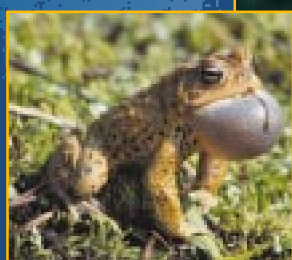
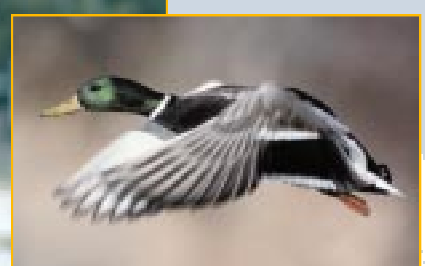
# Wildlife Resource Trends in the United States

Curtis H. Flather, Stephen J. Brady, and Michael S. Knowles





A Technical Document Supporting the 2000 USDA Forest Service RPA Assessment

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## Abstract

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This report documents trends in wildlife resources for the nation as required by the Renewable Resources Planning Act (RPA) of 1974. The report focuses on recent historical trends in wildlife as one indicator of ecosystem health across the United States and updates wildlife trends presented in previous RPA Assessments. The report also shows short- and long-term projections of some wildlife for documenting expected trajectories of resource change. National trends in four attributes of wildlife resources, including habitat, population, harvest, and users, set the context within which region-specific trends are presented. The data for this analysis came largely from information that currently exists within Forest Service and cooperating state and federal agency inventories. The report concludes with a synthesis of these trends as they relate to the concept of resource health. We highlight those trends that appear to indicate favorable, uncertain, or degraded resource conditions in an attempt to identify resource situations that warrant policy and management attention.



**Keywords:** wildlife resources, resource planning, national assessment, trends, wildlife populations, wildlife habitats, wildlife harvests, recreation participation, bird diversity, threatened and endangered species


## The Authors


**Curtis H. Flather** is a research wildlife biologist at the Rocky Mountain Research Station in Fort Collins, CO.

**Stephen J. Brady** is a wildlife biologist with the Natural Resources Conservation Service in Fort Collins, CO.

**Michael S. Knowles** is an information systems analyst with the Management Assistance Company of America at the Rocky Mountain Research Station in Fort Collins, CO.



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Cover photo of mule deer by Lane Eskew.

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## Contents

# Introduction

Over the last 60 years of "modern" wildlife management in the United States, researchers have learned much about how animals affect the structure and function of ecosystems (see Pimentel and others 1997). Animals pollinate, help germinate, and disperse many plant species; play important roles in nutrient flux and cycling; and help break down pollutants and organic wastes. They also provide recreational, commercial, and subsistence needs; contribute to spiritual and intellectual stimulation; and maintain biological diversity through interactions with their habitat (e.g., herbivory) and among species (e.g., predation). Despite a widespread awareness of wildlife's role in ecosystems, the magnitude of the ecologic and economic benefits attributable to wildlife is largely unrecognized by the public (Daily 1997).

Failure to recognize these benefits can lead to economic development that degrades ecosystems. Such development, in conjunction with increasing human populations and land use intensification, can stress ecosystems to a point where their ability to provide the aforementioned benefits is compromised (Rapport and others 1985; Pimentel and others 1997). Concern that the human enterprise may jeopardize the viability of ecosystems (Holdgate 1994; Vitousek and others 1997) is at the crux of a natural resource management shift that is now focusing on long-term sustainability of ecosystems (Lubchenco and others 1991; Christensen and others 1996).

Sustainability has been defined as "... the ability to maintain something undiminished over some time period" (Lélé and Norgaard 1996:355). Applied to natural resources, this definition implies that: (1) management will not degrade those systems being utilized (Lubchenco and others 1991), and (2) the current generation of humans will leave an equitable share of resources for future generations (Meyer and Helfman 1993). Although formal consideration of sustainability as a framework to guide natural resource management has received much discussion in the last decade, its roots actually extend back to classical economic theories proposed during the latter half of the 19th century (van den Bergh and van der Straaten 1994). As reviewed by Goodland (1995), the writings of T.R. Malthus and J.S. Mill during this time period emphasized concerns about exponential human population growth, a finite natural resource base, and the need for environmental protection to preserve human welfare. Natural resource management has had a similarly long history of advancing sustainability concepts: Sustainable forest management dates to the mid-1800s, maximum sustained yield was the accepted management paradigm for fisheries by the 1950s, and wildlife managers used the concept of reproductive surplus to recommend sustainable harvest strategies in the 1940s (Hilborn and others 1995). Given this history, it should not be surprising to see elements of the sustainable development philosophy incorporated into more recent legislation that guides federal natural resource policy and management. The Forest and Rangeland Renewable Resources Planning Act (RPA) of 1974, as amended by the National Forest Management Act (NFMA) of 1976, is one example of legislation that mandates elements consistent with a sustainable resource management perspective.

Our ability to manage ecosystems in a sustainable manner requires an analysis of the system's performance and health over broad spatial and temporal scales (Constanza 1992, Holling and Meffe 1996). The RPA directs the Forest Service to prepare such broad-scaled evaluations in the form of periodic national assessments of natural resources on the nation's 1.6 billion acres of forest and range lands. These assessments are to report on (1) the current status and condition of resources based on an analysis of recent historical trends and (2) the future resource situation based on trend projections (Cortner and Schweitzer 1981). Although these resource assessments were initially interpreted in the context of resource supplies and public demands (Hoekstra and others 1979), they now offer an opportunity to examine resource health in the context of sustainability. Sustainability is a broad concept that encompasses many aspects of human development, resource management, and ecological response. Therefore, sustainability is difficult to monitor directly without specificity. Assessments of sustainability have been couched in the concept of indicators (Hammond and others 1995), which ostensibly represent key measurable attributes that reflect ecosystem properties that are too difficult or costly to monitor directly (Noss 1990).

The primary objective of this report is to present recent historical and expected trends in wildlife resources across the United States as one set of ecosystem health indicators. Secondarily, this report also updates wildlife resource trends that have been presented as components of previous RPA Assessments (see, Flather and Hoekstra 1989; USDA Forest Service 1981). For the purposes of this report, wildlife are defined as free-ranging vertebrate and invertebrate taxa that inhabit primarily terrestrial ecosystems. Trends associated with aquatic species and their implications to aquatic system health are covered in a companion report (see Loftus and Flather [In press]). Although our definition of wildlife extends beyond the traditional game bird and mammal focus, available data on recent historical trends do have taxonomic biases that can be traced to the recreational and commercial importance of some species. Such biases notwithstanding, our intent is to discuss trends in wildlife resources across a broad taxonomic set.[1]

[1] *The scientific name for species mentioned in this Assessment can be found in Appendix A.*

We review trends in four attributes of wildlife resources: habitats, populations, harvests, and users. National-level summaries of emerging trends in these attributes set the context within which region-specific trends are presented and discussed. Each region constitutes a multi-state assessment area defined by the Forest Service for strategic planning purposes and includes the North, South, Rocky Mountain, and Pacific Coast Assessment regions (figure 1). In some instances, data on wildlife resources have been reported for regions that could not be reaggregated to conform to the Assessment region boundaries defined for this report (e.g., waterfowl flyways). Consequently, we also present wildlife resource trends for other geographic regions when they are well established in wildlife planning.

The time period over which we review wildlife trends varies among wildlife resource attributes and species. At a minimum, we have attempted to document those trends since the last RPA Assessment report (see Flather and Hoekstra 1989). In most cases, however, the data permitted us to examine trends over the last 20 to 30 years. Our review of the expected future condition of wildlife resource attributes was limited primarily to short-term (to the year 2000) and long-term (to the year 2045) projections provided by state wildlife agency personnel under an assumption that current resource management strategies would continue into the future.

The data that supported our examination of recent and expected trends in wildlife resources came largely from information that currently exists within Forest Service and cooperating state and federal agency inventories. In general, the data reviewed here were not collected specifically to support Forest Service RPA Assessments of wildlife resources. Although there are specific cases of standardized inventories that lend themselves to RPA Assessments, there is a dearth of nationally consistent environmental data from which to evaluate the ecological condition of the United States in general (Brown and Roughgarden 1990), and of wildlife resources in particular (Thomas 1990). Therefore, the extent to which we were able to discuss habitat, population, harvest, and user trends depended on the availability of information from diverse sources.

This report is organized into three major sections. We first review trends in broad classes of wildlife habitat, which provide an indication of land resources available to

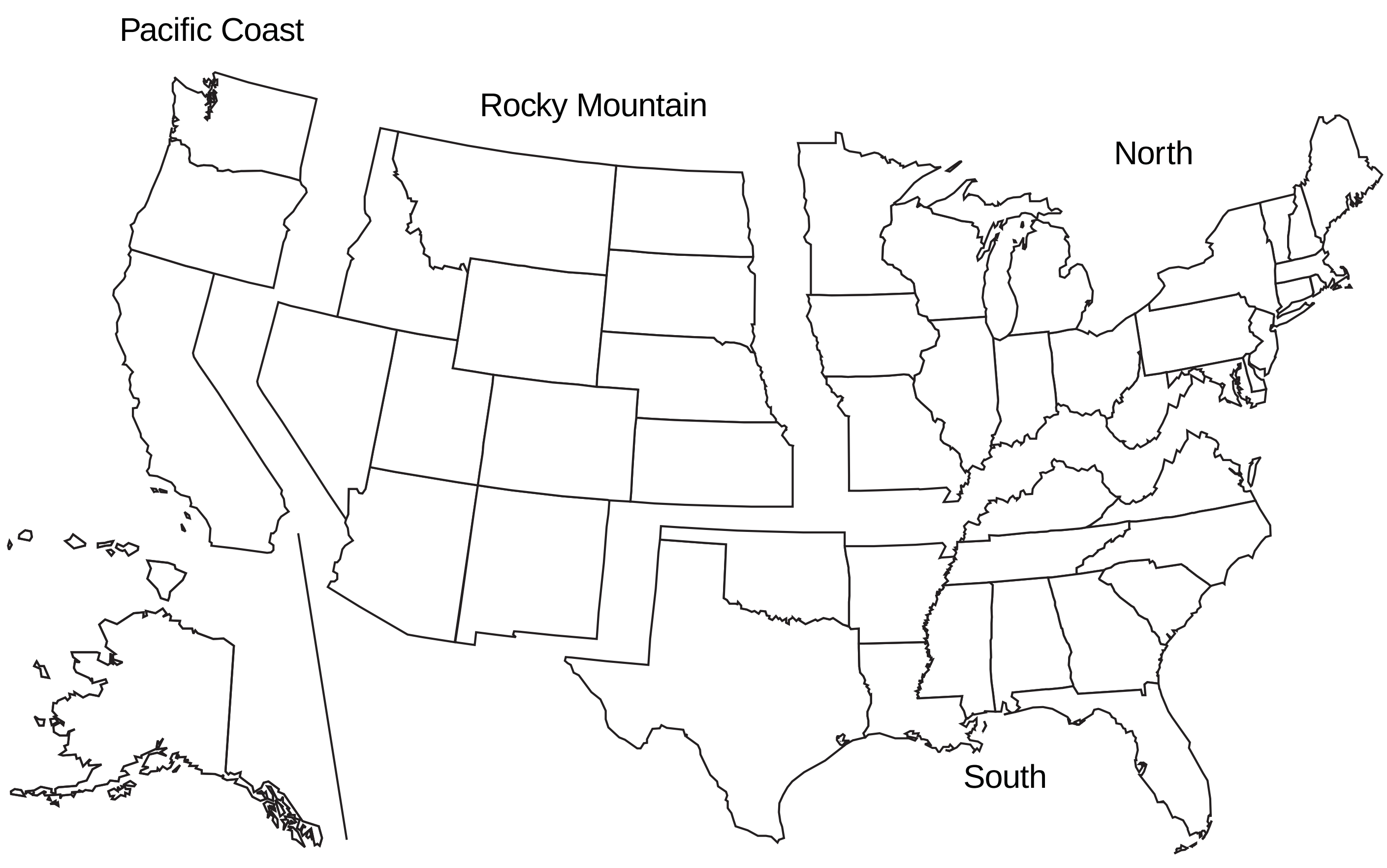


Figure 1. Forest Service RPA Assessment regions.

wildlife. Second, trends in populations and harvests are reviewed as indicators of species abundance and levels of resource extraction. This section also reviews distributional patterns of threatened and endangered species as a means of identifying those regions of the country where imperiled species are concentrated. Third, we present participation trends in wildlife-dependent recreation activities and trapping. We conclude by highlighting those trends that appear to indicate favorable, uncertain, or unfavorable resource conditions as a means of identifying those resource situations that could benefit from resource policy and land management attention.

# Habitat Trends

Wildlife habitat is defined as the biotic and abiotic attributes of a particular environment that, in total, make it possible for a specified species to inhabit that environment (Morrison and others 1998). Habitat is often delineated geographically as an area within which the mix of critical resources occur that provide for a species' survival and reproduction. Because habitat is species specific, and can vary regionally, there are very few national inventories of wildlife habitat per se. Even if such inventories existed, it would be difficult to gain a composite perspective of wildlife habitat condition by integrating habitat characteristics across species. For this reason, we have chosen to discuss recent trends in wildlife habitat using a broader definition of habitat—one that has its basis in a landscape-ecologic perspective.

Block and Brennan (1993) define macrohabitat as landscape features that are correlated with the distribution and abundance of species. Land use and land cover classifications are two important landscape features that can define macrohabitat classes, which can then be used to infer the expected species composition for a particular region. Using this approach, land use and land cover statistics over time can provide a basis for anticipating the response of species groups to changes in the amount and management of various land classes. Here we address the status of wildlife habitat by first examining general trends in land use and land cover. This is followed by a specific discussion of trends for forest, range, agriculture, and wetland habitats.

## General Land Use and Land Cover Trends

The distribution and abundance of wildlife is fundamentally affected by landscape structure, which is, in turn, affected by vegetation cover and the manner in which land is used by humans (Forman 1995, Janetos 1997). Human land use is the primary force driving changes in biological diversity (Vitousek and others 1997). Therefore, we examine recent trends in land use before reviewing changes in species populations.

At the spatial and temporal scales of this Assessment, we did not expect changes among major land use and land cover categories to be extensive because (1) considerable land transformations occurred before the availability of comprehensive land area statistics and (2) available statistics tend to report only the net change over time, which masks the dynamic nature of gross land area shifts that actually occur among individual land types. Statistics for land area change of major land uses compiled by the USDA Economic Research Service (1997) are consistent with this expectation (figure 2*a*). Area shifts in land

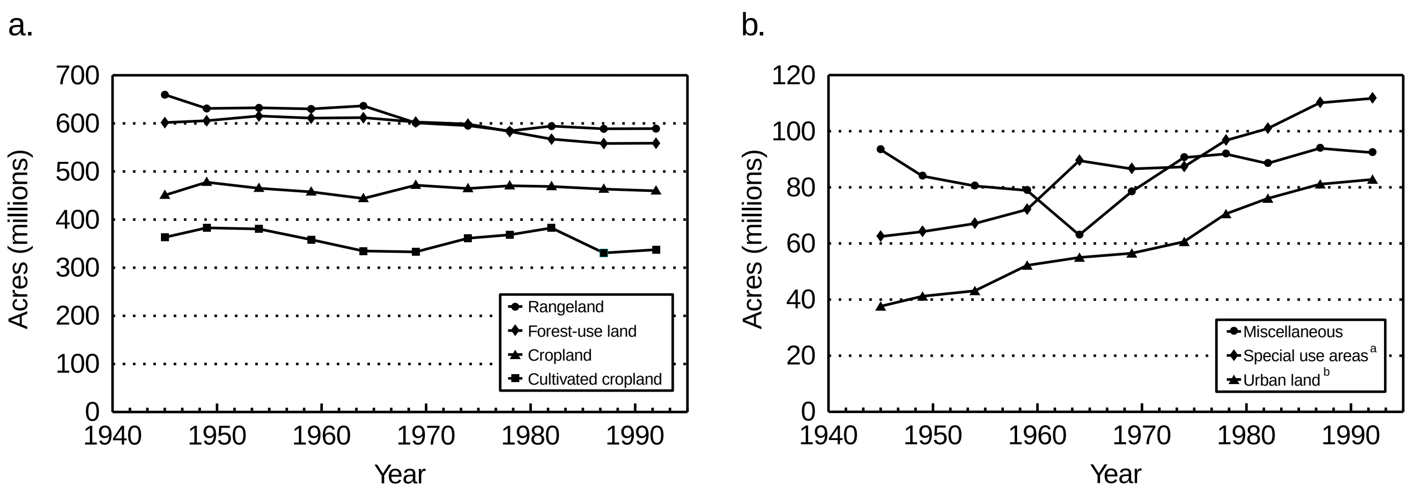


Figure 2. Trends in major (a) and minor (b) land uses for the conterminous United States (USDA Economic Research Service 1997). [a]Includes land in parks, wilderness areas, wildlife areas, national defense and industrial areas, and miscellaneous farmland uses. [b]Includes land in highways, roads, and railroads.

use from 1945 to 1992 have been less than 11% for the three major land uses that comprise 85% of the land area in the conterminous United States—namely forest land, rangeland, and cropland.

Forest land use, which comprises about 30% of the land base, has declined since 1945. Although land devoted to forest-use increased through the mid-1950s, it declined at an average annual rate of 4 million acres/year from the mid-1960s to 1987. This shift from forest to other land uses, however, must be interpreted cautiously. The conversion of land from forest-use does not necessarily reflect a conversion from forest cover. Much of the loss of forest-use land actually reflects a greater percentage of forests being classified as special use areas (e.g., parks, wilderness, wildlife areas) rather than an actual reduction of land with forest cover. It was not possible to determine the changes over time of land with forest cover using the USDA Economic Research Service (1997) data.

Rangeland comprises the largest area among major land uses, accounting for nearly a third of the total land base in the lower 48. Since 1945, rangeland has declined by nearly 11%, with the greatest decline occurring in the late 1960s. Since the late 1960s, rangeland area has fluctuated from a high of 601 million acres in 1969 to a low of 584.3 million acres in 1978.

Cropland, which comprises about one quarter of the land area in the lower 48 states, has shown essentially no change since 1945—a trend that belies the dynamic nature of cultivated land. The area of cropland that is cultivated for growing crops has actually declined by 7% over the 1945-1992 period. In addition, cultivated cropland has fluctuated over time due to the enrollment of land in various Federal programs that retire acres from production for specified periods of time.

Land use categories that comprise a smaller proportion of the land base show much greater relative acreage changes over time when compared to the major land use categories (figure 2*b*). This is particularly apparent with urban land uses (including transportation lands) that have increased consistently from 37.6 million acres in 1945 to nearly 83 million acres in 1992. This represents a 120% increase, although urban land comprised only about 4% of the land base in 1992. Other special use lands, which includes Federal and State parks, wilderness areas, wildlife areas, national defense areas, and miscellaneous farmland uses, also changed greatly over the 1945-1992 period relative to the other land use categories. The nearly 80% gain in this category (from 62.5 million acres in 1945 to 111.7 million acres in 1992) reflects the designation of lands for conservation and recreational purposes.

Recent trends in land use and land cover based on data from the 1992 National Resources Inventory (NRI; USDA Natural Resources Conservation Service 1994) are consistent with the previously reviewed long-term trends. The NRI is a periodic (5-year interval) inventory that monitors land use and land cover on nonfederal lands. We used the NRI to examine land use and land cover trends by Assessment region from 1982 to 1992. A detailed accounting of land use and land cover dynamics from 1982 to 1992 for the nation and by RPA region is provided in Appendix B.

Since 1982, cropland area has declined by 11%, with the loss being concentrated in the South and Rocky Mountain regions (table 1). This loss can be traced directly to the

Table 1--Recent (1982-1992) land area changes for major land uses on non-federal lands (USDA Natural Resources Conservation Service 1994).

| | Cropland (cultivated) | | | Cropland (uncultivated) | | | Pastureland | | | Rangeland | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Region | 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) |
| | thousand acres | | | | | | | | | | | |
| North | 130,082.2 | 121,081.8 | -9,000.4 (-6.9) | 24,159.4 | 24,646.3 | 486.9 (2.0) | 45,686.1 | 39,640.1 | -6,046.0 (-13.2) | 169.1 | 126.4 | -42.7 (-25.3) |
| South | 98,687.1 | 81,929.8 | -16,757.3 (-17.0) | 7,960.2 | 8,948.6 | 988.4 (12.4) | 64,944.8 | 65,356.1 | 411.3 (0.6) | 115,411.1 | 112,135.1 | -3,276.0 (-2.8) |
| Rocky Mtn. | 119,727.6 | 107,298.5 | -12,429.1 (-10.4) | 16,961.0 | 17,198.4 | 237.4 (1.4) | 15,682.1 | 15,717.5 | 35.4 (0.2) | 259,186.8 | 253,626.5 | -5,560.3 (-2.1) |
| Pacific Coast[a] | 17,352.2 | 14,869.3 | -2,482.9 (-14.3) | 5,622.6 | 5,977.4 | 354.8 (6.3) | 4,783.8 | 4,501.1 | -282.7 (-5.9) | 33,973.7 | 32,915.4 | -1,058.3 (-3.1) |
| Total U.S. | 365,849.1 | 325,179.4 | -40,669.7 (-11.1) | 54,703.2 | 56,770.7 | 2,067.5 (3.8) | 131,096.8 | 125,214.8 | -5,882.0 (-4.5) | 408,740.7 | 398,803.4 | -9,937.3 (-2.4) |

[a]Does not include Alaska

[b]Although the NRI is an inventory of nonfederal lands, it does report acres in federal ownership to maintain a complete accounting of land area in the United States. Land use and land cover characteristics are not inventoried on federally owned land.

Conservation Reserve Program (CRP), which enabled farmers to voluntarily retire highly erodible or environmentally sensitive cropland from production and convert it to perennial vegetation in exchange for annual rent payments. Although the gains in uncultivated cropland are consistent with the CRP retirements, the increased acreage in this land use class is due mostly to increases in hayland acreages and does not account for the CRP enrollment. Because of the long-term nature of CRP retirement contracts (10-15 years), the NRI chose to classify CRP lands as "other land" and the 35 million acres gained in this land use category is directly attributable to the CRP. Pastureland area also declined from 1982 to 1992 nationwide. Although there were minor gains in pastureland in the South and Rocky Mountains, the North lost more than 13% of its 1982 pastureland base.

Rangeland area declined in all Assessment regions since 1982 (table 1). We make a distinction between rangelands and patureland in that rangelands typically consist of native plant communities managed by the timing and intensity of grazing, but generally without agronomic management practices applied such as cultivation, planting, and fertilization. The greatest acreage declines occurred in the South and Rocky Mountains, which lost more than 3.2 and 5.5 million acres, respectively. The greatest proportional declines occurred in those regions with the fewest rangeland acres—declining by 3% in the Pacific Coast and by more than 25% in the North. Although rangeland habitats are rare in the North, this trend coupled with trends in pastureland appear to stem from natural successional processes in the absence of new disturbance (e.g., farm abandonment, fire).

Forest land changes varied among Assessment regions (table 1). Forest habitats increased by about 1.3 million acres in the North and by more than 410,000 acres in the South during the 1982-92 decade. Conversely, forest land area declined in the West, but by less than 2% in both the Rocky Mountain and Pacific Coast regions. Viewed nationally, these forest land dynamics negated each other, resulting in no net change of total forest area on nonfederal lands.

Urban land uses, including transportation networks, displayed consistent and substantial gains (greater than 10%) in all regions (table 1). The South witnessed notable urban growth, gaining nearly 7 million acres (a 24% increase) over the decade. The area covered by water showed increases in the North, South, and Rocky Mountains, with small declines in the Pacific Coast. Changes in water area are due to reservoir construction and to wet and dry climate cycles.

Finally, the amount of land in Federal ownership actually increased over this 10-year period (less than 1% nationwide), most notably in the North and South (table 1). Although the NRI is an inventory of nonfederal land, it does report land ownership changes involving federal lands. This is necessary to have a complete accounting of land area changes. However, once land is owned by the federal government the NRI no longer tracks land use and land cover characteristics on those acres.

## Forest Habitats

Forests in the United States are extensive and diverse with nearly a third of the landbase supporting forest

Table 1--(Cont'd.)

| Forest | | | Urban | | | Water | | | Other | | | Federal[b] | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) | 1982 | 1992 | Change (%) |
| thousand acres | | | | | | | | | | | | | | |
| 146,810.9 | 148,159.0 | 1,348.1 (0.9) | 31,177.2 | 35,235.2 | 4,058.0 (13.0) | 14,572.5 | 14,736.5 | 164.0 (1.1) | 17,214.4 | 25,849.3 | 8,634.9 (50.2) | 15,424.5 | 15,821.7 | 3,972.0 (2.6) |
| 176,921.1 | 177,334.4 | 413.3 (0.2) | 28,745.7 | 35,692.6 | 946.9 (24.2) | 20,417.9 | 21,176.9 | 759.0 (3.7) | 13,753.9 | 22,985.2 | 9,231.3 (67.1) | 4,690.7 | 25,973.8 | 1,283.1 (5.2) |
| 28,837.8 | 28,281.4 | -556.4 (-1.9) | 11,298.6 | 12,871.3 | 1,572.7 (13.9) | 9,031.2 | 9,143.0 | 111.8 (1.2) | 15,265.0 | 31,202.4 | 15,937.4 (104.4) | 273,458.5 | 274,109.6 | 651.1 (0.2) |
| 41,275.5 | 40,662.1 | -613.4 (-1.5) | 6,871.0 | 8,146.6 | 1,275.6 (18.6) | 3,948.3 | 3,806.5 | -141.8 (-3.6) | 6,540.3 | 8,527.7 | 1,987.4 (30.4) | 91,032.8 | 91,994.1 | 961.3 (1.1) |
| 393,845.3 | 394,436.9 | 591.6 (0.1) | 78,092.5 | 91,945.7 | 13,853.2 (17.7) | 47,969.9 | 48,862.9 | 893.0 (1.9) | 52,773.6 | 88,564.6 | 35,791.0 (67.8) | 404,606.5 | 407,899.2 | 3,292.7 (0.8) |

cover. This amounts to about 70% of the total land area in forest cover that was present at the time of European settlement (Powell and others 1993). Forest land is defined as land at least 10% stocked by forest trees of any size, including land that formerly had such cover and will be naturally or artificially regenerated to trees (USDA Forest Service 1981). It has been estimated that 90% of the resident or common migrant vertebrate species in the United States use forest habitats to meet at least part of their life requisites.[2]

Based on data from the NRI, just over 96% of the nonfederal land that was in forest cover in 1982 remained in forest cover in 1992 (Appendix B). Of the 14.7 million acres of forest land that was transformed, most (38%) was converted to urban or transportation lands. About 27% of converted forest land went into pasture or rangeland, and about 13% went into federal ownership. Only about 10% of the converted forest acres went into cropland.

Although these statistics imply a fairly stable forest land base, there have been some substantial shifts in the character of forest ecosystems that can greatly affect the distribution and abundance of wildlife. Based on forest inventory information for timberland (i.e., land capable of producing 20 cubic feet of wood per acre per year, and which is available for successive harvests of timber products [USDA Forest Service 1982]), changes in forest cover types and changes in successional stages reveal more dynamics in forest habitats than implied by general forest land area changes (tables 2 and 3). Unfortunately, forest inventory techniques, standards, and geographical definitions have changed over time making it difficult to interpret recent historical trends in forest cover types and successional stages. In addition, interpretation of these trends must be mindful of the fact that timberland can be converted to another status through the designation of parks and wilderness without affecting the forest cover type class or successional stage.

Forest cover types discussed here are those defined by the Forest-Range Environment Study (Garrison and others 1977). In the eastern United States, forests are dominated by the oak-hickory cover type which comprised

*[2] USDA Forest Service. 1979. The 1979 wildlife and fish data base. Data base stored at the Rocky Mountain Research Station.*

Table 2—Recent trends in eastern timberland area by forest ecosystem types (USDA Forest Service [1965, 1974, 1982]; Haynes [1990]; Powell and others [1993]).

| Region | Year | White-jack-red pine | Longleaf-slash pine | Loblolly-shortleaf pine | Spruce-fir | Oak-pine | Oak-hickory | Oak-gum-cypress | Elm-ash-cottonwood | Maple-beech-birch | Aspen-birch |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | thousand acres | | | | | | | | | |
| North[a] | 1963 | 10,680 | - | 3,818 | 19,623 | 2,266 | 58,896 | 1,678 | 18,301 | 32,812 | 23,715 |
| | 1970 | 11,910 | - | 3,422 | 18,899 | 4,085 | 55,536 | 1,361 | 21,971 | 30,657 | 20,484 |
| | 1977 | 11,455 | - | 3,423 | 17,552 | 4,170 | 49,956 | 623 | 19,074 | 35,821 | 19,243 |
| | 1987[c] | 13,349 | - | 2,340 | 16,825 | 3,550 | 47,124 | 795 | 11,283 | 43,384 | 17,774 |
| | 1992[c] | 12,906 | - | 2,310 | 17,752 | 3,371 | 49,431 | 709 | 11,053 | 46,053 | 16,387 |
| | % change[e] | 20.84 | - | -39.50 | -9.53 | 48.76 | -16.07 | -57.75 | -39.60 | 40.35 | -30.90 |
| South[b] | 1963 | 440 | 25,977 | 54,177 | 15 | 24,675 | 57,067 | 36,110 | 2,102 | 506 | - |
| | 1970 | 257 | 18,314 | 49,409 | 13 | 30,942 | 56,324 | 29,268 | 2,756 | 482 | - |
| | 1977 | 370 | 16,754 | 46,576 | 8 | 30,470 | 58,939 | 26,062 | 3,243 | 425 | - |
| | 1987[d] | 514 | 15,491 | 46,248 | 18 | 27,775 | 70,559 | 27,332 | 3,007 | 876 | - |
| | 1992[d] | 609 | 14,130 | 47,014 | 13 | 28,585 | 74,527 | 27,477 | 2,618 | 1,089 | - |
| | % change[e] | 38.41 | -45.61 | -13.22 | -13.33 | 15.85 | 30.60 | -23.91 | 24.55 | 115.22 | - |
| Total East | 1963 | 11,120 | 25,977 | 57,995 | 19,638 | 26,941 | 115,963 | 37,788 | 20,403 | 33,318 | 23,715 |
| | 1970 | 12,167 | 18,314 | 52,831 | 18,912 | 35,027 | 111,860 | 30,629 | 24,727 | 31,139 | 20,484 |
| | 1977 | 11,826 | 16,755 | 49,999 | 17,560 | 34,639 | 108,895 | 26,685 | 22,318 | 36,246 | 19,243 |
| | 1987 | 13,863 | 15,481 | 48,588 | 16,843 | 31,325 | 117,683 | 28,127 | 14,290 | 44,219 | 17,774 |
| | 1992 | 13,516 | 14,130 | 49,324 | 17,765 | 31,955 | 123,958 | 28,186 | 13,671 | 47,139 | 16,387 |
| | % change[e] | 21.55 | -45.61 | -14.95 | -9.54 | 18.61 | 6.89 | -25.41 | -33.00 | 41.48 | -30.90 |

[a]Includes ND, SD (east), NE, KS, and KY.
[b]Does not include KY.
[c]Does not include KY, includes SD (east and west).
[d]Includes KY.
[e]Percent change from 1963 to 1992.

about 35% of the total timberland in 1992. Not only is oak-hickory the most common cover type in the east, but more than 15 million acres have been added to this cover class since 1977 (table 2). The amount of timberland classified as maple-beech-birch has also increased greatly in the last few decades. Since 1970, the area of maple-beech-birch forest has increased by 16 million acres and it now comprises just over 13% of the eastern timberland base. Loblolly-shortleaf pine is currently the second most common eastern cover class (nearly 14% of the timberland base), but it has declined by nearly 15% since 1963. Other eastern cover types that have declined substantially since 1963 include longleaf-slash pine (-46%), elm-ash-cottonwood (-33%), aspen-birch (-31%), and oak-gum-cypress (-25%).

Forest cover types in the West are dominated by conifers (table 3). Douglas-fir, ponderosa pine, and fir-spruce together comprise nearly 70% of the total timberland area in the West. The area classified as Douglas-fir and ponderosa pine has declined by 3 and 10 million acres, respectively. The fir-spruce cover class has actually gained more than 11 million acres. Western hardwoods (e.g., oak, red alder, and aspen) are important for wildlife habitat and watershed protection. Western hardwoods increased in area from 1963 to 1992 by nearly 40% and have increased in value as fuelwood, lumber, specialty millstock, and pulp chips (Powell and others 1993). Western forest cover types that have declined by more that 25% since 1963 include western white pine, larch, ponderosa pine, lodgepole pine, and redwood.

Trends in forest cover classes that are of particular concern are those that have been identified to be critically endangered ecosystems. A critically endangered ecosystem, as defined by Noss and others (1995), is one in which the presettlement extent of the system has been reduced by more than 98%. Noss and others (1995) identified the following forest ecosystems as critically endangered: spruce-fir forests in the southern Appalachians, red and white pine forests in Michigan, longleaf pine forests and savannas in the southeastern coastal plain, slash pine rockland habitat in southern Florida, loblolly/shortleaf pine-hardwood forests in the west gulf coastal plain, Atlantic white-cedar in the Great Dismal Swamp of Virginia and North Carolina, and oak savannas in the Midwest, Willamette Valley, and foothills of the Oregon Coast Range. Of particular note in Noss' list are the southern pine types (loblolly-shortleaf and longleaf-slash) that we reported earlier as having lost substantial timberland acres (see table 2).

Stand size class is another characteristic of forest cover that can be used to describe the structure and age of forest ecosystems (table 4). Nationally, there have been substan-

Table 3—Recent trends in western timberland area by forest ecosystem types (USDA Forest Service [1965, 1974, 1982]; Haynes [1990]; Powell and others [1993]).

| Region | Year | Douglas fir | Ponderosa pine | Western white pine | Fir-spruce | Hemlock-Sitka spruce | Larch | Lodgepole pine | Redwood | Other softwood | Western hardwood |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | thousand acres | | | | | | | | | |
| Rocky[a] Mountain | 1963 | 13,447 | 18,881 | 2,360 | 8,962 | 200 | 2,669 | 13,163 | - | - | 5,941 |
| | 1970 | 11,885 | 14,454 | 631 | 9,800 | 896 | 2,032 | 9,940 | - | - | 4,272 |
| | 1977 | 12,220 | 14,673 | 320 | 10,124 | 1,246 | 1,749 | 9,816 | - | 507 | 4,555 |
| | 1987[b] | 13,304 | 13,714 | 260 | 11,009 | 1,489 | 1,749 | 9,397 | - | 301 | 4,810 |
| | 1992[b] | 13,817 | 14,237 | 191 | 11,196 | 1,573 | 1,742 | 9,106 | - | 671 | 4,960 |
| | % change[c] | 2.75 | -24.60 | -91.91 | 24.93 | 686.50 | -34.73 | -30.82 | - | 32.35 | -16.5 |
| Pacific Coast | 1963 | 23,905 | 17,116 | 2,643 | 6,654 | 9,808 | 863 | 2,633 | 1,596 | - | 5,146 |
| | 1970 | 18,902 | 13,509 | 198 | 8,029 | 9,922 | 711 | 3,294 | 803 | - | 8,545 |
| | 1977 | 18,677 | 11,976 | 126 | 9,732 | 11,620 | 683 | 2,919 | 662 | - | 10,308 |
| | 1987 | 19,023 | 10,927 | 14 | 15,843 | 9,495 | 852 | 2,178 | 1,102 | 492 | 11,028 |
| | 1992 | 20,718 | 11,015 | 13 | 15,748 | 6,715 | 350 | 1,997 | 1,148 | 492 | 10,346 |
| | % change[c] | -13.33 | -35.65 | -99.51 | 136.67 | -31.54 | -59.44 | -24.15 | -28.07 | 0 | 101.05 |
| Total West | 1963 | 37,352 | 35,997 | 5,003 | 15,616 | 10,008 | 3,532 | 15,796 | 1,596 | - | 11,087 |
| | 1970 | 30,787 | 27,963 | 829 | 17,829 | 10,818 | 2,743 | 13,234 | 803 | - | 12,817 |
| | 1977 | 30,897 | 26,649 | 446 | 19,856 | 12,866 | 2,432 | 12,735 | 662 | 507 | 14,862 |
| | 1987 | 32,327 | 24,641 | 274 | 26,852 | 10,984 | 2,601 | 11,575 | 1,102 | 793 | 15,838 |
| | 1992 | 34,535 | 25,252 | 203 | 26,945 | 8,290 | 2,091 | 11,103 | 1,148 | 1,164 | 15,306 |
| | % change[c] | -7.54 | -29.85 | -95.94 | 72.55 | -17.17 | -40.80 | -29.71 | -28.07 | 129.59 | 38.05 |

[a]Does not include ND, SD (east), NE, and KS.
[b]Does not include SD (east and west).
[c]Percent change from 1963 to 1992.

tial reductions (nearly 80%) in the number of acres that are nonstocked (i.e., forest cover that is less than 10% stocked with growing stock trees). The area in seedlings and saplings (less than 5 inches diameter at breast height [dbh]) has remained essentially unchanged from 1963 to 1992; poletimber (greater than 5 inches dbh, but less than sawtimber) has declined by more than 25%; and sawtimber (greater than 9 inches dbh for softwoods, greater than 11 inches dbh for hardwoods) has increased by nearly 25%.

Increases in sawtimber appear to be caused primarily by maturing eastern forests. The area of timberland in sawtimber has increased by more than 50% in the North and by nearly 30% in the South from 1963 to 1992 (table 4). In the West, the area in sawtimber increased by more than 10% in the Rocky Mountain region, but declined by nearly

Table 4—Trends in stand-size class by RPA Assessment region (USDA Forest Service [1965, 1974, 1982]; Haynes [1990]; Powell and others [1993]).

| Class | Year | Total U.S. | North[a] | South[b] | Rocky[c] Mountain | Pacific Coast |
|---|---|---|---|---|---|---|
| | | ---thousand acres--- | | | | |
| Sawtimber | 1963 | 208,945 | 52,974 | 68,828 | 38,639 | 48,504 |
| | 1970 | 215,876 | 58,949 | 74,041 | 36,555 | 46,321 |
| | 1977 | 215,435 | 59,098 | 71,246 | 38,545 | 46,545 |
| | 1987 | 242,449 | 74,548[d] | 78,321[e] | 41,981[f] | 47,599 |
| | 1992 | 259,879 | 81,116[d] | 88,975[e] | 43,339[f] | 46,449 |
| | % change[g] | 24.38 | 53.12 | 29.27 | 12.16 | -4.24 |
| Poletimber | 1963 | 164,794 | 64,808 | 71,580 | 19,063 | 9,343 |
| | 1970 | 126,794 | 60,156 | 46,151 | 12,129 | 8,256 |
| | 1977 | 135,610 | 55,543 | 58,316 | 11,708 | 10,042 |
| | 1987 | 136,773 | 60,445 | 54,888 | 9,454 | 11,986 |
| | 1992 | 120,788 | 48,120 | 53,348 | 8,653 | 10,667 |
| | % change[g] | -26.70 | -25.75 | -25.47 | -54.61 | 14.17 |
| Seedling sapling | 1963 | 99,573 | 39,327 | 49,254 | 4,352 | 6,640 |
| | 1970 | 131,368 | 49,223 | 67,578 | 5,229 | 9,337 |
| | 1977 | 115,032 | 46,676 | 53,286 | 4,955 | 10,115 |
| | 1987 | 92,436 | 31,547 | 44,883 | 5,323 | 10,683 |
| | 1992 | 101,417 | 30,743 | 53,736 | 5,501 | 11,437 |
| | % change[g] | 1.85 | -21.83 | 9.10 | 26.40 | 72.24 |
| Nonstocked | 1963 | 35,533 | 14,680 | 11,407 | 3,569 | 5,877 |
| | 1970 | 20,721 | 9,571 | 4,771 | 2,671 | 3,707 |
| | 1977 | 16,408 | 4,823 | 5,198 | 2,556 | 3,831 |
| | 1987 | 11,649 | 2,247 | 5,380 | 2,186 | 1,836 |
| | 1992 | 7,471 | 1,348 | 3,250 | 1,607 | 1,266 |
| | % change[g] | -78.97 | -90.82 | -71.51 | -54.97 | -78.46 |
| All | 1963 | 508,845 | 171,789 | 201,069 | 65,623 | 70,364 |
| | 1970 | 499,692 | 177,901 | 192,542 | 61,631 | 67,622 |
| | 1977 | 482,485 | 166,141 | 188,045 | 57,765 | 70,543 |
| | 1987 | 483,309 | 168,788 | 183,473 | 58,944 | 72,104 |
| | 1992 | 489,555 | 161,328 | 199,309 | 59,099 | 69,819 |
| | % change[g] | -3.79 | -6.09 | -0.88 | -9.94 | -0.77 |

[a] Includes ND, SD (east), NE, KS, and KY.
[b] Does not include KY.
[c] Does not include ND, SD (east), NE and KS.
[d] Does not include KY; includes SD (east and west).
[e] Includes KY.
[f] Does not include SD (east and west).
[g] Percent change from 1963 to 1992.

5% in the Pacific Coast over the same period. The acreage in poletimber has declined by more than 25% in all regions except the Pacific Coast where it has increased by 14%. Although the acreage in the seedling/sapling stage has remained stable at the national level, the North has actually lost 22% of the timberland area in this stand size class over the last three decades. This loss is more than offset by gains in the seedling/sapling stage across all other Assessment regions. The number of acres classified as nonstocked has declined in all regions since 1963, with the North (91% decline) and the Pacific Coast (78% decline) undergoing the greatest proportional reductions.

Although eastern forests appear to be maturing, stand size class does not adequately capture the structural characteristics of old-growth or ancient forests. Consequently, Noss and others (1995) identified old-growth eastern deciduous forests as critically endangered. Old-growth and ancient forests, in general, and old-growth ponderosa pine forests, in particular, were listed by Noss and others (1995) as endangered (i.e., reductions of 85 to 98%) in the western United States.

## Rangeland Habitats

Rangeland habitats include those areas where the potential natural vegetation is comprised predominately of grasses, grass-like plants, forbs, or shrubs and where herbivory was an important ecological disturbance shaping plant life history (Anderson and others 1976). Rangelands have traditionally been evaluated in terms of livestock production (Busby 1994). There is, however, a growing emphasis on evaluating rangeland habitats from a system health perspective (Committee on Rangeland Classification 1994) that includes a broader understanding of the importance of rangeland in the maintenance of biological diversity. The importance of this perspective becomes more meaningful if one considers that rangeland habitats are used, during some period of the year, by 84% of the mammals and 74% of the bird species that are inhabitants or common migrants in the United States.[3]

Although there is a growing recognition that assessing rangeland health is a critical conservation need, data constraints impede such assessments. According to the Committee on Rangeland Classification (1994), all existing national rangeland assessments suffer from the lack of current and comprehensive field data. Because rangeland inventories have used different methods and sources over time, it is not possible to aggregate these data for national summaries of trends in rangeland condition. For these reasons, our discussion of the status of rangeland habitats will be limited.

