between 1981 and 1992, considerable annual variation in nesting density and nest success has been found in several species of shorebirds (D. Troy, Troy Ecological Associates and British Petroleum Exploration, unpublished report). Much of this variation has been attributed to predation and environmental factors such as snow cover and temperature at the start of the breeding season. However, among eight species of intensively monitored shorebirds, only dunlin (*Calidris alpina articola*) have exhibited a general, but not significant, downward trend in nesting density over this 10-year period.

Detecting Future Trends

To conserve the tremendous biodiversity of our shorebird resources in western North America, we suggest a two-tiered monitoring program that addresses trends in both habitat availability and shorebird population size. In this program we should:

- Identify and map the current geographic extent and quality of breeding, staging, and wintering habitats important to shorebirds, particularly those species with relatively small populations or restricted habitat requirements;
- Monitor the extent and quality of these habitats, evaluating them at periodic intervals;
- Develop cooperative, international programs to monitor trends in shorebird populations;
- Monitor a representative sample of shorebird populations and evaluate trends in comparison with changes in critical habitats; and
- Establish cooperative, international agreements to protect critical breeding, staging, and wintering habitats, with priority given to those species with low numbers, specific habitat requirements, and immediate threats.

Recently developed technology and continental habitat mapping now provide the tools to identify and map the current extent of wetlands and other habitats important to shorebirds of western North America. By coupling this with current information on shorebird distribution and habitat requirements, we will be able to identify areas critical for shorebirds. The same technology can be used to monitor changes in these habitats over time.

Several existing programs can be adapted or modified to provide reliable information on trends in size of several shorebird populations. Each species needs to be evaluated individually to determine where it could be monitored most cost-effectively—breeding grounds, staging areas, or wintering grounds. Programs such as the International Shorebird Survey, Breeding Bird Survey, and Christmas Bird Count can be used to coordinate efforts of large numbers of volunteers to simultaneously collect information on several species of shorebirds. For many other species like the snowy plover, buff-breasted sandpiper, and bristle-thighed curlew—of particular concern or difficult to monitor with these programs—specific surveys need to be designed and repeated periodically to effectively monitor population trends.

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Raptors

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Raptors, or birds of prey, which include the hawks, falcons, eagles, vultures, and owls, occur throughout North American ecosystems. As predators, most of them kill other vertebrates for their food. Compared to most other animal groups, birds of prey naturally exist at relatively low population levels and are widely dispersed within their habitats. The natural scarcity of raptors, combined with their ability to move quickly, the secretive behavior of many species, and the difficulties of detecting them in rugged terrain or vegetation, all make determining their population status difficult.

As top predators, raptors are key species for our understanding and conservation of ecosystems. Changes in raptor status can reflect changes in the availability of their prey species, including population declines of mammals,

birds, reptiles, amphibians, and insects. Changes in raptor status also can be indicators of more subtle detrimental environmental changes such as chemical contamination and the occurrence of toxic levels of heavy metals (e.g., mercury, lead). Consequently, determining and monitoring the population status of raptors are necessary steps in the wise management of our natural resources.

Methods

We did not compile summary statistics or analyze data for any species; rather, we only have summarized the interpretations and analyses of others. Our summary of raptor status draws largely on the biological literature and on state and federal government reports. Much of this information is summarized in Johnsgard (1988), Palmer (1988), and White (1994) and in proceedings sponsored by the National Wildlife Federation (NWF 1988, 1989a, 1989b, 1990, 1991). Other information is from unpublished data (S.W. Hoffman, HawkWatch International; J.C. Bednarz, Arkansas State University; and W.R. DeRagon, U.S. Army Corps of Engineers).

Interpretations and analyses to determine raptor status and trends can be characterized in four general types: impressions of biologists and of other serious observers of wildlife; impressions or nonstatistical analyses of organized searches or of tallies of birds seen (e.g., Christmas Bird Counts); statistical analyses of intensive quantitative status surveys; and statistical analyses of standardized counts, incorporating estimates of the survey effort (e.g., number of persons, time expended, area covered).

Our conclusion about the status of each species (Table) is usually applied on a nationwide scale, but often must be qualified because of local or regional concerns. These reflect habitat modification or contamination for which we did not have information on a broader scale. We used statistical results when available, but usually our conclusions are based on impressions or qualitative analyses because only that is available on a scale across the species' range, or the United States.

Northern goshawk (*Accipiter gentilis*) in adult plumage, is an example of a raptor species for which there is concern about status.

Courtesy M.R. Fuller

Selected Species

Ospreys

Nesting ospreys (*Pandion haliaetus*) are concentrated along the Atlantic coast, Great Lakes, the northern Rocky Mountains, and in the Pacific Northwest. Most regional populations declined through the early 1970's, but the magnitude of decline varied, with the North Atlantic coast and Great Lakes being most severe. After the 1972 nationwide ban of the insecticide DDT, raptor productivity improved and population numbers increased in most areas. Ospreys also benefited from reservoir construction, especially in the West. Osprey numbers generally are stable, but in some areas they are still increasing. The large stick nests of ospreys, like those of bald eagles (*Haliaeetus leucocephalus*), are relatively conspicuous, thus aiding counts of occupied nests, which are used as a measure of population size. Counts from most states in the early 1980's provided an estimate of about 8,000 nesting pairs. Also, because several osprey populations were studied for many years, a general knowledge of their population dynamics permits a greater understanding of this species' status.

Snail Kite

The endangered snail kite (*Rostrhamus sociabilis*) breeds in central and southern Florida, the northern extent of the species' range, where it is associated with wetlands that are affected by management of water levels. From 1900 to 1960 the population declined; however, it then increased, and now remains stable with fluctuations from 300 to 800 birds (R.E. Bennetts, University of Florida, personal communication).

Bald Eagles

Many local bald eagle populations showed sharp declines (25% to 100%) from 1950 to the 1970's. Populations were adversely affected by shooting, habitat destruction, and organochlorine pesticides (primarily DDT). The bird was protected by the Bald Eagle Protection Act of 1940. In 1978 it was reclassified as endangered in 43 states and threatened in 5. With the documented effects of DDT on reproduction, early studies emphasized locating breeding pairs and monitoring reproductive success.

After the nationwide ban of DDT in 1972, bald eagle reproduction improved and populations began increasing. In 1981 about 1,300 pairs nested in the United States outside Alaska. The active protection of nesting habitat and release of hand-reared eagles aided the population increase. In 1993 at least 4,016 pairs of bald eagles nested in the contiguous United States, with an estimated additional 20,000-25,000 pairs in Alaska. Bald eagles nesting along the shorelines of Lakes Superior, Michigan, Huron, and Erie have lower reproductive rates and relatively high concentrations of the toxic DDE and PCB compounds (Bowerman 1993). Bald eagles nesting in Maine also have low reproductive success, probably because of environmental contaminants.

Habitat loss remains a threat in many areas. Historically there was a continuous (though scattered) distribution of bald eagles in the Southwest, south into Sonora and Baja California, Mexico, where now only a remnant population exists. Because population increases were not uniform throughout the range, the U.S. Fish and Wildlife Service has proposed downlisting this species from endangered to threatened in certain geographic areas.

Hawks

Populations of sharp-shinned hawks (*Accipiter striatus*) in the Midwest might be increasing, but analyses of eastern hawk migration count stations reveal a drop in numbers of juveniles, and blood samples collected from sharp-shinned hawks in the Northeast contained high DDE pesticide concentrations. Many other factors could be involved in a population decline, however. The sharp-shinned hawk provides an example of how monitoring can warn researchers of a potential, long-term decline in a regional population.

Similarly, the northern goshawk (*A. gentilis*) counts of eastern migrants suggest a stable population, but analyses of counts from the West reveal a decline. There is no widespread standardized design for surveying goshawks during the breeding season.

Habitat loss has reduced the number of Harris' hawks (*Parabuteo unicinctus*), whose northern range extent is the southwestern United States. Searches reveal that Harris' hawks have been extirpated from some areas such as the Colorado River Valley, California and Arizona, and that clearing of brush for agriculture likely has led to more than 50% reduction in Texas in the winter.

The biological status of the ferruginous hawk (*Buteo regalis*) remains uncertain because it is stable in some areas (e.g., Great Plains), but declining in other areas (e.g., half the western

Table. Status and trends of raptors in the United States.

Species	Status/trend	Comment*
Black vulture (*Coragyps atratus*)	Stable	Population estimation difficult because of flocking and wide-ranging behavior, secretive nesting
Turkey vulture (*Cathartes aura*)	Stable	
California condor (*Gymnogyps californianus*)	Endangered; extirpated from wild, 1987	Captive propagation and release underway
Osprey (*Pandion haliaetus*)	Increasing	Good information
Hook-billed kite (*Chondrohierax uncinatus*)	Unknown	Extreme northern range limit
American swallow-tailed kite (*Elanoides forficatus*)	Stable	Greatly reduced from historical range
White-tailed kite (*Elanus caeruleus*)	Increasing	Recent range expansion
Snail kite (*Rostrhamus sociabilis*)	Endangered, stable	Northern range limit
Mississippi kite (*Ictinia mississippiensis*)	Increasing	Range expansion
Bald eagle (*Haliaeetus leucocephalus*)	Threatened or endangered in contiguous U.S.; increasing	Status reassessment underway
Northern harrier (*Circus cyaneus*)	Stable	Nomadic, no standard survey; local concern
Sharp-shinned hawk (*Accipiter striatus*)	Stable	Regional differences
Cooper's hawk (*A. cooperii*)	Stable	
Northern goshawk (*A. gentilis*)	Unknown	C2; petition to list *A.g. laingi*; threatened
Common black hawk (*Buteogallus anthracinus*)	Stable	Limited distribution
Harris' hawk (*Parabuteo unicinctus*)	Stable	Fragmented distribution, northern range limit
Gray hawk (*Buteo nitidus or Austurina plagiata*)	Stable	C2; limited distribution, northern range limit
Hawaiian hawk (*B. solitarius*)	Endangered	Difficult to survey, limited distribution
Red-shouldered hawk (*B. lineatus*)	Stable	Local concern
Broad-winged hawk (*B. platypterus*)	Stable	Migration count decline in 1980's
Puerto Rican broad-winged hawk (*B.p. brunnescens*)	Unknown	C2; limited distribution
Short-tailed hawk (*B. brachyurus*)	Stable	Northern range limit; about <500 birds in U.S.
Swainson's hawk (*B. swainsoni*)	Unknown	C3; local concern
White-tailed hawk (*B. albicaudatus*)	Stable	Northern range limit, about 200-400 birds in U.S.
Zone-tailed hawk (*B. albonotatus*)	Stable	Northern range limit, about 100 pairs in U.S.
Red-tailed hawk (*B. jamaicensis*)	Stable	Local increases; Breeding Bird Survey data
Ferruginous hawk (*B. regalis*)	Unknown	C2
Rough-legged hawk (*B. lagopus*)	Stable	
Golden eagle (*Aquila chrysaetos*)	Stable	
Crested caracara (*Caracara plancus*)	Unknown	Northern range limit
American kestrel (*Falco sparverius*)	Stable	Breeding Bird Survey data
American kestrel, Florida (*F.s. paulus*)	Declining	C2
Merlins (*F. columbarius*)	Stable	
Aplomado falcon (*F. femoralis septentrionalis*)	Endangered	Northern range limit; captive propagation and release underway
American peregrine falcon (*F. peregrinus anatum*)	Endangered; increasing	
Arctic peregrine falcon (*F.p. tundrius*)	Threatened	Proposed to delist
Gyrfalcon (*F. rusticolus*)	Stable	
Prairie falcon (*F. mexicanus*)	Stable	
Barn owls (*Tyto alba*)	Stable	Local concern
Flammulated owl (*Otus flammeolus*)	Unknown	Recent surveys reveal more birds, larger range
Virgin Islands screech-owl (*O. nudipes newtoni*)	Unknown	C2; limited distribution
Eastern screech-owl (*O. asio*)	Stable	
Western screech-owl (*O. kennicottii*)	Stable	
Whiskered screech-owl (*O. trichopsis*)	Unknown	Northern range limit
Great horned owl (*Bubo virginianus*)	Stable	
Snowy owl (*Nyctea scandiaca*)	Stable	U.S. breeding, AK only
Northern hawk owl (*Surnia ulula*)	Unknown	U.S. breeding, AK, northern Minnesota
Northern pygmy owl (*Glaucidium gnoma*)	Unknown	Current survey efforts
Ferruginous pygmy owl (*G. brasilianum cactorum)*	Unknown	C2; northern range limit
Elf owl (*Micrathene whitneyi*)	Unknown	Current survey efforts
Burrowing owl (*Athene cunicularia*)	Declining	Local concern
Northern spotted owl (*Strix o. caurina*)	Threatened	Current survey efforts
Mexican spotted owl (*S.o. lucida*)	Threatened	Current survey efforts
California spotted owl (*S.o. occidentalis*)	Unknown	C2; current survey efforts
Barred owl (*S. varia*)	Stable	Western range expansion
Great gray owl (*S. nebulosa*)	Stable	
Long-eared owl (*Asio otus*)	Stable	Local concern
Short-eared owl (*A. flammeus*)	Stable	Local concern
Boreal owl (*Aegolius funereus*)	Stable	Population estimation difficult
Northern saw-whet owl (*A. acadicus*)	Stable	Concern in southeast AK

*Category 2 (C2)—Proposal to list is possibly appropriate but available data are not conclusive for threatened or endangered status.
Category 3 (C3)—Proven more abundant or widespread than previously believed or not subject to identifiable threat.

