Indicator 9. Population Levels of Representative Species from Diverse Habitats Monitored Across Their Range

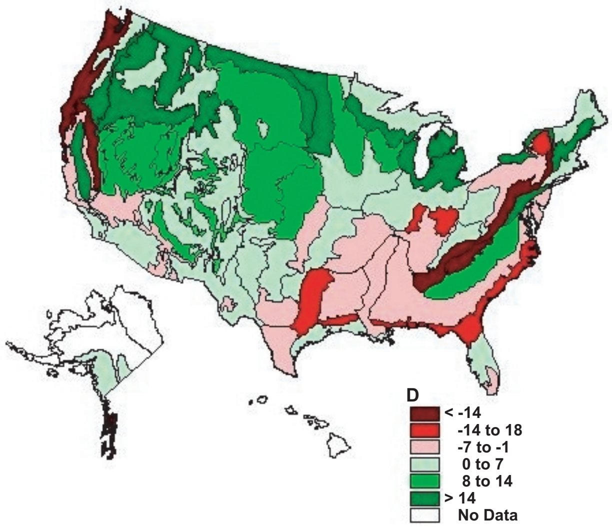


Figure 9-1. Difference (D) between the number of forest birds with significantly (P <0.1) increasing and decreasing population trends, by physiographic region, between 1966 and 2000, calculated from the Breeding Bird Survey (BBS) database.

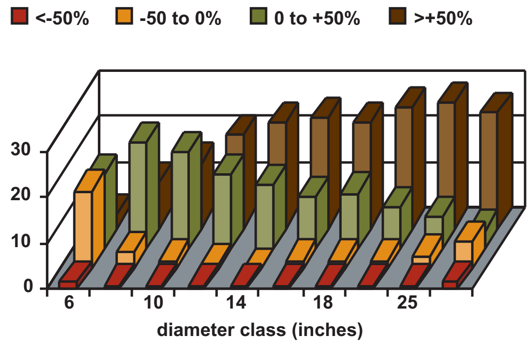


Figure 9-2. Number of tree species or groups of species in the Forest Inventory and Analysis database with decreasing and increasing stem numbers (a measure of tree population size), by diameter class, for trees >5 inches diameter breast height, between 1970 and 2002.

What Is the Indicator and Why Is It Important?

This indicator estimates population trends of selected species as a surrogate measure of genetic diversity. Decreases in genetic diversity as populations decline, particularly if associated with small populations, contribute to increased risk of extinction. This indicator also provides an important measure of general biodiversity, since changes in species abundance are a more sensitive measure of environmental stress than species richness alone.

What Does the Indicator Show?

Between 1966 and 2000, about 26 percent of bird species associated with forests increased and 27 percent decreased; for nearly half the species, no strong evidence existed for an increasing or decreasing trend. Physiographic regions with higher numbers of bird species with significantly decreasing trends compared to bird species with significantly increasing trends are clustered on the coastal regions and eastern third of the United States (figure 9-1). Most tree species or groups of species tracked by the Forest Inventory and Analysis program show increases of >50 percent in numbers of stems >12 inches in diameter between 1970 and 2002 (figure 9-2). State agency data indicate that populations of many big-game species increased in the last 25 years, but forest-dependent small-game species showed mixed trends.

Why Can't the Entire Indicator Be Reported at This Time?

Although it is not surprising that systematic inventories of obscure taxa (e.g., nonvascular plants, fungi, bacteria, nematodes, and arachnids) that would permit estimates of population trends over time are lacking, it is surprising that spatially and temporally extensive data for most other taxa are generally lacking as well. The paucity of population data for taxa other than bird species and a small subset of mostly big-game species points out the need to develop systematic strategies for monitoring population levels of other vertebrate, invertebrate, and plant taxa.

CRITERION 1: CONSERVATION OF BIOLOGICAL DIVERSITY

Indicator 9: Population Levels of Representative Species From Diverse Habitats Monitored Across Their Range

Carolyn Hull Sieg
Curtis H. Flather
Noah Barstatis

United States Department of Agriculture
Forest Service

ABSTRACT

Sieg, Carolyn Hull; Flather, Curtis H.; Barstatis, Noah. 2003. Criterion 1: Conservation of biological diversity. Indicator 9: Population levels of representative species from diverse habitats monitored across their range. In: Darr, D., compiler. Technical document supporting the 2003 national report on sustainable forests. Washington, DC: U.S. Department of Agriculture, Forest Service. Available: http://www.fs.fed.us/research/sustain/ [2003, August].

This indicator estimates population trends of selected species as a surrogate measure of genetic diversity. Decreases in genetic diversity as populations decline, particularly if associated with small populations, contribute to increased risk of extinction. This indicator also provides an important measure of general biodiversity, as changes in species abundances are a more sensitive measure of environmental stress than species richness alone. Between 1966 and 2000, about 26 percent of bird species associated with U.S. forests increased and 27 percent decreased; for nearly half the species there was no strong evidence for an increasing or decreasing trend. Physiographic regions where more forest birds declined than increased were clustered along the coast and eastern third of the U.S. The majority of tree species (or species groups) tracked by the Forest Inventory and Analysis program increased by >50 percent in numbers of stems >12 inches in diameter between 1970 and 2002. State agency data indicate that populations of many big game species increased in the last 25 years, but forest-dependent small game species showed mixed trends. The paucity of population data for taxa other than bird and tree species points out the need to develop systematic strategies for monitoring population levels of other animal and plant taxa.

Keywords: forest bird population trends, trends in tree stem counts, big game population trends, small game population trends, sustainability indicators, sustainable forest management.

Authors

Carolyn Hull Sieg is a research plant ecologist with the Rocky Mountain Research Station, USDA Forest Service, Flagstaff, AZ 86001.

Curtis H. Flather is a research wildlife biologist with the Rocky Mountain Research Station, USDA Forest Service, Fort Collins, CO 80526.

Noah Barstatis is biological technician with the Rocky Mountain Research Station, USDA Forest Service, Flagstaff, AZ 86001.

INTRODUCTION

Biological diversity has been defined as "... the variety of life and its processes" that encompasses "... the variety of living organisms, the genetic differences among them, and the communities and ecosystems in which they occur" (Keystone Center 1991:6). Over the last half-century, scientists and natural resource managers have learned much about how biodiversity contributes to human society, the economic significance of which can be considerable (Pimentel and others 1997). Most obviously, many of the goods that are harvested and traded in the human economy are a direct product of the biological diversity within ecosystems (Daily 1997). Biological diversity also provides indirect benefits to humans through its impact on important ecosystem functions (Risser 1995; Huston and others 1999; Naeem and others 1999), and less tangible, but equally important, benefits in the form of recreational opportunity, as well as spiritual and intellectual fulfillment (Postel and Carpenter 1997). Because intensive use of natural resources can stress ecosystems to a point where their ability to provide these benefits is compromised (Rapport and others 1985; Loreau and others 2001), it has been argued that the human enterprise may be jeopardizing the health and continued existence of some ecosystems (Vitousek and others 1997). This argument is the motivation behind a worldwide paradigm shift in natural resource management that is now focusing on long-term sustainability of ecosystems as the measure of responsible resource stewardship (Noble and Dirzo 1997). One of the fundamental goals emerging from the sustainable management paradigm is to use resources in ways that conserve biological diversity (that is, the variety of ecosystems, species, and genes) undiminished for future generations (Lubchenco and others 1991; Lélé and Norgaard 1996).

The nine indicators accepted by the Montréal Process countries for monitoring biological diversity trends consider ecosystem diversity (5 indicators), species diversity (2 indicators), and genetic diversity (2 indicators). This report focuses on one of the genetic diversity indicators – namely, population levels of representative species from diverse habitats monitored across their range. Our purpose is to provide the rationale for the use of population trends of forest dependent species[1] as an indicator of biological diversity, to review the data available on this indicator, and to present the findings from these data at national and sub-national scales. Finally, we will conclude with an evaluation of the data limitations and indicator adequacy, and propose a set of research topics directed at improving the use of population trends of representative species as an indicator of the status of biological diversity.

RATIONALE

Genetic diversity is a measure of the variability of genes among individuals in a species or population. Maintenance of genetic diversity is necessary to maintain

[1] A forest dependent species is any species that needs forest conditions for all or part of its requirements of food, shelter or reproduction (Report of the technical advisory committee to the working group on criteria and indicators for the conservation and sustainable management of temperate and boreal forests ["The Montréal Process"], Draft Version 3.0, September 25, 1996). We use the terms "forest dependent" and "forest associated" interchangeably throughout this report.

population fitness and is important in that it allows a species to evolve with changing environmental conditions and to decrease their susceptibility to pests and diseases (Schonewald-Cox and others 1983; Reed and Frankham 2003). Loss of genetic diversity is certainly a concern for species that are naturally rare, but it may be even more serious for relatively common species whose numbers have recently been severely reduced (Barrett and Kohn 1991). Genetic simplification has the potential to reduce population viability in a number of ways. In addition to reducing a species' ability to adapt to changing environmental conditions, a loss of genetic diversity can lead to higher rates of inbreeding or outbreeding and to the chance expression of deleterious genes (Wright 1977; Rieseberg 1991; Lande 1995). Inbreeding in butterfly species can lead to a reduction in some fitness components and result in higher extinction probabilities (Saccheri and others 1998). Outbreeding in some plant species can lead to a decline in fitness due to hydridization (Templeton 1986; Ellstrand and Elam 1993).

Unfortunately, data are not available to directly assess the status of genetic diversity of even well-studied animal species across broad geographic regions (Smith and Rhodes 1992), although there are a few exceptions (Millar and Libby 1991; Williams 2002). Therefore, members of the Montréal Process Working Group (2000) chose to use population levels of representative species from a diversity of habitats as a surrogate measure of genetic diversity. This indicator makes two important assumptions: first, that genetic diversity can be tracked by monitoring population levels; and second, that monitoring population levels of some subset of representative species will also provide an indication of the population response of other related species (Flather and Sieg 2000).

The motivation for the use of monitoring population levels as a surrogate for genetic diversity is simply that it is not feasible to monitor genetic diversity of the world's species. Ecological theory suggests that habitat loss and other factors can lead to population declines and degradation of genetic diversity (for example, Gilpin and Soulé 1986). The linkage between demographic decline and erosion of genetic diversity has been demonstrated for only a few species. Westemeir and others' (1998) study of prairie chickens *(Tympanuchus cupido pinnatus)* in the Midwest is one of the few that demonstrated the linkage of reduced genetic diversity associated with small isolated populations. Bellinger and other's (2003) study of the greater prairie chicken population in Wisconsin that has declined by nearly 50 percent since the 1950s documents reduced genetic variation in contemporary birds compared to those collected in 1951. Although quantitative data for other species that explore this relationship are sparse, it is expected that erosion of genetic diversity will be most relevant for small populations.

The use of representative species in Indicator 9 is based on the concept that a subset of species could be identified whose population levels would reflect the response of related species. Many forest dependent species rely on some particular forest structure, vegetation associations, or ecological processes; and these species are commonly associated with other species that are dependent on similar conditions (Montreal Process Working Group 2000). As it is not possible to monitor all species, the intent of this Indicator is to monitor population levels of one species across diverse habitats to represent the status of others associated with similar conditions. The concept of using representative or indicator species to reflect the status of a functional group of species is commonly recommended because inventory data are so sparse (Raven and Wilson 1992; Dale and others 2000). Unfortunately, the general applicability of using indicator species has not been verified (Flather and others 1997; van Jaarsveld and others 1998).

Aside from the intended purpose of Indicator 9 as a surrogate of genetic diversity, population trend is an important dimension of species diversity. Changes in species abundances are a more sensitive measure of environmental stress than species richness alone (Kempton 1979). Therefore, monitoring both species richness (Indicator 6 [Flather and others 2003a]) and species population trends provides a better assessment of general biodiversity trends (Flather and Sieg 2000). Identifying species whose population levels are declining is a potentially powerful early indicator of impending imperilment.

DATA SOURCES AND ANALYSIS APPROACH

Cost is the primary motivation for using population levels of representative species as a surrogate measure of genetic diversity. The techniques for measuring genetic variation are well established (Hedrick and Miller 1992), but it would be prohibitively expensive to periodically measure genetic variation in even a subset of species. Data on population trends are available for a much larger subset of species than data on genetic variability. Unfortunately, national level data that allow quantification of population trends of forest dependent species are restricted to only a few taxonomic groups and span only about 30 to 40 years. And given the available data, we did not attempt to identify any species or subsets of species as being representative of the other species. Our data selection should be viewed as an ad hoc compilation of available data that were consistent with the intent of this indicator, with emphasis on its use as a general biodiversity indicator.

The analyses discussed in this report stem primarily from three databases that provide national and regional estimates of species populations. First, the North American Breeding Bird Survey (BBS) was used to estimate population trends of native breeding bird species (Sauer and others 2001). The BBS is a geographically and temporally extensive survey that has been conducted since 1966 of more than 4,000 roadside routes that are randomly distributed within a degree block of latitude and longitude throughout the United States and southern Canada (Droege 1990). The sampling unit is a 24.5-mile route along which 50 three-minute point counts are conducted at 0.5-mile intervals. At each point-count stop, all birds seen or heard are recorded. Although the data are not without limitations (see Geographic Trends in Breeding Bird Populations below), the BBS is unique in that it provides a geographic depiction of population trends in avian species. DeGraaf and others (1991) was used to determine the set of bird species within the BBS that qualified as forest breeding.

We estimated the percentage of native forest-dependent bird species whose population numbers have not changed significantly ($P>0.1$), and those that have increased or decreased significantly ($P\leq0.1$) (see Sauer and others 1997), over three time periods: 1966-1979, 1980-2000, and 1966-2000. Species were counted as having significant population trends if the slope of the regression line differed from zero ($P\leq0.1$) (Geissler and Sauer 1990; Link and Sauer 1994). Including indicator variables for observers and omitting the first survey year for each observer on the route mitigated observer effects:

$$C_y = b_0^{y}\, b_1^{o_{1y}}\, b_2^{o_{2y}} \cdots b_n^{o_{ny}}\, \varepsilon_y$$

where
C_y = count in year y,
B_0 = slope term,
b_i = observer coefficient for observer i (i = 1, …, n), and
o_{iy} = 1 if observer i ran route in year y, 0 if not.

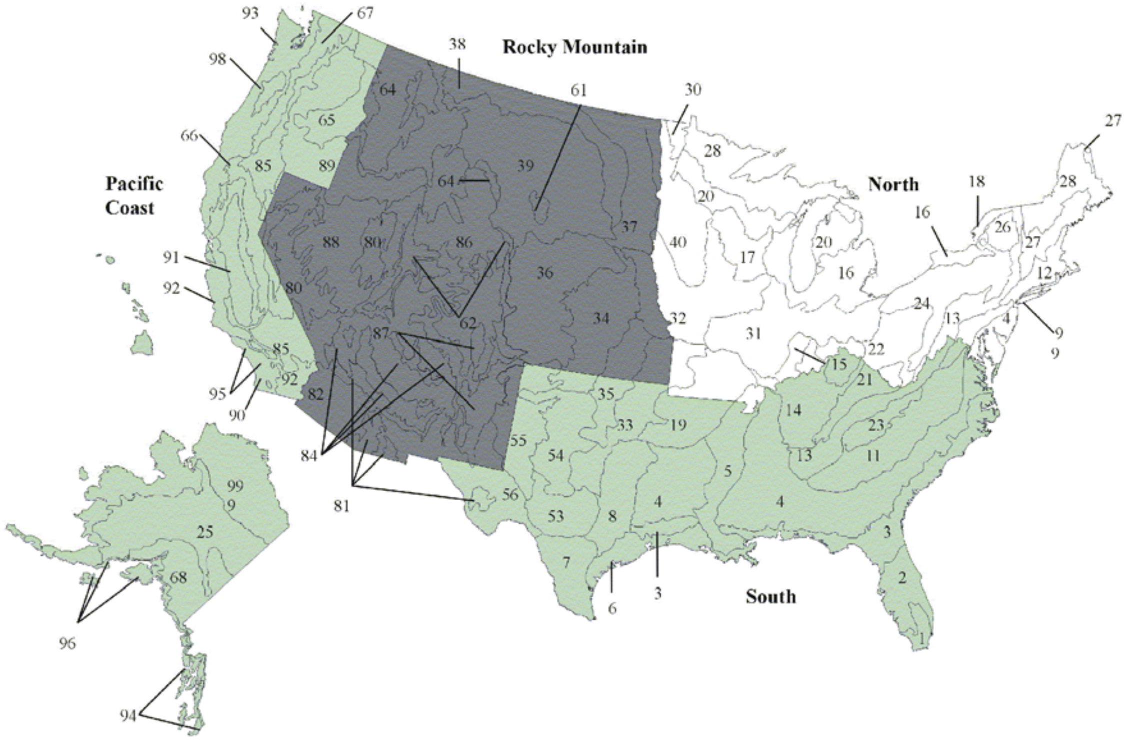
Population change was modeled on individual routes as a regression of the natural logarithm or counts plus 0.5 (to avoid domain errors) against year:

$$\ln(C_y + 0.5) = \ln(b_0)y + \ln(b_1)o_{1y} + \ldots + \ln(b_0)o_{ny}.$$

We present a national summary of these results. In addition, we calculated the difference (D) between the number of birds associated with forests with significantly ($P<0.1$) increasing and decreasing population trends. Values of $D>0$ indicate that among forest breeding birds with significant changes in population, most had increasing trends; values of $D<0$ indicate that a majority of birds with significant population trends were declining. We depict these results by physiographic regions defined by the BBS (Sauer and others 2001) and by the U.S. Forest Service regional planning boundaries to support the national resource assessment mandate (USDA Forest Service 2001). Physiographic regions and Forest Service regional planning boundaries are defined in figure 1.