Based on data from the NRI, there has been a 2.4% net loss of nonfederal rangeland area nationwide from 1982 to 1992 (see table 1). Of the rangeland acres converted, the largest proportion was converted to cropland (33%); nearly 20% was converted to federal ownership; 14% was converted to pastureland; just over 12% was converted to urban or transportation land; and nearly 9% was converted to forest (Appendix B).

Conversion of rangeland does not address the condition of those acres that remain in range habitats. Past evaluations of range condition are currently a topic of much debate and there is no agreement on how best to assess range condition (Committee on Rangeland Classification 1994). A recent examination of conservation treatment needs on private rangelands in the western United States showed that of the 394.8 million acres of private rangeland in the West, nearly 60% were in need of some management to correct for disturbances that were affecting productive capacity (USDA Natural Resources Conservation Service 1997:33). Brush and weed problems were implicated as the primary agent that reduced productivity on 17% of the rangeland acres; accelerated wind and water erosion was a problem on 23% of the acres; and multiple factors affected 18% of the acres. Unfortunately, the basis for assessments of rangeland condition has changed over time, making it impossible to determine whether rangeland health is improving or deteriorating.

In evaluating the state of rangeland habitats, it is important to recognize that recent land area dynamics or conservation needs assessments do not indicate the extent to which certain rangeland systems have been altered historically. Many rangeland ecosystems had undergone extensive conversions long before land base inventories were designed. In an examination of the status of the nation's ecosystems, Noss and others (1995:5) found that grasslands and shrublands were disproportionately represented among those ecosystems identified as critically endangered (i.e., more than 98% of their historic areal extent had been lost). Approximately 55% of the critically endangered ecosystems were grasslands and 24% were shrubland systems. For comparison, 17% of the critically endangered ecosystems were forests, 2% were forested wetland, and 2% were aquatic.

If we broaden Noss and others' (1995) list of critically endangered ecosystems to include those systems that are thought to be endangered (85-98% of their historic areal extent has been lost), then the distribution of these imperiled rangeland systems by Assessment region is about equal. Six occur in the Rocky Mountain region, seven occur each in the North and Pacific Coast regions, and nine occur in the South (table 5). This balance across Assessment regions was somewhat surprising given the

[3] *USDA Forest Service. 1979. The 1979 wildlife and fish data base. Data base stored at the Rocky Mountain Research Station.*

Table 5--Rangeland ecosystems considered to be endangered or critically endangered (from Noss and others 1995).

| Ecosystem | RPA Region where found | Comments |
|---|---|---|
| Tallgrass prairie east of the Missouri River | North, Rocky Mountain | Chapman (1984), and Klopatek and others (1979) have estimated that 99% of the tallgrass prairie east of the Missouri has been lost. |
| *Arundinaria gigantea* canebrakes in the Southeast | South | Extensive canebrake habitats are nearly all lost with the majority of the remaining habitat occurring as an understory plant or along fencelines (Platt and Brantley 1992) |
| Bluegrass savanna-woodland and prairies in Kentucky | South | Nearly all intact bluegrass savanna-woodland and native prairie have been lost[a]. Of the 2.6 million acres of native prairie, less than 200 acres remain (Mengel 1965, Kentucky Environmental Quality Commission 1992) |
| Black Belt prairies in Alabama and Mississippi and the Jackson Prairie in Mississippi | South | All but a few remnant patches of these prairie systems remain following conversion to agriculture (DeSelm and Murdock 1993) |
| Dry prairie habitats in Florida | South | Virtually all of the dry prairies in Florida have been lost to cattle pasture and agriculture (DeSelm and Murdock 1993) |
| Oak savanna in the Midwest | North | More than 99% of the original oak barrens and savannas have been lost in Wisconsin, Missouri, Michigan, and Minnesota (Nuzzo 1985, 1986, Nelson 1985, Henderson and Epstein 1995) |
| Wet and mesic coastal prairies in Louisiana | South | More than 99% of the original wet and mesic prairies have been lost in Louisiana[b] |
| Lakeplain wet prairie in Michigan | North | More than 99% of this habitat has been lost with only about 500 acres persisting (Chapman 1984) |
| Hempstead Plains grasslands on Long Island, New York | North | More than 99% of the grassland system has been lost (Niering 1992)[c] |
| Serpentine barrens, maritime heathland, and pitch pine-heath barrens in New York | North | A probable 98% loss of these habitats[c] |
| Prairies and oak savannas in the Willamette Valley and Coast Range foothills in Oregon | Pacific Coast | Since European settlement, more than 99% of these habitats have been lost (Ingersoll and Wilson 1991) |
| Palouse prairie | Pacific Coast, Rocky Mountain | Nearly all of the Palouse prairie has been lost to agricultural development throughout its range (Tisdale 1961) |
| California native grasslands | Pacific Coast | Of the original 22 million acres of native grasslands in California, 22 thousand acres remain (Kreissman 1991) |
| Alkali sink scrub in southern California | Pacific Coast | Alkali sink scrub habitats have been nearly extirpated in southern California (Freas and Murphy 1988) |
| Coastal strand in southern California | Pacific Coast | All coastal strand habitats in San Diego county have been lost[d] |
| Sagebrush steppe in the Intermountain west | Rocky Mountain | Most (greater than 99%) of the remaining sagebrush steppe habitats have been affected by grazing; 30% has been overgrazed; species composition is dominated by a few woody plants (West 1996) |

Table 5--(Cont'd.)

| Ecosystem | RPA Region where found | Comments |
|---|---|---|
| Basin big sagebrush in the Snake River Plain of Idaho | Rocky Mountain | Most of the big basin sagebrush habitats have been converted to agriculture (Hironaka and others 1983) |
| Coastal heathland in southern New England | North | About 90% of this habitat has been destroyed since the mid-1800s (Godfrey and Alpert 1985) |
| Limestone redcedar glades in Tennessee | South | About 90% of ecologically intact limestone cedar glades have been lost[e] |
| Calcareous prairie, Fleming glade, and stream terrace sandy woodland/savanna in Louisiana | South | 90 to 99% of these habitats have been lost[b] |
| Coastal sage scrub and coastal mixed chaparral in southern California | Pacific Coast | 70 to 90% of the southern California coastal scrub has been destroyed (Westman 1981, O'Leary 1990). Nearly 92% of the maritime sage scrub and 88% of coastal mixed chaparral has been lost in San Diego County[d] |
| Grassland and shrubland habitats of the lower Rio Grande River delta | South | 95% of the native habitat in the lower delta of the Rio Grande River has been lost; habitat that remains is highly fragmented (Riskind and others 1987) |
| Tallgrass prairie in general | North, South, Rocky Mountain | Of the nearly 145 million acres of tallgrass prairie, 90% has been lost; areas remaining in tallgrass prairie are highly fragmented (Madson 1990) |
| Native shrub and grassland steppe in Oregon and Washington | Pacific Coast | Greater than 90% of this habitat has been lost in Oregon and southwestern Washington (The Nature Conservancy 1992) |
| Low elevation grasslands in Montana | Rocky Mountain | 80 to 90% loss of this habitat in western Montana[f] |

[a]Personal communication. T. Bloom, Kentucky Nature Preserves Commission, Frankfort, KY. As cited in Noss and others (1995).

[b]Smith, L.M. 1993. Estimated presettlement and current acres of natural plant communities in Louisiana currently recognized by the Louisiana Natural Heritage Program, January 1993. Unpublished table. Louisiana Department of Wildlife and Fisheries, Natural Heritage Program, Baton Rouge. As cited in Noss and others (1995).

[c]Reschke, C. 1993. Estimated numbers of EOs, acreage, trends, and threats for selected New York natural communities. Unpublished report. New York State Department of Environmental Conservation, Natural Heritage Program, Lathan. As cited in Noss and others (1995).

[d]Oberbauer, T.A. 1990. Areas of vegetation communities in San Diego County. Unpublished report. County of San Diego, Department of Planning and Land Use, San Diego, CA. As cited in Noss and others (1995).

[e]Pyne, M.; Durham, D. 1993. Estimation of losses of ecosystems in Tennessee. Unpublished table. Tennessee Department of Environment and Conservation, Ecological Services Division, Nashville. As cited in Noss and others (1995).

[f]Chadde, S. 1992. Decline of natural ecosystems in Montana. Unpublished report. U.S. Forest Service, Missoula, MT. As cited in Noss and others (1995).

regional distribution of rangelands. About 53% of all rangelands are found in the Rocky Mountain region, 26% are found in the Pacific Coast, 15% are found in the South, and less than 1% occurs in the North (Bones and others 1989). Whereas rangeland habitats are relatively rare in the eastern United States, more than half of the endangered grassland systems listed by Noss and others (1995) occur there. This disproportionate representation of endangered grassland and shrubland systems in the East is likely a product of their rarity. Given that much of the East naturally supports forest vegetation, the maintenance of grassland and shrubland systems in these landscapes is due, in part, to periodic disturbance. If the frequency of natural disturbance events is altered, then grassland and shrubland systems may be lost to uninterrupted successional processes.

A review of the factors that have contributed to the loss of grass and shrublands appears to support this explanation. Noss and Peters (1995) identified agricultural development, fire suppression, urban development, and exotic species invasions as the primary agents of grass and shrubland system degradation and destruction. Tallgrass prairie habitats have been lost primarily to agricultural development. Eastern and Northwestern grasslands and savannas have been lost to urban development, agriculture, and fire suppression. These factors are also prominent in the loss of native California grasslands, but exotic species invasions have also played a major role in the loss of these grassland habitats (Barbour and others 1991).

Use of rangeland habitats by livestock has shown a general decline over the last two decades (figure 3). Nationally, the number of cattle has declined by about 22% from the mid-1970s to 1996 (figure 3*a*). Cattle numbers have also declined in each Assessment region. The greatest decline, both in terms of absolute numbers and relative change, occurred in the North with cattle numbers declining by 12.7 million head (-33%). Substantial declines in the number of cattle have also been reported in the South (-10.3 million head, or -21%). Reductions in the Rocky Mountain and Pacific Coast have been more moderate over this time period. Although the long-term trend has shown a decline, cattle numbers have increased since the last Assessment. Cattle numbers have increased by more than 5% nationwide since 1989 due to gains reported in the Rocky Mountain (+13%) and Southern (+8%) regions (figure 3*a*).

Sheep numbers have shown even greater declines when compared to cattle (figure 3*b*). Since 1975, the number of sheep has declined by nearly 52%, nationally. Although the greatest decline was observed in the Rocky Mountains (-56%), all Assessment regions have exhibited declines by more than 40% over the last 20 years. Unlike cattle, there is no evidence of any recent departure from this long-term decline.

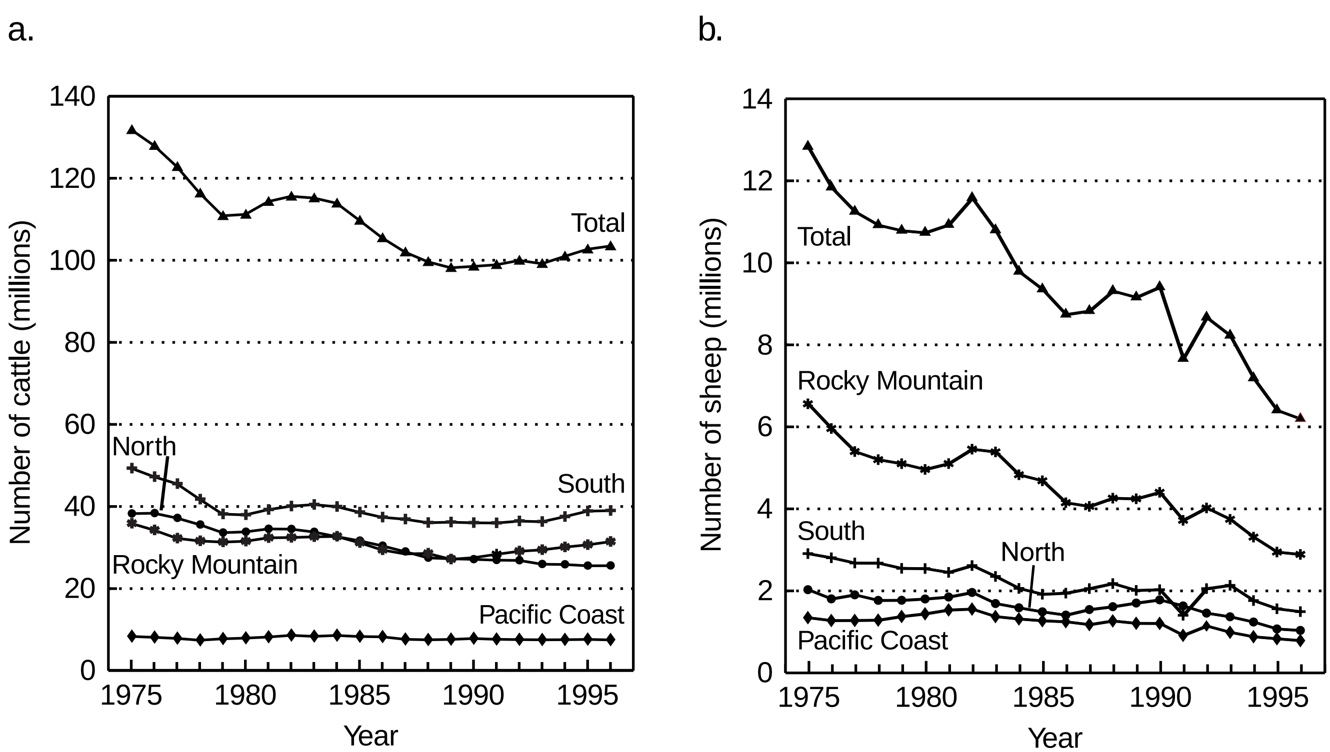


Figure 3. Trends in number of cattle (a) and sheep (b) from 1975 to 1996 for the United States and RPA Assessment region (Mitchell [In press]).

The declining trend in livestock numbers also is reflected in the permitted Animal Unit Months (AUMs) on Forest Service and Bureau of Land Management (BLM) lands (figure 4), although the magnitude of the decline is less. Total permitted AUMs have declined by 7 and 20% on Forest Service and BLM lands, respectively. The greatest proportional decline on Forest Service lands occurred in the South (-49%), followed by the Pacific Coast (-12%), North (-10%), and the Rocky Mountains (-5%). On BLM lands the number of permitted AUMs in the Rocky Mountains and Pacific Coast declined by 17 and 36%, respectively.

Livestock numbers may not accurately reflect use or quality of rangeland habitats. Livestock often are concentrated in feedlot production sites for all (e.g., hogs) or a portion (e.g., cattle) of the production cycle. New production and processing technologies have resulted in livestock concentrations that have not been observed in the past (USDA Natural Resources Conservation Service 1997). Although growth in concentrated animal production operations has the potential to reduce impacts associated with grazing on open range, such concentrations pose a different set of environmental impacts associated with animal manure and the contamination of aquatic habitats (USDA Economic Research Service 1997).

## Agricultural Habitats

Agricultural habitats consist primarily of lands used to produce food, feed, fiber, and oilseed crops, and therefore are described as cropland and pastureland. This excludes tree farms and commercial forests. Almost all croplands are owned by the private sector. Small amounts of cropland occur on federal wildlife refuges to produce food for migratory waterfowl and other species, but they will not be discussed here. Cropland can be readily divided into two categories: cultivated and noncultivated. Cultivated cropland is annually planted for commodities including rowcrops such as corn, soybeans, and cotton or small grains such as wheat and oats. Noncultivated cropland consists of land planted to multi-year or perennial crops such as hay, horticultural plants, and orchards. Pastureland is land used for livestock grazing and differs from rangeland in the level of management it receives. Pasturelands are planted primarily to introduced or domesticated native forage species and receive periodic cultural treatments such as tillage, fertilization, mowing, weed control, or irrigation (Jacoby 1989).

Agricultural lands provide food and cover used by many species of wildlife, and many agricultural cropping practices are used in managing habitats for some wildlife. Intensive agricultural land use becomes detrimental to

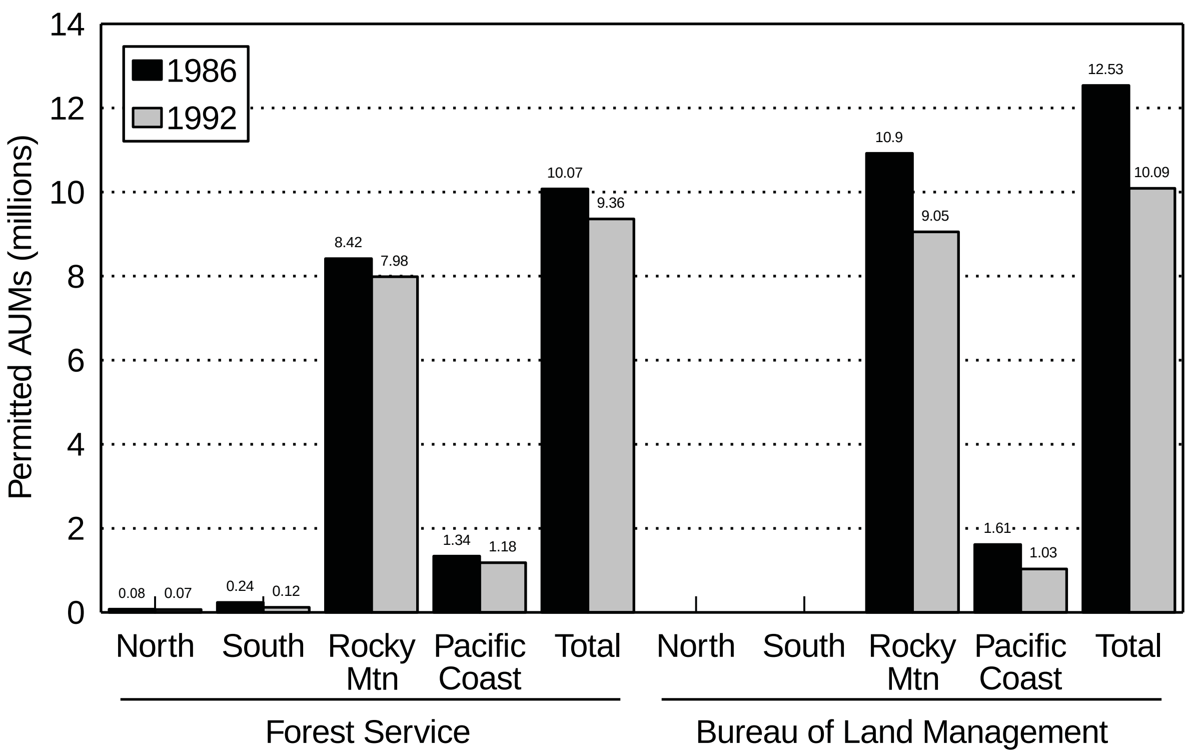


Figure 4. Number of permitted AUMs on public lands (Forest Service and BLM) in 1986 compared to 1992 (Mitchell [In press]).

wildlife when it is applied extensively over large contiguous tracts, when adjacent native habitats are converted to agricultural production, when land is farmed beyond its capacity resulting in excessive soil erosion, or when pesticides are applied inappropriately. Sound land stewardship that integrates agricultural production with habitat management and ecological processes can mitigate some of these adverse effects (Dumke and others 1981, Brady 1985, Warner and Brady 1994).

Cultivated cropland represented about 17%, noncultivated cropland represented about 3%, and pasture accounted for about 6% of the nonfederal land in the United States in 1992 (Appendix B). Another 2% of the nation's land (equal to about 10% of cultivated cropland) was enrolled in the CRP. The CRP removed highly erosive land out of crop production and established perennial vegetative cover for 10-15 years. During the decade from 1982 to 1992 there was a net decline in cultivated cropland of 11%. This net loss of 40.7 million acres resulted from 29.5 million new acres being converted to cultivated cropland while 70.1 million acres of cultivated cropland were converted to other uses (Appendix B). Of those 70.1 million acres of converted cropland, 43% was enrolled into the CRP, 26% was converted to noncultivated cropland, 16% went to pasture, and about 4% went to urban land.

Noncultivated cropland increased by 4% during the decade. This shift was the net result of 18.5 million acres of cultivated cropland being converted to noncultivated cropland and 14.5 million acres of noncultivated cropland being converted to cultivated cropland. These shifts may be partially explained by crop rotations that move land between crop production and hay.

There was a net decline in pasture of about 4% during the decade, but pasture use was more dynamic than this net change implies. Nearly 20% of the 1982 pasture land area was converted to another cover or use by 1992 (Appendix B). Most of the pasture loss went to cropland (29% cultivated, 15% noncultivated) while 30% of it reverted to forest land. A smaller acreage of pasture was converted into urban (9%), rangeland (5%), and CRP (5%).

Agricultural land is most abundant in the North with about 36% of the nation's total, followed by the South (31%), Rocky Mountain (28%), and the Pacific Coast (5.0%) regions (Appendix B). The North has 38% of the nation's cropland, followed by the Rocky Mountain (33%), South (24%), and Pacific Coast (5%) regions. The nation's pastureland is greatest in the South (52%), followed by the North (32%), Rocky Mountain (12%), and Pacific Coast (4%) regions.

All four regions exhibited declines in cultivated cropland and increases in noncultivated cropland during the decade. The South lost 17% of its cultivated cropland, followed by the Pacific Coast (-14%), Rocky Mountain (-10%), and North (-7%) (Appendix B). Increases in noncultivated cropland were greatest in the South showing a 12% increase, followed by the Pacific Coast (6%), North (2%), and Rocky Mountain (1%) regions. Pastureland increased slightly in the South (+1%) and Rocky Mountain (+<1%) regions, but declined in the North (-13%) and Pacific Coast (-6%) regions.

Apart from agricultural land area dynamics, the intensity with which land is managed for crop production also affects wildlife habitat quality. In general, more intensive use of agricultural lands as indicated by such attributes as farm size, nutrient inputs, pesticide use, soil erosion, and irrigation reduces the quality of wildlife habitats associated with agricultural lands (Allen 1995).

Farm size is an indirect indicator of the quality of agricultural habitats to wildlife. In order to capture the economic efficiencies associated with larger farm equipment and new technology, agricultural production has become concentrated in fewer and larger farms (Committee on Impacts of Emerging Agricultural Trends on Fish and Wildlife Habitat 1982). From 1940 to 1992 the number of farms has declined by nearly 75% (from 7 to 1.9 million farms), while the average size of farms has increased by nearly 200% (from 160 to 490 acres/farm) (USDA Economic Research Service 1997). The proportion of farms that are greater than 500 acres has increased at an accelerated pace since about 1960, while the proportion of farms in the smaller size classes (1 to 49 acres and 50 to 499 acres) has declined. Because of these changes, farms are now characterized by larger field sizes, reduced crop diversity, and a loss of small but important wildlife habitats such as fencerows, hedgerows, field border strips, wetlands, woodlots, and odd noncultivated areas within fields (Brady 1988; Warner and Brady 1994). With this form of agricultural intensification the amount of vertical and horizontal habitat diversity is reduced, which has been shown to degrade wildlife habitats in agriculturally dominated landscapes (Flather and others 1992).

In order to maintain high agricultural productivity, supplemental nutrient inputs are required on most agricultural lands. Since 1960, the amount of primary nutrient use (i.e., nitrogen, potash, and phosphate) increased from 7.5 million tons to 23.7 million tons in 1981--an increase of 216% (figure 5) . From 1981 to 1995, the amount of primary nutrient use has fluctuated around 20 million tons. Although the most recent decade shows constancy in amount of primary nutrient use, nitrogen applications show evidence of an increasing trend, having reached a maximum in 1994 of 12.6 million tons. Given that cropland and pastureland have declined in area since 1982, then the amount applied to each acre has actually increased, indicating an intensification of agricultural land management.

Pesticides have been the fastest growing input to agricultural production in the last 50 years. Early USDA benchmark surveys of pesticide use indicated that applications grew from 215 million pounds in 1964 to 572

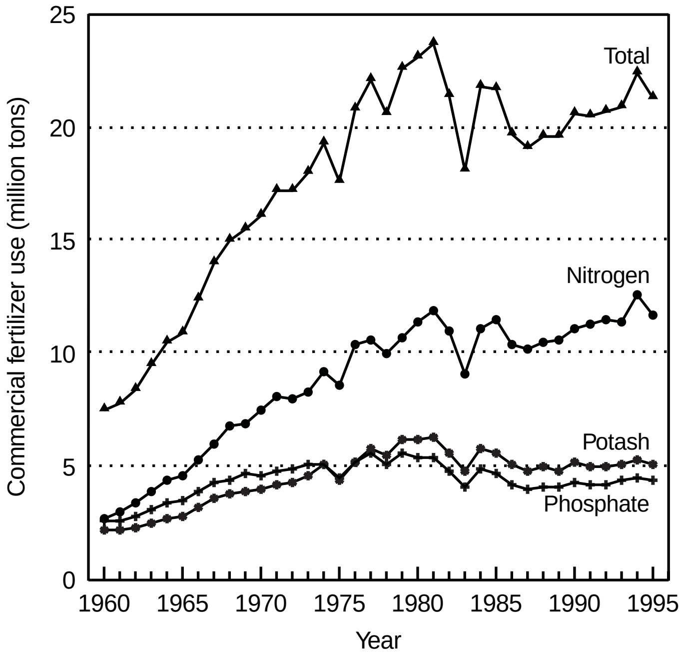


Figure 5. Trends in commercial fertilizer use among primary nutrients (nitrogen, potash, and phosphate) from 1960 to 1995 for the United States (USDA Economic Research Service 1997).

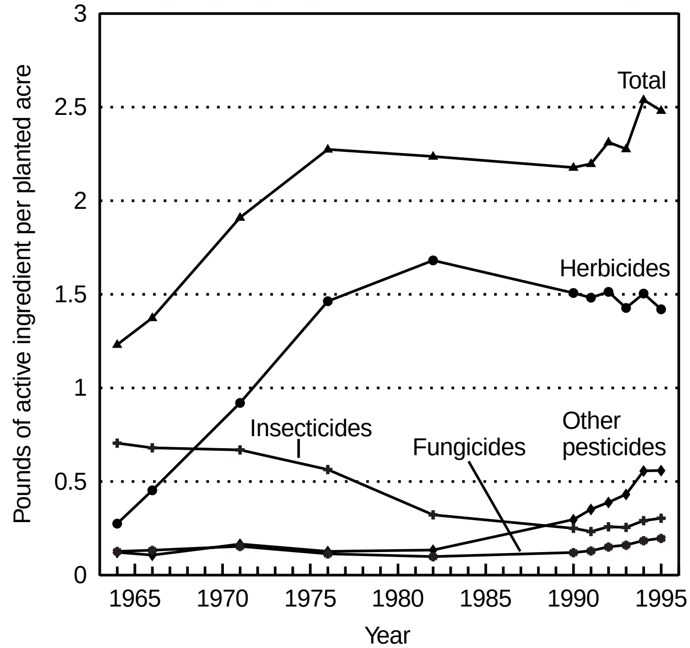


Figure 6. Trends in pesticide use from 1964 to 1995 for the United States (USDA Economic Research Service 1997).

million pounds in 1982 (USDA Economic Research Service 1997). It is difficult to interpret these trends since the amount of pesticides used is a function of acres planted, the proportion of acres treated, and the application rate per treated acre. Figure 6 shows the trend in pounds of active ingredients of pesticide per planted acre from 1964 to 1995. Over that period, application rates for all pesticides has more than doubled.[4] The greatest increases were observed in herbicides (+415%), other pesticides (+358%), and fungicides (+55%). The use of insecticides has declined by 57% since 1964 due to the replacement of organochlorine insecticides with other insecticides that can be applied at much lower rates (USDA Economic Research Service 1997:121). Recent trends in insecticide applications have shown a steady increase from 0.234 pounds of active ingredients per acre planted in 1990 to 0.305 pounds of active ingredients per acre planted in 1995--an increase of 30%.

Irrigation is an agricultural intensification that can negatively affect aquatic (e.g., reduced instream flows, reduced water quality) and terrestrial habitats (e.g., increased erosion, increased field salinity). Even though cropland area has declined during the 1982 to 1992 decade, total acres irrigated has increased slightly from 61.6 million acres to 62.1 million acres (table 6). Irrigated cropland has increased at the greatest rate in the North (+14%), where an additional 403,000 acres were irrigated in 1992 compared to 1982. Minor increases were observed in the South (+1%), with essentially no change in the Rocky Mountains. The Pacific Coast region did observe a net decline in irrigation of 309,000 acres (-2%). Although there was a net decline in irrigated cropland in the Pacific Coast, the proportion of cropland that was irrigated increased by more than 4% (table 6). Nationally, the proportion of cropland that was irrigated increased from 15% in 1982 to 16% in 1992 (table 6).

Increased soil erosion and resulting sediment deposition has been one of the effects of agricultural intensification throughout most of history. However, increased awareness on the part of agricultural producers, successful delivery of technical assistance, and recent USDA incentive programs linked to stewardship are paying off. In 1982, 73% of the cultivated cropland was experiencing sheet and rill erosion rates lower than the T-value (i.e., the tolerable limit) required to maintain productivity. By 1992 that level had increased to nearly 79% of the cultivated cropland (USDA Natural Resources Conservation Service 1994). Likewise, wind erosion rates on cultivated cropland in the plains states and other areas subject to wind erosion have declined. The proportion of cultivated cropland protected from wind erosion increased from 79 to 84% from 1982 to 1992 (USDA Natural Resources Conservation Service 1997). These improvements stem from improved technology applied on the land and the

[4] *As estimated for selected crops including corn, soybeans, wheat, cotton, potatoes, other vegetables, citrus fruit, apples, and other fruits (see USDA Economic Research Service 1997).*

conservation provisions of USDA Farm Programs since 1985, including the removal of 34 million acres of eroding cropland that was enrolled in the CRP.

Other measures of soil erosion that indicate better land stewardship are shown in table 7. Sheet and rill erosion, as estimated from the Universal Soil Loss Equation (USLE; Wischmeier and Smith 1978), has declined nationally and in all four RPA regions on both cultivated and noncultivated cropland from 1982 to 1992. In addition, wind erosion as estimated by the Wind Erosion Equation (WEQ; USDA Soil Conservation Service 1978) has declined in all regions except the Pacific Coast. Erosion rates declined between 1982 and 1992 primarily because many of the cropland acres with elevated erosion rates were entered into the CRP, removing them from cultivation and protecting them with perennial vegetation for 10-15 year contracts, beginning in 1986. Another indication of better stewardship are the acres of cropland with lower values of the erodibilitiy index (EI) between 1982 and 1992. The EI is computed using the soil, climatic, and topographic variables from the USLE and WEQ in the numerator and the T-value in the denominator. The EI calculated using this procedure does not include the effect of management practices such as contour farming

Table 6--Trends in irrigated cropland from 1982 to 1992 (USDA Natural Resources Conservation Service 1994).

| Region | Acres of irrigated cropland 1982 | 1992 | % change | Proportion of cropland irrigated 1982 | 1992 | change |
|---|---|---|---|---|---|---|
| | --------thousand acres------- | | | --------------------percent----------------- | | |
| North | 2,809.2 | 3,212.5 | 17.2 | 1.8 | 2.3 | 0.5 |
| South | 20,588.2 | 20,853.6 | 1.3 | 19.3 | 22.9 | 3.6 |
| Rocky Mountain | 25,818.3 | 25,839.3 | <1 | 18.9 | 20.7 | 1.8 |
| Pacific Coast | 12,438.9 | 12,129.6 | -2.5 | 54.1 | 58.2 | 4.1 |
| Total U.S. | 61,654.6 | 62,115.0 | <1 | 14.7 | 16.3 | 1.6 |

Table 7--Comparison of erosion indicators from 1982 to 1992 (USDA Natural Resources Conservation Service 1994).

| | Total U.S. 1982 | Total U.S. 1992 | North 1982 | North 1992 | South 1982 | South 1992 | Rocky Mountain 1982 | Rocky Mountain 1992 | Pacific Coast 1982 | Pacific Coast 1992 |
|---|---|---|---|---|---|---|---|---|---|---|
| USLE[a] (tons/acre/year) | 4.0 | 3.1 | 5.1 | 3.7 | 4.5 | 3.8 | 2.5 | 2.0 | 3.3 | 2.3 |
| WEQ[b] (tons/acre/year) | 3.4 | 2.4 | 1.5 | 1.3 | 4.2 | 2.7 | 5.1 | 3.7 | 1.9 | 2.1 |
| EI[c] < 2 | 81,400.0 | 78,645.3 | 48,440.7 | 47,329.3 | 16,690.2 | 15,665.0 | 5,422.4 | 5,058.4 | 10,846.7 | 10,592.6 |
| 2 ≤ EI < 5 | 134,023.7 | 125,738.6 | 51,032.2 | 49,080.4 | 38,800.6 | 34,606.8 | 40,182.4 | 38,693.0 | 4,008.5 | 3,358.4 |
| 5 ≤ EI < 8 | 80,281.0 | 72,328.2 | 18,597.9 | 17,251.3 | 20,235.6 | 16,864.4 | 38,623.8 | 35,758.0 | 2,823.7 | 2,454.5 |
| EI ≥ 8 | 124,847.6 | 105,238.0 | 36,170.8 | 32,067.1 | 30,920.9 | 23,742.2 | 52,460.0 | 44,987.5 | 5,295.9 | 4,441.2 |

[a]Sheet and rill erosion in tons/acre/year estimated from the Universal Soil Loss Equation (USLE).
[b]Wind erosion in tons/acre/year estimated from the Wind Erosion Equation. Erosion estimates from USLE and WEQ are not additive, but apply at different locations within the regions.
[c]EI is the erodibility index estimated from either of the two erosion equations. From the USLE the EI is estimated as: EI=RK(ls)T, where R=rainfall factor, K=soil erodibility factor, ls=length and percent slope factors, and T=tolerable soil loss limit. Units displayed are acres X 1,000. From the WEQ the EI is estimated as: EI=CI/T, where C is the climatic factor expressed as a percent, and I is the soil erodibility factor. Only the higher of the EI(USLE) or EI(WEQ) is reported for any particular location.

or conservation tillage; therefore it represents an index of potential erosion based upon natural conditions. The number of acres in each of the EI categories for cropland have been reduced since 1982 (table 7). EI values greater than or equal to 8 are considered to be highly erodible and those acres declined in all four Assessment regions with the national decline being 16%—a reduction that exceeds all other EI categories. Again this is the result of land-use shifts where the most erodible cropland acreage has been shifted to other uses, indicating that USDA programs since 1985 targeted those lands with the greatest potential for environmental damage.

USDA Farm Programs implemented since 1985 have had a positive effect on curbing soil erosion and on managing croplands. About 75-85% of farmers traditionally have participated in USDA commodity support programs (U.S. Department of Agriculture 1986). The particular programs that recently have had the greatest effect on encouraging conservation and stewardship of croplands include the CRP and the highly erodible land provisions of the 1985 Food Security Act (i.e., Conservation Compliance and Sodbuster).

The CRP has the greatest potential to directly improve wildlife habitat associated with agriculture. Participation in the CRP was greatest in the Rocky Mountain region with 45% of the nation's total, followed by the South (26%), North (24%), and Pacific Coast (5%). Land enrolled in the CRP has shown local benefits to many wildlife species, especially grassland nesting birds (Johnson and Schwartz 1993, Kimmel and others 1992), but the program's effect at the regional scale may not be as strong (Brady and Flather 1998). Although CRP has received much of the policy attention related to direct wildlife habitat benefits, other Farm Program provisions also have the potential to benefit wildlife habitat indirectly.

The highly erodible lands provisions of the 1985 Food Security Act included the "Sodbuster" and Conservation Compliance provisions. Sodbuster requires that lands newly converted to cropland are managed to control soil erosion within specified levels. Conservation Compliance required that all highly erodible (e.g., EI ≥8) croplands be farmed according to an approved plan that brings soil erosion losses to within specified acceptable levels. Both provisions were tied to other USDA programs such that participants in any of those programs must meet the soil erosion criteria specified in a mutually agreed upon conservation plan in order to receive USDA program benefits. The Conservation Compliance provisions will be effective as long as USDA continues to offer financial incentives for the numerous farm programs, but will lose effectiveness if USDA reduces program benefits in the future. Any benefits to wildlife from these programs were largely indirect in that they did not generally establish habitat as in the CRP (Brady 1988), but soil erosion was reduced and therefore sediment delivery to aquatic habitats was also reduced. Additionally, more careful selection of lands suitable for cultivation occurred and the conversion of highly erodible rangelands, pastures, and forests was likely reduced. While these provisions and the CRP were generally retained in the 1996 Farm Act, additional programs such as the Wildlife Habitat Incentives Program (WHIP) and the Environmental Quality Incentives Program (EQIP) will create additional USDA incentives to manage agricultural lands to benefit at least some species of wildlife.

## Wetland Habitats

Wetlands are characterized by constant or recurrent shallow inundation, or saturation, at or near the surface (Committee on Characterization of Wetlands 1995). For the purposes of classification, Cowardin and others (1979:3) have specified three diagnostic attributes of wetland habitats of which all wetlands exhibit at least one: "(1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soils; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year." Wetland ecosystems are generally very productive and they are often critical to flood and erosion control, aquifer recharge, and water purification (Mitsch and Gosselink 1986; Scodari 1997). The inherent productivity of wetlands supports a diversity of wildlife and fish that are important to commercial fisheries, furbearer harvests, waterfowl hunting, recreational fishing, and nonconsumptive outdoor recreation and study.

Despite these values, it was not long ago that Federal land policies encouraged the conversion of wetlands (Scodari 1997). During the early settlement period of America, wetlands were perceived as an impediment to economic development and up until the mid-1970s, wetland drainage and conversion was an accepted land use policy (Mitsch and Gosselink 1986). Of the 221 million acres of wetlands that occurred in the lower 48 states at the time of Colonial America, only 104 million acres remained by the mid-1980s (Dahl 1990). A study by Frayer and others (1983) estimated that the rate of wetland loss was nearly 460,000 acres/year from the mid-1950s through the mid-1970s. The annual rate of wetland loss had declined substantially from the mid-1970s to mid-1980s to about 260,000 acres/year (Dahl and Johnson 1991).

Agricultural development has been the primary economic force in the conversion of wetlands. Indeed, some of the most fertile and productive agricultural soils in such areas as the Corn Belt, Northern Plains, Mississippi

Delta, and the Southeastern states were wetlands that were drained for agricultural production (Brady and Flather 1994). According to Frayer and others (1983), conversion of wetlands to cropland production accounted for 87% of the wetland losses during the period 1954-1974. From the mid-1970s to mid-1980s, the role of agricultural development in wetland conversion had diminished to 54% (Dahl and Johnson 1991).

The most recent data on wetland habitat trends come from the 1992 NRI (USDA Natural Resources Conservation Service 1994). We analyzed wetland trends from 1982-1992 and found that the rate of wetland loss has continued to decline and that the land use activities responsible for wetland conversion have shifted. There was a net loss of nearly 791,000 acres of wetlands over the period for an average annual loss rate of 79,000 acres/year (figure 7)—a reduction of nearly 70% in the loss rate observed between the mid-1970s and mid-1980s. Among those Assessment regions that had a net loss of wetlands, the majority of lost wetlands (56%) occurred in the South, 41% occurred in the North, and the remaining 3% were lost from the Pacific Coast. The Rocky Mountain region actually had a small, yet statistically nonsignificant, net gain of wetland habitats. Of the wetlands that were converted, 85% were palustrine (also called inland freshwater wetlands) with the greatest losses of this type occurring in the South and North. An additional 10% of the lost wetlands were estuarine and were concentrated in the South, particularly along the Gulf Coast.

The reduced rate of wetland loss that has occurred in the last decade was caused, in part, by Federal Programs designed to conserve and restore wetland habitats (e.g., "Swampbuster" or the Wetlands Conservation provisions of recent Farm Acts since 1985). Of particular note is the Wetlands Reserve Program, which has restored approximately 665,000 acres of wetlands from 1992 through early-1999.[5] Nearly 55% of these restored wetland acres occurred in the South, 26% were in the North, 10% were in the Rocky Mountain region, and the remaining 9% occurred in the Pacific Coast.

Loss of wetlands by wetland class (i.e., the dominant life form or substrate composition [Cowardin and others 1979]) indicates that wetlands characterized by emergent vegetation accounted for 70% of lost wetlands while scrub-shrub wetlands accounted for an additional 29% of converted wetlands (table 8). Forested wetlands actually gained more than 218,000 acres nationwide. When examined regionally, forested wetlands showed varying dynamics. The South lost 51,000 acres of forested wetlands, with much of the loss occurring in the Southeast and Mississippi Delta regions (Cubbage and Flather 1993). The loss of forested wetlands in the South was more than offset by a gain of 304,000 acres of forested wetlands in the North. The large increase in forested wetlands in the North is likely due to successional changes from scrub-shrub to forested wetlands.

Land use activities that have caused the conversion of wetland habitats have shifted from agricultural to urban development (figure 7). Based on the 1992 NRI, urban and built-up land (i.e., urban and suburban areas, residential

[5] *USDA Natural Resources Conservation Service, Wetlands Reserve Program data. Report on file at the Rocky Mountain Research Station.*

Table 8--Estimates of wetland area by wetland class from 1982 to 1992 (USDA Natural Resources Conservation Service 1994).

| Region | Emergent 1982 | Emergent 1992 | Emergent Gain/loss | Emergent (% change) | Scrub-shrub 1982 | Scrub-shrub 1992 | Scrub-shrub Gain/loss | Scrub-shrub (% change) |
|---|---|---|---|---|---|---|---|---|
| | thousand acres | | | | | | | |
| North | 8,712.3 | 8,352.7 | -359.6 | (-4.1) | 5,042.6 | 4,862.7 | -179.9 | (-3.6) |
| South | 11,897.5 | 11,533.2 | -364.3 | (-3.1) | 1,374.7 | 1,272.9 | -101.8 | (-7.4) |
| Rocky Mtn. | 8,753.0 | 8,758.3 | 5.3 | (<0.1) | 511.6 | 509.6 | -2.0 | (-0.4) |
| Pacific Coast | 1,676.7 | 1,684.3 | 7.6 | (0.4) | 303.2 | 295.2 | -8.0 | (-2.6) |
| Total U.S. | 31,039.5 | 30,328.5 | -711.0 | (-2.3) | 7,232.1 | 6,940.4 | -291.7 | (-4.0) |

developments, industrial sites, roads, railroads, airports, golf courses, but excluding on-farm structures) were responsible for 57% of the wetland acres that were converted during the 1982-1992 period. Agricultural development now accounts for only 20% of the wetland conversions nationwide over the 1982-1992 period. The contribution of various land use activities to wetland conversions does vary regionally. Agricultural development was the most important factor in the Rocky Mountains (37% of regional wetland area lost) and least important in the Pacific Coast (10% of regional wetland area lost). Development associated with urban and built-up areas was clearly the most important in those regions with substantial coastal wetlands. Urban development in the North and South accounted for 63% and 58% of the wetland conversions, respectively. The prevalence of wetland losses due to urban development in regions with a lot of coastal wetlands is consistent with Culliton and others' (1990) finding that human population growth in coastal counties was four times the national average over the last 50 years.