Causes of Eagle Deaths

by
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The U.S. Department of the Interior has investigated the deaths of more than 4,300 bald and golden eagles *(Haliaeetus leucocephalus* and *Aquila chrysaetos*) since the early 1960's as part of an ongoing effort to monitor causes of wildlife mortality. The availability of dead eagles for study depends on finding carcasses in fair to good condition and transporting them to the laboratory. Such opportunistic collection and the fact that recent technological advances have enhanced our diagnostic capabilities, particularly for certain toxins, mean that results reported here do not necessarily reflect actual proportional causes of death for all eagles in the United States throughout the 30-year period. This type of sampling does, however, identify major or frequent causes of death.

Most diagnosed deaths of eagles in our study resulted from accidental trauma, gunshot, electrocution, and poisoning (Fig. 1). Accidental trauma, such as impacts with vehicles, power lines, or other structures, was the most frequent cause of death in both eagle species (23% of bald and 27% of golden). Gunshot killed about 15% of each species. Electrocution was twice as frequent in golden (25%) than in bald eagles (12%), probably because of the preference of golden eagles for prairie habitats and their use of utility poles as perches.

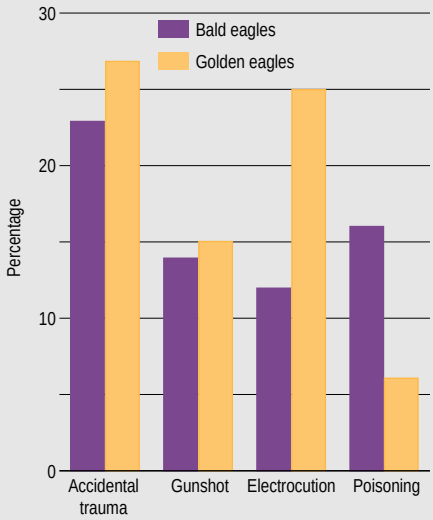

Fig. 1. Causes of mortality of bald and golden eagles over the past 30 years.

Lead poisoning was diagnosed in 338 eagles from 34 states (Fig. 2). Eagles become poisoned by lead after consuming lead shot and, occasionally, bullet fragments present in food items. Agricultural pesticides accounted for most remaining poisonings; organophosphorus and carbamate compounds killed 139 eagles in 25 states (Fig. 3). Eagles are exposed to these chemicals in a variety of ways, often by consuming other animals that died of direct poisoning or from baits placed to deliberately kill wildlife.

Overall, poisonings were more frequent in bald eagles (16%) than golden eagles (6%). The reasons for this are unclear, but may be related to factors that influence submission of carcasses for examination or differences in species' preferences for agricultural, rangeland, and wetland habitats.

Courtesy N. Runningen

Necropsy examination of a bald eagle at the National Wildlife Health Center, Madison, Wisconsin.

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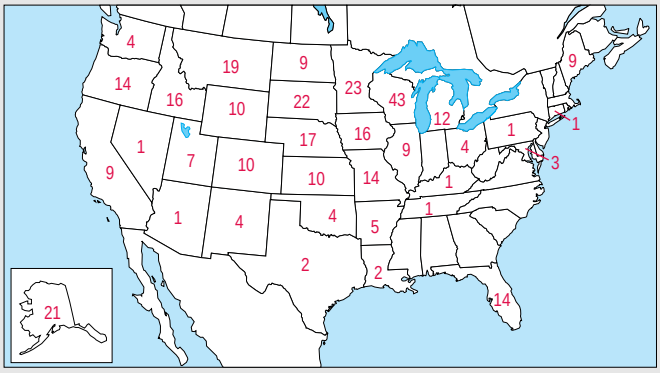

Fig. 2. Nationwide distribution of lead-poisoned eagles.

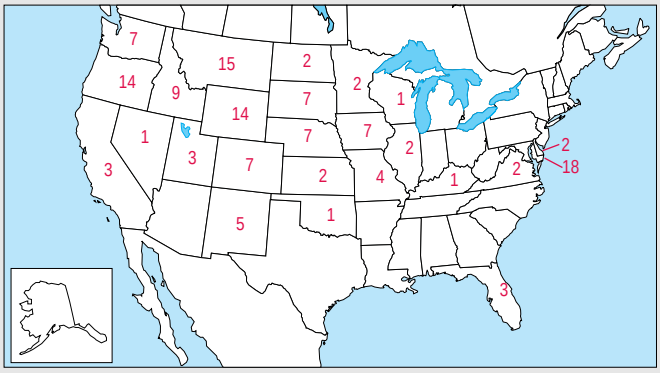

Fig. 3. Nationwide distribution of eagle poisonings caused by organophosphorus and carbamate pesticides.

states). Status determination is complicated by the low density of nesting birds and fluctuations in breeding associated with cycles of prey abundance. It remains in Category 2, i.e., possibly appropriate to propose to list but available data are not conclusive for threatened or endangered status.

Falcons

American peregrine falcon (*Falco peregrinus anatum*) populations declined as a result of contamination by DDT and other organochlorine pesticides. The species was extirpated as a breeding bird in the eastern United States and declared endangered elsewhere. Peregrine recovery has been accomplished in the eastern United States and supplemented in the West (except Alaska) by release of hundreds of peregrines bred in captivity. Now several generations originating from released peregrines have survived and produced young in the wild. In some locales (e.g., parts of California), however, young are still not produced at normal rates. In Alaska nesting numbers of the Arctic subspecies increased naturally, and it was downlisted to threatened status in 1984. Now the Artic peregrine falcon is proposed for removal from the Endangered Species List.

Owls

The distribution of the ferruginous pygmy owl (*Glaucidium brasilianum cactorum*) extends north only into southern Arizona and southern Texas, and concern exists about its status because of the fragmentation and loss of deciduous riparian woodlands and remnant mesquite habitat. The subspecies occurring there, the cactus ferruginous pygmy owl, was elevated from Category 2 as of March 1993 and is being considered for listing as threatened.

The spotted owl (*Strix occidentalis*) is being surveyed extensively and studied because the northern and Mexican subspecies are threatened. In the Pacific Northwest the threat to these owls is loss of old-growth forest, and in the Southwest, general loss of forest habitat. The attention focused on spotted owls has resulted in the only standardized, broad-scale survey of an owl species. Since 1968 the number of known owl nesting areas in Oregon has increased from 27 records (9 sightings, 18 specimens) to about 2,700 separate sites known to be occupied by pairs or single birds sometime within the last 5 years (E. Forsman, U.S. Forest Service, personal communication). This does not reflect an increase in owls; rather, it reflects our ignorance of owl numbers and distribution, largely resulting from lack of survey effort.

Conclusions

Raptors, as top predators, naturally occur at low densities relative to many other organisms. As a group, raptors are poorly surveyed and there are few quantitative data with which to determine their population status and trends. A summary of our assessment of the status and population trends of the 60 species and subspecies of raptors we considered (Table) includes the following: 2 are declining in numbers and 5 are increasing; 16 (27%) are thought to be stable; 19 (32%) are classified as stable, but this assessment is qualified because of local or regional concerns or poor information; the information for 12 (20%) is so poor that we could not determine their status; 7 (12%) of these species or subspecies are endangered or threatened; and 9 (15%) are in Category 2 or 3, reflecting recent concern that they might be endangered or threatened.

We must learn more about the distribution and population dynamics of all our raptor species. With knowledge of their status and trends and information about their distribution and habitat requirements, we can avoid expensive, disruptive, last-resort management of these birds. With knowledge of their ecology, we can conserve biodiversity.

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Return of Wild Turkeys

by
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The wild turkey (*Meleagris gallopavo*) is a large gallinaceous bird characterized by strong feet and legs adapted for walking and scratching, short wings adapted for short rapid flight, a well-developed tail, and a stout beak useful for pecking. These birds probably originated some 2 to 3 million years ago in the Pliocene epoch. Molecular data suggest this genetic line diverged from pheasant-like birds about 11 million years ago. There are two species in the genus, the wild turkey of the United States, portions of southern Canada, and northern Mexico; and the ocellated turkey (*M. ocellata*) in the Yucatan region of southern Mexico, Belize, and northern Guatemala. This article focuses on the return of the wild turkey.

Sources of Information

Historical information on turkeys comes from documented accounts of early explorers, which have been summarized by Mosby and Handley (1943) and Schorger (1966). Recent national population estimates are composite figures obtained from individual state wildlife management agencies. Researchers use many survey techniques including harvest estimates, brood counts, winter flock surveys, and hunter and landowner observations. Kennamer et al. (1992) recently summarized state estimates. At present, there is no consistent, widespread monitoring technique.

Courtesy G. C. Smith

Wild turkey (*Meleagris gallopavo*).

Life History

According to most accounts, wild turkeys were quite abundant at the time of European colonization of North America. Wild turkeys became a major food of these settlers as they moved westward across the forested eastern United States. Turkeys were also used for clothing, ornamentation, and food by many Native American tribes. As the nation grew in the 1800's, wild turkey numbers dwindled. The birds were harvested without restraint and marketed for human consumption. In addition, their forest habitat was cleared for agriculture and wood products. In the early 1900's, population numbers continued to decline. By 1920, wild turkeys were extirpated from 18 of the 39 states of their ancestral range (Mosby and Handley 1943).

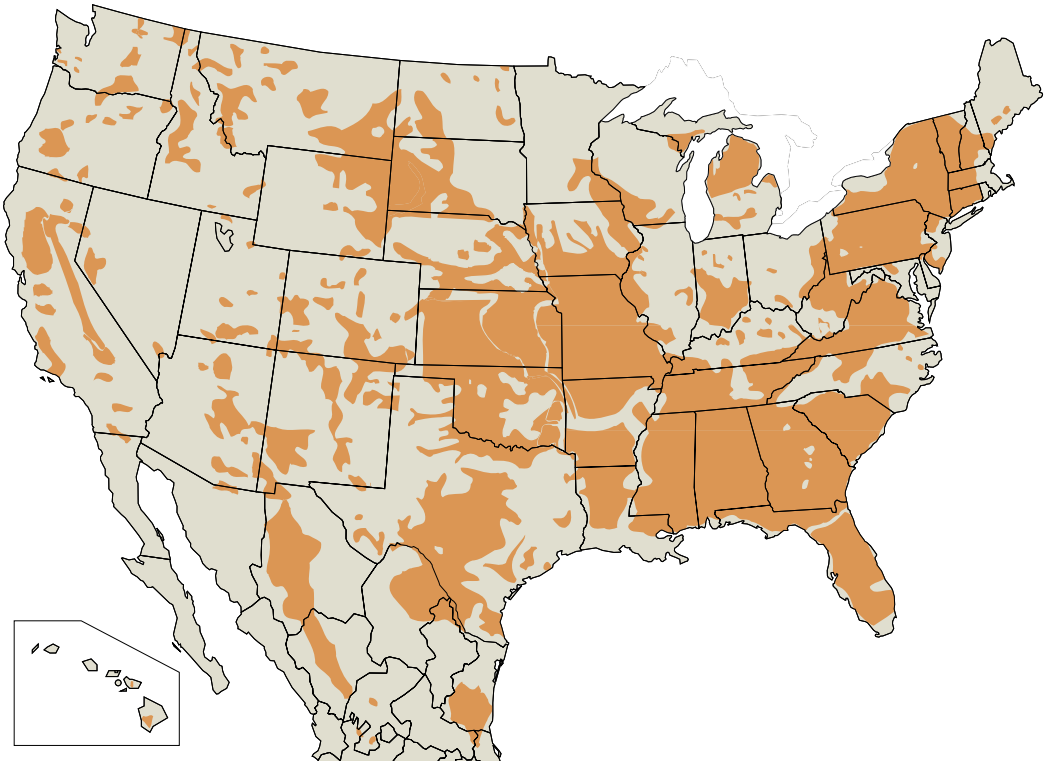

Fig. 1. Distribution of the wild turkey in the United States and Mexico in 1989 (Stangel et al. 1992).