Second, we used data from the USDA Forest Service Forest Inventory and Analyses (FIA) program to estimate changes in numbers of stems of tree species or groups of species between 1970 and 2002. The FIA (http://fia.fs.fed.us) currently has detailed data for trees >5 inches diameter, for areas classified as timberland (that is, land capable of producing 20 cubic feet of wood per acre per year, and which is available for successive harvests of timber products) (W.B. Smith and others 2001). In the eastern United States, timberland accounts for nearly 94 percent of the total forested land; in the western United States, excluding Alaska and Hawaii, timberland accounts for only about 40 percent of the forested land (W.B. Smith and others 2001). Forest types monitored by FIA, as defined by the Society of American Foresters (Eyre 1980), are associations or groups of tree species that are commonly found in forested communities ranging from single species to complex mixtures. Data were available for 27 tree species or species groups in the eastern United States (Appendix A) and 18 species or species groups in the western United States (Appendix B). To calculate number of stems for each species or species group, we divided the total volume of each diameter class by the regional average volume per tree in that diameter class in 2002. Data are summarized as the number of tree species or groups of species with decreasing and increasing numbers of stems, by diameter class, between 1970 and 2002. We present a national summary as well as summaries by Forest Service planning regions.

Figure 1. Physiographic regions defined by the Breeding Bird Survey (Sauer and others 2001) and U.S. Forest Service planning regions (USDA Forest Service 2001).



Physiographic regions:

1 Subtropical
2 Floridian
3 Coastal Flatwoods
4 Upper Coastal Plain
5 Mississippi Alluvial Plain
6 Coastal Prairies
7 South Texas Brushlands
8 East Texas Prairies
9 Glaciated Coastal Plain
10 Northern Piedmont
11 Southern Piedmont
12 Southern New England
13 Ridge and Valley
14 Highland Rim
15 Lexington Plain
16 Great Lakes Plain
17 Driftless Area
18 St. Lawrence River Plain
19 Ozark-Ouachita Plateau
20 Great Lakes Transition
21 Cumberland Plateau
22 Ohio Hills
23 Blue Ridge Mountains
24 Allegheny Plateau
25 Open Boreal Forest
26 Adirondack Mountains
27 Northern New England
28 N. Spruce-Hardwoods
29 Closed Boreal Forest
30 Aspen Parklands
31 Till Plains
32 Dissected Till Plains
33 Osage Plain-Cross Timbers
34 High Plains Border
35 Rolling Red Prairies
36 High Plains
37 Drift Prairie
38 Glaciated Missouri Plateau
39 Great Plains Roughlands
40 Black Prairie
53 Edward's Plateau
54 Rolling Red Plains
55 Staked Plains
56 Chihuahuan Desert
61 Black Hills
62 Southern Rockies
63 Fraser Plateau
64 Central Rockies
65 Dissected Rockies
66 Sierra Nevada
67 Cascade Mountains
68 Northern Rockies
80 Great Basin Deserts
81 Mexican Highlands
82 Sonoran Desert
83 Mojave Desert
84 Pinyon-Juniper Woodlands
85 Pitt-Klamath Plateau
86 Wyoming Basin
87 Intermountain Grasslands
88 Basin and Range
89 Columbia Plateau
90 S. California Grasslands
91 Central Valley
92 California Foothills
93 S. Pacific Rainforests
94 N. Pacific Rainforests
95 Los Angeles Ranges
96 S. Alaska Coast
98 Willamette Lowlands
99 Tundra

Our third source of information was the wildlife population trend database to support the Renewable Resources Planning Act National Assessment of wildlife resources (Flather and others 1999). State wildlife agencies were contacted to provide population estimates of commonly harvested wildlife species. We selected species associated with forested habitats including elk, deer (both mule deer and white-tailed deer), black bear, wild turkey, squirrel, and forest grouse (ruffed grouse, spruce grouse, and blue grouse) (scientific names in table 1). We calculated the percent change in estimated populations of these species or species groups between 1975 and 1993. These results are summarized nationally and by the Forest Service planning regions. In addition, we used data on American woodcock *(Scolopax minor)* abundance monitored through call-count surveys that provide an annual index of population size (Bruggink 1997). We calculated the percent change in woodcock populations between 1968 and 1996, and we summarize these results across the range of the woodcock (eastern half of the United States). Finally, we compared these results with population trends between 1966 and 2000 from the BBS database for species monitored on BBS transects (wild turkey, ruffed grouse, blue grouse and American woodcock).

Table 1. Estimated percent change in populations of selected forest-dependent wildlife species between 1975 and 1993, by Forest Service planning regions (USDA Forest Service 2001). Number of states reporting information is shown in parentheses. Region boundaries are defined in figure 1. Data are available upon request from Curt Flather (U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, CO 80526-1891, cflather@fs.fed.us).

	Forest Service planning region			
Species	Pacific Coast	Rocky Mountain	North	South
	Percent change (number of states reporting)			
Elk *(Cervus elaphus)*	12 (3)	109 (9)	512 (1)	(0)
Wild turkey *(Meleagris gallopavo)*	833 (3)	158 (10)	292 (15)	146 (13)
Deer (mule deer, *Odocoileus hemionus* and white-tailed deer, *O. virginianus)*	-8 (4)	29 (11)	87 (19)	198 (13)
Black bear *(Ursus americanus)*	54 (4)	50 (8)	97 (10)	184 (7)
Squirrels (*Sciurus* spp. plus red squirrel *(Tamiasciurus hudsonicus)*	(0)	-73 (1)	11 (3)	-12 (2)
Forest grouse (ruffed grouse *(Bonasa umbellus),* spruce grouse *(Falcipennis canadensis)*, and blue grouse *(Dendragapus obscurus)*	(0)	-51 (2)	-41 (6)	-57 (1)

RESULTS: INDICATOR INTERPRETATION

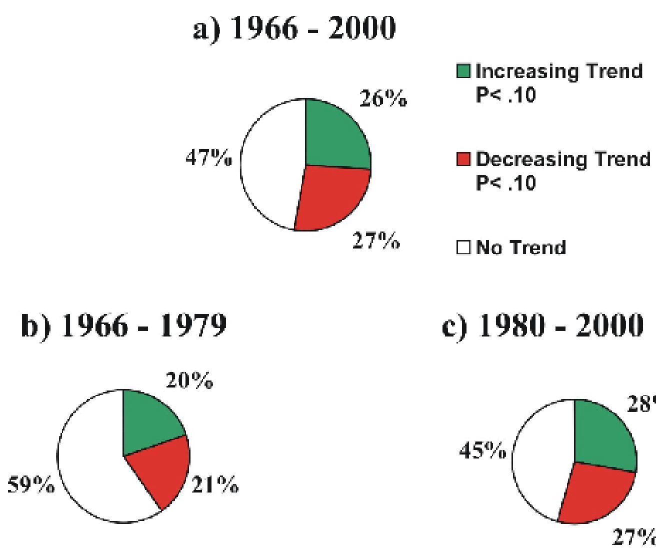
National Scale

Breeding Bird Population Trends

The BBS is one of the few databases available to examine spatial and temporal changes in population trends of native forest-dependent species in the United States. For the time period 1966 to 2000, 26 percent of the forest-dependent bird species increased, 27 percent of these species decreased, and population levels of 47 percent of the bird species did not change significantly (figure 2a). The percentage of species with decreasing population trends increased slightly from 21 percent between 1966 and 1979 (figure 2b) to 27 percent between 1980 and 2000 (figure 2c). The percentage of bird species whose population levels increased also increased from 20 percent between 1966 and 1979 to nearly 28 percent between 1980 and 2000. The biggest change between the two time periods was in the percentage of bird species with non-significant population trends. This percentage decreased from approximately 59 percent in the 1966 to 1979 time period to approximately 45 percent in the 1980 to 2000 time period.

The interpretation of population trends through time is seemingly simple; but without an understanding of the causes of population changes, this information will be difficult to interpret with respect to sustainability (Caughley 1994). Certainly a number of factors can contribute to population declines (or increases) of forest birds. For starters, not all native birds that nest in the forests of the United States winter here, and changes along migration corridors and in wintering ranges can affect population trends observed in the United States (for example, Askins and others 1990). An understanding of the processes that contribute to changes in population trends is critical for interpreting and applying these data (T.B. Smith and others 2001).

Figure 2. Percentage of breeding birds whose population trends have increased or decreased significantly ($P < 0.10$) (see Sauer and others 1997) over three time periods: a) 1966-2000, b) 1966-1979 and c) 1980-2000. Analyses are based on Breeding Bird Survey (BBS) data (Sauer and others 2001).

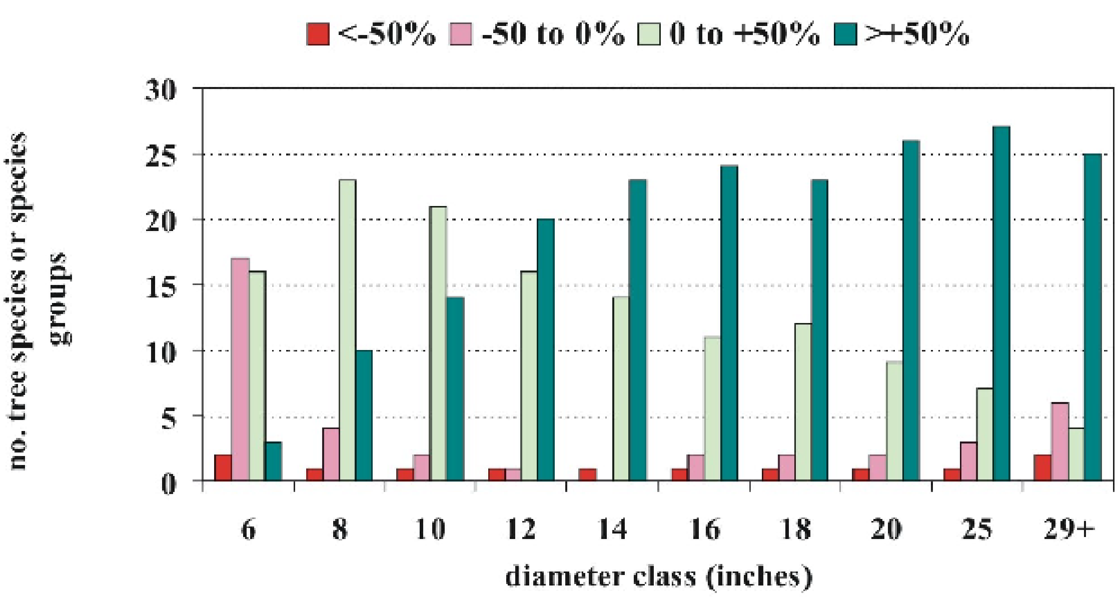


Numbers of Tree Stems Trends

The majority of tree species or groups of species tracked by the Forest Inventory and Analysis program increased by >50 percent in numbers of stems >12 inches in diameter between 1970 to 2002 (figure 3). For the 8- and 10-inch diameter classes, the majority of tree species or groups of species also increased, but to a lesser extent than the larger diameter classes. The number of tree species or groups of species that increased in the smallest diameter class monitored by the FIA was nearly equal to the number that decreased.

The number of stems in the various diameter classes can change as a result of a number of factors, including timber harvesting (or a lack thereof), other disturbances such as insect and disease outbreaks, fires, and drought, plus reclassification of timberland to reserve lands such as wilderness (Wright and Bailey 1982; Flather and others 1999; W.B. Smith and others 2001). Given the different needs and responses of individual species, it is not possible to describe a general pattern of number of stems by diameter classes that would fit every species. Therefore, there is a potential that the only meaningful standards for assessing whether tree populations are moving toward or away from sustainable management are frequency distribution patterns by diameter classes for each individual species or species groups. Aggregated frequency distributions that include tree species that respond differently to management and disturbances may not be appropriate.

Figure 3. Number of tree species or groups of species in the Forest Inventory and Analysis database with decreasing and increasing stem numbers (a measure of tree population size), by diameter class mid-points, for trees >5 inches diameter breast height, between 1970 and 2002.



Wildlife Population Trends

National population trends calculated from information provided by state wildlife agencies indicated that deer, elk, black bear, and wild turkey all increased between 1975 and 1993. Populations of wild turkeys expanded by 185 percent, and increases for deer, elk, and black bear were all between 70 and 90 percent. In contrast, using state data, nationwide squirrel populations were estimated to decline about 4 percent and forest grouse species declined 43 percent. American woodcock population estimates based on call count data were estimated to decline 24 percent between 1975 and 1993. BBS trend data indicated that wild turkey populations increased by an average of 12 percent annually ($P<0.001$) between 1966 and 2000 but the trend for American woodcocks was non-significant. Data for the two species of forest grouse monitored on BBS transects indicated that blue grouse populations declined ($P<0.001$) by about 4 percent annually between 1966 and 2000 and ruffed grouse population trends were not significant ($P>0.10$).

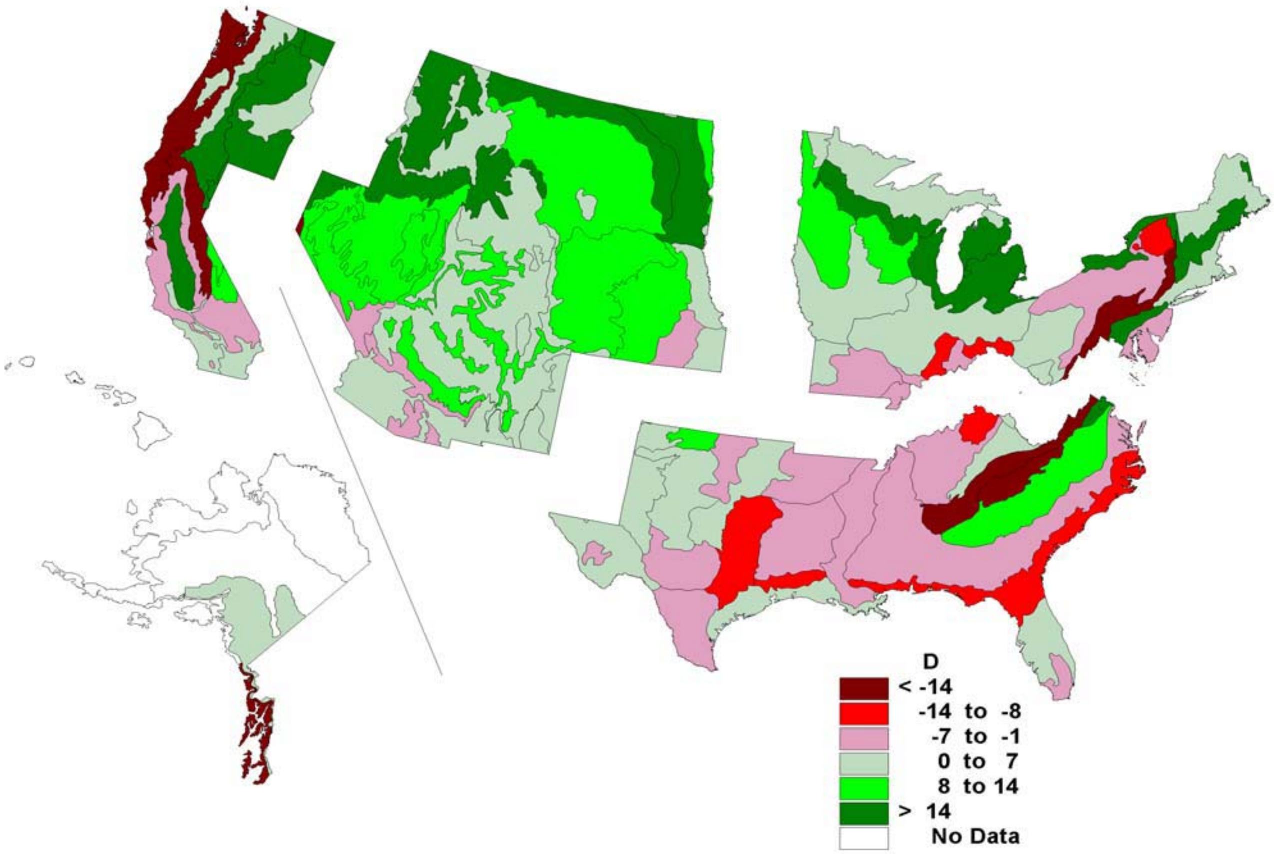
Interpretation of data on wildlife population trends calculated from information provided by state game agencies is confounded to some degree by the fact that data are not available for all species in all states and procedures for estimating populations are not standardized among states. However, other sources substantiate population trends suggested by these data for some species. According to Peek (1995), elk numbers are higher now than any time in the last century, and Vaughan and Pelton (1995) noted that black bear populations increased in 27 of 40 states reporting trends. Increases in wild turkey populations suggested by state data are substantiated by BBS analyses. Although woodcock population trends were not substantiated by BBS analyses, declines have continued in singing ground survey indices, base-year adjusted indices of daily and seasonal hunting success, and woodcock harvest (Kelley 2002). For other small game species, however, we were unable to substantiate population trends from independent sources.