Because the NRI inventories nonfederal land only, some caution is required in interpreting these trends. Some areas that were wetlands in 1982 came under federal ownership in 1992 and a small amount of federally owned land in 1982 went into nonfederal wetlands in 1992. We assumed that the wetland status of these areas was not affected by the change in land ownership and, therefore, they were not included in our estimates of loss, gain, or percent change depicted in table 8 and figure 7. Of the 394,500 acres of wetlands in 1982 that were converted to federal ownership by 1992, 87% occurred in the South, 7% occurred in the Pacific Coast, with the remainder being split between the North and Rocky Mountain regions.

## Implications of Habitat Trends to Wildlife

National trends in major land use categories (i.e., forest land, rangeland, and agricultural land) have shown relatively minor changes in total area over the last 50 years. Although the relative change in area over time has been small, the absolute number of acres that have shifted among the various land use activities has been substantial. As we reviewed earlier, the most notable recent land use shift has been caused by the Conservation Reserve Program, which retired approximately 36 million acres of cropland into perennial vegetative cover for 10- to 15-year contracts. Area changes in minor land use and land cover classes tended to show greater relative changes over time, with the 120% increase in urban land from 1945-1992 being particularly noteworthy. The increase in urban land also has become the primary factor in the continued loss of wetland habitats nationwide, with 57% of the wetlands lost between 1982 and 1992 being caused by conversion to urban and built-up land.

Land use activities fundamentally affect the composition and configuration of wildlife habitats. Therefore, the trends reviewed above presage the changes in wildlife populations and harvests that are reviewed in the next section. Of the land use and land cover changes that we reviewed earlier, those that are likely to significantly affect wildlife populations and harvests include the increase in urban and built-up land, the retirement of cropland acreage into the Conservation Reserve Pro-

Table 8 (Cont'd.)

| Wetland class | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Forested | | | Other | | | Total | | |
| 1982 | 1992 | Gain/loss (% change) | 1982 | 1992 | Gain/loss (% change) | 1982 | 1992 | Gain/loss (% change) |
| thousand acres | | | | | | | | |
| 23,657.6 | 23,961.2 | 303.6 (1.3) | 4,531.6 | 4,438.2 | -93.4 (-2.1) | 41,944.1 | 41,614.8 | -329.3 (-0.8) |
| 35,626.1 | 35,575.2 | -50.9 (-0.1) | 5,255.6 | 5,325.9 | 70.3 (1.3) | 54,153.9 | 53,707.2 | -446.7 (-0.8) |
| 384.9 | 368.0 | -16.9 (-4.4) | 3,003.9 | 3,030.1 | 26.2 (0.9) | 12,653.4 | 12,666.0 | 12.6 (<0.1) |
| 690.8 | 673.8 | -17.0 (-2.5) | 823.3 | 813.5 | -9.8 (-1.2) | 3,494.0 | 3,466.8 | -27.2 (-0.8) |
| 60,359.4 | 60,578.2 | 218.8 (0.4) | 13,614.4 | 13,607.7 | -6.7 (<-0.1) | 112,245.4 | 111,454.8 | -790.6 (-0.7) |

gram, changes in forest successional stages (as indicated by forest cover type changes and shifts among timber size classes), the extensive loss of grassland habitats, and the continued loss of wetland habitats. Based on these land use and land cover trends, we should expect increases in species that tolerate intensive land use activities, increases in species that are associated with agricultural habitats, decreases in species that are associated with grassland habitats and earlier successional stages of forest habitats (particularly in the North), and declines in species associated with wetland habitats (with the possible exception of species occurring in the northern plains states).

## Population and Harvest Trends

The ecosystems within the United States support some of the most diverse temperate forests, warm deserts, and shallow-water wetland systems found globally (Ricketts and others, 1999). Because of the diversity of resident and common migrant species that occur within the United States, we address the status and trends of populations and harvest by major species categories including: big game, small game, migratory game birds, furbearers, nongame, and threatened and endangered species. Esti-

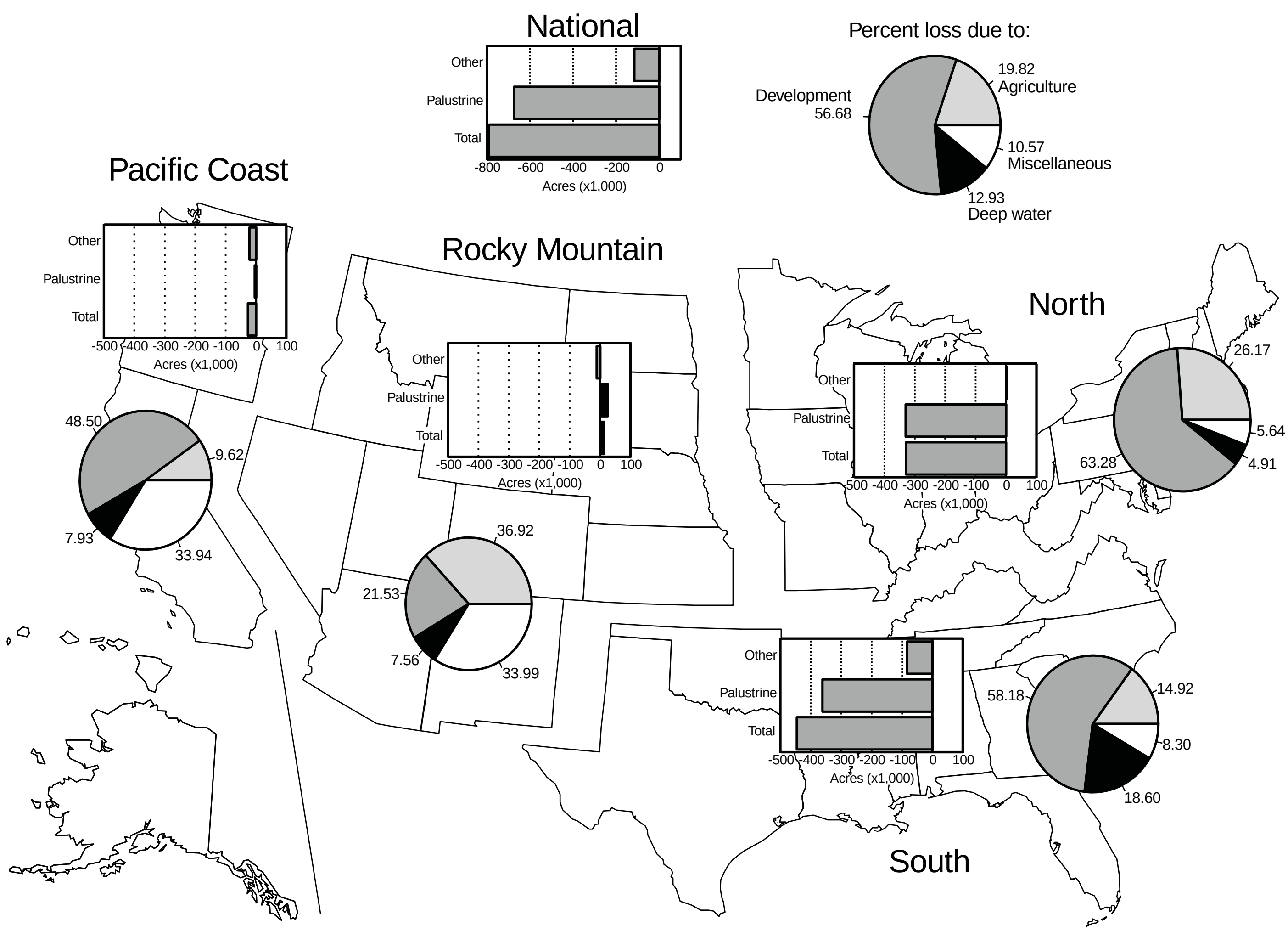


Figure 7. Net changes in wetland acres, and land use factors causing wetland loss from 1982 to 1992 for the United States and RPA Assessment region (USDA Natural Resources Conservation Service 1994).

mates of species populations and harvests (not including poaching) were compiled primarily from data provided by cooperating state and federal agencies. Because data sources vary by species categories, the details concerning source documents and data bases will be discussed within each species category section. Given the diversity of data sources that we used, it should not be surprising that data quality also varied greatly. In some cases, national inventories have been designed to provide statistically based estimates from which strong inferences on population size and trend can be made at state, regional, and national scales. In other cases the estimates are based on the best judgments of wildlife professionals and more emphasis should be placed on the direction of the trend rather than the actual magnitude of the estimates.

## Big Game Species

Big game are primarily comprised of large mammal species that are taken for sport or subsistence. Because of state agency convention, we also consider the wild turkey as a big game species. The species comprising big game were the first to stimulate widespread public interest in wildlife conservation and many of these species are now highlighted as wildlife management successes (Thomas 1990). For these reasons, the data from which recent historical trends could be examined were available over many states and over long time periods.

We compiled data on big game populations and harvests primarily from cooperating state wildlife agencies. We sent questionnaires developed cooperatively by the Forest Service and Natural Resources Conservation Service to state wildlife agency offices through the International Association of Fish and Wildlife Agencies. All states cooperated and responded to our request for information. The absence of data from certain states resulted from variation in the distribution of species or the lack of data for certain years. We included only those states that provided estimates for 1975 to 1990 (in 5-year intervals), and 1993 in the trend analysis. The states were also requested to provide short-term (≈10-year) and long-term (≈50-year) population projections assuming a continuation of the current management direction in each state. This resulted in state agency population and harvest projections, expressed as a percentage change from 1993 levels to the year 2000 and 2045. We projected population and harvest estimates by first calculating a weighted average percentage change for each species. The 1993 total population or harvest within each state served as the weights. Our weighted average percentage change for the year 2000 and 2045 was then applied to the 1993 estimate to project populations and harvests to those years. Population and harvest data were sufficient to analyze trends for five species or species groups including pronghorn, elk, deer, black bear, and wild turkey.

### *Big game populations*

Nationally, estimates of big game populations have increased substantially since 1975 (figure 8). Population increases have varied by species with wild turkey populations increasing by 211% over the 1975-1993 time period and pronghorn populations increasing by 56%. The estimated increase in wild turkey populations from state agency data is supported by the Breeding Bird Survey, which estimated an average annual increase in wild turkey detections of 13% from 1985 to 1996.[6] Deer populations (including both white-tailed and mule deer) have undergone the greatest increase with 8.1 million individuals being added to the total population of reporting states since 1975. Although there have been recent population gains among big game species, the rate of increase appears to have declined during the 1990s for all species except elk and black bear. In the 11 states reporting elk trends, populations have increased by more than 70%. As noted by Peek (1995), elk now occupy more suitable habitat and are more numerous than at any time since the turn of the century. The trend in black bear populations (+76% in 18 reporting states) is consistent with the findings of Vaughan and Pelton (1995) who found that of 40 states reporting estimates of black bear populations, 27 had increasing trends and only two had declining trends.

In the majority of cases, regional population trends are qualitatively consistent with national trends. Notable exceptions to this pattern include deer in the West, wild turkey in the Rocky Mountain region, and pronghorn in the South. Deer populations in the Rocky Mountain region have declined by about 11% from 1985 to 1993. Similar declines have been estimated in the Pacific Coast, with deer numbers declining by 12% from 1980 to 1993. Wild turkey populations increased steadily in the Rocky Mountains through the 1980s, but 1993 estimates indicated that the population in reporting states had declined by 30% during the early 1990s. The trend in wild turkey populations in the Rocky Mountains may be an artifact of the few number of states that reported population estimates for this species. The variability in pronghorn population in the South is also difficult to interpret since this region has relatively low populations of pronghorn that inhabit an area on the southeast periphery of the species' current range.

*[6] J.R. Sauer, pers. comm., U.S. Geological Survey, Biological Resources Division, Patuxent Wildlife Research Center, 1997. Breeding Bird Survey trend analysis. Data on file at the Rocky Mountain Research Station.*

Because population trends for many big game species are celebrated as wildlife management successes, and because big game contribute significantly to rural economies through recreational harvests, it can be difficult to accept that overabundant populations of some species in some regions can carry significant economic and ecological costs. Some of the more salient impacts caused by overabundant populations include increases in transmission of wildlife diseases to humans, direct human injury, collisions with vehicles or aircraft, and damage to agricultural crops, residential landscaping, and timber (Conover and others 1995). The overabundance of white-tailed deer populations has become so prevalent that it will likely represent one of the more important wildlife management problems during the next decade (Warren 1997).

Despite the significance of these impacts, there have been few studies that quantified their magnitude nationwide (Conover and others 1995). Romin and Bissonette (1996) examined deer-vehicle collisions and found that for 29 states reporting trends, the number of deer killed has increased an average of 210% per state from the early 1980s through the early 1990s. Much of this increase can be attributed to vehicle-collision records observed in the North (224% increase based on data from 18 states). Increases in the South (168% based on data from 4 states) and Rocky Mountains (81% based on data from 7 states)

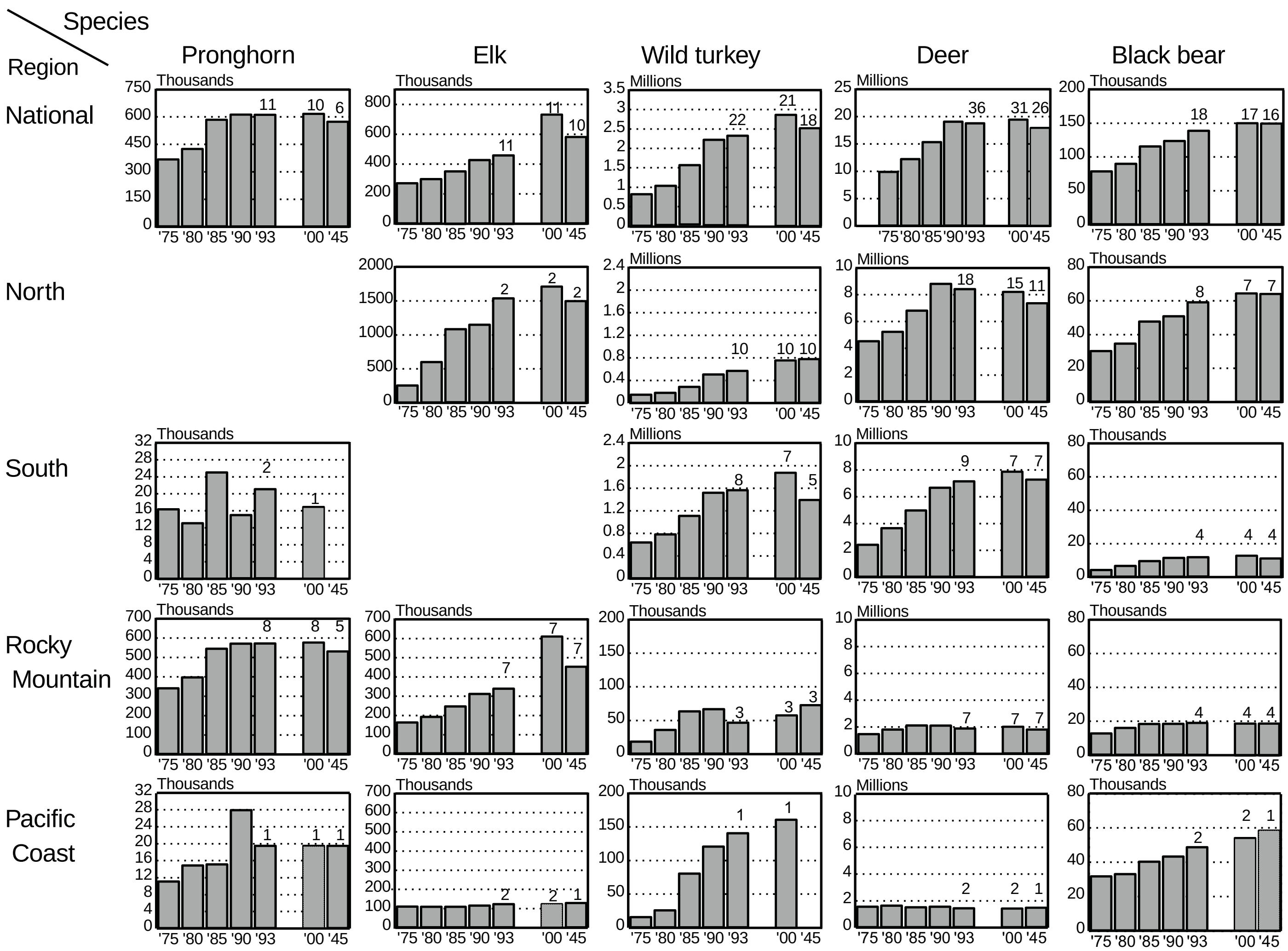


Figure 8. National and regional trends in big game populations for selected species. The number of states that provided historical estimates since 1975, short-term projections, and long-term projections are indicated above the 1993, 2000, and 2045 bar, respectively. Historical population estimates are summed across those states that provided data. Projections are based on a weighted average percentage change from 1993 to the year 2000 and 2045 for those states that provided projection estimates. The average percentage change was then applied to the 1993 population estimate in order to extrapolate a total projected population for those states that provided historical population estimates. The species group "deer" includes white-tailed deer and mule deer. (State wildlife agency data. Data on file at the Rocky Mountain Research Station.)

were much less than those reported in the North. The increase in vehicle-caused mortality of deer cannot be entirely attributed to overabundant populations. Increases in the number of roads and vehicles is also a factor contributing to high numbers of deer-vehicle collisions.

The economic impacts associated with deer-vehicle collisions is not trivial. A recent study by Conover and others (1995) found that the average repair bill per reported accident was $1,577 (in 1993 dollars), which when applied to the total number of reported accidents translated into a total annual cost of $1.1 billion. This is a conservative estimate of the total impact since many accidents are unreported and these figures do not include costs associated with human injury and deaths that also occur with deer-vehicle collisions (Conover and others 1995).

The need to control deer populations is likely reflected in both the short- and long-term projections of deer populations provided by state agency personnel (figure 8). Nationally, state agency biologists expect deer numbers to remain constant during the next 10 years and decline slightly over the next 50 years. The greatest long-term deer population decline is expected in the North where the overabundance problem is a key management concern. As with deer, state biologists expect pronghorn populations to show short-term stability, followed by slight, long-term declines. Both elk and turkey populations are expected to increase over the next 10 years followed by declines—trends that are driven primarily by the expected population patterns in the Rocky Mountain and Southern regions. Biologists from state wildlife agencies expect that bear populations will remain stable in the future in all regions except the Pacific Coast where increasing populations are anticipated.

### *Big game harvests*

Manipulation of harvests is an important management tool for achieving desired population levels of big game species (Caughley and Sinclair 1994). In addition, big game hunting expenditures ($9.7 billion in 1996) accounted for nearly 70% of the total specified expenditures for all types of hunting (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). Because of the management and economic importance of big game, harvest statistics from state agencies were much more complete than population statistics.

Over the last 20 years, harvests of common big game species have tended to parallel population trends (figure 9). The harvest rate, or the proportion of the population harvested, has varied from about 10% for black bear to nearly 20% for elk (table 9). Deer and turkey harvest trends at the national and regional levels are consistent with population trends. A total of 47 states provided deer harvest statistics with nearly 6 million deer harvested in 1993. Harvest rates for deer have averaged about 18% over the last 20 years, although the rate appears to be increasing over time. Nearly 90% of deer harvested came from the North and South regions. Turkey harvests were reported from 41 states and have increased by more than 190% since 1975. Although harvests have increased, the harvest rate has remained fairly constant (near 12%). The

Table 9--Harvest rates among big game species over time. (State wildlife agency data. Data on file at the Rocky Mountain Research Station.)

| | Pronghorn | | Elk | | Wild Turkey | | Deer | | Black Bear | |
|---|---|---|---|---|---|---|---|---|---|---|
| Year | n[a] | rate[b] | n | rate | n | rate | n | rate | n | rate |
| 1975 | 15 | 19.1 | 14 | 17.8 | 37 | 12.0 | 46 | 14.8 | 23 | 11.3 |
| 1980 | 14 | 15.4 | 12 | 18.1 | 32 | 14.9 | 45 | 14.7 | 21 | 10.0 |
| 1985 | 14 | 16.5 | 12 | 16.0 | 31 | 12.2 | 44 | 16.7 | 23 | 8.5 |
| 1990 | 11 | 12.9 | 10 | 22.4 | 25 | 11.1 | 37 | 20.5 | 21 | 10.0 |
| 1993 | 11 | 19.2 | 11 | 20.5 | 30 | 12.8 | 38 | 21.2 | 22 | 10.7 |
| Mean rate | | 16.6 | | 19.0 | | 12.5 | | 17.9 | | 10.1 |

[a]The number (n) of states reporting both population and harvest statistics for a given species in a given year.
[b]Harvest rate estimated as the proportion of the population harvested annually.

turkey population decline noted in the Rocky Mountain region does not appear to be an artifact of a small state sample as the harvest reported by 10 states within the region indicated an 8% decline during the 1990s.

Harvest trends that deviate from population trends were observed at the regional level. Elk populations in the Pacific Coast region have grown slightly from 1980 to 1990, yet harvests over that same period show declines. Similarly, bear harvests in the Rocky Mountain region have declined by 33% from 1980 to 1993, while populations have increased. Pronghorn populations in the South have been somewhat erratic since 1975, yet harvests have declined consistently and substantially (-61%) since 1980.

There are a number of reasons that may explain the divergence between big game harvest and population trends: change in regulations in response to public sentiment about harvest of certain species (e.g., black bear); reduced access to private lands; or reduced participation in hunting activities. These factors may also affect the ability of wildlife managers to control excessive populations of deer in the eastern United States (Warren 1997).

## Small Game Species

Species treated as small game typically include resident (native and introduced) game birds and mammals

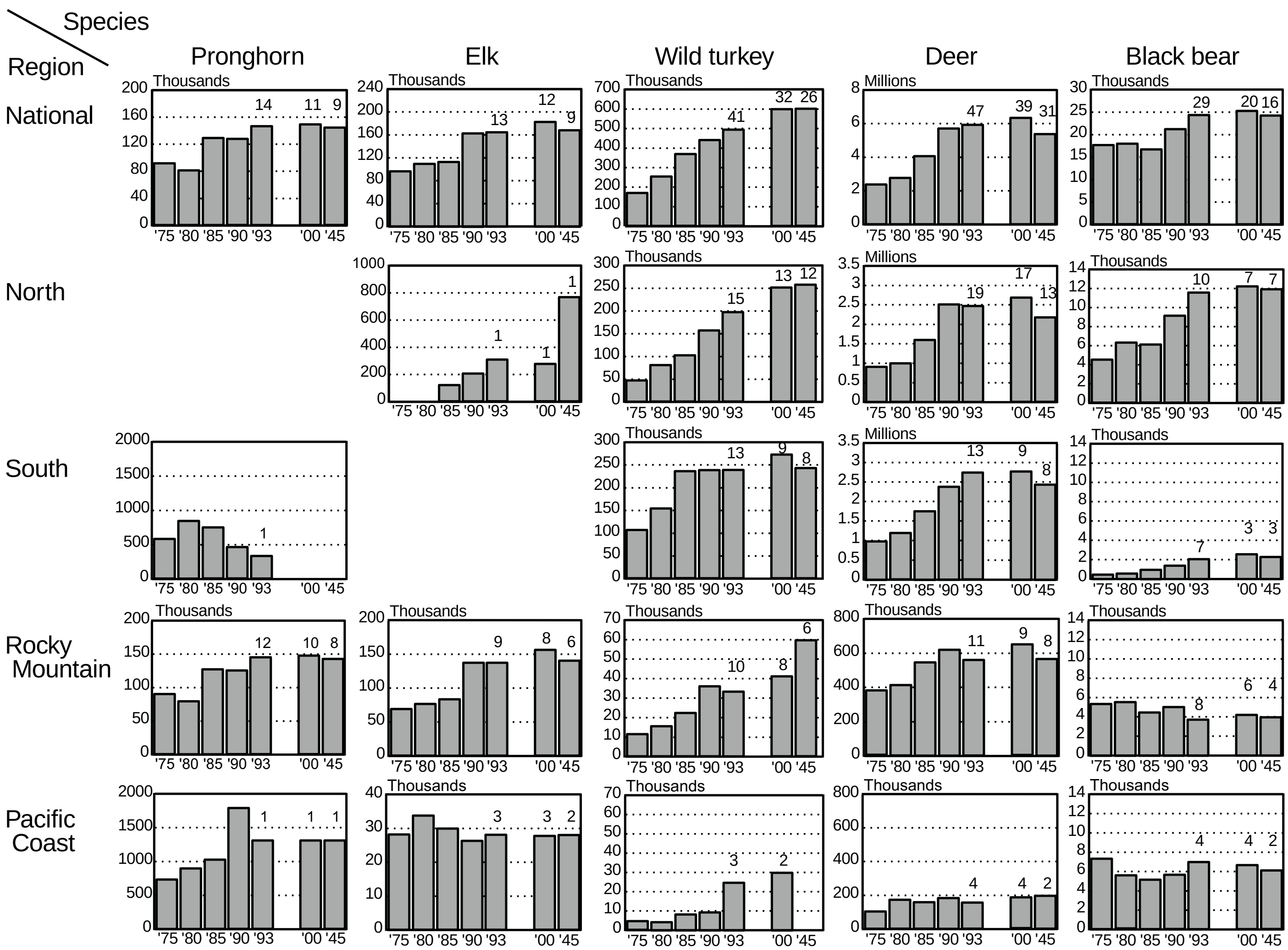


Figure 9. National and regional trends in big game harvests for selected species. The number of states that provided historical estimates since 1975, short-term projections, and long-term projections are indicated above the 1993, 2000, and 2045 bar, respectively. Historical population estimates are summed across those states that provided data. Projections are based on a weighted average percentage change from 1993 to the year 2000 and 2045 for those states that provided projection estimates. The average percentage change was then applied to the 1993 population estimate in order to extrapolate a total projected population for those states that provided historical population estimates. The species group “deer” includes white-tailed deer and mule deer. (State wildlife agency data. Data on file at the Rocky Mountain Research Station.)

that are associated with upland (forest, range, or agricultural) habitats. We compiled population and harvest statistics, as with big game, from cooperating state wildlife agencies. Questionnaires developed cooperatively by the Forest Service and Natural Resources Conservation Service were sent to state wildlife agency offices through the International Association of Fish and Wildlife Agencies. In addition, we supplemented state agency estimates of upland game bird populations with relative abundance data from the North American Breeding Bird Survey (see Droege [1990] for details about the Breeding Bird Survey).

There is some variation among state wildlife agencies as to the species that are managed as small game. For the purposes of this report, we review population and harvest statistics for quail, ring-necked pheasant, grouse, rabbit, hare, and squirrel. In those cases where state data were not distinguishable to the species level, we report trends for species groups that are taxonomically or ecologically similar. The species comprising these groups are described in table 10.

### *Small game populations*

Because few state wildlife agencies monitor small game populations, the trends that we review here should be interpreted carefully. With this caveat in mind, it appears that species associated with rangeland or agricultural habitats show evidence of declining populations, while species associated with forest habitats show mixed trends over time (figure 10).

Northern bobwhite, prairie grouse, hare, and western quail populations all show evidence of declines from the mid-1970s or early 1980s to the early 1990s. Among the five states that reported trends in bobwhite abundance, populations have declined by nearly 60% from about 28.5 million birds in 1975 to 12 million birds by 1993. Substantial declines in prairie grouse (approximately 75%) also occurred from 1975 to 1993 among three states, all within the Rocky Mountain region. Hare and western quail populations show evidence of declines, but the trends are mixed. Hare populations increased from the mid-1970s to the early 1980s after which they declined substantially (greater than 70%). Western quail also showed peak populations in the early 1980s followed by a nearly 85% decline over the next 10 years. Since 1990, however, western quail numbers have rebounded by 200% to population levels observed in the mid-1980s.

Table 10—Definition of small game species groups.

| Group name | Species |
|---|---|
| Cottontail | Species of the genus *Sylvilagus* |
| Hare | Species of the genus *Lepus* |
| Squirrel | Species of the genus *Sciurus* and red squirrel |
| Forest grouse | Ruffed grouse, spruce grouse, blue grouse |
| Prairie grouse | Greater prairie-chicken, lesser prairie-chicken, sharp-tailed grouse, sage grouse |
| Western quail | Montezuma quail, scaled quail, Gambel's quail, California quail, mountain quail |

Two species associated with agricultural habitats that have shown recent evidence of population increases are cottontail rabbit and ring-necked pheasant. Both species showed declines during the 1975 to 1985 decade, followed by population recovery. Cottontail populations are now estimated to exceed mid-1970 estimates nationwide, with the increase being attributed to population growth in the North. Pheasant populations have shown steady gains since 1985, due primarily to increases of 45% in the Rocky Mountain region. Pheasant numbers in the North remain low relative to state estimates in the early 1970s, although populations have increased slightly during the 1990s.

Forest associated species including squirrel and forest grouse show mixed trends among regions. Squirrel numbers show steady but slight gains in the North, declines in the Rocky Mountains, and declines since 1985 in the South. Forest grouse populations show a cyclical pattern, but there does appear to be evidence of a general population decline. Forest grouse population peaks and lows both decline in all regions where data were available.

Because so few states monitor small game populations (5 states per species nationwide, on average), data from the Breeding Bird Survey offer an important comparison for upland game birds (table 11). In general, trends from the Breeding Bird Survey were consistent with state agency population trends. Northern bobwhite relative abundance has declined significantly ($P < 0.05$) nationwide, in the North, and in the South. Nationwide, bobwhite have declined by 2.5% per year from 1966 to 1996, and have declined at an even greater rate since 1985 (-4.4% per year). Scaled quail show a similar pattern, declining by 3.4% per year since 1966 and declining by 5% per year since 1985. Ring-necked pheasant trends over the long-term differ dramatically from short-term trends. Over the 30-year period from 1966 to 1996, pheasants have shown significant declines of 1% per year. Since 1985, however, pheasant abundance has actually increased by 1.3% per year. The only other upland game species showing significant long-term declines is the blue grouse. Species that have shown significant population declines since 1985 include the gray and chukar partridge. No upland game bird has shown significant long-term population increases, but the California quail and greater prairie-chicken, along with ring-necked pheasant, have shown significant increases since 1985.

Factors affecting small game populations include weather, predation, and habitat quality. Given the reproductive potential of most small game species, increased predation in the absence of habitat degradation is unlikely to be responsible for declining trends, and populations are generally capable of recovering from declines attributable to inclement weather. Consequently, detectable abundance trends are likely habitat related (Edwards and others 1981). Brady and others (1998) examined the relations between land use and land cover and northern bobwhite abundance over the geographic range of the bird. They found that bobwhite abundance was generally higher in those areas less intensively used by agriculture. Areas with moderate physiographic relief and greater habitat diversity tended to support more bobwhite than did areas supporting intensive agriculture—findings that are consistent with Brennan's (1991) review of bobwhite declines.

Given the conversion of about 36 million acres of highly erodible cropland to perennial vegetation cover under the CRP, one might expect a short-term increase in bobwhite abundance in response to this program. We found no evidence for a positive bobwhite population response to the CRP. As noted earlier, bobwhite abundance has actually declined at a greater rate since the CRP (see 1985-1996 abundance trends in table 11) than over the last 30 years. The lack of a positive bobwhite population response to the CRP is likely due to the fact that bobwhite select for early stages of secondary succession characterized by perennial weeds and early woody shrub develop-

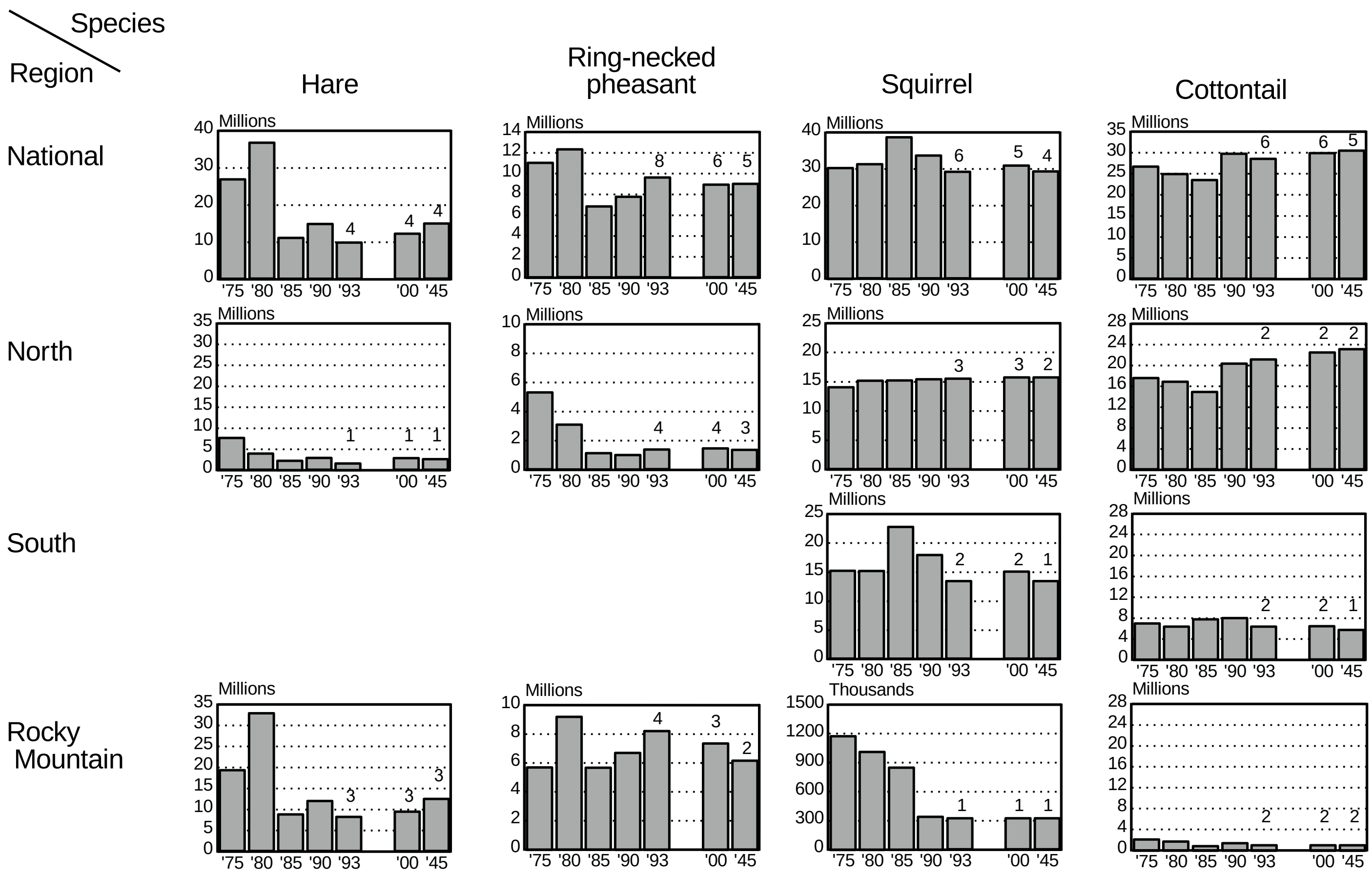


Figure 10. National and regional trends in small game populations for selected species groups. The number of states that provided to be consistent with Figures 8 and 9 historical estimates since 1975, short-term projections, and long-term projections are indicated above the 1993, 2000, and 2045 bar, respectively. Historical population estimates are summed across those states that provided data. Projections are based on a weighted average percentage change from 1993 to the year 2000 and 2045 for those states that provided projection estimates. The average percentage change was then applied to the 1993 population estimate in order to extrapolate a total projected population for those states that provided historical population estimates. (State wildlife agency data. Data on file at the Rocky Mountain Research Station.) Note that no states from the Pacific Coast region provided historical small game population estimates.

Table 11—Long-term (1966-1996) and short-term (1985-1996) population trends in upland game birds based on species observed on ≥ 15 routes (J.R. Sauer, pers. comm., U.S. Geological Survey, Biological Resources Division, Patuxent Wildlife Research Center, 1997).

| | Total U.S. | | | | North | | | | South | | | | Rocky Mountain | | | | Pacific Coast | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1966-1996 | | 1985-1996 | | 1966-1996 | | 1985-1996 | | 1966-1996 | | 1985-1996 | | 1966-1996 | | 1985-1996 | | 1966-1996 | | 1985-1996 | |
| Species | Trend[a] | P-value[b] | Trend | P-value | Trend | P-value | Trend | P-value | Trend | P-value | Trend | P-value | Trend | P-value | Trend | P-value | Trend | P-value | Trend | P-value |
| Gray partridge | 0.8 | 0.55 | -6.1 | <.01 | 1.3 | 0.58 | -7.9 | <.01 | | | | | 0.4 | 0.76 | -5.6 | 0.01 | | | | |
| Chukar | -5.8 | 0.10 | -9.4 | 0.04 | | | | | | | | | | | | | -6.2 | 0.08 | -8.5 | 0.11 |
| Northern bobwhite | -2.5 | <.01 | -4.4 | <.01 | -2.8 | <.01 | -2.6 | <.01 | -2.6 | <.01 | -5.6 | <.01 | -0.8 | 0.17 | 0.1 | 0.90 | | | | |
| Mountain quail | 1.5 | 0.14 | 1.6 | 0.39 | | | | | | | | | | | | | 1.5 | 0.15 | 1.6 | 0.38 |
| Scaled quail | -3.4 | <.01 | -5.0 | <.01 | | | | | -3.9 | <.01 | -6.8 | <.01 | -2.1 | 0.07 | -3.7 | 0.16 | | | | |
| California quail | 0.9 | 0.27 | 4.9 | <.01 | | | | | | | | | | | | | 0.8 | 0.29 | 4.7 | <.01 |
| Gambel's quail | 1.0 | 0.14 | -0.1 | 0.97 | | | | | | | | | 1.3 | 0.09 | 0.5 | 0.83 | -0.9 | 0.78 | -2.4 | 0.61 |
| Blue grouse | -3.4 | 0.02 | -1.9 | 0.62 | | | | | | | | | | | | | -3.3 | 0.05 | -0.7 | 0.87 |
| Ruffed grouse | -0.8 | 0.56 | 2.1 | 0.64 | 0.1 | 0.97 | 6.6 | 0.15 | | | | | -2.3 | 0.60 | -14.4 | 0.19 | | | | |
| Greater prairie-chicken | 5.0 | 0.50 | 20.2 | <.01 | | | | | | | | | | | | | | | | |
| Sharp-tailed grouse | 3.0 | 0.14 | 2.2 | 0.67 | | | | | | | | | 3.2 | 0.15 | 2.5 | 0.64 | | | | |
| Sage grouse | 1.7 | 0.55 | -17.0 | 0.09 | | | | | | | | | 1.6 | 0.57 | -16.9 | 0.10 | | | | |
| Ring-necked pheasant | -1.0 | 0.02 | 1.3 | 0.07 | -2.0 | 0.02 | 1.7 | 0.02 | 0.6 | 1.76 | -4.0 | 0.02 | -0.2 | 0.69 | 1.4 | 0.28 | -0.9 | 0.38 | 0.1 | 0.94 |

[a] *Trends are estimated using the route-regression method (Geissler and Sauer 1990). Regional trends are estimated as a weighted average of trends over individual routes using the estimating equations estimator (Link and Sauer 1994).*

[b] *The probability that the trend is equal to zero. Trends with P-values > 0.10 are not considered to be significantly different from zero.*

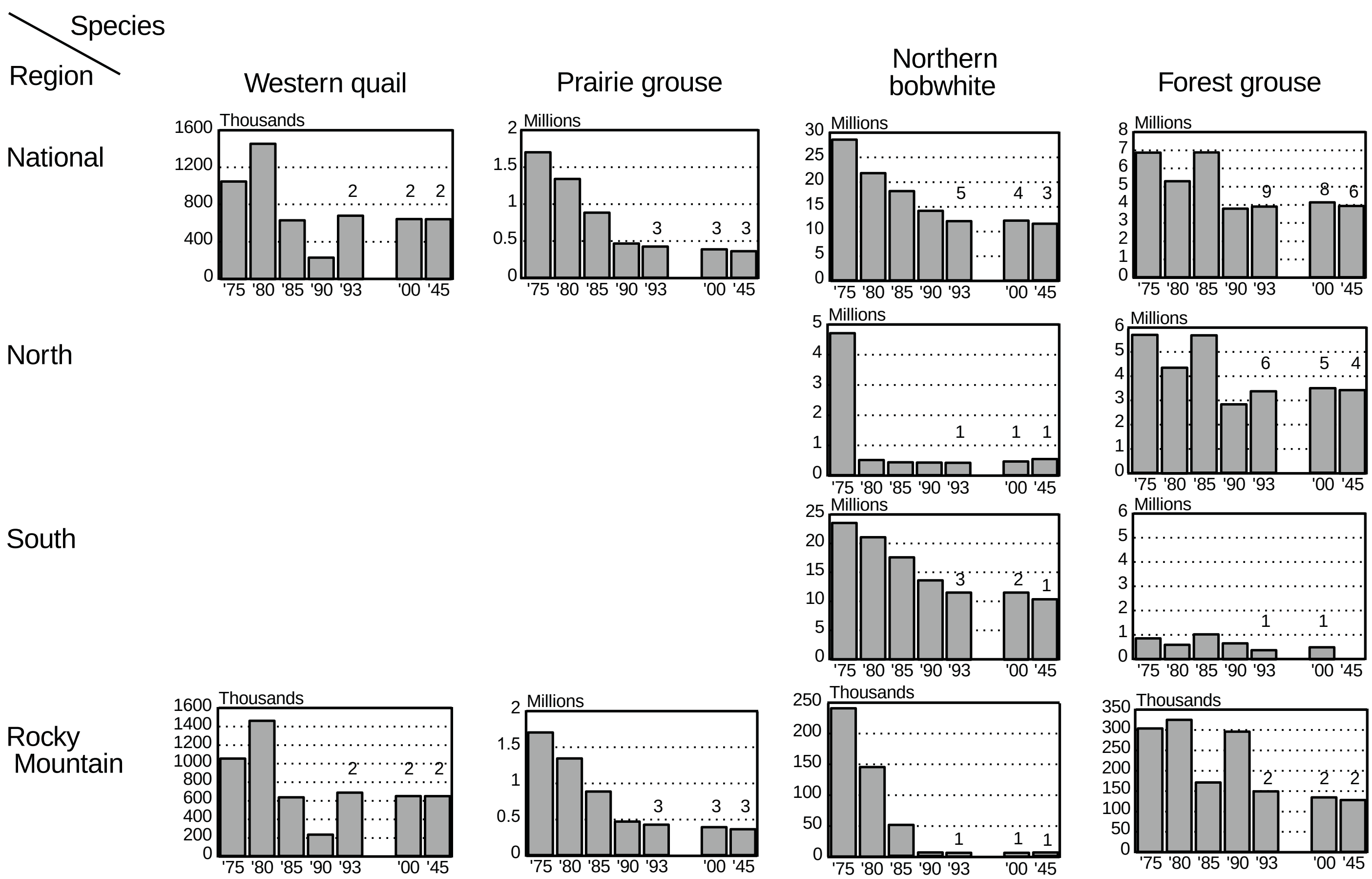


Figure 10. (Cont'd.)

ment (Klimstra and Roseberry 1975). Habitats with such characteristics have yet to develop on most CRP lands. The recent population recovery of pheasants does appear to be related to the CRP. For nesting, pheasants select for the type of grassland cover that is found in abundance on most CRP plots. Both the Breeding Bird Survey and state agency results reviewed here provide evidence that the CRP has had a regional and national affect on recent increases in ring-necked pheasant abundance.

