Courtesy S.L. Grace, NBS

Terrestrial Ecosystems

Science Editor

Raymond J. Boyd
Bureau of Land Management
Service Center
Denver Federal Center, Bldg. 50
Denver, CO 80225

Text by

B.D. Keeland
National Biological Service
Southern Science Center
700 Cajundome Blvd.
Lafayette, LA 70506

Overview

Whereas several other sections in this volume cover individual species or locations, the articles in this section address the integration of individual species into communities and ecosystems (*see* glossary). Terrestrial ecosystems include a rich variety of community types and cover a range extending from nearly aquatic wetlands along our coasts and myriad rivers, lakes, and streams, to mountain tops and arid, desert locations. The diversity of these ecosystems offers both challenge and opportunity. The challenge stems from the sheer number of potential ecosystems to be analyzed. Grossman and Goodin (this section) discuss 371 imperiled and critically imperiled communities, and state that this number represents only 10%-15% of all terrestrial communities. This implies a minimum of 2,500-3,500 individual terrestrial community types. Obviously, a single report cannot hope to address more than a few of these many terrestrial communities and ecosystems.

Discussions of biological diversity have traditionally revolved around the protection of individual species. More recently, we have begun to realize that protection of community or ecosystem diversity is equally important. Patchwork conversions of natural landscapes for agriculture, silviculture, and development result in a fragmentation that leaves small remnant areas of natural ecosystems (Burgess and Sharpe 1981). As these natural patches become smaller and more isolated, their ability to maintain healthy populations of many plant and animal species is reduced (Harris 1984). As individual species are lost from each fragment, the community changes and both species and ecosystem diversity are reduced. Thus, large numbers of natural ecosystems are now in danger.

Kendall (this section) discusses one such imperiled ecosystem. The whitebark pine (*Pinus albicaulis*) ecosystem of the western mountains is endangered because of the combined effects of an introduced disease and fire suppression. The effects of introduced diseases on natural species and ecosystems have been well documented. Several species, such as the American chestnut (*Castanea dentata*), have been virtually eliminated and other species have been greatly reduced by introduced diseases. The effects on ecosystems where these species were previously found have been dramatic (Shugart and West 1977).

Alteration of natural fire regimes has played a major role in the reshaping of natural ecosystems. In many systems a reduction in fire frequency can lead to invasion by fire-intolerant species and eventual loss of the original ecosystem.

This is shown by Henderson and Epstein (this section) in their discussion of how fire supression and other factors caused tremendous losses of oak savannas throughout the Midwest. In other systems, an increase in fire frequency can also lead to changes in ecosystem structure and function. Although we now realize that fire is a natural and necessary part of many ecosystems, it was not until after the devastating fires of Yellowstone National Park that the general public was alerted to the benefits of such fires (Elfring 1989). An effective fire-suppression program can allow accumulation of vast amounts of detritus (dead organic material such as leaves, branches, and stems). If this material is not consumed periodically by small fires burning along the forest floor, it will accumulate to the point of providing raw materials for an exceptionally intense fire that can burn tree crowns and destroy the existing forest. Ferry et al. (this section) discuss four fire-adapted ecosystems that have been affected by modified fire regimes and conclude, “Managers must balance the suppression program with a program of prescribed fire applied on a landscape scale if we are to meet our stewardship responsibilities.”

Numerous variables in addition to disease and fire affect our natural resources. These variables include pollution (Peterson, this section; Nash et al., this section), conversions to other uses, harvesting activities such as logging, and global climate change. Cole (this section) demonstrates that over the past 5,000 years change has been a natural part of our terrestrial ecosystems. Within a given ecosystem some species decline in importance while others increase over time, resulting in a change in the overall character of the ecosystem. A key feature to stand out in the 5,000-year chronology developed by Cole is that current rates of change are about 10 times higher than presettlement rates. Human intervention in one form or another is now the principal agent of change. Darr (this section) provides a review of U.S. Forest Service data and discusses changes being brought about by forestry-management practices. At a reduced spatial scale, Keeland et al. (this section) discuss changes within the forested wetlands of the southeastern United States. Forested wetlands have been especially reduced and fragmented as a result of land-use conversions, predominantly to agricultural activities.

A common thread here, as in all sections in this report, is that if unchecked, human activities will continue to result in an upset balance of species interactions, alteration of ecosystems, and extensive habitat loss.

References

Burgess, R.L., and D.M. Sharpe, eds. 1981. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York. 310 pp.

Elfring, C. 1989. Yellowstone: fire storm over fire management. BioScience 39(10):667-672.

Harris, L.D. 1984. The fragmented forest. The University of Chicago Press, Chicago. 211 pp.

Shugart, H.H., and D.C. West. 1977. Development of an Appalachian deciduous forest succession model and its application to assessment of the impact of the chestnut blight. Journal of Environmental Management 5:161-179.

U.S. Forest Resources

by
David R. Darr
U.S. Forest Service

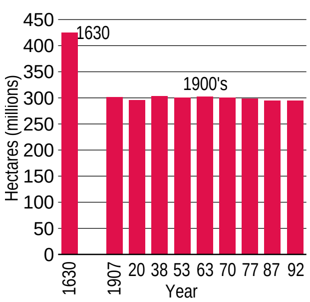


Fig. 1. Forest land area (Powell et al. 1993).

The Secretary of Agriculture is directed by law to make and keep current a comprehensive inventory and analysis of the present and prospective conditions of and requirements for the renewable resources of U.S. forests and rangelands. This inventory includes all forests and rangelands, regardless of ownership. The work is carried out by people in the Forest Inventory and Analysis program of the U.S. Department of Agriculture Forest Service (USFS).

Inventories provide key forest resource information for planners and policy makers. Increasingly, people turn to these inventories for information on biological diversity, forest health, and developmental decisions.

Information is collected from over 130,000 permanent sample plots selected to assure statistical reliability. Vegetation on the plots is measured on average about every 10 years. Characteristics of the vegetation and land are measured, including ownership, productivity for timber production, the kinds and sizes of trees, how fast trees are growing, whether any trees have died from natural causes, and whether any trees have been cut (USFS 1992).

Characteristics of Forest Land

Over the years, the U.S. forest cover has changed because of the way people use and manage forest land. Today, about 33% of the U.S. land area, or 298 million ha (737 million acres), is forest land, about two-thirds of the forested area in 1600 (Fig. 1). Since 1600, some 124 million ha (307 million acres) of forest land have been converted to other uses, mainly agricultural. More than 75% of this conversion occurred in the 19th century, but by 1920, clearing forests for agriculture had largely halted.

Some 34% of all forest land is federally owned and managed by the U.S. Forest Service, the Bureau of Land Management, and other federal agencies. The rest is owned by nonfederal public agencies, forest industry, farmers, and other private individuals. About 19 million ha (47 million acres; 6% of all U.S. forest land) are reserved from commercial timber harvest in wilderness, parks, and other land classifications.

Forest land is widely but unevenly distributed. North Dakota has the smallest percentage of forest cover (1%) and Maine has the greatest (89%). Forest areas vary greatly from sparse scrub forests of the arid interior West to the highly productive forests of the Pacific coast and the South, and from pure hardwood forests to multispecies mixtures and coniferous forests. In total, 52% of the forest land is east of the Great Plains states. In the East, the oak-hickory forest type group is most common, while in the West, the category referred to as “other softwoods” is most common.

U.S. forests provide wildlife habitat and thereby support biodiversity; take carbon out of the air and thus serve as carbon sinks; and provide the outdoor environments desired by many people for recreation.

Timberland forests are logged for lumber, plywood, and paper products. This timberland is generally the most productive and capable of producing at least 1.4 m^3 of industrial wood per hectare a year (20 ft^3/acre) and is not reserved from timber harvest (Powell et al. 1993). Two-thirds of the nation’s forested ecosystems (198 million ha or 490 million acres) are classed as timberland. Because of historical interest in timber production, more information is available for the characteristics of timber inventories on timberland than for other forest land.

Timberland ownership patterns vary throughout the United States. For the country as a whole, 73% of all timberland is owned by private individuals and firms. The remaining 27% is in federal, state, and other public ownerships. Much of the privately owned land is in the East and much of the national forest land is in the West (Fig. 2). Most of the publicly owned land is managed according to plans that account for the various uses and values provided by forest cover. Forest industry lands are generally managed with timber production being the main interest. Other private forest lands are managed for a variety of interests, reflecting the divergent views of the some 6 million owners in this category.

The nation’s timberland contains an estimated 24.3 billion m^3 (858 billion ft^3) of timber, of which 92% is in growing stock—live, sound trees suited for roundwood (timber) products. Softwoods such as pine are generally used to make lumber and plywood for use in construction. Hardwoods, such as oak, are used in making furniture and pallets. Both softwoods and hardwoods are used in manufacturing paper products. The nation’s softwood growing stock volume amounts to 57% of the total, with about 66% of this volume in the West. Total softwood growing stock volume has been slightly declining recently (Fig. 3). By contrast, hardwood growing stock volume increased 7% between 1987 and 1992. More than 90% of all hardwood timber volume is in the eastern United States.

Mortality, Growth, Harvest

Mortality is the result of natural causes such as insects, disease, fire, and windthrow. Between 1962 and 1986, mortality averaged 122 million m^3 (4.3 billion ft^3) per year. Mortality increased to 155 million m^3 (5.5 billion ft^3) in 1991, but was still less than 1% of the U.S. growing stock volume.

Net annual growth, which already has mortality subtracted out, totaled 612 million m^3 (21.6 billion ft^3) in 1991—about 2.7% of the growing stock inventory. Total growing stock growth declined about 2% between 1986 and 1991 (Fig. 4), the first decline in net annual growth since 1952. The decline between 1986 and 1991 occurred with softwoods, which declined 4.4% to 339 million m^3 (12 billion ft^3). Net annual growth for hardwoods increased 0.9%.

Removals from timber inventories are losses by other than natural causes (mortality) and include harvest of roundwood products. Timber removals from growing stock inventory in 1991 totaled 461.5 million m^3 (16.3 billion ft^3) or 2.1% of the inventory. Average timber removals have risen each decade since the 1950’s. Almost 55% of all timber removals came from the forests of the South, up from 45% in 1970. Twenty-three percent of all removals came from Pacific coast forests, 17% from the North, and 5% from forests in the Rocky Mountains. Softwoods accounted for two-thirds of all growing stock removals in 1991. Timber removals continued to be concentrated on private land in 1991.

The growth-removals ratio for the United States is greater than one for all species (1.3), for softwoods (1.1), and for hardwoods (1.8), which indicates that the timber inventory is increasing. In the 1920’s, timber growth was about one-half the rate of harvest. By the 1940’s, improved forest growth rates (partly because of forest protection from fire), as well as declines in harvest rates, resulted in timber growth and harvest coming into approximate balance.

References

Powell, D.S., J.L. Faulkner, D.R. Darr, Z. Zhu, and D.W. MacCleery. 1993. Forest resources of the United States, 1992. Gen. Tech. Rep. RM-234. U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 132 pp.

USFS. 1992. Forest Service resource inventories: an overview. Forest inventory, economics, and recreation research. U.S. Forest Service, Washington, DC. 39 pp.

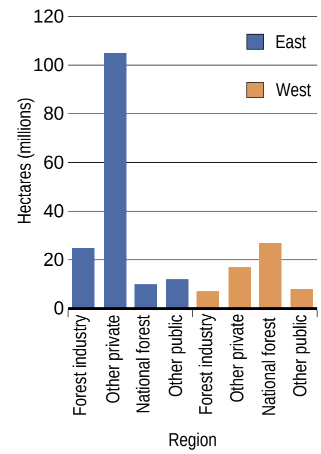


Fig. 2. Timberland ownership patterns by regions, 1992 (Powell et al. 1993).