Regional Scale

Geographic Trends in Breeding Bird Populations

Physiographic region analyses of the BBS data allowed us to depict the difference between the number of native avian forest species increasing and the number decreasing (figure 4). For the nation as a whole, the number of bird species that were increasing exceeded the number decreasing in the majority of physiographic regions, or about 70 percent of the total land area. In the majority of the physiographic regions in the Rocky Mountains, or 95 percent of the land area, the number of stable or increasing forest bird species exceeded the number of declining species. Physiographic regions where the number of decreasing bird species exceeded the number increasing were concentrated in the South assessment region (65 percent of the land area), followed by Pacific Coast (40 percent of the land area), North (30 percent of the land area), and Rocky Mountain region (5 percent of the land area). Physiographic regions where the number of decreasing species was greater than those increasing by >14 species were concentrated in the Pacific Coast (25 percent of the land area), with lesser amounts in the South (6 percent), North (3 percent), and Rocky Mountain region (1 percent).

Figure 4. Difference (D) between the number of forest birds with significantly (P≤ 0.1) increasing and decreasing population trends, by physiographic region, between 1966 and 2000, calculated from the Breeding Bird Survey (BBS) database (Sauer and others 2001). Also shown are the four Forest Service planning regions (USDA Forest Service 2001).

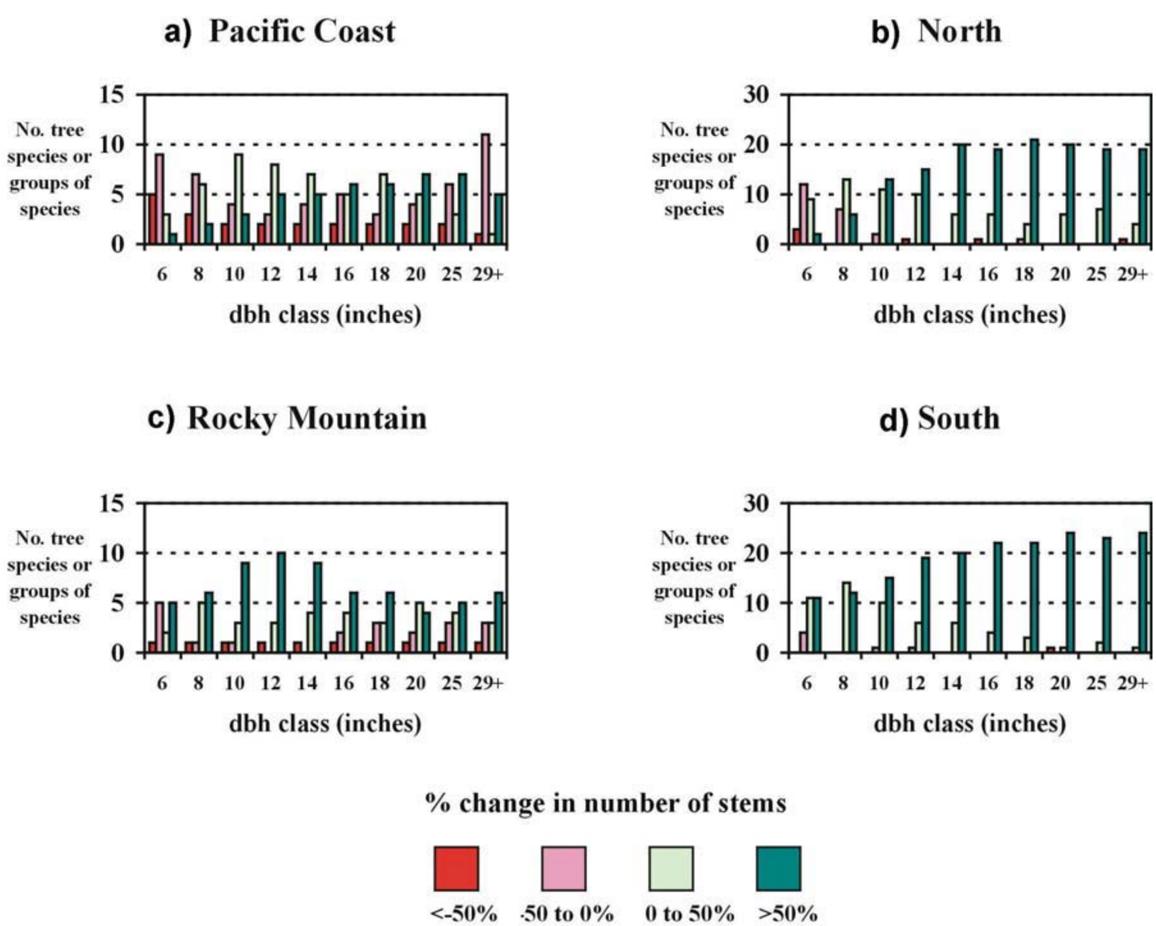


Although the BBS is a monitoring program that is unique in its geographic and temporal scope, a number of problems associated with its design and implementation warrant remark. First, not all bird species are monitored equally well by the BBS. Nocturnal or crepuscular species, cryptic species, some colonial nesting species, and species with restricted geographic ranges are not monitored well by the BBS (O'Connor and others 2000). The BBS database does not estimate and adjust population trends for these differences in detectability (Thompson 2002). Second, because the BBS is conducted along secondary roads, bird trends in relative abundance may not be representative of the regional bird pool because of biases associated with roadside habitats that may affect species detections (Robbins and others 1989; Bart and others 1995; Keller and Scallan 1999). Third, changes in observers over time can introduce spurious trends since observers can vary in their ability to detect birds; however, analytic methods have been developed to control for this nuisance variable (see Sauer and others 1994). Finally, survey routes can be closed (in total or in part) and relocated when urban encroachment results in traffic densities that limit aural detection of species. This relocation of routes could affect how representative BBS-based estimates of population trend are to land use and land cover changes that are occurring regionally.

Geographic Trends in Numbers of Tree Stems

Changes in the numbers of tree stems by diameter classes since 1970 for species and groups of species monitored by the FIA varied among the four Forest Service planning regions (figure 5). Trends in the North and South regions were similar to national trends, with the majority of tree species or groups of species increasing by >50 percent in numbers of stems in diameter classes 12 inches and greater. Many of the species with increasing stems since 1970 in most diameter classes in the eastern regions are late seral species, such as beech, that are increasing in response to fire suppression and natural succession; a number of early seral species, such as jackpine, are declining in most diameter classes (W.B. Smith and others 2001). Recent spruce budworm *(Choristoneura fumiferana)* outbreaks, and to a lesser degree, spruce beetle *(Dendroctonus rufipennis)* infestations, have contributed to declines in spruce/fir types in most diameter classes in the North region (Forest Health Protection 2003).

Figure 5. Number of tree species or groups of species in the Forest Inventory and Analysis database with decreasing and increasing stem numbers, by diameter class mid-points, for trees >5 inches diameter breast height, between 1970 and 2002, for the four Forest Service planning regions: a) Pacific Coast, b) North, c) Rocky Mountain, and d) South (USDA Forest Service 2001).



In Rocky Mountain states, the greatest number of tree species or groups of species increased by >50 percent in diameter classes between 10 and 18 inches; a fewer number increased by this magnitude in diameter classes >18 inches. In the Pacific Coast region, most species or groups of species increased by up to 50 percent in diameter classes between 10 and 14 inches, >50 percent in 16-, 20-, and 25-inch diameter classes, but decreased up to 50 percent in diameter classes >29 inches and <10 inches. High losses of Sitka spruce in all diameter classes in the Pacific region can be attributed in part to large areas being moved into reserved land (W.B. Smith and others 2001) as well as continuing spruce beetle infestations in Alaska and in some areas in Washington and Oregon (Forest Health Protection 2003).

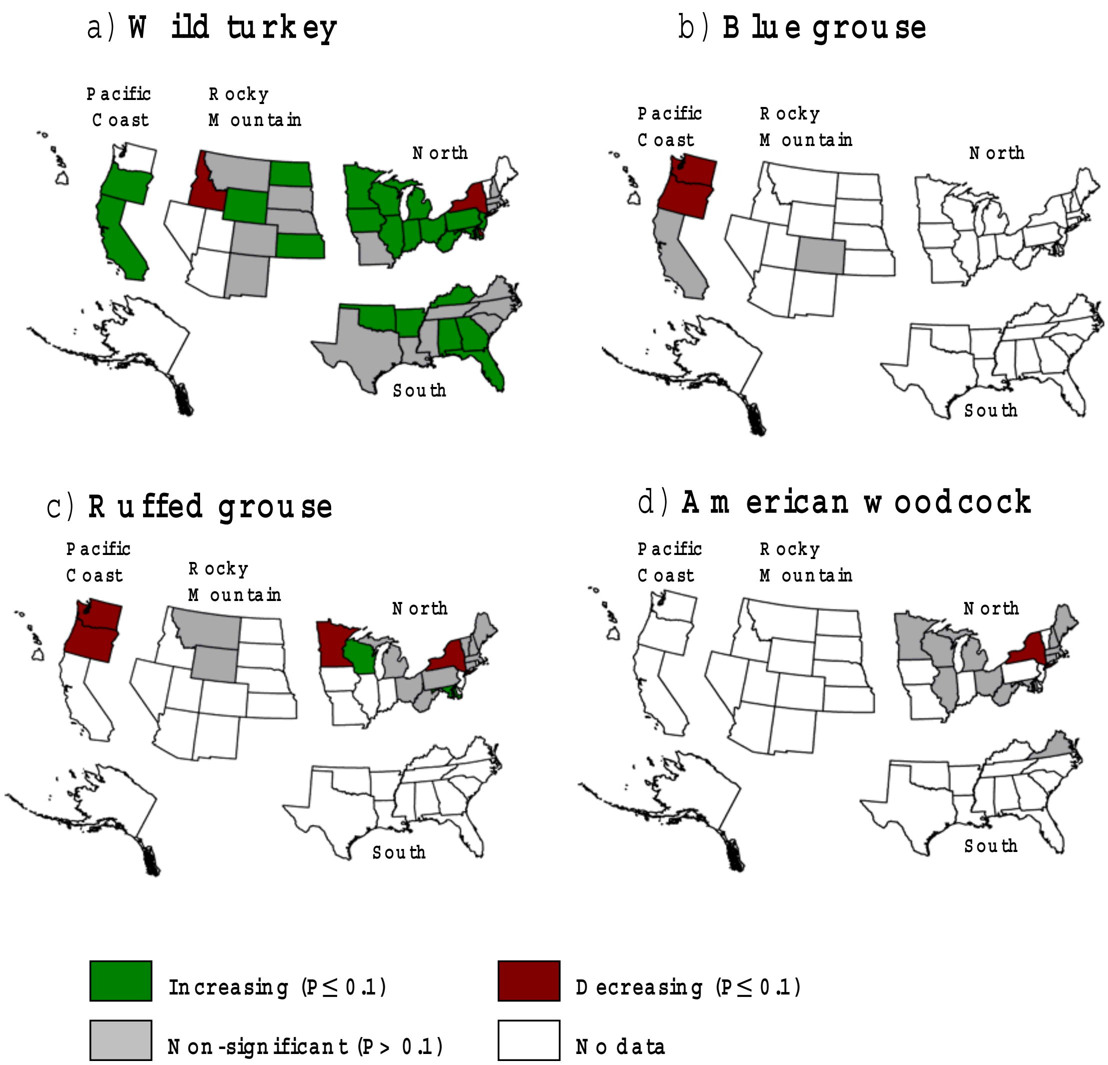
Although not well tested, commercially important tree species may prove to be valuable indices for monitoring their associated plant communities. Aside from the fact that trees are considered fiscally and politically important, they also play key ecological roles in their communities and we have more ecological and genetic information about them than nearly all other species in their communities (Millar and Libby 1991). The FIA data provides a pretty complete accounting of the status of forested lands in the eastern United States, and recent sampling that includes reserved land as well will increase the coverage in western states (W.B. Smith and others 2001). Unfortunately, at this point we lack a standard for comparing the current distribution of stems against some target distribution reflective of sustainable forestry. In the absence of an agreed-upon target stem distribution for each tree species or species group it will be difficult, if not impossible, to determine whether forest ecosystems are on a trajectory toward or away from a sustainable condition.

Geographic Trends in Populations of Selected Wildlife Species

Although more variable when considered region by region, the trends in estimated population levels of selected wildlife species reflect overall national trends (table 1). The highly variable regional results reflect, in part, low numbers of states reporting data for some regions. For example, the 512 percent increase in elk populations noted for the North region reflects information from only one state in that region. A greater number of states in the Pacific Coast and Rocky Mountain regions reported information; trends in elk populations in both these regions increased between 1975 and 1993. Most states also reported deer population estimates, and only the Pacific Coast estimates indicate declining populations between 1975 and 1993; in the other regions deer population trends are estimated to increase from 29 to 198 percent. Data on population estimates of black bear are given for most states, as well, and indicate increases ranging from 50 percent in the Rocky Mountain region to 184 percent in the South. For wild turkey, most states within each region provided data, but the estimated percent change varied from an increase of 146 percent in the South to an 833 percent increase in the Pacific Coast region (table 1). These results are substantiated to a large degree by BBS analyses. Wild turkey populations increased in 22 states between 1966 and 2000 and decreased in only two states (figure 6a).

Fewer states report data for small game species such as squirrels and forest grouse; none of the states in the Pacific Coast region provide data for these species. Changes in squirrel populations in the remaining regions are based on two or three states in each region and suggest declines of 73 percent in the Rocky Mountain region and 12 percent in the South but an increase of 11 percent in the North region. Changes in populations of forest grouse suggest declines in all regions; however, the Rocky Mountain estimate is based on data from only one state and the South estimate is based on data from two states. American woodcocks occur only in eastern portions of the United States. Call-count data suggest populations declined by 29 percent in central states and 37 percent in eastern states between 1975 and 1993. BBS analyses of forest grouse and American woodcock populations indicated variable trends from state to state (figure 6b, c, and d).

Figure 6. State-level population trends between 1966 and 2000 for a) wild turkey, b) blue grouse, c) ruffed grouse, and d) American woodcock, calculated from the Breeding Bird Survey database (Sauer and others 2001). Also shown are the four Forest Service planning regions (USDA Forest Service 2001).



INDICATOR EVALUATION

Indicator Adequacy

The use of Indicator 9 to track genetic diversity rests on the assumption that genetic diversity can be tracked by monitoring population levels of a species and that monitoring population levels of representative species will allow us to understand population responses of related species. Unfortunately, empirical support for either of these assumptions is lacking (Flather and Sieg 2000). Estimates of genetic erosion have been simulated for varying population sizes (for example, Lacy 1987), and genetic variation in at least one plant species has been correlated with population growth rate (Menges and Dolan 1998), but there are few estimates of the rate at which genetic diversity is lost or gained as a function of changing population levels (but see Westemeier and others 1998). In regard to the second assumption, the concept of using representative or indicator species to reflect the status of a functional group of species is commonly recommended because inventory data are so sparse (Raven and Wilson 1992; Dale and others 2000). Unfortunately, the general applicability of using indicator species has not been verified (Flather and others 1997; van Jaarsveld and others 1998).

Aside from the role of this indicator to track genetic diversity, data on species population trends might provide insights on other biodiversity trends. Unfortunately, the interpretation of the number of species with declining population trends, especially relative to the question of sustainable management, is not straightforward. This number can increase as the number of species on a trajectory toward imperilment increases. But an increasing number of species with declining population trends may not always be indicative of unsustainable management. Declines in some over-abundant species (such as white-tailed deer in some areas) might be indicative of a trajectory toward more sustainable management. Similarly, population declines of species associated with late successional forest types might indicate a more sustainable management strategy, such as re-introduction of fire in fire-prone ecosystems.

Data Limitations

Although it is not surprising that we lack systematic inventories of obscure taxa (for example, non-vascular plants, fungi, bacteria, nematodes, arachnids) that would permit estimates of population trends over time, it is surprising that we generally lack spatially and temporally extensive data for most other taxa as well (Flather and Sieg 2000). In addition to the databases we used in this paper, others such as the Global Population Dynamics Database (GPDD) (Inchausti and Halley 2001; NERC Centre for Population Biology, Imperial College 1999) have the potential to add to our knowledge of national population trends for a subset of taxa. Opportunistic databases such as the GPDD are limited in their usefulness in formal inference procedures, as these data represent a convenience sample as opposed to a probabilistic based sample (Anderson 2001).