After the early 1900's little change occurred in wild turkey distribution and populations until after World War II when resources were directed to restoring and managing the nation's wildlife populations, including the wild turkey. A technique that many state agencies believed to be promising, but did not work, was artificial propagation of game-farm or pen-raised turkeys. Turkeys raised in captivity were not properly imprinted on (recognition and attachment) wild hens and did not have the experience and survival skills necessary to live and reproduce in the wild.

Restoration through trapping wild turkeys in the wild and relocating them was the proper solution, but this technique was not easily accomplished with the wary bird. Development of the rapidly propelled cannon net, originally designed for capturing waterfowl, was a major factor in relocating large numbers of wild turkeys for restoration. Thousands of wild turkeys were captured or moved with this technique or variations of it; in addition, drop nets and immobilizing drugs were used.

Several other factors contributed to the return of the wild turkey: the maturing of the eastern forests, which had been almost eliminated; increased knowledge from research; spread of sound management practices; and better protection of new flocks vulnerable to poaching.

The restoration of the wild turkey is a great wildlife management success story. In the early part of this century only tens of thousands of wild turkeys were found in a few remote areas. By 1959 the total population approached one-half million (Kennamer et al. 1992), and by 1994 almost all of the forested eastern United States and much of the forested West had been restocked (Fig. 1), with the total population now probably approaching 4 million (Fig. 2). At present, there are viable wild turkey populations with hunting seasons in every state but Alaska, and the annual harvest exceeds one-half million turkeys. The state wildlife management agencies, aided by the National Wild Turkey Federation and supported by sportmen's dollars, undertook a tremendous task and achieved dramatically successful results (Dickson 1992). Turkey hunting continues to be pursued by millions of dedicated hunters.

Future population expansion is expected to be somewhat limited. Most suitable turkey habitat has been stocked, and, generally, populations in these areas have already gone through their high-productivity phase. Population expansion is also limited because appropriate habitat will be lost as the human population expands.

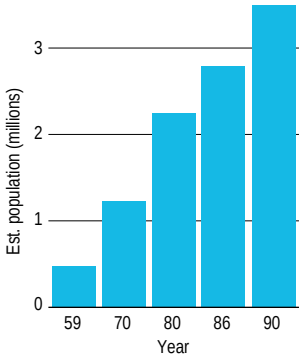

Fig. 2. Estimated U.S. wild turkey population, 1959-90 (from Kennamer et al. 1992).

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Mourning Doves

by
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The mourning dove (*Zenaida macroura*) is one of the most widely distributed and abundant birds in North America (Droege and Sauer 1990). It is also the most important U.S. game bird in terms of numbers harvested. The U.S. fall population of mourning doves has been estimated to be about 475 million (Tomlinson et al. 1988; Tomlinson and Dunks 1993).

The breeding range of the mourning dove extends from the southern portions of the Canadian Provinces throughout the continental United States into Mexico, the islands near Florida and Cuba, and scattered areas in Central America (Aldrich 1993; Fig. 1). Although some mourning doves are nonmigratory, most migrate south to winter in the United States from northern California to Connecticut, south throughout most of Mexico and Central America to western Panama.

Within the United States, three areas contain breeding, migrating, and wintering mourning dove populations that are largely independent of each other (Kiel 1959). In 1960 three areas were established as separate management units: the Eastern (EMU), Central (CMU), and Western (WMU; Fig. 1).

The two main tools used to manage mourning doves are an annual breeding population survey (known as the Mourning Dove Call-count Survey; Dolton 1993a, b) and harvest surveys. The Call-count Survey provides an annual index to population size as well as data for determining long-term trends in dove populations. State harvest surveys and the National Migratory Bird Harvest Information Program, begun in 1992, estimate dove harvest. In addition, recoveries from banded doves have provided vital information for managing the species (Hayne 1975; Dunks et al. 1982; Tomlinson et al. 1988).

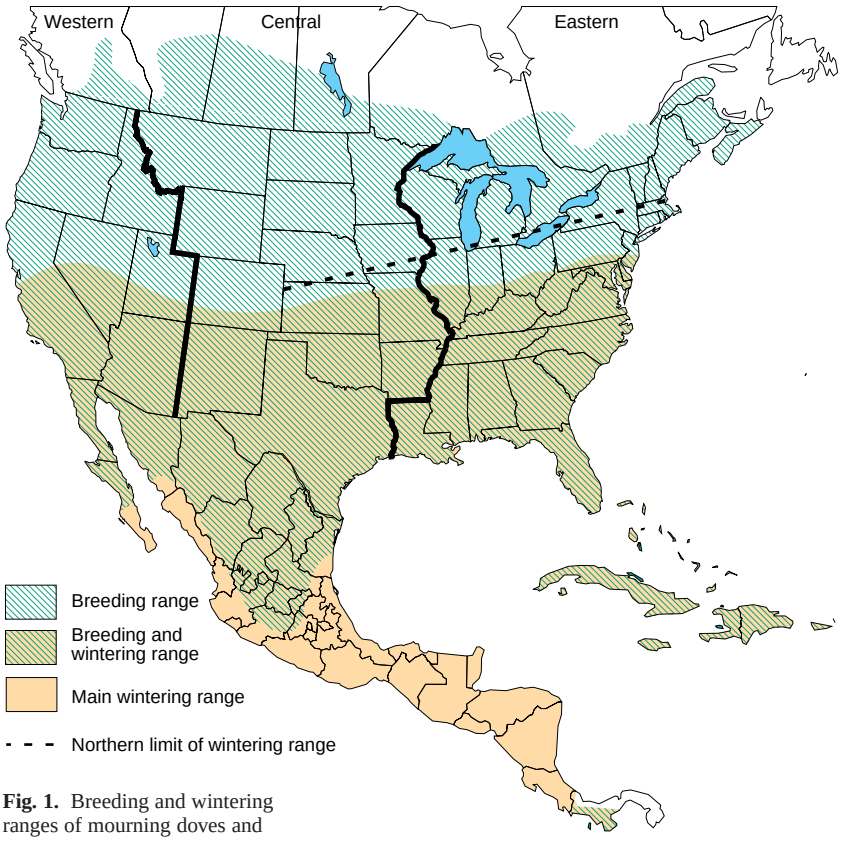

Fig. 1. Breeding and wintering ranges of mourning doves and mourning dove management units in the United States.

Status and Trends

The Eastern Management Unit includes 27 states—30% of the U.S. land area. The 1993 population indices were 18.3 doves heard and 14.9 doves seen per route (Dolton 1993b; Fig. 2). Both estimates are above the long-term trend estimates. Between 1966 and 1993, the population has been relatively stable. Dove harvest in the EMU was relatively constant from 1966 to 1987, with between 27.5 million and 28.5 million birds taken. The latest estimate, a 1989 survey, indicated that the harvest had dropped to about 26.4 million birds shot by an estimated 1.3 million hunters (Sadler 1993).

The Central Management Unit consists of 14 states containing 46% of the U.S. land area. Of the three units, the CMU has the highest mourning dove population index. The 1993 index for the unit of 23.9 doves heard per route is slightly below the long-term trend estimate (Dolton 1993b; Fig. 2). For doves seen, the estimate of 26.8 is also below what was expected. Even though there appears to be an increase in doves seen and a slight decrease in doves heard between 1966 and 1993, in statistical terms there is no significant trend indicated for either count. Although hunting pressure and harvest varied widely among states, dove harvest in the CMU generally increased between 1966 and 1987 to an annual average of about 13.5 million birds. In 1989 almost 11 million doves were taken by about 747,000 hunters (Sadler 1993).

The Western Management Unit comprises seven states and represents 24% of the land area in the United States. The 1993 population indices of 9.3 doves heard and 8.5 doves seen per route are slightly above their long-term trend estimates (Dolton 1993b; Fig. 2). Significant downward trends in numbers of doves heard and seen for the unit occurred between 1966 and 1993. From 1987 to 1993, however, a significant positive trend occurred in the unit although the indices were still below those of the 1960's. After a decline in the dove breeding population, dove harvest in the WMU declined significantly. In the early 1970's, about 7.3 million doves were taken by an estimated 450,000 hunters. By 1989, the harvest had dropped to about 4 million birds shot by about 285,000 hunters (Sadler 1993).

In summary, mourning dove populations in the EMU and CMU are relatively stable. Although the population of doves in the WMU declined from a high in the mid-1960's, it appears that it stabilized during the past 7-10 years. U.S. dove harvest appears to be decreasing. The mourning dove remains an extremely important game bird, however, especially since more doves are harvested than all other migratory game birds combined. A 1991 survey indicated that the mourning dove provided about 9.5 million days of hunting recreation for 1.9 million people (USFWS and U.S. Bureau of Census 1993).

Year-to-year population changes are normal and expected. Although populations are relatively stable in the Eastern and Central Management units, declining long-term trends in the past two decades are cause for concern in the Western Unit and in local areas elsewhere. A combination of factors may have been detrimental to dove populations in some areas: habitat and agricultural changes including loss of nesting habitat through reclamation and industrial and urban development, changes in agricultural practices that may have reduced food sources, and possibly overharvest of doves in local areas. In California, for example, many live oak trees have been cut for wood products resulting in a loss of nesting habitat. Reclamation projects or lowered water tables eliminated thousands of acres of mesquite nesting habitat in Arizona. Since many doves from the WMU winter in Mexico during a 5- to 6-month period each year, agricultural changes there may negatively affect doves.

In the CMU, agricultural changes were evaluated and compared with dove population trends in the eastern group of states (R.R. George, Texas Parks and Wildlife Department, unpublished data); mourning dove population

indices appeared to be most closely correlated with changes in number of farms (positive) or farm size (negative). In addition, an analysis identified number of farms and acres of soybeans, oats, and sorghum over time as good indicators of the number of doves heard.

Early records indicate that mourning doves were present, although not abundant, when the United States was settled by colonists (Reeves and McCabe 1993). The resulting clearing of forests, introduction of new food plants, grazing and trampling by livestock that promoted seed-producing plants used by doves, and the creation of stock ponds providing more widely distributed drinking water in the arid West all benefited the mourning dove so that they are probably more numerous now than in colonial times.

These birds are quite adaptable and readily nest and feed in urban and rural areas. The mourning dove has recently even expanded its range northward.

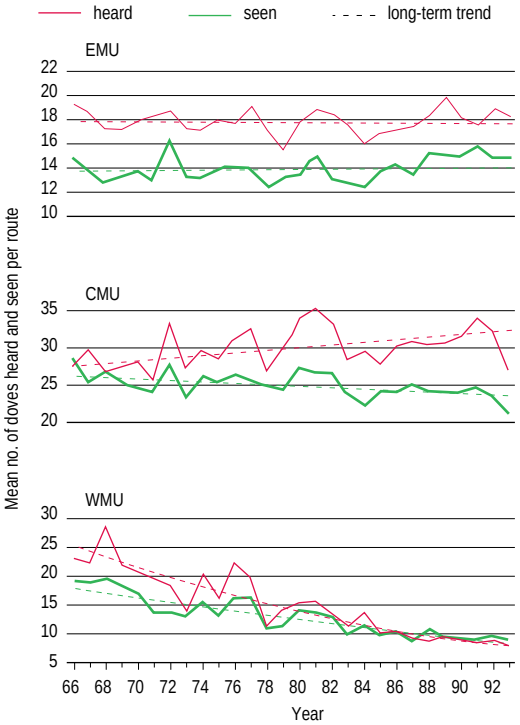

Fig. 2. Population indices of mourning doves in the Eastern (EMU), Central (CMU), and Western (WMU) Management units, 1966-93.

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Common Ravens in the Southwestern United States, 1968-92

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The common raven (*Corvus corax*) is a large black passerine bird found throughout the northern hemisphere including western and northern North America. Ravens are scavengers that frequently feed on road-killed animals, large dead mammals, and human refuse. They kill and eat prey including rodents, lambs (Larsen and Dietrich 1970), birds, frogs, scorpions, beetles, lizards, and snakes. They also feed on nuts, grains, fruits, and other plant matter (Knight and Call 1980; Heinrich 1989). Their recent population increase is of concern because ravens eat agricultural crops and animals whose populations may be depleted.

Ravens are closely associated with human activities, frequently visiting solid-waste landfills and garbage containers at parks and food establishments, being pests of agricultural crops, and nesting on many human-made structures. In two recent surveys in the deserts of California (FaunaWest Wildlife Consultants 1989; Knight and Kawashima 1993), ravens were more numerous in areas with more human influences, and were often indicators of the degree to which humans affect an area.