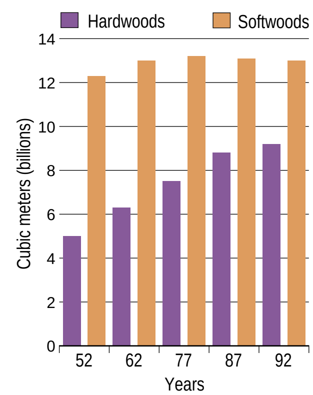


Fig. 3. Softwood and hardwood growing stock volume, selected years (Powell et al. 1993).

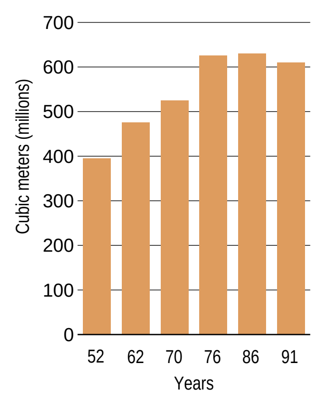


Fig. 4. Net annual growth, selected years (Powell et al. 1993).

For further information:

David R. Darr
U.S. Forest Service
PO Box 96090
Washington, DC 20090

Southern Forested Wetlands

by
B.D. Keeland
James A. Allen
Virginia V. Burkett
National Biological Service

European settlers in many parts of the southern United States encountered a landscape largely comprising forested wetlands. These wetlands were a major feature of river floodplains and isolated depressions or basins from Virginia to Florida, west to eastern Texas and Oklahoma, and along the Mississippi River to southern Illinois (Fig. 1). Based on the accounts of pre-20th-century naturalists such as Audubon, Banister, John and William Bartram, Brickell, and Darby, the flora and fauna of many wetlands were unusually rich even by precolonial standards (Wright and Wright 1932). These early travelers described vast unbroken forests of oaks, ashes, maples (*Quercus, Fraxinus, Acer*), and other tree species, many with an almost impassable understory of saplings, shrubs, vines, switch cane, and palmetto. Low swampy areas with deep, long-term flooding were dominated by baldcypress (*Taxodium distichum)* and tupelo (*Nyssa aquatica* or *N. sylvatica* var *biflora*) and typically had sparse understories.

Most southern forested wetlands fall in the broad category of bottomland hardwoods, characterized and maintained by a natural hydrologic regime of alternating annual wet and dry periods and soils that are saturated or inundated during a portion of the growing season. Variations in elevation, hydroperiod, and soils result in a mosaic of plant communities across a floodplain. Wharton et al. (1982) classified bottomland hardwoods into 75 community types, including forested wetland types such as Atlantic white cedar bogs (*Chamaecyparis thyoides*), red maple (*Acer rubrum* var *drummondii*) and cypress-tupelo swamps, pocosins, hydric hammocks, and Carolina bays.

Realistic estimates of the original extent of forested wetlands are not available because accurate records of wetlands were not maintained until the early 20th century, and many accounts of wetland size were little more than speculation (Dahl 1990). Klopatek et al. (1979) estimated the precolonial forested wetland area of the United States to be about 27.2 million ha (67.2 million acres), but Abernathy and Turner (1987) suggested that this figure was low because it ignored small isolated wetlands.

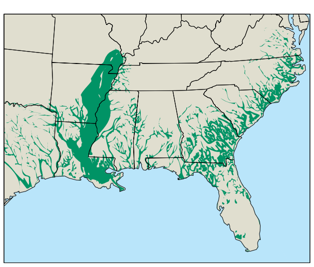


Fig. 1. Approximate distribution of forested wetlands along rivers and streams in the southeastern United States prior to European colonization (Putnam et al. 1960).

Status

Estimates of the current forested wetland area vary. Shaw and Fredine (1956) estimated that as of the mid-1950's, the United States had about 19.1 million ha (47.2 million acres) of forested wetlands. Frayer et al. (1983) reported a similar total, 20.1 million ha (49.7 million acres), as of the mid-1970's. Between 1940 and 1960, the area of southern bottomland hardwoods increased from about 14.8 to 15.1 million ha (36.6 to 37.3 million acres) but declined to 12.5 to 13.1 million ha (30.9 to 32.4 million acres) by the mid-1970's (Hefner and Brown 1985; Turner et al. 1988). By the mid-1980's, an additional 1.4 million ha (3.5 million acres) of forested wetlands were lost, mostly from the southeastern United States.

The Southeast (including Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, and Tennessee) makes up only 16% of the surface area of the conterminous United States yet accounts for about 47% of the total wetland area and 65% of the forested wetland area (Hefner and Brown 1985). Fifteen percent of the land surface of the Southeast can be categorized as wetlands, whereas only 5% of the land surface on a national basis is wetlands.

Before the mid-1970's, about 54% of palustrine wetland losses on a national basis were in forested areas. Palustrine wetlands include all nontidal wetlands dominated by trees, shrubs, persistent emergents, emergent mosses or lichens, and all such wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 ppt (Cowardin et al. 1979). Between the mid-1950's and the mid-1970's, more than 2.2 million ha (5.4 million acres) of palustrine forested wetlands were lost within the Southeast, accounting for 92% of the national loss for this wetland type (Hefner and Brown 1985). Since the mid-1970's, loss of forested wetlands has accounted for 95% of all palustrine wetland losses (Dahl et al. 1991).

Despite dramatic losses since the beginning of the colonial period, southern forested wetlands currently account for about 36% of all wetlands and 60% to 65% of all forested wetlands in the conterminous United States (Hefner and Brown 1985; Dahl et al. 1991). Although loss rates have declined recently, most wetland acreage lost every year in the United States is from southern forested wetlands (Alig et al. 1986).

The most dramatic wetland loss in the entire nation has occurred in the forested wetlands of the Lower Mississippi River Alluvial Floodplain (LMRAF). This vast wetland extends nearly 1,000 km (621 mi) from the confluence of the Mississippi and Ohio rivers to the Gulf of Mexico and originally covered more than 10.1 million ha (25.0 million acres; Hefner and Brown 1985). About 8 million ha (19.8 million acres) of this area were forested wetlands in Arkansas, Louisiana, and Mississippi. Recent estimates reveal that fewer than 2 million ha (4.9 million acres) of forested wetlands remain in the LMRAF (The Nature Conservancy 1992), and the remaining portions of the original area are extremely fragmented (Fig. 2) and have lost many of their original functions (Mitsch and Gosselink 1993). Also, alterations in hydrology and poor timber management practices have resulted in a degraded condition of many of the remaining forests (Alig et al. 1986).

Turner et al. (1988) reported annual loss rates of 3.1% for forested wetlands in Arkansas, 0.9% for Louisiana, and 0.5% for Mississippi from 1960 to 1975. Recent U.S. Forest Service inventories indicate continued annual loss rates of 0.7% and 1.0% for the oak-gum-cypress forest type in the Louisiana and Mississippi portions of the LMRAF (May and Bertelson 1986; Kelly and Sims 1989; Vissage et al. 1992).

Causes of Loss

Since colonial times, wetlands have been regarded as a menace and a hindrance to land development: wastelands that were valuable only if drained. During the mid-19th century, Congress passed the Swamp Lands Acts of 1849, 1850, and 1860, granting swamp and periodically flooded bottomlands to the states. Five southern states received 16.7 million ha (41.3 million acres) for draining. By 1960, over 40 million ha (98 million acres) of former wetland area in the United States were under drainage (Turner et al. 1988). Most wetlands were drained for conversion to agriculture; such conversions account for 87% of our national wetland losses.

Large-scale federal navigation, flood-control, and drainage projects have played a large role in these conversions by making previously flood-prone lands dry enough for planting crops (USDI 1988). Other losses have resulted from construction of flood-control structures and reservoirs, mining and petroleum extraction, and urban development. A 40% increase in the population of the South between 1960 and 1980 (Alig et al. 1986) has accelerated wetland losses.

Future Prospects

A significant future threat is global climate change; in particular, sea-level rise represents a direct threat to thousands of hectares of coastal wetlands (Titus et al. 1984). Although the main effects of sea-level rise would be seen in coastal marshes, extensive areas of bottomland hardwood and swamp forest in Florida and Louisiana could be affected by increased flooding and saltwater intrusion (Titus et al. 1984; Pezeshki et al. 1987; Conner and Brody 1989).

Legislation such as the Clean Water Act and the "Swampbuster" provision of the 1985 Public Law 100-233 "Farm Bill" has slowed, but not completely prevented, the loss of forested wetlands. In the future, however, the amount of new losses of forested wetlands may be of less concern than the fragmentation and degradation of the few remaining large wetland areas.

While the amount of forested wetlands in the South is expected to continue declining, there are good prospects for restoration in some areas. Recognition of the scale and effects of bottomland hardwood losses has resulted in interest in restoration techniques. Serious restoration began in the mid-1980's, when state and federal agencies began reforesting former agricultural lands (Haynes and Moore 1988; Savage et al. 1989; Newling 1990). The pace of reforestation picked up rapidly following the establishment of the Conservation Reserve Program (CRP) and later the Wetland Reserve Program, two federal agricultural programs that provide payment to private landowners who plant trees on a portion of their land. The combined efforts of the agencies and these two agricultural programs have resulted in the planting of about 65,000 ha (160,615 acres) of bottomland hardwood forests in the southern United States since 1985. Most restoration has occurred in the LMRAF.

Prospects for a similar rate of reforestation over the coming decade appear excellent. Federal and state natural resource agencies continue to reforest their lands. In addition, they have become heavily involved in promoting reforestation on private lands through initiatives such as the Wetland Reserve Program, the U.S. Fish and Wildlife Service's Partners for Wildlife Program, and the North American Waterfowl Management Plan.

Partnerships are being sought between the forest industry, individual landowners, universities, and several state and federal agencies. Examples of such partnerships include Scott Paper Company's enrollment of 27,500 ha (67,952 acres) near Mobile, Alabama, in the Gulf Coast Joint Venture of the North American Waterfowl Management Plan, and a reforestation research project being initiated in west-central Mississippi that involves International Paper Company, the National Council of the Pulp and Paper Industry for Air and Stream Improvement, six federal agencies, and two universities.

Although there is growing concern that many reforestation projects have not been fully successful, it is clear that when properly done, reforestation can yield impressive results in the LMRAF region (Allen 1990). The technical feasibility of reforestation, along with the current environment of federal, state, and private cooperation in much of the region, suggests that the LMRAF may be one of the best areas of the country to seriously attempt a net gain of wetlands.

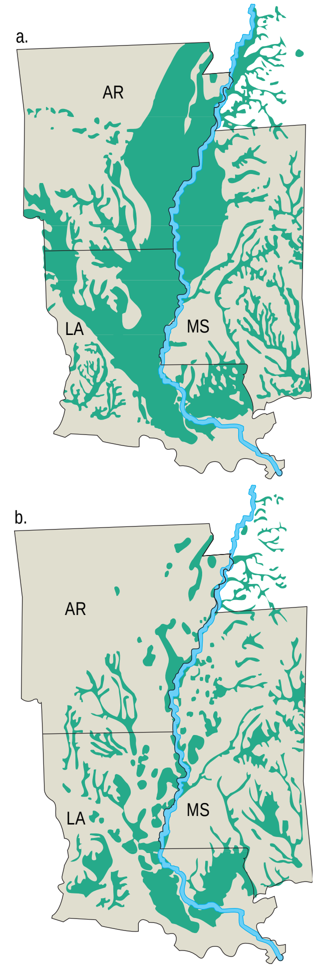


Fig. 2. Distribution of forested wetlands along the Lower Mississippi River: (a) Precolonial extent based on Putnam et al. (1960); (b) recent extent based on 1982 data (data source: U.S. Fish and Wildlife Service, Vicksburg, Mississippi).

References

Abernathy, Y., and R.E. Turner. 1987. US forested wetlands: 1940-1980. BioScience 37:721-727.

Alig, R.J., H.A. Knight, and R.A. Birdsey. 1986. Recent area changes in southern forest ownerships and cover types. U.S. Forest Service, Southern Forest Experiment Station Res. Paper. SE-260. 10 pp.

Allen, J.A. 1990. Establishment of bottomland oak plantations on the Yazoo National Wildlife Refuge complex. Southern Journal of Applied Forestry 14:206-210.