Recommendations for Improvement and Research Needs

Our evaluation of this indicator and the shortcomings discussed above revealed five areas of research needs: systematic monitoring strategies for taxa other than breeding birds; a test of the linkage between species population trends and genetic diversity; testing the validity of the "representative" (indicator) species concept; the role of thresholds; and standards to be applied in evaluating population trends of representative species relative to sustainable management.

The paucity of population data for taxa other than avian species and a small subset of mammals points out the need to develop systematic strategies for monitoring population levels of other vertebrate, invertebrate, and plant taxa. In the absence of these data, it is impossible to evaluate the relationship between population levels and genetic diversity, let alone the population viability of these groups (Flather and Sieg 2000). Although recent efforts have extended the BBS-type survey design to amphibians (see the North American Amphibian Monitoring Program, http://www.mp2-pwrc.usgs.gov/naamp/), there is a critical need to develop surveys that can quantify population trends for a much broader set of taxa (National Research Council 1992). Inventory programs such as the Monitoring Avian Productivity and Survivorship (MAPS) constant-effort bird banding stations (DeSante and others 1995) should also be expanded geographically and to other taxa as well. MAPS provides an index of annual reproductive success of birds and has expanded from 424 stations in 1996 to 467 in 1998 (DeSante and O'Grady 2000). Tracking population vital rates like annual reproductive success provides potentially valuable early warnings of population changes and may be the most sensitive indicator of population change that could be monitored.

Considering the sessile nature of plants and the ease of sampling them, data on plant populations besides trees are surprisingly sparse in the literature. Two aspects of the population biology of plants may account for the limited census data: soil seedbanks and clonal growth in some species (Barrett and Kohn 1991). Because of these aspects of plant populations, species richness (Indicator 6) and species that occupy a small portion of their former range (Indicator 8) may be more achievable indicators of plant biodiversity than population levels of representative species (Indicator 9). The development of several websites dedicated to an accounting of the distribution of North American flora should assist in efforts to quantify both current distributions and future assessments of the changes in the distribution of plant species. Both Flora of North America (http://hua.huh.harvard.edu/FNA/) and the plants database (http://plants.usda.gov/) provide distributional data on North American plant taxa.

The second research area relates to testing the validity of using measures such as population levels of representative species as surrogates for genetic diversity. There is evidence that, at least for some species like red pine *(Pinus resinosa)* that have almost no genetic variation, measures of population trends are poor indicators of genetic variability (Fowler and Morris 1977). And, on a related note, we have limited information on whether depression of genetic diversity does, necessarily, lead to decreases in fitness components. There are examples where population size and genetic diversity have been positively correlated; however, neither population size nor genetic diversity was related to fitness components such as seed germination rates of some rare plants (Lammi and others 1999).

The concept of identifying representative species to indicate the response of other species is intuitively appealing and widely suggested (Dale and others 2000). Under this concept, population trend data in well-studied taxa are assumed to reflect the pattern among the other taxa. Unfortunately, there has been little empirical evidence supporting the general applicability of the indicator taxon concept (Prendergast and others 1993; Flather and others 1997; van Jaarsveld and others 1998; Harcourt 2000; Ricketts and others 2002). The lack of general support for the indicator taxon concept notwithstanding, there may be circumstances and scales under which some taxa serve as useful surrogates for the diversity patterns of other taxa (Reid 1998; Allen and others 2001; Moritz and others 2001). The variability in conclusion regarding the utility of indicator taxa argues for a systematic research agenda that will permit the specification of those ecological conditions and scales for which indicator taxa is a tenable conservation concept.

There is a need to better understand at what population levels losses of genetic diversity induce deleterious effects, and whether population thresholds might exist. Increases in risk extinction are expected to occur with decreasing levels of genetic variability associated with small populations (Wright 1977) but may occur at higher population levels than expected (Lande 1995). Further, for some species, extinction rates may increase incrementally with population decline and then suddenly increase exponentially (Frankham 1995). Whether thresholds exist, and at what population levels increased extinction risks are likely to occur, are challenging but important questions relative to understanding the role of genetic variability in population persistence.

Finally, there is a need to identify standards for assessing how species population trends might reflect whether or not management is on a sustainable trajectory. Species population trends are a more sensitive measure of environmental stress than species richness alone, but the relationship between population trend and sustainability is not always straightforward and constant. It is likely that monitoring the population trends of a subset of species from diverse taxonomic groups, such as those identified as being at risk (Indicator 7 [Flather and others 2003b]) or those associated with habitats that are at risk (Indicator 2 [Smith 2003]), would provide a potential early warning indication of biodiversity trends.

REFERENCES

Allen, C. R.; Pearlstine, L. G.; Wojcik, D. P.; Kitchens, W. M. 2001. The spatial distribution of diversity between disparate taxa: spatial correspondence between mammals and ants across South Florida, USA. Landscape Ecology 16: 453-464

Anderson, D. R. 2001. The need to get the basics right in wildlife field studies. Wildlife Society Bulletin. 29: 1294-1297.

Askins, R. A.; Lynch, J. F.; Greenberg, R. 1990. Population declines in migratory birds in eastern North America. In: Power, D. M., ed. Current Ornithology, Vol. 7. New York, NY: Plenum Press: 1-57.

Barrett, S. C. H.; Kohn, J. R. 1991. Genetic and evolutionary consequences of small population size in plants: implications for conservation. In: Falk, D. A.; Holsinger, K. E., eds. Genetics and conservation of rare plants. New York, NY: Oxford University Press: 3-30

Bart, J.; Hofschen, J. M.; Peterjohn, B. J. 1995. Reliability of the Breeding Bird Survey: effects of restricting surveys to roads. Auk. 112: 758-71.

Bellinger, R.R., J.A. Johnson, J. Toepfer, and P. Dunn. 2003. Loss of genetic variation in greater prairie chickens following a population bottleneck in Wisconsin, U.S.A. Conservation Biology. 17:717-724.

Bruggink, J. G. 1997. American woodcock harvest and breeding population status, 1997. Washington, D.C.: USDI, Fish and Wildlife Service. 14 p.

Caughley, G. 1994. Directions in conservation biology. Journal of Animal Ecology. 63:215-244.

Coulombe, M.J. 1995. Sustaining the world's forests: the Santiago Agreement. Journal of Forestry. 93: 18-21.

Daily, G. C. 1997. Nature's services: societal dependence on natural ecosystems. Washington, DC: Island Press 392 p.

Dale, V. H.; Brown, S.; Haeuber, R. A.; Hobbs, N. T.; Huntly, N.; Naiman, R. J.; Riebsame, W. E.; Turner, M. G.; Valone, T. J. 2000. Ecological principles and guidelines for managing the use of land. Ecological Applications. 10: 639-670.

DeGraaf, R. M.; Scott, V. E.; Hamre, R. H.; Ernst, L.; Anderson, S. H. 1991. Forest and rangeland birds of the United States: natural history and habitat use. Agricultural Handbook 688. Washington, DC: U. S. Department of Agriculture, Forest Service. 625 p.

DeSante, D. F.; Burton, K. M.; Saracco, J. F.; Walker, B. L. 1995. Productivity indices and survival rate estimates from MAPS, a continent-wide programme of constant effort mist-netting in North America. Journal of Applied Statistics. 22: 935-947.

DeSante, D. F.; O'Grady, D. R. 2000. The monitoring avian productivity and survivorship (MAPS) program 1997 and 1998 report. Bird populations 5: 49-101.

Droege, S. 1990. The North American breeding bird survey. In: Sauer, J. R.; Droege, S., eds., Survey designs and statistical methods for the estimation of avian populations trends. Washington, DC: U.S. Fish and Wildlife Service Biological Report 90(1): 1-4.

Ellstrand, N. C.; Elam, D. R. 1993. Population genetic consequences of small population size: implications for plant conservation. Annual Review of Ecological Systems. 24: 217-242.

Eyre, F. H., ed. 1980. Forest cover types of the United States and Canada. Bethesda, MD: Society of American Foresters. 148 p. (1 map).

Flather, C. H.; Wilson, K. R.; Dean, D. J.; McComb, W. C. 1997. Identifying gaps in conservation networks: of indicators and uncertainty in geographic-based analyses. Ecological Applications. 7: 531-542.

Flather, C. H.; Brady, S. J.; Knowles, M. S. 1999. Wildlife resource trends in the United States: a technical document supporting the 2000 USDA Forest Service RPA Assessment. Gen. Tech. Rep. RMRS-GTR-33. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 79 p.

Flather, C. H.; Sieg, C. H. 2000. Applicability of Montreal Process Criterion 1 – conservation of biological diversity – to rangeland sustainability. International Journal of Sustainable Development and World Ecology. 7: 81-96.

Flather, C. H.; Ricketts, T. H.; Sieg, C. H.; Knowles, M. S.; Fay, J. P.; McNees, J. 2003a. Criterion 1: Conservation of biological diversity. Indicator 6: The number of forest-dependent species. In: Darr, D., compiler. Technical document supporting the 2003 national report on sustainable forests. Washington, DC: U.S. Department of Agriculture, Forest Service. Available: http://www.fs.fed.us/research/sustain/ [2003, August].

Flather, C. H.; Ricketts, T. H.; Sieg, C. H.; Knowles, M. S.; Fay, J. P.; McNees, J. 2003b. Criterion 1: Conservation of biological diversity. Indicator 7: The status (threatened, rare, vulnerable, endangered, or extinct) of forest-dependent species at risk of not maintaining viable breeding populations, as determined by legislation or scientific assessment. In: Darr, D., compiler. Technical document supporting the 2003 national report on sustainable forests. Washington, DC: U.S. Department of Agriculture, Forest Service. Available: http://www.fs.fed.us/research/sustain/ [2003, August].

FIA statistics, 2002 RPA online data, references, and a map of U.S. forest distributions (http://fia.fs.fed.us).

Forest Health Protection. 2003. Forest insect and disease conditions in the United States. Washington, DC: U.S. Department of Agriculture, Forest Service. 112 p.

Fowler, D.P.; Morris, R.W. 1977. Genetic diversity in red pine: evidence for low genetic heterogeneity. Canadian Journal of Forest Research. 7: 343-347.

Frankham, R. 1995. Inbreeding and extinction: a threshold effect. Conservation Biology. 9: 792-799.

Geissler, P. H.; Sauer, J. R. 1990. Topics in route-regression analysis. In Sauer, J. R.; Droege, S., eds. Survey Designs and Statistical methods of Estimation of Avian Population Trends. Washington, DC: U.S. Fish and Wildlife Service Biological Report 90(1): 54-57.

Gilpin, M. E.; Soulé, M. E. 1986. Minimum viable populations: processes of species extinction. In Soulé, M. E., ed. Conservation biology – the science of scarcity and diversity. Sunderland, MA: Sinauer Associates, Inc.: 19-34.

Harcourt, A. H. 2000. Coincidence and mismatch of biodiversity hotspots: a global survey for the order, primates. Biological Conservation. 93: 163-175.

Hedrick, P. W.; Miller, P. S. 1992. Conservation genetics: techniques and fundamentals. Ecological Applications. 2: 30-46.

Huston, M.; McVicker, G.; Nielsen, J. 1999. A functional approach to ecosystem management: implications for species diversity. In: Szaro, R. C.; Johnson, N. C.; Sexton, W. T.; Malk, A. J., eds. Ecological stewardship: a common reference for ecosystem management, Vol. II. Oxford, UK: Elsevier Science: 45-85.

Inchausti, P.; Halley, J. 2001. Investigating long-term ecological variability using the Global Population Dynamics Database. Science. 293: 655-657.

Keller, C. M. E.; Scallan, J. T. 1999. Potential roadside biases due to habitat changes along Breeding Bird Survey routes. Condor. 101: 50-57.

Kelley, J.R., Jr. 2002. American woodcock population status, 2002. Laurel, MD: USDI Fish and Wildlife Service. 16 p.

Kempton, R.A. 1979. Structure of species abundance and measurement of diversity. Biometrics. 35: 307-322.

Keystone Center. 1991. Biological diversity on federal lands: report of a Keystone Policy Dialogue. Keystone, CO: The Keystone Center. 96 p.

Lacy, R.C. 1987. Loss of genetic diversity from managed populations: interacting effects of drift, mutation, immigration, selection and population subdivision. Conservation Biology. 1: 143-145.

Lammi, A.; Siikamaki, P.; Mustajarvi, K. 1999. Genetic diversity, population size, and fitness in central and peripheral populations of a rare plant *Lychnis viscaria.* Conservation Biology. 13: 1069-1078.

Lande, R. 1995. Mutation and conservation. Conservation Biology. 9: 782-791.

Lélé, S.; Norgaard, R. B. 1996. Sustainability and the scientist's burden. Conservation Biology. 10: 354-365.

Link, W.A.; Sauer, J. R. 1994. Estimating equations estimates of trend. Bird Populations. 2: 21-32.

Loreau, M.; Naeem, S.; Inchausti, P.; Bengtsson, J.; Grime, J. P.; Hector, A.; Hooper, D. U.; Huston, M. A.; Raffaelli, D.; Schmid, B.; Tilman, D.; Wardle, D. A. 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. Science. 294: 804-808.

Lubchenco, J.; Olson, A. M.; Brubaker, L. B. [and others]. 1991. The sustainable biosphere initiative: an ecological research agenda. Ecology. 72: 318-325.

Menges, E.S.; Dolan, R. W. 1998. Demographic viability of populations of *Silene regia* in Midwestern prairies: relationships with fire management, genetic variation, geographic location, population size and isolation. Journal of Ecology. 86: 63-78.

Millar, C.I.; Libby, W. J. 1991. Strategies for conserving clinal, ecotypic, and disjunct population diversity in widespread species. In: Falk, D. A.; Holsinger, K. E., eds. Genetics and conservation of rare plants. Center of Plant Conservation. New York, NY: Oxford University Press: 149-170.

Montréal Process Working Group. 2000. Criteria and indicators for the conservation and sustainable management of temperate and boreal forests. Technical Notes, Criteria 1-6, March 30, 2000. (http://www.mpci.org/meetings/tac-mexico/tn1-6_e.html).

Moritz, C.; Richardson, K. S.; Ferrier, S.; Monteith, G. B.; Stanisic, J.; Williams, S. E.; Whiffin, T. 2001. Biogeographical concordance and efficiency of taxon indicators for establishing conservation priority in a tropical rainforest biota. Proceedings of the Royal Society of London, Series B 268: 1875-1881.

Naeem, S.; Chapin, F. S.; Costanza, R.; Ehrlich, P. R.; Golley, F. B.; Hooper, D. U.; Lawton, J. H.; O'Neill, R. V. O'Neill; Mooney, H. A.; Sala, O. E.; Symstad, A. J.; Tilman, D. 1999. Biodiversity and ecosystem functioning: maintaining natural life support processes. Issues in Ecology No. 4. 11 p.

National Research Council. 1992. Conserving biodiversity: a research agenda for development agencies. Washington, D.C.: National Academy Press. 127 p.

NERC Centre for Population Biology, Imperial College. 1999. The Global Population Dynamics Database. http://www.sw.ic.ac.uk/cpb/cpb/gpdd.html.

Noble, I. R.; Dirzo, R. 1997. Forests as human-dominated ecosystems. Science. 277:522-525.

O'Connor, R. J.; Dunn, E.; Johnson, D. H.; Jones, S. L.; Petit, D.; Pollock, K.; Smith, C. R.; Trapp, J. L.; Welling, E. 2000. A programmatic review of the North American Breeding Bird Survey. U.S. Department of the Interior, U. S. Geologic Survey, Patuxent Wildlife Research Center. (http//www.mp2-pwrc.usgs.gov/bbs/bbsreview).

Peek, J. M. 1995. North American elk. In: LaRoe, E.T.; Farris, G.S.; Puckett, C.E.; Doran, P. D.; Mac, M. J., eds. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. Washington, DC: USDI, National Biological Service: 115-116.

Pimentel, D.; Wilson, C.; McCullum, C.; Huang, R.; Dwen, P.; Flack, J.; Tran, Q.; Saltman, T.; Cliff, B. 1997. Economic and environmental benefits of biodiversity. BioScience. 47: 747-757.

Postel, S.; Carpenter, S. 1997. Freshwater ecosystem services. In Daily, G. C., ed. Nature's services: societal dependence on natural ecosystems. Washington, DC: Island Press: 195-214.

Prendergast, J. R.; Quinn, R. M.; Lawton, J. H.; Eversham, B. C.; Gibbons, D. W. 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature. 365: 335-337.

Rapport, D. J.; Regier, H. A.; Hutchinson, T. C. 1985. Ecosystem behavior under stress. American Naturalist. 125: 617-640.