State agency projections of small game populations indicate only minor changes in population (figure 10). For most species, expected future changes in small game abundance are less than 10% from 1993 estimates. Forest grouse species, western quail, and squirrel populations are expected to remain stable in the future. Hare and cottontail populations are expected to increase over the next 50 years. State biologists expect prairie grouse, northern bobwhite, and ring-necked pheasant populations to decline. Given recent historical trends (i.e., since 1990), the projected decline in pheasant numbers is somewhat surprising and may reflect biologists' perceptions that CRP lands will be brought back into agricultural production in the future.

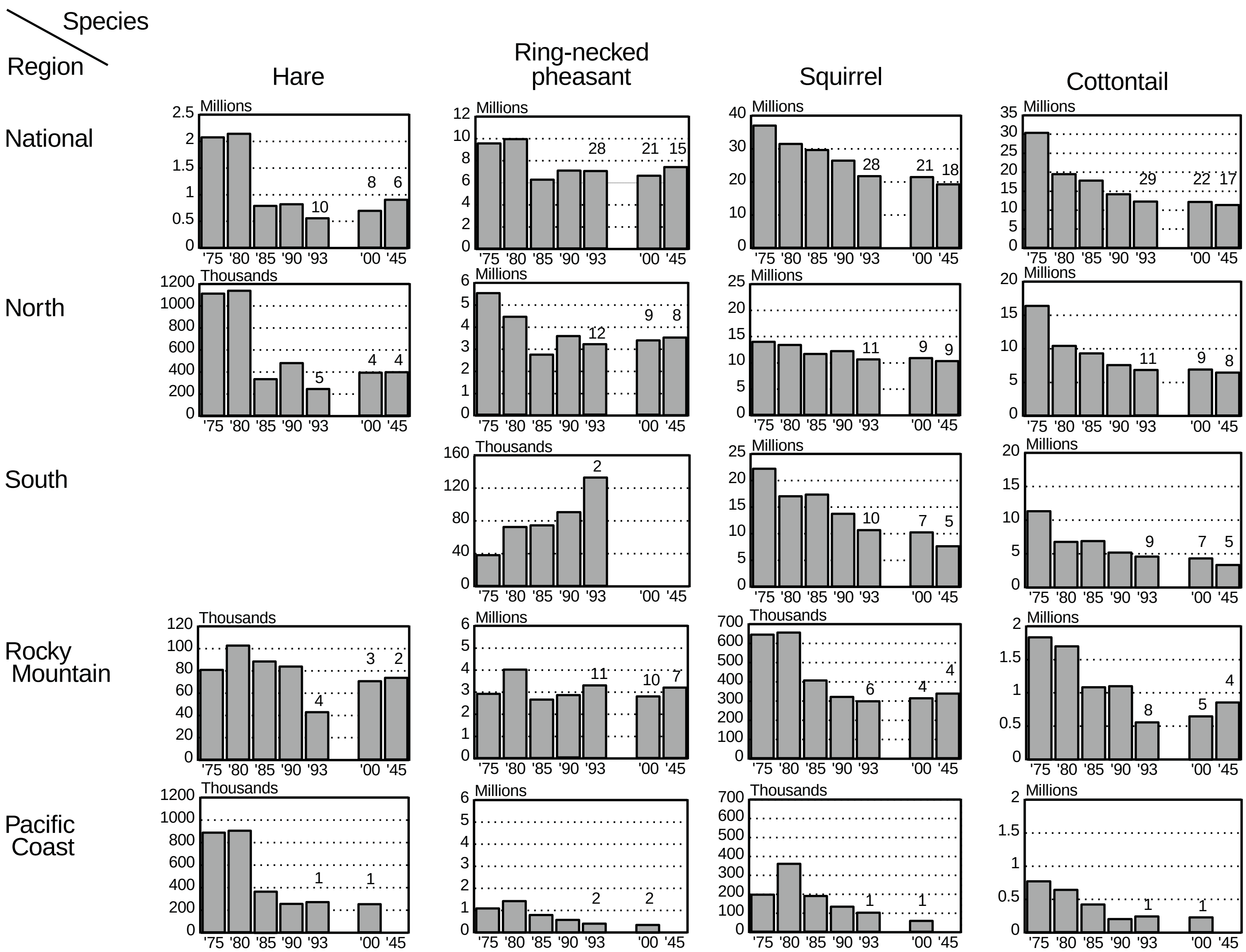


Figure 11. National and regional trends in small game harvests for selected species groups. The number of states that provided to be consistent with figures 8 and 9 historical estimates since 1975, short-term projections, and long-term projections are indicated above the 1993, 2000, and 2045 bar, respectively. Historical population estimates are summed across those states that provided data. Projections are based on a weighted average percentage change from 1993 to the year 2000 and 2045 for those states that provided projection estimates. The average percentage change was then applied to the 1993 population estimate in order to extrapolate a total projected population for those states that provided historical population estimates (State wildlife agency data. Data on file at the Rocky Mountain Research Station).

### *Small game harvests*

Many more states were able to provide historical trends in small game harvest (21 states per species nationwide, on average) compared to population estimates. Based on data from states that provided both population and harvest estimates, about 15 to 20% of the small game population is harvested each year—ranging from a low of about 3% for hare, to a high of 31% for ring-necked pheasant (table 12). Because so few states provided both population and harvest estimates, the relation between harvest and population trends is difficult to estimate. Harvest trends that deviate from population trends could: (1) reflect a more representative sample of states from which to estimate harvest, or (2) may reflect changes in the number of hunters that are actually pursuing small game for reasons independent of population status (e.g., limited access to private land that permits hunting). For these reasons, comparisons of harvest and population trends must be interpreted cautiously.

Northern bobwhite, ring-necked pheasant, hare, and forest grouse had harvest trends that were qualitatively similar to population trends (figure 11). Among the 25 states providing harvest data on northern bobwhite, total annual harvest has declined by more than 50% from 1975 to 1993. The greatest decline occurred in the South where the number of bobwhite annually harvested in nine states declined from nearly 17 million in 1975 to just over 6 million in 1993. There is concern that if current population trends continue, the opportunity to hunt bobwhite across much of the bird's range could be lost in the near future (Brennan 1991). Although pheasant harvests have declined in the last 20 years, the trend since 1985 shows increasing numbers of pheasants being taken by small game hunters. This pattern is particularly evident in the Rocky Mountain region where CRP lands are concentrated. Although

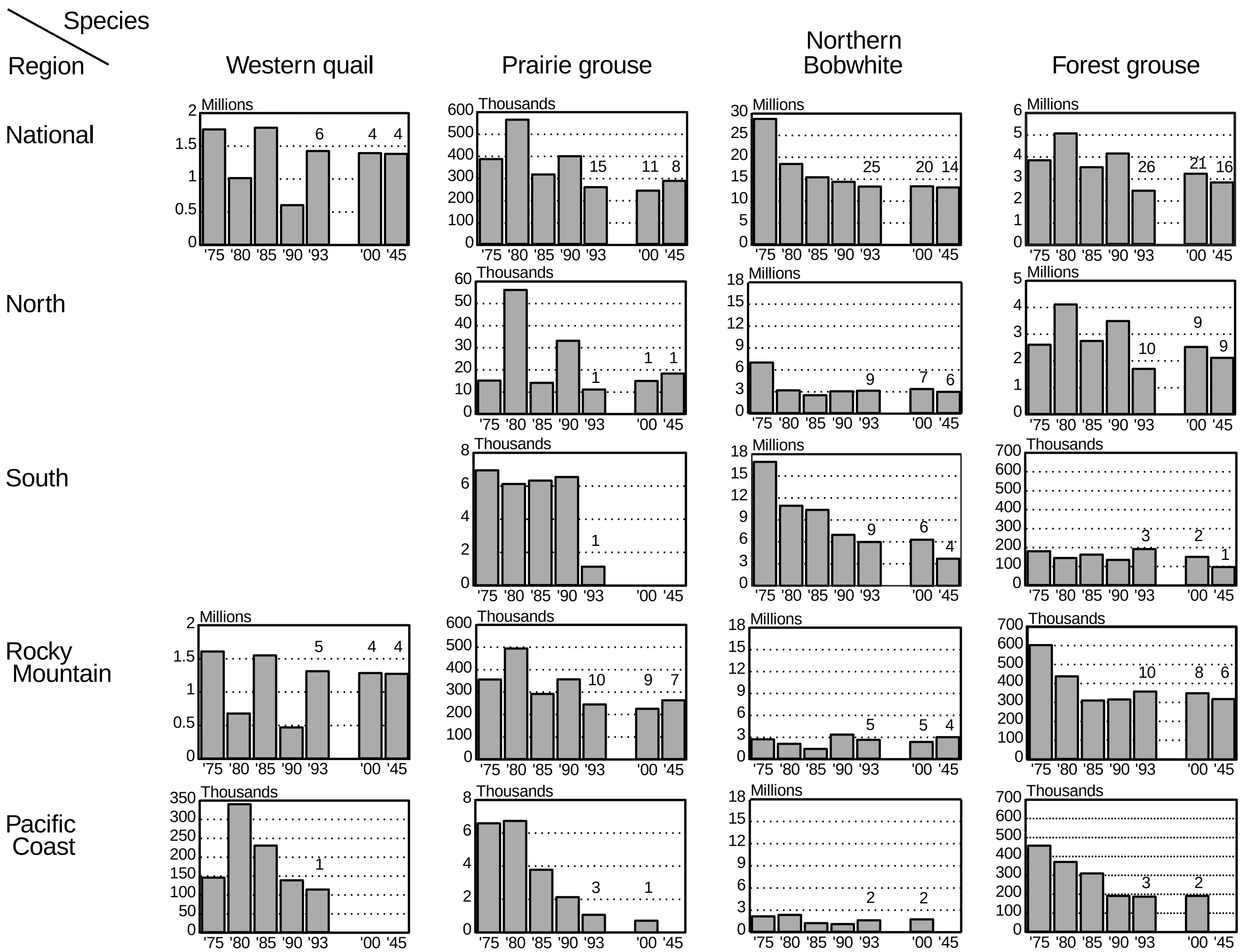


Figure 11. (Cont'd.)

relatively few pheasants are harvested in the South, harvests have increased consistently since 1975. Conversely, pheasant harvests in the Pacific Coast have declined by 70% since 1980. Hare harvests declined abruptly between 1980 and 1985, a pattern that was observed in both the North and Pacific Coast regions. Forest grouse harvests paralleled the cyclical pattern that was observed in the population data—a pattern that is driven by harvest trends in the North. The Pacific Coast and Rocky Mountain regions show consistent declines in forest grouse harvests with the exception of the most recent period in the Rocky Mountain region when harvests have increased slightly.

Harvest and population trends for cottontail, squirrel, prairie grouse, and western quail differed (compare figures 10 and 11). While cottontail and squirrel populations were somewhat stable over the last 20 years, annual harvests of these species groups have declined nationally and regionally. Over this period, cottontail and squirrel harvests have declined by nearly 60 and 40%, respectively. In both species groups, the national harvest trend was dominated by the pattern that has occurred in the eastern regions. Prairie grouse and western quail harvests also differ from their respective population trends by expressing a cyclical pattern that was not evident in the abundance estimates. The cycles in the harvest data notwithstanding, harvests of these species groups do appear to be declining over the last 20 years.

There are some notable differences between harvest and population projections among small game species as well (compare figures 10 and 11). Whereas state agencies expect pheasant populations to decline in the long-term, pheasant harvests are expected to increase over the same projection period. A similar difference was observed with prairie grouse harvests, which state agencies expect to increase over the next 50 years while populations are projected to decline. Conversely, cottontail harvests are expected to decline nationally even though populations are expected to remain stable or increase slightly.

## Migratory Game Bird Species

Federal authority to conserve and manage migratory birds is rooted in a series of statutes that were passed in the early 1900s (Migratory Bird Act of 1913, Migratory Bird Treaty Act of 1918, Migratory Bird Conservation Act of 1929). These early international agreements were with Great Britain on behalf of Canada, with subsequent treaties established with Mexico (1936), Japan (1972), and the Soviet Union (1976) (Chandler 1985). The primary objective of these treaties is the protection and conservation of migratory bird populations. Harvesting of migratory birds in a manner that is consistent with protection is a secondary objective. The long history of migratory bird management in North America that was initiated by these historic agreements has resulted in the development of, perhaps, the premier monitoring system for continentally distributed species in the world (Nichols and others 1995). Consequently, population and harvest estimates are among the most extensive (in time and geographically) and the most reliable for resource planning.

"Migratory game birds" refers to a collection of species that include waterfowl (ducks, geese, and swans) and webless migratory species including mourning dove and woodcock. Population and harvest trends come primarily from annual reports published by the U.S. Fish and Wildlife Service and from the North American Waterfowl Plan (U.S. Department of the Interior, Environment Canada, and Secretaria de Desarrollo Social México 1994).

Table 12--Harvest rate among small game species over time. (State wildlife agency data. Data on file at the Rocky Mountain Research Station.)

| | Hare | | Ring-necked pheasant | | Squirrel | | Cottontail | | Western quail | | Prairie grouse | | Northern bobwhite | | Forest grouse | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | n[a] | rate[b] | n | rate | n | rate | n | rate | n | rate | n | rate | n | rate | n | rate |
| 1975 | 8 | 5.1 | 22 | 34.9 | 16 | 20.4 | 17 | 17.6 | 3 | 9.0 | 11 | 13.8 | 20 | 21.4 | 17 | 18.9 |
| 1980 | 4 | 1.5 | 9 | 29.6 | 8 | 27.9 | 7 | 17.3 | 2 | 10.3 | 2 | 13.1 | 4 | 18.5 | 8 | 38.7 |
| 1985 | 5 | 2.4 | 12 | 37.0 | 12 | 22.1 | 11 | 21.0 | 2 | 10.2 | 5 | 16.9 | 9 | 19.4 | 10 | 25.5 |
| 1990 | 3 | 2.0 | 7 | 26.1 | 7 | 23.0 | 8 | 11.5 | 2 | 12.5 | 2 | 12.0 | 6 | 20.8 | 8 | 45.7 |
| 1993 | 4 | 1.8 | 10 | 25.7 | 7 | 19.6 | 8 | 10.6 | 3 | 11.7 | 4 | 14.0 | 7 | 23.0 | 10 | 29.3 |
| Mean rate | | 2.8 | | 31.0 | | 22.1 | | 16.2 | | 10.3 | | 14.0 | | 20.7 | | 27.8 |

[a]The number (n) of states reporting both population and harvest for a given species in a given year.
[b]Harvest rate estimated as the proportion of the population harvested annually.

### *Duck populations*

The status of duck populations varies among species, but trends do suggest substantial increases in duck abundance for many species over recent years (figure 12). The 1996 estimate of 37.5 million ducks is 16% higher than the long-term average (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996) and is approaching population levels that have not been observed since the early 1970s and late 1950s. These trends contrast sharply with those reviewed in the previous RPA Assessment (Flather and Hoekstra 1989), which showed declining trends overall and a record low population of ducks estimated in 1985.

The recent increase in duck populations is likely a function of many factors and it has been difficult to draw strong conclusions regarding cause-and-effect because of the retrospective nature of past studies (Nichols and others 1995). One factor that appears to play an important role in explaining yearly variation in duck numbers is the number of small wetland habitats early in the breeding season. The increasing trend in wetland habitats observed during the 1990s due to unusually wet conditions corresponds to a period of rapid population growth in ducks (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996). CRP acreage in the northern Great Plains of the U.S. also provided additional nesting cover.

Populations of the 10 principal duck species indicated that many species have increased to the point where they exceed their long-term average (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996). Mallard, green-winged teal, and redhead had breeding populations well above their long-term means in 1996. Furthermore, record numbers of blue-winged teal, gadwall, northern shoveler, and canvasback were also observed in 1996. Species remaining below their long-term mean include the American wigeon, scaup, and northern pintail. Northern pintail breeding populations are particularly troubling as they have declined from a high of about 10 million birds in the mid-1950s to 2.7 million birds in 1996. Because pintails have early nest initiation dates, Beauchamp and others (1996) have hypothesized that pintails, when compared to other species, may be disproportionately exposed to increased predation and nest destruction from farm practices, especially early spring plowing.

For the purposes of planning, duck population projections are discussed in terms of long-term management goals rather than predicted population levels. The North American Waterfowl Plan (U.S. Department of the Interior, Environment Canada, and Secretaria de Desarrollo Social México 1994) specifies an overall goal of 62 million

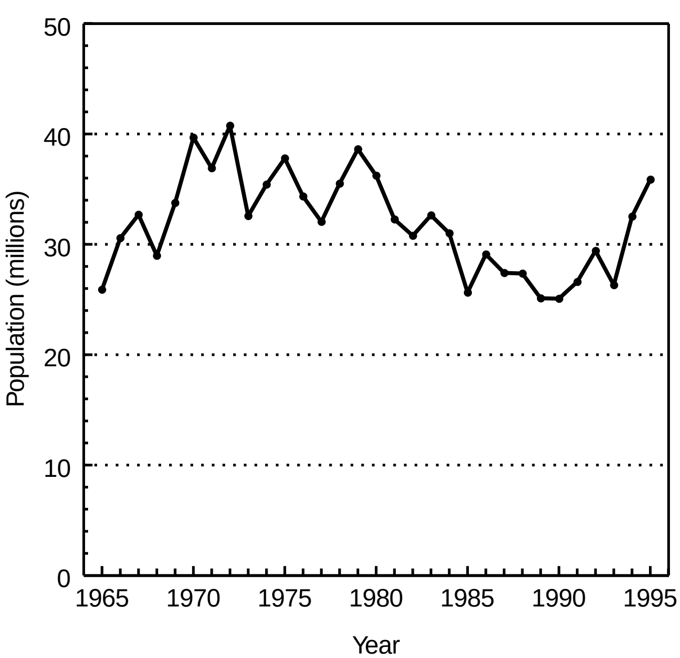


Figure 12. Trends in duck populations from 1965 to 1996 (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996).

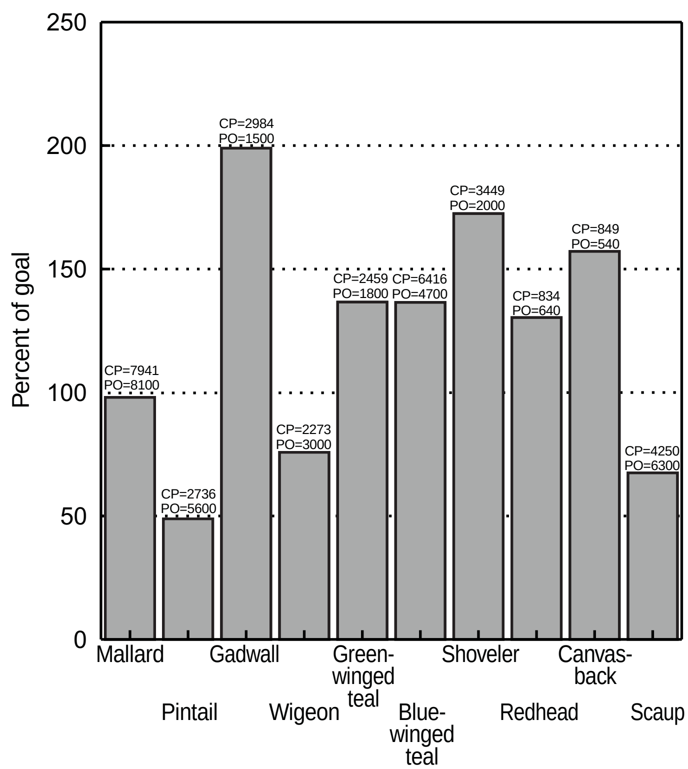


Figure 13. The relation between current (1996) duck populations (CP) and population objectives (PO) for the year 2001 for the 10 principal duck species established by the North American Waterfowl Plan. Population estimates at the top of each bar are in thousands of ducks. The y-axis reflects the degree (measured as a percentage) to which current populations meet population objectives (U.S. Department of the Interior, Environment Canada, and Secretaria de Desarrollo Social México 1994; USDI Fish and Wildlife Service and Canadian Wildlife Service 1996).

breeding ducks under average environmental conditions. The baseline reference period for establishing this goal was the mean estimated during the 1970-1979 period based on the observation that duck numbers during the 1970s generally met the needs of all users. Based on the goals established for each species, the 1996 population estimates exceeded those goals for 6 of the 10 principal duck species (figure 13 on previous page). Only the pintail, American wigeon, and scaup remain below population objectives. Mallard populations essentially meet the breeding population objectives for this species.

### *Duck harvests*

Management of duck populations through the annual establishment of harvest regulations has been a key federal activity directed at ensuring healthy duck populations. Early in the history of migratory game bird management, surveys and band returns indicated that waterfowl populations followed four major flyways as they migrated from their breeding to wintering habitats (Lincoln 1935). The four flyways are identified by the major north-south water courses and include the Atlantic, Mississippi, Central, and Pacific flyways. Because waterfowl have long been managed according to flyways, harvest trends will be reviewed according to administrative flyway boundaries rather than by RPA Assessment regions.

National duck harvests since the early 1960s do track duck population estimates closely (figure 14). After increasing substantially in the 1960s, harvests remained relatively stable through the 1970s when an average of 13.5 million ducks were harvested annually. During the 1980s, harvests underwent a near monotonic decline until 1988 when the total duck harvest was 4.7 million birds.

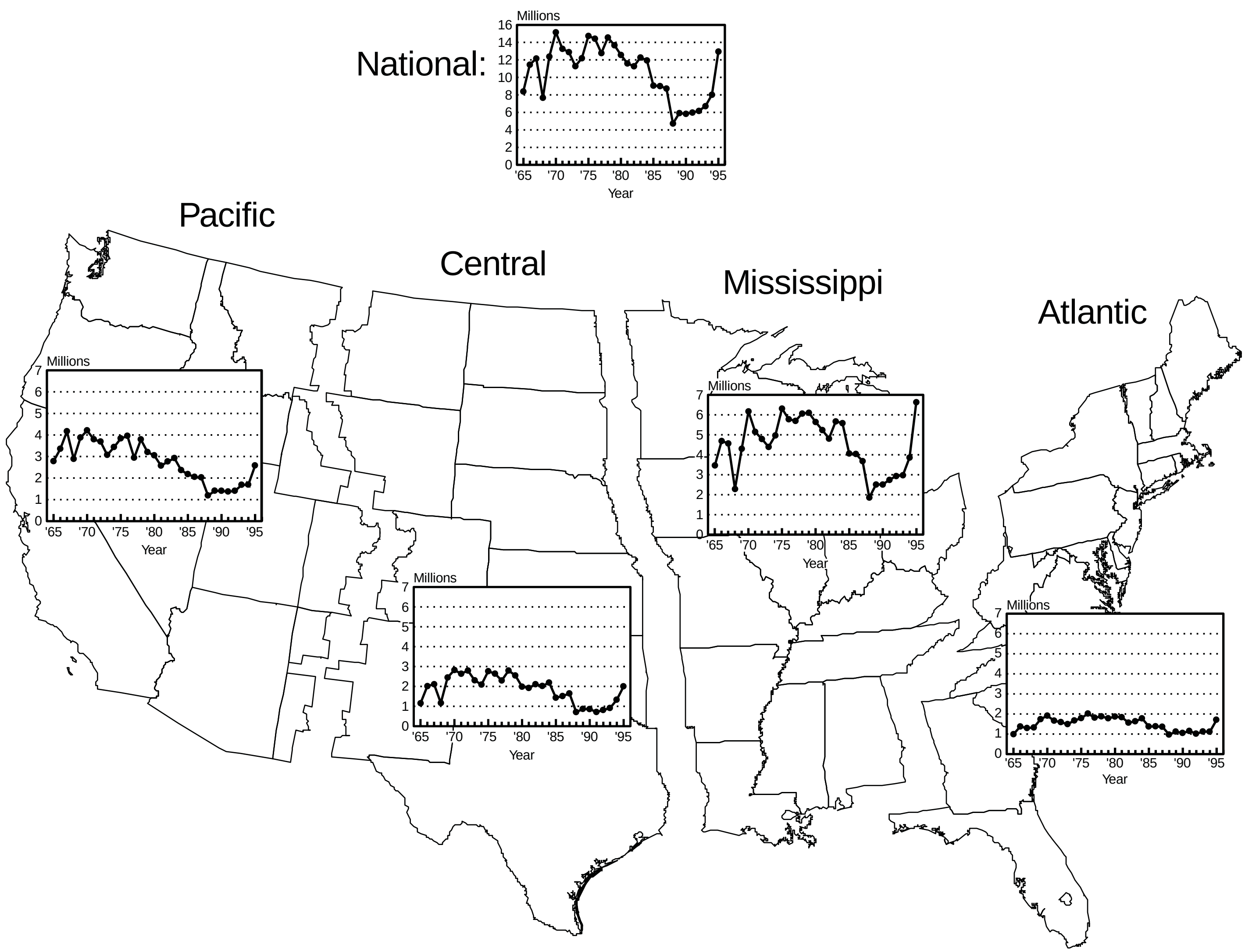


Figure 14. Trends in duck harvests from 1965 to 1995 by administrative flyway (P. Padding, pers. comm., USDI Fish and Wildlife Service, Office of Migratory Bird Management, 1996).

Now that population numbers are more favorable, harvests have increased to levels observed during the 1970s. The 62% increase in duck harvests from 1994 to 1995 was the greatest annual increase on record.

Duck harvests by flyway show no qualitative deviation from the noted national trends. Over the last 25 years, 41% of the national harvest was taken in the Mississippi flyway, followed by the Pacific (27%), Central (17%), and Atlantic (15%). All four flyways show a pattern of high harvests during the 1970s, followed by substantial declines through much of the 1980s, and substantial harvest increases during the 1990s. Duck harvests in the Mississippi flyway had the greatest increases from 1988 to 1995 (nearly a 260% gain), with 1995 being a record duck harvest of 6.6 million birds.

## *Goose and swan populations*

Species comprising this group of migratory game birds include Canada geese, snow geese, Ross' geese, greater white-fronted geese, brant, and tundra swans. Because most geese and swans breed outside of the area that has been traditionally surveyed to estimate breeding waterfowl populations, most population estimates are derived from surveys conducted on migration and wintering areas (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996). Of the 29 populations of geese and swans that are monitored with these surveys, 11 have shown significant increasing trends in overall numbers over the last 10 years, 12 have had stable populations, two populations have declined, and four have unreliable trend data. Consequently, nearly 80% of goose and swan populations are increasing or stable. The two declining populations are both Canada geese—the Atlantic population and the dusky Canada goose. All other populations of Canada geese have increasing or stable trends (figure 15), and Canada geese are likely more abundant now than they ever have been in the past (Rusch and others 1995). Because of the high variability in tundra swan population estimates, both the Eastern and Western populations show no detectable trend (figure 16). During the 1990s, tundra swan population estimates averaged 88,000 for Eastern populations and 67,000 for Western populations.

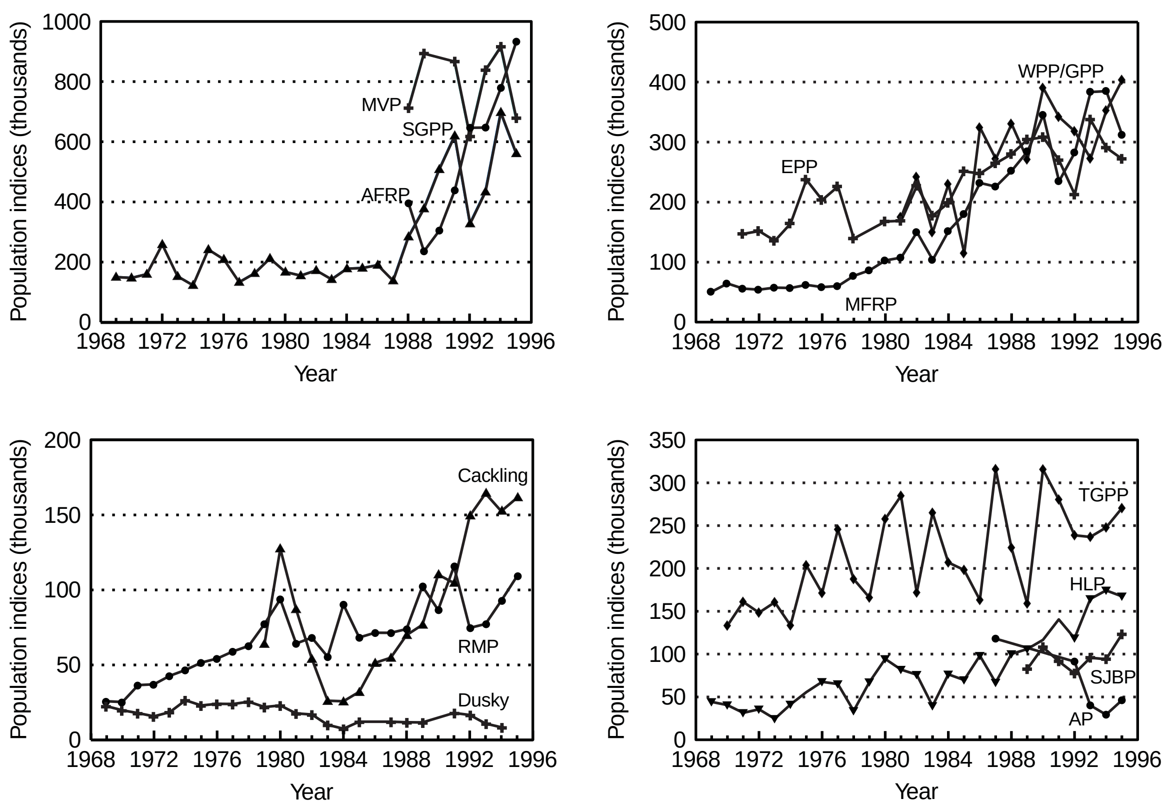


Figure 15. Trends in Canada goose population indices of abundance (in thousands) from 1969 to 1995. Acronyms are defined as follows: MVP=Mississippi Valley Population, SGPP=Shortgrass Prairie Population, AFRP=Atlantic Flyway Resident Population, EPP=Eastern Prairie Population, WPP/GPP=Western Prairie Population/Great Plains Population, MFRP=Mississippi Flyway Resident Population, Cackling=Cackling Canada goose, RMP=Rocky Mountain Population, Dusky=Dusky Canada goose, TGPP=Tallgrass Prairie Population, HLP=Hi-line Population, AP=Atlantic Population, SJBP=Southern James Bay Population (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996).

Of the 29 goose and swan populations that are monitored, 28 have population goals specified in the North American Waterfowl Management Plan (U.S. Department of the Interior, Environment Canada, and Secretaria de Desarrollo Social México 1994). A total of 12 populations are at, or exceed, the population goals specified in the plan, and several populations exceed the goals by greater than 50% (Shortgrass Prairie and Rocky Mountain populations of Canada geese, Midcontinent population of lesser snow geese, Ross' goose, and the Eastern Midcontinent population of White-fronted geese).

Recent population increases in many goose populations, however, cannot always be interpreted as a favorable resource situation. As reviewed by Ankney (1996), there is growing evidence that some populations exceed that which is acceptable on biological, esthetic, or economic criteria. One of the factors contributing to these burgeoning populations is agricultural activities that provide wintering geese with an abundance of waste grain in proximity to human-caused open water habitats (Rusch and others 1995). Increased survivorship over the winter leads to increasing numbers of birds on breeding colonies, which is causing severe breeding habitat degradation in some areas (Ankney 1996).

### *Goose and swan harvests*

Record numbers of geese have been harvested nationally for three consecutive years starting in 1993, reaching 2.4 million birds by 1995 (figure 17, see facing page). The national trend is driven primarily by the pattern of goose harvests in the Central and Mississippi Flyways, which accounted for 80% of the birds harvested in 1995. Because of the dominating influence of harvests in the interior of the country, the national trend masks the pattern of harvests in the Pacific and Atlantic Flyways. Harvests of geese in the Pacific Flyway have remained fairly stable since the early 1970s. Conversely, goose harvests in the Atlantic Flyway have been declining since the mid-1980s. After reaching a peak harvest of about 550,000 birds in 1983, goose harvests in the Atlantic Flyway have declined to nearly 180,000 birds in 1995. The trend in the Atlantic Flyway harvests is a significant departure from the trend observed in the last RPA Assessment (Flather and Hoekstra 1989), which indicated substantial increases in goose harvests through the mid-1980s.

The harvest trends for tundra swan (figure 18) show a pattern that is not expected given the population status of this species. Both the eastern and western populations of tundra swans have shown relatively stable, although variable, abundance over the last decade and their populations either exceed or meet population goals set in the North American Waterfowl Plan. Despite this stability, the harvest of eastern tundra swans has increased substantially (from 34 birds in 1983 to more than 5,000 birds in 1994) from the mid-1980s through the 1990s. Harvests of the western population have declined by about 50% when one compares a 3-year average harvest centered on 1980 and 1994.

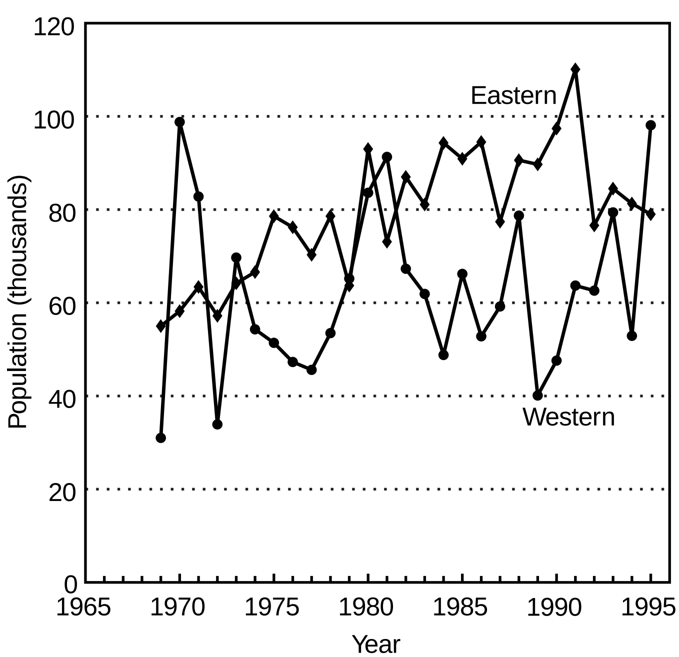


Figure 16. Trends in tundra swan populations from 1969 to 1995 for eastern and western populations (USDI Fish and Wildlife Service and Canadian Wildlife Service 1996).

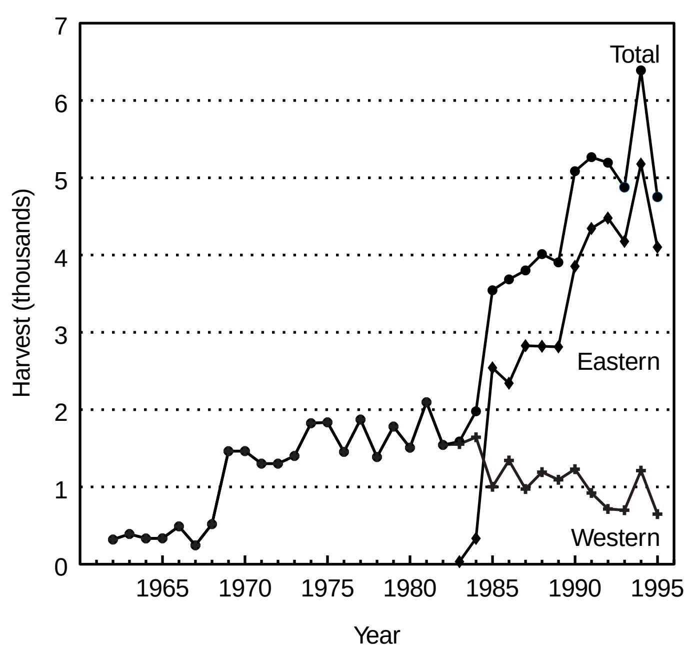


Figure 18. Harvest trends from 1962 to 1995 for eastern and western populations of tundra swans (D. Sharp, pers. comm., USDI Fish and Wildlife Service, Office of Migratory Bird Management, 1998).

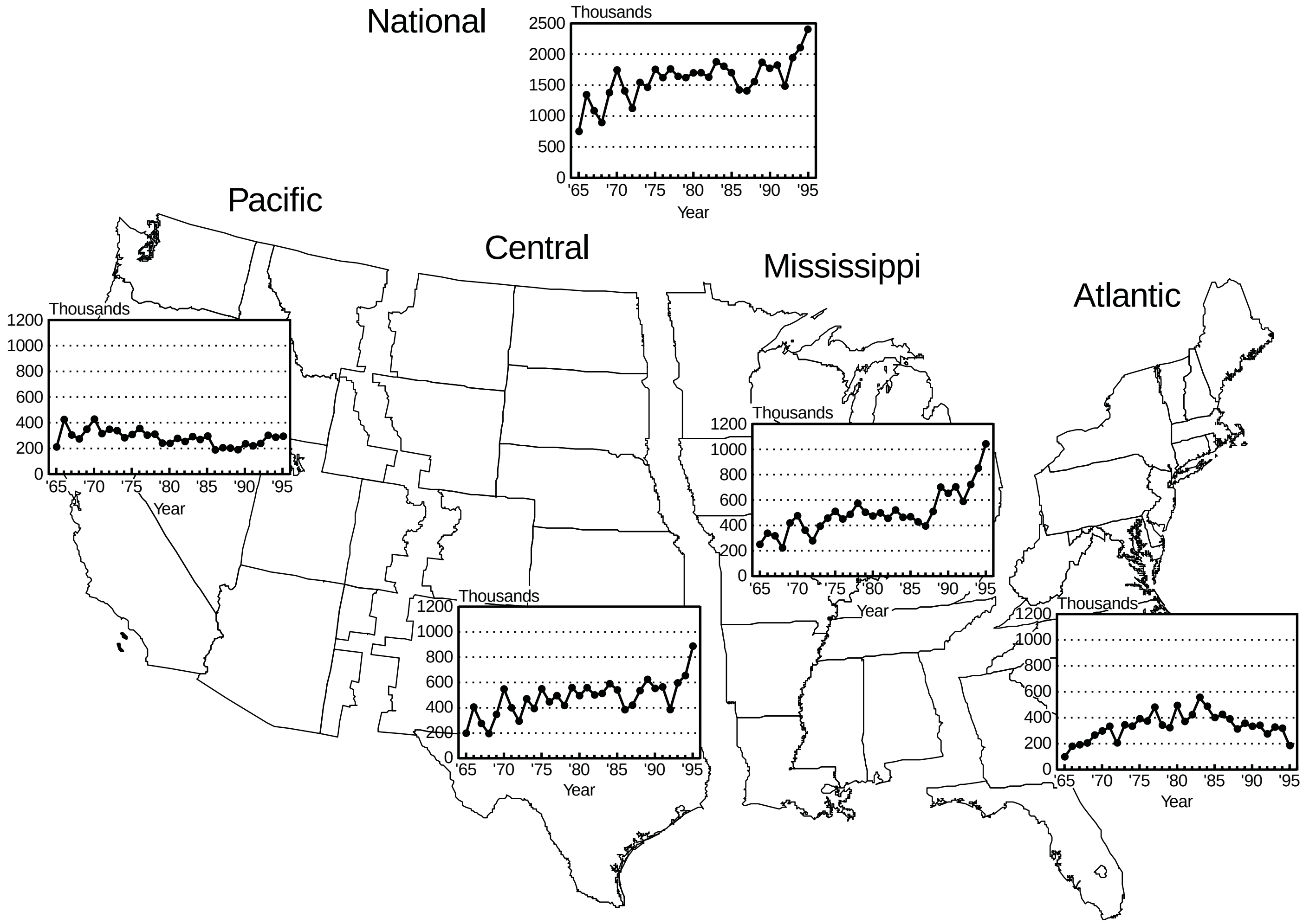


Figure 17. Trends in goose harvest from 1965 to 1995 by flyway (P. Padding, pers. comm., USDI Fish and Wildlife Service, Office of Migratory Bird Management, 1996).