Conner, W.H., and M. Brody. 1989. Rising water levels and the future of southeastern Louisiana swamp forests. Estuaries 12:318-323.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service FWS/OBS-79/31. 103 pp.

Dahl, T.E. 1990. Wetlands losses in the United States 1780's to 1980's. U.S. Fish and Wildlife Service, Washington, DC. 21 pp.

Dahl, T.E., C.E. Johnson, and W.E. Frayer. 1991. Status and trends of wetlands in the conterminous United States, mid-1970's to mid-1980's. U.S. Fish and Wildlife Service, Washington, DC. 28 pp.

Frayer, W.E., M.J. Monahan, D.C. Bowden, and F.A. Graybill. 1983. Status and trends of wetlands and deepwater habitats in the conterminous United States, 1950's to 1970's. Colorado State University, Fort Collins. 31 pp.

Haynes, R.J., and L. Moore. 1988. Reestablishment of bottomland hardwoods within national wildlife refuges in the Southeast. Pages 95-103 *in* Proceedings of a conference: increasing our wetland resources. National Wildlife Federation—Corporate Conservation Council, Washington, DC.

Hefner, J.M., and J.D. Brown. 1985. Wetland trends in the southeastern United States. Wetlands 4:1-11.

Kelly, J.F., and M. Sims. 1989. Forest resources of Mississippi. USDA Forest Service, Southern Forest Experiment Station, Res. Bull. SO-147. 63 pp.

Klopatek, J.M., R.J. Olson, C.J. Emerson, and J.L. Jones. 1979. Land use conflicts with natural vegetation in the United States. Environmental Conservation 6:192-200.

May, D.M., and D.F. Bertelson. 1986. Forest statistics for Louisiana parishes. USDA Forest Service, Southern Forest Experiment Station, Res. Bull. SO-115. 59 pp.

Mitsch, W.J., and J.G. Gosselink. 1993. Wetlands, 2nd ed. Van Nostrand Reinhold, New York. 722 pp.

Newling, C.J. 1990. Restoration of bottomland hardwood forests in the Lower Mississippi Valley. Restoration and Management Notes 8:23-28.

Pezeshki, S.R., R.D. Delaune, and W.H. Patrick, Jr. 1987. Response of baldcypress (*Taxodium distichum* L. var. *distichum*) to increases in flooding salinity in Louisiana's Mississippi River Deltaic Plain. Wetlands 7:1-10.

Putnam, J.A., G.M. Furnival, and J.S. McKnight. 1960. Management and inventory of southern hardwoods. U.S. Forest Service, Agriculture Handbook 181. 102 pp.

Savage, L., D.W. Pritchett, and C.E. Depoe. 1989. Reforestation of a cleared bottomland hardwood area in northeast Louisiana. Restoration and Management Notes 7:88.

Shaw, S.P., and C.G. Fredine. 1956. Wetlands of the United States. Their extent and their value to waterfowl and other wildlife. U.S. Fish and Wildlife Service Circ. 39. 67 pp.

The Nature Conservancy. 1992. The forested wetlands of the Mississippi River, an ecosystem in crisis. The Nature Conservancy, Baton Rouge. 24 pp.

Titus, J.G., T.R. Henderson, and J.M. Teal. 1984. Sea level rise and wetlands loss in the United States. National Wetlands Newsletter 6(5):3-6.

Turner, R.E., S.W. Forsythe, and N.J. Craig. 1988. Bottomland hardwood forest land resources of the southeastern United States. Pages 13-28 *in* J.R. Clark and J. Benforado, eds. Wetlands of bottomland hardwood forests. Elsevier, Amsterdam, The Netherlands.

USDI. 1988. The impact of federal programs on wetlands, Vol. 1. The Lower Mississippi Alluvial Plain and the Prairie Pothole region. U.S. Department of the Interior, Washington, DC. 114 pp.

Vissage, J.S., P.E. Miller, and A.J. Hartsell. 1992. Forest statistics for Louisiana parishes–1991. U.S. Forest Service, Southern Forest Experiment Station Resour. Bull. SO-168. 65 pp.

Wharton, C.H., W.M. Kitchens, and T.W. Sipe. 1982. The ecology of bottomland hardwood swamps of the Southeast: a community profile. U.S. Fish and Wildlife Service. FWS/OBS-81/37. 133 pp.

Wright, A.H., and A.A. Wright. 1932. The habitats and composition of the vegetation of Okefinokee Swamp, Georgia. Ecological Monographs 2:109-232.

For further information:

B.D. Keeland
National Biological Service
Southern Science Center
700 Cajundome Blvd.
Lafayette, LA 70506

Rare Terrestrial Ecological Communities of the United States

by

Dennis H. Grossman

Kathleen Lemon Goodin

The Nature Conservancy

Federal agencies and conservation organizations have shifted their focus from managing individual species to managing entire ecosystems to protect biological diversity and conserve natural resources. Although ecological communities provide a more appropriate level of biological organization for characterizing ecosystems than individual species, the lack of a standard ecological community classification has impeded progress for ecosystem protection and management.

The Nature Conservancy and the Association of Natural Heritage Programs and Conservation Data Centers (Natural Heritage Network) have developed a framework for the classification of ecological communities. The first product from this effort is a preliminary list of rare terrestrial communities across the conterminous United States. This list was completed for the U.S. Fish and Wildlife Service (Grossman et al. 1994). This article provides a summary of the information from the Grossman et al. report, including a review of the status of information concerning rare communities of the United States, an analysis of regional patterns of rarity, and a discussion of the application of this information toward protection efforts. The use of ecological communities as a coarse conservation unit promotes conservation of the underlying ecological processes and biotic interactions that sustain the ecosystems across the landscape and ensures protection of biological diversity and rare species.

The Nature Conservancy/Natural Heritage Network Classification System

The basic goal of the community classification effort is to provide a complete listing of *all* communities that represent the variation in ecological systems. The classification hierarchy for terrestrial communities is based on the biological characteristics of existing vegetation types. These types range from early successional through climax associations and include seral stages that are maintained by natural and human-induced management and disturbance regimes.

The classification hierarchy is partitioned into terrestrial, aquatic, and subterranean "systems." The upper levels of the terrestrial system have been derived through the modification of United Nations Educational, Scientific and Cultural Organization (1973) and Driscoll et al. (1984) and refer to the physiognomic attributes (structural) of the vegetation. The two finest levels of the classification hierarchy are based on floristic analysis and are determined through the identification of diagnostic species (Westhoff and van der Maarel 1973).

Ranking System

The Nature Conservancy and Natural Heritage Network rank all elements of natural biological diversity according to their relative rarity and vulnerability to aid in ranking critical areas for conservation. The community ranks are consistent with the overall conservation ranking approach applied to all elements of natural diversity within The Nature Conservancy/Natural Heritage Network methodology (Master 1991). The communities described in this report have been ranked G1 and G2 according to The Nature Conservancy/Natural Heritage Network ranking system (*see* Table 1).

Listing Globally Rare Community Types

The development of the list of rare communities proceeded from the identification of rare communities at the state level, to the production of regional classifications of the rare state types, and finally to the generation of a consistent list of rare communities at the national level. Most state heritage programs have developed a classification system at the state level; these systems are based on available data and literature, input from experts, and field verification. State conservation ranks have been assigned to most of these communities based on the analysis of existing information.

Table 1. The Nature Conservancy/Natural Heritage Network conservation ranks for rare communities.

Rank	Definition
G1	Critically imperiled globally because of extreme rarity. Generally five or fewer occurrences or less than about 800 ha (or 2,000 acres) or because of some factor making the community particularly vulnerable to extinction.
G2	Imperiled globally because of extreme rarity. Generally 6-20 occurrences or 800-4,000 ha (2,000-10,000 acres) or because of some factor making the community very vulnerable to extinction throughout range.

A preliminary list of G1 and G2 communities was compiled by each of the Nature Conservancy's science regions (Table 2) through a detailed evaluation of all rare state types reported by the state heritage programs in each region. Each rare state type was reclassified to conform to the classification and nomenclature standards of the national framework. Rare communities that cross regional boundaries were identified and re-classified as necessary to produce the national list of G1 and G2 communities.

Table 2. The Nature Conservancy science regions.

East	Southeast	Midwest	West
Connecticut	Alabama	Illinois	Alaska*
Delaware	Arkansas	Indiana	Arizona
Maine	Florida	Iowa	California
Maryland	Georgia	Kansas	Colorado
Massachusetts	Kentucky	Michigan	Hawaii*
New Hampshire	Louisiana	Minnesota	Idaho
New Jersey	Mississippi	Missouri	Montana
New York	North Carolina	Nebraska	Nevada
Pennsylvania	Oklahoma	North Dakota	New Mexico
Rhode Island	South Carolina	Ohio	Oregon
Vermont	Tennessee	South Dakota	Utah
Virginia	Texas	Wisconsin	Washington
West Virginia			Wyoming

*Not included in this report.

Patterns of Community Rarity

Within the lower 48 United States, 371 globally rare terrestrial vegetated communities have been documented (Grossman et al. 1994). Preliminary evaluation of the proportion of G1 and G2 types indicates that these will account for about 10%-15% of all terrestrial communities. It is premature to attempt detailed national analysis and synthesis of existing data because of the preliminary nature of the overall classification and the unevenness in available community information among regions. We can, however, provide a preliminary examination of the relative proportion of rare communities in each physiognomic class within each region.

Eastern Region

Fourteen percent of the nationally rare communities occur in the eastern region (Anderson et al. 1994). Many new community types are still being identified. Most of the rare communities reported from the eastern region were forest, followed by sparse woodland and herbaceous types (Figure). The rarity of these communities is either related to the suitability of

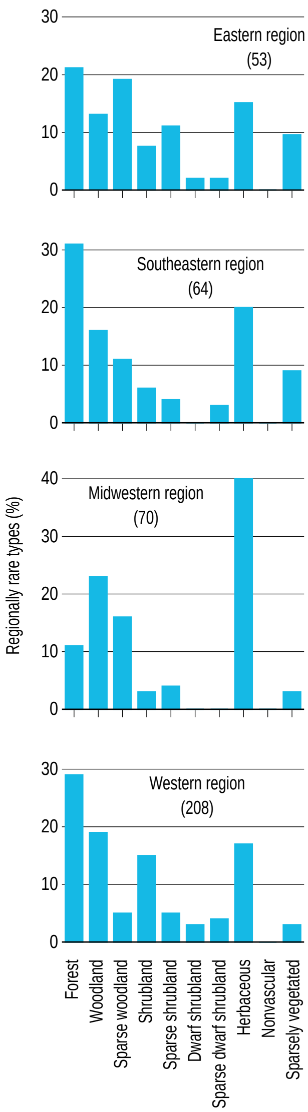


Figure. Rare terrestrial communities by region and physiognomic class. Numbers in parentheses refer to number of community types.

these habitats for land conversion or due to association with naturally rare habitats.

Southeastern Region

The rare communities occurring in the southeastern region account for about 18% of the nationally listed types (Patterson et al. 1994). This region is dominated by forest, and numerous diverse and intact rare forest associations remain. Most of the rare types fall within the forest class, followed by the herbaceous and woodland classes (Figure). The communities within the herbaceous class remain poorly defined throughout this region, but this class still represents a large portion of the rare types. We believe that the total number of types and the number of rare types within the herbaceous class will increase as additional information becomes available. Fire suppression has threatened many of the woodland types, and the actual number of rare woodland types is also presumed higher than now reported.

Midwestern Region

About 19% of the communities in the list of nationally rare communities occur in the midwestern region (Ambrose et al. 1994). Although the proportion of rare community types in this region is relatively small because of the historically coarse level of classification for this region, the magnitude of land conversion to agricultural production is staggering. The herbaceous class accounts for 40% of the rare types in the midwestern region, and the woodland and sparse woodland types make up another 38% (Figure). The rare herbaceous types reflect the remnant patches of the once-extensive prairie province. The woodland and sparse woodland communities have been heavily affected by the disruption of historical fire regimes and agricultural development.