Annual Breeding Bird Surveys (BBS) conducted nationwide by the U.S. Fish and Wildlife Service (USFWS) indicated that raven

populations in several parts of the country significantly increased during 1965-79 (Robbins et al. 1986). This increase concerns resource managers because ravens feed on agricultural crops and animal species of interest to humans. For instance, in the deserts of the southwestern United States, ravens prey on young desert tortoises (*Gopherus agassizii*; Berry 1985; Fig. 1), which in the Mojave and Colorado deserts are listed as a threatened species by the USFWS (Federal Register 1990). Because of high levels of raven predation on tortoises, the Bureau of Land Management has taken action to reduce this predation (BLM 1990, 1994). We report here on a 24-year trend in raven abundance along roadsides in the deserts of the southwestern United States and surrounding regions, where increasing raven populations interest resource management agencies (BLM 1990; USFWS 1994).

Courtesy W. I. Boarman, NBS

Fig. 1. Juvenile desert tortoise shell found beneath an active raven nest. The hole in the shell was probably pecked open by a raven to eat the organs.

Our analysis of BBS 1968-92 data focuses on arid lands and neighboring habitats in California, Nevada, Utah, and Arizona. We used data from 137 39.2-km (24.5-mi) routes within the following BBS strata: Great Basin Desert; mountain highlands of Arizona; Sonoran-Colorado Desert; Mojave Desert; basins and ranges, including portions of the northern Mojave and Great Basin deserts; Central Valley; and southern California grasslands, California foothills (southern California routes only), and Los Angeles ranges combined into one (coastal southern California).

Status and Trends

Between 1968 and 1992, the latest year for which data were available, raven populations increased significantly ($P \leq 0.01$) throughout the study area (Fig. 2), in spite of relatively high variances among routes. Raven sightings increased 76-fold in the Central Valley of California, 14-fold in the Sonoran-Colorado Desert, and 10-fold in the Mojave Desert over the 24-year period. Statistically significant but lower increases in raven populations were experienced in the heavily urbanized coastal southern California strata. The results for the mountain highlands stratum are questionable because of a low number of routes (n = 7; B. Peterjohn, NBS, personal communication).

In three studies, raven numbers were highest along powerlines, intermediate along highways, and lowest in open desert areas (Austin 1971; FaunaWest Wildlife Consultants 1989; Knight and Kawashima 1993). These reports and observations of raven use of human-based resources for food, water, and nesting substrate (Knight and Call 1980; FaunaWest Wildlife Consultants 1989; Heinrich 1989) suggest that high raven populations are a result of human subsidies (Boarman 1993).

Increased raven populations may be a concern for threatened and endangered species if increased numbers of ravens result in greater predation. In California alone, there are 96 threatened or endangered species, some of which are or may be at risk of increased raven predation if raven populations continue to grow. On San Clemente Island, ravens are a predator of the endangered San Clemente Island loggerhead shrike (*Lanius ludovicianus mearnsi*), and along coastal California they prey on endangered populations of the California least tern (*Sterna antillarum browni*; Belluomini 1991). The carcasses of 11 chuckwallas (*Sauromalus obesus*), a candidate species for listing as threatened or endangered by the USFWS, were recently found beneath one raven nest (personal observation). This finding may be a rare occurrence, but if raven populations continue to increase, more ravens may begin to prey on chuckwallas. We are conducting more research to understand the foraging ecology and population biology of ravens and their effects on their prey populations. This research will help us determine how much of a threat ravens pose to the region's biodiversity and learn how to reduce these effects.

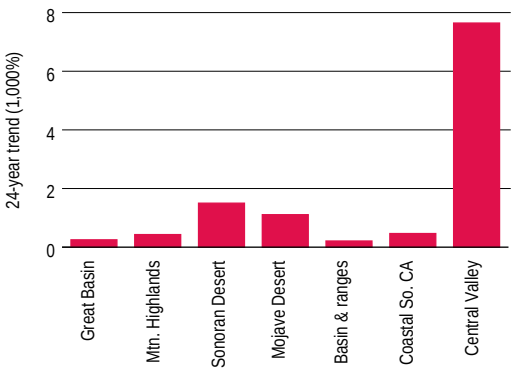

Fig. 2. A 24-year trend in the average (mean) number of raven sightings within each stratum studied.

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Mississippi Sandhill Cranes

by
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Resident sandhill cranes formed a continuous population in Georgia and Florida and widely separated populations along the Gulf Coastal Plain of Texas, Louisiana, Mississippi, and Alabama (Figure). The Mississippi sandhill crane (*Grus canadensis pulla*) was one of the widely separated populations on the Coastal Plain that bred in pine savannas in southeastern Mississippi, just east of the Pascagoula River to areas just west of the Jackson County line, south to Simmons Bayou, and north to an east-west line 8-16 km (5-10 mi) north of VanCleave.

Agricultural and industrial development including World War II ship building, fire suppression, and forestry practices destroyed much of the sandhill crane's habitat in Jackson County, Mississippi. The U.S. Fish and Wildlife Service (USFWS) added the Mississippi sandhill crane to the endangered species list in 1973 and established the Mississippi Sandhill Crane National Wildlife Refuge in 1974. The USFWS began captive breeding at the Patuxent Wildlife Research Center (PWRC) in 1965 to protect the subspecies during habitat restoration and to provide stock for reintroduction.

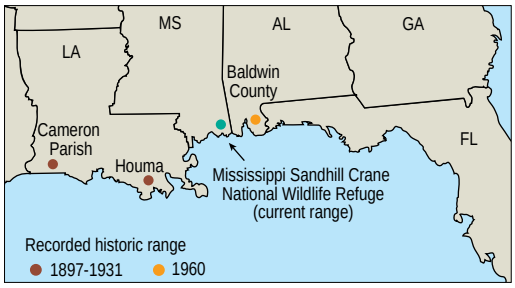

Figure. Range of Mississippi sandhill cranes.

Morphological, physiological, and genetic differences exist among crane subspecies (Aldrich 1972). Mississippi birds mature earlier and begin egg production about 6 weeks later than Florida sandhill cranes. Genetic studies (Dessauer et al. 1992; Jarvi et al. 1994) show a level of heterozygosity (*see* glossary) in the wild Mississippi population about half that in other sandhill cranes. As in other small populations, cranes seem to have genetic weaknesses. In the captive population, for example, 17% of all birds die from detectable heart murmurs and when released to the wild, 36% with heart murmur and 83% without heart murmurs survive for 1 year after release.

Status and Trends

Population Decline

In the 1800's the species was abundant enough for farmers to consider it a pest. Although population studies only started recently, it appears the population has been small for most of this century (Table 1).

Table 1. Estimated numbers of Mississippi sandhill cranes on the Mississippi Sandhill Crane National Wildlife Refuge, 1929-93.

Year	Wild	Captive released	Total
1929			50 -100
1949			50+
1969			50-60
1975			30-50
1978			40-50
1979			40-50
1980			50
1981	41	9	50
1982	41	9	50
1983	34	9	43
1984	27	13	40
1985	13	19	32
1986	23	18	41
1987	17	16	33
1988	21	23	44
1989	21	33	54
1990	24	49	73
1991	19	73	92
1992	20	88	108
1993	20	115	135

Until the 1940's, the human population in Jackson County was small, and the remnant population of Mississippi sandhill cranes remained stable. The suitable pine savanna habitat shrunk from over 40,500 ha (100,000 acres) in 1940 to 10,530 ha (26,000 acres) in the 1960's, which were designated as critical habitat by the USFWS. The USFWS requested a population study in 1960 when Mississippi proposed building Interstate Highway 10 through the last of the crane habitat. The Nature Conservancy, the U.S. Department of Transportation, and State of Mississippi donated land to the refuge.

Recent Reintroductions

The first releases of hand-reared birds failed. Thus, releases of Mississippi sandhills on the refuge during the 1980's were birds raised by their parents or surrogate parents. These parent-reared birds proved wilder than the hand-reared birds and adapted well to the pine savanna. Unfortunately, the parent-rearing technique reduced production and increased expenses.

The PWRC developed a new hand-rearing technique that visually isolated chicks from humans and imprinted them on adult sandhill cranes in the chick-rearing area. Caretakers dressed in sheets to hide their human form when handling birds, and encounters with cranes were limited. Juveniles were placed in socialization pens in the fall to form three cohorts (parent-reared, hand-reared, and a mixed group). A gentle release on the refuge allowed the birds to leave the release pen when ready and to return for food for a period after release. Surprisingly, a greater percentage of hand-reared birds has survived than the parent-reared birds, although both groups have paired and produced fertile eggs. The releases increased the refuge population from 44 in 1988 to 135 in 1993 (Table 1).

Status in Jackson County, Mississippi

The population decline of the Mississippi sandhill crane reflects the loss of the mesic and hydric pine savanna once abundant in the area. Savannas occur on coastal terraces, elevated ridges, and uplands. Fire frequency and intensity, combined with soil type and hydrology, provide successional regulation of the savanna. Woody, forested communities replace the savanna without fire. Before ditching, the flat topography of the terraces allowed sheet flow of water across the terraces and supported extensive areas of open savanna. When the refuge was established, about 75% of the crane savannas had been destroyed (by residential or commercial development) or changed to one of several different forest types. Only 5% of the original savanna type that supported the cranes remains on the Gulf Coastal Plain. For this reason, Mississippi sandhill cranes now occur only on the refuge and adjacent private lands in southeastern Mississippi.

The Mississippi sandhill crane population nests only on the 7,813-ha (19,300-acre) refuge. The only other large tract of remnant savanna that might be suitable nesting habitat exists southeast of the refuge on the proposed Grand Bay National Wildlife Refuge. Savanna used by the Mississippi sandhill crane exists as highly fragmented remnants that the refuge must manage to provide nesting, foraging, and roosting sites (Table 2).

Mortality and natural recruitment may also restrict population viability. Predation (primarily mammalian) causes high mortality during the first year of life. Other factors that may limit populations include tumors, contaminants, microbial pathogens, and parasites. The prevalence of tumors in the wild Mississippi sandhill crane population far exceeds that expected in other birds and mammals.

Table 2. Mississippi sandhill crane nesting sites on refuge, by habitat.

Type of habitat	Number	Percentage
Open savanna	82	49
Swamp edges	62	38
Pine plantations	12	7
Forest edges	8	5
Cleared lands	2	1

Research Needs

Research needs include assessing the effects of prescribed burns and other mechanical techniques on habitat restoration and crane use; assessing the effects of water levels, water-level fluctuations, and hydrology on crane nesting and fledging success; determining the level of propagation and captive release conditioning needed to maintain population size during restoration; developing genetic management to protect the gene pool; and determining disease and contaminant sources for tumors and poor reproductive success in captive and wild flocks.

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Piping Plovers

by
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Editor's note: This paper is largely a synopsis of a paper by Haig and Plissner (1993) in *Condor*.

The piping plover (*Charadrius melodus*) is a wide-ranging, beach-nesting shorebird whose population viability continues to decline as a result of habitat loss from development and other human disturbance (Haig 1992). In 1985 the species was listed as endangered in the Great Lakes Basin and Canada and threatened in the northern Great Plains and along the U.S. Atlantic coast. The U.S. Fish and Wildlife Service (USFWS) is proposing that birds in the northern Great Plains also be listed as endangered.

Each year, many breeding areas are censused and some winter surveys are conducted. In 1991 biologists from Canada, the United States, Mexico, and various Caribbean nations carried out a simultaneous census of piping plovers at all known breeding and wintering sites. Census goals were to establish baseline population levels for all known piping plover sites and to census additional potential breeding and wintering sites (Figure).

Status

This census covered 2,099 sites, resulting in the highest number of breeding and wintering piping plovers ever recorded. It will be repeated three or four more times over the next 15-20 years for more accurate assessment of population trends.

Winter Census

The total number of wintering birds (3,451) reported constituted 63% of the breeding birds (5,486) counted (Tables 1, 2). Most birds (55%; N = 1,898) were found along the Texas coast where the census concentrated on birds in previously uncensused stretches of Laguna Madre's back bays. The highest concentration of birds in local sites was also reported in Texas (Haig and Plissner 1993). Although the 1991 census discovered more wintering birds than had been previously reported, a large proportion of piping plovers were not seen in the winter census.

Better census efforts in Louisiana, northern Cuba, and on many of the smaller Caribbean islands may reveal additional winter sites. Previous reviews of their distribution did not indicate that birds moved farther south than the Caribbean (Haig and Oring 1985). Relatively few birds are seen on the Atlantic coast in winter, a contrast to the 36% of plovers that breed along the Atlantic coast. Thus, the largest gap in our understanding of piping plover distribution during winter appears to be in locating winter sites for Atlantic coast breeders.

