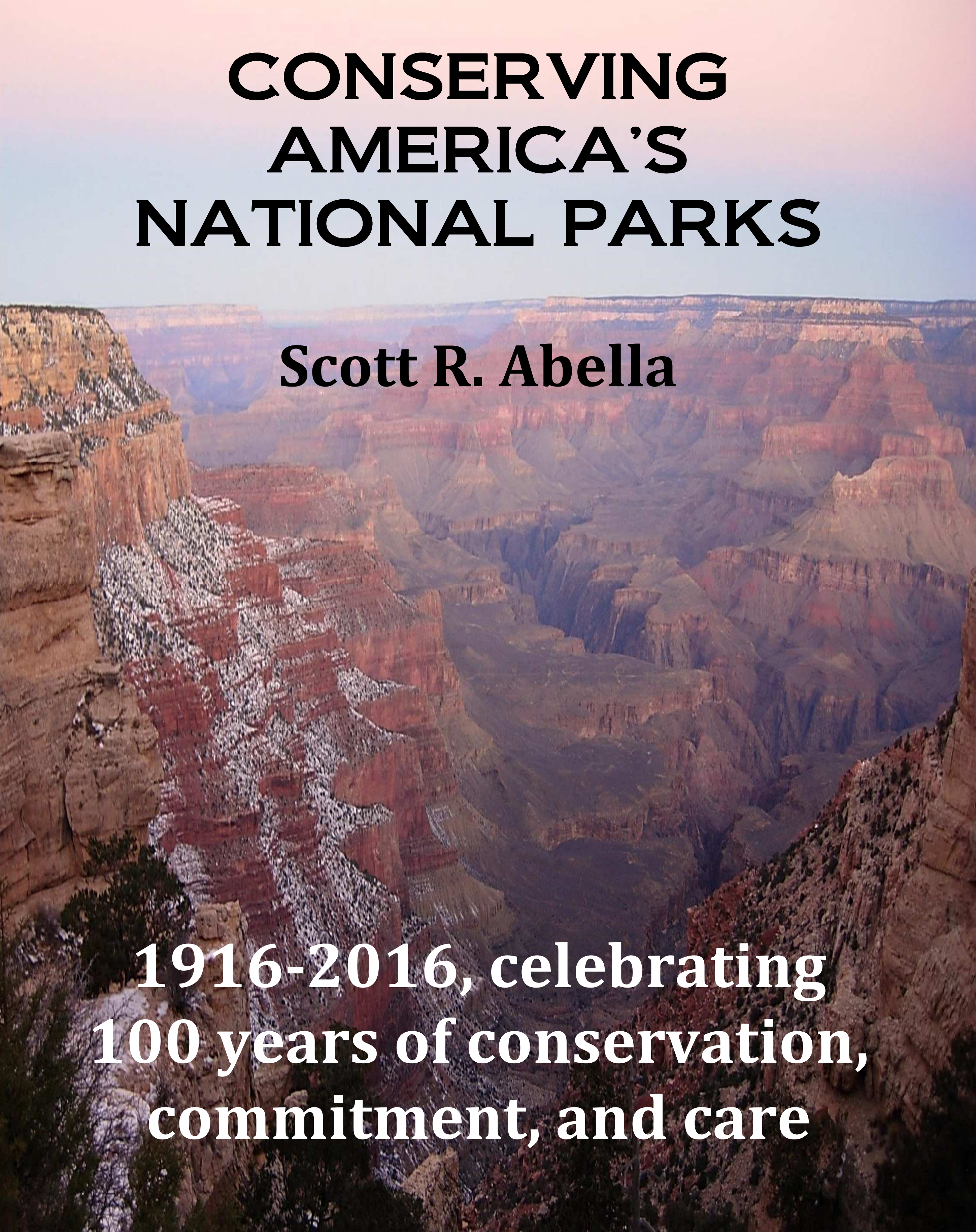

CONSERVING AMERICA'S NATIONAL PARKS

Scott R. Abella

1916-2016, celebrating 100 years of conservation, commitment, and care

PREFACE

With 2016 marking the 100-year centennial of the National Park Service, there are many successes in national park conservation to celebrate. Through dedicated efforts of people and resilience of animals, wildlife species including wolves and panthers, long absent from parks, have returned. In the largest such project in U.S. history, the removal of two dams has restored a free-flowing river and salmon runs in Olympic National Park for the first time since 1912. Over 300,000 eastern hemlock trees were saved from a devastating pest through heroic efforts by park managers. To buffer coasts from rising seas, parks are continuing to restore coastal wetlands. Outcomes of these efforts are thrilling. This book shares and illustrates many other inspiring stories through 247 photos, maps, and sketches.

There are also formidable challenges to conserving parks. Land uses pre-dating park designation, shifting roles of wildfire, invasion by earthworms and other non-native species, drought and altered freshwater supply, collapse of coral reefs, mercury contamination, and climate change are among the many challenges defining modern park conservation. Yet, carefully implemented projects show promise for conserving ecosystems in parks faced with these challenges. *Conserving America's National Parks* tells conservation stories emerging in national parks and their influence on the future of the national park system.

This book is designed to be readable and inspiring for diverse audiences interested in national parks and conservation, whether readers are students, interested citizens, natural area managers, scientists, policymakers, or anyone in between.

Wolf pups, which have returned to Yellowstone and Grand Teton National Parks (National Park Service).

Some wildlife species of national parks. Top: badger and bear cubs (photos courtesy of Grand Teton National Park). Middle: deer fawn (by C. Ballou, Big Bend National Park) and Florida panther kittens (courtesy of U.S. Fish and Wildlife Service). Bottom: coyote pup (by J. Good, Yellowstone National Park).

Conserving America's National Parks

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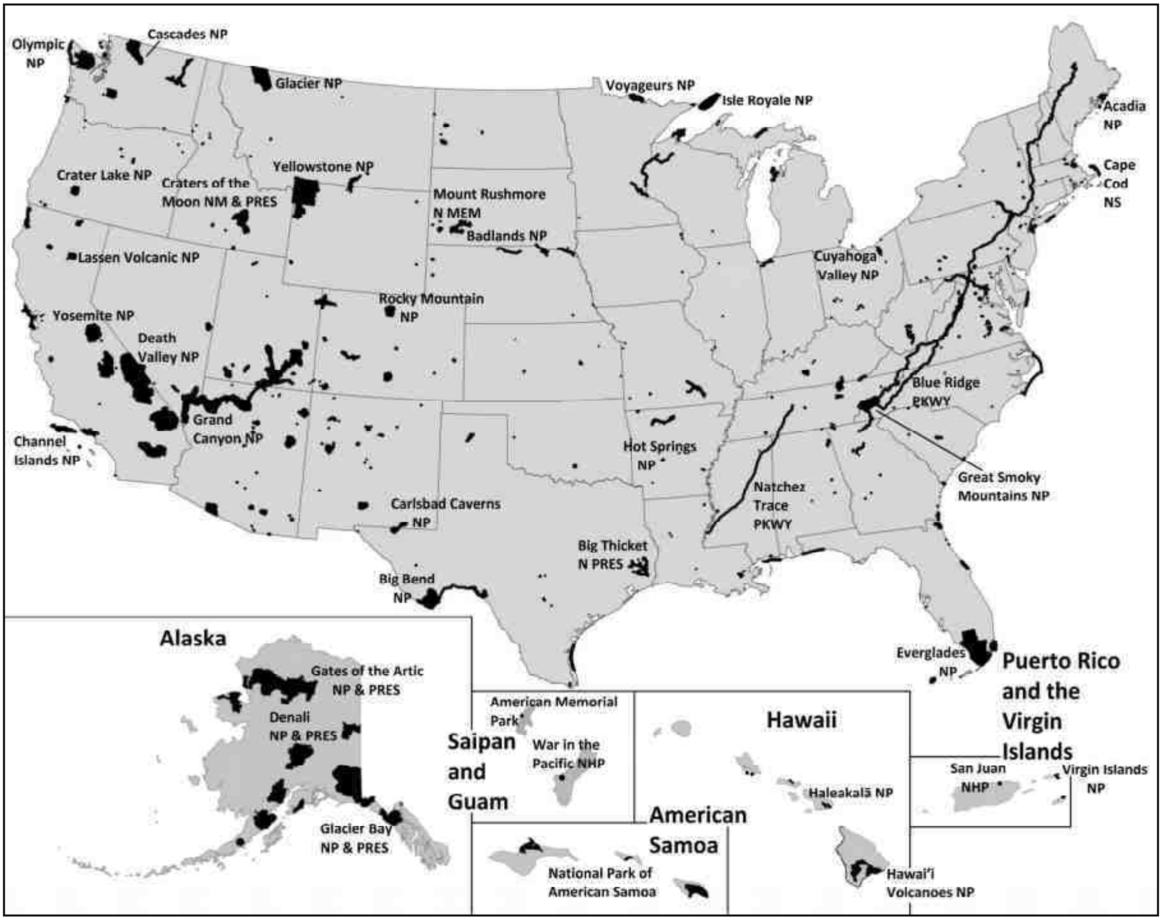
CONTENTS

NATIONAL PARKS OF THE UNITED STATES



The national park system contains 408 parks, with some examples labeled on the map. Abbreviations: MEM, memorial; NHP, national historical park; NM, national monument; NP, national park; NS, national seashore; PKWY, parkway; and PRES, preserve.

1 THE LEGACY

In an extraordinary turn of events, the United States created Yellowstone National Park in 1872 as the first national park in the world. Intensive exploitation of natural resources was in full swing at that time in the United States. Logging forests, grazing livestock, hunting and trapping animals, clearing land for farms, and manipulating waterways were altering natural ecosystems. Against this backdrop, the U.S. Congress approved the establishment of Yellowstone National Park. While a prime motivation for Yellowstone's designation was economic to promote tourism and business for railroads, it still departed from the traditional thinking.[8] In Yellowstone's case, its scenery preserved in place would be its value, rather than extracting or using its natural resources.

Not much thought then was given to ecology and the fact that ecosystems continually change. Parks were mostly viewed as unchanging scenic landscapes. We know this to be untrue today, but at the time, Charles Darwin's book *On the Origin of Species* had been published only 13 years earlier in 1859.[6] *On the Origin of Species* outlined the change and evolution of species, which now underpins modern biology.[2] No matter the motivations or level of consideration given to conserving species in 1872, early efforts to designate national parks set in motion the conservation of natural areas still unfolding today. It is all the more remarkable to consider the permanency of these early efforts through inconceivable political and social change. At 144 years old, Yellowstone National Park has been through the tenures of 27 U.S. presidents and 73 changeovers of the U.S. Congress. Since record keeping began in 1904, 167 million people have visited Yellowstone through 2014. Visitation increased from 13,727 people in 1904 to 3.5 million in 2014.

Fig. 1.1. Dana Meadows and a tributary of the Tuolumne River near Tioga Pass, Yosemite National Park, California. Yosemite was designated in 1890 as the third national park (photo by S.R. Abella).

Following Yellowstone, five more national parks were created by 1902: Sequoia and Yosemite in California plus the small General Grant National Park later incorporated into Kings Canyon in 1940, Mount Rainier in Washington, and Crater Lake in Oregon (Fig. 1.1). A 1906 Antiquities Act and its shrewd use by 26th President Theodore Roosevelt, a conservation proponent, increased the rate of park establishment. By 1916, 23 more parks were designated, such as Zion and Petrified Forest, raising the total to 29 parks.

Who would actually manage the parks? Yellowstone was already 45 years old by 1916 and was initially overseen by the U.S. army. The 1916 Organic Act created the National Park Service to manage the parks. The act also mandated that while providing for the people's enjoyment, the parks be conserved "unimpaired" for future generations. What unimpaired means has been debated in the century since. Today, unimpaired is generally considered successful conservation of natural and cultural features for which a park was designated (such as fossils of Petrified Forest National Park) or that are critical to a park's integrity.[4] Owing to interrelationships on the landscape, like between the health of soil and the integrity of fossils, many natural features are critical to park integrity.

New parks continued to be created after formation of the National Park Service. By 1950, over 150 parks had been designated. This total included several parks in the East established to provide accessible nature close to urban areas. Some of these parks were Shenandoah (Virginia), Great Smoky Mountains (North Carolina/Tennessee), and Mammoth Cave (Kentucky) designated in 1926; and in 1936, the Blue Ridge Parkway winding from North Carolina to Virginia (Fig. 1.2).

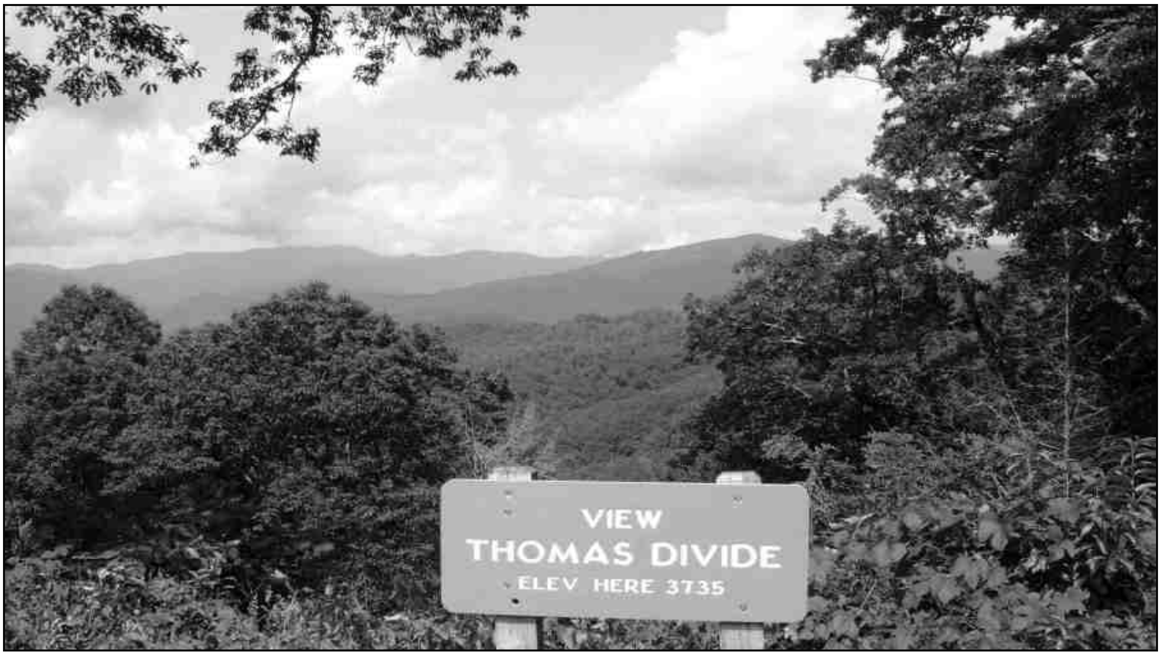


Fig. 1.2. Along the Blue Ridge Parkway, here northwest of Asheville, North Carolina (S.R. Abella).

As of September 2015, the national park system had 408 parks. This includes some of the newest parks designated in December 2014, like Harriet Tubman Underground Railroad National Historical Park in Maryland, Valles Caldera National Preserve in north-central New Mexico, and Tule Springs Fossil Beds National Monument adjacent to Las Vegas, Nevada (Fig. 1.3). In July 2015, Waco Mammoth National Monument was added near Waco, Texas, to conserve the remains of ancient mammoths, camels, and saber-toothed cats. National parks are located in all 50 states and the U.S. territories of Guam, American Samoa, Puerto Rico, Saipan, and the Virgin Islands.

Fig. 1.3. One of the newest national parks, Tule Springs Fossil Beds National Monument, Las Vegas, Nevada. The pile in the foreground that appears to be rocks is instead remains of a mammoth skull, preserved in the white gypsum soil since the ice age (S.R. Abella).