Western Region

Most rare and threatened types identified in the national list of rare communities (about 56%) occur in the western region (Reid et al. 1994). This reflects the region's rich base of ecological and biological data and the consistent application of a detailed level of community classification, as well as a high level of natural diversity in this large region. Most rare types in the western region occur within the forest class, followed by the woodland, herbaceous, and shrubland classes (Figure). Fire suppression as a widespread forest-management practice over many decades has pushed many forest types to this status of rarity. Flood-control and water-diversion projects have similarly affected many of the forest and woodland riparian types. The rarity of the herbaceous communities across the western region is reported to be primarily the result of overgrazing and, to a lesser degree, direct agricultural conversion.

No regions reported rare communities in the nonvascular class and few were documented within the dwarf shrubland, sparse dwarf shrubland, and sparsely vegetated class. This result may not reflect the actual status of rare communities in these classes throughout the United States but rather the shortage of available information.

Knowledge Gaps

The rare communities for several states are not documented at this time. This does not mean there are no rare communities in those states but instead indicates the lack of available information. These knowledge gaps were documented during the listing of rare communities. Information gaps at the state level included incomplete or overly coarse classifications, lack of conservation ranks, and the lack of time and support for field verification. Those states where significant work remains are listed in Table 3.

Many communities recognized as rare still require additional work to complete their classification, ranking, and description process. The number of communities in this group presently totals 482.

Limitations

The number of rare communities varies among regions, reflecting unevenness in the quantity and quality of community information among the regions, along with varying levels of classification development and subtle differences in procedures for conservation ranking. To some extent, the regional variation also reflects the actual differences in ecological and biological diversity, the results of landscape fragmentation, and land-cover conversion.

While rarity of ecological communities is critical information for biodiversity conservation and management, appropriate protection and management activities should be determined for each individual rare community. Communities assigned a rank of G1 or G2 are very rare and occur generally within a restricted range of environmental conditions. These ranks do not reflect why a particular community is rare; such analysis, however, is fundamental to setting guidelines for protection and long-term management.

Some communities are naturally rare because of their association with an uncommon habitat. For example, the rarity of the inland salt marsh association (*Scirpus maritimus-Atriplex patula-Eleocharis parvula* herbaceous vegetation) has been documented, but the community is not noticeably rarer than it was 100 years ago.

This kind of community occurs on saturated saline mud flats associated with rare inland salt springs in Illinois, Michigan, and New York (Ambrose et al. 1994; Anderson et al. 1994). The environmental characteristics that support this biological association have similarly restricted the use of this habitat for agricultural production and most other types of land conversion, although some communities have been degraded by salt-extraction operations. Though this community is unlikely to disappear because of human-induced disturbance, individual communities should be protected from degradation due to incompatible land use.

In contrast, the mesic tall-grass prairie association (*Andropogon gerardii-Sorghastrum nutans-Sporobolus heterolepis* [*Liatrus* spp.-*Silphium laciniatum*] herbaceous vegetation) in the Midwest was common a century ago but is very rare today. The existing occurrences of this association type represent remnants of a community whose acreage has rapidly declined because of the value of its habitat for agricultural production (Ambrose et al. 1994). It has also suffered from the large-scale alteration of historical fire regimes. Rare communities such as this are quite threatened and require immediate protection and management.

Future

The list of nationally rare communities will help ensure their recognition and set priorities for their protection, an important step for conservation. Even if the list of rare communities were complete, however, it would still be insufficient to conserve and manage biological diversity. A comprehensive national conservation strategy for all communities, including common ones, is necessary to protect and manage the full spectrum of biological diversity and ecological systems.

The development of a standard community classification system has dramatically increased our capability to make better informed conservation and ecosystem management decisions at multiple geographic scales. The synthesis of existing community data on nationally rare types has identified the strengths and weaknesses of the existing information base, information that will help us decide how to accumulate and analyze data to fill critical gaps in our knowledge.

The acquisition and management of the ecological and biological data needed to complete the national classification represent a major challenge. The success of many ecosystem management initiatives will depend upon this information. A concerted cooperative effort is necessary to conserve and manage our biological and ecological resources.

Table 3. Knowledge gaps related to state community classification, ranking, and inventory.

State classification not completed	No state ranks	Coarse state classification	Limited inventory
Maryland	California	California	Alabama
Rhode Island	Georgia	Florida	Georgia
South Dakota	Maryland	North Dakota	North Dakota
West Virginia	Rhode Island	Texas	South Dakota
	South Dakota	Virginia	
	Tennessee		
	West Virginia		

References

Ambrose, D., J. Drake, and D. Faber-Langendoen. 1994. Rare plant communities of the United States: midwestern region. *In* D.H. Grossman, K.L. Goodin, and C.L. Reuss, eds. Rare plant communities of the conterminous United States: an initial survey. The Nature Conservancy, Arlington, VA. In press.

Anderson, M.G., L.A. Sneddon, and K.J. Metzler. 1994. Rare plant communities of the United States: eastern region. *In* D.H. Grossman, K.L. Goodin, and C.L. Reuss, eds. Rare plant communities of the conterminous United States: an initial survey. The Nature Conservancy, Arlington, VA. In press.

Driscoll, R.S., D.L. Merkel, R.L. Radloff, D.E. Snyder, and J.S. Hagihara. 1984. An ecological land classification framework for the United States. U.S. Forest Service Miscellaneous Publ. 1439.

Grossman, D.H., K.L. Goodin, and C.L. Reuss, eds. 1994. Rare plant communities of the conterminous United States: an initial survey. The Nature Conservancy, Arlington, VA. In press.

Master, L. 1991. Assessing threats and setting priorities for conservation. Conservation Biology 5(4):559-563.

Patterson, K.D, D.J. Allard, and S. Landaal. 1994. Rare plant communities of the United States: southeastern region. *In* D.H. Grossman, K.L. Goodin, and C.L. Reuss, eds. Rare plant communities of the conterminous United States: an initial survey. The Nature Conservancy, Arlington, VA. In press.

Reid, M., L. Engelking, and P. Bourgeron. 1994. Rare plant communities of the United States: western region. *In* D.H. Grossman, K.L. Goodin, and C.L. Reuss, eds. Rare plant communities of the conterminous United States: an initial survey. The Nature Conservancy, Arlington, VA. In press.

United Nations Educational, Scientific and Cultural Organization (UNESCO). 1973. International classification and mapping of vegetation. Series 6, Ecology and conservation. United Nations Educational, Scientific and Cultural Organization, Paris. 93 pp.

Westhoff, V., and E. van der Maarel. 1973. The Braun-Blanquet approach. Pages 626-828 *in* R.H. Whittaker, ed. Handbook of vegetation science. Part V. Ordination and classification of communities. Junk, The Hague.

For further information:

Dennis H. Grossman
The Nature Conservancy
1815 N. Lynn St.
Arlington, VA 22209

Altered Fire Regimes Within Fire-adapted Ecosystems

by
Gardner W. Ferry
Robert G. Clark
Roy E. Montgomery
Bureau of Land Management
Robert W. Mutch
U.S. Forest Service
Willard P. Leenhouts
U.S. Fish and Wildlife Service
G. Thomas Zimmerman
National Park Service

Fires ignited by people or through natural causes have interacted over evolutionary time with ecosystems, exerting a significant influence on numerous ecosystem functions (Pyne 1982). Fire recycles nutrients, reduces biomass, influences insect and disease populations, and is the principal change agent affecting vegetative structure, composition, and biological diversity. As humans alter fire frequency and intensity, many plant and animal communities are experiencing a loss of species diversity, site degradation, and increases in the size and severity of wildfires. This article examines the role fire plays in the ecological process around which most North American ecosystems evolved.

The five plant communities selected for study were the sagebrush steppe, juniper woodlands, ponderosa pine forest, lodgepole pine forest, and the southern pineland (Fig. 1). Status and trends of altered fire regimes in fire-adapted ecosystems highlight the role that fire plays in wildland stewardship. Fire regimes are considered as the total pattern of fires over time that is characteristic of a region or ecosystem (Kilgore and Heinselman 1990).

Sagebrush-grass Plant Communities

Greater frequency of fire has seriously affected the sagebrush steppe during the last 50 years (Table). One such community, the semiarid intermountain sagebrush (*Artemisia* species) steppe, encompasses about 45 million ha (112 million acres). After repeated fires, non-native European annual grasses such as cheatgrass (*Bromus tectorum*) and medusahead (*Taeniatherum caput-medusae*) now dominate the sagebrush steppe (West and Hassan 1985). It is unclear whether cheatgrass invasion, heavy grazing pressure, or shorter fire return intervals initiated the replacement of perennial grasses and shrubs by the non-native annual grasses. It is clear, however, that wildfires aid in replacing native grasses with cheatgrass, as well as causing the loss of the native shrub component (Whisenant 1990). Inventories show that cheatgrass is dominant on about 6.8 million ha (17 million acres) of the sagebrush steppe and that it could expand into an additional 25 million ha (62 million acres) in the sagebrush steppe and the Great Basin sagebrush type (Pellant and Hall 1994).

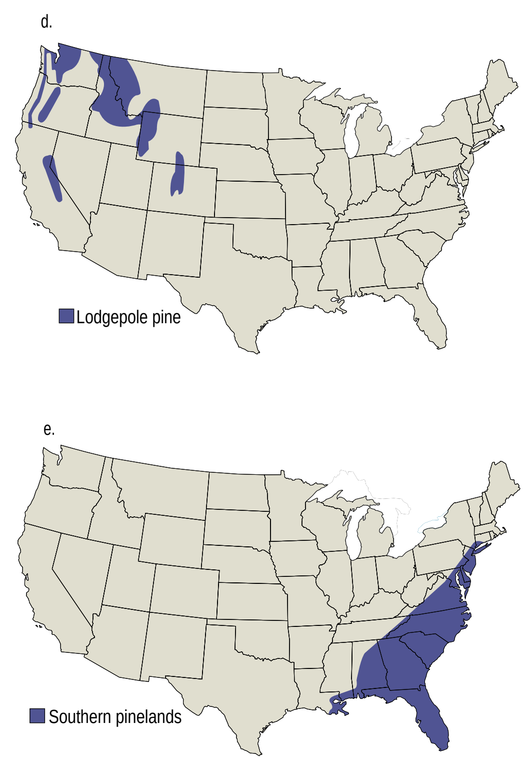


Fig. 1. Range of: a —sagebrush steppe; b — juniper woodlands; c — ponderosa pine; d — lodgepole pine; and e — southern pineland communities in the United States.

Table. Increase in the number of wildfires and area burned on sagebrush steppe in Idaho (data from the Bureau of Land Management, Idaho State Office, Boise).

	1950-59	1960-69	1970-79	1980-89
Number of wildfires	Data incomplete	1,344	1,406	2,334
Area burned (ha)	751,000	663,000	900,000	1,316,000

Western Juniper Woodlands

Juniper woodlands occupy 17 million ha (42 million acres) in the Intermountain region (West 1988). Juniper species common to this region are western juniper (*Juniperus occidentalis*), Utah juniper (*J. osteosperma*), single-seeded juniper (*J. monosperma*), and Rocky Mountain juniper (*J. scopulorum*). Presettlement juniper woodlands were usually savanna-like or confined to rocky outcrops not typically susceptible to fire (Nichol 1937).

Juniper woodlands began increasing in both density and distribution in the late 1800's (R.F. Miller, Eastern Oregon Agricultural Research Center, unpublished data; Fig. 2) because of climate, grazing, and lack of fire (Miller and Waigand 1994). Warm and wet climate conditions then were ideal for juniper and grass seed production. Fire frequency had decreased because the grazing of domestic livestock had greatly reduced the grasses and shrubs that provided fuel, and relocation of Native Americans eliminated an important source of ignition. Continued grazing and 50 years of attempted fire exclusion have allowed juniper expansion to go unchecked.