Raven, P. H; Wilson, E. O. 1992. A fifty-year plan for biodiversity surveys. Science. 258: 1099-1100.

Reed, D. H. and R. Frankham. 2003. Correlation between fitness and genetic diversity. Conservation Biology. 17: 230-237.

Reid, W. V. 1998. Biodiversity hotspots. Trends in Ecology and Evolution. 13: 275-280.

Ricketts, T. H.; Daily, G. C.; Ehrilch, P. R. 2002. Does butterfly diversity predict moth diversity? Testing a popular indicator taxon at local scales. Biological Conservation.103: 361-370.

Rieseberg, L. H. 1991. Hydridization in rare plants: insights from case studies in *Cercocarpus* and *Helianthus*. In Falk, D.A.; Holsinger, K.E., eds. Genetics and conservation of rare plants. Center of Plant Conservation. New York, NY: Oxford University Press: 171-181

Risser, P. G. 1995. Biodiversity and ecosystem function. Conservation Biology. 9: 742-746.

Robbins, C. S.; Droege, S; Sauer, J. R. 1989. Monitoring bird populations with Breeding Bird Survey and atlas data. Annals Zoological Fennici. 26: 297-304.

Schonewald-Cox, C.M.; Chamers, S.M; MacBryde, B.; Thomas, L., eds. 1983. Genetics and conservation. Menlo Park, CA: The Benjamin/Cummings Publishing Co. 722 p.

Saccheri, I., Mikko, K.; Kankare, M.; Vikman, P.; Fortelius, W.; Hanski, I. 1998. Inbreeding and extinction in a butterfly population. Nature. 392: 491-494.

Sauer, J. R.; Peterjohn, B. G.; Link, W. A. 1994. Observer differences in the North American Breeding Bird Survey. Auk 111: 50-62.

Sauer, J. R.; Hines, J. E.; Gough, G.; Thomas, I.; Peterjohn, B. G. 1997. The North American breeding bird survey and results analysis. Version 96.4. Laurel, MD: Patuxent Wildlife Research Center (http://www.nbs.gov/bbs/bbs.html).

Sauer, J. R.; Hines, J. E.; Fallon, J. 2001. The North American Breeding Bird Survey, Results and Analysis 1966 – 2000. Version 2001.2, Laurel, MD:USDI Geological Survey, Patuxent Wildlife Research Center (http://www.nbs.gov/bbs/bbs.html).

Sieg, C. H.; Flather, C. H.; McCanny, S. 2000. Recent biodiversity patterns in the Great Plains: implications for restoration and management. Great Plains Research. 9: 277-313.

Smith, W.B. 2003. Criterion 1: Conservation of biological diversity. Indicator 2: Extent of area by forest type and by age class or successional stage. In Darr, D., compiler. Technical document supporting the 2003 national report on sustainable forests. Washington, DC: U.S. Department of Agriculture, Forest Service. Available: http://www.fs.fed.us/research/sustain/ [2003, August].

Smith, W. B.; Vissage, J. S.; Darr, D. R. Darr; Sheffield, R. M. 2001. Forest statistics of the United States, 1997. Gen. Tech. Rep. NC-219. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Experiment Station. 191 p.

Smith, M. H.; Rhodes, O. E. 1992. Genetics and biodiversity in wildlife management. Transactions of the North American Wildlife and Natural Resources Conference. 57: 243-51.

Smith, T. B.; Kark, S.; Schneider, C. J.; Wayne, R. K.; Moritz, C. 2001. Biodiversity hotspots and beyond: the need for preserving environmental transitions. Trends in Ecology and Evolution. 16: 431.

Templeton, A.R. 1986. Coadaptation and outbreeding depression. In: Soulé, M. E., ed, Conservation biology – the science of scarcity and diversity Sunderland, MA: Sinauer Associates: 105-116.

Thompson, W.L. 2002. Towards reliable bird surveys: accounting for individuals present but not detected. Auk. 119:18-25.

U.S. Department of Agriculture, Forest Service. 2001. 2000 RPA assessment of forest and rangelands. Washington, DC: U.S. Department of Agriculture, Forest Service. 78 p.

van Jaarsveld, A. S.; Freitag, S.; Chown, S. L.; Muller, C.; Koch, S.; Hull, H.; Bellamy, C.; Krüger, M.; Endrödy-Younga, S.; Mansell, M. W.; Scholtz, C. H. 1998. Biodiversity assessment and conservation strategies. Science. 279: 2106-2108.

Vaughan, M. R.; Pelton, M. R. 1995. Black bears in North America. In: LaRoe, E. T.; Farris, G. S.; Pucket, C. E.; Doran, P. D.; Mac, M. J., eds. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. Washington, DC: USDI, National Biological Service: 100-103.

Vitousek, P. M.: Mooney, H. A.; Lubchenco, H. A.; Melillo, J.A. 1997. Human domination of earth's ecosystems. Science. 277: 494-499.

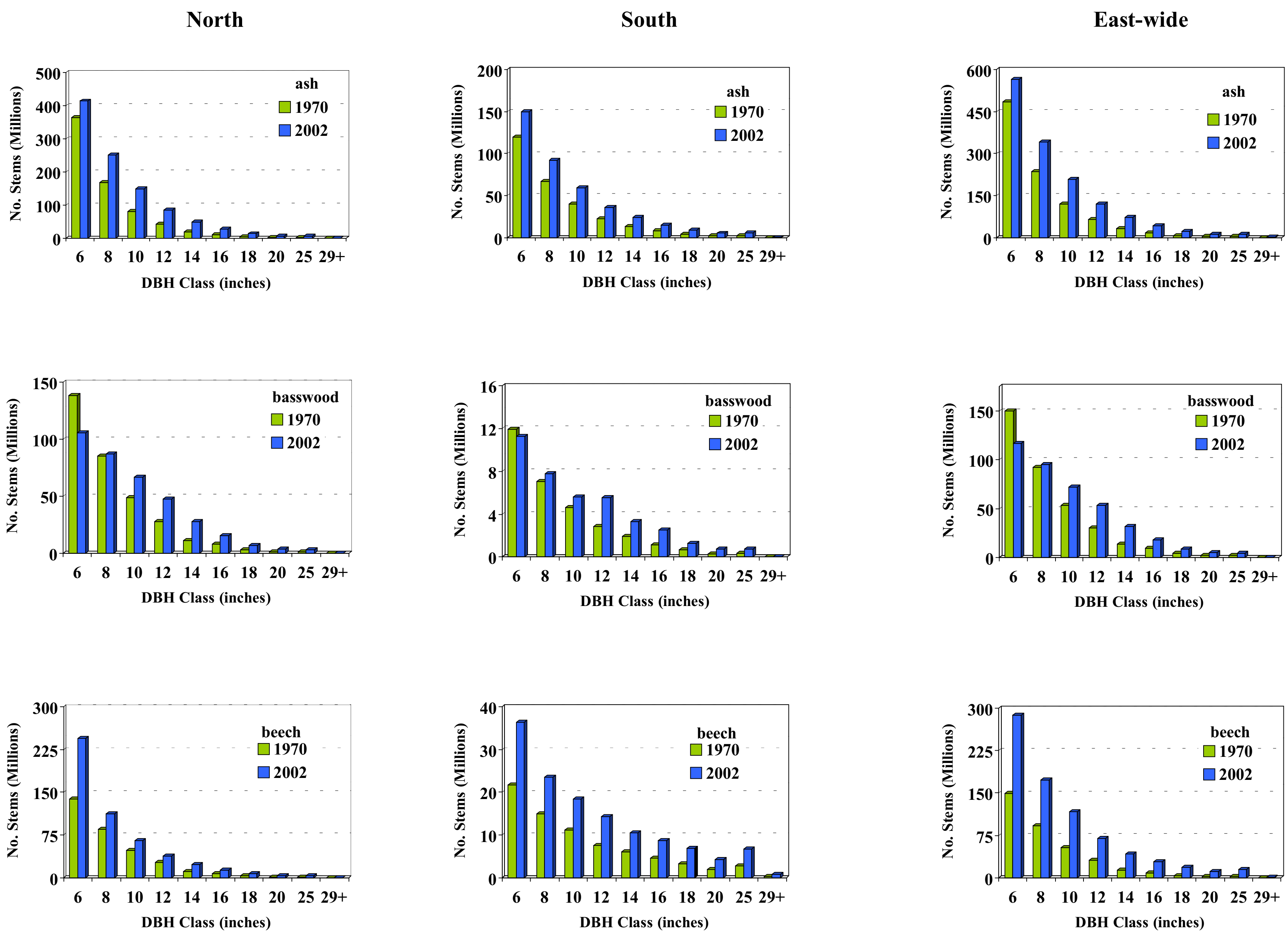
Westemeier, R. L.; Brawn, J. D.; Simpson, S. A.; Esker, T. L.; Jansen, R. W.; Walk, J. W.; Kershner, E. L.; Bouzat, J. L.; Paige, K. N. 1998. Tracking the long-term decline and recovery of an isolated population. Science. 282: 1695-1698.

Williams, B. L. 2002. Conservation genetics, extinction, and taxonomic status: a case history of the regal fritillary. Conservation Biology. 16: 148-157.

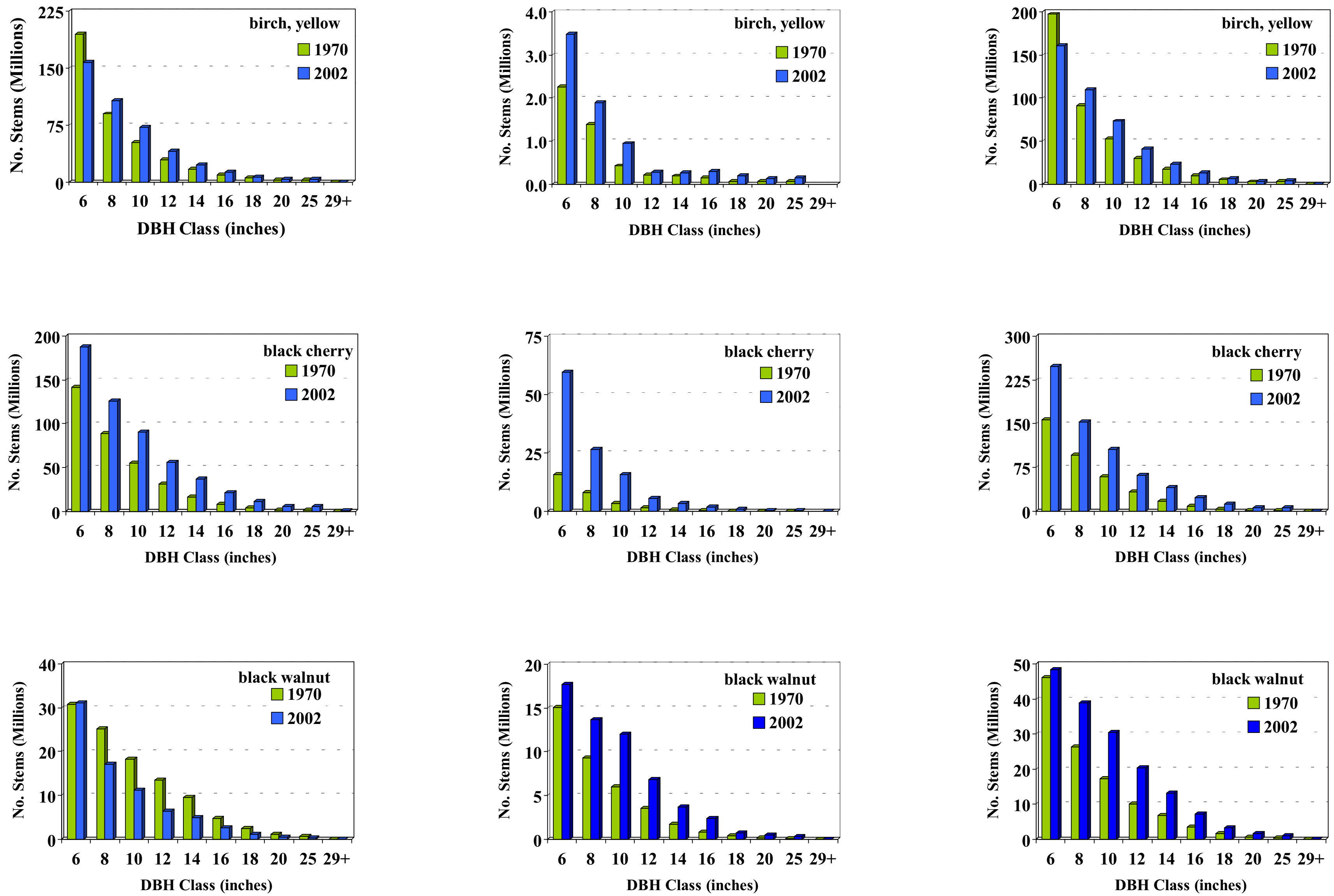
Wright, H.A.; A. Bailey. 1982. Fire Ecology – United States and Southern Canada. New York, NY: John Wiley & Sons. 501 p.

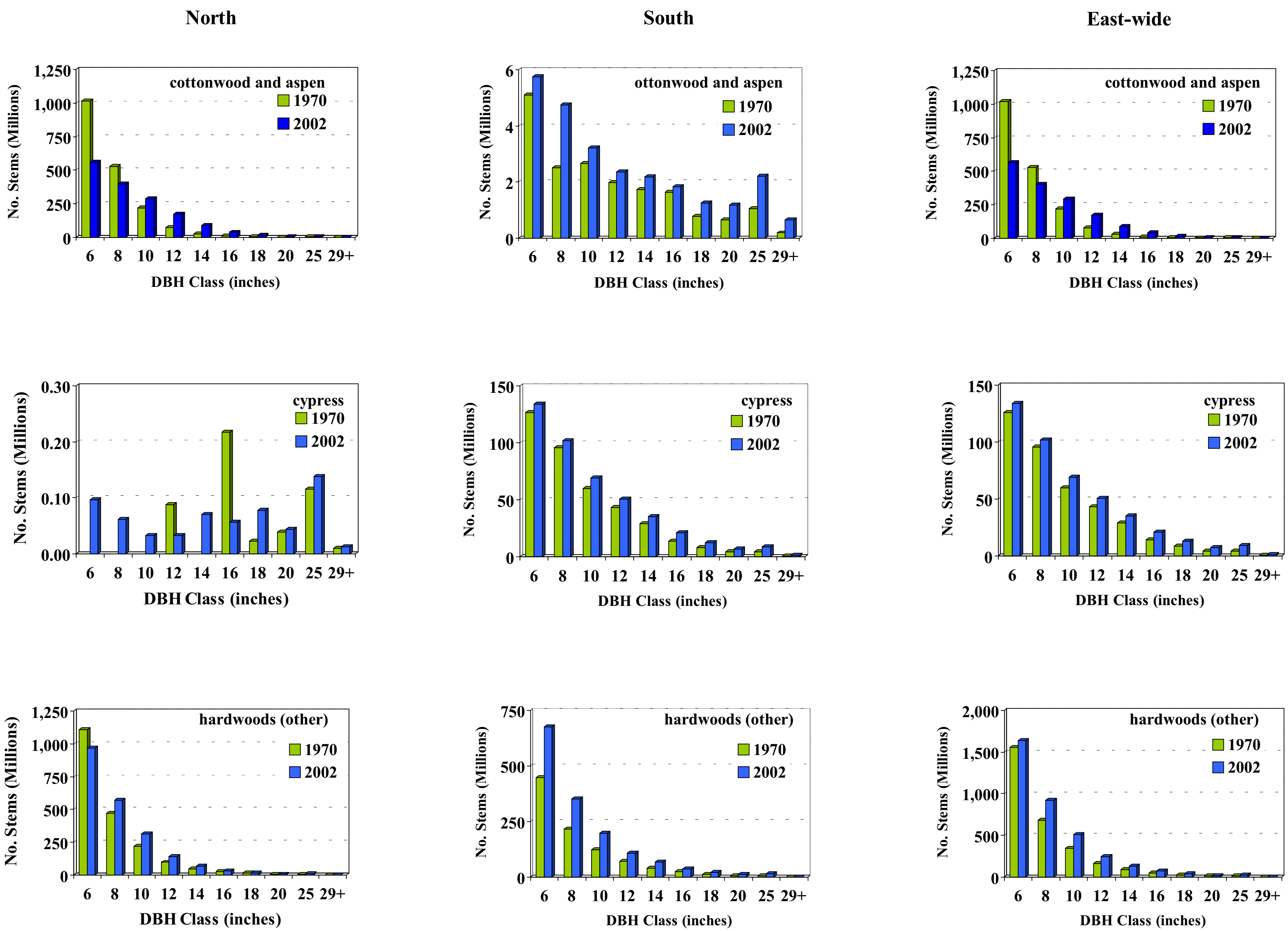
Wright, S. 1977. Evolution and the genetics of populations. Vol. 3: experimental results and evolutionary deductions. Chicago, IL: University of Chicago Press. 614 p.