The 408 units of the national park system include 28 different types of designations and do not always have the words "national park" in their name. Some of the different park designations in the system and managed by the National Park Service include: national monuments, national recreation areas, national rivers, national seashores, national preserves, national historical parks, national battlefields, and others (Fig. 1.4). The designations define the general focus of a park, but all 408 units, no matter their designation, are managed under the National Park Service policy of being conserved unimpaired for future generations. All of the parks in the national park system are referred to as national parks for convenience throughout the book, except when identifying specific parks by name.

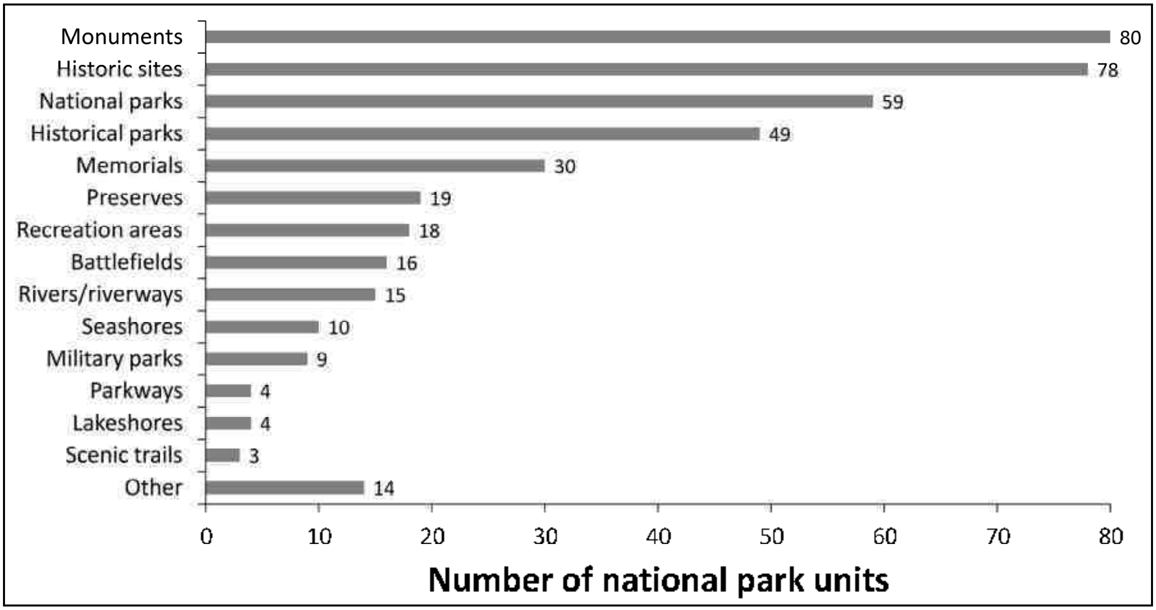


Fig. 1.4. Breakdown of the 408 national park units by their designation. The three designations with the most units are national monuments, national historic sites, and national parks. Numbers at the end of bars are the exact numbers of units (compiled from the National Park Service).

In an interesting twist, many of the national battlefields, historical parks, and recreation areas also contain unique natural features for which they are not necessarily known. For example, once the scene of the first major battle west of the Mississippi River in 1861 during the American Civil War, Wilson's Creek National Battlefield near Republic, Missouri, harbors one of the largest populations of Missouri bladderpod (*Lesquerella filiformis*). This rare plant species inhabits prairies only of southwestern Missouri to northwestern Arkansas.[11] Near Santa Fe, New Mexico, Pecos National Historical Park conserves over a thousand years of human history. The park contains structures of the Native American Pueblo people, Spanish missionaries, and the battlefield of Glorieta Pass, one of the most important American Civil War battles in the West.[7] The park simultaneously conserves a rich landscape of ponderosa pine (*Pinus ponderosa*) forest and grassland (Fig. 1.5). This book includes stories from parks well known for their natural features, such as Yellowstone, and less well known, but where conservation activities for natural features are occurring.

The 408 national parks total 32 million hectares (80 million acres), or 3.3% of the area of the United States. Alaska's 23 parks contain 66% of the total national park area. In the lower 48 states, the 385 national parks occupy 11 million hectares (27 million acres), or 1.4% of the area of the lower 48 states. This is equivalent to the size of the state of Virginia.

Although national parks comprise only a small portion of the United States, what they contain is remarkable. The parks have examples of the smallest, largest, tallest, lowest, and oldest things in the United States and in some cases the world. For example, national parks contain the highest point in North America (Mt. McKinley, Denali National Park, Alaska), the lowest point (Badwater Basin, Death Valley National Park, California), the world's largest land carnivores (polar bears, *Ursus maritimus*, national parks in Alaska), the longest cave in the world (Mammoth Cave National Park, Kentucky), and the world's tallest tree (a redwood, *Sequoia sempervirens*, Redwood National Park, California).

Fig. 1.5. Ceremonial kiva constructed by the Pueblo people within a mixed wooded-grassland landscape of Pecos National Historical Park, New Mexico. A historic Spanish church also is on site (S.R. Abella).

Biodiversity of U.S. national parks is tremendous, including for species of global conservation concern. An inventory of bees since 2010 in 46 national parks has already identified 700 species of bees, including ones previously unknown to science.[5] Bees are of global concern for pollinating plants in natural, agricultural, and horticultural settings. Amphibians and reptiles provide important functions within ecosystems and their presence is a barometer of the health of the environment. An inventory of 16 southeastern parks identified 123 species of native amphibians and reptiles, a quarter of the nation's total.[10] Great Smoky Mountains National Park contains an astounding 18,545 known species, which might be only 20% of the total number of species the park contains.[9]

What makes parks so valuable to the country, even to people who may not care much about nature? National parks provide a recreation resource for 280 million visitors annually. In the process, national parks contributed $30 billion to the U.S. economy in 2014 through visitor spending in gateway communities near parks, businesses supporting the parks, and job creation.[3] The National Park Service also encourages and facilitates the education of all age groups interested in history, cultural heritage, nature, and science. Parks are often core areas around which conservation programs are developed for natural resources critical to society. One of many examples is freshwater. National parks are located around some of the nation's most important groundwater sources, lakes, reservoirs, and rivers. Storing carbon in the soil and living organisms, thus taking it out of the air to ameliorate climate change, is another benefit the parks provide with national and global significance. National parks near and far from urban areas are a main part of U.S. air quality monitoring programs. Monitoring air pollution in parks has detected good and bad trends in air quality linked with human well-being, including for rural residents affected by moving polluted air masses. Additionally, the natural conditions of national parks provide valuable baseline information to optimize production on economically "working" landscapes, such as sustainable fisheries, industrial timberlands, and livestock rangelands.

Fig. 1.6. Fox in Big Bend National Park, Texas (National Park Service photo). National parks are refuges for predators such as foxes.

Notwithstanding the utilitarian benefits of parks, ultimately national parks represent a small portion of the country where the primary goal is conserving natural and cultural features for their inherent value (Fig. 1.6). As this book illustrates, this does not necessarily mean keeping nature in some type of unchanging state. Trying to do that can itself be unnatural. Indeed, the continuous evolution of species has enabled life to persist through Earth's changing environments. One of the greatest challenges to conserving natural features in parks is identifying which changes are taking place and whether they are consistent with natural processes. For instance, increasing amounts of toxic mercury in animals due to pollution by humans is an undesirable, unnatural change. Species transported to another continent on an airplane or a ship, intentionally or unintentionally by humans, would likewise not be a natural change.[1] The species would not have arrived at that time on their own. Such introduced species can devastate native ecosystems.

Nature in national parks is supposed to be *authentic*. This means an ideal of natural processes operating (like natural fires) and species evolving in response to natural changes in the environment. In modern conservation, certain practices of Native Americans, like intentionally burning some ecosystems, are not excluded from natural processes. This is because ecosystems we still have today evolved with the practices for thousands of years.

The challenges to conserving national parks are numerous and seem to be intensifying. Altered roles of fire, removal and addition of species, obstruction of rivers by dams, air pollution, and climate change are some of the many threats to the natural features that parks are supposed to protect. Even park scenery, once viewed as unchanging and unrelated to ecology, is now inseparable from conserving ecosystems. Glacier National Park, so named for its ice-covered mountain scenery, is one of several examples of the inseparability of scenery and changing environmental conditions. National parks have not been immune to widespread ecological problems, such as collapses of coral reef ecosystems and the devastation of certain western forests from unnatural fires. This adds to concern regarding what type of nature might still be intact within parks in the coming decades.

It might become easy to lose hope about the future of conserving natural features in parks were it not for exceptional conservation success stories. Several of these stories are shared in this book. There are many examples of improving ecological conditions during good stewardship of ecosystems, including in some cases restoring habitat that had been lost from parks. This book provides a balanced portrayal of the formidable threats parks face and some inspiring reversals of those threats.

The next chapters each cover an individual conservation challenge or natural feature. Chapter 2 describes how land uses before parks were designated continue to affect contemporary conservation. In Chapter 3, the shifting role of wildfire in parks is illustrated, including circumstance where more, less, or different types of fire are needed to restore natural conditions. Chapter 4 describes predatory animals (such as wolves and bobcats) and changes that have occurred when predators are removed or reintroduced. Chapters 5 and 6 cover the effects of species unnaturally introduced to parks, such as invasive plants, pythons, earthworms, and ants. Chapter 7 focuses on forests, including some of the oldest trees in parks and heroic efforts by park managers to protect forests from damaging pests. Including the largest project of removing a dam from a river in U.S. history, Chapter 8 shares several projects successfully conserving freshwater ecosystems in parks. Chapter 9 highlights overall improvements in park air quality, while increases in some pollutants are cause for concern. Hurricanes, rising sea levels, degradation of coral reefs, and other conservation challenges in parks along ocean coasts are the subject of Chapter 10. Chapter 11 provides a long-term context for contemporary climate change, effects that climate change may be having in parks, and tradeoffs of potential conservation strategies in a changing climate. The last chapter, Chapter 12, illustrates ways that active conservation measures can meet present and upcoming challenges for conserving parks. Each chapter starts with an overview and then shows examples from parks, ending with a prognosis for the future.

Units of measurement throughout the book are provided as both metric and English units for ease of interpretation. Measurement units are usually spelled out, except kilometer can be abbreviated as km and square kilometer (a square 1 km long on each side) as km^2. Supporting references and further reading are cited using superscript numbers that correspond with references by chapter provided at the end of the book.

2 BEFORE THEY WERE PARKS

Ancient human land uses – such as intensive agriculture by the Maya civilization in Central America and clearing of forests in France by the Romans 2,000 years ago – have left legacies on the landscape still evident in contemporary ecosystems.[14] The relatively recent Euro-American settlement during the last 400 years in the United States has similarly imprinted the landscape. Settlement sometimes accentuated earlier land-use legacies of Native Americans, and in other cases, created different legacies. Legacies originating before parks were established have carried forward into today's parks.

In some parks, these legacies are obvious. Indeed, the legacies themselves have spurred designation of parks. Examples include: flat-topped pyramidal mounds, up to 17 meters (55 feet) tall, built 1,000 years ago by Native Americans and now protected in Ocmulgee National Monument near Macon, Georgia; earthen pueblos constructed by the Pueblo people in the 1100s in Wupatki National Monument near Flagstaff, Arizona; and many historical and battlefield parks conserving features of human construction. Some of these features are the 1777-1778 winter encampment of the Continental Army during the Revolutionary War (Valley Forge National Historical Park, southeastern Pennsylvania) and a 298 km (185 mile) long canal between Washington, D.C. and western Maryland in use from 1830 to 1924 (Chesapeake and Ohio Canal National Historical Park). Often, however, land-use legacies escape superficial visual identification. Despite their subtlety, land-use legacies continue affecting park landscapes in ways we are still trying to understand.

Some of the many past human activities that have left imprints in today's national parks are creation of infrastructure (such as roads, homesteads, and agricultural fields), logging, grazing livestock, initiating changes in species abundance or extinctions, introducing non-native species, and altering fire occurrence. It can be surprising to learn that seemingly "pristine" parks have a long history of human manipulation. This manipulation has sometimes promoted highly valued ecosystems, sometimes degraded ecosystems, and sometimes just made parks different than they would be without past human activities.

While not all areas in today's parks are heavily influenced by land uses pre-dating park designation, land-use legacies are so pervasive that they are central to understanding conservation needs of parks. The following sections discuss examples of land-use legacies, their ecological effects, and how they influence contemporary park conservation.

Infrastructure

Contemporary vegetation in Cape Cod National Seashore, Massachusetts, reflects a human imprint of woodland clearing for agriculture, roads, and fences in the centuries before the park was created in 1961.[10] For at least 9,000 years after the last major ice age, the area now inside the park was dominated by woodlands of pitch pine (*Pinus rigida*) and oaks (*Quercus* species) under land use by Native Americans that numbered 500 people in 1698.

After two centuries of Euro-American settlement, the area contained an extensive network of roads, fences, and buildings by 1850. At that time, 44% of woodland now on the landscape was plowed for agricultural fields or livestock pastures, 42% was repeatedly logged (but not plowed), and 14% was used for diverse purposes. When people began abandoning their farms later in the 1800s and early 1900s, natural reforestation began. The park has more woodland now than the area did in the mid-1800s (Fig. 2.1).

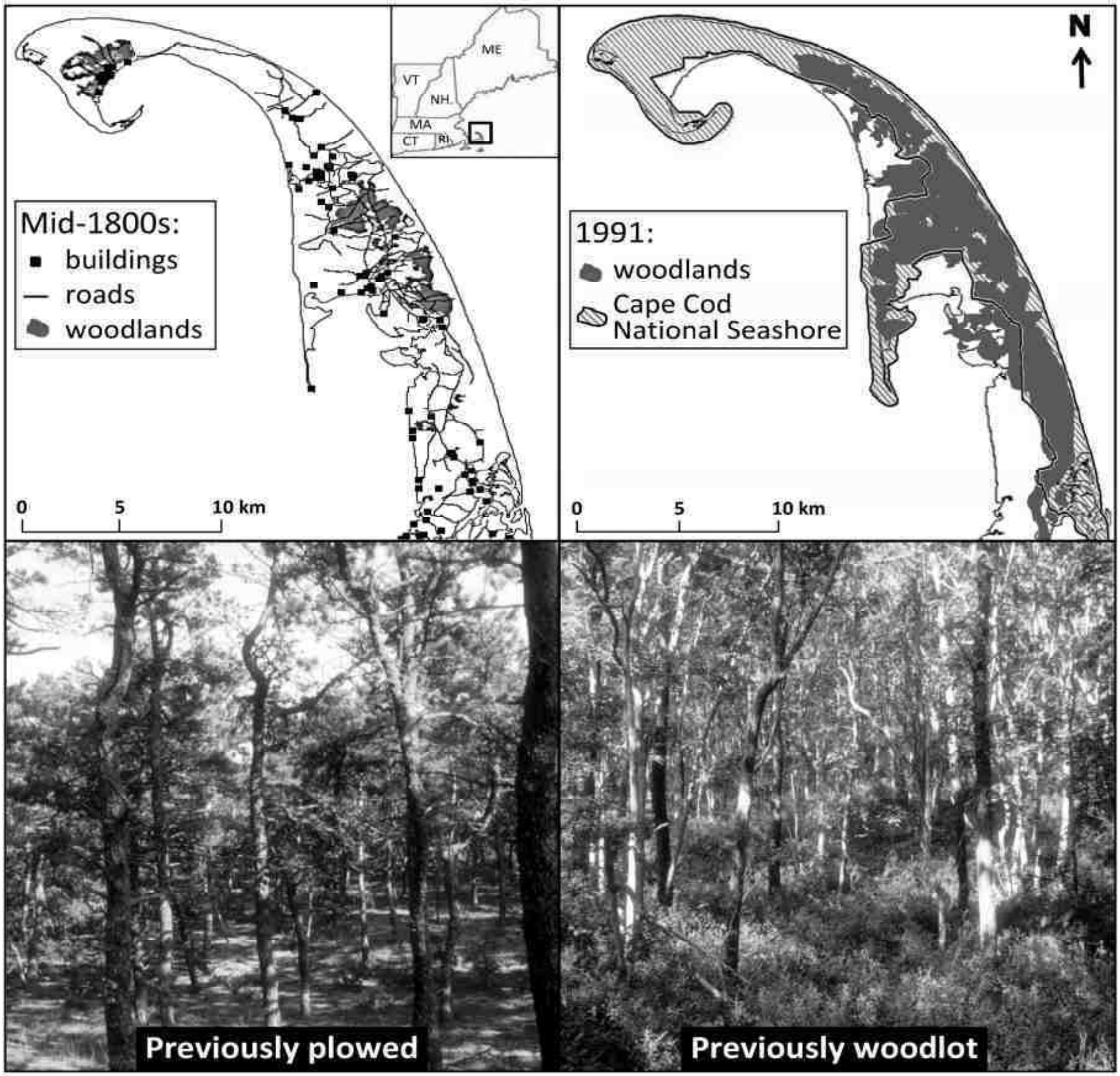


Fig. 2.1. Mid-1800s buildings, roads, and woodlands (compared to 1991), Cape Cod National Seashore, Massachusetts. Maps based on Eberhardt et al. (2003),[10] using data provided by Harvard Forest. Photos: contemporary woodlands with different historical land uses (photos courtesy of D.R. Foster).