Ponderosa Pine Forest

Decreases in fire frequency are also seriously affecting ponderosa pine (*Pinus ponderosa*) forests, a common component on about 16 million ha (40 million acres) in the western United States. Historically, the ponderosa pine ecosystem had frequent, low-intensity, surface fires that perpetuated park-like stands with grassy undergrowth (Barrett 1980). For six decades, humans attempted to exclude fire on these sites (OTA 1993). Fifty years ago, Weaver (1943) stated that complete prevention of forest fires in the ponderosa pine region had undesirable ecological effects and that already-deplorable conditions were becoming increasingly serious. Today, many ponderosa pine forests are overstocked, plagued by epidemics of insects and diseases, and subject to severe stand-destroying fires (Mutch et al. 1993).

Lodgepole Pine Forest

Like ponderosa pine forests, lodgepole pine (*Pinus contorta*) forests are experiencing a change in structure, distribution, and functioning of natural processes because of fire exclusion and increases in disease. Wildfire may be the most important factor responsible for establishment of existing stands (Wellner 1970). Historical stand-age distributions in lodgepole pine forests indicated an abundance of younger age classes resulting from periodic fires. Fire exclusion, by precluding the initiation of new stands, is responsible for a marked change in distribution of age classes in these forests (Fig. 3).

Dwarf mistletoe (*Arceuthobium americanum*), the primary disease of lodgepole pine, also has a profound effect on forest structure and function, although it occurs slowly. Data show that chronic increases of dwarf mistletoe are partly due to the exclusion of fire (Zimmerman and Laven 1984) because fire is the natural control of dwarf mistletoe and has played a major role in the distribution and abundance of current populations and infection intensities (Alexander and Hawksworth 1975). As the frequency and extent of fire have decreased in lodgepole pine stands over the last 200 years, dwarf mistletoe infection intensity and distribution are clearly increasing (Zimmerman and Laven 1984).

Southern Pinelands

In contrast to the juniper, ponderosa pine, and lodgepole pine communities, fire frequencies have not drastically decreased in the 78 million ha (193 million acres) of southern pinelands. These pinelands are composed of diverse plant communities associated with longleaf *(Pinus palustris*), slash (*P. elliotti*), loblolly (*P. taeda*), and shortleaf pines (*P. echinata*). Fire has continued on an altered basis as an ecological process in much of the southern pinelands; historically, fire burned 10%-30% of the forest annually (Wright and Bailey 1982); the southern culture never effectively excluded fire from its pinelands (Pyne 1982), although human-ignited fires have partially replaced natural fires. Consequently, the amount of fire has been reduced and the season of burns has changed from predominately growing-season to dormant-season (fall or winter) fires (Robbins and Myers 1992). Altering the burning season and frequency has significantly affected southern pineland community structure, composition, and biological diversity (Fig. 4).

Implications

The role of fire becomes more complex as it interacts with land management. Maintaining interactions between disturbance processes and ecosystem functions is emphasized in ecosystem management. It is vital for mangers to

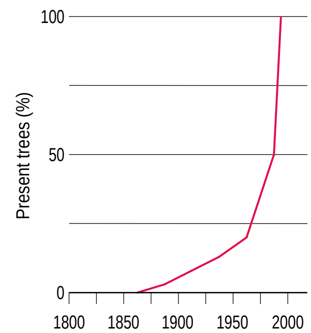


Fig. 2. Cumulative establishment of western juniper on Steens Mountain, Oregon (adapted from R.F. Miller, Eastern Oregon Agricultural Research Center, unpublished data).

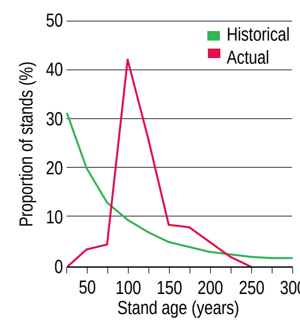


Fig. 3. Historical and actual age-class distributions of lodgepole pine forest.

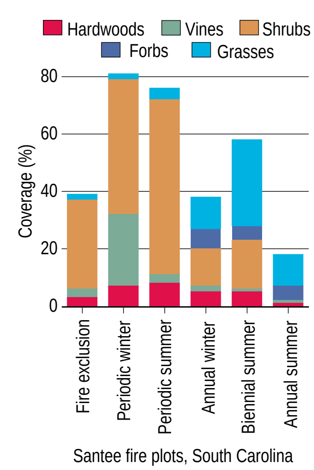


Fig. 4. Understory plant crown coverage after 30 years of burning (Waldrop et al. 1987).

recognize how society influences fire as an ecological process. In addition, managers must uniformly use information on fire history and fire effects to sustain the health of ecosystems that are both fire-adapted and fire-dependent. Managers must balance the suppression program with a program of prescribed fire applied on a landscape scale if we are to meet our stewardship responsibilities.

For further information:

Gardner W. Ferry
Bureau of Land Management
Division of Fire and Aviation
Policy and Management
3905 S. Vista Ave.
Boise, ID 83705

References

Alexander, M.E., and F.G. Hawksworth. 1975. Wildland fires and dwarf mistletoe: a literature review of ecology and prescribed burning. U.S. Forest Service Gen. Tech. Rep. RM-14. Rocky Mountain Forest Range Experiment Station, Fort Collins, CO. 12 pp.

Barrett, S.W. 1980. Indians and fire. Western wildlands 6(3):17-21.

Kilgore, B.M., and M.L. Heinselman. 1990. Pages 297-335 *in* J.C. Hendee, G.M. Stankey, and R.C. Lucas, eds. Fire in wilderness ecosystems. Wilderness management. 2nd ed. North American Press, Golden, CO.

Miller, R.F., and P.E. Waigand. 1994. Holocene changes in semi-arid pinyon-juniper woodlands: response to climate, fire and human activities in the Great Basin. Biological Science 44 (7). In press.

Mutch, R.W., S.F. Arno, J.K. Brown, C.E. Carlson, R.D. Ottmar, and J.L. Peterson. 1993. Forest health in the Blue Mountains: a management strategy for fire-adapted ecosystems. U.S. Forest Service Gen. Tech. Rep. PNW-310. 14 pp.

Nichol, A.A. 1937. The natural vegetation of Arizona. University of Arizona Tech. Bull. 68.

OTA. 1993. Preparing for an uncertain climate. 2 vols. U.S. Congress, Office of Technology Assessment. Washington, DC.

Pellant, M., and C. Hall. 1994. Distribution of two exotic grasses on public lands in the Great Basin: status in 1992. *In* Proceedings of a Symposium on Ecology, Management and Restoration of Intermountain Annual Rangelands. Intermountain Forest and Range Experiment Station, Ogden, UT.

Pyne, S.J. 1982. Fire in America: a cultural history of wildland and rural fire. Princeton University Press, Princeton, NJ. 653 pp.

Robbins, L.E., and R.L. Myers. 1992. Seasonal effects of prescribed burning in Florida: a review. Tall Timbers Research, Inc. Miscellaneous Publ. 8. 96 pp.

Waldrop, T.A., D.H. Van Lear, R.T. Lloyd, and W.R. Harms. 1987. Long-term studies of prescribed burning of loblolly pine forests of the southeastern Coastal Plain. U.S. Forest Service Southeastern Forest Experiment Station Gen. Tech. Rep. SE-45. Asheville, NC. 23 pp.

Weaver, H. 1943. Fire as an ecological and silvicultural factor in the ponderosa-pine region of the Pacific Slope. Journal of Forestry 41:7-14.

Wellner, C.A. 1970. Fire history in the northern Rocky Mountains. Pages 41-64 *in* The role of fire in the Intermountain West. Proceedings of the Intermountain Fire Research Council Symposium, Missoula, MT.

West, N.E. 1988. Intermountain deserts, shrub steppes, and woodlands. Pages 209-230 *in* M.B. Barbour and W.D. Billings, eds. North American terrestrial vegetation. Cambridge University Press, New York.

West, N.E., and M.A. Hassan. 1985. Recovery of sagebrush-grass vegetation following wildlife. Journal of Range Management 38:131-134.

Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. Pages 4-10 *in* E.D. McArthur, E.M. Romney, S.D. Smith, and P.T. Tueller, eds. Proceedings of a Symposium on Cheatgrass Invasion, Shrub Die-off, and Other Aspects of Shrub Biology and Management. U.S. Forest Service Gen. Tech. Rep. INT-276. Intermountain Forest and Range Experiment Station, Ogden, UT.

Wright, H.A., and S.W. Bailey. 1982. Fire ecology: United States and southern Canada. John Wiley and Sons, New York. 501 pp.

Zimmerman, G.T., and R.D. Laven. 1984. Ecological interrelationships of dwarf mistletoe and fire in lodgepole pine forests. Pages 123-131 *in* F.G. Hawksworth and R.F. Scharpf, technical coordinators. Biology of dwarf mistletoes: Proceedings of the Symposium. U.S. Forest Service Gen. Tech. Rep. RM-111. Rocky Mountain Forest Range Experiment Station, Fort Collins, CO. 131 pp.

Vegetation Change in National Parks

by
Kenneth Cole
National Biological Service

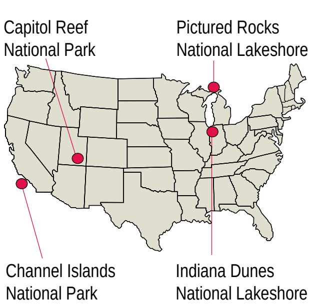


Fig. 1. Four national park units studied.

Natural ecosystems are always changing, but recent changes in the United States have been startlingly rapid, driven by 200 years of disturbances accompanying settlement by an industrialized society. Logging, grazing, land clearing, increased or decreased frequency of fire, hunting of predators, and other changes have affected even the most remote corners of the continent. Recent trends can be better understood by comparisons with more natural past trends of change, which can be reconstructed from fossil records. Conditions before widespread impacts in a region are termed "presettlement"; conditions after the impacts are "postsettlement."

Fossil plant materials from the last few thousand years are used to study past changes in many natural areas. Pollen buried in wetlands, for example, can reveal past changes in vegetation (Faegri and Iversen 1989), and larger fossil plant parts can be studied in deserts where the fossilized plant collections of packrats, called packrat middens, have been preserved (Betancourt et al. 1990).

This article summarizes the rates of vegetation change in four national park areas over the last 5,000 years as reconstructed from fossil pollen and packrat middens. These four national park areas from different ecological regions (Fig. 1) demonstrate the flexibility of these paleoecological techniques and display similar results.

Northern Indiana Prairie

A 4,500-year history of vegetation change was collected from Howes Prairie Marsh, a small marsh surrounded by prairie and oak savanna in the Indiana Dunes National Lakeshore near the southern tip of Lake Michigan. Only 40 km (25 mi) from Chicago, this area has been affected by numerous impacts from settlements but still supports comparably pristine tall-grass prairie vegetation as well as the endangered Karner blue butterfly

Courtesy W. L. Loope

Coring a small pond at Pictured Rocks National Lakeshore.

(*Lycaeides melissa samuelis*). Although this site has experienced more disturbances than any of the others described here, it is a most valuable site because of its many species (Wilhelm 1990) and its tall-grass prairie vegetation that has been nearly eliminated elsewhere.

The many historical impacts to this area make it a good source for studying past changes. Past amounts of pollen from the primary plant taxa are illustrated in Fig. 2. Many changes occurred before settlement, but more rapid changes occurred in the last 140 years.

Past rates of change in vegetation can be measured by summing the relative change in each plant type between successive samples and then dividing by the number of years between samples. The technique is similar to that used by Jacobson and Grimm (1986).

Although these changes had been occurring throughout the last 4,500 years, the postsettle-

Courtesy K. Cole, NBS

Experimental prairie fire at Indiana Dunes National Lakeshore.

ment rates of change are at least 10 times greater than the presettlement rates of change (Fig. 3a). The rates of change have been declining over the last 50 years, but are still far greater than the presettlement rates of change.