### *Dove and woodcock populations*

Both mourning dove and American woodcock abundance is monitored through call-count surveys that provide an annual index of population size (Dolton and Smith 1997; Bruggink 1997). National trends in population indices for both species show evidence of declines, although the magnitude of the decline appears to be much less for mourning dove (figure 19) than woodcock (figure 20). This pattern is confirmed by the Breeding Bird Survey data which indicates that doves are declining annually at a rate of 0.3% compared to a 3.2% decline for woodcock over the 1966 to 1996 period.[7]

[7] *J.R. Sauer, pers. comm., U.S. Geological Survey, Biological Resources Division, Patuxent Wildlife Research Center, 1997. Breeding Bird Survey trend analysis. Data on file at the Rocky Mountain Research Station.*

Mourning dove call count data has shown evidence of declining populations during the last 10 years in the Eastern and Central management units, with long-term (over the last 30 years) declines being detected in the Central and Western units (figure 19). Declines in the West have been estimated at 2.4% annually since 1966 (Dolton and Smith 1997). Although doves are adaptable to human dominated lands (Dolton 1995), intensification of some agricultural practices may be negatively impacting the breeding populations throughout much of the bird's range. Although agricultural land in the Eastern management unit is positively related to dove populations (because much of the East is forested and agriculture creates the edge habitats and food sources selected by doves), increased uses of herbicides and insecticides (see figure 6) along with shifts in the specific types of crops planted may be related to observed dove declines in the East (Martin and Sauer 1993). In the Central management unit, the number of farms (a factor positively associated

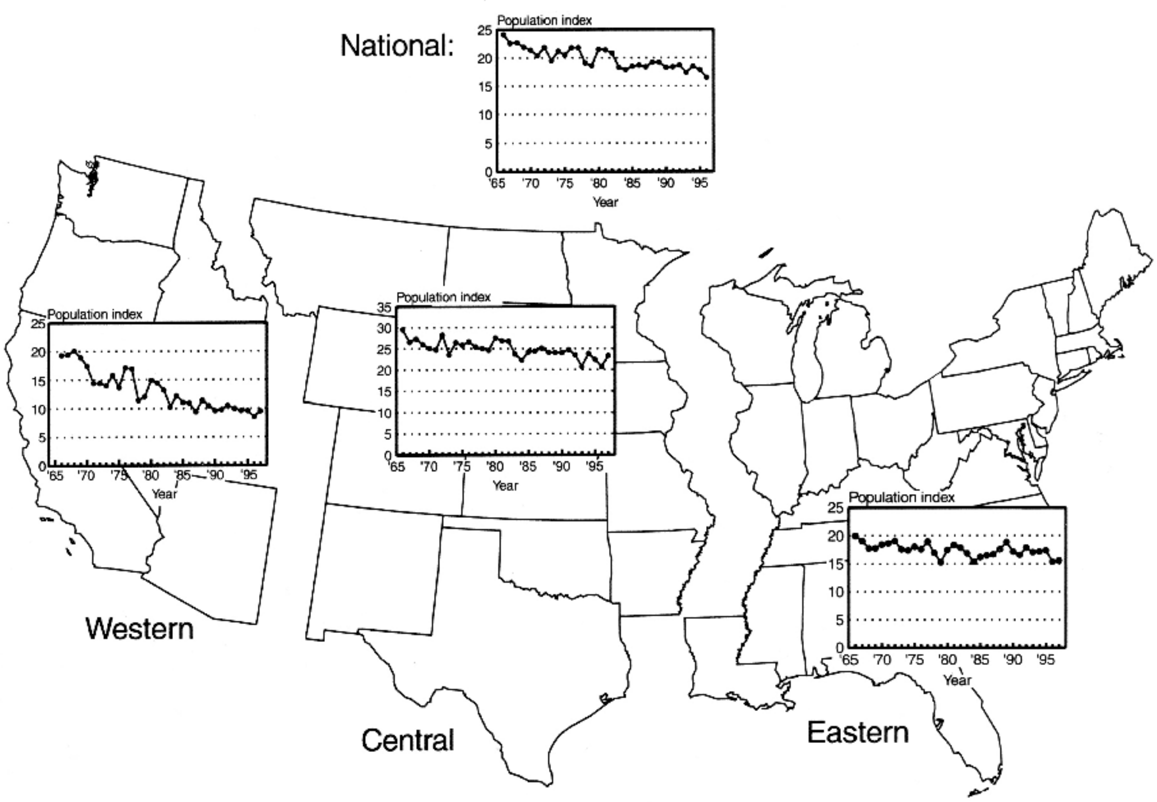


Figure 19. Mourning dove population trends from 1966 to 1996 by management unit (Dolton and Smith 1997; Dolton, pers. comm., U.S. Fish and Wildlife Service, 1997).

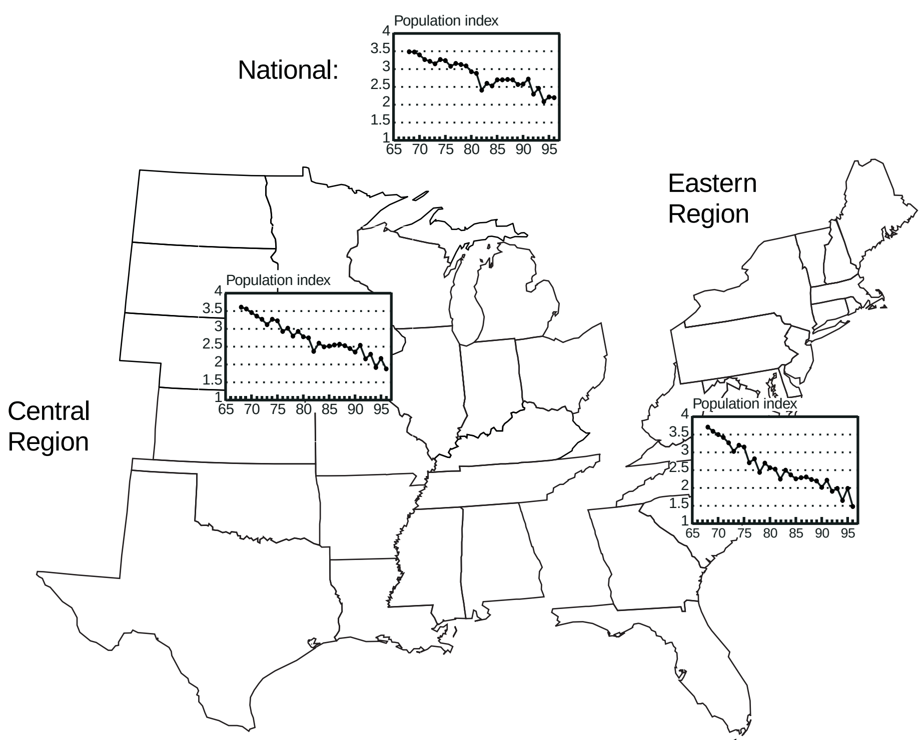


Figure 20. Woodcock population trends from 1968 to 1996 by management unit (Bruggink 1996).

with dove abundance) and the size of farms (a factor negatively associated with dove abundance) were the two environmental attributes that were consistently associated with dove population indices over time (Tomlinson and Dunks 1993). The trend toward fewer and larger farms has almost certainly had a negative impact on dove populations in this region. The reasons for declining dove populations in the West are more diverse. Certainly the trend toward fewer and larger farms has also negatively impacted dove numbers in the Western management unit, but the clearing of shrublands and live oak trees, the conversion of small-grain farming to cotton in some areas, the conversion of fallow fields and pastureland to cropland, and the increased use of pesticides and herbicides have all likely contributed to population declines in the West (Reeves and others 1993).

Call-count trends for woodcock show remarkable consistency in the temporal pattern of decline in both the Eastern and Central management units (figure 20). Long-term trends since 1968 indicate that the number of calling woodcock heard have declined by 2.5% and 1.6% per year in the Eastern and Central regions, respectively (Bruggink 1996). Within the last 10 years, declines in the Eastern (-3.2%/year) and Central (-3.7%/year) units indicate that the rate of decline may be accelerating. Woodcock select early successional stages of second-growth hardwood forests associated with fields and forest openings on mesic sites. As with the mourning dove, the widespread declines in woodcock breeding populations are thought to be related to a deterioration of breeding habitats due to forest succession and land use intensification (Straw and others 1994).

### *Dove and woodcock harvests*

More mourning doves are harvested than any other game bird. About 50 million birds were harvested annually nationwide in the early 1970s and early 1980s with the greatest number of harvested birds coming from the Eastern management unit (figure 21). In the absence of national surveys that monitor dove harvest, the trend data reported here represent compilations from state

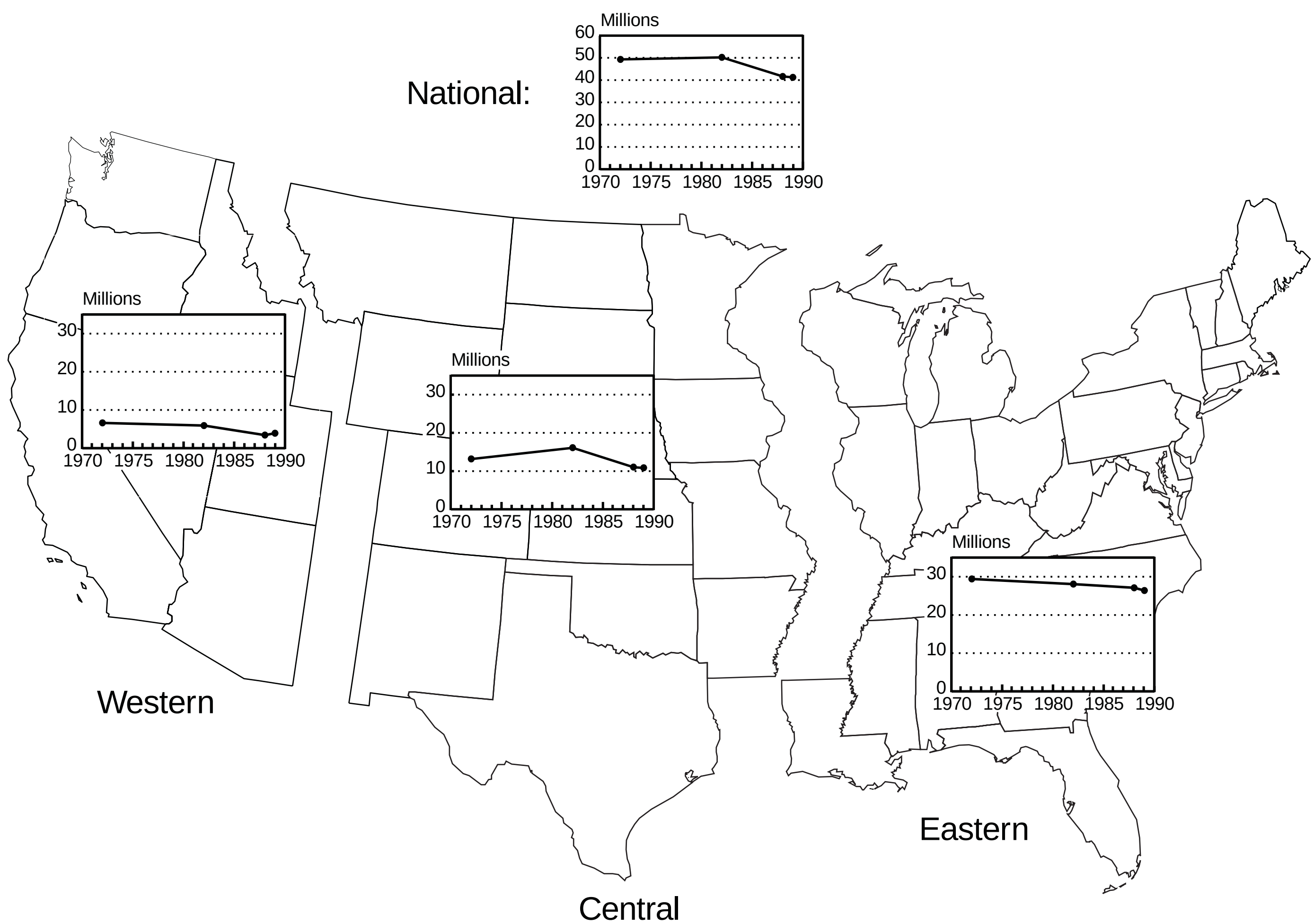


Figure 21. Mourning dove harvest trends from 1972 to 1989 by management unit (Sadler 1993).

wildlife agencies and should be interpreted carefully (Sadler 1993). Given recent dove population trends, it is not surprising that estimates for the 1988 and 1989 seasons show declining numbers of harvested birds. The total harvest has declined by about 6% in the Eastern management unit, and by more than 30% in the Central and Western management units from the mid-1980s to the late 1980s.

Unlike doves, woodcock harvests are monitored annually through wing-collection surveys that are used to estimate the seasonal bag indices. Estimates are adjusted to a 1969 base-year to facilitate temporal comparisons (Bruggink 1997). Data since 1965 indicate that total seasonal bag indices have declined by 75% and 63% in the Eastern and Central management units, respectively (figure 22). Record low seasonal bag indices were observed in both management units in 1996. Although the results from the wing-collection survey provide approximate trends in harvest success, they should be evaluated judiciously because of the non-random sampling procedure by which participants are selected (Straw and others 1994; Bruggink 1997).

## Furbearer Species

Collectively, furbearers constitute a resource that is valued ecologically, recreationally, and commercially. The management of furbearer resources in the United States has recently undergone a period of rapid change stemming from public sentiments that have spawned new domestic and international legislation regulating fur

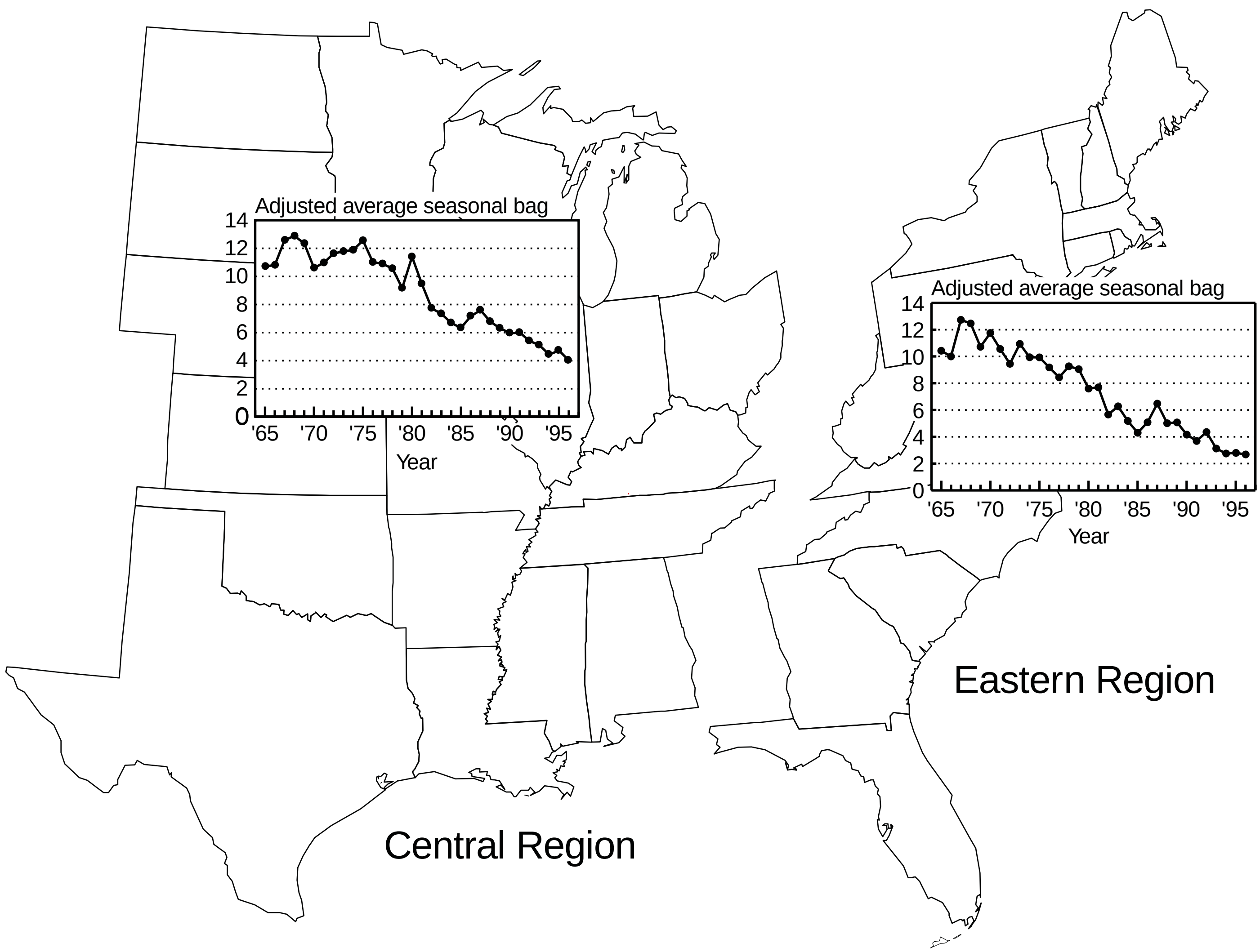


Figure 22. Woodcock harvest trends from 1965 to 1996 by management unit (Bruggink 1997).

trapping activities (Siemer and others 1994). Not only is the management of furbearers controversial (Gentile 1987), but the secretive nature of these species (Deems and Pursley 1983) make it difficult to evaluate the population status of most species. For many species the only available information is on temporal trends in harvests, which is likely more reflective of fur prices than population status. The conflict associated with trapping and the limited amount of information on resource status make furbearer management uncertain and particularly contentious.

### *Furbearer populations*

There have been very few comprehensive examinations of trends in furbearer populations nationwide. The results from two national summaries (Sisson-Lopez 1979, Deems and Pursley 1983) were discussed in the previous RPA Assessment (see Flather and Hoekstra 1989). A more recent compilation of furbearer population status was completed for the Fur Resources Committee of the International Association of Fish and Wildlife Agencies.[8] A telephone survey of state agency furbearer biologists was used to assess the population status of those species that were identified to be important based on factors such as population status, harvest levels, damages, or management activities. The population status for the three most important species, as determined by each state, was evaluated based on the state biologists' ranking of the population relative to the carrying capacity of the habitat within the state. Biologists also provided their professional judgment on the likely short-term (next 10 years) population projection and the reasons contributing to that future trend. Data were obtained from all states except Hawaii.

We mapped the current population status for those furbearer species that were reported by at least 10 states including beaver, raccoon, muskrat, coyote, bobcat/lynx, and red/gray fox. State populations of most furbearers included in the survey were estimated to be at or above carrying capacity (figure 23). Several species have the potential to cause significant economic damage (e.g., beaver, coyote) or can be a public health concern (e.g., raccoon) when populations exceed a level that the habitat can support. Few states reported furbearer populations that were below carrying capacity, with beaver (four states) and muskrat (three states) having the greatest number of states reporting below-capacity populations.

Even though state agency biologists thought that many furbearer populations exceeded the capacity of the habitat, many biologists project populations to continue to increase (figure 23). For example, of the 28 states that listed beaver as one of the most important furbearers, about 70% are expecting population increases in the next 10 years. The pattern was similar for raccoons except that proportionately more states are expecting raccoon populations to decline due to disease outbreaks. In addition to improving habitat conditions, decreased harvest caused by low fur prices were prominent reasons cited for population increases.

### *Fur harvest and price*

Data on furbearer harvest trends are more complete than data on population levels. Data on the number of pelts taken by species and the average prices were provided by the Fur Resources Committee of the International Association of Fish and Wildlife Agencies[9] as derived from State Agency personnel.

The national trend in fur harvests has continued the decline observed in the previous RPA Assessment (Flather and Hoekstra 1989). Apart from a short-term deviation during 1986-87 period when harvests increased by about 50%, harvests have declined from a peak of 20 million pelts in 1980 to a low of 3 million pelts in 1991 (figure 24). Since 1991 there have been modest increases in fur harvest, reaching 6 million pelts in 1995. Harvest trends for the two most commonly harvested species, muskrat and raccoon, are consistent with the national trend for all species (figure 24).

There are no notable deviations from the national trend within RPA regions (figure 24). Aside from the fact that the North accounts for more than 60% of the total number of pelts harvested nationwide, all regions showed peak harvests during the late 1970s that were followed by declines through the early 1990s.

Prices that trappers have received for their pelts is a strong determinant of harvest. The price is in turn driven by consumer demand for fur products, which is affected by economic expansion, discretionary income levels, consumer preferences, and cyclical weather patterns.[10] Prices are obviously correlated with the harvest pattern (figure 25). Peak prices in the late 1970s and the mid-1980s correspond to peak harvest periods. Unlike harvest, average pelt prices do show regional variability that is associated with differences in species composition. Both the Pacific Coast and Rocky Mountain Region have had higher average pelt prices than the North or the South since 1970. However, the total value of fur harvests (pelt price x number of pelts) remains higher in eastern regions

*[8] Unpublished report, Southwick Associates. 1993. 1993 state and provincial survey of furbearers with emphasis on nuisance animals. Report on file at the Rocky Mountain Research Station.*

*[9] R.G. Frederick, pers. comm., Department of Experimental Statistics, Louisiana State University, 1997. Data on file at the Rocky Mountain Research Station.*

*[10] Unpublished report, Southwick Associates. 1993. An economic profile of the U.S. fur industry. Report on file at the Rocky Mountain Research Station.*

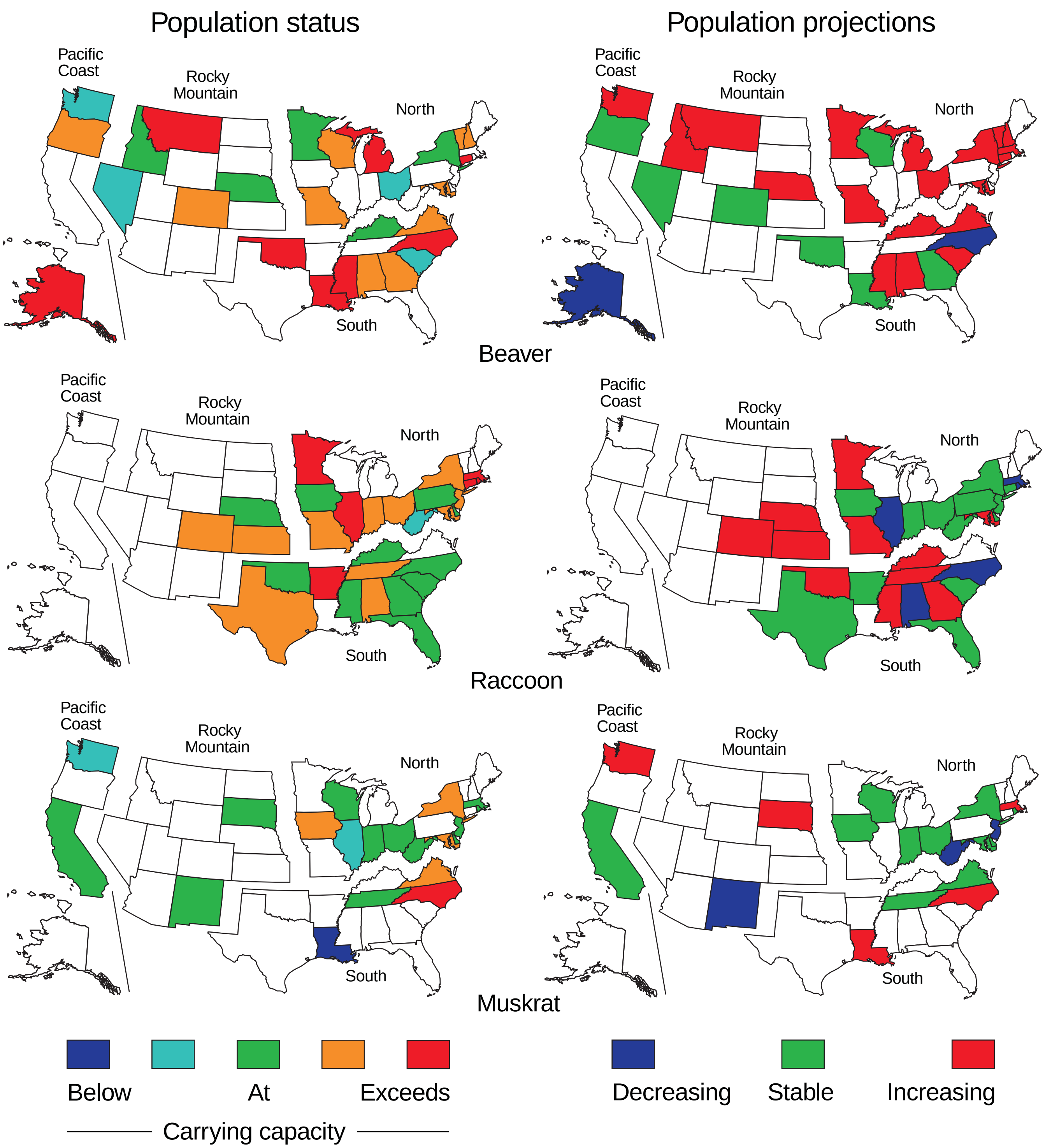


Figure 23. Furbearer population status in relation to carrying capacity and projected (10 year) population trends for selected species (Unpublished report, Southwick Associates 1993. Report on file at the Rocky Mountain Research Station).

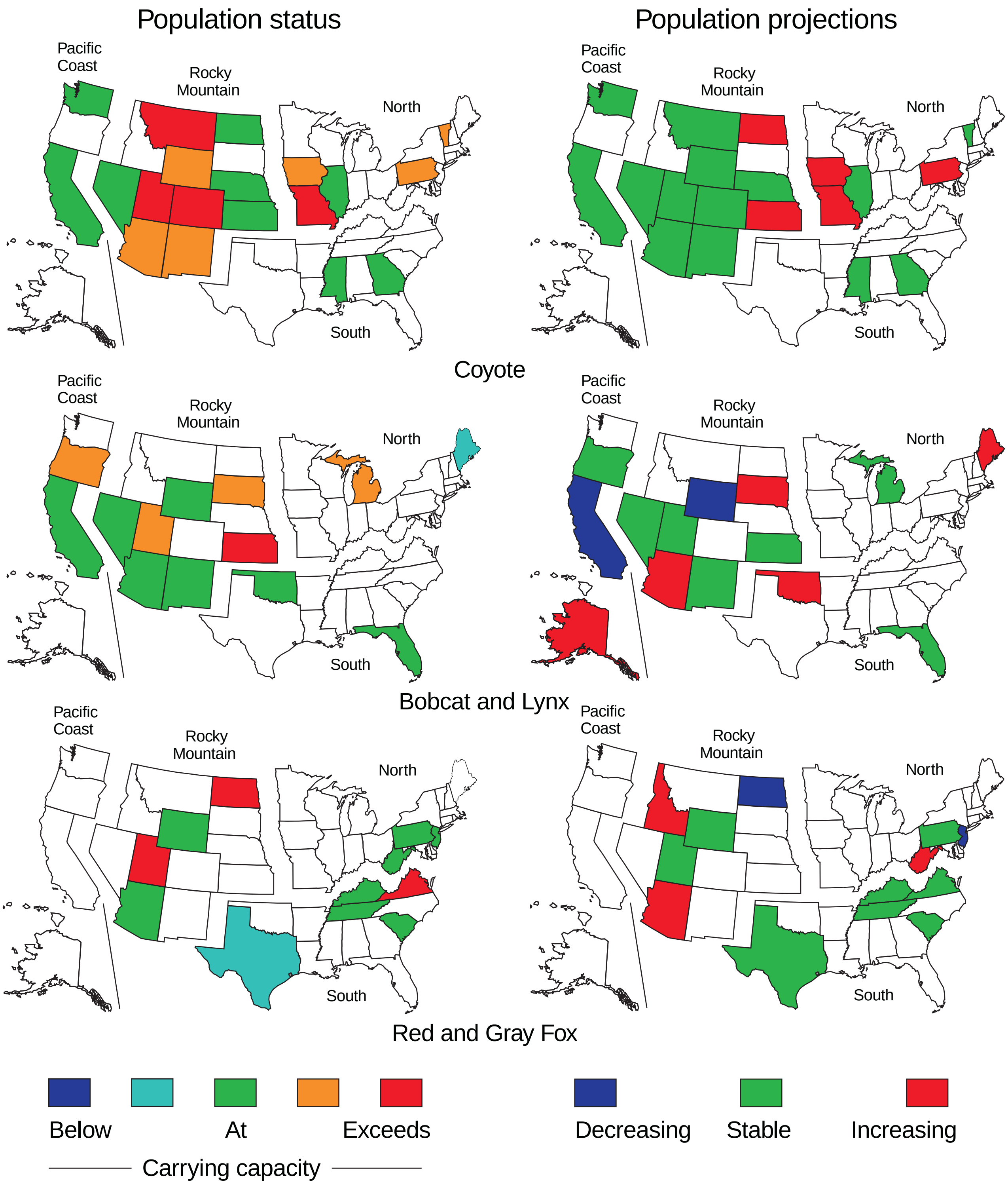


Figure 23. (Cont'd.)

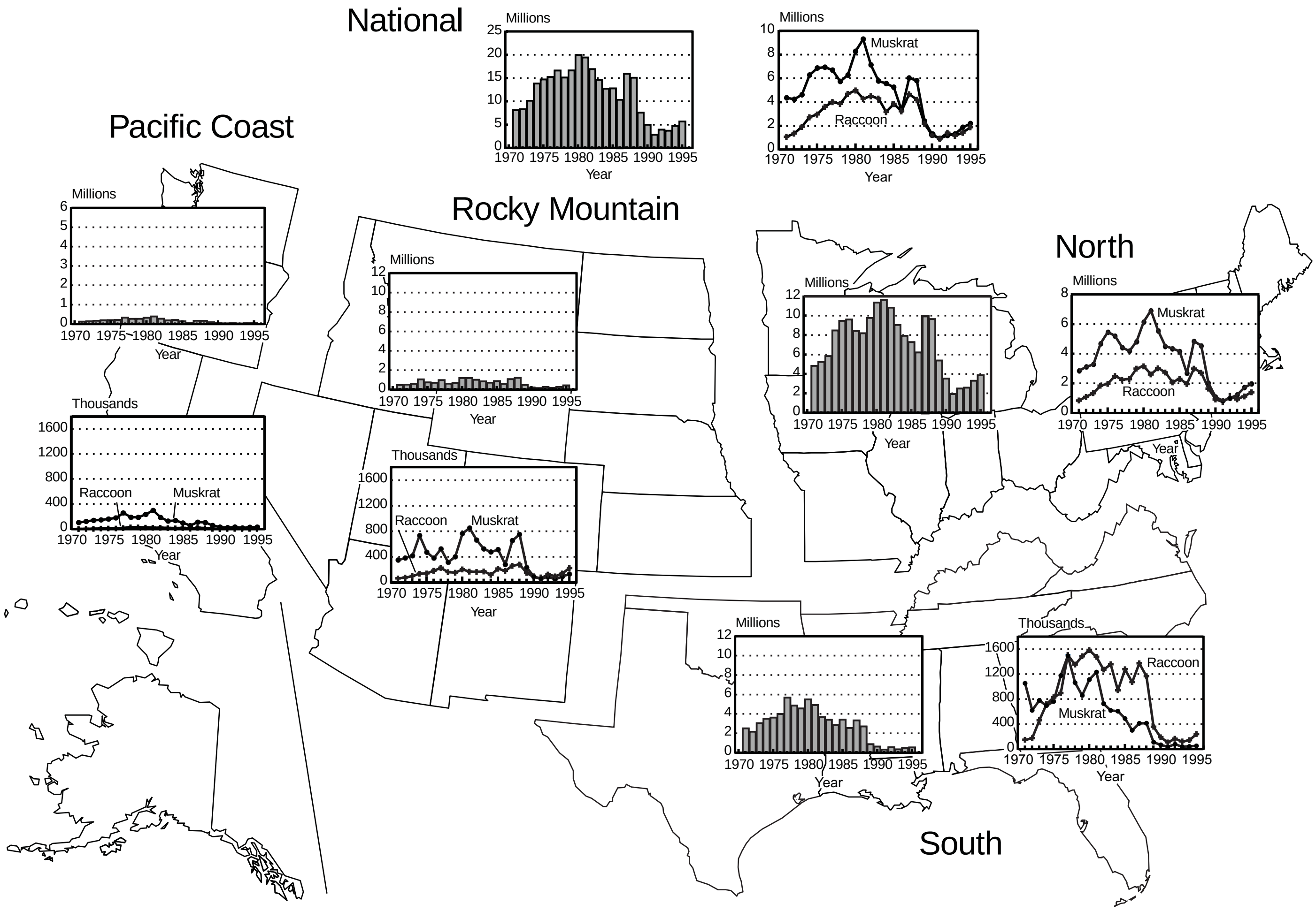


Figure 24. Trends in total fur harvest (bar graph), and the harvest of muskrat and raccoon (line graph), by RPA Assessment region from 1971 to 1995 (R.G. Frederick, pers. comm., Department of Experimental Statistics, Louisiana State University, 1997).

because of the large number of pelts sold. After showing a variable price pattern through the late 1980s in all regions, average pelt prices have remained relatively stable since 1990 at levels that are about 60% lower than those observed during the late 1970s and early 1980s (figure 25).

Although furs harvested by trapping remain an important source of pelts, most pelts used in fur garment manufacturing are produced by fur farms that raise primarily mink and fox.[11] Recent trends in fur farm characteristics track well with those trends observed for trapped furs (table 13). In 1987, 1,027 farms raised 4.12 million mink pelts worth about $177 million ($43.00/pelt). For comparison, 362,000 mink were trapped in 1987 for a total value of $7.6 million ($23.26/pelt). By 1990, 771 farms (down 25% from 1987) raised 3.37 million pelts with a total value of $85.8 million ($25.50/pelt). During the same year, trappers harvested 142,000 mink with a total value of $3.1 million ($25.38/pelt). From 1987 to 1990, trapped mink dropped from about 8% of the total mink harvest (trapped + farmed) to about 4% of the total mink harvest.

Increasingly, the global economy is affecting the North American fur industry. The fur industry must continually demonstrate compliance with the humane trapping standards adopted by the European Economic Community (1991) if European markets are to remain open to North American fur products. Similarly, fur imports from countries with lower labor costs are continually challenging North American manufacturers to control costs to remain competitive.[12]

*[11] Unpublished report, Southwick Associates. 1993. An economic profile of the U.S. fur industry. Report on file at the Rocky Mountain Research Station.*

*[12] Unpublished report, Southwick Associates. 1993. An economic profile of the U.S. fur industry. Report on file at the Rocky Mountain Research Station.*

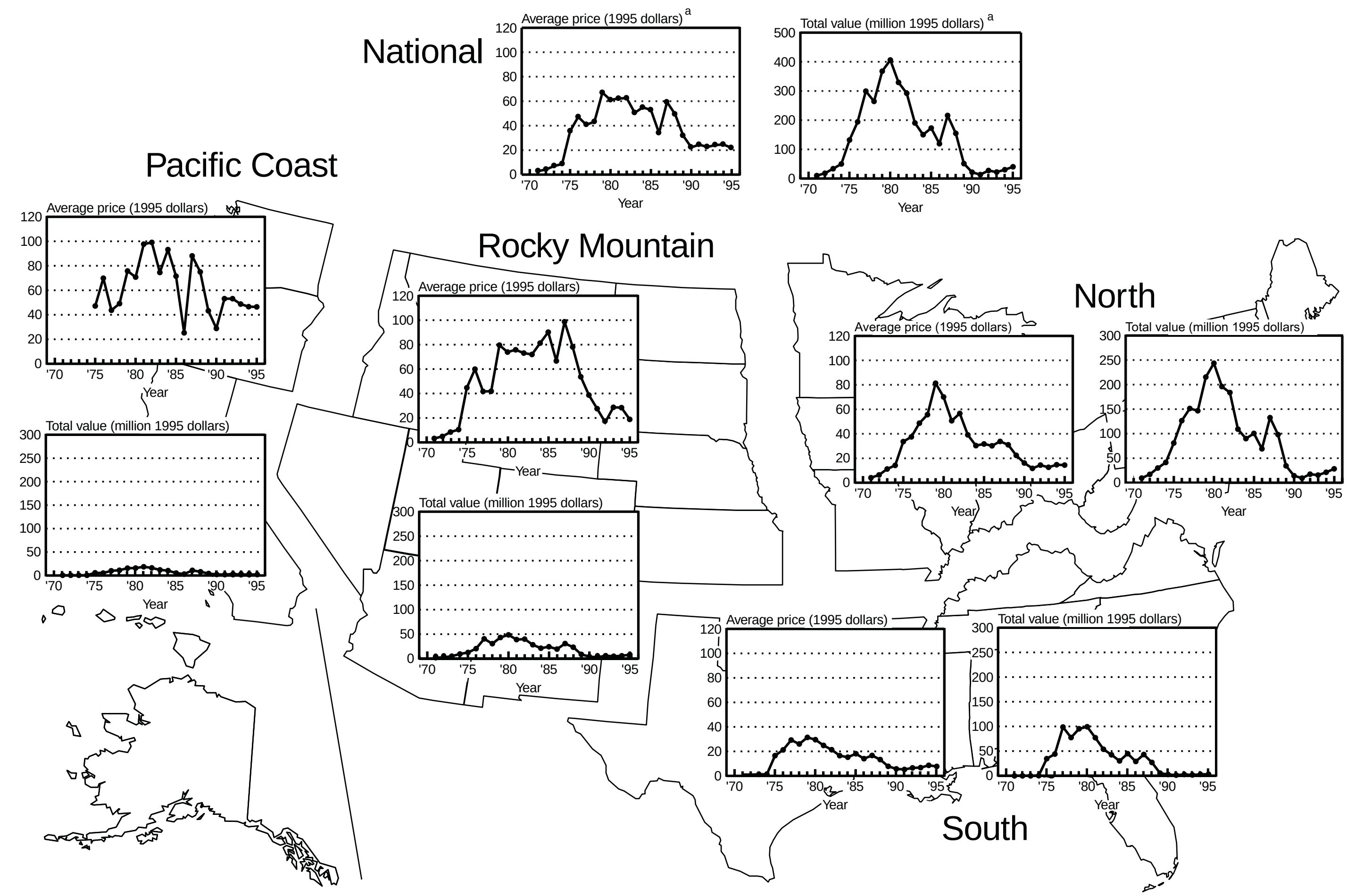


Figure 25. Trend in average pelt price (across all species) and total fur value by RPA Assessment region from 1971 to 1995. Both price and value have been adjusted for inflation (1995 constant dollars) (R.G. Frederick, pers. comm., Department of Experimental Statistics, Louisiana State University, 1997). [a]Average price and total value have been adjusted for inflation based on producer price indices for "hides, skins, leather, and related products" (Council of Economic Advisors 1997).

Table 13—Trends in mink and fox fur farms. (Unpulbished report, Southwick Associates [1993]. Report on file at the Rocky Mountain Research Station.)

| Species/farm characteristics | ------------------ Year ------------------ | |
|---|---|---|
| | 1987 | 1990 |
| **Mink** | | |
| number of farms | 1,027 | 771 |
| number of pelts | 4.12 million | 3.37 million |
| value of pelts | $177.2 million | $85.8 million |
| pelt price | $43.00 | $25.50 |
| **Fox** | | |
| number of farms | 112 | 84 |
| number of pelts | -[a] | - |
| value of pelts | $19.3 million | $9.3 million |
| pelt price | - | - |

[a]No data.

## Nongame Species

For our purposes, nongame species are defined as those terrestrial vertebrate species that are not consumptively taken for sport, subsistence, or profit. As such, nongame species comprise the majority of the approximately 3,000 vertebrates that are resident or seasonal inhabitants within the United States (Flather and Hoekstra 1989). Unfortunately, there are very few data sources available for most nongame species that would permit an exploration of national and regional population trends. One taxonomic group where sufficient population information does exist to support broad-scale analyses of abundance trends is birds.

### *Breeding bird populations*

The North American Breeding Bird Survey (BBS) was established in 1966 to provide data on breeding bird populations along road-side routes across the continental

United States and southern Canada. Routes occur along secondary roads, are 24.5 miles long, and are surveyed once a year in June. During each survey, all birds seen or heard along the route are counted at 50 stops placed at 0.5-mile intervals. More than 3,400 routes allowed us to examine the relative abundance trends for approximately 400 species nationwide for the period 1966 to 1996.[13] For details concerning the design and implementation of the BBS, see Droege (1990).

Relative abundance trends for each species was summarized two ways. First, we estimated the number of species with statistically significant increasing, decreasing, or stable trends nationwide and for each region. Secondly, we grouped birds according to life-history characteristics including: nest type/location (cavity, open cup, ground or low, midstory or canopy nesters), migration status (neotropical migrant, short-distance migrant, permanent resident), and breeding habitat (woodland, shrubland, grassland, wetland or open water, urban nesting) according to the classification of Peterjohn and Sauer (1993). The number of species with increasing, decreasing, or stable trends was estimated separately for each life-history group. To qualify for analysis, each bird species had to have been detected on at least 15 routes. We counted species as having an increasing or decreasing trend if the slope of the regression differed from zero with probability $P \leq 0.10$. Regression slopes that were not significantly different from 0 ($P > 0.10$) were counted as stable populations.

Nationally, the majority of surveyed species (n=194 species, 48% of all species analyzed) have had stable trends in relative abundance. The number of species with increasing (n=109) and decreasing (n=103) trends were nearly equal (figure 26, table 14). Regionally, the North had the greatest proportion of increasing species (31%) and the South had the greatest proportion of decreasing species (35%). Both the Rocky Mountain and Pacific Coast regions were characterized by a high proportion of species with stable trends (64 and 61%, respectively).

Among the 12 bird life-history groups we examined (figure 26 and table 14), those with the greatest proportion of declining bird species were observed among species that nest in or around urban areas or human settlement (54%), nest in grassland habitats (44%), or nest on or near the ground (36%). Those life-history groups with the greatest proportion of species with increasing trends included wetland or open water nesting species (40%), cavity nesting species (37%), or short-distance migrant species (31%). We did not observe strong evidence for consistent declines among those species that winter in the neotropics. Nearly 55% of neotropical migrant species have had stable populations during the 1966-1996 period. However, of the species with significant trends, 27% of neotropical migrants had declining trends compared to 19% with increasing trends.

Given the habitat trends reviewed earlier, the abundance trends for urban- and wetland-associated species were surprising. Both of these species groups show trends that deviate from those expected given the trends in habitat. Nearly 54% of those species associated with urban and intensively farmed habitats showed significantly declining trends over the 1966-1996 period, despite the fact that urban habitats have increased substantially over this period (see figure 2, table 1). Similarly, a high proportion of wetland-associated species have shown abundance increases (40%) despite declines in wetland habitats (see figure 7). Although the pattern observed for urban species may be associated with "clean farming" practices that have increased nest disturbance and reduced the amount of waste grains and other foods that support these species (Sauer and others 1997), it may also be a function of biases associated with the location of routes and BBS observers. Because urban associated species are not of interest to most birders, it has been speculated that BBS observers may detect fewer individuals of this species group over time because they "tune out" these species in favor of other species (Sauer and others 1997). There is also a chance that populations of urban species show a decline because routes have had to be relocated because urban encroachment and increased traffic noise makes it difficult to detect species. Similar cautions apply to wetland associated species (Sauer and others 1997). However, there have been several agricultural programs (e.g. CRP, Wetlands Reserve Program) that have improved nesting conditions associated with wetland habitats (Robinson 1997). In both cases it will be difficult to resolve these apparent contradictions between abundance and habitat trends in the absence of information on habitat along BBS routes.