In the park today, woodlands growing on soil not previously plowed have different tree overstories and understory plants than woodlands on previously plowed soil. Woodlands never plowed have mixed pine-oak overstories, whereas previously plowed woodlands have more pine and less oak. The shrubs teaberry (*Gaultheria procumbens*), hillside blueberry (*Vaccinium pallidum*), and trailing arbutus (*Epigaea repens*) are abundant in understories of never-plowed woodlands but are sparse in previously plowed woodlands. In contrast, previously plowed woodlands have an abundance of the grasses little bluestem (*Schizachyrium scoparium*) and wavy hairgrass (*Deschampsia flexuosa*) and the low-growing shrub striped wintergreen (*Chimaphila maculata*). When we know what to look for, previously plowed or unplowed woodlands can be visually distinguished on the current landscape (Fig. 2.1).

What are the implications of these land-use legacies for conservation in the park today? After the park was established in 1961, natural reforestation continued, but at the expense of open woodland containing plants promoted by fire and past woodcutting. The now uncommon open woodland provides habitat for rare plant and animal species designated as conservation priority in the northeastern United States. But, by being partly derived from past human use, does conserving these habitats conflict with a mandate of managing for natural conditions? These considerations represent tradeoffs in contemporary conservation and invoke reflection on what elements of the landscape we seek to maintain. Where perseverance of some grassland and open woodland habitat is desired, clearing small patches of trees or using fire may keep some areas open.[26]

On the other side of the country in California's Death Valley National Park, brief mining boom towns have imprinted the landscape for more than a century.[33] In January 1906, two prospectors located gold deposits in the northern Panamint Mountains within what would become the park 27 years later. By July 1906, 40 mining claims were registered, and the town of Skidoo was established about half a mile from the mines. Quickly, Skidoo contained streets, 130 buildings of framed canvas or wood, and a population of 450 people by May 1907. Just as quickly, abandonment of the town began that summer when businesses moved or went bankrupt during the national financial panic of 1907.

Measuring the vegetation on the current landscape and re-taking historical photos demonstrated that cover of perennial plants on the former townsite is similar to the plant cover found on adjacent undisturbed areas. However, plant cover on the former townsite is provided by different species than on the undisturbed areas, even a century after the town was abandoned (Fig. 2.2 left column).

In another example, copper deposits in the Black Mountains in southeastern Death Valley attracted thousands of prospectors in 1905.[33] One of the resulting towns, Kunze, was established in 1906 and supported travel by both horses and automobiles on its streets. An enormous sum of over $50 million (in 1906 dollars) was invested in area mines during 1906-1907, but ore was not found that could be profitably mined. As fast as it originated, the town was abandoned in 1908. On the current landscape, cheesebush (*Hymenoclea salsola*) dominates the former townsite, creating a shrubland differing from the creosote bush (*Larrea tridentata*) and spiny menodora (*Menodora spinescens*) shrubland on nearby undisturbed areas

(Fig. 2.2 right column). Soil compaction, requiring a century to alleviate through natural processes, could have partly limited the establishment of certain plant species on the former town roads. Such long recovery times reveal an important lesson for how long-lived impacts will likely be for new disturbances. For example, bulldozing vegetation and soil for solar energy development has been proposed to occur near boundaries of California desert parks.

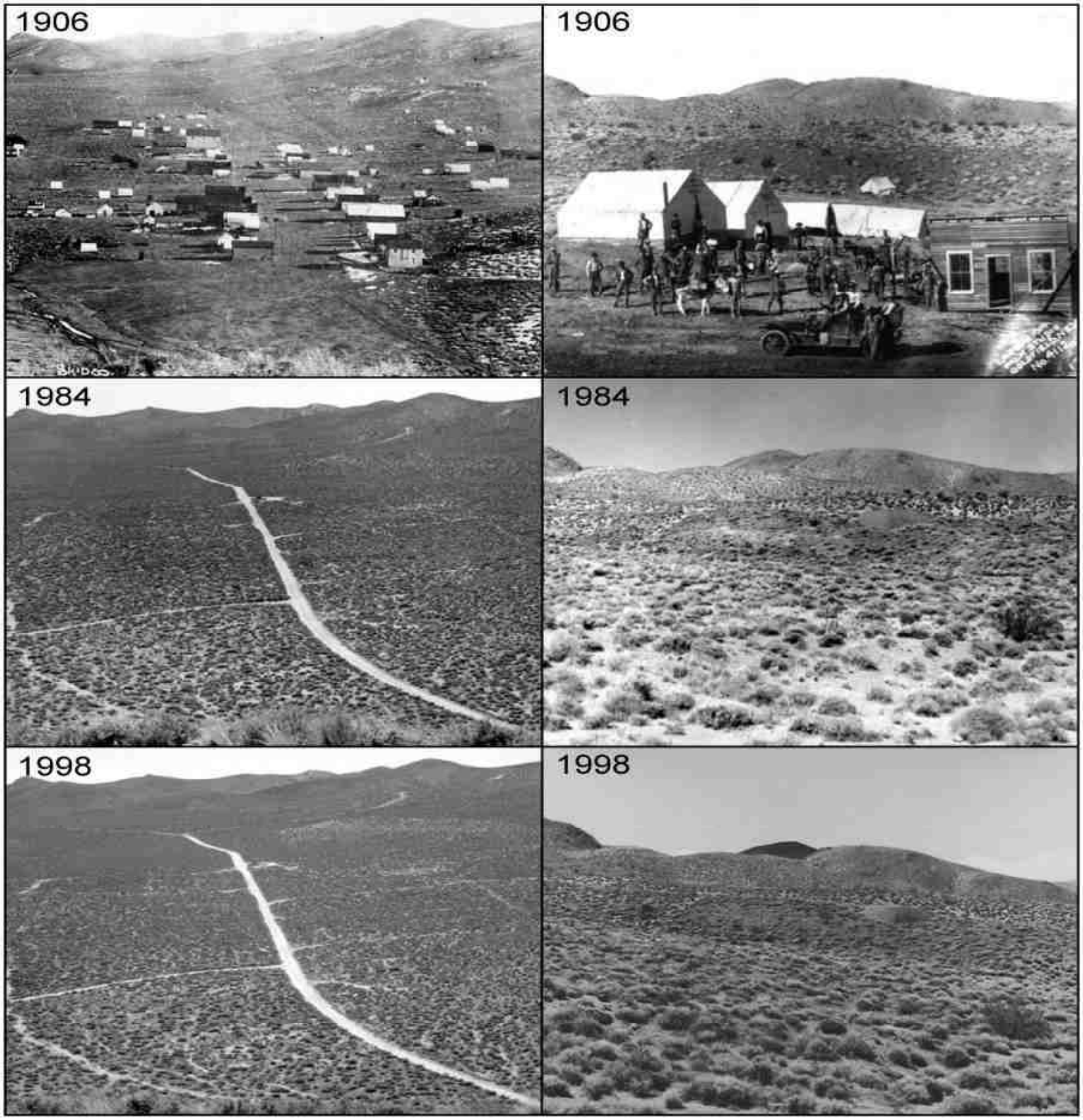


Fig. 2.2. Change after abandonment of the mining towns Skidoo (left photos) and Kunze (right photos), Death Valley National Park, California. From Webb et al. (1988).[33]

Logging

Logging extensively occurred in many forested parks before their establishment. Forty percent of Great Smoky Mountains National Park in North Carolina/Tennessee had incurred corporate logging prior to the final establishment of the park in 1934.[27] This logging was often mechanized. Trees cut by hand, using cross-cut saws, were affixed to aerial cableways or dragged along the ground ("skidded") using steam-powered machines. The logs were moved to railroads for transport to sawmills. Logging frequently cut down all trees via clearcutting. In some areas, trees were only selectively cut, leaving "inferior" trees. Additionally, 31% of land had diffuse disturbance (including earlier, non-mechanized logging or livestock grazing) and 9% had concentrated human settlement. As a result, only 20% of the park in 1934 had forest in a near virgin state (Fig. 2.3).

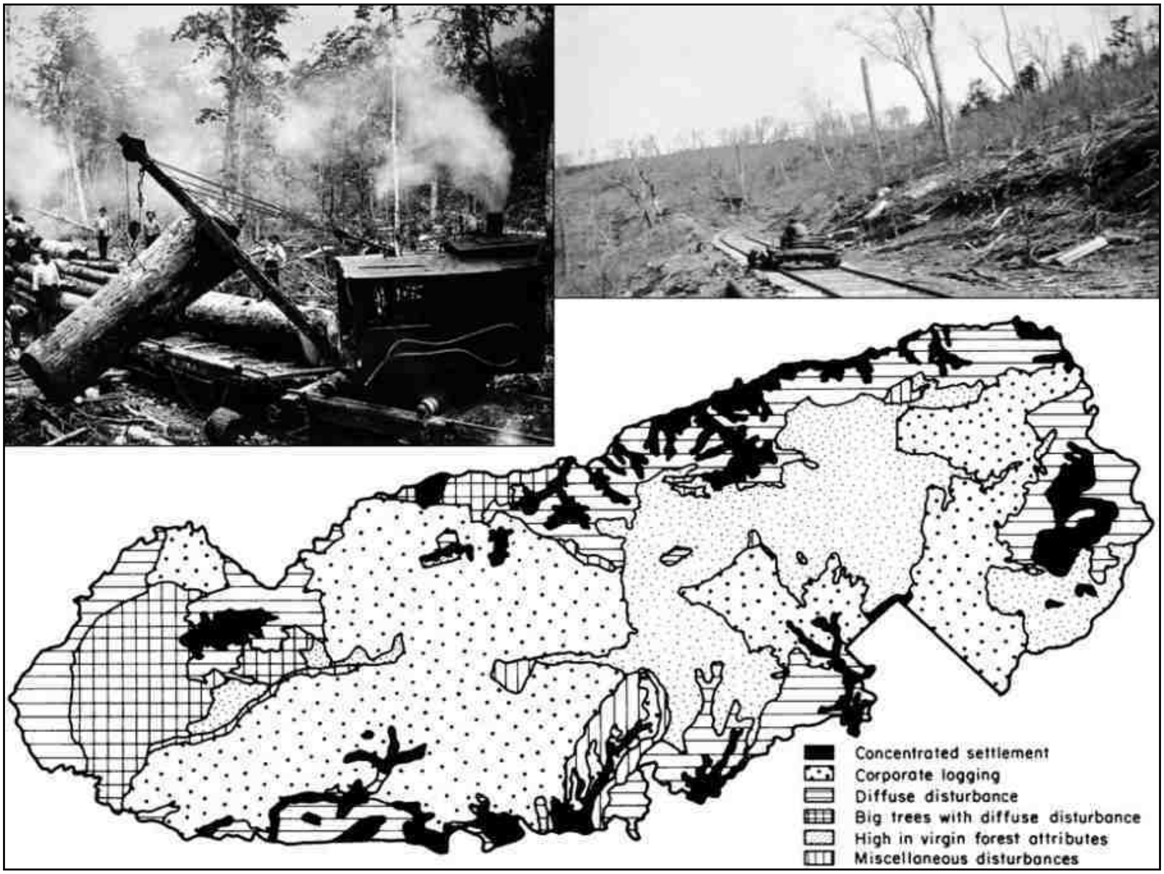


Fig. 2.3. Human disturbances in Great Smoky Mountains National Park at the time of its establishment in 1934. Only 20% of the park had not been logged or otherwise substantially disturbed. Map adapted from Pyle (1988).[27] Photos, courtesy of the National Park Service, show historical logging operations loading logs by steam engine onto flatbed rail cars (left photo) and cutover forest (right photo).

How does this logging legacy influence the contemporary park landscape? Not surprisingly, previously logged forests, now 80 years old, contain 50% fewer large trees greater than 50 centimeters (20 inches) in trunk diameter than unlogged forests.[31] Also, logged forests contain proportionally more yellow birch (*Betula alleghaniensis*), red maple (*Acer rubrum*), and sweet birch (*Betula lenta*). These trees initially colonize after forest disturbances in the southern Appalachian Mountains where the park is located. Overall though, tree composition of the logged and unlogged forests is not that dissimilar.

Likewise, bird communities are not that different. Three common bird species – dark-eyed junco (*Junco hyemalis*), black-throated blue warbler (*Setophaga caerulescens*), and winter wren (*Troglodytes hiemalis*) – are more frequent in unlogged forests. However, another species – Scarlet tanager (*Piranga olivacea*) – is more frequent in logged forests. Of 68 bird species, 54 occurred in both logged and unlogged forests, two only in unlogged forests, and 12 only in logged forests. Thus, 79% of bird species were shared by logged and unlogged forests.

Other parts of the ecosystem displayed greater difference between forest types. The total number of salamanders was twice as high in unlogged compared to logged forests.[18] A species only inhabiting the southern Appalachian region, the red-cheeked salamander (*Plethodon jordani*), was eight times more abundant in unlogged forests. Compared to streams draining logged forests, four times more woody debris occurred in the channels of streams draining unlogged forests. The abundance of woody debris can affect habitat quality for fish.[30] However, streams in unlogged forests contained 38% fewer aquatic invertebrate organisms than streams in logged forests. These examples illustrate that different components and species of an ecosystem respond differently to land-use legacies.

Legacies of logging are more visible in the Great Lakes region. In Michigan's Upper Peninsula along Lake Superior, Pictured Rocks National Lakeshore was established in 1966. Logging began 86 years earlier, with cutting of eastern white pine (*Pinus strobus*) in 1880.[15] Logs were moved into Lake Superior, where they were assembled into rafts covering acres of lake surface. The log rafts were floated to sawmills, such as at Grand Marias and Sault St. Marie. Logs were also moved via logging railroads.

The 2,800-hectare (7,000-acre) Kingston Plains, around the eastern section of the park, supported one of the most extensive white pine stands in the Upper Peninsula until the plains were logged by 1890.[15] As was common, a series of severe fires occurred after logging, fueled in part by slash (branches and other woody debris) left by logging operations. Today, the area is a "stump prairie," consisting of grasses, bracken fern (*Pteridium aquilinum*), shrubs, and scattered pine stumps reminding us of the former forest (Fig. 2.4). Compared to nearby areas where forest reestablishment has occurred, soils of the stump prairie lack a surface layer of partly decomposed organic matter and have less organic carbon in subsurface layers.[3] These differences may reflect the influence of post-logging fires and subsequent effects that either forest or prairie vegetation had on soil development. While old-growth pine stands in the Great Lakes region are now rare, the legacy of logging created unique habitats, such as stump prairies, that appear persistent.