Northern Michigan Forest

A similar analysis was carried out on pollen from a small bog (unofficial name: 12-Mile Bog) surrounded by pine forest along the southern shore of Lake Superior (Fig. 3b). This site, within Pictured Rocks National Lakeshore, was more severely affected by logging and slash burning in the 1890's than by the periodic wildfires that characterized this forest earlier, but it has been protected for the last 80 years. The magnitude of change caused by the crude logging and slash burning of the logging era was far greater than any recorded during the 2,500 years since Lake Superior receded to create the forest of white and red pine (*Pinus strobus* and *P. resinosa*).

As in the Indiana Dunes, rates of change have declined during the last 60 years, and the forest is now very similar to the forest of 2,000 years ago. Thus, although the area is still changing at a rate far above normal, it has begun to recover through protection.

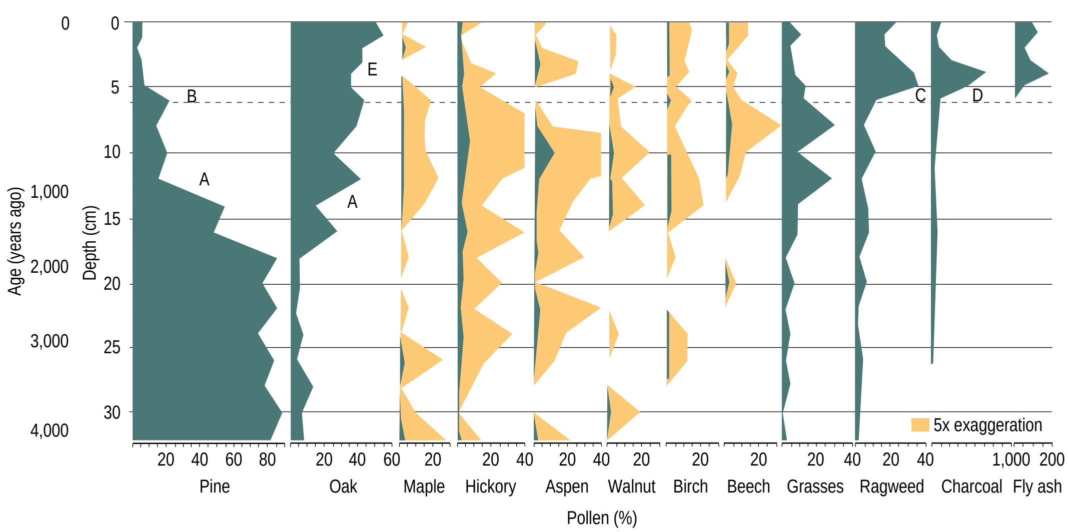


Fig. 2. Selected taxa of fossil pollen recorded from Howes Prairie in the Indiana Dunes. The percentage of total pollen representing each plant is graphed along a vertical time axis. The dotted line shows the sedimentary horizon representing settlement of the region (about A.D. 1850). Major changes indicated by letters: A — decline in pine and increases in oak and grasses due to plant succession and climate change; B — decline in pine due to logging of white pine in mid-1800's; C — increase in ragweed from cleared farm fields and increase in fly ash from the development of the steel industry in Gary, IN (22 km away) in the late 1800's (Cole et al. 1990); D — increase in charcoal particles as steam railroads ignite nearby drained wetlands and subsequent decline in charcoal as steam power ends and wildfires are controlled; and E — decline in oak as frequent fires top-kill mature trees followed by increase in oak as periodic prairie fires are extinguished.

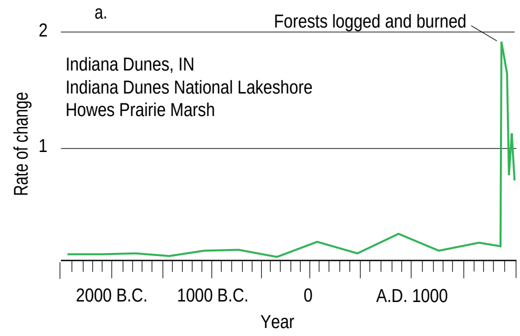


Fig. 3a. Rates of change from a tall-grass prairie and oak savanna in the Indiana Dunes, IN, based on pollen from tree species.

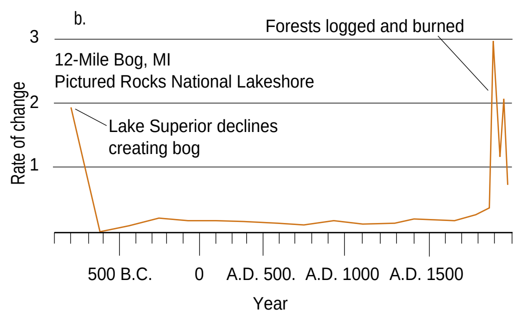


Fig. 3b. Rates of change from a boreal pine forest at Pictured Rocks, MI, based on pollen from tree species.

California Coastal Sage Scrub

Fossil pollen was analyzed from an estuary on Santa Rosa Island off the coast of southern California (Cole and Liu 1994). The semi-arid landscape around the estuary is covered with coastal sage scrub, chaparral, and grassland. This site, within Channel Islands National Park, is one of the least affected areas in this region of rapidly expanding urbanization, although the island's native plants and animals were not well adapted to withstand the grazing of the large animals introduced with the ranching era of the 1800's. This island, which had no native large herbivores, became populated with thousands of sheep, cattle, horses, goats, pigs, deer, and elk. The National Park Service is removing many of the large herbivores, although most of the island remains an active cattle ranch.

All pollen types from 33 samples spread over the last 4,600 years were analyzed. The rates of change in the pollen were similar to those observed from the other sites (Fig. 3c).

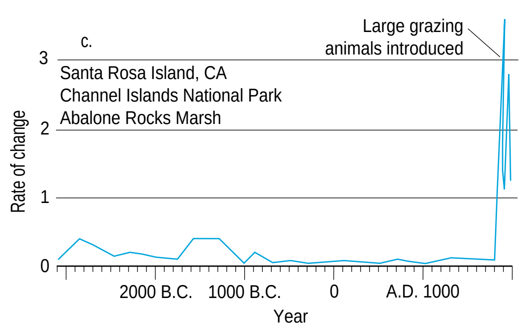


Fig. 3c. Rates of change from coastal sage scrub on Santa Rosa Island, CA, based on pollen from an estuary.

Southern Utah Desert

Because fossil pollen is usually preserved in accumulating sediments of wetlands, different paleoecological techniques are necessary in arid areas. In western North America, fossil deposits left by packrats (*Neotoma* spp.) have proven a useful source of paleoecological data (Betancourt et al. 1990). Past desert vegetation can be reconstructed by analyzing bits of leaves, twigs, and seeds collected by these small rodents and incorporated into debris piles in rock shelters or caves. These debris piles can be collected, analyzed, and radiocarbon dated.

The vegetation history of a remote portion of Capitol Reef National Park (Hartnett Draw) was reconstructed through the analysis of eight packrat middens ranging in age from 0 to 5,450 years (Cole 1995). The vegetation remained fairly stable throughout this period until the last few hundred years. The most recent deposits contain many plants associated with overgrazed areas such as whitebark rabbitbrush (*Chrysothamnus visidiflorus*), snakeweed (*Gutterezia sarothrae*), and greasewood (*Sarcobatus vermiculatus*), which were not recorded at the site before settlement.

Conversely, other plants that are extremely palatable to grazing animals were present throughout the last 5,450 years, only to disappear since settlement. Plant species preferred by sheep and cattle, such as winterfat (*Ceratoides lanata*) and rice grass (*Stipa hymenoides*), disappeared entirely, while many other palatable plant species declined in abundance after 5,000 years of comparative stability.

The past rates of vegetation change for this site were calculated in a manner similar to the fossil pollen records (Fig. 3d). Although the rate of change calculation is less precise than the fossil pollen records because there were fewer samples, the results show a similar pattern. The rate of vegetation change is highest between the two most recent records.

Although this area is still grazed by cattle today through grazing leases to private ranchers from the National Park Service, much of the

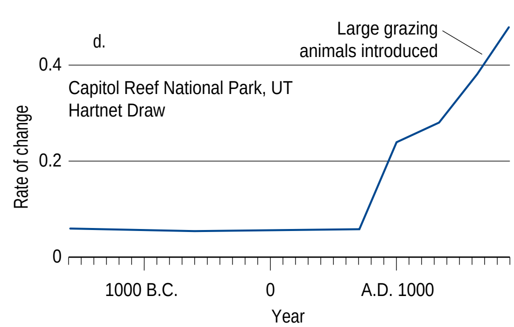


Fig. 3d. Rates of change from desert vegetation at Capitol Reef, UT, based on plant fossils in packrat middens.

severe damage was probably done by intensive sheep grazing during the late 1800's when the entire region was negatively affected by open-land sheep ranching. We cannot yet demonstrate whether the grazing effects are continuing or if the site is improving, although reinvasion of palatable species is unlikely in the face of even light grazing. Severe overgrazing is required to eliminate abundant palatable species, but once they are eliminated, even light grazing can prevent their restoration.

Implications

Wise land management decisions are more likely to be made if land managers understand a site and are able to place the status quo into a historical perspective. Because most of the damage to these four sites occurred before the 20th century, land managers might not even be aware of the tremendous changes that have occurred were it not for these fossil records. Since the ultimate goal for the management of many areas is to mitigate settlement impacts and return the land to its presettlement status, detailed knowledge of the effects of settlement is imperative.

In all study areas, postsettlement rates of change were at least 10 times higher than the presettlement rates of change. Thus, the changes now being observed in even remote natural ecosystems are unlike former natural changes. Some areas are continuing to change at rapid rates, while other areas, which have not been disturbed as recently, are stabilizing. The climatic warming projected for the next 50 years may exacerbate these ongoing changes, but will be only one of many variables operating in the unplanned redesign of our natural ecosystems. Land managers need to understand the nature and severity of the effects of settlement to return the land to its presettlement condition.

Courtesy K. L. Cole, NBS

Twenty-eight thousand-year-old packrat midden at Capitol Reef National Park; orange notebook is 6 inches high.

References

Betancourt, J.L., T.R. Van Devender, and P.S. Martin, eds. 1990. Packrat middens: the last 40,000 years of biotic change. The University of Arizona Press, Tucson. 467 pp.

Cole, K.L. 1995. A survey of the fossil packrat middens and reconstruction of the pregrazing vegetation of Capitol Reef National Park. National Park Service Res. Rep. In press.

Cole, K.L., D.R. Engstrom, R.P. Futyma, and R. Stottlemyer. 1990. Past atmospheric deposition of metals in northern Indiana measured in a peat core from Cowles Bog. Environmental Science and Technology 24:543-549.

Cole, K.L., and G. Liu. 1994. Holocene paleoecology of an estuary on Santa Rosa Island, California. Quaternary Res. 41:326-335

Faegri, K., and J. Iversen. 1989. Textbook of pollen analysis. Wiley and Sons, New York. 328 pp.

Jacobson, G.L., Jr., and E.C. Grimm. 1986. A numerical analysis of Holocene forest and prairie vegetation in central Minnesota. Ecology 67:958-966.

Wilhelm, G.S. 1990. Special vegetation of the Indiana Dunes National Lakeshore. Indiana Dunes National Lakeshore Research Program, Rep. 90-02, Porter, IN.

For further information:

Kenneth L. Cole
National Biological Service
Cooperative Park Studies Unit
University of Minnesota
Department of Forest Resources
115 Green Hall
St. Paul, MN 55108

Air Pollution Effects on Forest Ecosystems in North America

by
David L. Peterson
National Biological Service

Air pollution poses a threat to forest ecosystems in several regions of North America. Although there are isolated impacts downwind from point sources such as industrial operations, the major impacts are from regional-scale exposure to ambient ozone and acid precipitation. Acidic deposition (including sulfur and nitrogen deposition) is fairly high in the northeastern United States and southeastern Canada, although symptomatic injury and changes in forest growth have not been clearly linked to a particular pollutant. Recent evidence, however, indicates that long-term inputs of acid precipitation may be altering the chemical equilibrium of some soils, which could result in a nutritional imbalance in trees.