The trends in grassland nesting birds also warrant examination with respect to the habitat trends reviewed earlier. Over the 1966 to 1996 period, grassland nesting birds showed the lowest proportion of species with increasing trends (11%) and the second highest proportion of species with declining trends (44%)—a pattern consistent with Knopf's (1995) findings. Given the retirement of nearly 36 million acres of cropland that was planted to perennial cover in the late 1980s as part of the CRP, one would expect grassland bird populations to have increased in response to this program. Evidence of a positive population response by grassland birds to the CRP has been observed at local (Johnson and Swartz 1993) and regional (Reynolds and others 1994) scales, but on initial inspection appears to be inconsistent among species at the national scale using the BBS. However, by viewing the

*[13] J.R. Sauer, pers. comm., U.S. Geological Survey, Biological Resources Division, Patuxent Wildlife Research Center, 1997. Breeding Bird Survey trend analysis. Data on file at the Rocky Mountain Research Station.*

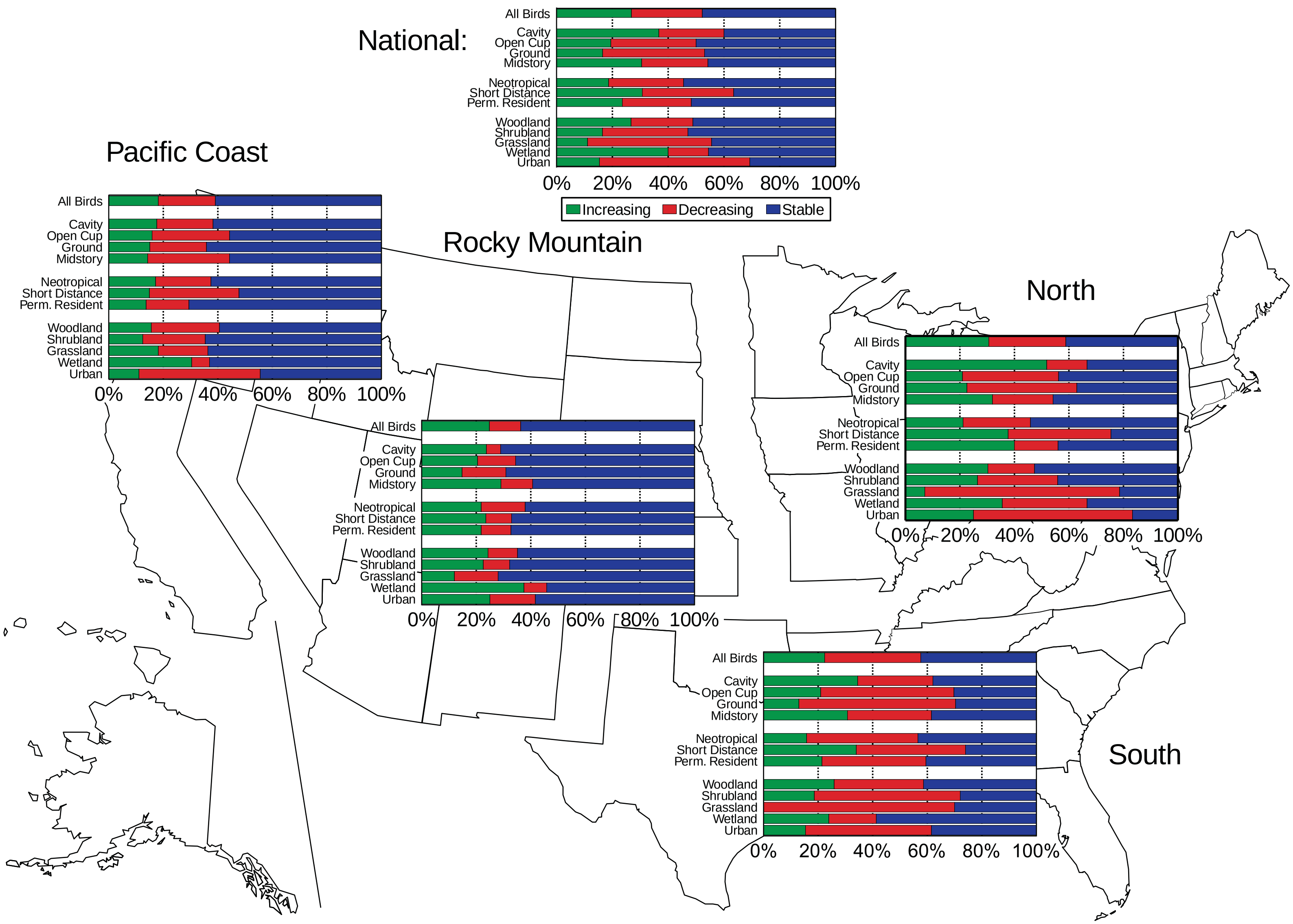


Figure 26. The proportion of bird species with increasing, decreasing, and stable trends from 1966 to 1996 by RPA Assessment region. Bird species have been grouped by broad life history characteristics including nest type/location (cavity, open cup, ground or low, midstory or canopy nesters), migration status (Neotropical migrant, short-distance migrant, permanent resident), and breeding habitat (woodland, shrubland, grassland, wetland or open water, urban nesting). Species were counted as increasing or decreasing if the trend was significantly different from 0 at P≤0.1 (J.R. Sauer, U.S. Geological Survey, Biological Resources Division, Patuxent Wildlife Research Center, 1997).

BBS over the last 30 years, more recent trends may be masked by the variation over this long time period. When we estimated the trends in grassland nesting birds from 1985 to 1996, a period of time when the CRP was implemented, we do see some evidence for a positive effect at a national scale. Over this period, the proportion of grassland nesting birds with upward population trends increased to 19%, while the proportion of species with declining trends decreased to 22%.

Abundance trends among species groups vary considerably by RPA region. A greater proportion of bird abundance trends in the South declined when compared to other regions. The average proportion[14] of species with declining trends across all groups was 44% in the South. Greater than 40% of the species showed declining trends for grassland nesting birds (70%), ground nesting birds (57%), shrubland nesting birds (53%), open-cup nesters (49%), urban nesting birds (46%), and neotropical migrants (41%). The only groups in the South that had more species with increasing than decreasing trends were cavity and wetland nesting species.

[14] *Estimated as a weighted average where the number of species in each species group served as the weight.*

The North had the next highest average proportion of species with declining trends (34%). As in the South, species groups with a relatively high proportion of species with declining trends included grassland nesting birds (71%), urban nesting birds (58%), and ground nesting birds (40%). Half of the bird groups in the North showed a greater proportion of increasing species and include cavity nesters, midstory and canopy nesters, permanent residents, woodland associated species, and wetland associated species.

The average proportion of species with declining trends were much lower in the Pacific Coast (26%) and Rocky Mountains (13%) than in the eastern regions. The only species groups in the Rocky Mountain region that showed more decreasing than increasing species were grassland and ground nesting species. Again, the trend in grassland nesting birds warrant remark. Because many of the acres enrolled in CRP are concentrated in the Rocky Mountain region (USDA Natural Resources Conservation Service 1997), we may expect short-term trends in grassland birds to deviate from the 30-year trend. From 1966 to 1996, 12% of the grassland nesting species had increasing trends and 16% had declining trends in the Rocky Mountain region. From 1985 to 1996, the number of grassland nesting

Table 14--Number of breeding bird species with increasing, decreasing, and stable trends from 1966 to 1996 by life-history characteristic for the total U.S. and RPA Assessment region (J.R. Sauer, pers. comm., U.S. Geological Survey, Biological Resources Division, Patuxent Wildlife Research Center, 1997).

| | Total U.S. | | | | North | | | |
|---|---|---|---|---|---|---|---|---|
| | Total Species | Increasing species (%) | Decreasing species (%) | Stable species (%) | Total species | Increasing species (%) | Decreasing species (%) | Stable species (%) |
| **All species** | 406 | 109 (26.8) | 103 (25.4) | 194 (47.8) | 209 | 64 (30.6) | 59 (28.2) | 86 (41.1) |
| **Nest type/location** | | | | | | | | |
| Cavity | 60 | 22 (36.7) | 14 (23.3) | 24 (40.0) | 27 | 14 (51.9) | 4 (14.8) | 9 (33.3) |
| Open cup | 180 | 35 (19.4) | 55 (30.6) | 90 (50.0) | 105 | 22 (21.0) | 37 (35.2) | 46 (43.8) |
| Ground/low | 115 | 19 (16.5) | 42 (36.5) | 54 (47.0) | 62 | 14 (22.6) | 25 (40.3) | 23 (37.1) |
| Midstory/canopy | 118 | 36 (30.5) | 28 (23.7) | 54 (45.8) | 72 | 23 (31.9) | 16 (22.2) | 33 (45.8) |
| **Migration status** | | | | | | | | |
| Neotropical | 134 | 25 (18.7) | 36 (26.9) | 73 (54.5) | 85 | 18 (21.2) | 21 (24.7) | 46 (54.1) |
| Short distance | 104 | 32 (30.8) | 34 (32.7) | 38 (36.5) | 61 | 23 (37.7) | 23 (37.7) | 15 (24.6) |
| Permanent resident | 89 | 21 (23.6) | 22 (24.7) | 46 (51.7) | 25 | 10 (40.0) | 4 (16.0) | 11 (44.0) |
| **Breeding habitat** | | | | | | | | |
| Woodland | 131 | 35 (26.7) | 29 (22.1) | 67 (51.1) | 76 | 23 (30.3) | 13 (17.1) | 40 (52.6) |
| Shrubland | 85 | 14 (16.5) | 26 (30.6) | 45 (52.9) | 34 | 9 (26.5) | 10 (29.4) | 15 (44.1) |
| Grassland | 27 | 3 (11.1) | 12 (44.4) | 12 (44.4) | 14 | 1 (7.1) | 10 (71.4) | 3 (21.4) |
| Wetland/open water | 90 | 36 (40.0) | 13 (14.4) | 41 (45.6) | 45 | 16 (35.6) | 14 (31.1) | 15 (33.3) |
| Urban | 13 | 2 (15.4) | 7 (53.8) | 4 (30.8) | 12 | 3 (25.0) | 7 (58.3) | 2 (16.7) |

species with increasing trends grew to 28% while the number with declining trends decreased slightly to 12% providing very tentative evidence for a regional affect of CRP. Because other factors could have also changed during this time period (e.g., weather), it is not possible, in the absence of further research, to attribute grassland bird population increases to the CRP.

### *Bird diversity and land use*

Diversity (i.e., the patterns of species distributions and abundances) has long been a central issue of ecological study (May 1986) and is receiving a renewed and urgent interest (Rosenzweig 1995) by conservation planners in response to global degradation of biological diversity. Much of the focus on conserving biological diversity has been in the species-rich tropics (Raven 1988; Allan and Flecker 1993). Although the tropics clearly warrant conservation attention, it is important to acknowledge the extent to which natural environments have been altered by human activity in temperate regions (Carlson 1988:22; Ricketts and others 1999). Franklin (1988:166) observed that "[m]any ecosystems and organisms in temperate zones have been entirely eliminated, and most remaining examples of natural ecosystems are fragmented and highly modified." Concern exists that as land management intensifies to support growing human populations, natural variety will be reduced, leading to diminished productivity or limited capacity to recover from natural or human-caused disturbance. Ecosystem simplification under resource extraction and land use intensification is, in fact, the essence of the biological diversity issue (Roberts 1988; Raven 1990; Koshland 1991; Ehrlich and Wilson 1991; Nelson and Serafin 1992), but empirical investigation of this pattern across broad geographic regions has been limited. Identifying and monitoring elements of biological diversity is a complex undertaking at any scale of analysis and particularly difficult over large geographic areas (Lubchenco and others 1991).

Complexity notwithstanding, a basic ecological inventory is fundamental to describing and understanding regional patterns of diversity. Unfortunately, much of the information that could support macroecological investigation of biodiversity patterns is restricted to a subset of well-studied taxa. One well-studied taxon that does lend itself to investigation of regional diversity patterns is birds. Despite this taxonomic limitation, birds have been found to be useful indicators of broad-scale habitat changes

Table 14 (Cont'd.)

| South | | | | Rocky Mountain | | | | Pacific Coast | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Total species | Increasing species (%) | Decreasing species (%) | Stable species (%) | Total species | Increasing species (%) | Decreasing species (%) | Stable species (%) | Total species | Increasing species (%) | Decreasing species (%) | Stable species (%) |
| 210 | 47 (22.4) | 74 (35.2) | 89 (42.4) | 270 | 67 (24.8) | 31 (11.5) | 172 (63.7) | 220 | 40 (18.2) | 46 (20.9) | 134 (60.9) |
| 29 | 10 (34.5) | 8 (27.6) | 11 (37.9) | 38 | 9 (23.7) | 2 (5.3) | 27 (71.1) | 34 | 6 (17.6) | 7 (20.6) | 21 (61.8) |
| 86 | 18 (20.9) | 42 (48.8) | 26 (30.2) | 122 | 25 (20.5) | 17 (13.9) | 80 (65.6) | 88 | 14 (15.9) | 25 (28.4) | 49 (55.7) |
| 54 | 7 (13.0) | 31 (57.4) | 16 (29.6) | 81 | 12 (14.8) | 13 (16.0) | 56 (69.1) | 53 | 8 (15.1) | 11 (20.8) | 34 (64.2) |
| 65 | 20 (30.8) | 20 (30.8) | 25 (38.5) | 86 | 25 (29.1) | 10 (11.6) | 51 (59.3) | 70 | 10 (14.3) | 21 (30.0) | 39 (55.7) |
| 76 | 12 (15.8) | 31 (40.8) | 33 (43.4) | 87 | 19 (21.8) | 14 (16.1) | 54 (62.1) | 64 | 11 (17.2) | 13 (20.3) | 40 (62.5) |
| 50 | 17 (34.0) | 20 (40.0) | 13 (26.0) | 85 | 20 (23.5) | 8 (9.4) | 57 (67.1) | 67 | 10 (14.9) | 22 (32.8) | 35 (52.2) |
| 42 | 9 (21.4) | 16 (38.1) | 17 (40.5) | 55 | 12 (21.8) | 6 (10.9) | 37 (67.3) | 51 | 7 (13.7) | 8 (15.7) | 36 (70.6) |
| 58 | 15 (25.9) | 19 (32.8) | 24 (41.4) | 74 | 18 (24.3) | 8 (10.8) | 48 (64.9) | 64 | 10 (15.6) | 16 (25.0) | 38 (59.4) |
| 43 | 8 (18.6) | 23 (53.5) | 12 (27.9) | 62 | 14 (22.6) | 6 (9.7) | 42 (67.7) | 48 | 6 (12.5) | 11 (22.9) | 31 (64.6) |
| 10 | | 7 (70.0) | 3 (30.0) | 25 | 3 (12.0) | 4 (16.0) | 18 (72.0) | 11 | 2 (18.2) | 2 (18.2) | 7 (63.6) |
| 46 | 11 (23.9) | 8 (17.4) | 27 (58.7) | 48 | 18 (37.5) | 4 (8.3) | 26 (54.2) | 46 | 14 (30.4) | 3 (6.5) | 29 (63.0) |
| 13 | 2 (15.4) | 6 (46.2) | 5 (38.5) | 12 | 3 (25.0) | 2 (16.7) | 7 (58.3) | 9 | 1 (11.1) | 4 (44.4) | 4 (44.4) |

(Koskimies 1989). What follows is a review of several case studies that address the impacts of land use activities on several attributes of breeding bird diversity. These case studies focus primarily on patterns of association among birds and land use in the eastern United States.

The simplest description of bird diversity patterns across the United States is species richness. Based on BBS data, bird species richness is high in the Northeast, along the southern Appalachians, in areas of the Pacific Northwest, and along the tidewater area of northeastern North Carolina and the Chesapeake Bay (figure 27). Areas of relatively low bird species richness are located along the shortgrass prairie of the western Great Plains and in the desertic basin and range region of the Southwest. Because some species are always missed during bird count surveys, it is important to note that the bird richness estimates depicted in figure 27 are minimum estimates of species richness. Although these bird richness estimates are biased low, they do depict the relative richness among regions accurately (Sauer and others 1997).

There are many causes for the regional variation in bird richness depicted in figure 27. Fundamentally, differences in the amount of energy received at any given location have been shown to explain much of the regional variation in species richness for many taxa, including birds (Currie 1991, Brown 1995). In order to explore the potential impacts of land use practices on observed bird richness, there is a need to first control for the inherent variation in species richness that is unrelated to land use activities. This was accomplished by first estimating the expected species pool under natural conditions from range map data, and comparing the expected species composition to that observed on BBS routes within various physiographic strata. This approach resulted in an estimated ratio of bird species counts from the BBS to the expected species count—a ratio that has been interpreted as an index of bird community integrity (for details see Flather and others 1992).

Flather and others (1992) used an index of vertical habitat complexity as their measure of land use intensification. The habitat structure index was calculated as the ratio of observed habitat layers (vertical vegetation strata including tree canopy, tree bole, shrub layer, soil surface vegetative layer, and subsurface) at site to the expected number of habitat layers under the potential natural vegetation defined by Küchler (1964). Bird community

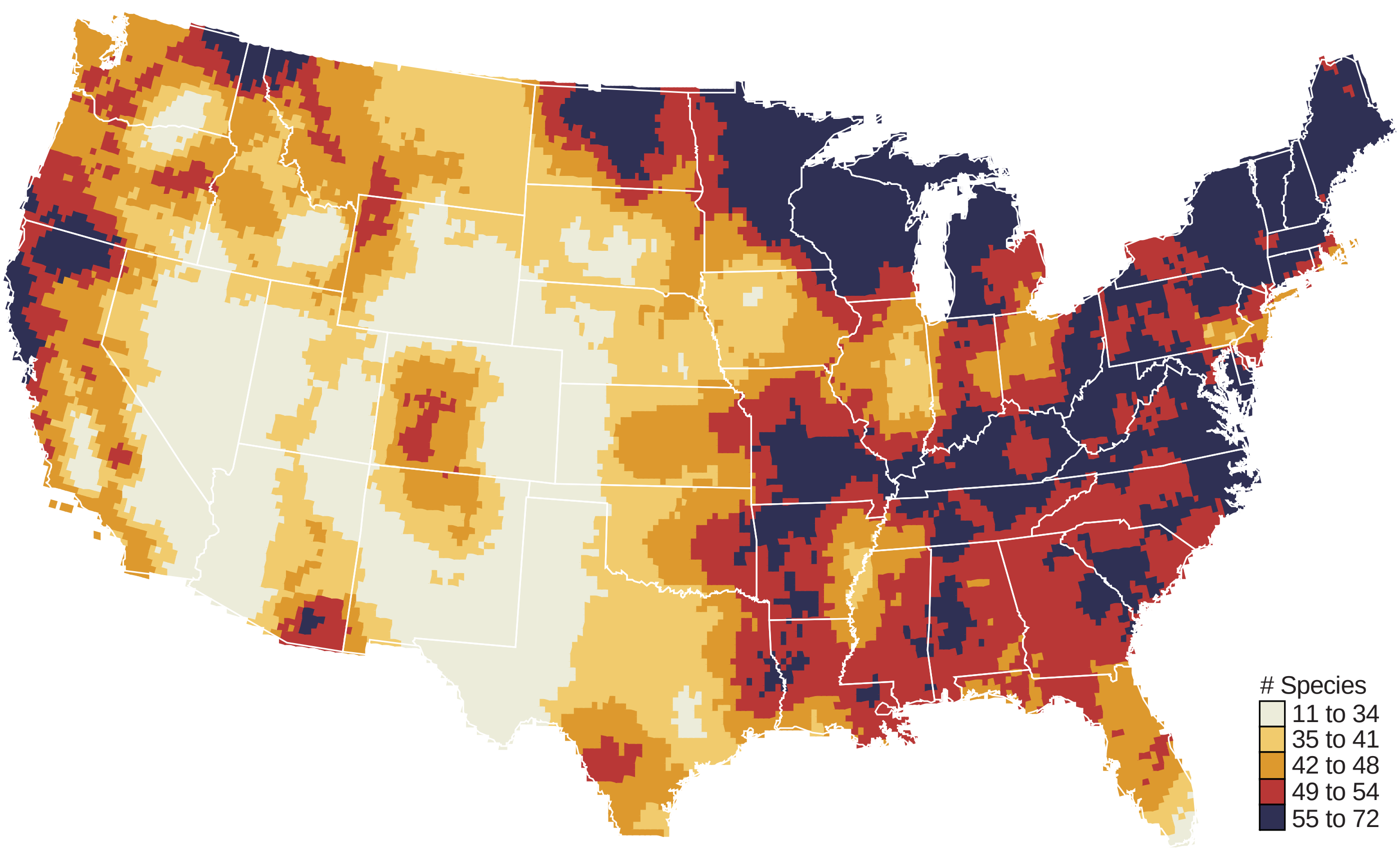


Figure 27. Pattern of bird species richness based on species counts observed on BBS routes.

integrity was found to be positively associated with the habitat structure index (figure 28) within 35 physiographic strata located in the forested region of the eastern United States. However, only 34% of the total variation in bird community integrity was accounted for by the vertical habitat structure index. More complex multiple linear regression models that incorporated landscape composition (i.e., number of land types, land type dominance, and the proportion of urban land) and the spatial structure of forest patches resulted in a model that accounted for 63% of the variation in bird species integrity. These results support the hypothesis that as land uses intensify regionally, a smaller proportion of the expected bird community is observed in any given year, at least for forested ecosystems across the eastern United States.

Changes in bird diversity can result from changes in species richness (as we reviewed above), or it can result from changes in the relative commonness and rarity of species comprising the assemblage. Changes in the commonness and rarity among species found in a given geographic area is often termed the evenness component of species diversity (Magurran 1988). It has been hypothesized that conversion of natural habitats to human-dominated land uses is associated with increased dominance of opportunistic species (sometimes these are exotic species) that can exploit disturbed habitats and increased rarity of those native species considered to be habitat specialists. This hypothesis is derived from the stress ecology literature (Barrett and others 1976; Odum 1985; Rapport and others 1985) and was tested by Flather (1996) using breeding birds in forested ecosystems in the eastern United States.

The evenness component of bird diversity was based on species-accumulation curves (see Flather 1996 for details). As one samples bird species from a particular assemblage, new species are initially encountered rapidly. As samples accumulate, the rate of encounter declines and the total number of species in the collection is approached asymptotically. Species-accumulation curves were derived using statistical simulation within physiographic strata across forested regions of the East using the BBS. All accumulation curves were fit empirically with a single functional form, which would permit comparisons of parameters that describe each curve among physiographic strata. The parameter related to the rate of species accumulation was estimated for each physiographic stratum.

For each physiographic stratum, Flather (1996) also estimated the proportion of cropland and urban land within each stratum as an index of land use intensity. Physiographic strata with a greater proportion of agricultural and urban land uses accumulated species more slowly than landscapes that retained a greater proportion of natural habitats (figure 29). Those strata with the lowest rates of species accumulation (lower right in figure 29) are found primarily in the eastern portion of the

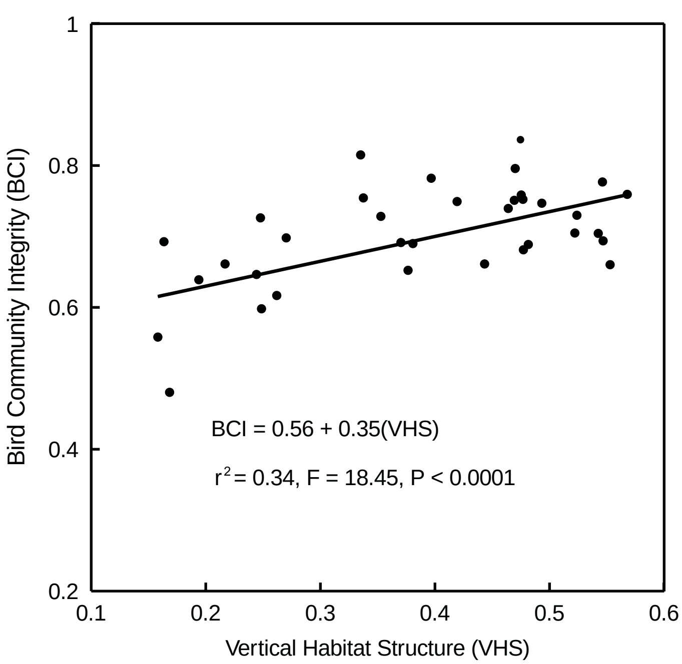


Figure 28. The relation between bird community integrity and vertical habitat structure for 35 physiographic strata located in forested ecosystems of the eastern United States (Flather and others 1992).

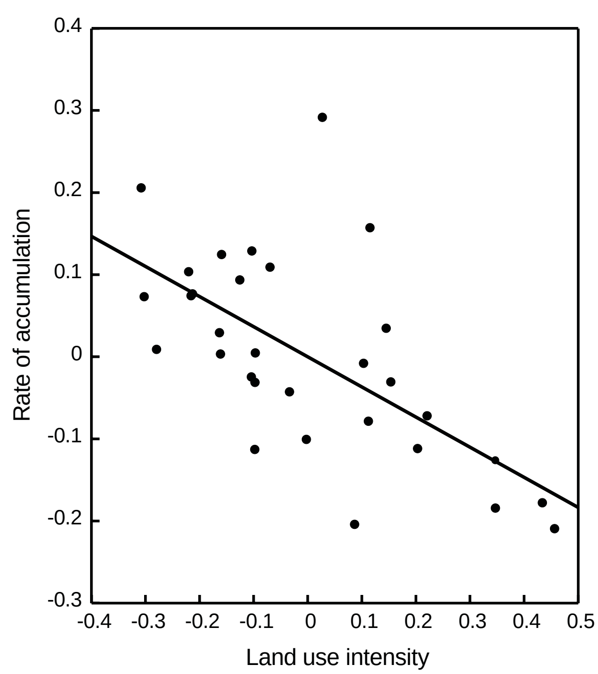


Figure 29. Rate of bird species accumulation as a function of land use intensity (Flather 1996).

United States corn belt. These strata are important centers for agriculture and they are also heavily urbanized. Strata that had high rates of species accumulation were not clustered geographically and included the Western Coastal Plain, the Southern Piedmont, and the Highland Rim and Pennyroyal area of central Tennessee and Kentucky. One physiographic stratum that accumulated species at a rate much greater than expected given the proportion of land under crop and urban development was a region in northwestern Kentucky and southern Indiana (the data point in figure 29 with the greatest rate of species accumulation). One possible explanation for this pattern is that this landscape includes a fairly high proportion of National Forest land that has retained some of the natural vegetation characteristics of this region. Consequently, the National Forest may be serving as a source habitat for bird species that would otherwise be rare or absent from this physiographic region given its current pattern of land use on private lands.

The relation between land use intensity and species-accumulation rate was consistent with predictions of ecosystem behavior under anthropogenic land use stress. Rapport and others (1985) noted that physical restructuring associated with land use conversion from natural habitats (e.g., forest and wetland) to human-dominated land uses (e.g., agriculture and urban land) is associated with a degradation of species diversity. The reduced rate of species accumulation under land use intensification is indicative of a disparate species-abundance distribution, with the bird assemblage being comprised of a greater proportion of rare and distributionally restricted species, and a greater dominance (i.e., high abundance) of a few opportunistic species that do well in simpler, disturbed habitats (Urban and others 1987; Cotgreave and Harvey 1994). This latter pattern is illustrated well by the proportion of individuals observed on BBS routes that are exotic species (figure 30). Areas of the country dominated by agricultural or urban land use observe a relatively high proportion of exotic birds. Recall that the corn belt region was characterized by relatively low species accumulation rates, which is explained in part by the numerical dominance of exotic species in these areas.

The case studies reviewed above show that geographic variation in bird diversity attributes was associated with variation in landscape structure attributes. A question that still remains, is how bird diversity changes over time and whether (and to what extent) those temporal changes can be attributed to changes in landscape structure. A

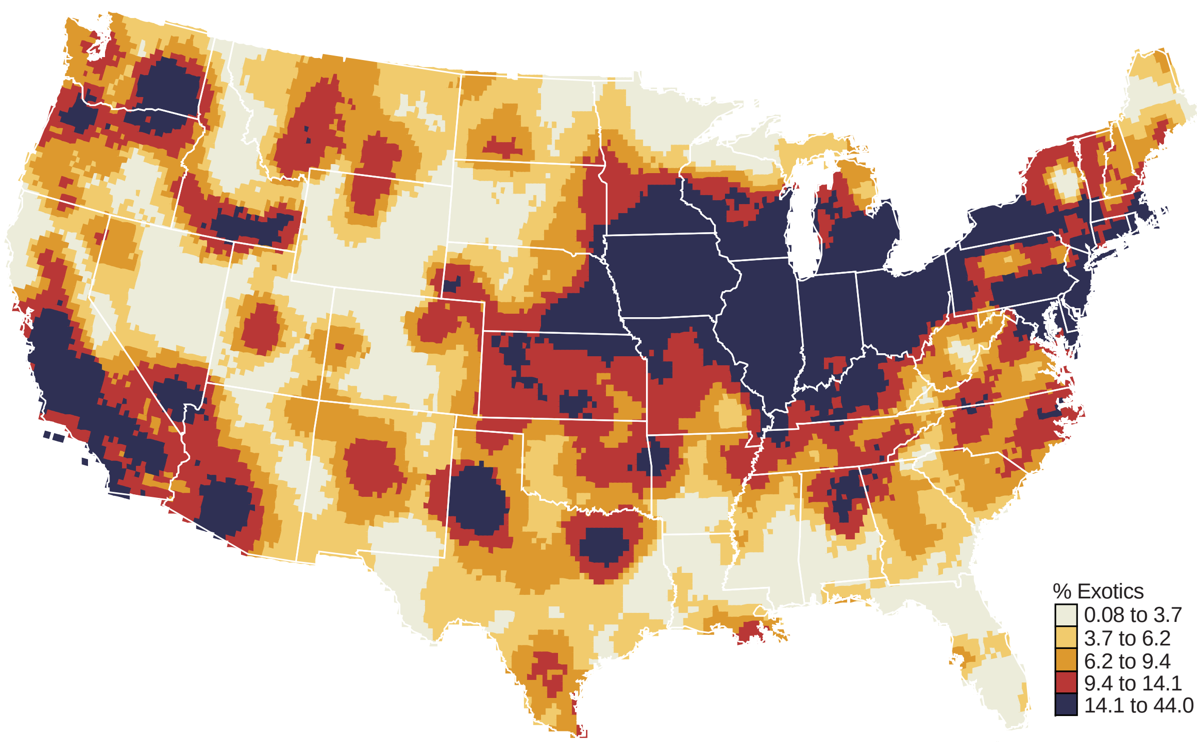


Figure 30. Proportion of total individual birds observed along BBS routes that were exotics based on a 10-year mean from 1980 to 1990.

prediction from the stress ecology literature is that land use intensification can destablize populations resulting in marked increases in the amplitude of population fluctuations (Rapport and others 1985). If this is true, then large scale monitoring efforts like the BBS would be expected to see greater incidence of local extinction events (i.e., variable bird species occurrence among survey routes).

To test this prediction, Boulinier and others (1998) estimated the temporal variation in species richness for those bird species that are thought to be sensitive to the area of forest patches (as defined by Robbins and others 1989) along BBS routes in the mid-Atlantic region of the eastern United States. Landscape metrics were estimated from high-altitude aerial photography from a 19.7 km scene (half the length of a BBS route) centered on each route. The average size of forest patches within each scene was used as an index of forest fragmentation. Comparison of temporal variation in species richness with forest fragmentation supported the hypothesis that temporal variability of forest bird richness over a 22-year period was associated with landscape structure. In particular, species richness of area-sensitive forest birds was more variable in landscape scenes with small average forest patches (i.e., high fragmentation) than in scenes with large average forest patch sizes. These results confirm that more fragmented landscapes not only reduce the numbers of some forest breeding birds, but fragmentation also may be responsible for destabilizing bird community structure through increased rates of local extinction and turnover.

The results from these case studies all emphasize the need to empirically examine the relationships between biological diversity and landscape structure. Results also support the notion that the composition, extent, and spatial layout of land types have the potential to fundamentally affect the flow of ecosystem goods and services in a region (Daily and others 1997). Land use intensification results in simplified bird communities with higher temporal variation in composition, which may presage broader-scale extinction events. However, because the composition of species assemblages varies geographically (Huston 1994), relationships with land use activities may also vary. For example, it is known that neotropical migrant birds comprise the majority of species (>50%) in the East yet comprise a much smaller proportion of the bird species inhabiting the Great Plains (about 10%) or the West (< 15%) (MacArthur 1959). In a study of landscape structure influences on eastern breeding birds, Flather and Sauer (1996) found that the abundance of neotropical migrant birds as a group was more sensitive to landscape structure than were permanent residents or temperate migrants. Neotropical migrants tended to be more abundant in landscapes with a greater proportion of forest and wetland habitats, fewer edge habitats, larger forest patches, and forest habitats well dispersed throughout the landscape. Permanent residents showed few correlations with landscape structure, and temperate migrants were associated with habitat diversity and edge attributes rather than with the amount, size, and dispersion of forest habitats. Given the variation in how species with different life history strategies (e.g., migration status) were associated with the arrangement of land use and land cover, the patterns observed in the eastern United States may not hold for other geographic areas.

Research that has investigated the influence of land use and land cover on bird species at macroecological scales has focused primarily on spatial variation—that is, trying to explain why bird abundance or bird communities vary from one locale to the next. What is currently lacking is a commensurate set of investigations that examine the relationship between landscape structure and bird distribution and abundance over time. In order to more fully examine the relationships between landscapes and biological diversity, there is a critical need for temporally consistent land base inventories that permit the estimation of changes in landscape composition and configuration.

### *Amphibian populations*

Within the borders of the continental United States there are approximately 230 species of amphibians (140 salamanders and 90 frogs and toads) (Bury and others 1995). The activities of humans have long been thought to affect amphibian populations, yet measures to conserve amphibians have lagged relative to other taxa (Bury and others 1980). One manifestation of this conservation neglect has been the absence of long-term data that monitors the status of amphibian populations (Pechmann and others 1991).

Many reports have documented local and regional amphibian declines in both disturbed and pristine habitats (see Blaustein and Wake 1990). However, evaluating this apparent widespread decline is made difficult by the dearth of geographically widespread and long-term survey data, which make the status of amphibian populations one of the more controversial issues in conservation biology (Reed and Blaustein 1995). Although there is general agreement that habitat destruction and other human-related impacts have resulted in some local population reductions or extinction (Pechmann and Wilbur 1994), there is less agreement concerning how widespread the decline is. The specific factors that may be contributing to the purported decline in amphibian numbers are disputed, as well. Part of the difficulty is that some amphibian populations exhibit large natural population fluctuations that may make it difficult to detect human-caused declines—a pattern that further highlights the need for long-term monitoring (Pechmann and others 1991; Hecnar and M'Closkey 1996).

Although continuous long-term records of amphibian populations are lacking, comparisons between contemporary and historical surveys do offer a more temporally extensive perspective. In a study of regional frog fauna in the Yosemite National Park area, Drost and Fellers (1996) found that five of seven frog and toad species had suffered serious declines from 1915 to 1992. These declines have occurred in sites that have been relatively undisturbed over the period of comparison, but the cause of the declines remains undetermined.

The lack of basic monitoring information notwithstanding, there have been attempts to compile status evaluations from professional herpetologists over large geographic areas. A report from the Declining Amphibian Populations Task Force summarized the findings of this compilation as of December 1993 (Vial and Saylor

Table 15—Amphibians whose population status is of conservation concern[1] (from Vial and Saylor 1993).

| Species | Pacific Coast | Rocky Mountain | North | South |
|---|---|---|---|---|
| *Acris crepitans* | | | * | |
| *Acris crepitans blanchardi* | | | * | |
| *Ambystoma californiense* | * | * | | |
| *Ambystoma cingulatum* | | | | * |
| *Ambystoma gracile* | * | * | | |
| *Ambystoma jeffersonianum* | | | * | |
| *Ambystoma laterale* | | | * | |
| *Ambystoma mabeei* | | | | * |
| *Ambystoma macrodactylum croceum* | * | | | |
| *Ambystoma macrodactylum sigillatum* | * | * | | |
| *Ambystoma maculatum* | | | * | |
| *Ambystoma platineum* | | | * | |
| *Ambystoma talpoideum* | | | * | |
| *Ambystoma tigrinum* | | * | * | * |
| *Ambystoma tigrinum stebbinsi* | | * | | |
| *Ambystoma tremblayi* | | | * | |
| *Aneides aeneus* | | | * | |
| *Aneides ferreus* | * | | | |
| *Ascaphus truei* | * | * | | |
| *Batrachoseps aridus* | * | * | | |
| *Batrachoseps attenuatus* | | | | |
| *Batrachoseps campi* | * | * | | |
| *Batrachoseps pacificus pacificus* | * | | | |
| *Batrachoseps simatus* | * | | | |
| *Batrachoseps* sp. 1 | * | | | |
| *Batrachoseps* sp. 2 | * | | | |
| *Batrachoseps* sp. 3 | * | | | |
| *Batrachoseps* sp. 4 | * | | | |
| *Batrachoseps* sp. 5 | * | | | |
| *Batrachoseps stebbinsi* | * | | | |
| *Bufo alvarius* | * | | | |
| *Bufo boreas* | * | * | | |
| *Bufo canorus* | * | | | |

| Species | Pacific Coast | Rocky Mountain | North | South |
|---|---|---|---|---|
| *Bufo exsul* | * | * | | |
| *Bufo hemiophrys baxteri* | | * | | |
| *Bufo houstonensis* | | | | * |
| *Bufo microscaphus* | | * | | |
| *Bufo microscaphus californicus* | * | * | | |
| *Bufo microscaphus microscaphus* | * | * | | |
| *Bufo nelsoni* | * | * | | |
| *Bufo wrighti* | * | | | |
| *Cryptobranchus alleganiensis* | | | * | |
| *Cryptobranchus bishopi* | | | | * |
| *Desmognathus auriculatus* | | | | * |
| *Desmognathus fuscus* | | | | * |
| *Desmognathus fuscus conanti* | | | * | |
| *Dicamptodon copei* | * | | | |
| *Dicamptodon ensatus* | * | * | | |
| *Ensatina eschscholtzii croceator* | * | * | | |
| *Ensatina eschscholtzii klauberi* | * | * | | |
| *Eurycea longicauda* | | | * | |
| *Eurycea lucifuga* | | | * | |
| *Gastrophryne carolinensis* | | | * | |
| *Gyrinophilus porphyriticus* | | | * | |
| *Hemidactylium scutatum* | | | * | * |
| *Hydromantes brunus* | * | * | | |
| *Hydromantes platycephalus* | * | * | | |
| *Hydromantes shastae* | * | * | | |
| *Hyla andersonii* | | | * | |
| *Hyla avivoca* | | | | * |
| *Hyla chrysoscelis* | | | * | |
| *Hyla crucifer* | | | * | |
| *Hyla gratiosa* | | | * | * |
| *Necturus maculosus* | | | * | |
| *Necturus* spp. | | | | * |
| *Notophthalmus viridescens* | | | * | |
| *Notophthalmus perstriatus* | | | | * |

1993). Because this synthesis was based on combining results from independent local studies, it lacked the rigor that can be achieved with regional or national monitoring protocols. Therefore, the patterns summarized here must be viewed as tentative.

A total of 120 species and subspecies were determined by Vial and Saylor (1993) to have downward population trends in the continental United States. The greatest number of species or subspecies with evidence of declining populations occurred in the Pacific Coast region, followed by the North, Rocky Mountain, and the South (table 15). The number of genera determined to have species or subspecies with declining populations indicates that the North has the greatest number of genera followed by the Pacific Coast and South, and the Rocky Mountains. Given that the greatest diversity of amphib-

Table 15 (Cont'd.)

| Species | Pacific Coast | Rocky Mountain | North | South |
|---|---|---|---|---|
| *Phaeognathus hubrichti* | | | | * |
| *Plethodon caddoensis* | | | | * |
| *Plethodon dunni* | * | | | |
| *Plethodon elongatus* | * | * | | |
| *Plethodon fourchensis* | | | | * |
| *Plethodon hubrichti* | | | | * |
| *Plethodon kiamichi* | | | | * |
| *Plethodon larselli* | * | | | |
| *Plethodon nettingi* | | | * | |
| *Plethodon ouachitae* | | | | * |
| *Plethodon shenandoah* | | | | * |
| *Plethodon stormi* | * | * | | |
| *Plethodon vandykei* | * | | | |
| *Plethodon wehrlei* | | | * | |
| *Pseudacris (Hyla) regilla* | * | | | |
| *Pseudacris brachyphona* | | | * | |
| *Pseudacris feriarum* | | | * | |
| *Pseudacris streckeri* | | | * | |
| *Pseudacris streckeri illinoensis* | | | * | * |
| *Pseudacris streckeri sterckeri* | | | | * |
| *Pseudacris triseriata* | | | * | |
| *Pseudotriton montanus* | | | * | |
| *Pseudotriton ruber* | | | * | |
| *Rana a. aurora* | * | * | | |
| *Rana a. draytoni* | * | * | | |
| *Rana areolata* | | | * | |
| *Rana aurora* | * | | | |
| *Rana blairi* | | * | * | |
| *Rana boylii* | * | | | |
| *Rana capito capito* | | | | * |
| *Rana capito* | | | | * |
| *Rana cascadae* | * | * | | |
| *Rana chiricahuensis* | | * | | |
| *Rana clamitans* | | | * | |
| *Rana muscosa* | * | * | | |
| *Rana pipiens* | * | * | * | |
| *Rana pretiosa* | * | * | | |
| *Rana spp.* | | * | | |
| *Rana subaquavocalis* | | * | | |
| *Rana sylvatica* | | | * | |
| *Rana tarahumarae* | | * | | |
| *Rana utricularia* | | | * | |
| *Rana virgatipes* | | | * | * |
| *Rana yavapaiensis* | * | * | | |
| *Rhyacotriton olympicus* | * | | | |
| *Rhyacotriton variegatus* | * | * | | |
| *Scaphiopus bombifrons* | | | * | * |
| *Scaphiopus couchii* | * | * | | |
| *Scaphiopus hammondii* | * | * | | |
| *Scaphiopus holbrookii* | | | * | |
| *Scaphiopus holbrookii holbrookii* | | | * | |
| *Taricha torosa torosa* | * | * | | |
| *Typhlotrition spelaeus* | | | | * |
| Total number of species/ subspecies: | 51 | 39 | 42 | 26 |
| Number of genera: | 14 | 12 | 17 | 14 |

*[1]Species of conservation concern were defined from Vial and Saylor (1993) and include those species with the following status categories: Decline observed, federal or state threatened and endangered, vulnerable, rare, sensitive, regional concern, locally absent, extinct, federal candidate species, state species of special concern, state critical, state vulnerable, and candidate for state status. Species with International Union for the Conservation of Nature and Natural Resources (IUCN) categories of extinct, extinct in the wild, critical, endangered, vulnerable, and susceptible were also included.*

ian species occurs in the southeastern United States (Currie 1991), it is somewhat of a surprise that there are not more species with declining trends in the South.