Fig. 2.4. Stump prairie in 2011, occupying former eastern white pine forest, Pictured Rocks National Lakeshore, Michigan. Photo taken and provided by B. Leutscher, National Park Service.

Livestock Grazing

Livestock have been moved through, kept, or become feral within park lands for sometimes centuries before park designations. Spurred by a demand for beef to support gold miners in California, thousands of cattle were moved 2,400 km (1,500 miles) from Texas through western states to California in the mid-1800s.[25] There was incentive to do this: a cow bought for $5 to $15 in Texas sold for $60 to $150 in California. After these early cross-country cattle drives, the number of livestock in western states continued increasing. Arizona, which now contains 22 national park units, had an official estimate of 721,000 cattle on its rangelands in 1891, but some ranchers estimated the actual number was 1.5 million.[25] Now home to 14 national parks starting with El Morro National Monument in 1906, New Mexico contained a whopping 2 million sheep and 500,000 cattle in the 1880s.[29] These numbers increased even further to 3.5 million sheep and 1 million cattle in 1900. Thousands of livestock had already been kept in New Mexico since at least the 1500s, and according to some reports, concern about overgrazing was noted as early as 1630.[29]

While we may view livestock grazing as primarily a western U.S. activity, it was common in many eastern parks and Hawaii as well. For example, an average of 5 cattle/km^2 (13 cattle/square mile) and 6 pigs/km^2 (16 pigs/square mile) were kept from 1860 to 1940 in what is now Ozark National Scenic Riverways, designated in 1964 in southern Missouri.[16]

Cumberland Island National Seashore, established in 1972 off the coast of Georgia, has contained feral pigs, introduced by Spanish colonists, since at least the 1600s.[32] The 14,737-hectare (36,415-acre) seashore contained 2 pigs/km^2 (5 pigs/square mile) in 1980 when the National Park Service was removing pigs from the island. The island still presently contains pigs. Brought by Polynesians to Hawaii, pigs inhabited Hawaii Volcanoes and Haleakala National Parks even longer – over 1,000 years.[31]

Major livestock species presently inhabiting U.S. wildlands are cattle (*Bos taurus*), sheep (*Ovis aries*), goats (*Capra aegagrus hircus*), domestic pigs (*Sus scrofa domesticus*, also called hogs or swine, and descended from European wild boar), European wild boar (*Sus scrofa*), feral burro (*Equus asinus*, also called donkeys), and feral horses (*Equus ferus caballus*). One or more of these species has been in – or still reside in – many national parks. None of the species are native to the United States. Because so few areas exist for comparison that have been free of livestock influences, it is difficult to evaluate the effects of livestock grazing especially when it occurred for centuries before parks were designated.

One of the largest and longest-term grazing exclosures in the West enabled a unique assessment of long-term protection from grazing in Chaco Culture National Historical Park, northern New Mexico.[13] Created in 1907 to include 8,600 hectares (21,250 acres), the unfenced park continued to be grazed by cattle, sheep, and goats. To keep these livestock out, park managers fenced the entire park by 1948 and they also fenced an additional 5,000 hectares (12,350 acres) later added to the park. At the time of the assessment in 1999-2000, the original area had been fenced for 50 years and the additional area up to five years.

Shrub and perennial grass cover was greatest in areas ungrazed for 50 years at four of six study sites.[13] However, cover in ungrazed areas was similar to, or lower than, areas currently grazed at the other two sites. Cover of biological soil crusts – consisting of lichens, mosses, and bacteria growing on the surface of soil – was greater in long- and short-term protected areas compared to grazed areas at all sites. Biological soil crusts provide important functions, such as stabilizing soil to limit erosion. Overall, the number of plant species present and the cover of soil crust were greatest in areas ungrazed for 50 years, intermediate with short-term exclusion of grazing, and least in grazed areas.

Other responses to grazing could have occurred historically in areas with different climates, soils, vegetation types, evolutionary histories of grazing, and types of grazing animals. For example, another common response to grazing is an increase in the number of plant species present in an area. This can result from grazing preventing a few plant species from dominating and excluding others, keeping vegetation open to enable small plants to grow. One consistent effect of grazing is its removal of plant material. Depending on many factors like quality of forage, a mature sheep on western rangelands can eat 1 to 3 kilograms (2 to 6 pounds, excluding moisture) of plant material daily.[7] Cattle can eat 3 to 15 kilograms (6 to 34 pounds) daily. In addition to removing plant material, historical grazing operations built thousands of miles of fences and manipulated streams and springs to provide livestock with water.[1] Many park ecosystems remain altered from a long legacy of historical grazing.

Species Removals

Humans have removed individuals of a species and entire species from landscapes long before parks were established. Commissioned by third president Thomas Jefferson to map a route across the continent, a 33-person, 1805-1806 expedition, led by Meriwether Lewis and William Clark, relied on hunting wild game for food and diligently recorded their animal kills. During the 4,500-km (3,000-mile) northern round trip through what became North Dakota, Montana, Idaho, Oregon, and Washington, the expedition passed near or through lands later designated as Theodore Roosevelt, Yellowstone, and Glacier National Parks, Big Hole National Battlefield, and Lewis and Clark National Historical Park (Fig. 2.5).

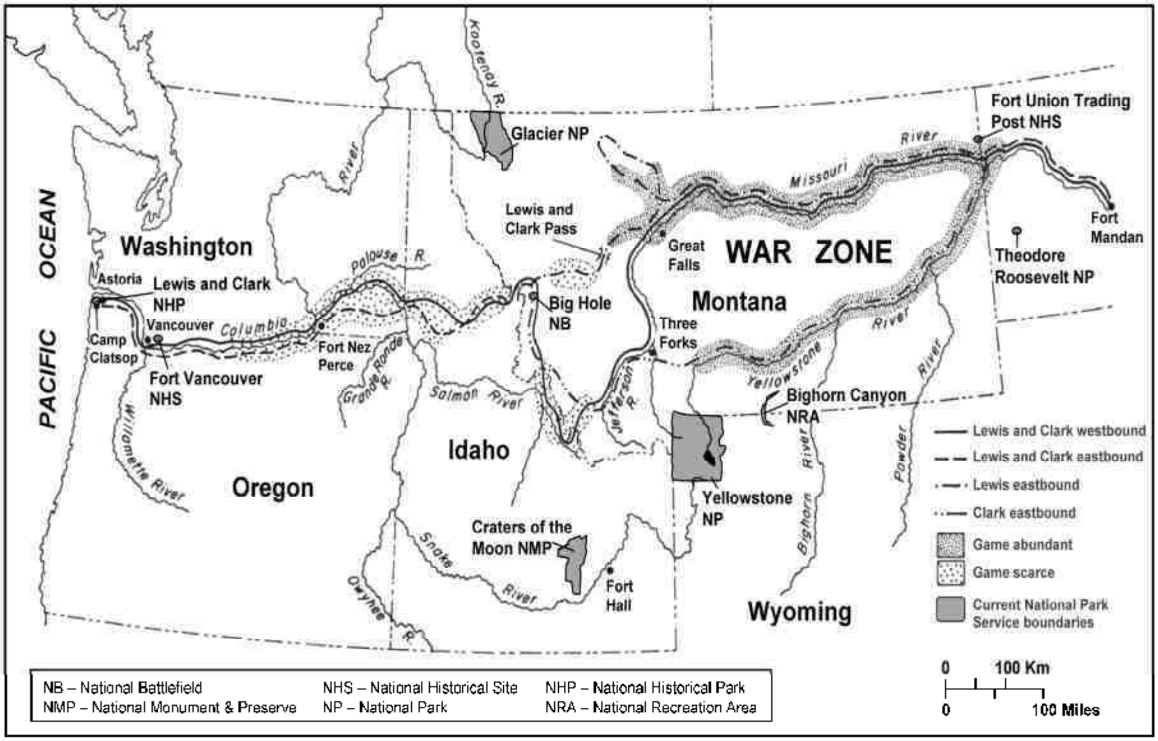


Fig. 2.5. Route of the Lewis and Clark Expedition, 1805-1806. Game animals were abundant in "war zones" separating Native American tribes. Adapted from Martin and Szuter (1999).[22]

In areas densely populated by Native Americans, game was scarce, and the expedition purchased dogs and horses for food. Game was plentiful in "war zones" between Native American tribes, presumably where hunting pressure was reduced.[22] Along the Yellowstone and Missouri Rivers (present-day eastern Montana), for example, the expedition killed 191 deer (*Odocoileus* species), 51 elk (*Cervus elaphus*), 55 bison (*Bison bison*), nine pronghorn (*Antilocapra americana*), and 12 bear (*Ursus* species) in 50 days during summer 1806. In their journals, Lewis and Clark noted that they believed locations of Native American tribes were related to abundance of game.[22] Their own game kills show early manipulation of wildlife 66 years before designation of Yellowstone, the first national park in 1872.

In another example, the passenger pigeon's (*Ectopishes migratorius*) habitat encompassed 2.5 million km^2 (1 million square miles) of deciduous forest in eastern North America. Although this enormous range overlaps with an astonishing 271 national park units, the species has never inhabited any park (Fig. 2.6). Remarkably, the passenger pigeon went from the most abundant bird in the world – with a population estimated at billions in the 1800s – to extinct from Earth in 1914.[6] How did this happen? It is currently believed that the passenger pigeon's demise resulted from hunting and habitat alteration by humans, coinciding with a natural population low cycle that reduced the population below a level needed for sustainability.[17] The species' population size is believed to have long fluctuated in cycles corresponding with good crops of nuts from the trees American chestnut (*Castanea dentata*), oak (*Quercus* species), and American beech (*Fagus grandifolia*). Owing to the passenger pigeon's social behavior and breeding requirements, enormous breeding populations were likely necessary for sustaining the species. During the 1800s, humans killed millions of passenger pigeons by shooting or catching them with nets. Humans also cleared and fragmented the eastern forest by the late 1800s, changing habitat conditions.

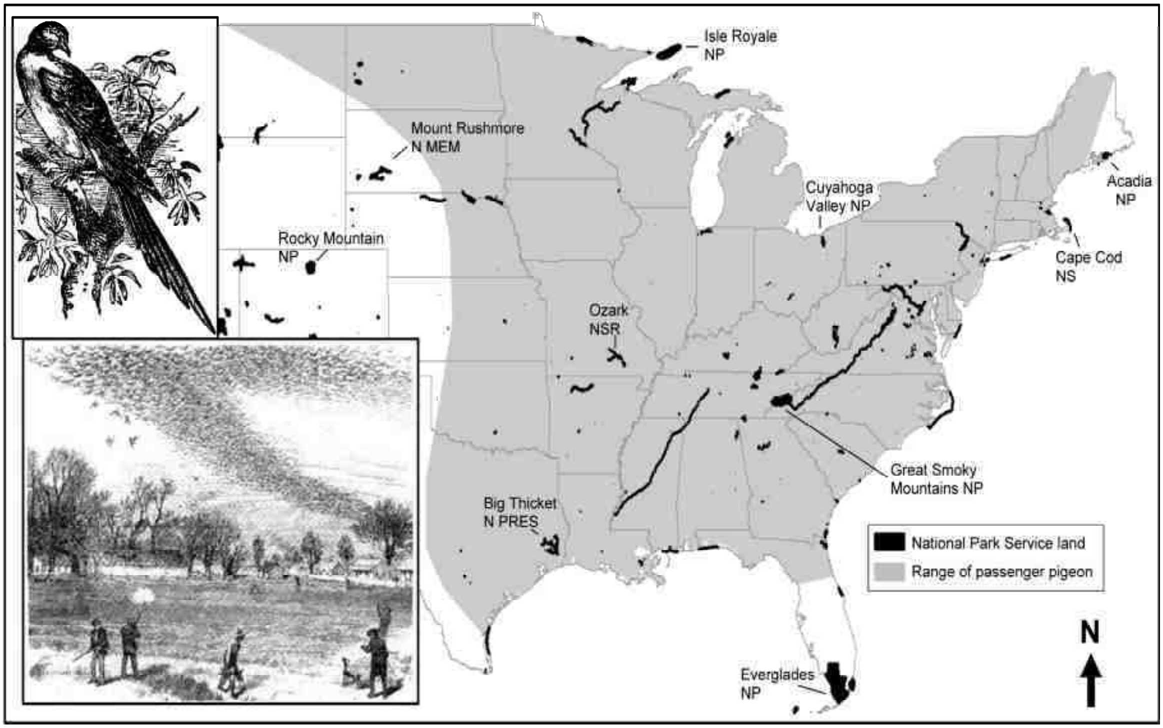


Fig. 2.6. Former range of the extinct passenger pigeon overlaid with contemporary national parks. Upper left drawing depicts a passenger pigeon; bottom left, shooting flocks of passenger pigeons (drawings in public domain). Range map based on Ellsworth and McComb (2003)[11] from earlier sources.

Before its extinction, how might the passenger pigeon have influenced the environment and other species? A typical mature pigeon weighed 300 grams (0.7 pounds) and could fly 60 to 140 km/hour (37 to 87 miles/hour).[6] To meet its energy demand, a passenger pigeon is believed to have consumed a volume of 70 milliliters (one-third of a cup) of food daily, or 16 acorns of red oak (*Quercus rubra*).[11] With a red oak forest producing about 150,000 acorns/hectare (60,000 acorns/acre), a population of 3 billion passenger pigeons could have consumed the total acorn production from the equivalent of 300,000 hectares (700,000 acres) of oak forest each day.[11] In addition to food consumption, roosting by the enormous flocks potentially affected 38,000 km^2 (15,000 square miles) of forest annually. Roosting likely damaged forest canopies by breaking branches and small trees, with effects similar to periodic windstorms that keep forests open. The feces and urine falling from the flocks may have altered the abundance and species composition of forest understory plants by the sheer depth of the deposits and chemical changes in the forest floor.

Passenger pigeons, in conjunction with Native Americans, might also have transported seeds of the nut-producing tree species.[34] American beech and oaks expanded their ranges at 200 to 350 meters/year (700 to 1,100 feet/year) at the end of the last major ice age 11,000 years ago. This is faster than expected for trees with large nuts not dispersed via wind. It is possible that the passenger pigeon's extinction has reduced the capacity of these trees to move in response to future climate change.

The passenger pigeon's extinction is forever, right? Maybe not. The science of de-extinction – bringing extinct species back to life – is no longer just fiction. Extinct animals have already been cloned and brought back to life using DNA preserved in labs. This was done, for example, in 2003 using a goat-hybrid surrogate mother to give birth to an extinct goat, the bucardo (*Capra pyrenaica pyrenaica*), formerly inhabiting the Pyrenees Mountains of France and Spain.[12] The newborn lived 10 minutes. While much more work is needed to develop the techniques, the de-extinction discussion has shifted from *can we* to *should we*.

De-extinction is controversial. The passenger pigeon is one of the species proposed for de-extinction, and preliminary work has already begun.[12] It is not known, however, how many pigeons would need to be "brought back to life" to result in a sustainable wild population. This number could be millions to billions. Where would all these birds be or go? Eastern North America is more urbanized now than in the early 1900s during the extinction, and forests providing nuts have declined overall. In fact, one of the pigeon's main food sources – American chestnut – no longer even exists as a dominant overstory tree because of an introduced blight. Other questions include how reintroducing pigeons could affect other species, particularly other birds now endangered, and whether effort is better used preventing more species from going extinct in the first place. "De-extincting" a few individuals in captivity differs from reestablishing functioning wild populations, especially when the factors causing the species to go extinct have not changed. While no evidence exists that de-extinction can reverse the global problem of species extinctions related to human activities, the future possibilities of de-extinction at the very least are intriguing.