Breeding Census

All known piping plover breeding sites were censused in 1991 (Table 2). Piping plovers were widely distributed in small populations across their breeding range (Figure); most adults (63.2%) bred in the northern Great Plains and prairies of the United States and Canada. Thirty-six percent were found on the Atlantic coast and

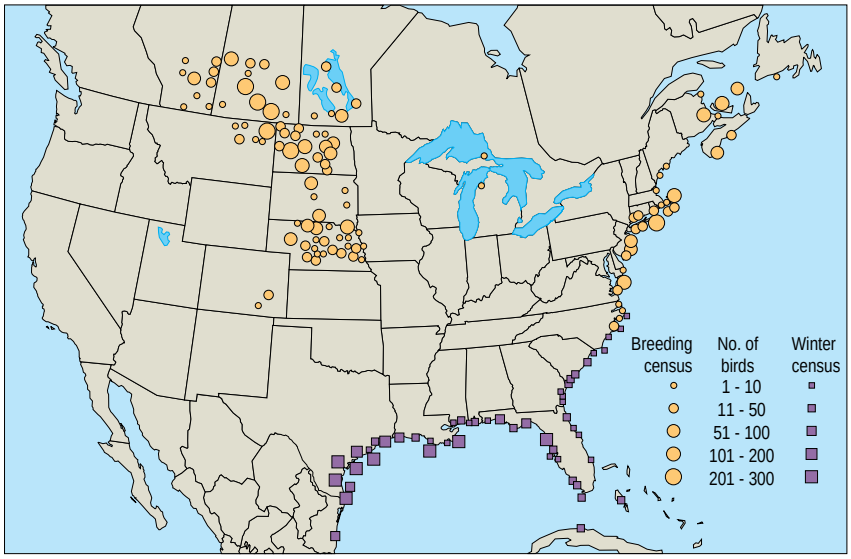

Figure. Distribution of piping plovers throughout the annual cycle in 1991.

Table 1. Numbers of wintering piping plovers and sites where birds occurred in 1991.

Location	Birds	Sites
U.S. Atlantic		
North Carolina	20	7
South Carolina	51	8
Georgia	37	6
Florida	70	9
Total	178	30
U.S. Gulf		
Florida	481	31
Alabama	12	1
Mississippi	59	7
Louisiana	750	23
Texas	1,904	64
Total	3,206	126
Mexico Gulf	27	4
Caribbean		
Bahamas	29	1
Turks and Caicos	0	0
Cuba	11	1
Jamaica	0	0
Puerto Rico	0	0
Cayman Islands	0	0
Total	40	2
Combined total	3,451	162

less than 1% occurred on the Great Lakes. Sites with the highest concentrations of breeding birds also were found in the northern Great Plains (also known in Canada as the Great Prairie); however, each local population consisted of only a small (less than 8%) proportion of the total breeding population. Local populations were even smaller on the Atlantic coast.

Migration Areas

Atlantic coast piping plovers are commonly seen on east coast beaches during spring and fall migration. Migration routes of inland birds are poorly understood, however. Only a few occurrences of piping plovers have been reported at seemingly appropriate inland migration sites such as Kirwin National Wildlife Refuge in Kansas, Cheyenne Bottoms Wildlife Management Area in Kansas, and Great Salt Plains National Wildlife Refuge in Oklahoma. It appears that inland birds may fly nonstop to gulf coast sites.

Trends

Because simultaneous, species-wide censuses were not conducted in the past, assessing population trends is difficult. Examination of long-term census data at specific sites is useful in some cases. Most midcontinent sites that have been monitored for 10 years or more have experienced a decline (Table 3). The cumulative effects of problems in the prairies have been modeled, and results indicate that piping plovers in the Great Plains are now declining by 7% annually (Ryan et al. 1993), a devastating trend for the species. Atlantic coast numbers remain stable; however, there has been unprecedented effort to protect piping plovers along the U.S. Atlantic coast. Results from previous censuses (Table 3) should be considered rough population estimates; as is true with many bird species, we have little information regarding the intensity of census efforts in those population estimates.

Threats

In the northern Great Plains, water-level regulation policies on the major rivers (e.g., Platte, Missouri) serve as a direct source of chick mortality and an indirect source of habitat loss through vegetation encroachment and flooding (Schwalbach 1988; Sidle et al. 1992). We know that because 20% of northern Great Plains (Great Prairie) birds use river sites, loss of productivity on rivers such as the Missouri can

Table 2. Piping plover breeding census, 1991.

Location	Adults	Sites where piping plovers occurred
Atlantic Coast		
Canada		
New Brunswick	203	24
Newfoundland	7	1
Nova Scotia	113	34
Prince Edward Island	110	20
Quebec	76	11
St. Pierre/Miquelon	4	2
Canada Atlantic total	513	92
U.S.		
Maine	38	8
Massachusetts	293	50
Rhode Island	47	7
Connecticut	67	7
New York	338	69
New Jersey	280	22
Delaware	10	3
Maryland	35	1
Virginia	270	14
North Carolina	86	14
South Carolina	2	1
U.S. Atlantic total	1,466	196
Atlantic total	1,979	288
Great Lakes		
Duluth, MN	0	0
Wisconsin	1	1
Michigan	39	14
Long Point, Ontario	0	0
Great Lakes total	40	15
Northern Great Plains/Prairie		
Canada Prairie		
Alberta	180	27
Saskatchewan	1,172	71
Manitoba	80	12
Lake of Woods, Ontario	5	1
Canada Prairie total	1,437	111
U.S. Great Plains		
Montana	308	39
North Dakota	992	115
South Dakota	293	47
Lake of Woods, MN	13	1
Colorado	13	4
Nebraska	398	106
Iowa	13	2
Kansas	0	0
Oklahoma	0	0
U.S. Great Plains total	2,030	314
Combined totals		
Canada	1,950	203
United States	3,536	525
Total	5,486	728

significantly affect annual productivity for the species. A similar threat to piping plovers occurs on Lake Diefenbaker in Saskatchewan, the largest piping plover breeding site in the world, where each year water levels are raised soon after parents have laid their clutches, resulting in a loss of all nests.

Avian and mammalian predation is a problem throughout the species' breeding range, although population numbers appear to be stabilizing on the Atlantic coast and the Great Lakes as a result of using predator exclosures over nests (Rimmer and Deblinger 1990; Mayer and Ryan 1991; Melvin et al. 1992). Human disturbance continues to be a problem on the Atlantic coast (Strauss 1990), and in the Great Lakes, piping plovers may also be suffering from a lack of viable habitat (Nordstrom 1990). Comparison of food availability at northern Great Plains sites with Great Lakes sites indicated lower diversity and abundance of invertebrates on the Great Lakes. Finally, recent evidence suggests that Great Lakes birds may be suffering from high levels of toxins (i.e., PCB's), which may be a prime factor in low productivity and population growth (USFWS, East Lansing, Michigan, personal communication).

The discovery of the high proportion of wintering piping plovers on algal and sand flats has significant implications for future habitat protection. Current development of these areas on

Courtesy S. Haig

Piping plover (*Charadrius melodus*).

Laguna Madre in Texas and Mexico, increased dredging operations, and the continuous threat of oil spills in the Gulf of Mexico will result in serious loss of piping plover wintering habitat.

In summary, piping plovers suffer from many factors that may cause their extinction in the next 50 years. Most devastated are the Great Lakes and northern Great Plains birds whose viability is severely threatened. Unfortunately, recovery is hindered by a lack of knowledge about the winter distribution, status of winter sites, adequate water-management policy in western breeding sites, and direct human disturbance on the Atlantic coast.

Table 3. Changes in numbers of piping plovers at specific breeding areas.*

Location	1st est.		2nd est.		1991 census	% Change 1st est. 1991	% Change 2nd est. 1991
	Year	No.	Year	No.			
Atlantic Coast							
Newfoundland	1968	30	1984	4	7	-72	+75
Cadden Beach, Nova Scotia	1976	56	1983	28	20	-64	-29
Maine	1976	48	1982	12	38	-21	+217
Rhode Island	1945	80	1983	20	47	-41	+135
Connecticut	1980	40	1983	34	67	+68	+97
Long Island, NY	1939	1,000	1983	200	338	-66	+69
New Jersey	1980	118	1983	64	280	+137	+338
Delaware	1978	80	1984	18	10	-88	-44
Maryland	1972	85	1984	25	35	-59	+40
Great Lakes							
Michigan	1979	77	1982	14	39	-49	+179
Wisconsin	1900	140	1983	6	1	-99	-83
Northern Great Plains/Prairie							
Big Quill Lake, Saskatchewan	1978	210	1984	186	151	-28	-19
Chain Lakes, Alberta	1976	50	n.a.	n.a.	9	-72	n.a.
Lake Manitoba, Manitoba	1980	27	1984	9	3	-89	-67
Lake of the Woods, MN	1982	44	1986	32	13	-70	-59
Niobrara River, NE 1978	1981	92	1985	100	110	+20	+10

*Sources are listed in Haig and Oring (1985) and Haig and Plissner (1993).

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California Condors

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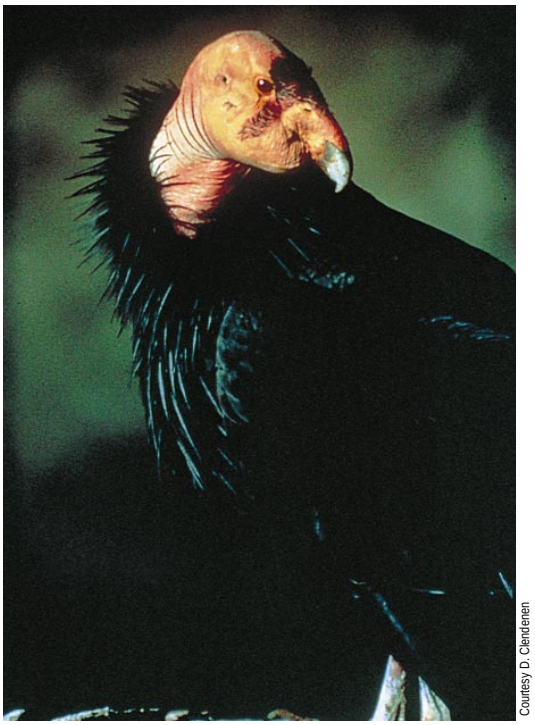
The California condor (*Gymnogyps californianus*) is a member of the vulture family. With a wingspan of about 3 m (9 ft) and weighing about 9 kg (20 lb), it spends much of its time in soaring flight visually seeking dead animals as food. The California condor has always been rare (Wilbur 1978; Pattee and Wilbur 1989). Although probably numbering in the thousands during the Pleistocene epoch in North America, its numbers likely declined dramatically with the extinction of most of North America's large mammals 10,000 years ago. Condors probably numbered in the hundreds and were nesting residents in British Columbia, Washington, Oregon, California, and Baja California around 1800. In 1939 the condor population was estimated at 60-100 birds, and its home range was reduced to the mountains and foothills of California, south of San Francisco and north of Los Angeles.

Conservation to halt the condor's decline included establishing the Sisquoc (1937) and Sespe (1947) condor sanctuaries within the Los Padres National Forest, obtaining fully protected status under California Fish and Game Code (1953), placement on California's first state endangered species list (1971), and, finally, being listed by the federal government under the Endangered Species Act of 1973 (Wilbur 1978). The success of these efforts could not be judged, however, because verifiable status and trends data did not become available until 1982. By using these data, we confirmed the decline in condor numbers over the past 50 years was even greater than thought.

Population estimates before 1939 were based entirely on guesswork and interpretation of the fossil record, historical accounts, museum collections, or anecdotal observations by early naturalists and scholars. We believed there were fewer condors because they were no longer seen in many areas where they were once commonly observed. The condor's plight generated widespread interest among conservationists to know the actual population size and its rate of decline.

Koford (1953) conducted the first major life-history study of the California condor and provided the first documented enumeration of the species. His count was based on numbers seen in the largest single flocks with an unspecified adjustment for condors not seen. Another estimate in 1965 (Miller et al. 1965) compared flock sizes seen in the late 1950's and early 1960's with those reported by Koford.

A yearly survey was begun by volunteers in 1965 and continued through 1981 (except for 1979). This survey used multiple observers at strategic sites who counted all condors seen for a 2-day period in October (Mallette and Borneman 1966; Wilbur 1980). The yearly population estimates of this October survey were quite different from year to year and failed to provide any statistical measures of variability, although results did show a gradual downward trend in condor numbers.



Courtesy D. Clendenen

California condor (*Gymnogyps californianus*).

The annual October survey was replaced in 1982 by a counting method (Snyder and Johnson 1985) using photographs of soaring condors to recognize differences in feather patterns. This method allowed individuals to be identified and counted. Although an improvement over previous techniques, this method is time consuming and only works when there are few animals. The photographic census was discontinued after 1985 because all condors had been marked with uniquely colored and numbered tags and radio transmitters.