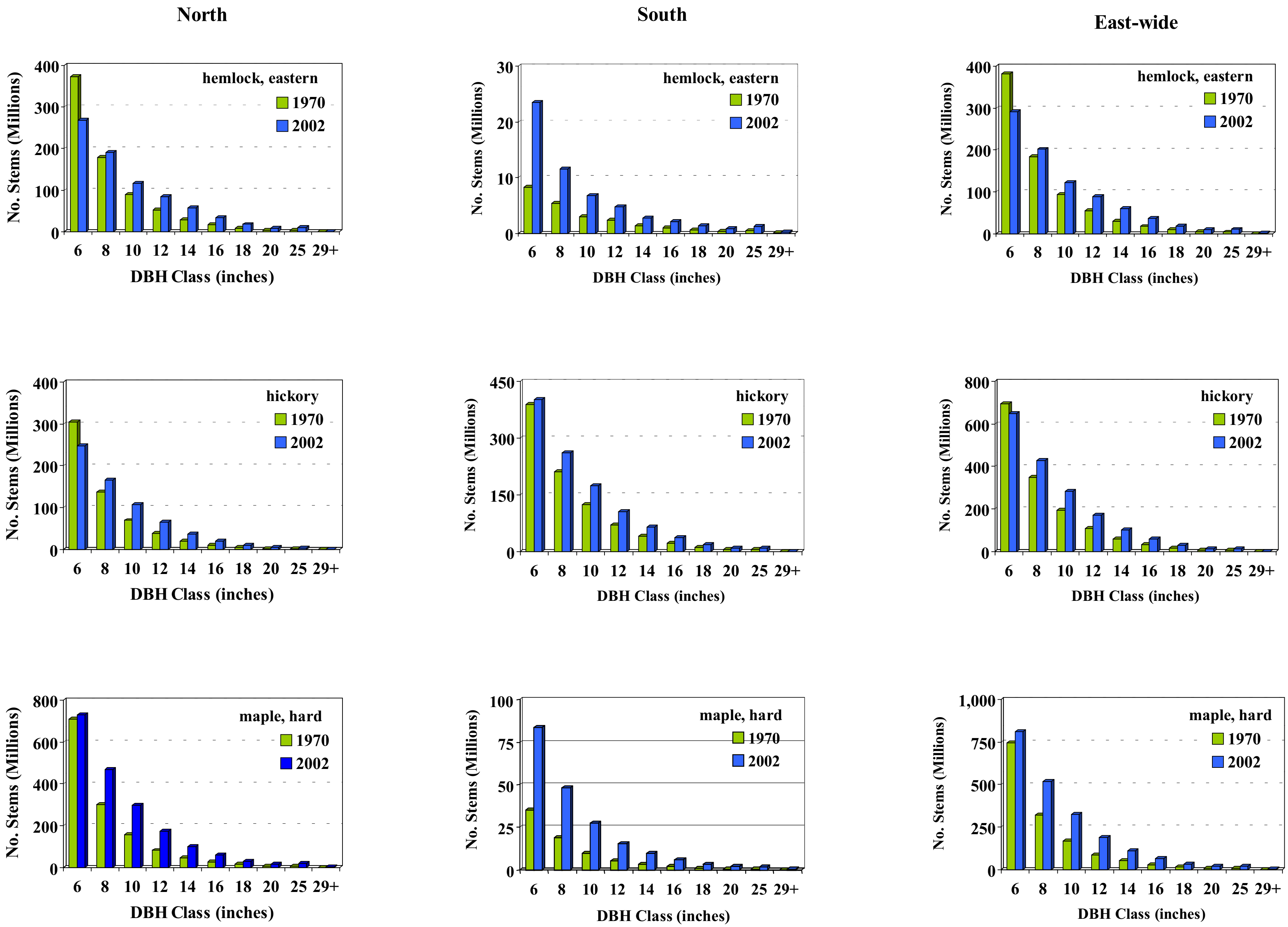
APPENDIX A (following pages). Number of stems >5 inches, by diameter class mid-points, in 1970 and 2002, for tree species or groups of species, for the two eastern U.S. Forest Service planning regions (North and South) (USDA Forest Service 2001), plus east-wide summaries. Data are from the Forest Inventory and Analysis database and are available upon request from Brad Smith (U.S. Forest Service, 1400 Independence Ave. S.W., Washington, DC 20250-1119, bsmith@fs.fed.us). See figure 1 for boundaries of U.S. Forest Service planning regions.

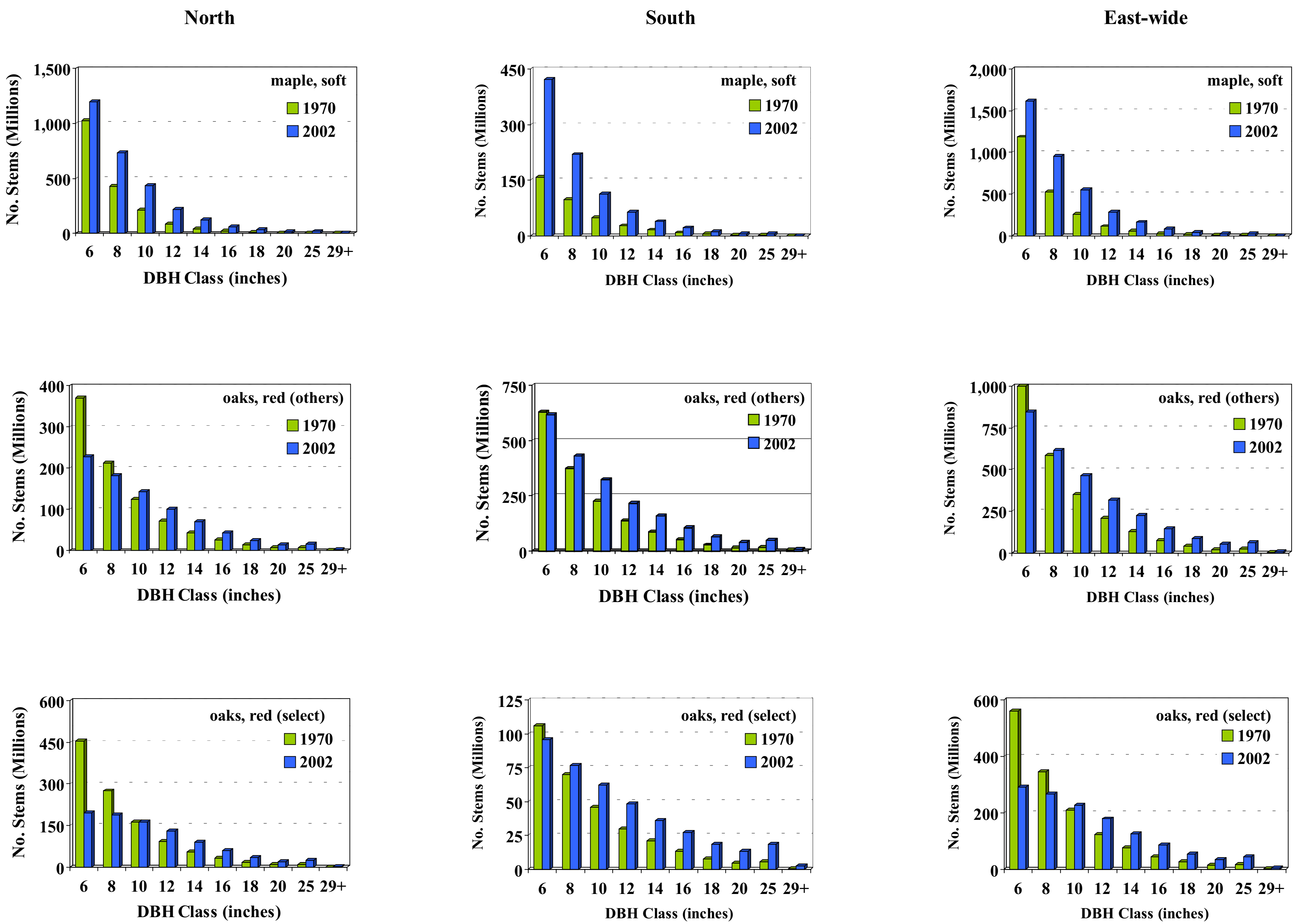

North
South
East-wide
ash
1970
2002
basswood
1970
2002
beech
1970
2002
No. Stems (Millions)
DBH Class (inches)
6 8 10 12 14 16 18 20 25 29+
0 100 200 300 400 500
0 50 100 150 200
0 150 300 450 600
0 50 100 150
0 4 8 12 16
0 50 100 150
0 75 150 225 300
0 10 20 30 40
0 75 150 225 300

North | South | East-wide

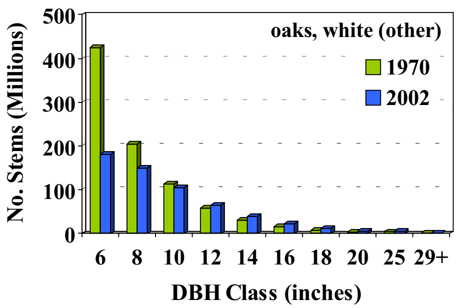


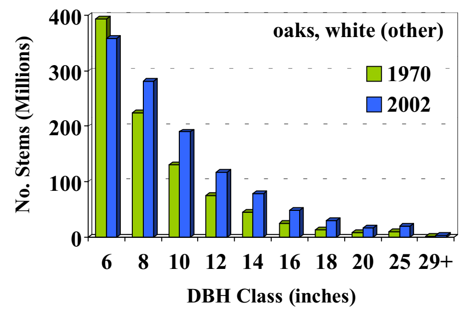

North
South
East-wide
cottonwood and aspen
ottonwood and aspen
cypress
hardwoods (other)
1970
2002
No. Stems (Millions)
DBH Class (inches)
6
8
10
12
14
16
18
20
25
29+
0
250
500
750
1,000
1,250
2
4
6
0.00
0.10
0.20
0.30
50
100
150
1,500
2,000

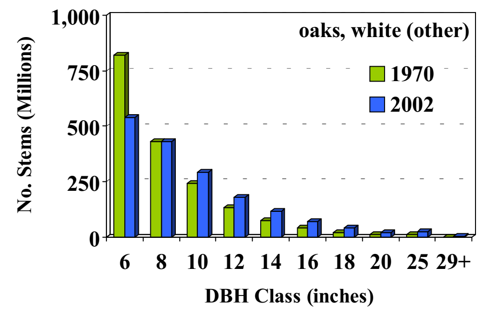

North
South
East-wide
hemlock, eastern
hickory
maple, hard
1970
2002
No. Stems (Millions)
DBH Class (inches)
6 8 10 12 14 16 18 20 25 29+
0 100 200 300 400
0 10 20 30
0 150 300 450
0 200 400 600 800
0 25 50 75 100
0 250 500 750 1,000

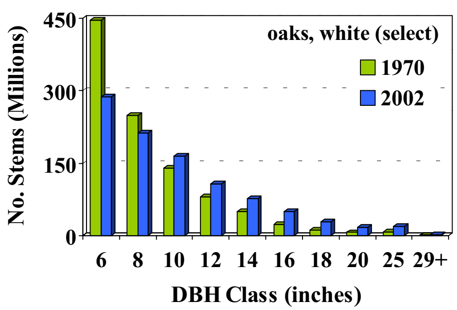

North
South
East-wide
maple, soft
oaks, red (others)
oaks, red (select)
1970
2002
No. Stems (Millions)
DBH Class (inches)
6
8
10
12
14
16
18
20
25
29+
0
150
300
450
500
1,000
1,500
2,000
250
750
100
200
400
600
25
50
75
125