Introduced Species

Introduction of species not native to the United States has quickly and pervasively altered parks. Decimation of the tree American chestnut by the introduced fungus *Cryphonectria parasiticus* (creating chestnut blight) is an example.[19] This Asian pathogen was unintentionally imported to the United States on Asian chestnut seedlings and was discovered in 1904 at the Bronx Zoological Park in New York City. Spreading quickly, the pathogen killed an estimated 4 billion American chestnuts by 1960. There was no control available for the blight and American chestnut had virtually no resistance. In today's forests, living American chestnut root systems remain common, which keep sending up sprouts typically killed before they produce seed. Once a major species from Mississippi to Maine and forming up to half of the entire forest canopy of east-central forests, the American chestnut is now relegated to small-statured stump sprouts.

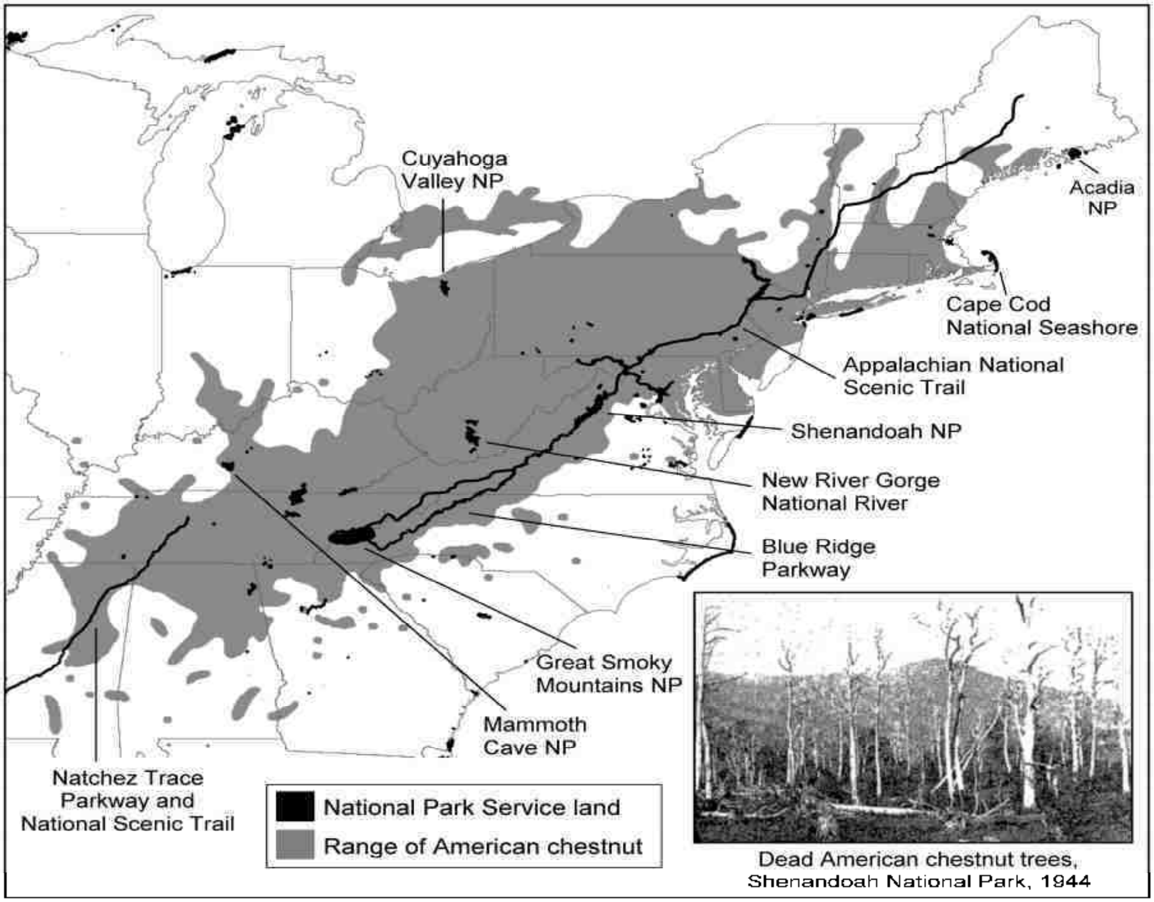


Fig. 2.7. Range of American chestnut in the eastern United States, with some national park units labeled. Bottom right photo courtesy of the National Park Service (Wilhelm 1973).[35]

Just how abundant was American chestnut? About 183 parks, or 46% of the National Park Service's total number of parks, overlap with American chestnut's range (Fig. 2.7). These include many small historical parks, where goals include historical authenticity, and most of the largest eastern parks. Based on trees recorded at property boundaries of land deeds filed between 1824 and 1877 in Edmonson County, Kentucky, where Mammoth Cave National Park was established in 1926, chestnut was the ninth most abundant tree species.[24] Specifically on moderately dry upper slopes, chestnut formed the dominant tree canopy.

A mid-1930s forest inventory in the then newly created (1934) Great Smoky Mountains National Park, in North Carolina/Tennessee, found American chestnut to be even more abundant there at the time the blight arrived.[21] American chestnut occupied 416 (30%) of the 1,378 inventory sites, more than the next most frequent species, chestnut oak (*Quercus prinus*), that occupied 296 sites (21%). American chestnut averaged a density of 33 trees/hectare (13 trees/acre), 12% of the trees of the entire park.

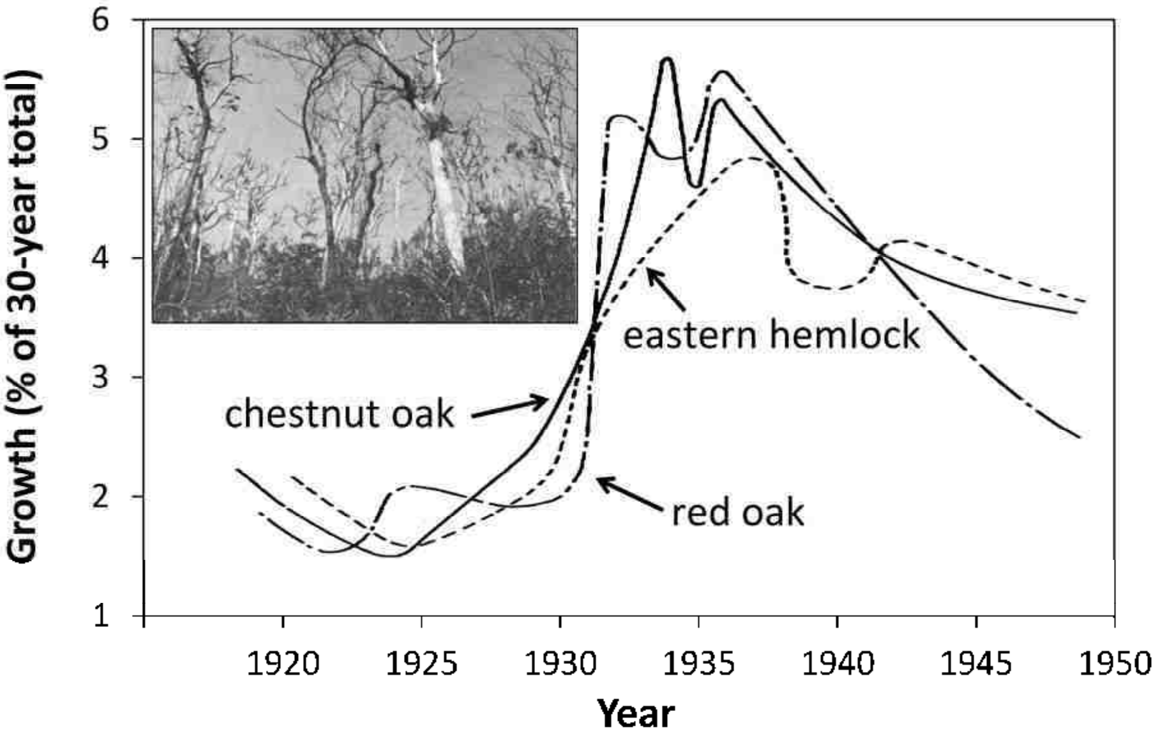


Fig. 2.8. Growth response of neighboring trees to the death of American chestnut trees after arrival of the chestnut blight, around 1930, in Great Smoky Mountains National Park. Data from Woods and Shanks (1959).[36] Photo, courtesy of the National Park Service, shows canopy openings created by chestnut blight.

Decimation of American chestnut transformed composition of the eastern forest. Openings created by the death of American chestnut trees in Great Smoky Mountains National Park were partly filled by increased growth of neighboring trees (Fig. 2.8). Saplings of other tree species, such as red maple (*Acer rubrum*) and sourwood (*Oxydendrum arboreum*), also became established in the openings. A consequence of chestnut blight was the removal of a consistent source of nuts (chestnuts) eaten by wildlife.[8] Replacement tree species were not nut producers, or, in the case of oaks, did not produce nuts (acorns) as reliably among years as American chestnut had. Populations of white-footed mice (*Peromyscus leucopus*), eastern chipmunks (*Tamias striatus*), and gray squirrels (*Sciurus carolinensis*) – small mammals which ate chestnuts – were estimated to have their carrying capacity reduced by 50% from pre-blight levels.[8] While the 80 years since decimation of American chestnut might seem a long time, it is less than the life spans of a single generation of eastern tree species, which can live several hundred years. We must recognize that even if there was no climate change, forests in eastern parks would still be changing from the effects of losing American chestnut before the parks were even established.

The story of the American chestnut is not over, however. Many efforts starting as early as 1909 have sought to develop blight-resistant American chestnut, such as through breeding the few potentially naturally blight-resistant trees, hybrid breeding with resistant Asian chestnut trees, and engineering genetic resistance into American chestnut.[19] For example, by repeatedly breeding the blight-resistant Chinese chestnut (*Castanea mollissima*) and American chestnuts together, a blight-resistant hybrid, still 94% American chestnut, has been developed. Small test plantings of hybrid trees have already occurred, such as at Purdue University experimental sites in central Indiana.[4]

Challenges to reestablishing American chestnut in eastern forests are formidable.[18] Public acceptance of hybrid and genetically engineered organisms is uncertain. Obtaining sufficient blight-resistance seedlings to conduct even small plantings is difficult and costly. Other native tree species now grow where American chestnut formerly did, often limiting potential locations for chestnut plantings to disturbed areas where an existing forest has been removed. Many other introduced pests, ranging from pathogens to gypsy moths (*Lymantria dispar*), now also afflict American chestnut, including trees resistant to the original blight. A recent symposium regarding American chestnut in national parks noted that, while interpretations of National Park Service management guidelines can vary, many aspects of establishing blight-resistant chestnut trees in parks are consistent with reestablishing natural forests.[9] Despite the obstacles, working to rectify the pre-park demise of an iconic species is an exciting prospect.

Alteration of Fire Occurrence

Fires shape the development of vegetation and associated wildlife habitat. When humans influence fires, humans directly mold ecosystems. Native Americans are believed to have long used fire across the landscape to communicate, clear forest for crops, maintain open habitat, and promote desired tree species such as oaks for food.[2] Early Euro-American settlement frequently continued introducing fire to eastern forests.

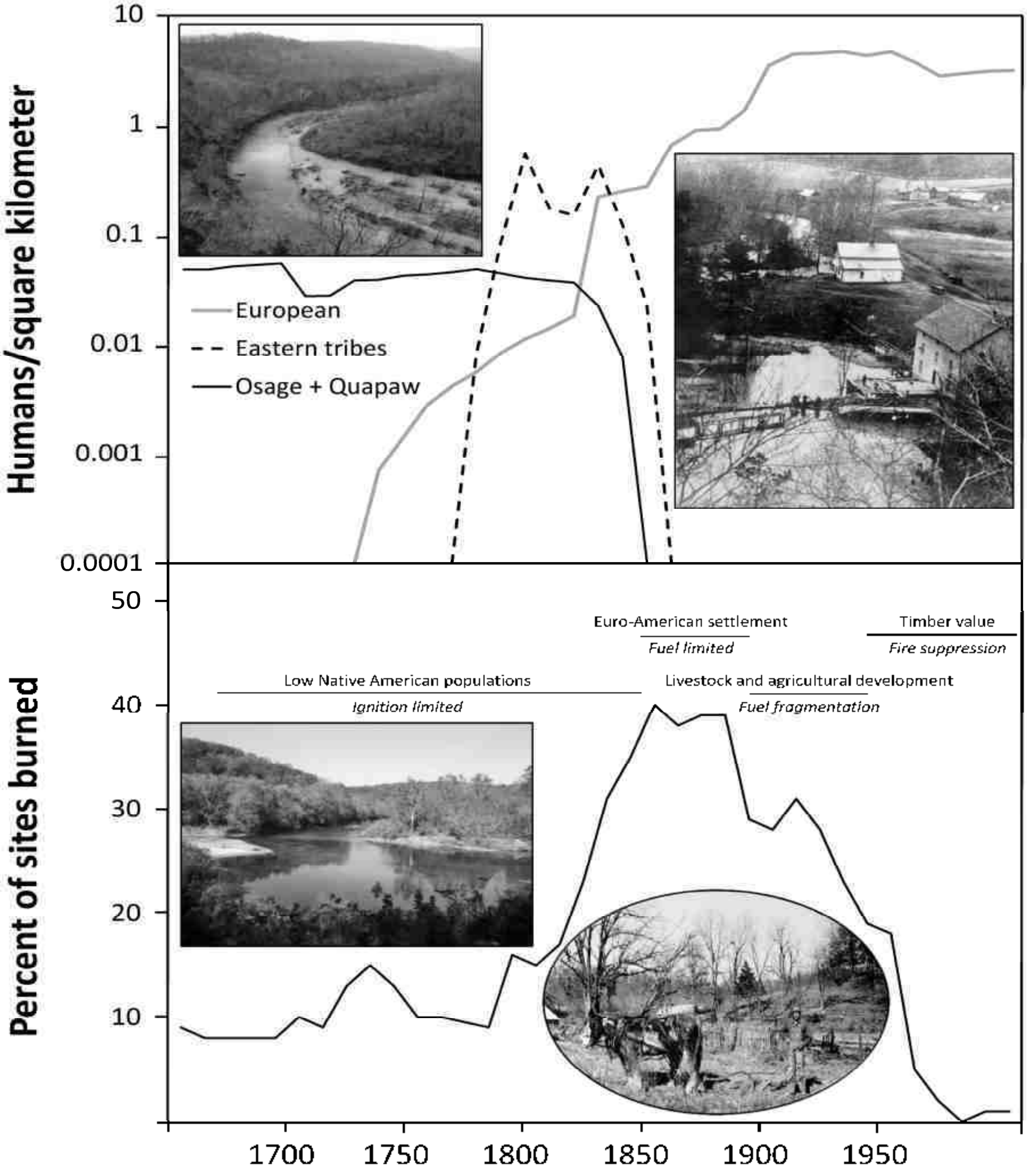


Fig. 2.9. Human population trends and fire frequency from 1650 to 1990 in Ozark National Scenic Riverways, Missouri. Data from Guyette et al. (2002).[16] *Left photos: contemporary views of the Jacks Fork (top) and confluence of the Jacks Fork and Current Rivers (bottom). Right photos: early 1900s views of Alley Mill (top) and plowing a field at Nichols farm (bottom). Photos from the National Park Service.*

In the Ozark Mountains of southern Missouri, included in Ozark National Scenic Riverways, humans have sculpted fire patterns for centuries.[16] Few lightning-ignited fires occur in this area, so nearly all fires are of human origin. Native American populations were low during the early and mid-1700s, owing to warfare between tribes, Euro-American disease, and westward migration. Still, 10% of the area burned annually (Fig. 2.9). Westward migration of eastern tribes, including the Cherokee, Delaware, and Shawnee, correlated with increased fire frequency in the late 1700s and early 1800s. Fire frequency was even higher – with 35% of areas burning annually – during the rest of the 1800s dominated by Euro-American settlement. There were likely so many ignitions during this period that wildfires were not limited by ignition sources, but rather by vegetation re-growth to provide fuel. Prevalence of fire began declining in the early 1900s during agricultural development, which reduced the continuity of fuel and the ability of fire to spread across the landscape. Thereafter, fire virtually ceased during an era of fire exclusion, when any human or lightning ignitions were extinguished as quickly as possible. Excluding all wildfires was the dominant wildfire management policy in the United States during the 1900s.