Elevated levels of ozone have resulted in stress in several forest ecosystems of North America: (1) those adjacent to Mexico City (extensive mortality and reduced growth); (2) those in mountains of the Los Angeles Basin in California (mortality and growth reductions); (3) those in the central and southern Sierra Nevada (some reduced growth and widespread symptomatic injury); (4) those in the Rincon Mountains of Arizona (some symptomatic injury); and (5) those in the Great Smoky Mountains (some symptomatic injury).

Recent growth reductions and changes in forest health have been reported for several locations in North America although the role of air pollution in these "declines" must be evaluated in the context of a stress complex that includes climate, stand dynamics, and site factors. Although some lichens are known to be sensitive to air pollution, there is relatively little information on the effects of air pollutants on forest species other than trees. Only if monitoring programs are implemented soon will it be possible to detect how long-term pollutant deposition affects forest health and productivity.

For further information:

David L. Peterson
National Biological Service
Cooperative Park Studies Unit
University of Washington
AR-10
Seattle, WA 98195

Air Quality in the National Park System

by
Bruce Nash
Kathy Tonnessen
National Biological Service
David Joseph
Miguel Flores
National Park Service

The National Park Service (NPS) Organic Act and the federal Clean Air Act require the NPS to protect the natural resources of the lands it manages from the adverse effects of air pollution. The NPS established a program to measure ozone—the air pollutant that is most widespread and injurious to human health and vegetation—at more than 40 monitoring sites within the National Park System.

NPS sites in southern and central California, the Great Lakes region, and the northeast and east-central United States generally record the highest ozone concentrations in the NPS network. Ozone levels exhibit strong seasonal and diurnal temporal trends, and year-to-year variation may be significant (Figure).

The 1987-91 NPS trend in maximum ozone concentrations closely resembles the corresponding trend for the entire nation. The National Biological Service (NBS) National Air Quality Research Program sponsors surveys to document ozone injury to vegetation. Current monitoring concentrates on sensitive indicator plants, including hardwoods and some herbaceous plants in the eastern United States and conifers in the West. Controlled fumigation studies have confirmed that elevated ambient ozone levels can cause decreased growth rate, decreased biomass, and premature defoliation in sensitive species such as black cherry (*Prunus serotina*), American sycamore (*Platanus occidentalis*), yellow-poplar (*Liriodendron tulipifera*), and ponderosa pine (*Pinus ponderosa*).

Acid deposition is a regional pollutant monitored at 30 NPS units as part of the National Atmospheric Deposition Program (NADP). Ten years of wet deposition (e.g., pollutants that may come down in rain or snow) data permit researchers to estimate loading of nitrate, sulfate, and hydrogen ions to sensitive ecosystems. NADP data show that the NPS units with the greatest acid loading are in the eastern United States, with Acadia, Cape Cod, Shenandoah, and Great Smoky Mountains national parks showing annual average wet deposition pH values of 4.4-4.6. These values do not reflect the contributions of cloudwater, fogwater, and dry deposition (e.g., particles and gases) to the total loading of acids, nitrogen, and sulfate to ecosystems that are sensitive to acidic inputs. NADP samplers do not measure snow efficiently and do not account for the effect of snowmelt pulses on sensitive alpine lakes and streams in the spring at high-elevation sites in the Sierra Nevada, the Cascades, and the Rocky Mountains. Research at Shenandoah National Park has shown that deposition-driven episodes in streams can result in pH levels low enough to affect native fish species.

Any assessment of ecosystem health must consider the composition of the atmosphere and its interactions with the biological and physical components of the ecosystem under investigation. Although we have some understanding of the biological effects of air pollution, more studies are necessary to ensure the protection of our natural resources.

For further information:

Bruce L. Nash
National Biological Service-AIR
National Air Quality Research Program
PO Box 25287
Denver, CO 80225

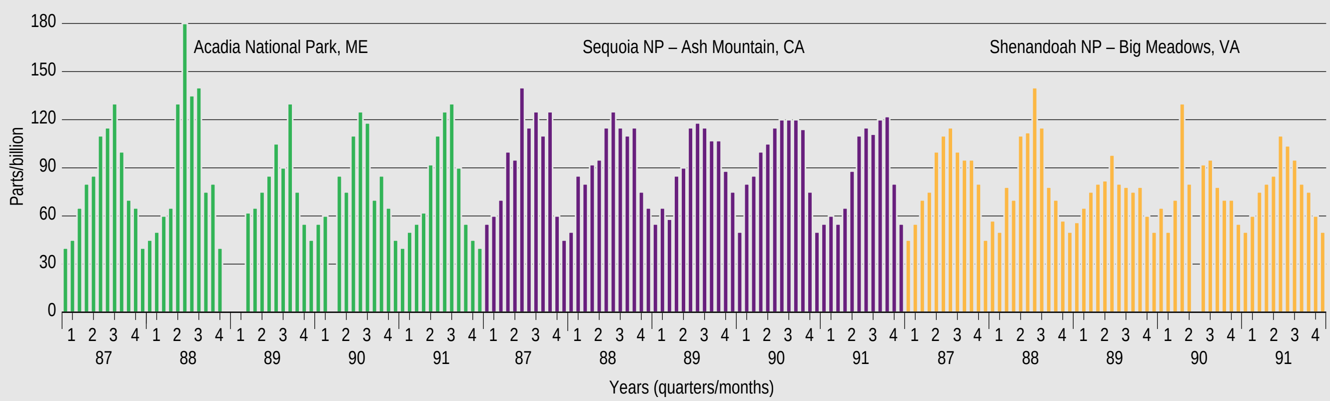


Figure. Highest daily 1-h ozone concentration per month.

Whitebark Pine: Ecosystem in Peril

by
Katherine C. Kendall
National Biological Service

Whitebark pine (*Pinus albicaulis*) is well-suited to harsh conditions and populates high-elevation forests in the northern Rocky Mountain, North Cascade, and Sierra Nevada ranges (Fig. 1a). Whitebark pine seeds are unusually large, highly nutritious, and are a preferred food for grizzly bears (*Ursus arctos*) and many other animals (Kendall and Arno 1990). These pine trees (Fig. 2) are adapted to cold, dry sites and pioneer burns and other disturbed areas. At timberline, they grow under conditions tolerated by no other tree species, thus playing an important role in snow accumulation and persistence. Because few roads occur in whitebark pine ecosystems and because the tree's wood is of little commercial interest, information on the drastic decline of this picturesque tree has only recently emerged.

Threats

Whitebark pine is threatened by an introduced disease and fire suppression. In its northern range, many whitebark pine stands have declined by more than 90% (Fig.1a). The most serious threat to the tree is from white pine blister rust (*Cronartium ribicola*), a non-native fungus that has defied control. Fewer than one whitebark pine tree in 10,000 is rust-resistant. Mortality has been rapid in areas like western Montana, where 42% of whitebark pine trees have died from the disease in the last 20 years; 89% of the remaining trees are infected with rust (Fig. 3; Keane and Arno 1993). Although drier conditions have slowed the spread of blister rust in whitebark pine's southern range, infection rates there are increasing and large die-offs are eventually expected to occur (Fig. 1b).

Before fire suppression, whitebark pine stands burned every 50-300 years. Under current management, they will burn at 3,000-year intervals. Without fire, seral whitebark pine trees are replaced by shade-tolerant conifers and become more vulnerable to insects and disease.

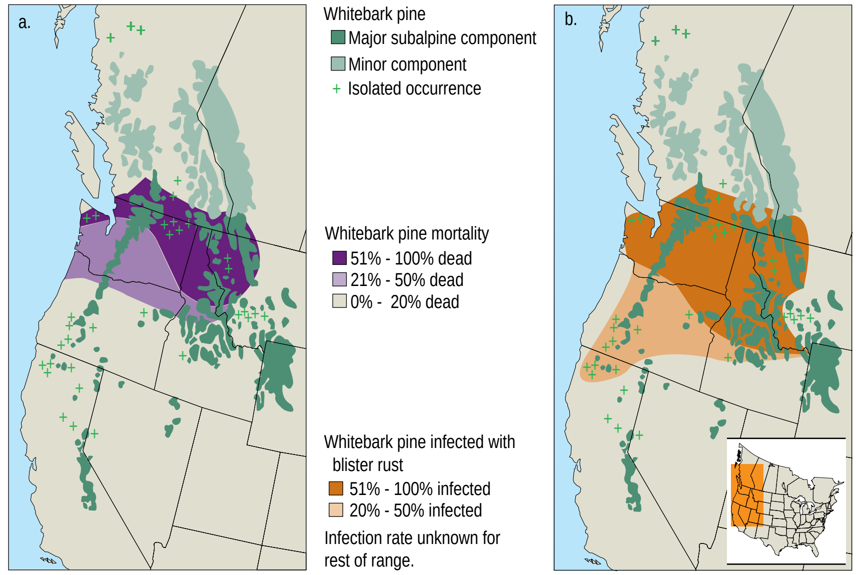


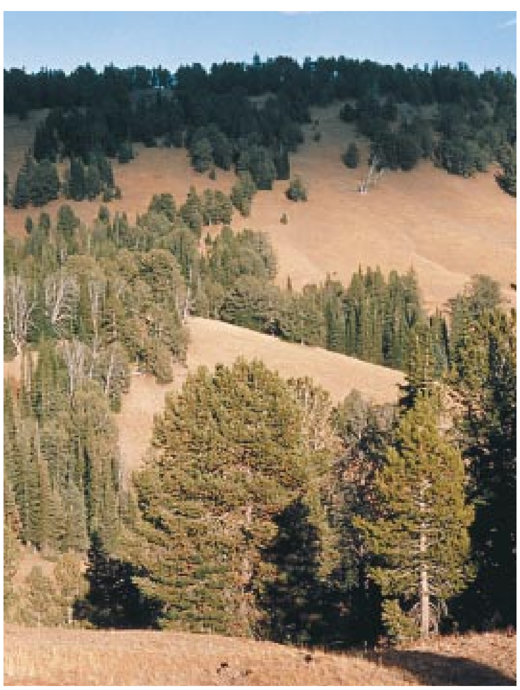
Fig. 1. (a) Natural distribution of whitebark pine (Arno and Hoff 1989; Olgilvie 1990) with mortality zones. Mortality level is the proportion of trees dead from all causes since presettlement. **(b)** White pine blister rust infection rates in whitebark pine. Blister rust is present but infection rates are unknown in Canada and the southern United States.

Repercussions

The alarming loss of whitebark pine has broad repercussions: mast for wildlife is diminished and the number of animals the habitat can support is reduced. Such results hinder grizzly bear recovery and may be catastrophic to Yellowstone grizzlies for whom pine seeds are a critical food. Predicted changes in whitebark pine communities include the absence of reforestation of harsh sites after disturbance and the lowering of treelines. In addition, stream flow and timing will be altered as snowpack changes with vegetation.

Implications

Whitebark pine will be absent as a functional community component until rust-resistant strains evolve. Natural selection could be speeded with a breeding program like that developed



Courtesy D. Reinhart

Fig. 2. Healthy whitebark pine stand in Yellowstone National Park not yet affected by the introduced disease, white pine blister rust.

Courtesy R. McClelland

Fig. 3. Dead whitebark pine trees in Glacier National Park.

for western white pine (*P. monticola*), which also suffers from rust. In some areas where whitebark pine is regenerating, its competitors should be eliminated. To perpetuate whitebark pine at a landscape scale, fires must be allowed to burn in whitebark pine ecosystems.

Isolated populations may become extinct where mountain pine beetle or other agents kill remaining resistant trees. To prevent loss of genetic diversity, seeds of these pines should be collected throughout the species' range and stored as insurance against catastrophic events. To guide restoration efforts, more information is needed on whitebark pine's historical distribution, trends throughout its range, and rust epidemic dynamics.

For further information:

Katherine C. Kendall
National Biological Service
Glacier Field Station
Glacier National Park
West Glacier, MT 59936

References

Arno, S.F., and R.J. Hoff. 1989. Silvics of whitebark pine (*Pinus albicaulis*). U.S. Forest Service Gen. Tech. Rep. INT-253. 11 pp.