Trends

Data used to determine the population size of California condors before 1982 (Figure) were biased for many reasons. Foremost was the fact that no surveyors could explain how they used the number of condors they saw to estimate how many condors actually existed. Nor could they say how sure they were of being right. Consequently, the severity of the decline and number of condors dying were grossly underestimated. Because management was unaware of the severity of the decline and urgency of the crisis, critical decisions to save the condors

were delayed. For example, the ability to recognize individuals based on methods that started in 1982 (Table) allowed us to realize we had lost five adult condors (about 30% of the wild population) during winter 1984-85. Understanding the critical nature of this loss ultimately led to the decision to capture the remaining wild birds.

As of January 1994 there were 66 birds, and the future of the captive population appears bright. The World Center for Birds of Prey in Boise, Idaho, became the third captive site in September 1993, joining the San Diego Wild Animal Park and the Los Angeles Zoo. The George Miksch Avian Research Center in Bartlesville, Oklahoma, is scheduled to become the fourth captive breeding facility in 1994. We expect all captive flocks to do well and continue to increase, providing young birds for release in California as well as yet-to-be selected sites in Arizona and New Mexico.

Timely and accurate status and trends data will continue to be important to the condor recovery program as more birds are released. Not only will these data be needed to monitor the success of the release, but also they are essential for identifying problems, which is especially critical because no known or suspected mortality factors in California have been significantly reduced, much less eliminated. The relocation of all released California condors to a site near the Sisquoc Sanctuary after the death of the fourth bird (three lost to powerline collisions) reflects the close monitoring necessary to ensure that appropriate actions can be taken as quickly as possible.

With the wild population consisting of only nine young birds with a restricted range and still dependent on artificial feeding stations, conventional radiotelemetry and tagging have been adequate. As the number of birds increase and their territories expand, however, conventional methods for monitoring and locating birds will be unable to fulfill the recovery program's needs. For the release program to succeed, we will need to identify and remove or avoid key mortality factors such as the powerline collision hazard at the first site. To accomplish this, we will need to monitor and locate dozens of individual condors scattered over a million or more hectares. Equipment to do this exists but has not been modified or adequately tested for use on condors. Eventually a simple, inexpensive survey procedure will be needed to track the wild condor population as it increases and starts reproducing. Developing these procedures now is essential.

Courtesy D. Clendenen

California condors have a wingspan of about 3 m or 9 ft.

Year	No. of captive birds	No. of wild birds	Total
1982	3	21	24
1983	9	16	25
1984	16	11	27
1985	21	6	27
1986	25	2	27
1987	27	0	27
1988	28	0	28
1989	32	0	32
1990	40	0	40
1991	52	0	52
1992	56	7	63
1993	66	9	75

Table. Status of the wild and captive California condor populations, 1982-93.

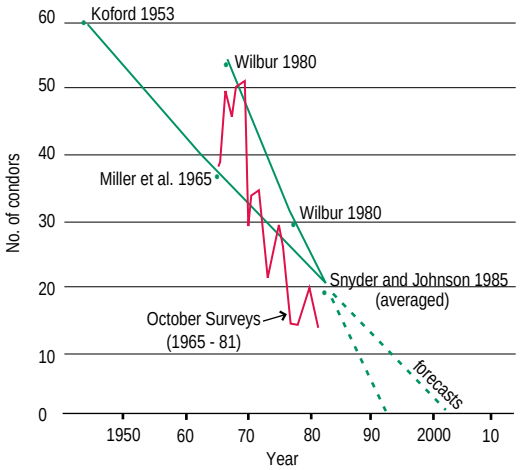


Figure. Estimates of the California condor population, 1945-82 (Snyder and Johnson 1985). Used with permission from the *Condor*©.

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Audubon's Crested Caracara in Florida

by
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Audubon's crested caracara (*Caracara plancus audubonii*) is a species characteristic of the grassland ecosystems of central Florida and is one of the state's most distinctive birds. The Florida population is threatened and widely separated from the main species' range, which extends from extreme southwestern Louisiana, southern Texas, and southern Arizona to the tip of South America, including Tierra del Fuego and the Falkland Islands. Another isolated population occurs on Cuba and the Isle of Pines.

Courtesy J.N. Layne

Audubon's crested caracara (*Caracara plancus audubonii*) in Florida.

The number of Florida caracaras is believed to have undergone a substantial decline from the early historic level in the 1950's and 1960's (Layne in press), with the total state population estimated at 250 in the early 1950's (Sprunt 1954) and fewer than 100 birds in the late 1960's (Heinzman 1970). Based on the apparent continuing decrease in its numbers, Florida's population of Audubon's crested caracara was federally listed as threatened in 1987 (Federal Register 1987). As part of a general study of the life history, ecology, and behavior of the caracara in Florida, I monitored its distribution and population status from 1972 to 1991.

Information was obtained from road and off-road searches in all parts of the known range; systematic roadside and aerial surveys in a 5,116-km^2 (1,975-mi^2) area within the core portion of the range; published records; museum specimens; and sighting reports from over 500 cooperators. Logistical limitations prevented surveying the entire potential Florida range thoroughly enough in any given year to obtain a reasonably accurate picture of the distribution and total population. Thus, estimates of the statewide distribution and numbers were based on records combined over 5-year periods: 1972-76, 1977-81, 1982-86, and 1987-91. Searches were most intensive from 1972 to 1981 and in the final period 1987-91. Because areas along public roads were surveyed more intensively than those remote from highways, there was a lower probability of detecting caracaras whose territories did not overlap roads than those whose territories included roads. This bias appeared to be at least partially compensated for by a tendency of caracaras to concentrate along highways because of the attraction of roadkills as a food source.

Status and Trends

The breeding range of Audubon's crested caracara in Florida (Fig. 1), based on records from the most recent 5-year period of the study (1987-91), did not differ significantly from that during 1973-76 (Layne 1978). Caracaras were documented in 20 counties in central peninsular Florida, with most locations in the same 5-county area as in the earlier years. Counties with 10% or more of the 183 estimated locations during 1987-91 included (number of locations in parentheses) Glades (41), Highlands (34), Okeechobee (23), and Osceola (18). The data indicate no obvious change has occurred in the overall range or core area of the distribution of the caracara in Florida from that shown by Howell (1932). As there had been relatively little alteration of the natural habitats of the state up to that time, Howell's range map is assumed to reflect the early historical distribution.

The estimated number of adult caracaras during 5-year intervals from 1972 to 1991 ranged from 196 to 312 (Fig. 2). The variation between periods reflects differences in sampling effort rather than changes in actual numbers. Thus, the adult population over the 20-year period appears to have been stable with a minimum of about 300 individuals in 150 territories. Further evidence that the population remained generally stable between 1972 and

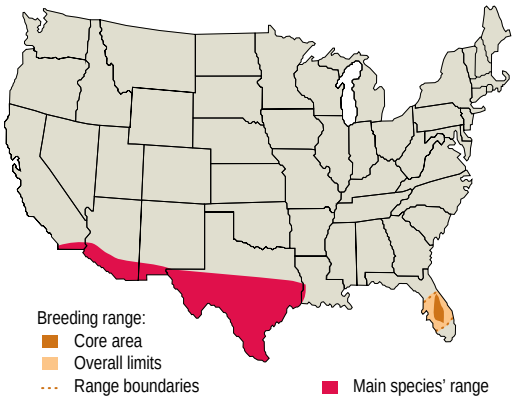

Fig. 1. Breeding range of Audubon's crested caracara in Florida based on records from 1987 to 1991; range boundaries shown by Howell (1932), and main species' range in western United States.

1991 is the similarity in adult-immature age ratios during this interval (Fig. 2). Although immatures could not be censused accurately because they tend to wander individually or in aggregrations after the break up of family groups, they are believed to have numbered between 100 and 200 in any one year, giving a total statewide population of 400-500.

The estimate of the minimum adult population includes single adults observed in an area only once during a 5-year interval as representing a pair on an established territory. Assuming that such individuals were actually unmated transients reduces the estimated adult population to about 230 individuals. Regardless of which estimate of the adult population during 1972-91 is accepted, it is highly unlikely that the Florida population was reduced to fewer than 100 birds between 1967 and 1970 (Heinzman 1970).

Although the range of Audubon's crested caracara in Florida appears to have remained unchanged for the past 60 years and numbers have been stable over at least the past 20 years, the future status of the population is still of concern. Most birds occur on private ranchlands subject to habitat degradation or loss from intensification of agricultural practices or other development. The most immediate threat is large-scale conversion of native range and improved pasture habitats to citrus groves.

A decline in the Florida caracara population within the next 10 years appears likely if citrus conversion and other habitat losses continue at the present rate. Because caracaras are relatively long-lived and strongly attached to their territories, residents may persist in a territory despite unfavorable changes, but may not be replaced by new individuals when they finally leave or die. The result may be a significant time lag before the effects of deleterious habitat changes are reflected in an actual population decline. The magnitude of the time lag in detection of any trend in the Florida distribution and population of Audubon's crested caracara also will depend upon the effectiveness of future monitoring efforts.

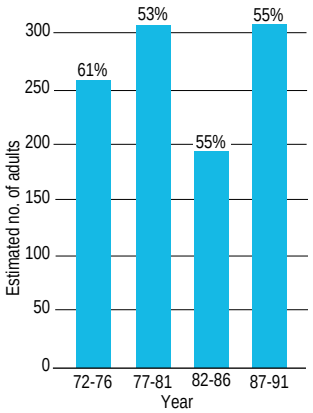

Fig. 2. Estimated numbers of adult Audubon's crested caracaras in Florida over 5-year intervals from 1972 to 1991, based on the assumption that localities where adults were recorded represent territories occupied by an adult pair. Percentage of locations that had immature birds versus those that had adults are given above bars.

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Puerto Rican Parrots

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Since the arrival of Columbus in Puerto Rico, the Taino Indian has disappeared and the parrot has just barely survived (Wadsworth 1949; Snyder et al. 1987). The Puerto Rican parrot (*Amazona vittata*) had shared its habitat with the peaceful Taino Indians for centuries before the arrival of European settlers in the Caribbean.

Status and Trends

Upon arrival of the Spanish in 1493, the Puerto Rican parrot lived in all major habitats of Puerto Rico and the adjacent smaller islands of Culebra, Mona, Vieques, and possibly the Virgin Islands (Snyder et al. 1987). Parrots occupied eight major climax or old-growth forest types (Little and Wadsworth 1964) that covered Puerto Rico and were interspersed only by small, scattered, sandy, or marshy areas near the coast (Snyder et al. 1987). Parrots nested in cavities of large trees that were plentiful throughout the forests. Fertile, moist lowland forests in the coastal plain as well as forested mountain valleys contained much of the fruits and seeds necessary to feed a thriving parrot population. The forests of Puerto Rico probably supported a parrot population of 100,000-1,000,000 at the end of the 15th century (Snyder et al. 1987; Wiley 1991).

Little habitat change occurred in Puerto Rico during the first 150 years of European settlement. By 1650 the Spanish population had increased to 880 (Snyder et al. 1987); parrots still occupied all major habitats and were plentiful (Fig. 1). During the next two centuries the human population soared to almost 500,000 (Fig. 1), and clearing for agriculture, especially in the lowlands, eradicated forests in Puerto Rico (Wadsworth 1949). By 1836 reports by

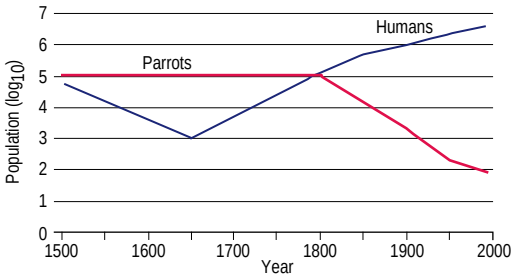

Fig. 1. Population trends of humans and Puerto Rican parrots since 1500 (Snyder et al. 1987 and U.S. Census data; all data for the year 2000 are projected). Populations are converted to $\log_{10}$ for showing trends.

Moritz, a German naturalist, indicated that the Puerto Rican parrot population had begun to decline (Snyder et al. 1987).

By 1900 the human population had doubled to a million (Fig. 1). About 76% of the land area of Puerto Rico had been converted from forest to agriculture (Snyder et al. 1987); less than 1% of the old-growth forest remained after more than 400 years of European civilization. At this time, the parrot population must have been low, but no data exist. By 1937 U.S. Forest Service (USFS) rangers estimated the Puerto Rican parrot population at about 2,000 birds (Wadsworth 1949). A few years later, parrots were found only in the Luquillo Mountains, formerly a forest reserve of the Spanish Crown and now managed by the USFS. This area contained the last forest habitat suitable for Puerto Rican parrots.