The legacy of human fires creates an interesting dilemma for current forest conservation.[20] Forests of oak and shortleaf pine (*Pinus echinata*), promoted by human fires for centuries, are replaced by other tree species better adapted to today's fire-free environment. Yet, the fire-dependent oak and pine forests are currently valued for their wildlife habitat and may be among the forest types best adapted to potentially more severe droughts or more variable weather during climate change. Current management could allow continuation of the trend for oak-pine forests to be replaced by other species. Or, reintroducing fire or clearing forest patches could promote oak-pine forest.

In another example, Mount Rushmore National Memorial was established in 1938 after the completion of a famous sculpture of four U.S. presidents carved into a granite cliff in the Black Hills of South Dakota. The ponderosa pine (*Pinus ponderosa*) forest of the 517-hectare (1,278-acre) park surrounding the sculpture had at least one spreading fire on average every 16 years between 1530 and 1890.[5] Here, lightning is common, although ignitions by the Sioux or other tribes could have augmented lightning fires. Unlike in the Missouri Ozarks, the frequency of spreading fires abruptly declined with Euro-American settlement beginning in the 1880s. Introduction of livestock may have quickly reduced grasses and other fuels and was followed by the 1900s policy of fire exclusion.

Cessation of fire beginning 50 years before the park was established has changed the forest around Mount Rushmore. Ponderosa pine, alleviated from having its seedling establishment kept in check by frequent fires, increased from 280 trees/hectare (113 trees/acre) in 1870 to 1,300 trees/hectare (526 trees/acre) in 2005.[5] This shifted the fuel conditions away from ones that could only support surface fires burning along the ground and leaving large trees alive, to those that could support severe fires capable of burning through interconnected tree canopies and killing large trees.

As detailed in Chapter 3, fuel buildup in the absence of natural fires is pervasive in many parks. While the large, severe wildfires increasingly occurring in western forests might be exacerbated by the generally warm, dry climate during the 2000s, without fuel there is no fire. We cannot solely blame climate change, because it is actually fuel built up during the legacy of fire exclusion that is fueling many of today's severe forest fires.

Significance of Events Before Park Designation

Only 10 (2%) of the 408 parks currently in the national park system were established by 1905. The United States had existed for 130 years by then. Euro-American exploration and settlement had already occurred in some areas for centuries.

Land use before park establishment does not necessarily detract from value of current parks. In fact, it may be heartening that a park such as Cape Cod National Seashore could be viewed as "more natural" today than the area was in the 1800s. The park has more area in woodland now than in 1850 at the peak of clearing for agriculture and pasture.[10] Similarly, recovering desert shrubland now occupies former roadways and townsites constructed during early 1900s mining booms before Death Valley National Park was established in 1934.[33] Initial Euro-American settlement may even have accentuated Native American practices that promoted development of some valued ecosystems, like oak forests.[2] In other cases, park ecosystems are just different than they would be had previous use not occurred. Although most of Great Smoky Mountains National Park was logged and American chestnut was lost, second-growth forests now occupy the park. Trees are still small, but the park's current forest is one of the most species-rich forests in the entire country.

Some historical legacies, however, degrade parks and continue creating challenges for park conservation. Former agricultural fields and pastures in Olympic National Park, Washington, and Grand Teton National Park, Wyoming, are dominated by non-native plants rather than native vegetation.[28,23] Excluding fire for over a century from western dry forests has resulted in many parks facing deforestation from severe fire, unless forest thinning or careful prescribed fires are implemented to reduce fuel. Native species removals or extinctions have changed the ecology of parks, such as elimination of top predators, resulting in unnatural irruptions of prey animals. Fortunately, potential exists to reestablish natural processes such as fire, reintroduce native species, and reverse degradation.

Pre-park legacies will probably remain important in park conservation because many legacies have not stopped. As Chapter 7 describes, for example, chestnut blight was just one of many damaging forest pests that continue to be introduced via global commerce. We also need to be careful about automatically ascribing all of today's ecological changes to climate change. Instead, shifting ecosystems likely at least partly stem from previous or ongoing land-use legacies. Moreover, when new parks are added to the national park system, they will be starting from a baseline including both historical and more recent land uses. The longevities of the effects of past land uses further indicate how effects of management activities (including no action) done today can persist far into the future.

3 FIRE

Fire has shaped the evolution of most park ecosystems, and so has recent alteration to fire regimes. A fire regime includes the frequency, spatial pattern, seasonality, and severity of fire occurrence.[33] Frequency is how often a particular area burns and is called a fire interval. Spatial pattern describes the size of fires and where on the landscape fires occur. Seasonality is when during the year fires happen. Severity is the relative effect a fire has, defined using measurements like the proportion of trees killed or the amount of organic matter burned in the soil. Severity is a function both of the intensity of a fire (heat output and duration) and the susceptibility of an ecosystem to fire.

Fire regimes are also broadly classified as ground fires (burning along the soil surface, not into the tops of tall trees or shrubs) or stand-replacing fires. Stand-replacing fires remove an existing patch of trees or other vegetation. These fires are also termed crown fires when they burn the tops of trees in forests, tall shrubs in shrublands, and grasses in grasslands. Some ecosystems are clearly dominated by either ground or crown fires. Other ecosystems sustain mixtures of both types, sometimes even within the same fire. Burning along the ground or in crowns within the same fire depends on variation in fuels that the flames encounter, topography, and weather conditions as fires move across landscapes.

Different parks have various historical fire regimes shaping the development of species and ecosystems. For example, ponderosa pine (*Pinus ponderosa*) forests in Grand Canyon National Park, Arizona, predominately had frequent, low-severity fires (leaving most large trees alive) during summer for millennia before the late 1800s.[13] In contrast, southern California shrubland of Santa Monica Mountains National Recreation Area had periodic, severe fires in fall during Santa Ana winds.[38] When reporting contemporary wildfires, the national news media rarely or never mentions how behavior of a current fire compares to behavior of historical fires in an ecosystem. Such information is essential for evaluating whether a current fire truly is "natural." Periodic (though not too frequent) crown fire in shrubland of the Santa Monica Mountains is natural, whereas a crown fire in ponderosa pine forest at the Grand Canyon is unnatural and destructive.

Based on fire frequency and severity, generalized historical fire regimes have been mapped across the United States (Fig. 3.1). Parks in the northern Great Lakes region and Northeast, such as the northern hardwoods and boreal (also called taiga) forest of Maine's Acadia National Park, had infrequent, stand-replacing fires. Much of the rest of the eastern forest – dominated by oaks, American chestnut (*Castanea dentata*), and pines – generally had frequent, low-severity fires that usually did not kill large trees. Great Smoky Mountains National Park is an example. Southern Florida parks generally had frequent, severe fires. Fire regimes in the West were highly variable. Infrequent, stand-replacing fires typified Yellowstone National Park. Glacier National Park supported a mixture of frequent ground fires and infrequent stand-replacing fires. Desert parks, such as Death Valley, are thought to have had infrequent fire, owing to little production of fuel in the dry environment.

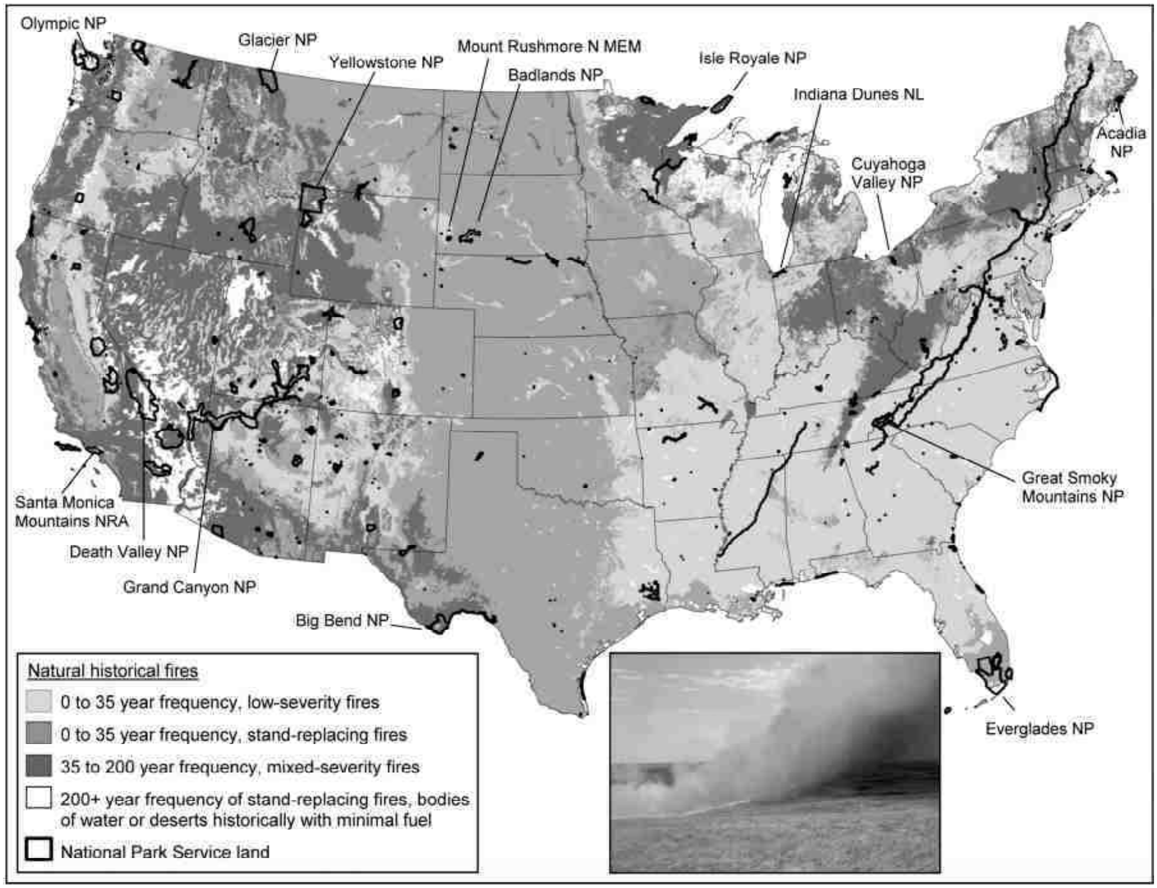


Fig. 3.1. Generalized historical fire regimes of the United States, with some example parks mapped. Source data from Schmidt et al. (2002).[28] Photo: prairie fire, Badlands National Park, South Dakota (M. Carlbom, National Park Service).

Both historical and present fire regimes result from three main factors: fuel characteristics, ignitions, and climate. The greater the amount of fuel, generally the greater the severity of fire. Likewise, the greater the continuity of fuel across the landscape, generally the more readily fire spreads. Main sources of ignitions are lightning and humans, but lava (such as in Hawaii Volcanoes National Park) and other sources do ignite fires. Increasing ignitions can increase fire frequency, but this depends on whether ignitions coincide with fuel and weather conditions conducive to fire spread. Climate shapes fire regimes through its effect on vegetation as fuel, ignitions (including storms producing lightning), and shorter term weather like dry periods.

Changes in even one of these main factors (fuel, ignitions, or climate) can dramatically change fire regimes. Changes in multiple factors, like increased fuel loads coinciding with increased ignitions or warmer temperatures, can synergistically alter fire and its effects.

The United States has had a mercurial relationship with wildfire, which has affected national parks. Early Euro-American settlement accentuated burning by Native Americans in certain areas and disrupted it in others. Widespread logging in the late 1800s and early 1900s was associated with large, severe fires (Fig. 3.2). Slash – branches and unwanted debris left after logging – altered fuel conditions and coincided with lightning or human ignitions and dry summers. In eastern Wisconsin, the October 1871 Peshtigo Fire scorched 0.5 million hectares (1.2 million acres), twice the size of Rhode Island. This fire killed 1,500 people and remains the deadliest fire in U.S. history. Also in 1871, the Port Huron Fire burned 0.5 million hectares (1.2 million acres) in eastern Michigan, near Flint and Detroit.

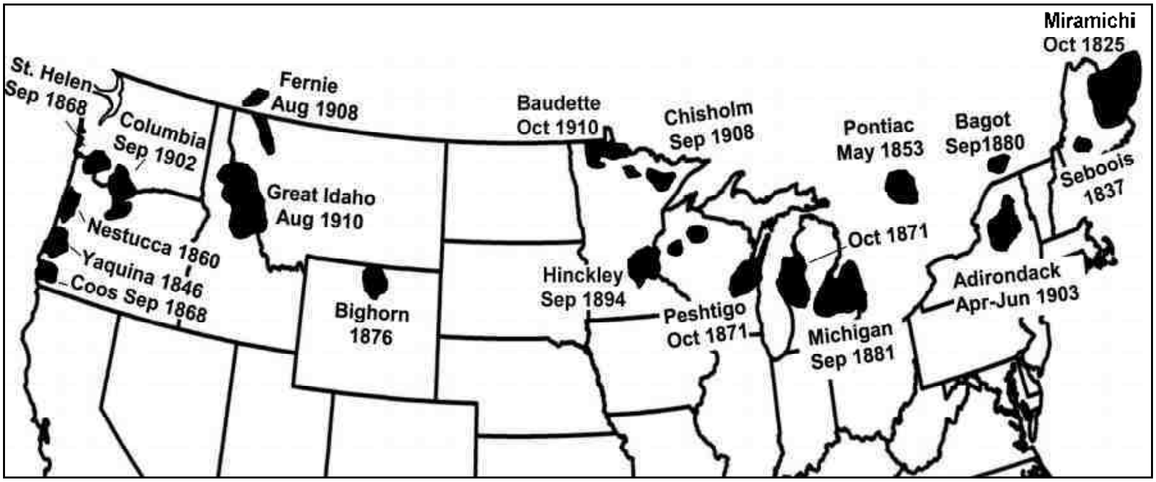


Fig. 3.2. Large fires between 1825 and 1910. Adapted from Plummer (1912).[24]

Severe fire was viewed as problematic by the late 1800s, but there was disagreement regarding what to do about it. This was exemplified by a meeting in 1890 of foresters Gifford Pinchot and Bernhard Fernow with Secretary of the Interior John Noble to advocate a "forest protection" policy to exclude fire completely.[25] This view partly stemmed from forestry perspectives in Europe that fires were destructive and even "uncivilized." At the meeting, however, famous explorer John Wesley Powell countered the view that all fire was bad. Instead, he advocated a policy of light burning (ground fires) for many forests to keep fuel loads low and render forests relatively immune to crown fires.

The fire season of 1910 changed the debate. The Great Idaho Fire, together with a conglomerate of several thousand smaller fires collectively named the Great Fire, burned an area the size of Connecticut (1.2 million hectares, 3 million acres) in the northern Rocky Mountains. The fires started from sparks via railroads, other machines, and lightning, coinciding with dry weather and ample fuel partly from post-logging slash. Burning across northeastern Washington, Idaho, and western Montana, the fires destroyed seven towns. With 78 firefighters killed,[25] the Great Fire was the deadliest event for firefighters in U.S. history until the September 11, 2001 attack on the World Trade Center.

In this historical context, it may not be surprising that after 1910 the United States entered a policy era of extinguishing any lightning or human ignitions on public lands. Many ecologists and some foresters continued pointing out benefits of having natural fires in ecosystems, but fire suppression remained the exclusive policy for fire management on public lands for most of the 1900s. By 1910, livestock grazing (by reducing fuel) and other land uses had already disrupted fire regimes for over 30 years in some western forests. This was subsequently reinforced by the fire exclusion policy. When the National Park Service was formed as a federal agency in 1916, it too adopted the policy of fire suppression.