Many factors have been hypothesized to explain the declines in amphibians. Direct habitat destruction or contamination have been implicated in several documented population declines (Petranka and others 1993; Hecnar and M'Closkey 1996). As noted by Pechmann and Wilbur (1994), there is little contention that habitat loss and pollution are contributing agents, but the widespread nature of declines and population reductions in less disturbed areas point to broader environmental agents. Acidification (Dunson and others 1992; Wissinger and Whiteman 1992), ultraviolet-B radiation (Blaustein and others 1994*a*), introduced predators (e.g., bullfrogs and fish) (Fisher and Shaffer 1996), and pathogens (Blaustein and others 1994*b*) have all been proposed as likely candidates. Still other researchers are proposing interactive effects among these agents (Long and others 1995). While it seems likely that many factors are contributing to the decline in amphibians, the relative importance of each factor will probably depend on species and locale. Furthermore, it is unlikely that determining causation in amphibian declines will be resolved without long-term monitoring and experimental manipulation (Blaustein 1994).

## Imperiled Species

An estimated global extinction rate that appears to be unprecedented in geological time (May 1990) has heightened concern for increasing rarity among the nation's biota. Moreover, this elevated extinction rate is being attributed to the activities of humans rather than the result of some calamitous natural disaster (Pimm and others 1995). Traditionally, conservation efforts to slow the loss of biodiversity have focused on species with few remaining individuals under the assumption that they are the most vulnerable to extinction (Sisk and others 1994). Consequently, rarity has been used as an important criterion for identifying which species should be the focus of conservation efforts.

The Endangered Species Act of 1973 (ESA) has epitomized this species-by-species conservation strategy (Doremus 1991). The ESA and its subsequent amendments codified broad-ranging protection for all species, plant or animal, encompassing species in immediate danger of extinction and species that may be threatened with extinction in the foreseeable future. The rate at which species are being listed as threatened and endangered has raised concerns that the current single-species emphasis to recovery will be overwhelmed by the sheer number of species requiring protection.

Much of what is reviewed in this section represents an update of Flather and others (1994). That report reviewed species listing trends and the geographic distribution of threatened and endangered species based on that set of species that were listed as threatened or endangered as of August 1992. Since that time, more than 360 species have been added to the endangered species list. We reiterate some of the past trends noted in Flather and others (1994) for historical completeness, but also discuss those trends that have manifested since that earlier report.

### *Recent trends in species listings*

Over the last two decades the number of threatened and endangered species annually listed has varied greatly.[15] During the 1981 and 1983 calendar years, there was a net addition of only four species, while greater than 120 species were listed in 1994. Based on cumulative plots of species listed since July 1976, we identified three phases in the listing history (figure 31*a*).

Phase 1 (July 1976 to October 1979) was characterized by episodes of listing activity. The first plant listings occurred during phase 1 and the pulse of new listings toward the end of this phase was primarily due to a mass listing of cactus species that were threatened by domestic and international commerce (USDI Fish and Wildlife Service 1979). Overall, 94 species were added to the list during this time period.

The 56-month phase 2 (November 1979 to July 1984) is conspicuous by its inactivity. The paucity of new listings is explained, in part, by 1978 amendments that required both designation of critical habitat with due consideration to economic impact and numerous new stipulations for hearings and local notice (Kohm 1991). Logistically, these changes were burdensome and led to a very protracted process for evaluating the merits of listing proposals. A net gain of 38 species was observed during phase 2.

From August 1984 until the moratorium on species listings in April 1995 (phase 3), the species listing rate increased greatly with a total net addition of 651 species. The increase can again be traced to ESA amendments passed in 1982 and 1988 that were directed at expediting the listing process (Kohm 1991). Plant species were listed at twice the rate of animal species and comprised the majority (55%) of the listed biota by the end of this listing phase (figure 31*a*). The listing rate for vertebrates and invertebrates was nearly equal. Fish species now dominate numerically among vertebrate taxa, and they have been listed at a rate exceeding twice that of other vertebrates (figure 31*b*). Invertebrates comprised 13% of the

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*[15] In July of 1976, the U.S. Fish and Wildlife Service initiated publication of a technical bulletin to assist in information exchange among agencies and organizations involved or interested in the Endangered Species Program (USDI Fish and Wildlife Service 1976). We used these technical bulletins to chronicle shifts in the emphasis given to major taxa and changes in species listing rates.*

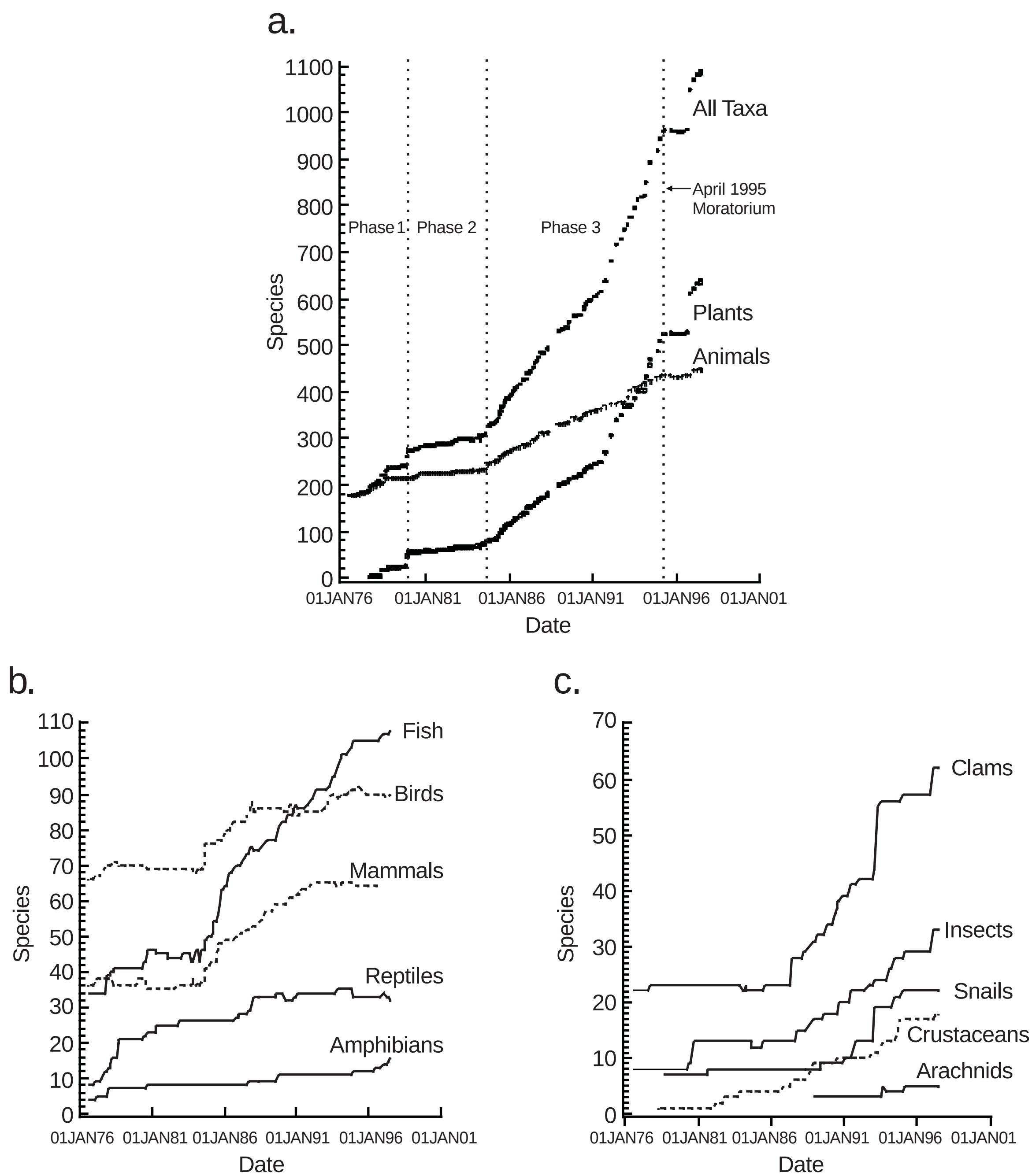



Figure 31. Cumulative plots of the number of species listed as threatened or endangered from July 1976 - June 30, 1997 for (a) plants and animals, (b) vertebrates, and (c) invertebrates (U.S. Fish and Wildlife Service Endangered Species Technical Bulletins)

listed taxa and more clams were added (35 new species listed) during this phase than any other animal taxon except fish (figure 31*c*). Under the criterion of "numbers listed," these trends suggest a taxonomic shift away from highly visible and charismatic species.

We don't know whether the moratorium on species listings that was passed by Congress in April 1995 (USDI Fish and Wildlife Service 1995) marks the beginning of a new phase. Early trends indicate that listing has resumed at a rate consistent with pre-moratorium levels (figure 31*a*). The current backlog of species (238 species awaiting final listing decisions, 182 candidate species awaiting listing proposals [USDI Fish and Wildlife Service 1996]) could certainly sustain the pre-moratorium listing rate for many years. Whether new listings will sustain that rate observed during phase 3 will depend largely on congressional appropriations and the amendments that will shape the ESA following the long awaited reauthorization.

Even in the absence of future listings, the present number of threatened and endangered species exceeds the capacity of the U.S. Fish and Wildlife Service to fully implement the provisions of the Act (Smith and others 1993). This, along with inevitable interspecific conflicts that surface when species are considered separately (Losos 1993), has raised the question of whether a broader inspection of the ecology and recovery needs of multiple species would improve the efficiency with which species conservation can be achieved (Marcot and others 1994; Committee on Scientific Issues in the Endangered Species Act 1995).

### *Geographic patterns in species endangerment*

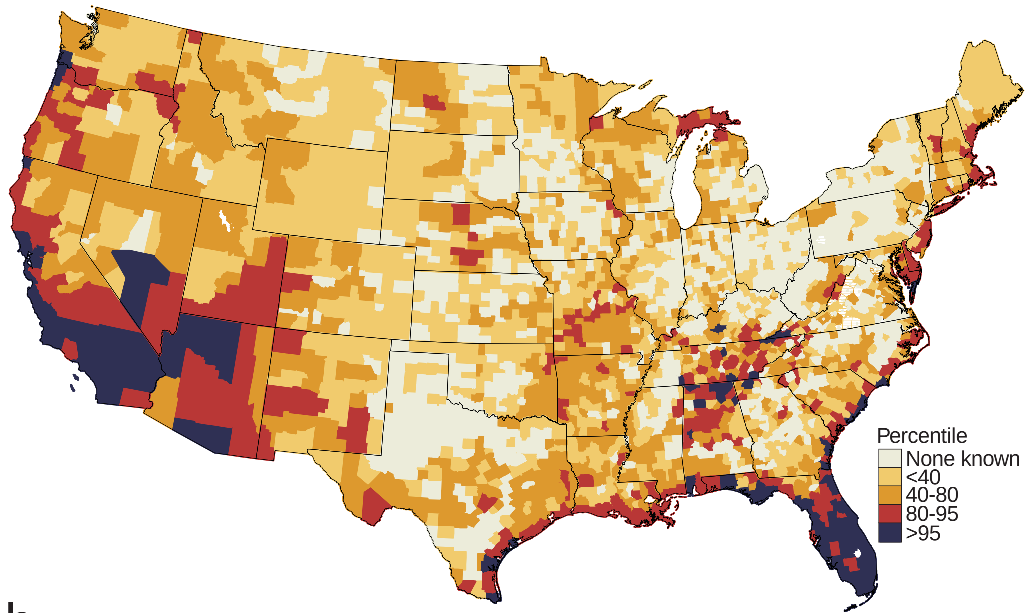
One approach that has the potential to help achieve multiple species recovery is to focus conservation efforts in those areas where threatened and endangered species are concentrated. In a study of 631 listed species that occur in the conterminous United States, Flather and others (1998) found that endangered species are concentrated in distinct geographic regions (figure 32*a*) including the southern Appalachians, coastal areas, and the arid Southwest.

Geographic areas with a concentration of endangered species, also called "hotspots," were identified by Flather and others (1998) using the criteria specified in Prendergast and others (1993)—namely, an arbitrarily defined upper percentile of sample units ranked by species counts. To account for the disparity in county area across the United States, counties were partitioned into "large-area" and "small-area" sets at a threshold of about 910,000 acres. This resulted in 45% of the conterminous United States being categorized as large-area counties. This was done because the alternative, a simple conversion to county density, produced an eastern bias and concealed known concentrations of endangered species in the arid southwest (Hallock 1991). Counties were then ranked within these large- and small-area sets according to the number of threatened and endangered species that occurred within their boundaries. Endangerment hotspots were initially located by mapping the top 5% of large- and small-area counties (i.e., those counties where the greatest number of listed species were found). The areal extent of hotspots was then defined by the top 20% of counties that were also contiguous to, or formed distinct physiographic clusters with, those counties in the top 5%. A land resource classification system developed by the USDA Soil Conservation Service (1981) was used to group those counties that supported many endangered species into regions that had similar climate, physiography, soils, vegetation, and land use. This led to the identification of 12 hotspots of high species endangerment (figure 32*b*). Of the total hotspot area (estimated as the sum of the area of all counties within defined hotspots), 40% occurs in the Rocky Mountain region, 29% occurs in the Pacific Coast, 28% occurs in the South, and less than 2.5% occurs in the North.

The endangered species inhabiting hotspots showed varying degrees of biological similarity as defined by taxonomic composition, prevalence of endemism, land type associations, and factors contributing to species endangerment. The listed species occurring within eastern coastal hotspots were characterized by relatively high proportions of birds and reptiles (particularly nesting marine turtles), were associated with forest and aquatic habitats, and have been negatively impacted by human development activities (e.g., residential and urban development, shoreline modification). Of the eastern coastal hotspots, Peninsular Florida had the highest level of endemism with 78% of the species having their distribution restricted to this hotspot. Endangered species located in the arid southwest hotspots were characterized by high proportions of plants and fish, were associated with rangeland and wetland habitats, and have been impacted by extractive resource uses (e.g., grazing, mining, water diversion) and the introduction of exotic species. The California hotspots were distinguished from other regions by the relatively high proportion of insect fauna that are listed, high endemism (greater than 55% across both regions), and a mixture of forest and rangeland habitat associates. Like the east coast, residential and industrial development was the most common factor contributing to species endangerment in California; however, the introduction of exotic species has also had a major impact on listed species populations. For a more detailed discussion of biological characteristics of endangerment hotspots see Flather and others (1998).

Apart from the biological characteristics shared by species occurring within endangerment hotspots, the geographic proximity of these regions to National Forest System lands is also of interest (figure 33). Among eastern hotspots, approximately 5% of the total hotspot area is included in National Forest System lands. Among western hotspots the overlap is 17%. Hotspots where the percentage of National Forest System lands exceeds 10% includes the Southern Appalachians (11%), Northern California Coastal Mountains and Valleys (14%), Colorado and Green River Plateaus (16%), Southern California Mountains and Valleys (17%), Arizona Basin and Range (23%), and Northern Pacific Coast Range and Valleys (40%). Although the hotspots are defined at a very geographically coarse scale, this analysis of overlap does indicate which endangerment regions are likely to be influenced by management on the nation's National Forests and Grasslands.

Mapping regions where many endangered species cooccur can be used to geographically identify those ecoa.




b.

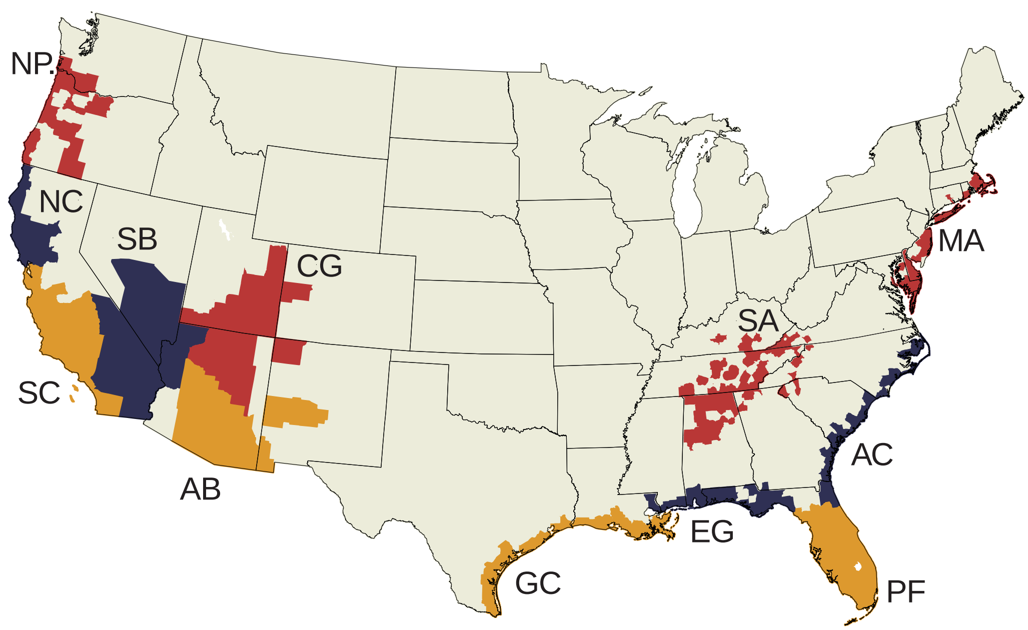



Figure 32. The geographic distribution of threatened and endangered species. The map depicted in (a) shows counties in percentile classes after ranking both large- and small-area counties (to account for differential county area) according to the number of threatened and endangered species that occurred within their boundaries. The map depicted in (b) identifies endangerment hotspots. Labels refer to endangerment regions as follows: **NP**=Northern Pacific Coast Range and Valleys, **NC**=Northern California Mountains and Valleys, **SC**=Southern California Mountains and Valleys, **SB**=Sonoran Basin and Range, **CG**=Colorado and Green River Plateaus, **AB**=Arizona Basin and Range, **GC**=Gulf Coast Marsh and Prairie, **EG**=Eastern Gulf Coast Flatwoods, **SA**=Southern Appalachians, **PF**=Peninsular Florida, **AC**=Atlantic Coast Flatwoods, **MA**=Mid-Atlantic and Northern Coastal Plain (Flather and others 1998).

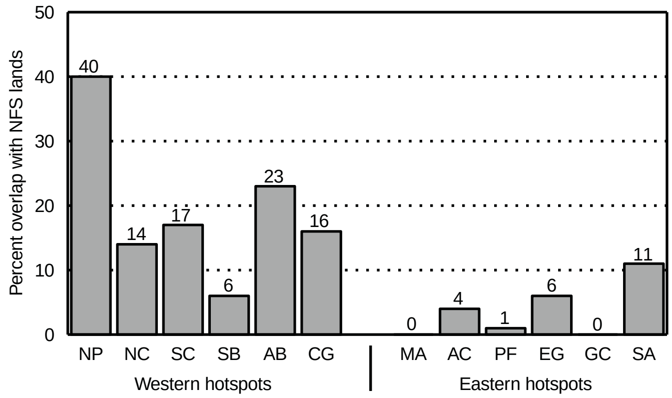



Figure 33. Percent of hotspot area that occurs on National Forest System lands. Hotspot codes are as defined in figure 32.

logical systems that are subject to high levels of endangerment stress. Knowledge of where listed species are concentrated can guide where habitat protection efforts (Orians 1993) or habitat conservation plans (Pulliam and Babbit 1997) are likely to affect the greatest number of imperiled species. Location alone, however, says little about why listed species are concentrated where they are and what population threats will have to be addressed by recovery efforts. Geographic patterns of endangered species occurrence coupled with information on those factors that have contributed to species endangerment are prerequisite to developing integrative conservation strategies (Falk 1990). They also help define characteristics of species or environments that are susceptible to endangerment (Slobodkin 1986) and are thus a means of anticipating where species endangerment may be a problem in the future.

### *Projections of endangerment hotspots*

Hof and others (1998; 1999) combined the descriptive geography of the preceding section with environmental information (e.g., climate, human populations, land use activities) across the conterminous United States for the development of statistical models that yielded predictions of where future concentrations of endangered species may occur. Statistical models relating environmental and land use characteristics to the density of threatened and endangered plants and animals were generated from a county-level data set. Because county area varies greatly, county-level data were interpolated to a uniform grid (3280 cells) across the conterminous United States. In an effort to highlight areas where species endangerment may be concentrated in the future, we used these statistical models, in conjunction with projections of human populations, land use and land cover (i.e., agriculture, rangeland, forest, urban, wetland, water), and resource extraction (timber, beef cows, irrigation, and minerals) to the year 2020 to project changes in the density of threatened and endangered species. For a detailed methodological discussion of model estimation and projections see Hof and others (1998; 1999).

Geographic areas where threatened and endangered plant and animal species are expected to increase, called future hotspots, were determined by the criteria specified by Prendergast and others (1993). The top 5% of grids with the greatest increase in plant or animal species were identified and mapped.

Future hotspots of plant species endangerment were primarily concentrated in the Southern region (figure 34*a*). Many of the areas where the greatest number of new plant species are projected to occur are found in areas where there is already a high concentration of listed species (compare with figure 32*a*). However, there are new areas of concentration projected to occur in the Southern Piedmont, the desertic mountains and basins of west Texas, and the Edwards Plateau region of west-central Texas.

Future endangerment hotspots for animals are clearly concentrated in the Pacific Coast region (figure 34*b*). The greatest absolute increase is projected to occur in coastal Washington and Oregon, the California Central Valley, and the region defined by the upper Columbia Plateau and Palouse Prairie in eastern Oregon and Washington. These future hotspots in the Pacific Coast region tend to emphasize areas where animal endangerment is currently a concern—the exception being the emergence of a potentially new area of animal species concentration in eastern Oregon and Washington.

## Trends in Wildlife Use

How humans have valued wildlife resources has shifted over time. Utilitarian and commodity-oriented valuations of wildlife that were dominant at the turn of the century have declined over time, suggesting that Americans are broadening their attitudes toward wildlife (Kellert 1987, Gray 1993). This broader base of values held by the public for wildlife indicate the growing importance of this resource to human social and economic welfare. Based on data from the most recent survey of hunting and wildlife-associated recreation (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997), hunters alone spent nearly 21 billion dollars on equipment and travel in 1996. Similarly, wildlife viewers spent 29 billion dollars in 1996 to observe, feed, or photograph wildlife. Although these direct expenditures are useful indicators of economic impact (e.g., the contribution of wildlife-oriented recreation to local and regional economies), it does not quan-

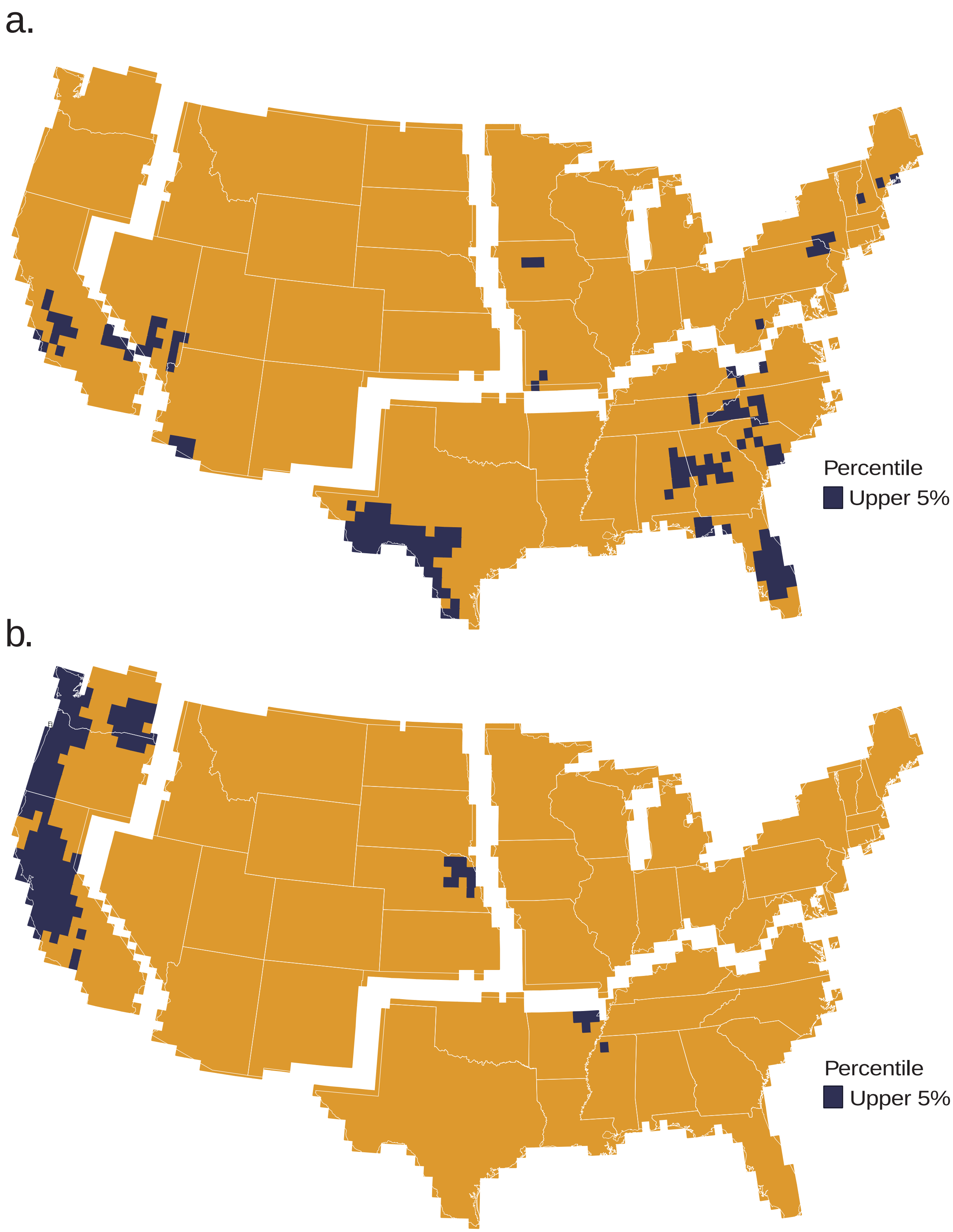



Figure 34. Areas projected to have the greatest increase (i.e., upper 5% of all grids) in plant (a) and animal (b) endangered species for each RPA Assessment region by the year 2020 (Hof and others 1999).

tify the net economic value of these activities to the individual participant.

In the context of wildlife-oriented recreation, net economic value measures the dollar amount that individuals would be willing to pay, over and above what they actually spend, to participate in some activity (Loomis 1993). Net economic value is a better measure of economic benefit to individuals than direct expenditures, and the total value to society can be estimated by summing net economic value across all individuals that participated (Waddington and others 1994). Based on a 1991 survey of people who participated in wildlife-associated recreation (USDI Fish and Wildlife Service and USDC Bureau of the Census 1993), the mean annual net economic value to each deer hunter varied from $768 in Maryland to $168 in Iowa (Waddington and others 1994). If we multiply the total number of deer hunters in each state by the mean estimates of net economic value, then the total net economic benefit to society attributable to deer hunting is estimated to be $5.2 billion. Estimates of the net economic value to wildlife observers were similar in magnitude, varying between $766 in Indiana to $106 in North Dakota. Total societal benefits associated with wildlife observation ($10.2 billion) are higher than deer hunting due to the greater number of participants.

Apart from the socioeconomic benefits attributable to wildlife resources, state wildlife agencies derive much of their revenues for managing these resources from license sales and excise taxes on firearms and ammunition. Consequently, the number of persons participating in wildlife-associated recreation not only affects rural economies and societal values but also affects state revenues for managing those resources.

## Hunting

Since 1955, trends in wildlife-oriented recreation activities have been monitored by the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation. Although methods have evolved, the survey is one of the oldest and most comprehensive recreation surveys that permits the examination of long-term trends in hunting participation by the American public. Because survey methodology has changed over time, trend analyses require adjustments to make participation estimates comparable among survey years (see USDI Fish and Wildlife Service and USDC Bureau of the Census 1997: Appendix B).

The number of hunters and the days spent hunting peaked in 1975. Since that time the number of hunters has

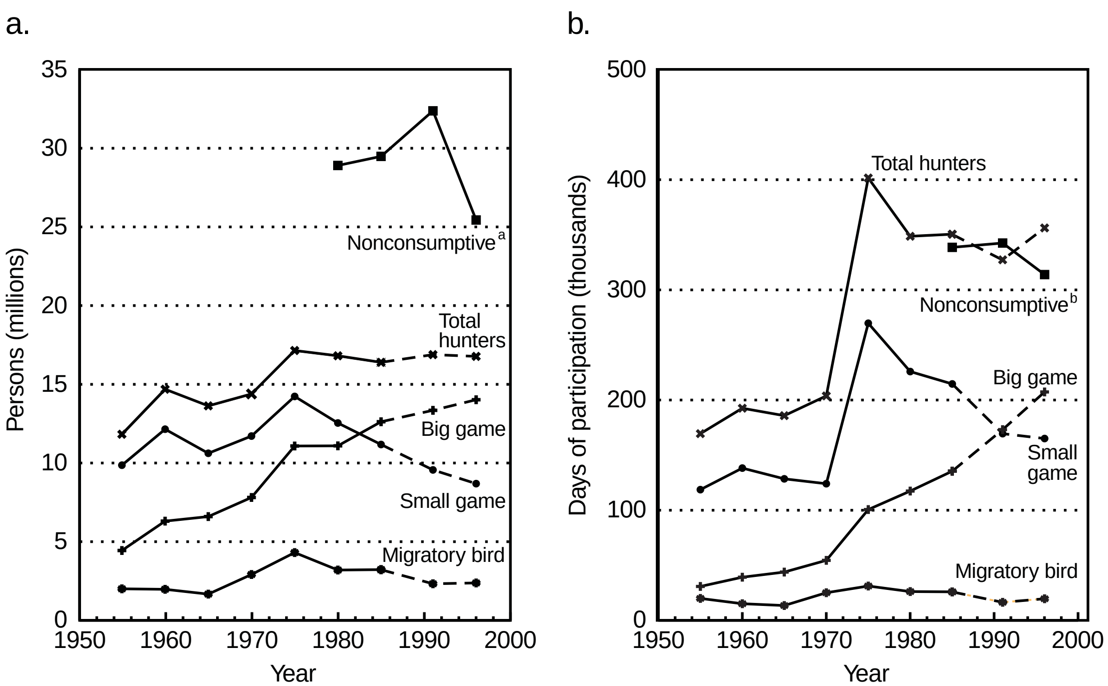



Figure 35. Trends in (a) the number of persons, and (b) the number of days that persons (≥12 years old) spent participating in recreational activities dependent on wildlife. Dashed lines represent extrapolated estimates of participation in hunting activities for 1991 and 1996 based on percentage changes applied to 1985 estimates (USDI Fish and Wildlife Service and USDC Bureau of the Census 1993; USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). [a]Trend for nonconsumptive participants was provided by Richard Aiken (pers. comm., USDI Fish and Wildlife Service, 1997). [b]Trends in nonconsumptive days have not been adjusted for differences in survey methodology.

remained relatively stable, while the number of days spent hunting has declined by about 11% from 1975 to 1996 (figure 35 on facing page). Trends in total hunting, however, hide the diverse trends associated with the type of hunting. Participation in big game hunting has increased in every survey period since 1955. Over the entire 40 year period that the surveys have been conducted, the number of big game hunters has increased by more than 210%. The number of days spent hunting big game has increased at an even greater rate with more than a 570% increase since 1955. The number of days devoted to hunting big game species has increased to the point where more days are spent hunting big game than any other category of hunting. In contrast, the number of small game hunters has declined at a nearly constant rate since the mid-1970s. The number of persons participating in and the number of days spent hunting small game have both declined by nearly 40% from 1975 to 1996. Over the same time period, there has been a similar proportionate decline in migratory game bird hunters (-45%) and days of participation (-37%); however, the most recent survey period indicates slightly increasing participation levels in migratory bird hunting.

Regional trends in the number of hunters were obtained by summing state-level estimates for those states comprising each RPA Assessment region. Because state-level numbers have not been adjusted for differences in survey design, we were only able to examine the 1991 and 1996 surveys that share a common methodology. Short-term regional hunter trends were consistent with the national trends in most cases (table 16). Exceptions include small game hunters in the West and migratory bird hunters in the Rocky Mountain region. The number of small game hunters increased in both the Rocky Mountain and Pacific Coast regions, despite substantial national declines in small game hunters. The increased number of small game hunters in the Rocky Mountains, however, did not keep pace with overall population growth resulting in a declining proportion of the population participating from 6.8 to 6.3%. Migratory game bird hunters in the Rocky Mountain Region declined by 31% compared to a 2% gain nationally.

Many factors influence hunting participation rates, including wildlife population status, the availability of places to hunt, human demographics, and socioeconomics. The participation trends observed are consistent with those expected given our previous review of population status. The two most commonly sought big game species are deer (95% of all big game hunters hunt deer) and wild turkey (19% of big game hunters) (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). The populations of these species have been increasing substantially over the last 20 years. Conversely, many of the species sought by small game hunters have been declining in some regions, which may discourage participation.

Changes in populations of harvested species, however, do not explain entirely the trends in hunting participation. In 1996, 51% of all hunters hunted only on private land, 17% hunted only on public land, and 30% hunted on both public and private land (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). Given the importance of private land to hunters, access to private lands has the potential to affect participation in hunting. However, only 2.5% of hunters in 1996 who would have liked to have hunted more indicated that access or too few places to hunt was a constraint (USDI Fish and Wildlife

Table 16--Regional trends in the number of hunters and nonconsumptive users 16 years old and older (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997).

| | Big game | | | Small game | | | Migratory bird | | | Nonconsumptive | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Region | 1991 | 1996 | Change (%) | 1991 | 1996 | Change (%) | 1991 | 1996 | Change (%) | 1991 | 1996 | Change (%) |
| | thousands | | | | | | | | | | | |
| Total U.S.<br>(% of population) | 10,745<br>(5.7) | 11,288<br>(5.6) | 543 (5.1) | 7,642<br>(4.0) | 6,945<br>(3.4) | -697 (-9.1) | 3,009<br>(1.6) | 3,073<br>(1.5) | 64 (2.1) | 29,999<br>(15.8) | 23,652<br>(11.7) | -6,347 (-21.2) |
| North | 5,790<br>(6.7) | 5,879<br>(6.6) | 89 (1.5) | 4,025<br>(4.6) | 3,586<br>(4.1) | -439 (-10.9) | 892<br>(1.0) | 1,040<br>(1.2) | 148 (16.6) | 14,344<br>(16.6) | 11,043<br>(12.5) | -3,301 (-23.0) |
| South | 3,648<br>(6.2) | 4,041<br>(6.2) | 393 (10.8) | 2,574<br>(4.3) | 2,262<br>(3.5) | -312 (-12.1) | 1,494<br>(2.5) | 1,493<br>(2.3) | -1 (-<1) | 7,635<br>(12.9) | 6,441<br>(10.0) | -1,194 (-15.6) |
| Rocky Mountain | 1,405<br>(9.9) | 1,544<br>(9.6) | 139 (9.9) | 968<br>(6.8) | 1,010<br>(6.3) | 42 (4.3) | 417<br>(2.9) | 289<br>(1.8) | 128 (-30.7) | 2,951<br>(20.8) | 2.488<br>(15.4) | -463 (-15.7) |
| Pacific Coast | 691<br>(2.3) | 865<br>(2.7) | 174 (25.2) | 428<br>(1.5) | 533<br>(1.7) | 105 (24.5) | 331<br>(1.1) | 386<br>(1.2) | 55 (16.6) | 5,034<br>(17.1) | 3,648<br>(11.5) | -1,386 (-27.5) |

Note: Detail does not add to total because of multiple responses. U.S. totals include responses from participants residing in the District of Columbia.

Service and USDC Bureau of the Census 1997). Far more common reasons for not being able to hunt as much as one would like were "family or work obligations" (66%) and "not enough time" (29%). Lack of leisure time appears to be a more influential factor in hunting participation rates than restricted access. One must recognize, however, that the reasons given for not participating more in hunting only address the perspectives of those persons who hunted in 1996; it does not provide insights into why people who would like to hunt did not hunt at all in 1996.

Increasing costs associated with hunting also may be reducing participation. In 1991, a total of 572,000 hunters owned or leased land for hunting and spent on average $5,267 per hunter (USDI Fish and Wildlife Service and USDC Bureau of the Census 1993). The number of hunters with expenditures for owning or leasing land grew to 1.4 million in 1996 (a 2.5 fold increase), yet the average amount spent declined to $2,203 per hunter (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). Trends observed for private land user fees show a different pattern. Like those that bought or leased land for hunting, the number of persons that paid private land users-fees increased from 703,000 hunters in 1991, to 930,000 hunters in 1996. Unlike owning and leasing land, the fees paid for private land access doubled from 1991 to 1996 (each hunter spent an average of $173 in 1991 compared to $348 in 1996). Clearly the number of hunters that own, lease, or pay private access fees for hunting purposes has increased dramatically in the last five years.

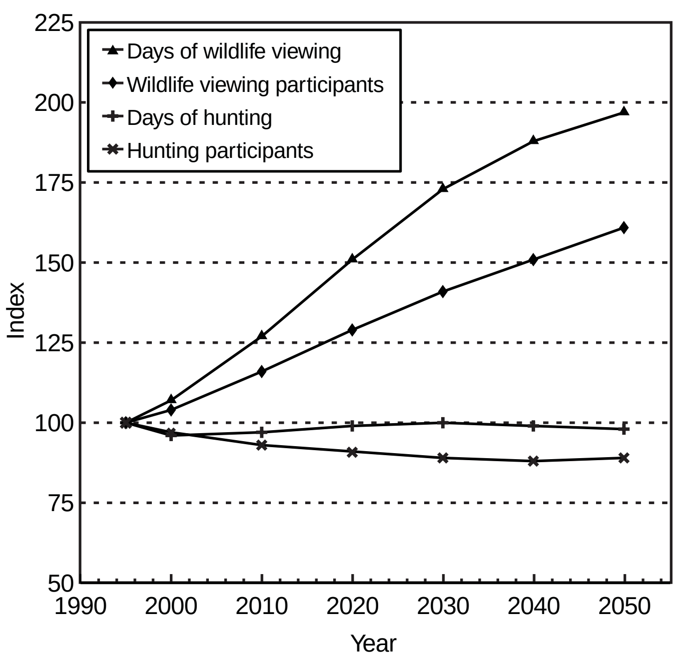



Figure 36. Projected indexed trends (1995=100) in hunting and nonconsumptive wildlife activities from 1995 to 2050 (Bowker and others 1999).

Whether these costs are becoming prohibitive, and therefore reducing hunting participation rates, will require further research. It is interesting to note, however, that only 7% of those who hunted in 1996, and would have liked to have hunted more, cited cost as a constraint (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997).

In an effort to understand some of the demographic and socioeconomic factors that influence a person's decision on whether to participate in hunting, Bowker and others (1999) developed empirical models of participation and recreation intensity (i.e., days of participation). They found that the factors most associated with hunting behavior were gender, race, and population density. Males, whites, and persons living in rural settings are more likely to hunt than females, nonwhites, or persons living in areas with high population densities. Bowker and others (1999) used these empirically estimated relationships to project hunting participation to the year 2050 based on projections of demographic and socioeconomic model variables. Projected number of hunters and days of hunting (figure 36) do not indicate any major deviation from the historical trends observed in the last 15 years. The number of hunters is expected to decline slightly (11% decline from 1995 to 2050), a decline that is attributed to the increasing proportion of nonwhites in the population, increasing human populations, and the associated decline in rural residents. The number of days spent hunting by participants is projected to remain relatively stable over the projection period (figure 36).

Regional projections do vary around the national pattern (table 17). The number of hunters is expected to increase by 20% in the Rocky Mountain region, to remain stable in the North, and to decline by 36% in the South and Pacific Coast regions (Bowker and others 1999). The number of days spent hunting is expected to increase in both the Rocky Mountains and the North (22% and 12%, respectively), and to decline in the South and Pacific Coast (-30% and -19%, respectively).

Table 17—Regional projections of participation in hunting and nonconsumptive wildlife recreation from 1995 to 2050. Estimates reflect the percent change over the 55-year period (Bowker and others 1999).

| Activity | North | South | Rocky Mountain | Pacific Coast |
|---|---|---|---|---|
| Days of wildlife viewing | 76 | 120 | 94 | 114 |
| Wildlife viewing participants | 40 | 86 | 70 | 77 |
| Days of hunting | 12 | -30 | 22 | -19 |
| Hunting participants | -1 | -36 | 20 | -36 |

## Nonconsumptive Wildlife Recreation

The number of people who take trips for the primary purpose of viewing, feeding, or photographing wildlife is substantial. In 1996, nearly 24 million people participated in what is now being called wildlife watching (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). Most nonconsumptive wildlife recreation involves the simple observation of wildlife (97%). Substantially fewer individuals photograph (51%) or feed (42%) wildlife on trips away from their residence. Recent historical participation trends in this activity, however, appear to contradict the conventional wisdom that nonconsumptive activities dependent upon wildlife are becoming more popular among U.S. citizens (Duffus and Dearden 1990). Although the number of persons and days of participation increased through the 1980s, participation in this activity appears to have declined substantially since 1991. From 1991 to 1996, the number of nonresidential (activities taking place away from the residence) wildlife watchers declined by 21% and the number of days spent viewing wildlife away from the home declined by 8% (figure 35). Although it is more common to participate in nonconsumptive recreation than to hunt, the rate of decline in wildlife watching exceeds that of all types of hunting.

Trends by Assessment region also indicate declining participation in wildlife viewing (table 16), but the magnitude of the decline does vary. The South and the Rocky Mountain regions showed the least decline in the number of persons participating in wildlife watching (both declined by nearly 16%). The North declined at a rate similar to the national trend (-23%), while the Pacific Coast region declined at the greatest rate (-27%). The reasons for these declines is unknown but may be related to some of the same issues that affect participation in consumptive recreational activities, namely a reduction in leisure time available to devote to wildlife watching (see Goodale 1991; Schor 1991).