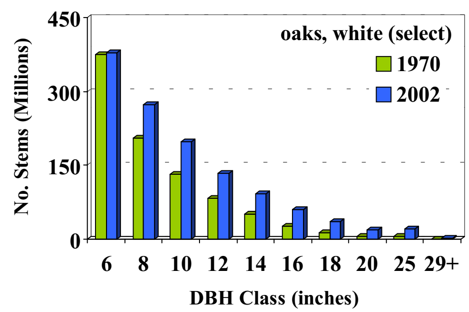
North | **South** | **East-wide**

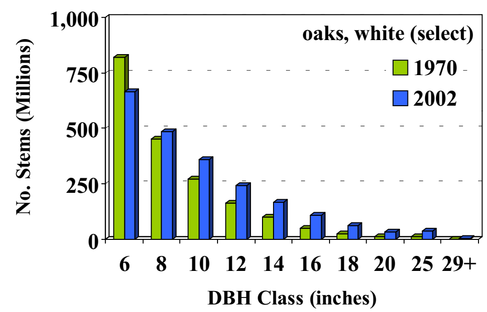


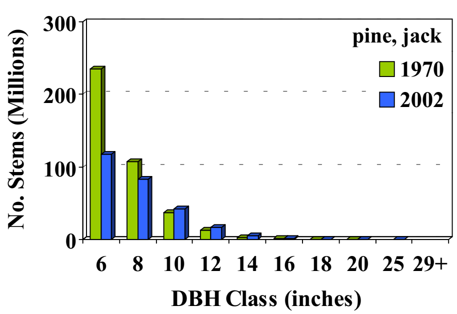


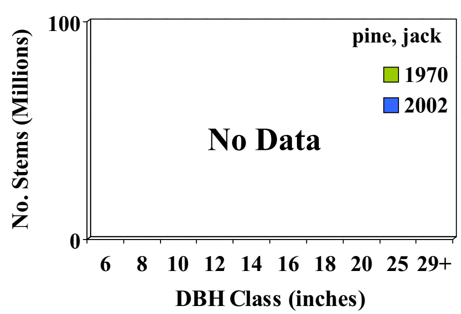


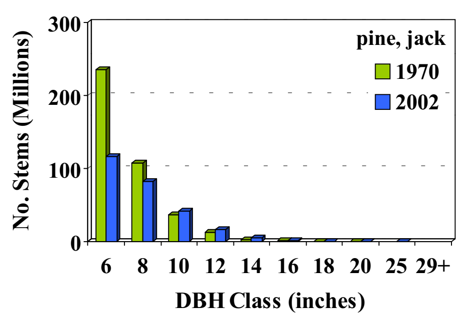




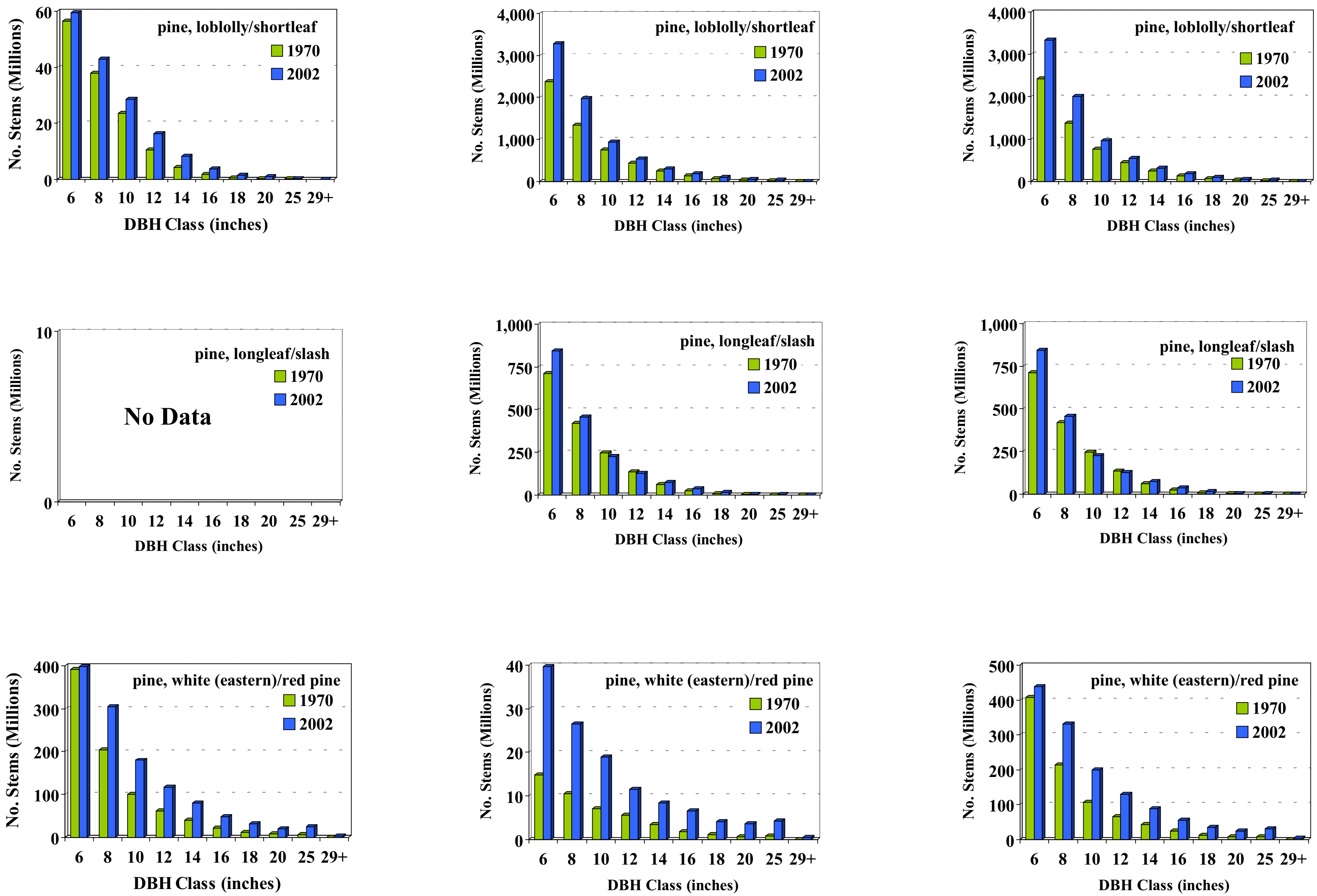


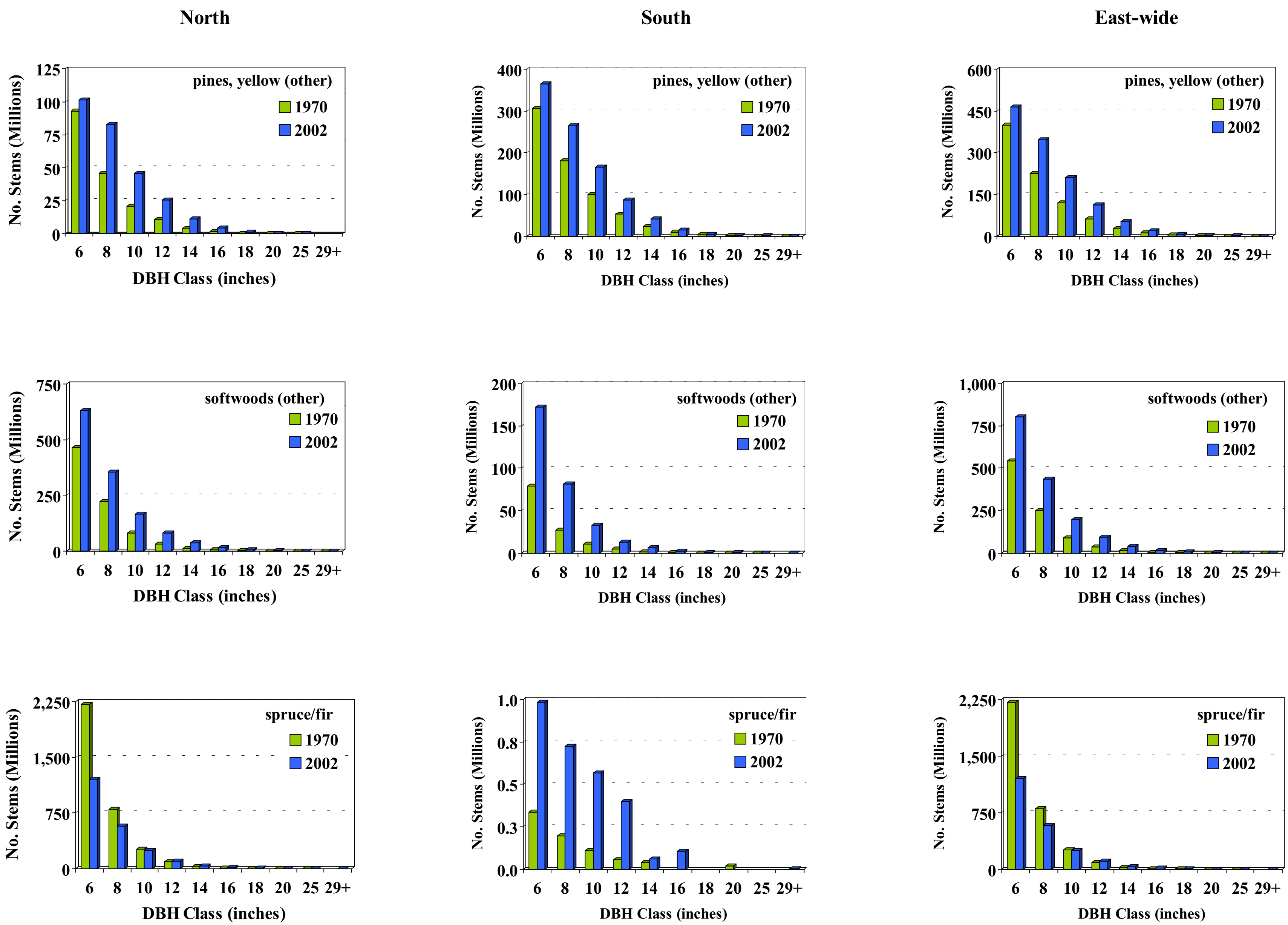




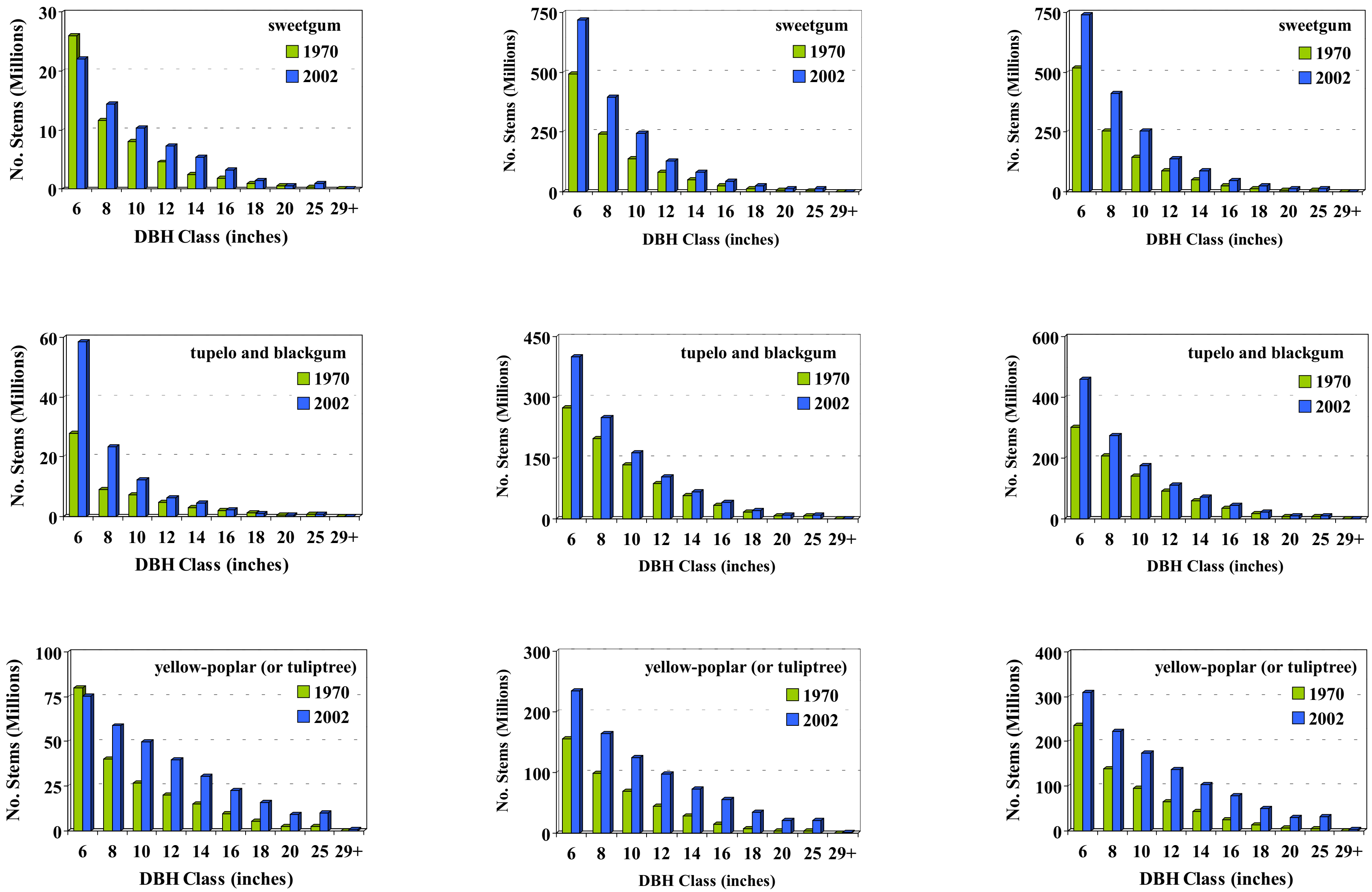


North South East-wide



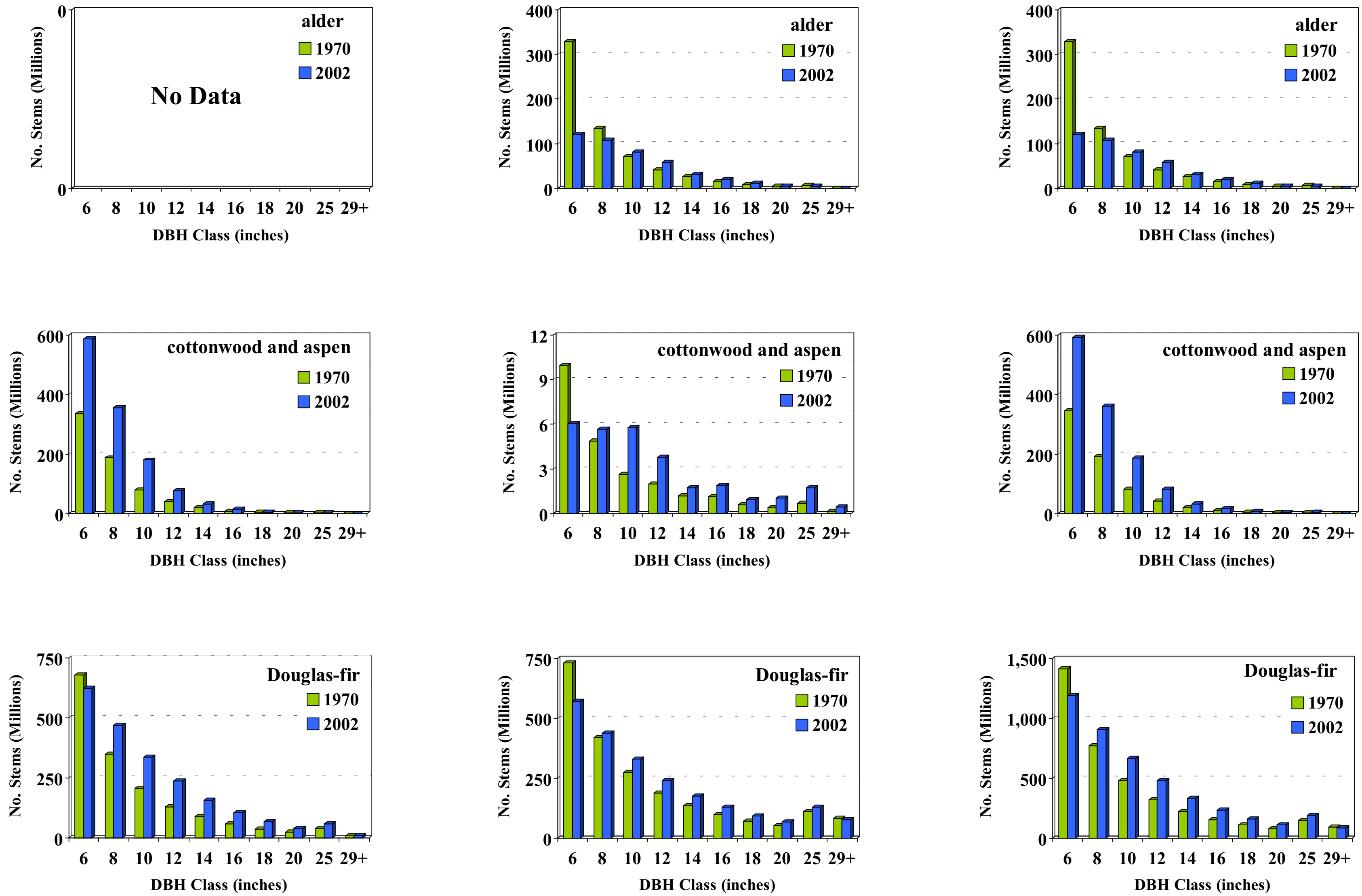

North
South
East-wide
pines, yellow (other)
softwoods (other)
spruce/fir
1970
2002
No. Stems (Millions)
DBH Class (inches)
6 8 10 12 14 16 18 20 25 29+
0 25 50 75 100 125
0 100 200 300 400
0 150 300 450 600
0 250 500 750
0 50 100 150 200
0 250 500 750 1,000
0 750 1,500 2,250
0.0 0.3 0.5 0.8 1.0

North | South | East-wide

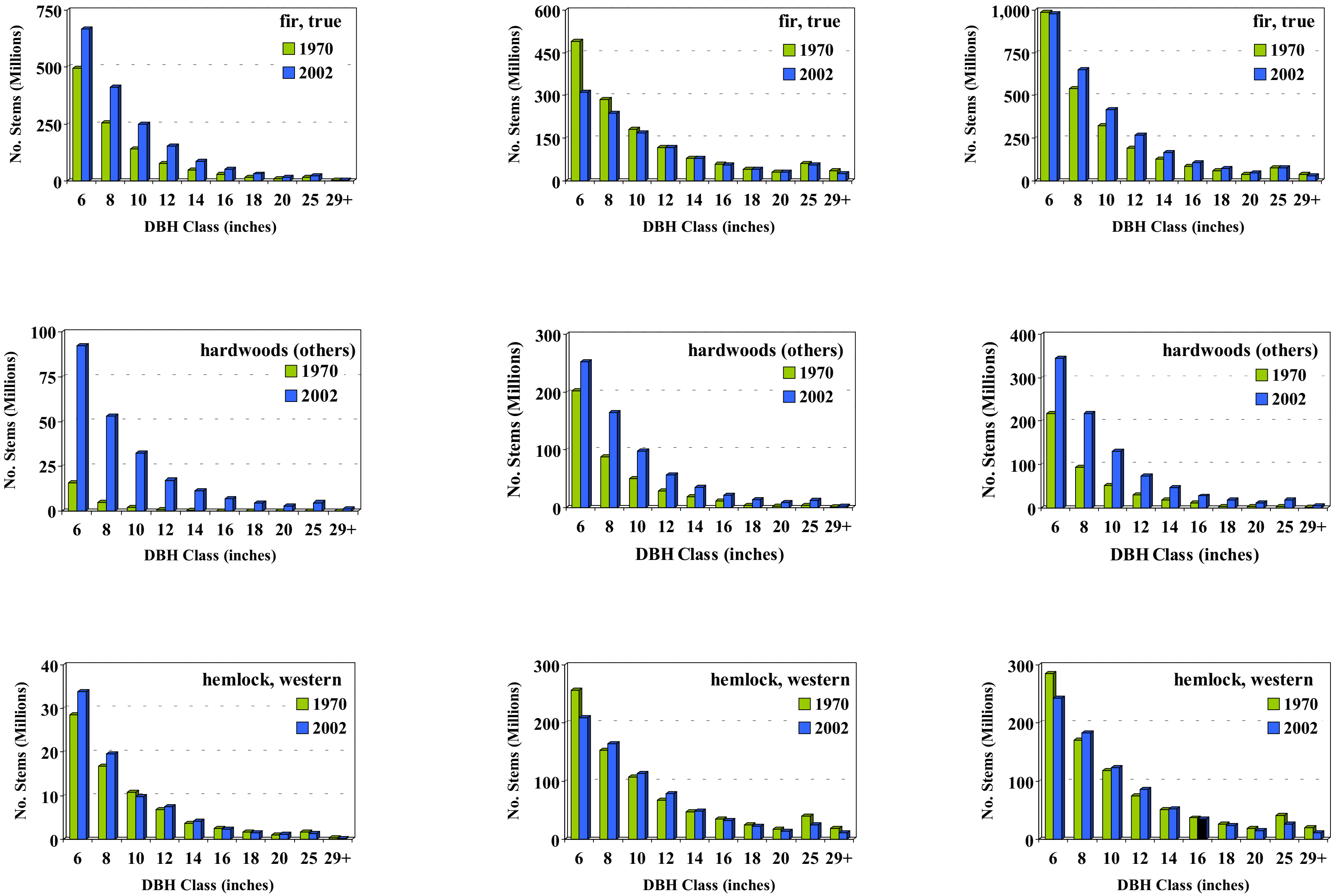


APPENDIX B (following pages). Number of stems >5 inches, by diameter class mid-points, in 1970 and 2002, for tree species or groups of species, for the two western U.S. Forest Service planning regions (Pacific Coast and Rocky Mountain) (USDA Forest Service 2001), plus west-wide summaries. Data are from the Forest Inventory and Analysis database and are available upon request from Brad Smith (U.S. Forest Service 1400 Independence Ave. S.W., Washington, DC 20250-1119, bsmith@fs.fed.us). See figure 1 for boundaries of U.S. Forest Service planning regions.