Effectiveness and effects of fire suppression on public lands have varied across the country. In Santa Monica Mountains National Recreation Area, for example, fires continued in southern California chaparral during the 1900s despite attempted fire suppression.[20] In other cases, fire exclusion may not yet have affected parks that had historical regimes of infrequent, stand-replacing fires. Woodlands of pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) in Glen Canyon National Recreation Area (southeastern Utah/northern Arizona), that naturally go centuries between stand-replacing fires, still seem minimally influenced by fire exclusion.[11] In many formerly frequent-fire eastern forests, fire has been absent since the early 1900s. This partly stemmed from fire suppression and partly from curtailing intentional forest burning by humans.

Fig. 3.3. Change in forest openness in Yosemite Valley, California. Photos from the National Park Service.

Fire suppression in western dry forests (such as those with ponderosa pine) has been highly effective, at least until recently, and its legacy effects continue. These forests, formerly with frequent, low-severity fire, have responded to fire exclusion by accumulating hazardous fuel loads that burn in stand-replacing fires (Fig. 3.3). This is generally a different response than in the East. Eastern forests partly respond to fire exclusion by actually becoming less flammable, via changes in the species composition of the vegetation and the fuels.

The United States still has an uncomfortable relationship with wildfire that varies from park to park. Ironically, continued fire suppression in western dry forests is now important to avoid undesirable effects of severe fire, until fuels can be safely reduced. Fire suppression also is needed to reestablish natural fire regimes in other parks, such as Santa Monica Mountains National Recreation Area, where current fire frequencies appear unnaturally high because of increased ignitions via humans.[38] Maintaining uncontrollable stand-replacing fire in parks where it is natural is a major challenge. Consider the urbanization around parks, park infrastructure like visitor centers, and the safety of millions of human visitors in parks.

Understanding historical fire regimes is a first step for conserving species and ecosystems that have evolved with the fires. As a result, fire history studies have been conducted in several parks. Researchers use four main techniques to reconstruct past fires for comparison with the present: 1) fire scars on trees; 2) charcoal deposited in the soil from past fires; 3) ages of woody plants; and 4) written, oral, or photographic documentation.

In forests with ground fires, the lower trunk of some trees can be scarred by a passing fire.[9] A fire creates an initial wound in the wood, which the tree heals around. Subsequent fires enlarge the wound, creating a blackened burn-out area in the wood, termed a fire scar (Fig. 3.4). Fire scars occur on ponderosa pine in western parks, pitch pine (*Pinus rigida*) and other pines in eastern parks, and on some other (but not all) tree species.

Black marks signifying a fire can be dated from the tree rings of fire scar samples. Fig. 3.5 shows a cross-section of a fire scar from a ponderosa pine in Mount Rushmore National Memorial, South Dakota.[4] This tree grew from a seed that germinated in 1474. During its 442-year life, the tree recorded 10 fires that burned its trunk (but that it survived) and created the fire scar. These fires occurred between 1652 and 1890, for an average of one fire every 24 years. Fires abruptly ceased after 1890, coinciding with livestock grazing and the subsequent policy of fire exclusion. Because not all fires leave scars on every tree, analyzing fire scars is not a perfect method for reconstructing fire history. Nevertheless, fire scars are one of the most accurate techniques for reconstructing ground-fire history, enabling remarkable insights for many forests.[9]

Fig. 3.4. Fire scar on the lower trunk of a pitch pine, Great Smoky Mountains National Park. A sample was removed on the right to date fires. Photo courtesy of R. Klein, National Park Service.

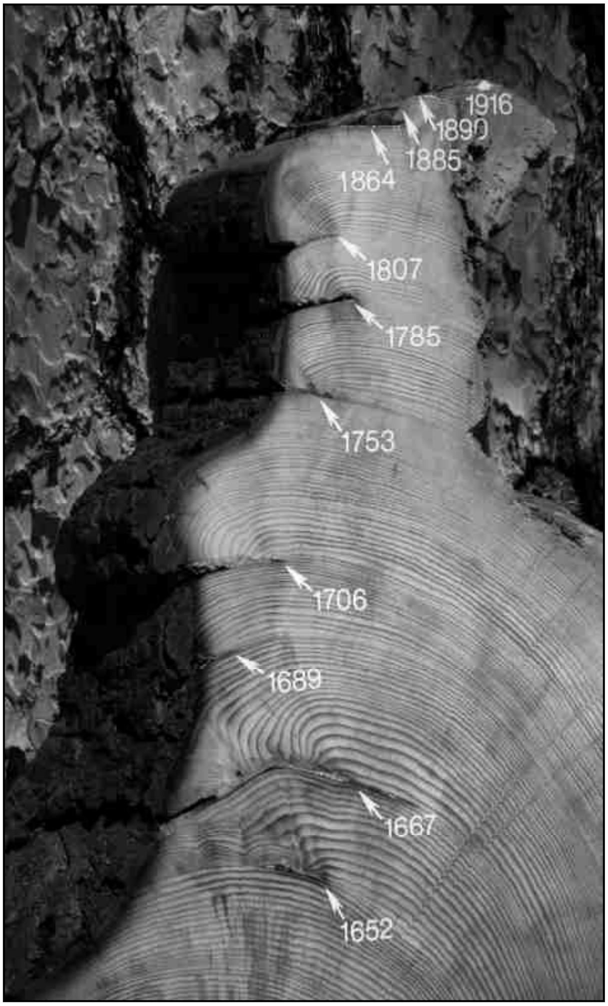


Fig. 3.5. Sanded and dated fire scar sample from a ponderosa pine, Mount Rushmore National Memorial, South Dakota. Years indicate recorded fires. Near the outer bark, 1916 was when the tree died, at age 442. Photo by and used with permission from P.M. Brown, Rocky Mountain Tree-Ring Research.

Charcoal deposited in the soil from burning of organic matter can be dated using radiocarbon techniques to estimate fire occurrence. Movement or loss of charcoal from the soil is a disadvantage of the technique, but it is useful for estimating fire history without fire-scarred trees. Likewise, if used carefully, the ages of shrubs or trees can indicate past fire. If the ages of individuals in a shrubland or forest are relatively uniform, this can indicate that some event removed the previous vegetation and allowed a new cohort of individuals to become established at the same time. Analysis of age patterns is particularly useful to identify stand-replacing fire when it is relatively certain that it was fire (and not some other type of event) that removed the previous vegetation.

Based on historical fire regimes and departure of modern conditions from them, fire management varies from park to park. The next sections illustrate the diverse roles of fire in shaping the evolution of park ecosystems, effects of recent alteration to fire regimes, and park projects restoring fire as a natural process.

First Burn in the Everglades

When thinking of where the first intentional reintroduction of fire to a national park occurred, a regional wetland might not be the first location that comes to mind. Everglades National Park conducted the first prescribed burn in National Park Service history in 1958.[22] This was 48 years into the general policy of fire exclusion and was the first reintroduction of fire as a natural process. The 1958 burn was near the Pineland Trail, in the east-central part of the park. This area contains pine rockland forest on exposed limestone, slightly elevated from surrounding wetland prairies. In only a decade without fire, other tree species begin replacing overstories of slash pine (*Pinus elliottii*). The pine forest, however, is valued for its high plant diversity and unique habitat.

Following the 1958 burn, Everglades National Park's prescribed burning program was off and running. From 1958 to 1971, the park conducted 102 prescribed fires covering 11,300 hectares (28,000 acres).[22] Then, from 1972 to 1974, 124 prescribed fires burned 10,900 hectares (27,000 acres). Also during this time, some lightning ignitions were allowed to continue burning. From the inception of the park in 1947 through 1999, at least 1,600 fires burned over 560,000 hectares (1.4 million acres).[30] This total includes some areas burned multiple times. A given site in the park burned, on average, every 23 years, between 1947 and 1999. In addition to the pine rocklands, wetlands and coastal prairies regularly burned when low fuel moisture promoted fire spread (Fig. 3.6).

Fig. 3.6. Everglades National Park: Top left: prescribed burning sawgrass over surface water using an airboat for ignition and control in 1975. Top right: ignited via helicopter, prescribed fires in sawgrass and wet prairie. Bottom left: fire-dependent pine rocklands, Long Pine Key Nature Trail. Big Cypress National Preserve: Bottom right: prescribed burning using drip torches. Top photos from Wade et al. (1980),[37] and bottom photos from the National Park Service (bottom right by C. Derman).

Everglades National Park and adjacent Big Cypress National Preserve continue having among the most active fire programs in the National Park Service. Over 70,000 hectares (175,000 acres) of prescribed or managed wildfires (allowed to spread or even augmented via prescribed fire) burned in 2011 and 2012 in Everglades National Park. In Big Cypress from 2008 to 2013, prescribed fires burned between 8,000 and 33,000 hectares (20,000 and 83,000 acres) each year.

An important ecological function of these fires is keeping the vegetation open. This function might have effects as far reaching as facilitating water flow through the Everglades and benefitting the endangered Florida panther (*Puma concolor coryi*). Fewer than 100 Florida panthers are estimated to exist in southern Florida wildlands today, after habitat alteration and extensive hunting (with bounties even placed on panthers) reduced their numbers in southern states from the 1800s to mid-1900s. Sightings of 17 Florida panthers in Big Cypress National Preserve during the 1990s were most frequent in pinelands burned within the previous year.[8] Panthers hunting prey animals attracted to re-growing vegetation might partly account for the association between panthers and recent burns.

Fire-Dependent Savanna and Prairie at Indiana Dunes National Lakeshore

A mosaic of oak savanna with prairie occupied the U.S. Midwest in a transition between deciduous forest to the east and prairie to the west. The Midwestern savanna region extended south to north from Texas to Minnesota and neighboring states east and west. With agricultural development, urbanization, and the absence of fire, less than 1% of the once vast ecosystem remained in any type of semi-natural state by 1985.[23]

Midwestern savannas depend on fire for their existence. The region's climate is capable of supporting deciduous forests, including non-oak species such as red maple (*Acer rubrum*) and black cherry (*Prunus serotina*). In as few as 10 years without fire, open savannas and prairies incur infilling by dense canopies of oaks and less fire-tolerant species such as red maple. The dense tree canopies quickly shade out plants requiring lots of sunlight.[1] Native Americans, and later, Euro-American settlers, are believed to have ignited many of the fires historically keeping the vegetation open. Genesis of the savanna ecosystem is considered intricately linked to humans for millennia. The biotic assemblages that evolved under this human influence are diverse and priorities for conservation.

Southeast of Chicago along Lake Michigan's southern shoreline, the 4,266-hectare (10,541-acre) Indiana Dunes National Lakeshore is in the savanna region. Unlike many other areas, the lakeshore continued having fires through the mid-1900s. In the north-central part of the lakeshore, for example, Howes Prairie burned 11 times between 1900 and 1972, or one fire every 6.5 years.[7] After 1972, fire suppression was more vigilant, and vegetation changed quickly. In 1986, the National Park Service initiated prescribed burns to conserve open habitats before forest developed with trees too large to be readily killed by fire. Because the lakeshore is interspersed with adjacent urban developments, these burns required careful planning to avoid dispersing smoke into urban neighborhoods.

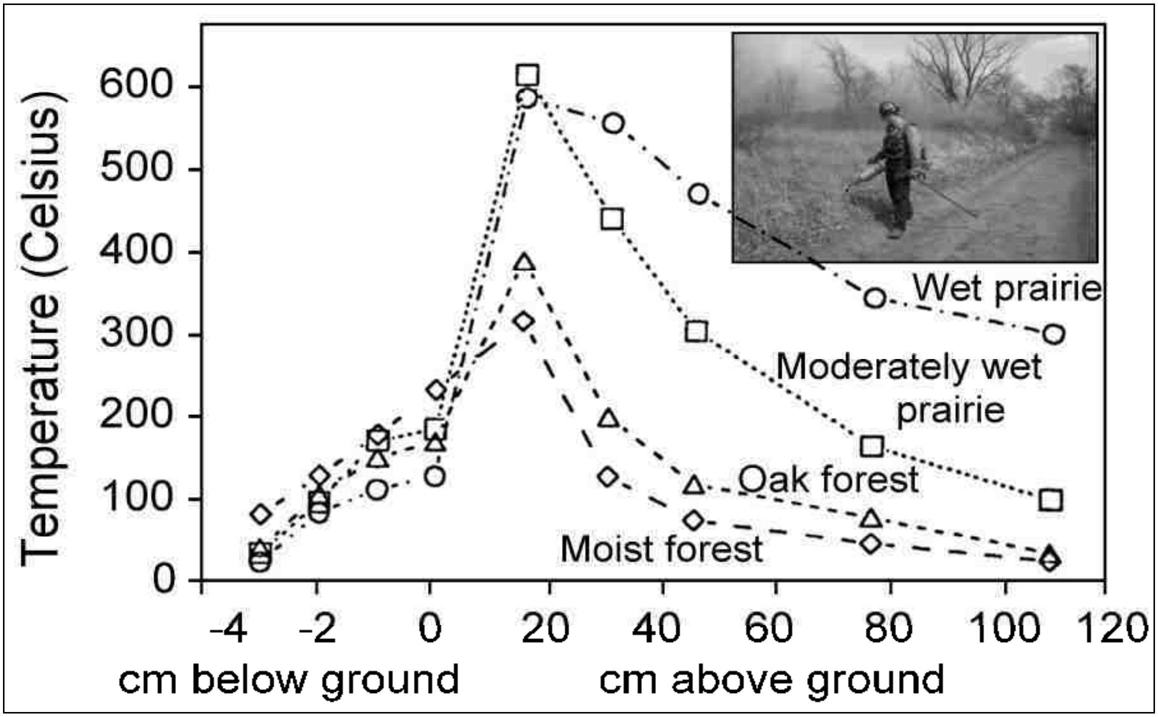


Fig. 3.7. Temperatures below ground in soil and at different heights (measured in centimeters) above ground during prescribed burning in vegetation types of Indiana Dunes National Lakeshore. Data from Cole et al. (1992).[6] Photo, from the National Park Service, shows igniting a prescribed fire in the lakeshore.

Using aluminum tags with paints that melt at different temperatures, scientists measured temperatures at several locations during 1986 and 1988 prescribed burns at Howes Prairie (Fig. 3.7). The soil was well insulated during fire, and temperatures did not always reach 100 degrees Celsius (212 degrees Fahrenheit). Temperatures peaked 20 centimeters (8 inches) above the ground and were greatest in prairies with the most fuel, reaching 600 degrees Celsius (1,100 degrees Fahrenheit). Temperatures of at least 180 degrees Celsius (350 degrees Fahrenheit) were required to kill the aboveground stems of tree saplings up to 5 centimeters (2 inches) in stem diameter (root systems often survived insulated in the soil). Only the hottest temperatures could kill the largest stems. Multiple burns achieved the goal of reducing the dense tree canopies that grew during the fire-free period (Fig. 3.8).

The mosaics of prairies and savannas, combined with denser oak woodlands forming with less frequent fire, increase wildlife diversity. Different groups of bird species were related to different parts of the gradient from open prairies to forests that dynamically occur in different areas and during different time periods in the same area.[15] Food specialization partly related to where birds occurred. Ground-feeding birds inhabited prairies, while birds specializing in bark insects preferred densely wooded areas. Each habitat type provided unique features important to conserving the park's total diversity of birds.

Bees, currently of major conservation concern around the world, were also most abundant in open prairie-savanna habitat and in frequently burned areas.[16] This might relate to open conditions promoting nectar plants. Researchers recorded 4,631 individual bees of 170 species at their sample sites across the prairie and forest habitats of the lakeshore.

These examples illustrated a dynamic interplay among fire frequency, the resulting mosaic of vegetation types, and different groups of wildlife species. Taking no management action reduced habitat diversity by extensively converting open habitat to dense forest. Active management, via reintroducing fire, promoted species diversity.