Population surveys of the Puerto Rican parrot were not conducted until the 1950's. Early estimates of the parrot population in Puerto Rico are based on few written records and general observations (Snyder et al. 1987), knowledge of the parrot's biology, and extrapolation of population surveys conducted by Rodríguez-Vidal (1959). During the 1950's, Rodríguez-Vidal of the Puerto Rico Department of Agriculture and Commerce conducted the first extensive study of the Puerto Rican parrot. He reported a population of 200 Puerto Rican parrots by the mid-1950's (Fig. 2). About 20 years later the population had dwindled to 14 individuals that inhabited an isolated rain forest of the Luquillo Mountains.

Courtesy J.M. Meyers, NBS

Puerto Rican parrot (*Amazona vittata*).

In 1968 Kepler, U.S. Fish and Wildlife Service (USFWS), organized parrot surveys by placing observers at strategic sites, including overlooks from prominent rocks, road-cuts, and building roofs. Snyder et al. (1987) improved the survey method in 1972 by constructing 10 treetop lookouts in areas of major parrot use. Parrot surveys are conducted from these platforms during the breeding season and pre- and postbreeding season (Snyder et al. 1987). Observers collect information on parrot numbers, directions, and their distance from the platform by the time of day. By 1993 this treetop lookout system was expanded to 38 platforms (Vilella and García 1994).

In 1968 implementation of the Puerto Rican Parrot Recovery Plan began; it is a cooperative effort of scientists and managers of the Puerto Rico Department of Environmental and Natural Resources, USFS (Caribbean National Forest and International Institute of Tropical Forestry), USFWS Puerto Rican Parrot Field Office, and the National Biological Service. After the recovery program began, the parrot population increased to 47 birds by 1989 (Wiley 1980; Lindsey et al. 1989; Meyers et al. 1993); however, about 50% of the population was destroyed by Hurricane Hugo that same year. A small population of 22-24 individuals remained in late 1989 (Fig. 2). Since then, the population recovered to 38-39 by early 1994 (F.J. Vilella, USFWS, personal communication). After the hurricane, the number of successful nesting pairs increased from a maximum of 5 to 6 pairs from 1991 to 1993 (Meyers et al. 1993; Vilella and García 1994).

Research and Management

Puerto Rican parrots declined in relation to the increasing human population (Fig. 1). Conversion of forests to agriculture and loss of forest habitat, on which the species depended for food and nest cavities, was the primary cause for decline. Shooting parrots for food or protection of crops and capture for pets were secondary causes for decline. The remnant parrot population in the Luquillo Mountains was further stressed when trails and roads were created and when human uses of the forest timber were encouraged in the early 1900's (Snyder et al. 1987). Storms before the arrival of Europeans probably had little effect on the parrot population because the population was more widespread, and hurricanes tend to affect only a small geographic area. Severe hurricanes in 1898, 1928, 1932, and 1989 reduced small, now-isolated populations even further. The apparent ability of the population to rebound after these storms is suggested by increases in the parrot population and in nesting pairs after

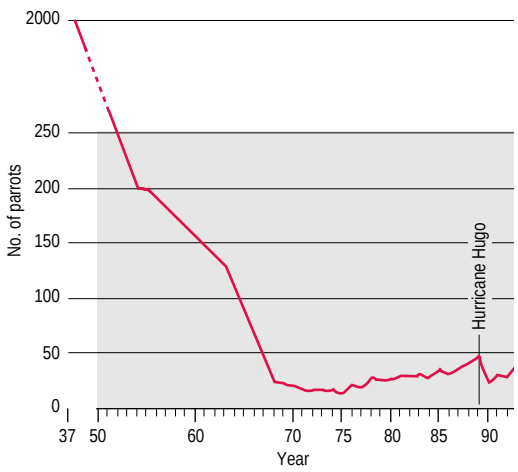

Fig. 2. Population trends of the Puerto Rican parrot in the 20th century.

Hurricane Hugo hit the island in 1989 (Meyers et al. 1993).

Intense research and management strategies during the last 27 years have prevented the extinction of the Puerto Rican parrot. Much of the effort to rebuild the population has involved research and management of nesting sites (Wiley 1980; Snyder et al. 1987; Lindsey et al. 1989; Wiley 1991). Predators, such as black rats (*Rattus rattus*) and pearly-eyed thrashers (*Margarops fuscatus*), have been controlled (Snyder et al. 1987). Bot fly (*Philornis* spp.) infestations of nestlings are still a minor problem (Lindsey et al. 1989). Management of nests by fostering captive-reared young into wild nests, guarding nests, controlling honey bees (*Apis mellifera*), improving and maintaining existing nest cavities, and creating enhanced nesting cavities should increase the population of the Puerto Rican parrot (Wiley 1980; Lindsey et al. 1989; Wiley 1991; Lindsey 1992; Vilella and García 1994).

Hurricanes will continue to threaten the wild population of the Puerto Rican parrot. Researchers estimate that storms equal to the intensity of Hugo (sustained winds of 166 km/h or 104 mi/h) occur at least every 50 years in northeastern Puerto Rico (Scatena and Larsen 1991). The risk of extinction caused by hurricanes will be reduced by establishing a geographically separated wild population (USFWS 1987).

Introduced parrots and parakeets are common in Puerto Rico, including some of the genus *Amazona*. Monitored populations of these non-native birds have increased from 50% to 250% during 1990-93 (J.M. Meyers, National Biological Service, unpublished data). If they expand their ranges to include older forests, these populations may pose a threat to the Puerto Rican parrot by introducing diseases and by competing for resources. At present, none of the introduced *Amazona* populations are found near the Luquillo Mountains; however, orange-fronted parakeets (*Aratinga canicularis*) have foraged and nested in these mountains at lower elevations (J.M. Meyers, NBS, unpublished data).

As the Puerto Rican parrot population increases, it is possible that suitable nesting sites may limit population growth. Before this occurs, research and management should concentrate on increasing the wild population. The ability of the Puerto Rican parrot to expand its population in a manner similar to the exotic parrots in Puerto Rico, in a variety of natural and human-altered environments, should not be underestimated and may be the key to its recovery.

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Red-cockaded Woodpeckers

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The red-cockaded woodpecker (RCW; *Picoides borealis*) is a territorial, nonmigratory, cooperative breeding species (Lennartz et al. 1987). Ecological requirements include habitat for relatively large home ranges (34 to about 200 ha or 84 to about 500 acres; Connor and Rudolph 1991); old pine trees with red-heart disease for nesting and roosting (Jackson and Schardien 1986); and open, parklike forested landscapes for population expansion, dispersal (Connor and Rudolph 1991), and necessary social interactions.

Historically, the southern pine ecosystems, contiguous across large areas and kept open with recurring fire (Christensen 1981), provided ideal conditions for a nearly continuous distribution of RCWs throughout the South. Within this extensive ecosystem red-cockaded woodpeckers were the only species to excavate cavities in living pine trees, thereby providing essential cavities for other cavity-nesting birds and mammals, as well as some reptiles, amphibians, and invertebrates (Kappes 1993). The loss of open pine habitat since European settlement precipitated dramatic declines in the bird's population and led to its being listed as endangered in 1970 (Federal Register 35:16047).

We obtained historic RCW distribution data, arranged by state and county, from published sources (Jackson 1971; Hooper et al. 1980), and interviews with various red-cockaded woodpecker experts. Current distribution and abundance data were obtained from natural resource agencies and knowledgeable biologists. Most records were reported between January 1993 and March 1994, and most represent direct census data. Specific references are available from R. Costa (Table).

Several terms are used to describe red-cockaded woodpecker abundance. "Group" refers to birds that cooperate to rear the young from a single nest. It usually consists of a breeding male and female, and zero to four helpers, usually the group's male offspring from previous breeding seasons. For reporting purposes, single bird groups (usually male) are tallied. The collection of cavity trees used by a group for nesting and roosting is the "cluster." Although single tree clusters do occur, typically each cluster consists of 2 to more than 15 cavity trees and may occupy 2 to more than 4 ha (5 to more than 10 acres). Each group normally occupies and defends only one cluster. "Population" refers to the aggregation of groups that are more distant than 29 km (18 mi) from the nearest group. A single isolated group may constitute a population.

Courtesy J. Hanula and K. Franzreb, USFS

Red-cockaded woodpecker (*Picoides borealis*).

Table. Number of red-cockaded woodpecker active clusters, by state and land ownership category, for various years between 1990-94.*

State	Ownership			
	Federal	State	Private	Total
Alabama	150	8	25	183
Arkansas	35	0	121	156
Florida	1,063	128	94	1,285
Georgia	431	2	218	651
Kentucky	5	0	0	5
Louisiana	422	10	73	505
Mississippi	152	0	22	174
North Carolina	408	162	163	733
Oklahoma	0	9	1	10
South Carolina	456	39	186	681
Tennessee	1	0	0	1
Texas	218	26	61	305
Virginia	0	0	5	5
Total	3,341	384	969	4,694

*For information on references, contact R. Costa.

Historical Distribution and Abundance

The historical range of this species covered southeast Virginia to east Texas and north to portions of Tennessee, Kentucky, southeast Missouri, and eastern Oklahoma (Figure). The range included the entire longleaf pine ecosystem, but the birds also inhabited open shortleaf, loblolly, and Virginia pine forests, especially in the Ozark-Ouachita Highlands and the southern tip of the Appalachian Highlands.

Red-cockaded woodpecker abundance was described variously as fairly common (Woodruff 1907), locally common (Murphey 1939), common (Chapman 1895), or abundant (Audubon 1839). Occasional occurrences were noted for New Jersey (Hausman 1928), Pennsylvania (Gentry 1877), Maryland (Meanly 1943), and Ohio (Dawson and Jones 1903).

The distribution map (Figure) displays only counties for which specimens or reliable sources can be cited. The gaps in the distribution undoubtedly contained red-cockaded woodpeckers in the past. Most counties without documented occurrences are found in the longleaf pine-shortleaf pine-loblolly pine-hardwoods transition areas in the east gulf region (Figure), where richer soils and rolling topographies were associated with intense agriculture and interrupted fire regimes. Such areas possibly supported smaller populations that were quickly lost with the forest clearing and therefore were never recorded.

Status and Causes of Decline

Red-cockaded woodpeckers survive as very small (1-5 groups) to large (groups of 200 or more) populations. There are at least small populations in most states with historical occurrences (Table). Except for a population of about 90 groups in southern Arkansas and northern Louisiana, the largest populations are found within the historical longleaf pine ecosystem. Other populations outside the longleaf pine range consist of fewer than 20 groups in single or several adjacent counties. Within the longleaf range, there are 4 populations with more than 200 groups and 11 populations with more than 100 groups; all but one are found on federal lands. The remaining longleaf pine-associated populations are small and isolated. Such small populations are threatened by adverse effects of demographic isolation, increased predation and cavity competition, and stochastic (random) natural events such as hurricanes.

The decline of the red-cockaded woodpecker coincided with the loss of the longleaf ecosystem. As forests were cleared, birds were isolated in forest tracts where unmerchantable trees were left. Aerial and ground photographs from the 1930's show that scattered medium to large trees (0.4-2 per ha or 1-5 per acre) were left in many stands. The culled trees (undoubtedly including red-cockaded woodpecker cavity trees) provided residual nesting and foraging habitat for the birds. In some places these trees remain and are used by red-cockaded woodpeckers today.

Since the 1950's, on lands managed for forest products, the forest structure and composition changed in conjunction with clearcutting, short timber rotations, conversion of longleaf stands to other pine species, and "clean" forestry practices (removal of cavity, diseased, or defective trees). These practices eliminated much of the remaining red-cockaded woodpecker habitat. Additionally, aggressive fire suppression promoted the development of a hardwood midstory in pine forests. The adverse impacts of a dense midstory on RCW populations are well-documented (Connor and Rudolph 1989; Costa and Escano 1989).