Keane, R.E., and S.F. Arno. 1993. Rapid decline of whitebark pine in western Montana: evidence from 20-year remeasurements. Western Journal of Applied Forestry 8(2):44-47.

Kendall, K.C., and S.F. Arno. 1990. Whitebark pine—an important but endangered wildlife resource. Pages 264-273 *in* U.S. Forest Service Gen. Tech. Rep. INT-270.

Olgilvie, R.T. 1990. Distribution and ecology of whitebark pine in western Canada. Pages 54-60 *in* U.S. Forest Service Gen. Tech. Rep. INT-270.

Oak Savannas in Wisconsin

by
Richard A. Henderson
Eric J. Epstein
Wisconsin Department of Natural Resources

Oak savanna is a term given to a loosely defined, yet well-recognized, class of North American plant communities that were part of a large transitional complex of communities between the vast treeless prairies of the West and the deciduous forests of the East. This system was driven by frequent fires and possibly influenced by large herbivores such as bison and elk. A wide range of community types was found within this transitional complex; collectively, they represented a continuum from prairie to forest. The term "savanna" is generally applied to a small group of related community types in the middle portion of this continuum.

Savannas all have a partial canopy of open grown trees and a varied ground layer of prairie and forest herbs, grasses, and shrubs, as well as plants restricted to the light shading and mix of shade and sun so characteristic of savanna. Oaks were clearly the dominant trees, and, hence, the common use of the term oak savanna. Definitions of savanna tree cover range from 5% to 80% canopy; however, the lower canopy covers of 5%-50% or 5%-30% are more widely used criteria. Savanna types range from those associated with dry, gravelly, or sandy soils; those on rich, deep soils; and those on poorly drained, moist soils.

Oak savannas have probably been in North America for some 20-25 million years (Barry and Spicer 1987), expanding and contracting with climatic changes and gaining and losing species (on a geologic time scale) through evolution and extinction. For the past several thousand years, such savannas have existed as a relatively stable band of varying width and continuity, from northern Minnesota to central Texas (Figure).

At the time of European settlement (ca. 1830), oak savanna covered many millions of hectares. It varied somewhat in species composition from north to south and east to west, but

Figure. Gross range of presettlement oak savanna in central North America (adapted from Nuzzo 1986 and Smeins and Diamond 1986). The shaded area represents the general region in which oak savannas occurred, although this region was not uniformly savanna. Significant blocks of nonsavanna vegetation, such as prairie or forest, were also present within it. Nor was oak savanna totally restricted to this region. Small, disjunct outliers existed as far east as Ohio and as far west as the Dakotas.

structure and functions were probably similar throughout. In the upper Midwest (Minnesota, Wisconsin, Michigan, Iowa, Illinois, Indiana, and Missouri) there were an estimated 12 million ha (29.6 million acres) of oak savanna (Nuzzo 1986). Wisconsin's portion was 2.9 million ha (7.3 million acres; Curtis 1959). As the Midwest's rich soils were used for agriculture and fire was suppressed, this ecosystem all but disappeared from the landscape throughout its range. Today, oak savanna is a globally endangered ecosystem.

Status

In the early to mid-19th century, the oak savanna ecosystem was thoroughly fragmented and nearly totally destroyed throughout its range. Most of its acreage suffered from clearing and plowing, overgrazing, or invasion by dense shrub and tree growth caused by lack of fire, lack of grazing, or both. Consequently, oak savanna now shares equal billing with tall-grass prairie as the most threatened plant communities in the Midwest and among the most threatened in the world. Only a little more than 200 ha (500 acres) of intact examples of oak savanna vegetation are listed in the Wisconsin State Natural Heritage Inventory, or less than 0.0001 (0.01%) of the original 2.9 million ha (7.3 million acres)—a fate repeated throughout this community's entire range (Johnson 1986; Smeins and Diamond 1986). A tallying of known oak savanna sites in the upper Midwest (Missouri northward) in 1985 (Nuzzo 1986) listed only 133 sites totaling 2,600 ha (6,420 acres), or only 0.0002 (0.02%) of the estimated presettlement extent of the community. Most of what remains are dry and wet savanna types. Richer, more productive soil savanna is now nearly nonexistent.

Fortunately, most of the biota that was associated with savanna, especially the vertebrates, have either adapted to the changed landscape or have managed to survive in suboptimal habitat (e.g., the fringes of other less devastated communities, such as oak forests). This situation is precarious for many species, however, and their long-term future is doubtful. Vertebrates have been successful because major elements of the savanna structure are still well represented in various edge habitats, including wooded pastures, lawns, and woodlots. The fact that the plant species may be different in surrogate savannas has not affected savanna vertebrate species for the most part.

Oak savanna vegetation has not fared as well. Many species that were probably savanna specialists are now uncommon and are found only in the fringes and openings of oak woods, brushy areas, and lightly grazed pastures. A few examples are giant false-foxglove (*Aureolaria grandiflora*), yellow pimpernel (*Taenidia integerrima*), pale Indian-plantain (*Cacalia atriplicifolia*), New Jersey tea (*Ceanothus americanus*), sessile-leaved eupatorium (*Eupatorium sessilifolium*), and white death-camas (*Zigadenus elegans*). Two likely savanna specialists, purple milkweed (*Asclepias purpurascens*) and wild hyacinth (*Camassia scilloides*), are now listed as endangered in Wisconsin. Three others—kitten-tails (*Besseya bullii*), cream gentian (*Gentiana alba*), and Virginia lespedeza (*Lespedeza virginica*)—are listed as threatened.

Most bird species found in Wisconsin savannas are still doing well today (e.g., American robin [*Turdus migratorius*], indigo bunting [*Passerina cyanea*], and brown thrasher [*Toxostoma rufum*]). Only one oak savanna/woodland bird, the passenger pigeon (*Ecopistes migratorius*), has become extinct, and another, the wild turkey (*Meleagris gallopavo*), was extirpated but is now restored; however, both of these were lost because of unregulated hunting and not because of habitat loss.


Courtesy R. Henderson, Wisconsin Department of Natural Resources

Oak savanna.

Recently, however, a number of savanna birds have not thrived or have begun to decline throughout their range, including the northern flicker (*Colaptes auratus*), red-headed woodpecker (*Melanerpes erythrocephalus*), vesper sparrow (*Pooecetes gramineus*), northern bobwhite (*Colinus virginianus*), warbling vireo (*Vireo gilvus*), and field sparrow (*Spizella pusilla*). The orchard oriole (*Icterus spurius*) and yellow-breasted chat (*Icteria virens*) are on Wisconsin's list of special concern. The loggerhead shrike (*Lanius ludovicianus*) and barn owl (*Tyto alba*) are on Wisconsin's endangered species list, and Bell's vireo (*Vireo bellii*) is now on Wisconsin's threatened species list (D.W. Sample and M.J. Mossman, Wisconsin Department of Natural Resources, personal communication, 1994). Although loss of habitat has not been the cause of decline in all these species, it certainly is affecting many of them. The abandonment and loss of savanna/woodlot pastures in the past few decades may be playing a role in some of the recent declines of savanna bird species.

Most amphibian and reptile species that were closely associated with the historic oak savanna in Wisconsin are doing at least moderately well today, although two reptiles associated with savanna habitat are on the Wisconsin list of endangered species and are suffering from habitat loss: the western slender glass lizard (*Ophisaurus attenuatus*) and the eastern massasauga rattlesnake (*Sistrurus catenatus*). The eastern massasauga is also under consideration for federal listing.

Our knowledge of oak savanna invertebrates is limited; we know little about what species were characteristic or restricted to oak savanna, let alone their current status. Some reliable status information does exist for savanna Lepidoptera (moths and butterflies), however; of this group, the Karner blue butterfly (*Lycaeides melissa samuelis*) is listed as federally endangered while phlox flower moths (*Schinia indiana*) and tawny crescent butterflies (*Phyciodes batesii*) are under consideration for federal listing. The frosted elfin butterfly (*Incisalia irus*) is listed as threatened in Wisconsin, and four savanna skippers (*Erynnis persius*, *Hesperia leonardus*, *H. metea*, and *Atrytonopsis hianna*) and the buck moth (*Hemileuca maia*) are considered rare in the state. Other globally rare insects thought to have been part of the oak savanna ecosystem include the federally listed American burying beetle (*Nicrophorus americanus*) and the red-tailed leafhopper (*Aflexia rubranura*), which is under consideration for federal listing.

Threats

Threats to the future survival of oak savanna throughout its range can be summarized into four categories. The first, loss of recovery opportunities, can be attributed to accelerating succession to tree and shrub species that produce dense shade; a lack of recruitment and eventual death of long-lived plants surviving now only in suboptimal habitat; changes in pasturing practices through either increasing or decreasing grazing pressure; and an increasing rate of rural home building in key savanna areas. The second threat is lack of understanding about the community by the public, professional resource managers, and scientists. Resistance to the use of prescribed fire, especially in wooded areas, and lack of understanding about the importance of fire in maintaining biodiversity are the third threat; invasion by aggressive non-natives (i.e., honeysuckle, buckthorn, and reed canary grass) is the fourth.

Recovery Potential

In the absence of active management, the future of oak savanna looks bleak in Wisconsin and throughout its entire range. The increasing abandonment of lightly to moderately grazed wooded pastures and the accelerating succession of oak woodlots toward heavy shade-producing trees and shrubs are likely to lead to the further decline and possible loss of much of the remaining savanna flora and fauna, including eventual declines of the oaks themselves.

If oak savanna habitats are actively managed, however, their recovery potential in Wisconsin and throughout their range is substantial (Holtz 1985; Bronny 1989; R.A. Henderson, Wisconsin Department of Natural Resources, unpublished data). Many degraded sites in the dry and wet ends of the spectrum can be recovered with relative ease. Mesic, richer soil savannas will take more time and work, but recovery is still feasible. The native species that formerly inhabited oak savannas can be reintroduced with a reasonable amount of effort (Packard 1988), but the options available are quickly decreasing.

In Wisconsin alone, hundreds if not thousands of hectares of overgrown but still retrievable oak savanna exist on public lands and thousands more on private lands. Although Wisconsin may be above average in potential for savanna recovery, similar situations exist in other states. Much of this land, especially low productivity sites, could be restored within a few decades simply by thinning trees, brushing, and burning. Well-drained, rich soil sites, however, will require more work and time to restore. Some plant reintroductions may be necessary, but much can be accomplished with fire alone. Light grazing may also have potential as a savanna management tool.

References

Barry, A.T., and R.A. Spicer. 1987. The evolution and palaeobiology of land plants. Croom Helm, London. 309 pp.

Bronny, C. 1989. One-two punch: grazing history and the recovery potential of oak savanna. Restoration and Management Notes 7(2):73-76.

Curtis, J.T. 1959. The vegetation of Wisconsin. University of Wisconsin Press, Madison. 657 pp.

Holtz, S.L. 1985. Cutting and burning a degraded oak barrens: management techniques that stimulate natural disturbance. M.S. thesis, University of Wisconsin, Madison. 80 pp.

Johnson, F.L. 1986. Oak-hickory savannahs and transition zones: preservation status and management problems. Pages 345-347 *in* D.L. Kulhavy and R.N. Conner, eds. Wilderness and natural areas in the eastern United States: a management challenge. Stephen F. Austin State University, Nacogdoches, TX.

Nuzzo, V.A. 1986. Extent and status of Midwest oak savanna: presettlement and 1985. Natural Areas Journal 6(2):6-36.

Packard, S. 1988. Just a few oddball species: restoration and the rediscovery of the tallgrass savanna. Restoration and Management Notes 6(1):13-20.

Smeins, F.E., and D.D. Diamond. 1986. Grasslands and savannahs of east central Texas: ecology, preservation, and management. Pages 381-394 *in* D.L. Kulhavy and R.N. Conner, eds. Wilderness and natural areas in the eastern United States: a management challenge. Stephen F. Austin State University, Nacogdoches, TX.

For further information:

Richard A. Henderson
Wisconsin Department of Natural Resources
Bureau of Research
1350 Femrite Dr.
Monona, WI 53716