Projections of participation in nonconsumptive wildlife recreation indicate that wildlife watching is expected to gain in popularity in the future. Both the number of persons participating in wildlife viewing and the number of days devoted to wildlife observation are expected to increase substantially by 2050 (Bowker and others 1999). The number of participants is projected to increase by 61%, while the number of days is projected to increase by 97% (figure 36). Gender (more women participate than men) and age (greater participation among older age classes) were significant predictors of participation in wildlife viewing activities. The aging of the U.S. population over the next several decades is the main reason this activity is expected to gain in popularity. Regional participation projections indicate that the South and Pacific Coast are expected to see the greatest gains in the popularity of wildlife watching (table 17).

A comparison of recent participation trends and participation projections reveals a discrepancy between what has occurred in the recent past and the expected future with respect to nonconsumptive wildlife recreation. An explanation for this discrepancy is likely multifaceted. The historical trends and participation projections are based on two different survey methodologies that vary in their definition of nonconsumptive wildlife recreation.[16] In addition, the recreation projections reported here were based on cross-sectional (i.e., for one point in time) patterns of association between participation in various activities, and demographic and socioeconomic factors. As a result, information concerning recent historical participation patterns are not factored into the projection modeling approach. Regardless of the reason, this discrepancy highlights the need for further research on what motivates people to participate in nonconsumptive wildlife recreation activities.

## Trapping

Although most persons participating in trapping do so primarily for sport and wildlife appreciation rather than income (USDI Fish and Wildlife Service 1988; Siemer and others 1994), we discuss trapping as a commercial venture given that pelts are sold in the marketplace regardless of the personal motivations for participating in this activity.

From 1970 to 1995, the number of trapping licenses sold has peaked twice—once during the period 1979-1981 and again in 1987, both at about 400,000 licenses (figure 37). Although many trappers participate for recreational reasons, their participation is affected by pelt prices (figure 37). Peak license sales occurred following a rapid increase in pelt prices (measured as the average across all species). From 1978 to 1979, and from 1986 to 1987, average prices rose in excess of 75%. After a period of relative high license sales in the mid- to late-1980s, participation in trapping has declined by nearly 60% from the peak in 1987. Concurrent with the drop in participation, average pelt prices also dropped by 50%. The number of trappers and average pelt price has remained relatively stable during the 1990s at about 160,000 license holders and an average pelt price of just over $20. An impending European Union ban on imported furs from animals caught in leghold traps has the potential to further reduce price and participation in trapping.[17]

*[16] M. Bowker, pers. comm., USDA, Forest Service, Southern Research Station, 1998.*

*[17] Unpublished report, Southwick Associates. 1993. An economic profile of the U.S. fur industry. Report on file at the Rocky Mountain Research Station.*

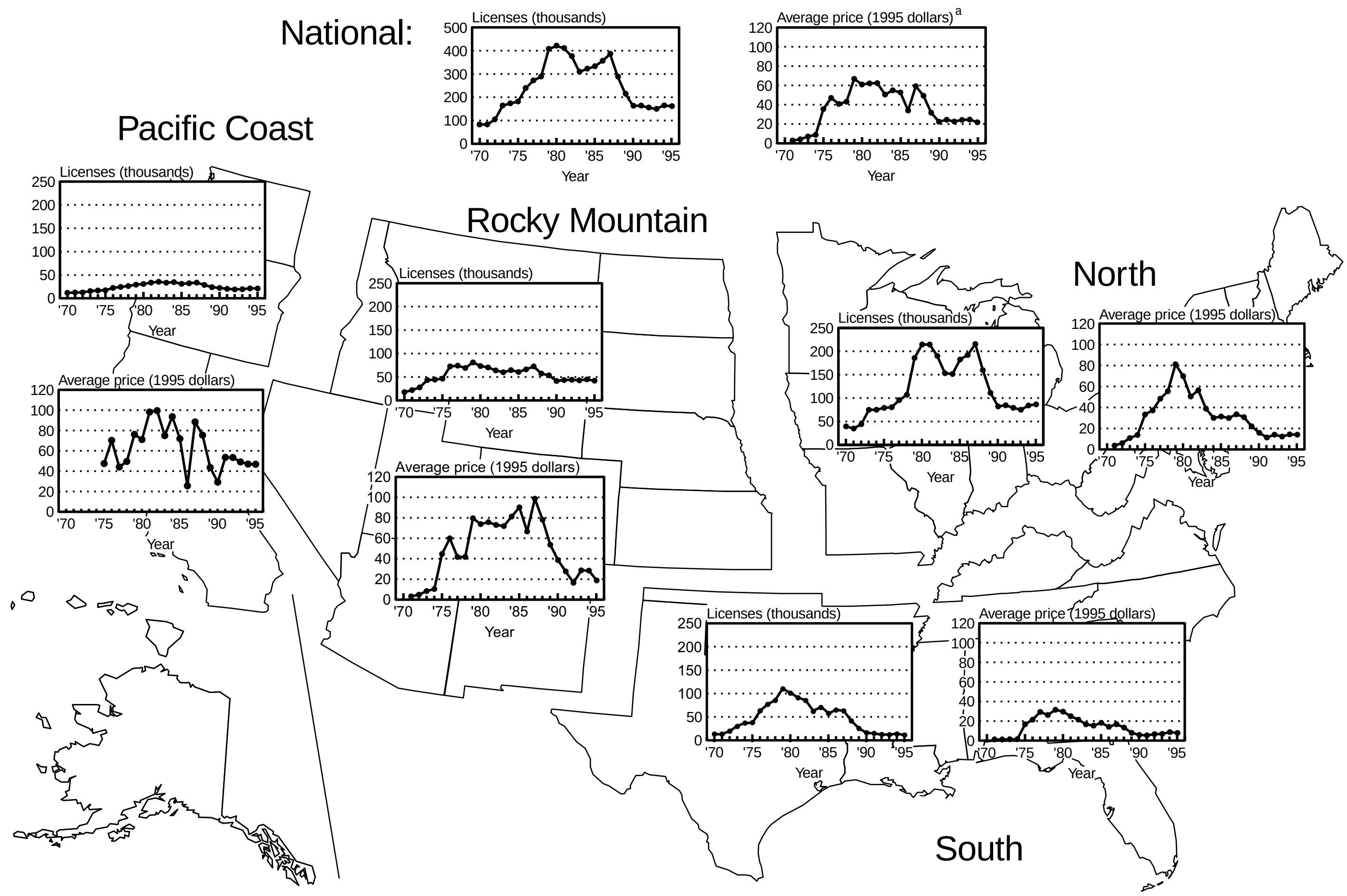



Figure 37. National and regional trends in trapping licenses and average pelt prices from 1971 to 1995 (R.G. Frederick, pers. comm., Department of Experimental Statistics, Louisiana State University, 1997).

The relationship between pelt prices and participation was also observed in a study of New York fur trappers by Siemer and others (1994). They found that nature appreciation was the strongest motivation to trap and that obtaining an income ranked low as a motivational factor. However, among inactive trappers, which constitutes about half of the license holders, low pelt prices were cited as the most important factor explaining their inactivity.

Regional patterns in trappers and price do not show strong deviation from national trends (figure 37). Because nearly half of all trappers reside in the North, the trends observed in the North are most consistent with those observed nationally. Peak prices in pelts that occurred in the late-1970s was primarily an eastern phenomenon, while the peak prices that occurred in the late-1980s is attributed to prices observed in the western United States.

## Wildlife-Associated Recreation on National Forest System Lands

Public lands in general, and National Forest System (NFS) lands in particular, provide diverse opportunities to wildlife recreationists. The National Forest System comprises 191 million acres on 156 National Forests and 19 National Grasslands. Although concentrated among western states, these lands are well distributed throughout the country. Because NFS lands are widely distributed (nearly all U.S. residents live within 300 miles of NFS land [Hunt 1988]), they represent a valuable network for supporting wildlife-associated recreational opportunities.

In 1996 nearly 50% of all hunters spent some time on public lands and 30% of hunting days were devoted entirely, or in part, to hunting on public lands (USDI Fish and Wildlife Service and USDC Bureau of the Census

1997). In a recent examination of the 1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation, Maharaj and Carpenter[18] estimated that more than 10% of the days spent hunting were spent on National Forest lands—a percentage that varied from a low of just over 7% in the South to a high of 27.5% in the Pacific Coast (table 18). Furthermore, hunting on National Forests was estimated to generate about 2.1 billion dollars in retail sales, with hunters spending an average of $76 per person per day (table 18).

The proportion of nonresidential nonconsumptive wildlife recreationists that visited publics lands is much greater than hunters. In 1996, nearly 85% of wildlife-watching participants recreated on public land (USDI Fish and Wildlife Service and USDC Bureau of the Census 1997). Maharaj and Carpenter[18] also observed an increased proportion of nonconsumptive wildlife participant on NFS lands relative to hunters. In 1996 nearly 17% of the total days spent viewing wildlife took place on National Forests with a low of nearly 11% in the North and a high of nearly 27% in the Rocky Mountains (table 18). Although the total retail sales attributed to wildlife viewing on National Forests is about the same as that estimated for hunters (2.1 billion dollars), the average expenditure per participant per day was lower ($40) (table 18).

[18] *Unpublished report, Maharaj, V.; Carpenter, J. 1999. 1996 economic impact of fishing, hunting, and wildlife-related recreation on National Forest lands. Report prepared by American Sportfishing Association for the USDA Forest Service's Wildlife, Fish, and Rare Plants Staff. Washington, DC.*

## Conclusions

Our purpose here was to review recent trends in wildlife resources, including habitats, populations, harvests, and users. The motivation for this review stems ultimately from the legislated requirements specified in the RPA. However, resource management philosophies are undergoing a period of introspection and considerable thought is being directed toward the question of how to incorporate sustainability concepts into resource management decisions (Lubchenco and others 1991; Costanza and others 1992; Levin 1993; Holdgate 1994). Therefore, we have also been motivated to present resource trend information that is relevant to evaluations of sustainable management and ecosystem health. Clearly, a comprehensive evaluation of sustainability cannot be accomplished by looking only at wildlife resources. However, because human land use activities affect the amount, distribution, and quality of habitat, which affects the distribution and abundance of animals, trends in wildlife resource attributes do appear to reflect some properties of ecosystem health.

Because wildlife resources are diverse, synthesis of our resource trends is burdensome owing to the sheer number of habitat, species, and user groups discussed. Consequently, it is difficult to evaluate how the findings from this report collectively relate to resource sustainability. What is needed is an organizing framework around which to summarize the results we present. That framework

Table 18--Summary of participation and economic impact of hunting and nonconsumptive wildlife recreation on National Forest System lands in 1996 (Maharaj and Carpenter[a] and USDI Fish and Wildlife Service and USDC Bureau of the Census 1997).

| | Hunting | | | | Nonconsumptive | | | |
|---|---|---|---|---|---|---|---|---|
| Region | Days on National Forest System lands (x1,000) | % of total[b] days | Retail sales (x 1,000 dollars) | Expenditures per person per day (dollars) | Days on National Forest System lands (x1,000) | % of Total[b] Days | Retail sales (x 1,000 dollars) | Expenditures per person per day (dollars) |
| Total U.S. | 27,797 | 10.8 | 2,111,387 | 76 | 53,014 | 16.9 | 2,135,445 | 40 |
| North | 9,205 | 8.1 | 501,459 | 54 | 14,801 | 10.9 | 406,768 | 27 |
| South | 7,338 | 7.2 | 469,551 | 64 | 13,584 | 16.6 | 594,755 | 44 |
| Rocky Mountain | 6,366 | 25.5 | 646,518 | 102 | 11,696 | 26.9 | 554,918 | 47 |
| Pacific Coast | 4,887 | 27.5 | 493,860 | 101 | 12,932 | 25.0 | 579,005 | 45 |

[a]Unpublished report, Maharaj, V.; Carpenter, J. 1999. 1996 economic impact of fishing, hunting, and wildlife-related recreation on National Forest lands. Report prepared by American Sportfishing Association for the USDA Forest Service's Wildlife, Fish, and Rare Plants Staff. Washington, DC.
[b]Estimate of the percentage of days that are spent recreating on National Forest System lands relative to the total number of days spent hunting or participating in nonresidential nonconsumptive wildlife recreation.

would include identifying a set of resource indicators that would signal the health of wildlife resources. International efforts, coordinated by the United Nations, have been initiated to develop a set of criteria and indicators for the conservation and sustainable management of forest ecosystems. As reviewed in Coulombe (1995), the United States is currently participating in such an effort directed at temperate and boreal forests through what is referred to as the "Montreal Process." An initial set of sustainability indicators was specified and agreed to by participating Montreal Process countries in a formal document entitled the "Santiago Declaration" (so named because agreement was reached at a meeting in Santiago, Chili, in February 1995). Although the Santiago Declaration has proposed an initial set of indicators for temperate and boreal forests, the actual set of indicators that will be monitored will vary from country to country and will change based on scientific review. One of the difficulties in indicator development is that the concepts of sustainability and ecosystem health lack strict definition. There is little agreement on the set of measurable indicators that will permit an unambiguous evaluation of whether human activities are diminishing system health or sustainability (Lélé and Norgaard 1996).

In the absence of a recognized set of wildlife indicators, we have organized our findings around three, albeit ad hoc, categories of resource condition. For each wildlife resource attribute (i.e., habitat, population, harvest, and users), we list those findings that we felt were indicative of a favorable, uncertain, or unfavorable resource condition. Although many definitions of system health and sustainability have been proposed (Dovers and Handmer 1993; Bertollo 1998), there is little consensus on those evaluation criteria that would allow these concepts to be implemented in resource planning (Gatto 1995; Lélé and Norgaard 1996). In the absence of operational definitions of system health and sustainability, our assignment of a particular resource trend to a condition category was necessarily subjective. An "unfavorable" resource condition was defined as those cases where recent historical trends indicated severe rarity (of habitat or species) caused by human activities or a trajectory toward rarity. We defined historical resource trends as "favorable" if they were recovering from an unfavorable state or if current resource status was meeting long-term goals as specified by managers. "Uncertain" resource conditions involved those where the empirical basis for evaluating national and regional trends was inadequate or there were conflicting resource trends that prevented assignment to either the favorable or degraded condition classes.

In light of these resource condition definitions, the "user of wildlife resources" attribute warrants some remarks. Humanity is a component of ecosystem structure and function. In fact, most ecosystem dynamics cannot be understood without accounting for human activities (Vitousek and others 1997). Consequently, trends in human uses of wildlife can indicate if wildlife habitat or populations are sufficient to meet human demands or if there are shifts in how wildlife resources are valued by the public. This can affect the economies of human communities that offer wildlife-related goods and services to the public, which affects how land is used (e.g., intensive agriculture vs. natural area) and the focus of wildlife management (e.g., hunting vs. wildlife observation).

A summary of how we interpreted recent wildlife resource trends with respect to our resource condition classes is provided in table 19. Notable favorable resource trends include the stability of forest habitats in general; recent gains in forested wetlands; recent improvement in wildlife habitats associated with agriculture due to the CRP; increases in big game, waterfowl, and some small game species; breeding bird populations in general; and participation in big game hunting. Unfavorable resource situations that warrant management focus include an imbalance in the mix of forest cover and age classes; the high number of endangered rangeland ecosystems; wetland habitats; some small game and migratory game bird populations; grassland nesting birds; and the increasing number of species being added to the threatened and endangered species list. Uncertain resource conditions often were related to inadequate inventories including rangeland condition; amphibian and furbearer population status; and mammalian small game species. Other resource trends that we considered uncertain include recent gains in agricultural and wetland habitats attributable to agricultural programs that may not continue in the future; overabundance of certain big game and waterfowl populations; declining participation in hunting and trapping; and trends in wildlife viewing, which has declined recently but is projected to increase greatly in the future.

Managing wildlife resources can become only more challenging in the future as a growing human population competes for the ecosystem goods and services that are provided by forest and rangelands. Because the Forest Service has resource stewardship responsibility for a substantial portion of the nation's land base (191 million acres), it has the opportunity to shape the wildlife resource situation on National Forests and Grasslands in the future. This opportunity also extends to cooperative assistance programs on private lands and to the promotion of research within and outside the agency. This Assessment has tried to provide a technical and objective basis for identifying wildlife resource issues that should be considered in the Forest Service's strategic planning process.

Table 19--Summary of wildlife resource attribute status. Numbers appearing parenthetically are a page number index.

| | Resource condition | | |
|---|---|---|---|
| Resource attribute | Favorable | Uncertain | Unfavorable |
| Habitat | **Forest** area has remained stable for the past several decades. Commercial forest land has declined, but this was caused by conversion to special uses including wildlife habitat protection areas (4, 6). Most of this conversion occurred in the Rocky Mountain and Pacific Coast regions.<br><br>Use of **rangelands** by livestock appears to be declining (12), but it does vary by Assessment region. Cattle numbers have increased in the Rocky Mountains and South since the last Assessment (12).<br><br>**Agricultural habitat quality** has increased recently due to the enrollment of highly erodible cropland into the Conservation Reserve Program (CRP)--with much of this conversion occurring in the Rocky Mountain region (4-5, 14, 17).<br><br>**Erosion rates** have declined over the last decade nationally and in all Assessment regions (16).<br><br>No net loss of **wetlands** in the Rocky Mountain region from 1982 to 1992 (18) and an increase of **forested wetlands** in the North (18). | **Forest** cover types and age classes have shifted over time (7-9). Such shifts will benefit some species and harm others.<br><br>Although **rangeland** area has declined recently (4, 9), there is a general lack of data that permitted an evaluation of range condition (9). Much of the decline in rangeland area has occurred in the South and Rocky Mountains (5).<br><br>Land that has remained as **cultivated cropland** appears to be used more intensively. Farm size, nutrient inputs, and pesticide use have all increased (14-15), but there is little direct evidence concerning the impacts of these changes on wildlife at the national level.<br><br>**Wetland** habitats have been lost at a declining rate because of agricultural programs (17-18). Whether those wetlands associated with agricultural lands uses will remain as wetlands in the future is uncertain and depends on the provisions of future Farm Bills and the economy. | **Forest** cover type conversions that are of particular concern in the East are the loss of early successional (aspen-birch) and young age classes in the North, and the loss of pine and bottomland hardwoods in the South (7, 9). In the West, forest cover types that have declined by more than 25 percent include western white pine, larch, ponderosa pine, lodgepole pine, and redwood (7).<br><br>There is a disproportionately high number of **rangeland** ecosystems that are considered endangered including tall grass prairie and savanna habitats (9, 12).<br><br>More than half of the **wetlands** that were present at colonial times have been lost (17). Although the rate of loss has declined over the last decade, **wetlands** continue to be converted, with much of the loss occurring in the South (18). Urban development has become the primary agent of wetland conversion (18-19).<br><br>**Forested wetlands** declined in the South by 51,000 acres (18). |
| Population/harvest | Many **big game** species have shown increases in both population and harvests since the last Assessment (21, 23). Notable among these are deer, elk, wild turkey, black bear, and pronghorn.<br><br>Some **small game** species, including ring-necked pheasant and cottontail, show some evidence of population increases that may be attributable to the CRP (25). | **Deer** populations may be overabundant in some portions of their range (particularly the North) and represents an important issue that is now receiving management focus (22-23).<br><br>**Deer** and **wild turkey** in the Rocky Mountain region show recent evidence of potential population and harvest declines (21, 23-24). | **Small game** populations and harvests for those species associated with grassland, early successional, and farmland habitats have declined (25-28, 29-30). Of particular concern are the trends in northern bobwhite (25, 29-30). |

Table 19 (Cont'd.).

| Resource attribute | Resource condition<br>Favorable | Uncertain | Unfavorable |
|---|---|---|---|
| Population/harvest (Cont'd.) | Trends in **duck** populations have increased substantially since the last Assessment (31). Populations of 6 of the 10 principal duck species exceed those established for long-term planning (31-32).<br><br>**Duck** harvests have shown recent gains, particularly in the Mississippi flyway (32-33).<br><br>Nearly 80 percent of **goose** and **swan** populations are either increasing or stable (33). There have been record numbers of geese harvested since 1993--a pattern driven by the harvest levels in the Central and Mississippi flyways (34).<br><br>About 75 percent of **all breeding birds** monitored had stable or increasing trends (44). The North had the greatest proportion of species with increasing trends (44). | Mammalian **small game** species are not monitored well, making it difficult to assess population status of these species (25-26).<br><br>**Small game** species associated with forest habitats, including squirrel and forest grouse, show mixed trends (25).<br><br>Recent increases in **goose** populations may be exceeding the capacity of the breeding habitat (34).<br><br>**Mourning dove** population trends have shown slight declines nationally (35).<br><br>Population trends in **furbearers** are not monitored well (39) and harvest appears to be driven by pelt price rather than population status (39). Thus it is difficult to evaluate the status of furbearer populations.<br><br>There is preliminary evidence that the CRP may have increased **grassland nesting bird** populations in the Rocky Mountain region (44-45).<br><br>There are numerous accounts of local declines in **amphibian** populations, but in the absence of a national inventory it is difficult to assess the status of amphibian populations nationally (51-54). | Populations of **northern pintail** are a notable exception to duck population status. Populations have continued to decline and are now more than 70 percent below populations observed in the mid-1950s (31).<br><br>Harvests of **Canada geese** in the Pacific flyway have remained stable while harvests in the Atlantic flyways have declined in the last decade (34). The only two populations of Canada geese with declining populations are the dusky and Atlantic populations, which occur in the Pacific and Atlantic flyways, respectively (33).<br><br>In the Western management unit, **mourning dove** populations have been declining at an average annual rate of 2.4 percent since 1966 (35), and harvests have declined by more than 30 percent since the early 1970s (38).<br><br>**Woodcock** population indices appear to be declining at a more accelerated rate (37), and harvests have declined by more than 60 percent in the Eastern and Central management units (38).<br><br>**Grassland nesting birds** as a group showed a high proportion (44 percent) of species with declining trends (44). More than 70 percent of grassland nesting birds in the North and South have had long-term declines (45).<br><br>Case studies of **bird communities** in the eastern United States indicate that landscapes under intensive land use (i.e., urban and agriculture) tended to have lower bird community integrity (48-49), a higher proportion of rare species (49), a higher proportion of |

Table 19 (Cont'd.)

| Resource attribute | Resource condition: Favorable | Uncertain | Unfavorable |
|---|---|---|---|
| Population/harvest (Cont'd.) | | | exotic individuals (50), and more variable species composition (i.e., high local extinction rates) (50-51) than landscape that supported greater amounts of natural vegetation.<br><br>The rate at which species have been listed as **threatened and endangered** has increased greatly in the last 10 years (54-55. Endangered species are concentrated along coastal areas, the arid Southwest, and the southern Appalachians (56-58). |
| Users of wildlife resources | Participation in **big game hunting** has increased greatly since the mid-1950s (61). The greatest proportional gains in big game hunting from 1991 to 1996 occurred in the Pacific Coast (+24 percent), followed by the South (+11 percent), and the Rocky Mountain region (+10 percent) (61).<br><br>The number **of migratory game bird hunters** has declined substantially since the mid-1970s but shows evidence of rebounding during the 1990s (61). | **Hunting** in general is declining in popularity (60-61). This trend appears to be caused by the dramatic decline in the number of **small game hunters** (61). This trend may reflect a decline in wildlife populations or a shift in recreational preferences.<br><br>The number of **trappers** has declined substantially since the last Assessment and may be affected further by international agreements concerning the use of leghold traps (63). It is unclear whether this trend reflects a decline in furbearer resources or a change in human preferences.<br><br>Participation in **wildlife viewing** has shown evidence of declining participation during the 1990s (63). The decline was greatest in the Pacific Coast and North (63). However, of all wildlife-dependent recreation activities, participation in nonconsumptive wildlife recreation is expected to increase at the greatest rate over the next 50 years (63). | |

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## Appendix A: Scientific names of species mentioned in the text.

| Common name | Scientific name | Common name | Scientific name |
|---|---|---|---|
| **Plants** | | Goose, Ross' | *Chen rossii* |
| Alder, Red | *Alnus rubra* | Goose, Snow | *Chen caerulescens* |
| Ash | *Fraxinus* spp. | Grouse, Blue | *Dendropagus obscurus* |
| Aspen | *Populus* spp. | Grouse, Ruffed | *Bonsa umbellus* |
| Beech | *Fagus grandifola* | Grouse, Sage | *Centrocercus urophasianus* |
| Birch | *Betula* spp. | Grouse, Sharp-tailed | *Pedioecetes phasianellus* |
| Cottonwood | *Populus* spp. | Grouse, Spruce | *Canachites canadensis* |
| Cypress | *Taxodium distichum* | Mallard | *Anas platyrhynchos* |
| Elm | *Ulnus* spp. | Partridge, Gray | *Perdix perdix* |
| Fir | *Abies* spp. | Pheasant, Ring-necked | *Phasianus colchicus* |
| Fir, Douglas | *Pseudotsuga menziesii* | Pintail, Northern | *Anas acuta* |
| Gum | *Nyssa* spp., *Liquidambar styraciflua* | Prairie-chicken, Greater | *Tympanuchus cupido* |
| Hemlock | *Tsuga* spp. | Prairie-chicken, Lesser | *Tympanuchus pallidicinctus* |
| Hickory | *Carya* spp. | Quail, California | *Callipepla californica* |
| Larch | *Larix* spp. | Quail, Gambel's | *Callipepla gambelii* |
| Maple | *Acer* spp. | Quail, Montezuma | *Cyrtonyx montezumae* |
| Oak | *Quercus* spp. | Quail, Mountain | *Oreortyx pictus* |
| Oak, Live | *Quercus virginiana* | Quail, Scaled | *Callipepla squamata* |
| Pine | *Pinus* spp. | Redhead | *Aythya americana* |
| Pine, Jack | *Pinus banksiana* | Scaup | *Aythya* spp. |
| Pine, Loblolly | *Pinus taeda* | Shoveler, Northern | *Anas clypeata* |
| Pine, Lodgepole | *Pinus contorta* | Swan, Tundra | *Cygnus columbianus* |
| Pine, Longleaf | *Pinus palustris* | Teal, Blue-winged | *Anas discors* |
| Pine, Ponderosa | *Pinus ponderosa* | Teal, Green-winged | *Anas crecca* |
| Pine, Red | *Pinus resinosa* | Turkey, Wild | *Meleagris gallopavo* |
| Pine, Shortleaf | *Pinus echinata* | Wigeon, American | *Mareca americana* |
| Pine, Slash | *Pinus elliottii* | Woodcock, American | *Scolopax minor* |
| Pine, Western White | *Pinus monticola* | **Mammals** | |
| Pine, White | *Pinus strobus* | Bear, Black | *Ursus americana* |
| Redcedar | *Juniperus virginianus* | Beaver | *Castor canadensis* |
| Redwood | *Sequoia sempervirens* | Bobcat | *Lynx rufus* |
| Sagebrush, Big Basin | *Artemisia tridentata tritendata* | Cottontail | *Sylvilagus* spp. |
| Spruce | *Picea* spp. | Coyote | *Canis latrans* |
| Spruce, Sitka | *Picea sitchensis* | Deer | *Odocoileus* spp. |
| **Birds** | | Deer, Mule | *Odocoileus hemionus* |
| Bobwhite, Northern | *Colinus virginianus* | Deer, White-tailed | *Odocoileus virginianus* |
| Brant | *Branta bernicla* | Elk | *Cervus elaphus* |
| Canvasback | *Aythya valisineria* | Fox, Gray | *Urocyon cinereoargenteus* |
| Chukar | *Alectoris chukar* | Fox, Red | *Vulpes vulpes* |
| Dove, Mourning | *Zenaida macroura* | Hare | *Lepus* spp. |
| Gadwall | *Anas strepera* | Lynx | *Lynx canadensis* |
| Goose, Canada | *Branta canadensis* | Mink | *Mustela vison* |
| Goose, Cackling Canada | *Branta canadensis minima* | Muskrat | *Ondatra zibethicus* |
| Goose, Dusky Canada | *Branta canadensis occidentalis* | Pronghorn | *Antelocapra americana* |
| Goose, Greater White-fronted | *Anser albifrons* | Raccoon | *Procyon lotor* |
| | | Squirrel | *Sciurus* spp. |

**Appendix B:** Land use and land cover changes on nonfederal lands from 1982 to 1992 for the nation and each RPA Assessment region. Entries in each table reflect the acres (x 1,000) that were converted from the 1982 land type specified by the rows, to the 1992 land type specified by the columns. For example, in the first row of Table B.1, there were 365.8 million acres of cultivated cropland in 1982. More than 80% of those acres that were cultivated cropland in 1982 remained cultivated in 1992 (295.7 million acres). Of the 70.1 million acres of cultivated cropland that were converted to another land use in 1992, 30.2 million acres were converted to CRP, 18.5 million acres were converted to noncultivated cropland, 11.2 million acres were converted to pasture, 3.1 million acres were converted to urban, 2.5 million acres were converted to forest, and 1.7 million acres were converted to rangeland.

Table B.1—Land use changes from 1982 to 1992 in the United States (USDA Natural Resources Conservation Service 1994).

1992 (acres X 1,000)

| 1982 (acres X 1,000) | Cult. cropland | Noncult. cropland | Pasture | Range | Forest | Misc. | Urban | Water | Federal | CRP | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Cult. | 295,704.3 | 18,503.1 | 11,247.3 | 1,677.4 | 2,500.8 | 1,781.2 | 3,139.7 | 363.4 | 751.2 | 30,180.7 | 365,849.1 |
| Noncult. | 14,536.1 | 32,163.0 | 3,518.8 | 413.5 | 632.7 | 487.9 | 996.8 | 62.0 | 256.6 | 1,635.8 | 54,703.2 |
| Pasture | 7,816.3 | 3,976.1 | 104,011.1 | 1,475.2 | 8,161.5 | 1,379.3 | 2,405.5 | 317.5 | 302.3 | 1,252.0 | 131,096.8 |
| Range | 4,484.8 | 1,194.1 | 2,416.1 | 391,513.9 | 1,523.7 | 1,095.4 | 2,107.9 | 348.1 | 3,301.4 | 755.3 | 408,740.7 |
| Forest | 1,135.4 | 311.8 | 2,838.4 | 1,136.1 | 379,138.3 | 1,160.9 | 5,558.2 | 462.7 | 1,968.7 | 134.8 | 393,845.3 |
| Misc. | 850.4 | 327.3 | 808.0 | 270.7 | 1,425.5 | 48,349.6 | 290.8 | 103.2 | 268.7 | 79.4 | 52,773.6 |
| Urban | 216.1 | 34.8 | 84.7 | 90.7 | 216.6 | 21.6 | 77,424.9 | 2.1 | 0.7 | 0.3 | 78,092.5 |
| Water | 112.8 | 40.8 | 99.2 | 260.7 | 180.3 | 53.2 | 5.5 | 47,203.9 | 12.4 | 1.1 | 47,969.9 |
| Federal | 323.2 | 219.7 | 191.2 | 1,965.2 | 657.5 | 195.5 | 16.4 | 0 | 401,037.2 | 0.6 | 404,606.5 |
| Total | 325,179.4 | 56,770.7 | 125,214.8 | 398,803.4 | 394,436.9 | 54,524.6 | 91,945.7 | 48,862.9 | 407,899.2 | 34,040.0 | $1.938 \times 10^{6}$ |
| %[a] | 16.8 | 2.9 | 6.5 | 20.6 | 20.4 | 2.8 | 4.8 | 2.5 | 21.0 | 1.8 | 100.1[d] |
| %Change[b] | -11.1 | +3.8 | -4.5 | -2.4 | +0.1 | +3.3 | +17.7 | +1.9 | +0.8 | na[c] | na |

[a]Percent of the total land base in 1992
[b]Percent change from 1982 to 1992
[c]Not applicable
[d]Does not sum to 100 because of rounding

Table B.2—Land use changes from 1982 to 1992 in the North region (USDA Natural Resources Conservation Service 1994).

1992 (acres X 1,000)

| 1982 (acres X 1,000) | Cult. cropland | Noncult. cropland | Pasture | Range | Forest | Misc. | Urban | Water | Federal | CRP | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Cult. | 109,062.9 | 9,026.6 | 2,919.3 | 0 | 355.3 | 747.0 | 1,287.5 | 90.1 | 87.1 | 6,506.4 | 130,082.2 |
| Noncult. | 7,572.4 | 12,956.1 | 1,641.6 | 0 | 390.2 | 306.3 | 377.1 | 18.3 | 21.7 | 875.7 | 24,159.4 |
| Pasture | 3,528.4 | 2,255.2 | 34,037.2 | 3.0 | 3,698.3 | 712.3 | 714.6 | 74.2 | 58.7 | 604.2 | 45,686.1 |
| Range | 8.1 | 7.4 | 10.1 | 123.4 | 18.3 | 1.2 | 0.6 | 0 | 0 | 0 | 169.1 |
| Forest | 328.6 | 168.2 | 664.6 | 0 | 142,788.8 | 511.4 | 1,834.0 | 113.3 | 355.1 | 46.9 | 146,810.9 |
| Misc. | 414.3 | 192.1 | 301.5 | 0 | 699.5 | 15,474.7 | 81.0 | 5.0 | 8.0 | 38.3 | 17,214.4 |
| Urban | 103.3 | 18.3 | 30.5 | 0 | 77.3 | 6.7 | 30,939.9 | 0.5 | 0.4 | 0.3 | 31,177.2 |
| Water | 35.3 | 11.2 | 22.5 | 0 | 51.8 | 15.7 | 0.5 | 14,435.1 | 0.4 | 0 | 14,572.5 |
| Federal | 28.5 | 11.2 | 12.8 | 0 | 79.5 | 2.2 | 0 | 0 | 15,290.3 | 0 | 15,424.5 |
| Total | 121,081.8 | 24,646.3 | 39,640.1 | 126.4 | 148,159.0 | 17,777.5 | 35,235.2 | 14,736.5 | 15,821.7 | 8,071.8 | 425,296.3 |
| %[a] | 28.5 | 5.8 | 9.3 | <0.1 | 34.8 | 4.2 | 8.2 | 3.4 | 3.7 | 1.9 | 99.8[d] |
| %Change[b] | -6.9 | +2.0 | -13.2 | -25.2 | +0.9 | +3.3 | +13.0 | +1.1 | +2.6 | na[c] | na |

[a]Percent of the total land lease in 1992
[b]Percent change from 1982 to 1992
[c]Not applicable
[d]Does not sum to 100 because of rounding

Table B.3—Land use changes from 1982 to 1992 in the South region (USDA Natural Resources Conservation Service 1994).

1992 (acres X 1,000)

| 1982 (acres X 1,000) | Cult. cropland | Noncult. cropland | Pasture | Range | Forest | Misc. | Urban | Water | Federal | CRP | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Cult. | 76,370.9 | 2,589.0 | 6,427.8 | 763.0 | 2,127.0 | 578.9 | 1,257.4 | 197.4 | 275.6 | 8,100.1 | 98,687.1 |
| Noncult. | 1,030.1 | 5,260.1 | 875.7 | 47.5 | 219.8 | 59.7 | 297.2 | 26.1 | 46.4 | 97.6 | 7,960.2 |
| Pasture | 2,467.2 | 762.6 | 53,872.5 | 958.3 | 4,291.0 | 501.6 | 1,360.8 | 216.1 | 141.7 | 373.0 | 64,944.8 |
| Range | 1,041.2 | 153.8 | 1,520.4 | 110,074.3 | 758.5 | 296.7 | 941.7 | 204.3 | 202.1 | 218.1 | 115,411.1 |
| Forest | 769.1 | 124.6 | 2,100.4 | 140.3 | 168,902.5 | 534.2 | 3,189.7 | 309.1 | 766.8 | 84.4 | 176,921.1 |
| Misc. | 169.1 | 45.0 | 436.7 | 88.2 | 673.0 | 12,104.5 | 106.2 | 75.4 | 42.6 | 13.2 | 13,753.9 |
| Urban | 33.2 | 4.5 | 40.7 | 9.1 | 117.3 | 4.1 | 28,536.5 | 0 | 0.3 | 0 | 28,745.7 |
| Water | 30.0 | 5.4 | 49.1 | 50.1 | 107.7 | 17.4 | 3.1 | 20,148.5 | 5.5 | 1.1 | 20,417.9 |
| Federal | 19.0 | 3.6 | 32.8 | 4.3 | 137.6 | 0.6 | 0 | 0 | 24,492.8 | 0 | 24,690.7 |
| Total | 81,929.8 | 8,948.6 | 65,356.1 | 112,135.1 | 177,334.4 | 14,097.7 | 35,692.6 | 21,176.9 | 25,973.8 | 8,887.5 | 551,532.5 |
| %[a] | 14.9 | 1.6 | 11.8 | 20.3 | 32.1 | 2.6 | 6.5 | 3.9 | 4.7 | 1.6 | 100.0 |
| %Change[b] | -17.0 | +12.4 | +0.6 | -2.8 | +0.2 | +2.5 | +24.2 | +3.7 | +5.2 | na[c] | na |

[a]Percent of the total land base in 1992
[b]Percent change from 1982 to 1992
[c]Not applicable

Table B.4—Land use changes from 1982 to 1992 in the Rocky Mountain region (USDA Natural Resources Conservation Service 1994).

1992 (acres X 1,000)

| 1982 (acres X 1,000) | Noncult. cropland | Cropland | Pasture | Range | Forest | Misc. | Urban | Water | Federal | CRP | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Cult. | 96,800.7 | 5,612.9 | 1,609.9 | 576.5 | 1.8 | 376.9 | 418.6 | 68.1 | 347.2 | 13,915.0 | 119,727.6 |
| Noncult. | 5,080.4 | 9,647.0 | 850.5 | 309.1 | 12.9 | 82.1 | 176.4 | 12.8 | 156.5 | 633.3 | 16,961.0 |
| Pasture | 1,583.4 | 780.7 | 12,262.3 | 399.5 | 43.2 | 76.7 | 178.2 | 20.0 | 67.3 | 270.8 | 15,682.1 |
| Range | 3,212.4 | 869.4 | 776.6 | 249,600.8 | 395.7 | 622.2 | 732.4 | 131.4 | 2,335.1 | 510.8 | 259,186.8 |
| Forest | 26.4 | 14.2 | 16.8 | 588.0 | 27,507.9 | 42.9 | 171.0 | 21.0 | 446.3 | 3.3 | 28,837.8 |
| Misc. | 229.2 | 66.3 | 49.4 | 156.3 | 33.0 | 14,495.5 | 61.7 | 7.8 | 137.9 | 27.9 | 15,265.0 |
| Urban | 70.2 | 10.0 | 11.0 | 73.4 | 13.9 | 1.9 | 11,116.6 | 1.6 | 0 | 0 | 11,298.6 |
| Water | 40.9 | 13.0 | 9.0 | 64.1 | 6.0 | 17.6 | 0 | 8,880.3 | 0.3 | 0 | 9,031.2 |
| Federal | 254.9 | 184.9 | 132.0 | 1,858.8 | 267.0 | 124.9 | 16.4 | 0 | 270,619.0 | 0.6 | 273,458.5 |
| Total | 107,298.5 | 17,198.4 | 15,717.5 | 253,626.5 | 28,281.4 | 15,840.7 | 12,871.3 | 9,143.0 | 274,109.6 | 15,361.7 | 749,448.6 |
| %[a] | 14.3 | 2.3 | 2.1 | 33.8 | 3.8 | 2.1 | 1.7 | 1.2 | 36.6 | 2.0 | 99.9[d] |
| %Change[b] | -10.4 | +1.4 | +0.2 | -2.1 | -1.9 | +3.8 | +13.9 | +1.2 | +0.2 | na[c] | na |

[a]Percent of the total land lease in 1992
[b]Percent change from 1982 to 1992
[c]Not applicable
[d]Does not sum to 100 because of rounding

Table B.5—Land use changes from 1982 to 1992 in the Pacific Coast region (USDA Natural Resources Conservation Service 1994).

1992 (acres X 1,000)

| 1982 (acres X 1,000) | Cult. cropland | Noncult. cropland | Pasture | Range | Forest | Misc. | Urban | Water | Federal | CRP | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Cult. | 13,469.8 | 1,274.6 | 290.3 | 337.9 | 16.7 | 78.4 | 176.2 | 7.8 | 41.3 | 1,659.2 | 17,352.2 |
| Noncult. | 853.2 | 4,299.8 | 151.0 | 56.9 | 9.8 | 39.8 | 146.1 | 4.8 | 32.0 | 29.2 | 5,622.6 |
| Pasture | 237.3 | 177.6 | 3,839.1 | 114.4 | 129.0 | 88.7 | 151.9 | 7.2 | 34.6 | 4.0 | 4,783.8 |
| Range | 223.1 | 163.5 | 109.0 | 31,715.4 | 351.2 | 175.3 | 433.2 | 12.4 | 764.2 | 26.4 | 33,973.7 |
| Forest | 11.3 | 4.8 | 56.6 | 407.8 | 39,939.1 | 72.4 | 363.5 | 19.3 | 400.5 | 0.2 | 41,275.5 |
| Misc. | 37.8 | 23.9 | 20.4 | 26.2 | 20.0 | 6,274.9 | 41.9 | 15.0 | 80.2 | 0 | 6,540.3 |
| Urban | 9.4 | 2.0 | 2.5 | 8.2 | 8.1 | 8.9 | 6,831.9 | 0 | 0 | 0 | 6,871.0 |
| Water | 6.6 | 11.2 | 18.6 | 146.5 | 14.8 | 2.5 | 1.9 | 3,740.0 | 6.2 | 0 | 3,948.3 |
| Federal | 20.8 | 20.0 | 13.6 | 102.1 | 173.4 | 67.8 | 0 | 0 | 90,635.1 | 0 | 91,032.8 |
| Total | 14,869.3 | 5,977.4 | 4,501.1 | 32,915.4 | 40,662.1 | 6,808.7 | 8,146.6 | 3,806.5 | 91,994.1 | 1,719.0 | 211,400.2 |
| %[a] | 7.0 | 2.8 | 2.1 | 15.6 | 19.2 | 3.2 | 3.9 | 1.8 | 43.5 | 0.8 | 99.9[d] |
| %Change[b] | -14.3 | +6.3 | -5.9 | -3.1 | -1.5 | +4.1 | +18.6 | -3.6 | +1.1 | na[c] | na |

[a]Percent of the total land lease in 1992
[b]Percent change from 1982 to 1992
[c]Not applicable
[d]Does not sum to 100 because of rounding

RMRS

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Forest Service
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