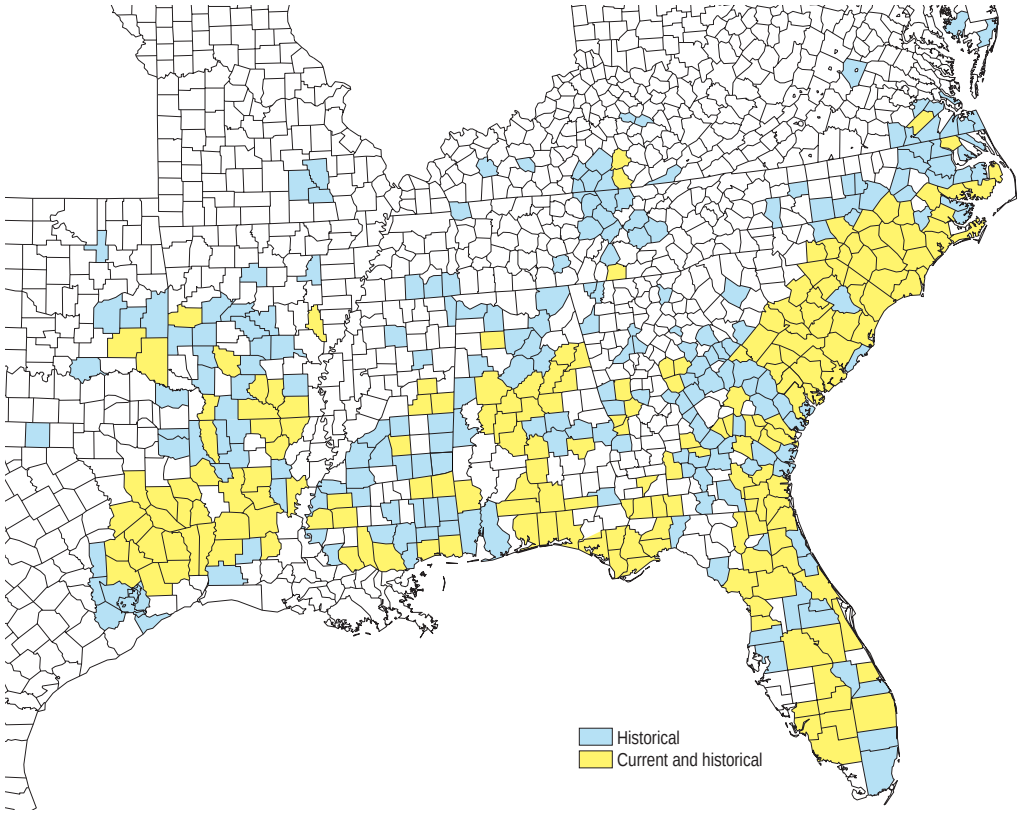

Figure. Distribution of red-cockaded woodpeckers by county and state. Most historical RCW records are cited from Jackson 1971 and Hooper et al. 1980. For information on references, contact R. Costa.

Recent Developments and the Future

The Red-cockaded Woodpecker Recovery Plan (USFWS 1985) specifies that rangewide recovery will be achieved when 15 viable populations are established and protected by adequate habitat management programs. The recovery populations are to be distributed across the major physiographic provinces and within the major forest types that can be managed to sustain viable populations. Each recovery population will likely require 400 breeding pairs (or 500 active clusters, as some clusters are occupied by single birds or contain nonbreeding groups) to ensure long-term population viability (Reed et al. 1993; Stevens, in press). At a density of 1 group/80-120 ha (200-300 acres; USFWS 1985; USFS 1993), landscapes of at least 40,000 ha (100,000 acres) will be needed to support viable populations. Most forested pine areas large enough to supply this habitat are on public, mostly federal, lands.

With two exceptions (Hooper et al. 1991; USFS, Apalachicola National Forest, FL, unpublished data), there is no evidence that red-cockaded woodpecker populations can expand to viable levels without considerable human intervention. Conversely, numerous population extirpations have been documented (Baker 1983; Costa and Escano 1989; Cox and Baker, in press). Ensuring the survival of the species, even in the short term (50 years), will require landscape-scale habitat and population management to provide the forest structure and composition needed for nesting and foraging habitat and population expansion; and to manage limiting factors (primarily a lack of suitable cavity trees, cavity competition, and demographic isolation) that can extirpate small populations. Both strategies are part of management guidelines drafted by several federal land stewards (USFS 1993; U.S. Army 1994; USFWS 1994).

These ecosystem management plans promote practices that minimize landscape fragmentation, retain suitable numbers of potential cavity trees well distributed throughout the landscape, and restore the original forest cover by planting the appropriate pine species. They recommend the use of growing-season fires to control hardwoods, create open forest conditions, and begin to restore the understory plant communities of the pine ecosystems. Stabilization and growth of small high-risk populations will be aided by creating artificial red-cockaded woodpecker cavities (Copeyon 1990) and translocating juvenile birds from stable larger populations into small ones (Rudolph et al. 1992). Technologies that minimize or eliminate predation and competition problems are available (Carter et al. 1989).

During the past 4-7 years, several populations have stabilized or increased (Gaines et al., in press; Richardson and Stockie, in press) as a result of implementing conservation biology principles—that is, integrating available technology with the species' life history and ecological requirements. The limited number of juvenile birds, however, may hinder recovery progress in all populations simultaneously.

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Southwestern Willow Flycatchers in the Grand Canyon

by
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The southwestern willow flycatcher (*Empidonax traillii extimus*) occurs, as its name implies, throughout most of the southwestern United States (Fig. 1). It is a Neotropical migrant songbird, i.e., one of many birds that return to the United States and Canada to breed each spring after migrating south to the Neotropics (Mexico and Central America) to winter in milder climates. In recent years, there has been strong evidence of declines in many Neotropical migrant songbirds (e.g., Finch and Stangel 1993), including the southwestern willow flycatcher (Federal Register 1993). The flycatcher appears to have suffered significant declines throughout its range, including total loss from some areas where it historically occurred. These declines, as well as the potential for continued and additional threats, prompted the U.S. Fish and Wildlife Service (USFWS) to propose listing the southwestern willow flycatcher as an endangered species (Federal Register 1993).

The southwestern willow flycatcher is one of four distinct races of willow flycatchers that breed in North America. All races breed in shrubby or woodland habitats, usually adjacent to, or near, surface water or saturated soil. Riparian areas—woodland and shrub areas along streams and rivers—are particularly favored. In fact, the southwestern willow flycatcher is a riparian obligate, breeding only in riparian vegetation. It prefers tall, dense willows and cottonwood habitat where dense vegetation continues from ground level to the tree canopy. Southwestern willow flycatchers appear to breed in stands of the exotic and invasive tamarisk (*Tamarix* spp.) only at locations above 625 m (2,051 ft) elevation (Federal Register 1993), and where the tamarisk stands have suitable structural characteristics (Fig. 2). Thus, many areas dominated by tamarisk are not suitable flycatcher habitat. Being a riparian obligate, the southwestern willow flycatcher is particularly sensitive to the alteration and loss of riparian habitat (including tamarisk invasion), which is a widespread and pervasive problem throughout the Southwest.

Because of the decline and precarious status of southwestern willow flycatchers, it is important to document the status of the species, where it occurs, how many individuals are present, and where they are successfully breeding. Information on trends is also important in managing and protecting the species. Grand Canyon

Fig. 1. Breeding distribution of the southwestern willow flycatcher. Dotted line represents areas where distribution is uncertain.

Courtesy M.K. Sogge, NBS

Fig. 2. Southwestern willow flycatcher breeding territory in tamarisk habitat along the Colorado River in the Grand Canyon.

National Park, the USFWS, and the U.S. Bureau of Reclamation have been regularly monitoring the status of the southwestern willow flycatcher in the Grand Canyon since 1982. The National Biological Service's Colorado Plateau Research Station at Northern Arizona University has conducted this monitoring since 1992. The Grand Canyon is one of the few areas with such a long record of willow flycatcher population data; the only others are the Santa Margarita and Kern rivers in southern California.

Methods

Our monitoring program involved intensive surveys of about 450 km (280 mi) of the Colorado River in Arizona between Glen Canyon Dam (Lake Powell) and upper Lake Mead. This portion of the river flows from elevation 945 m (3,100 ft) at the dam to 365 m (1,200 ft) at Lake Mead. We walked through or floated along all potential southwestern willow flycatcher habitat patches along the river corridor and looked and listened for willow flycatchers. Although willow flycatchers look very similar to several other flycatchers, they can be readily identified by their distinctive "fitz-bew" song. To increase the chance of detecting resident flycatchers, we played a tape recording of willow flycatcher songs and calls (Fig. 3) as we moved through our survey areas. This technique usually elicits a response from any resident southwestern willow flycatchers that may be present (Tibbitts et al. 1994). We conducted surveys from May through July at about 160 habitat patches each year (1992 and 1993), and made repeated trips to each site (Sogge et al. 1993).

Courtesy M.K. Sogge, NBS

Fig. 3. Surveyor broadcasting taped vocalizations and looking for response from willow flycatchers.

Status and Trends

Surveys conducted between 1982 and 1991 looked only at the upper 114 km (71 mi) of the river and counted primarily singing males. Within this same stretch, we detected only two singing male willow flycatchers in 1992, and three in 1993. These willow flycatchers were found only in the dense riparian habitat dominated by tamarisk, but including some willows along the river corridor above 860 m (2,800 ft) elevation. The breeding population of southwestern willow flycatchers in the Grand Canyon was very low: we found only one nest in 1992, and only three in 1993. Worse yet, each of the three 1993 willow flycatcher nests was brood-parasitized by brown-headed cowbirds (*Molothrus ater*), and none produced young willow flycatchers. With such a small breeding population, and the potential for severe loss of breeding effort due to cowbirds, there is concern over the continued survival of the species within Grand Canyon.

Based on comparison with past willow flycatcher surveys in the Grand Canyon (river mi 0-71; Brown 1988, 1991), willow flycatchers have declined since the mid-1980's (Fig. 4). Because we could conduct more surveys and our methods were more likely to detect flycatchers than the pre-1992 surveys (conducted without using tape playback), the population decline of the southwestern willow flycatcher in Grand Canyon may be even more dramatic than our data indicate.

We did find willow flycatchers in areas of the river corridor where surveys had not been previously conducted: three in 1992 and five in 1993. Two other willow flycatchers were also found during separate bird studies on the river corridor. These birds were found in tamarisk (above 530 m; 1,900 ft) or willow (below 530 m; 1,900 ft) habitats. None of these willow flycatchers established territories or bred, however, and most were probably migrants simply passing through the area (Sogge et al. 1993).

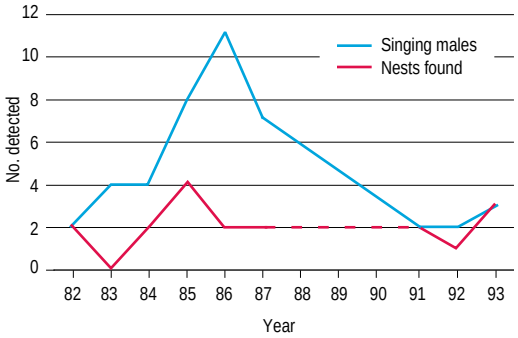

Fig. 4. The numbers of singing male southwestern willow flycatchers and flycatcher nests detected in the Grand Canyon (river mi 0 to 71), 1982-93. Dotted lines represent years when surveys were not conducted.

The low breeding population, historical declines, and potentially limited productivity in the Grand Canyon reflect the plight of the southwestern willow flycatcher throughout its range. Declines have been noted virtually everywhere the flycatcher occurs, and threats to its survival are widespread and immediate. As human activities such as urbanization, water diversion, agriculture, and grazing in riparian areas continue in the Southwest, so do the loss and alteration of riparian habitat. Vital wintering habitat in Mexico and Central America is also being lost to similar human activities.

Brood parasitism by brown-headed cowbirds is another significant threat to southwestern willow flycatchers within the Grand Canyon and in many other areas. In fact, cowbirds may be one of the greatest threats in areas where breeding habitat is protected, such as the Grand Canyon and other national parks and protected areas. Cowbirds lay their eggs in the nests of other birds (the host), who subsequently abandon the nests or raise the cowbird chicks. Female cowbirds will sometimes remove or destroy host eggs, and cowbird chicks often monopolize the parental care of the hosts. Thus, cowbird parasitism can reduce the number of host young produced, and in some cases, cowbirds may be the only young successfully raised by the host. Such effects have been recorded for southwestern willow flycatchers in the Grand Canyon and in other areas as well (Federal Register 1993). Conversely, control and removal of cowbirds have resulted in local increases in southwestern willow flycatchers and other songbirds. Cowbird brood parasitism is related to riparian loss and fragmentation because cowbird parasitism is highest in fragmented habitats.

The southwestern willow flycatcher is a unique and valuable part of the riparian community in the Southwest. Although recent and planned future surveys provide important status and distributional information on the flycatcher in the Grand Canyon and a few other areas within Arizona, there is a critical need for basic surveys and ecological research (including the effect of brown-headed cowbirds) on this species throughout most of its range, particularly in New Mexico, southern Utah, and Colorado. As a riparian obligate species whose continued existence is directly tied to the future of our remaining riparian habitats, its precarious status and historic decline help illustrate the need for riparian preservation and management. Such management is important not only for the southwestern willow flycatcher, but also for all plant and animal species that make up and depend on these valuable riparian areas.

Courtesy I.C. Taft

Southwestern flycatcher (*Empidonax traillii extimus*).

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