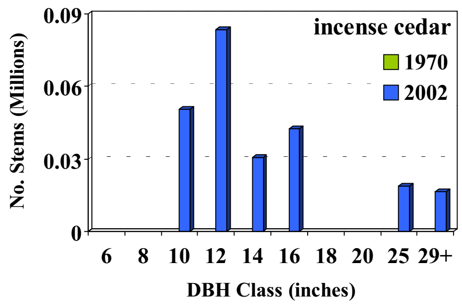
Rocky Mountain | Pacific Coast | West-wide

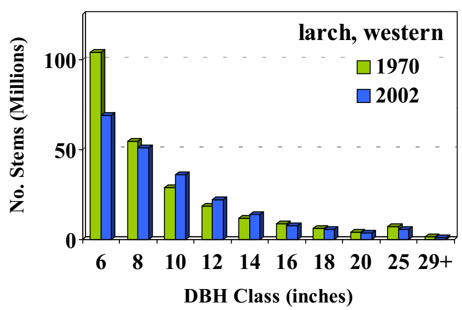


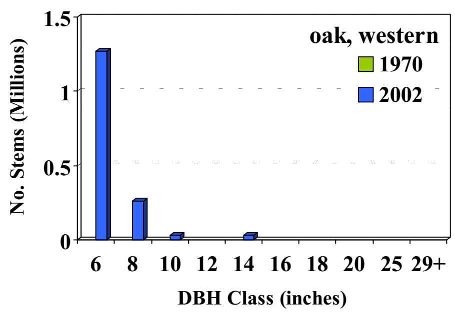
Rocky Mountain | **Pacific Coast** | **West-wide**



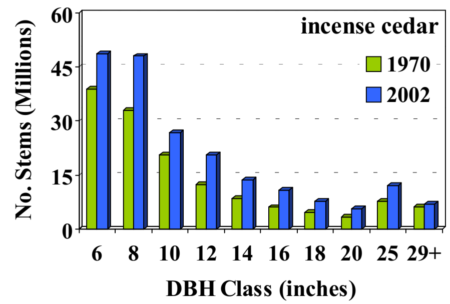
Rocky Mountain

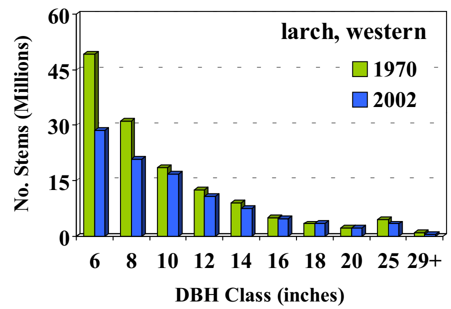


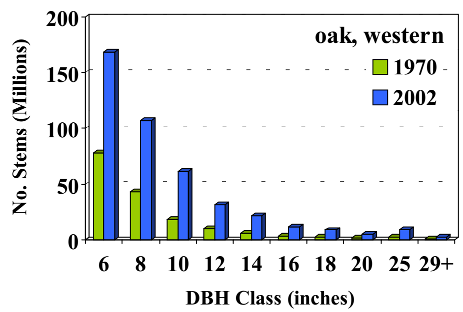




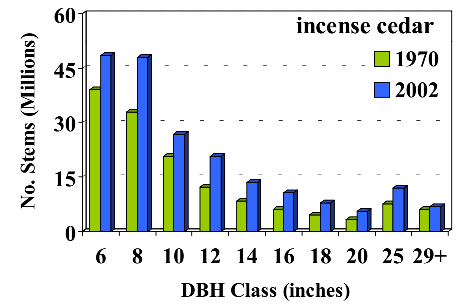
Pacific Coast

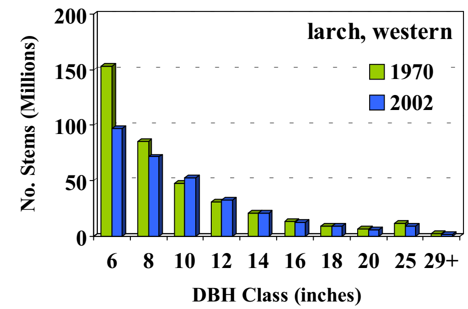


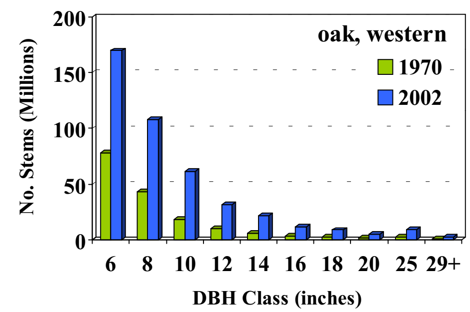




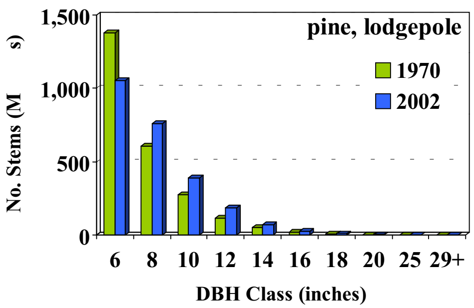
West-wide

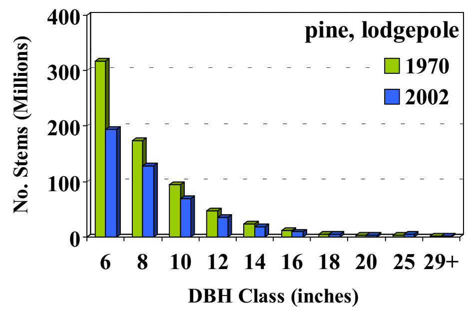


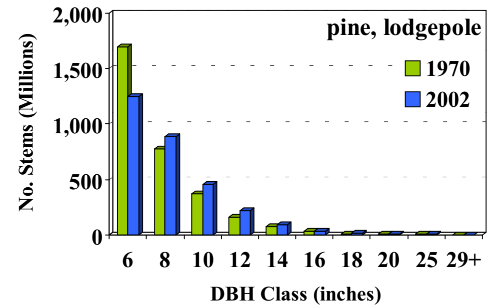


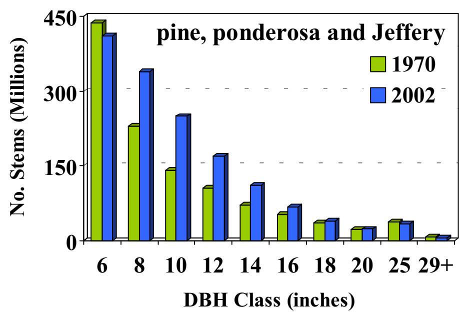


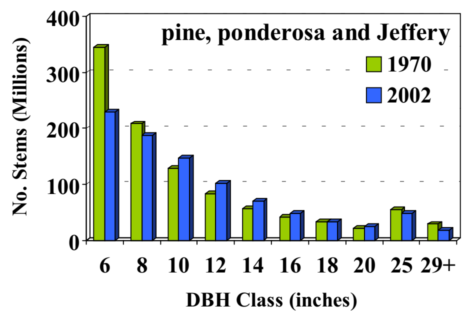
Rocky Mountain | **Pacific Coast** | **West-wide**

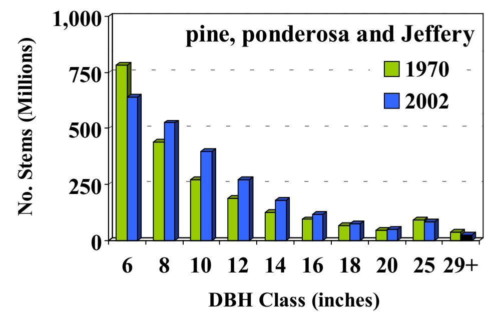


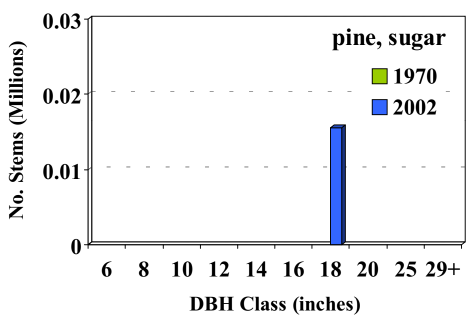


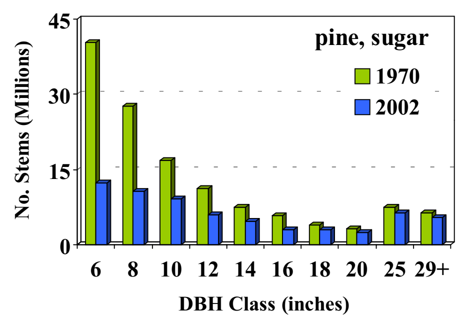


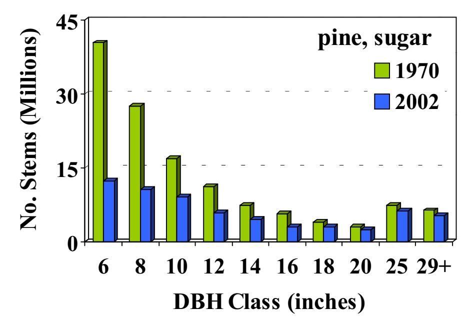




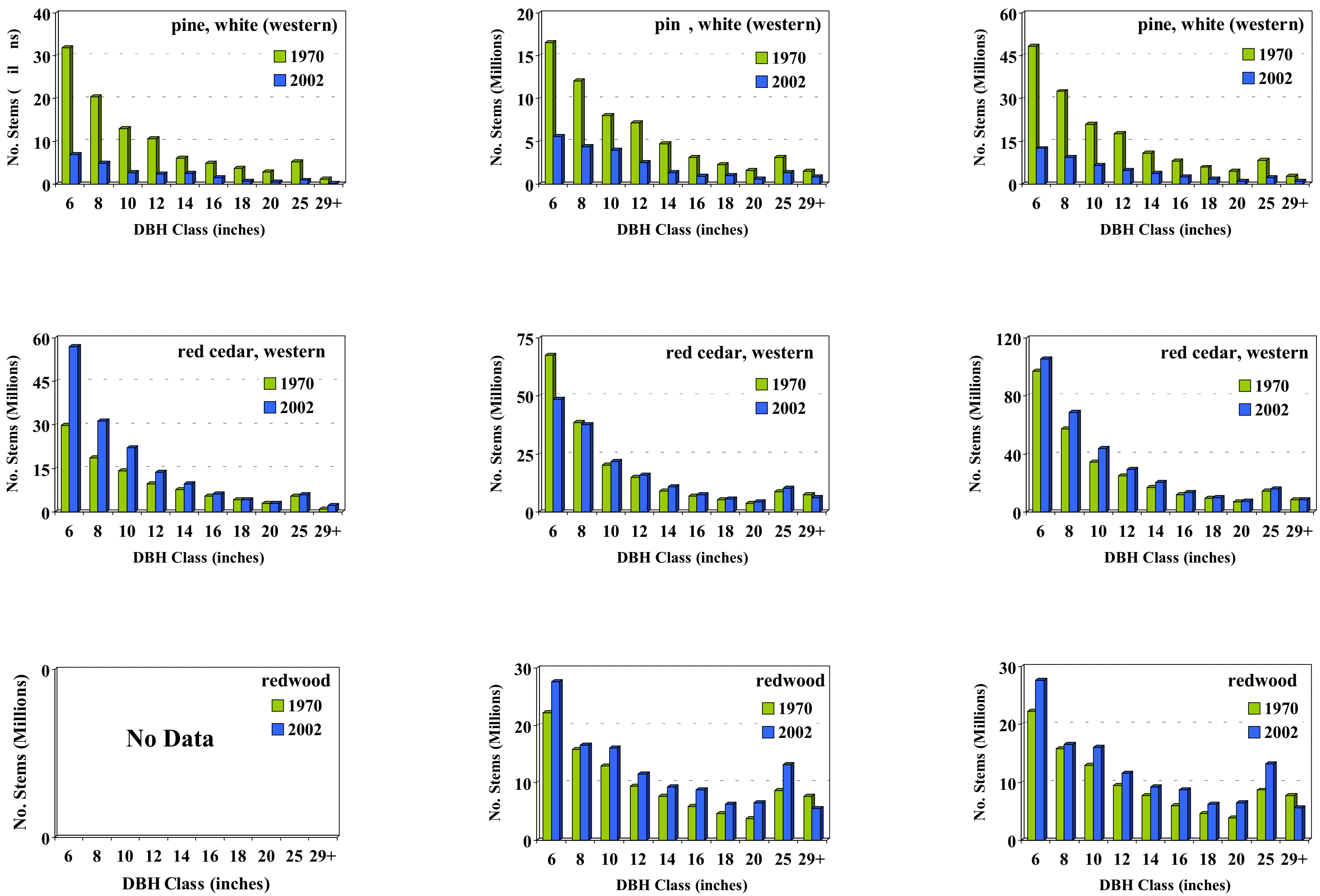




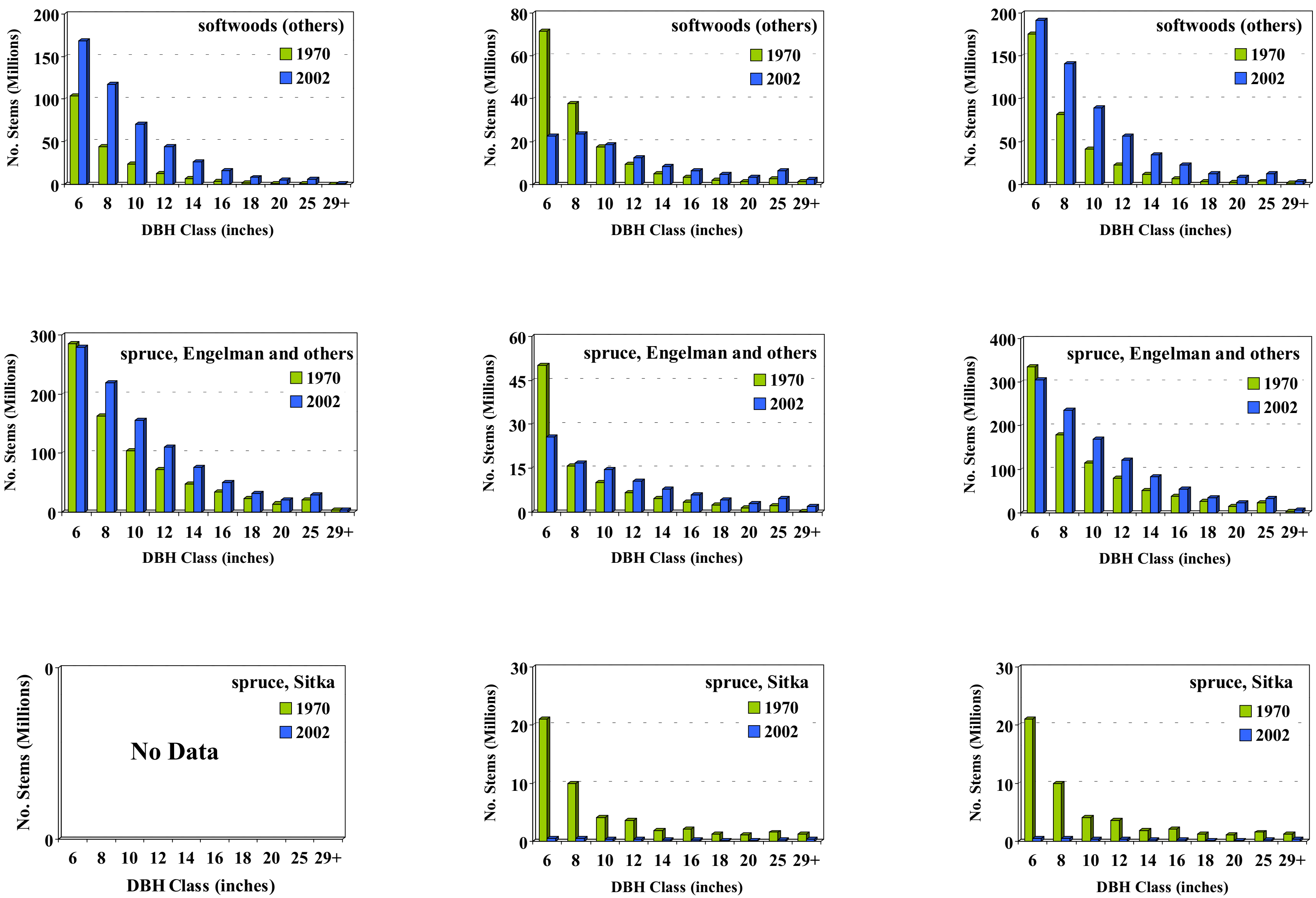




Rocky Mountain **Pacific Coast** **West-wide**



Rocky Mountain | **Pacific Coast** | **West-wide**



BLANK