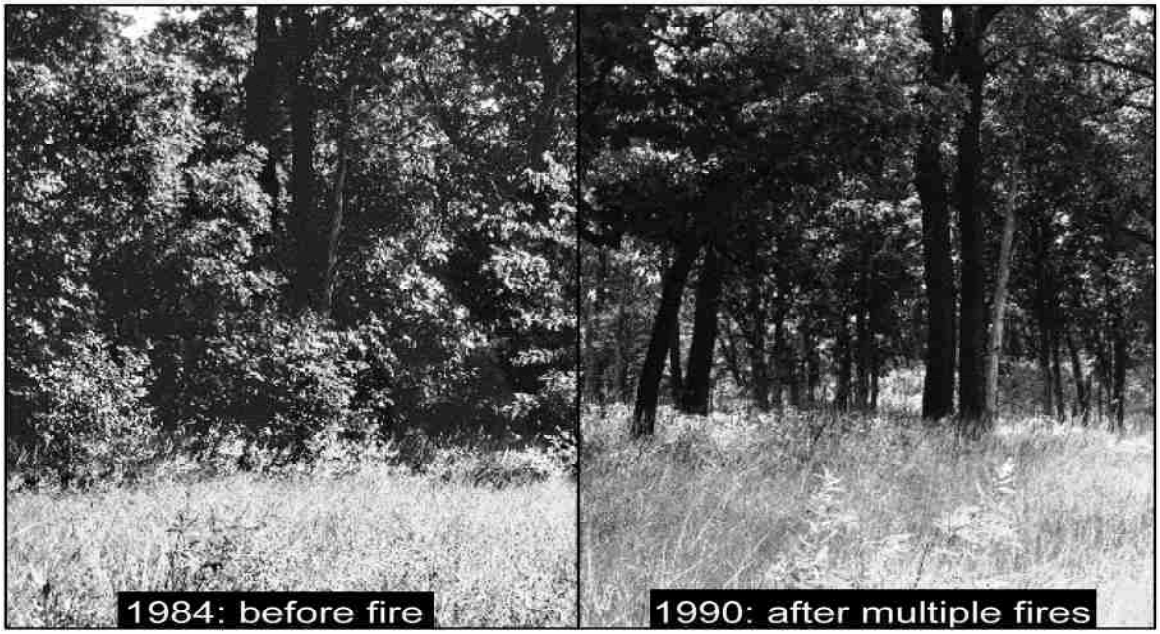


Fig. 3.8. Using prescribed fire to convert dense forest to more open oak savanna and woodland, Indiana Dunes National Lakeshore. Photos by K.L. Cole (U.S. Geological Survey).

Frequent Fire at Grand Canyon and Giant Sequoia National Parks

Western forests of ponderosa pine and mixtures of other conifer tree species ("mixed conifer forest") on relatively dry sites at mid-elevations are currently the most challenging forests for fire management in the lower 48 states. These forests inhabit elevations above more arid vegetation (including pinyon-juniper woodlands) and below other forests, such as spruce-fir, that occupy the highest mountain peaks.[3] Covering an enormous area in the West (25 million hectares, or 60 million acres), ponderosa pine-mixed conifer forests occur from the Rocky Mountains west to coastal mountains inland from the Pacific Ocean.[29]

Many ponderosa pine and dry mixed conifer forests historically had mostly ground fires occurring frequently, every few years to decades. Certain forests may have had some crown fire, combined with ground fire. Regardless of the specific historical fire regime, most ponderosa pine and dry mixed conifer forests have had no fire since at least the early 1900s. In the absence of reoccurring fires, the number of small trees and quantities of fuel (pine needles, dead wood, and other woody debris) have increased dramatically.[36] These elevated fuel loads have converted much of the West to a stand-replacing fire regime, incongruent with a historical regime of low-severity ground fires in many areas.

Containing vast ponderosa pine and mixed conifer forests within its 0.5 million hectares (1.2 million acres), Grand Canyon National Park in northern Arizona exemplifies conservation needs of these forests. On the South Rim near Grandview Point, analyzing fire-scarred ponderosa pines revealed that 26 ground fires burned between 1679 and 1887 across an 800-hectare (2,000-acre) site.[13] These fires scarred at least 10% of trees and occurred every seven years, on average, during the 208-year period. Spreading fires abruptly ceased after 1887. Over the next 110 years to 1997, the forest "missed" 16 natural fires that would have occurred if the historical frequency of fire had continued.

After 1887 and through the 1900s, the density of trees increased sevenfold.[12] In addition to being at risk of deforestation from stand-replacing fire, the contemporary forest supported minimal understory plant and insect diversity. It is interesting how different a forest the 4 million visitors annually to the Grand Canyon see today, compared to the forest there under frequent fire in 1887.

Can't fire simply be reintroduced to these forests using prescribed burning or allowing lightning fires to burn? In some cases, yes, such as where there had been a few fires during the 1900s, reducing fuel accumulation. In many other cases, however, the fuel loads are now so hazardous that reintroducing fire is dangerous – both to old trees of conservation priority and to human visitors and structures.

Ironically, cutting trees can be central to forest conservation and an important step for safely reestablishing ground fires as natural processes in South Rim ponderosa pine forests. One of the initial restoration tree thinning projects on the South Rim, proposed by the National Park Service in the late 1990s, met public opposition to cutting trees in a national park.[14] The project was delayed, the main focus on ecological restoration abandoned, and a revised project was proposed with an objective simply to reduce risk to park visitors from hazardous fuels. Three treatments in the revised project were implemented: 1) a minimal tree thinning where no trees greater than 13 centimeters (5 inches) in stem diameter were cut, which was followed by a prescribed burn; 2) a prescribed burn without tree cutting; and 3) an untreated control. Tree thinning was performed using chain saws in 2002. Due to the hazardous fuel conditions, fire managers conducted prescribed burning in November 2003 under weather conditions capable of supporting only low-intensity fire. During the burns, winds were light, the maximum air temperature was 8 degrees Celsius (46 degrees Fahrenheit), and lengths of flames were only 10 to 30 centimeters (4 to 12 inches).

The treatments only slightly affected forest structure and crown-fire risk.[14] Thinning plus burning reduced the number of trees by 45%. But, because only the smallest saplings were allowed to be cut, the number of trees remaining was still six times greater than in 1887 before fire exclusion. The prescribed burn (with no tree cutting) reduced the number of trees by only 23%, and seven times more trees remained than in 1887. Burning did reduce thickness of the layer of needles and debris on the forest floor by 2 centimeters (1 inch). Via removing the smallest trees, treatments also subtly raised the height of the lowest tree canopy by 1.5 meters (5 feet). Raising the lower canopy is important to reduce the risk of crown fire, by decreasing ladder fuels that enable fire to move from the ground up into the tallest tree crowns. However, treatments did not affect fuel in the upper forest canopy above the lower canopy. This means that if fire reached the forest canopy, crown fire could move just as easily through the treated area as it could in areas not treated.

Overall, the revised project minimally affected the forest despite the major effort expended to implement the project. Thinning projects in ponderosa pine forests also can be quite expensive – it is not uncommon for the projects to cost $1,000/acre of forest. Part of the reason for the high cost is that mainly small-diameter wood is produced, for which there is little current market or industry in the region to use the wood, and the material must be transported long distances. Because of this, the small trees and slash generated by the projects are often simply piled on site and burned for disposal.

Fuel reduction and fire risk are only part of the story. Forest canopies must be sharply reduced, to near pre-fire-exclusion levels, to increase understory plants or insects. The forest is more than just trees. Projects that minimally affect forest canopies have little benefit to many non-tree species – which are most species of the forest (Fig. 3.9).

Fig. 3.9. Burned forest retaining high tree density and minimal understory vegetation, South Rim, Grand Canyon National Park, Arizona. Photo by S.R. Abella.

Conducting tree thinning incrementally in phases over time has been proposed, but there are disadvantages to this approach. If crown fire risk is not reduced by the initial treatments, then stand-replacing fire can remove the forest in the meantime and make later incremental thinning moot. Each project costs money, uses planning resources, affects park visitor use, and creates chance to accidentally introduce non-native species.

The most practical, effective strategies for restoring Grand Canyon ponderosa pine forests include judicious use of forest thinning to substantially reduce tree density, and long-term prescribed burning, together with allowing some lightning ignitions to burn. These strategies, including more effective forest thinning than was used in the initial project, are currently used in the park. Not everyone will agree with cutting trees in a national park. However, restoration forest thinning is useful for reestablishing open forests, resembling evolutionary habitat of the forests, in areas where fire is initially ineffective or poses unacceptable risk to ecosystems and people. Following thinning, fire can ideally be a natural process sustaining open forests and species benefitting from fire.

An interesting ecological response to reintroducing fire in Grand Canyon forests is that some plant species become prevalent only after fire. Eight years after a 1993 prescribed burn at Swamp Ridge, North Rim, there was an average of 40 species/0.1 hectare (40 species/0.25 acre).[19] This was 60% more than in a nearby unburned area. Many of the additional species in the burned area were annual (living only one year) and perennial wildflowers. Wildflowers most common after fire were rabbit-tobacco (*Pseudognaphalium macounii*), rock phacelia (*Phacelia egena*), spreading groundsmoke (*Gayophytum diffusum*), and fireweed (*Chamerion angustifolium*).

Seeds of some species in ponderosa pine forests have been triggered to germinate following exposure to smoke.[2] Recently, an Australian research group discovered a specific chemical compound (out of hundreds) in smoke thought to promote seed germination in numerous species across continents.[10] The role that smoke may have in stimulating germination of plants after fire in Grand Canyon forests needs more evaluation. We can say, though, that reestablishing fire as a natural process has promoted plant diversity, which probably benefits insects and other species.

To the west in California's Sierra Nevada Mountains, Sequoia National Park has among the oldest prescribed fire programs (after the Everglades) in the national park system. By the 1960s, exclusion of formerly frequent fire was recognized as a major threat to many park forests (Fig. 3.10 left side). In particular, groves of giant sequoia trees (*Sequoiadendron giganteum*) are a magnificent park resource. The largest sequoia trees have trunk diameters of 10 meters (33 feet) at the base, are 80 meters (264 feet) tall (equivalent to a 26-story building), and can live 3,000 years.[34] Losing these trees is essentially permanent on a human time scale, as 100 human generations are required for the oldest trees to develop.

The Giant Forest, a 350-hectare (860-acre) sequoia grove in the northwestern part of the park, contains General Sherman, one of Earth's tallest trees. Fire-scarred trees revealed that ground fires occurred in the grove every three years in the two millennia before the 1800s.[34] The last 150 years, with little fire, are the most anomalous in the last 2,000 years.

To reduce accumulated ladder fuel posing a threat to giant sequoias, prescribed burns and allowance of some lightning fires to spread were implemented after 1968 in Sequoia and adjacent Kings Canyon National Park.[21] Requiring multiple low-intensity burns, these fires over time decreased ground and ladder fuels (Fig. 3.10 right side). Additionally, 100,000 giant sequoia seedlings/hectare (40,000 seedlings/acre) became established after some of the burns.[21] Most of the seedlings naturally die, but the mass germination highlights suitability of the post-fire environment for giant sequoia seed germination.

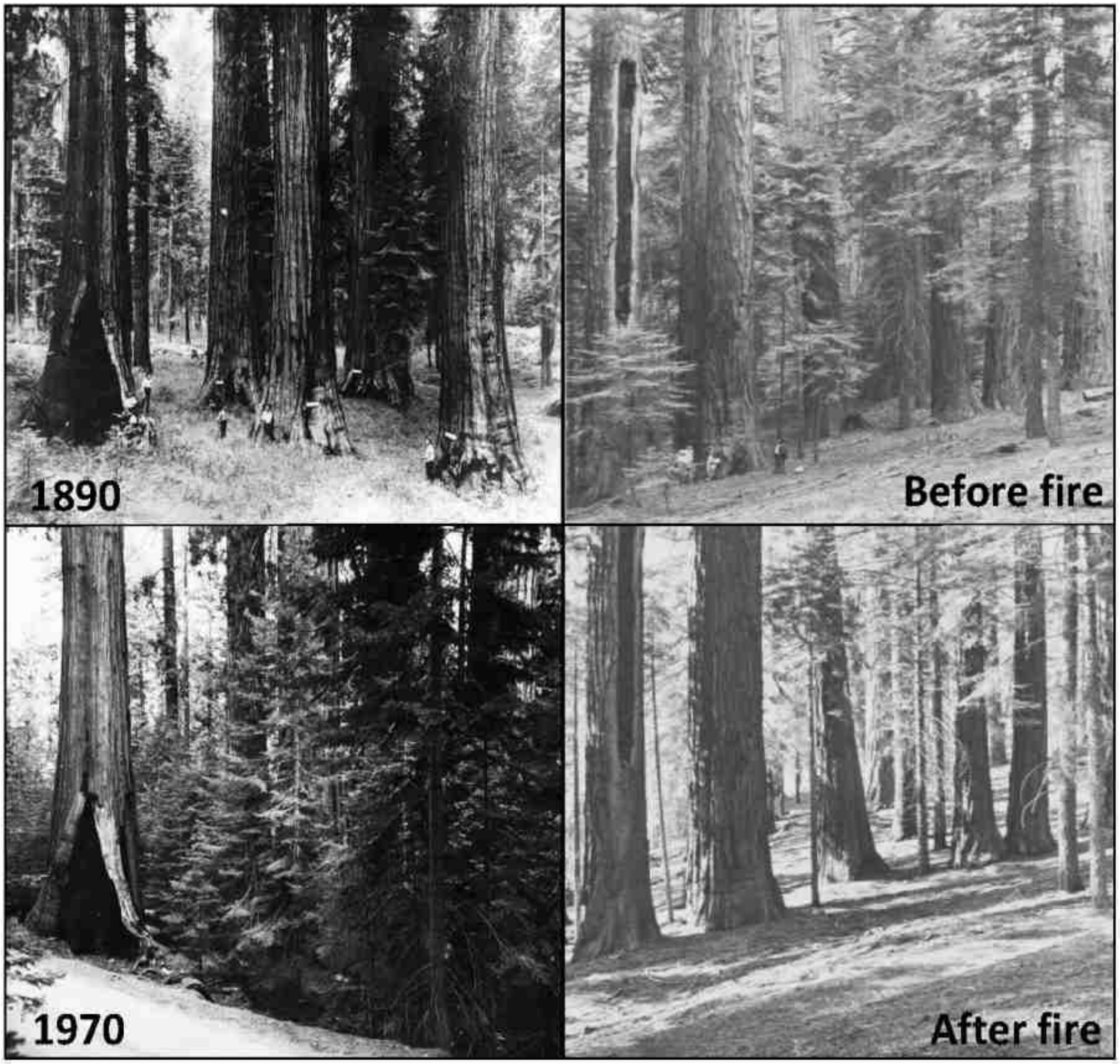


Fig. 3.10. Left side: increasing tree density and ladder fuel in Sequoia National Park, California. Right side: before and one year after prescribed fire in a giant sequoia grove along Redwood Mountain Trail, Kings Canyon National Park, California. Photos from Stephenson (1999)[31] and Kilgore and Biswell (1971).[21]

The amount of fire restored to Sequoia and Kings Canyon National Parks is a major accomplishment. It also illustrates a challenge. Despite burning an average of 1,504 hectares/year (3,716 acres/year) since 1968, this is only 15% of the estimated area burned annually before Euro-America settlement around 1860 (Fig. 3.11). A concern with this is that the park continues to fall behind in area burned, resulting in a "fire debt," and further accumulation of fuel across the landscape.[5] Striking a balance between prescribed burning new areas for the first time, versus re-burning areas, is challenging. Vegetation grows and produces fuel at a faster rate than it decomposes. Re-occurring fires need to periodically decrease fuel, or fuel will accumulate.

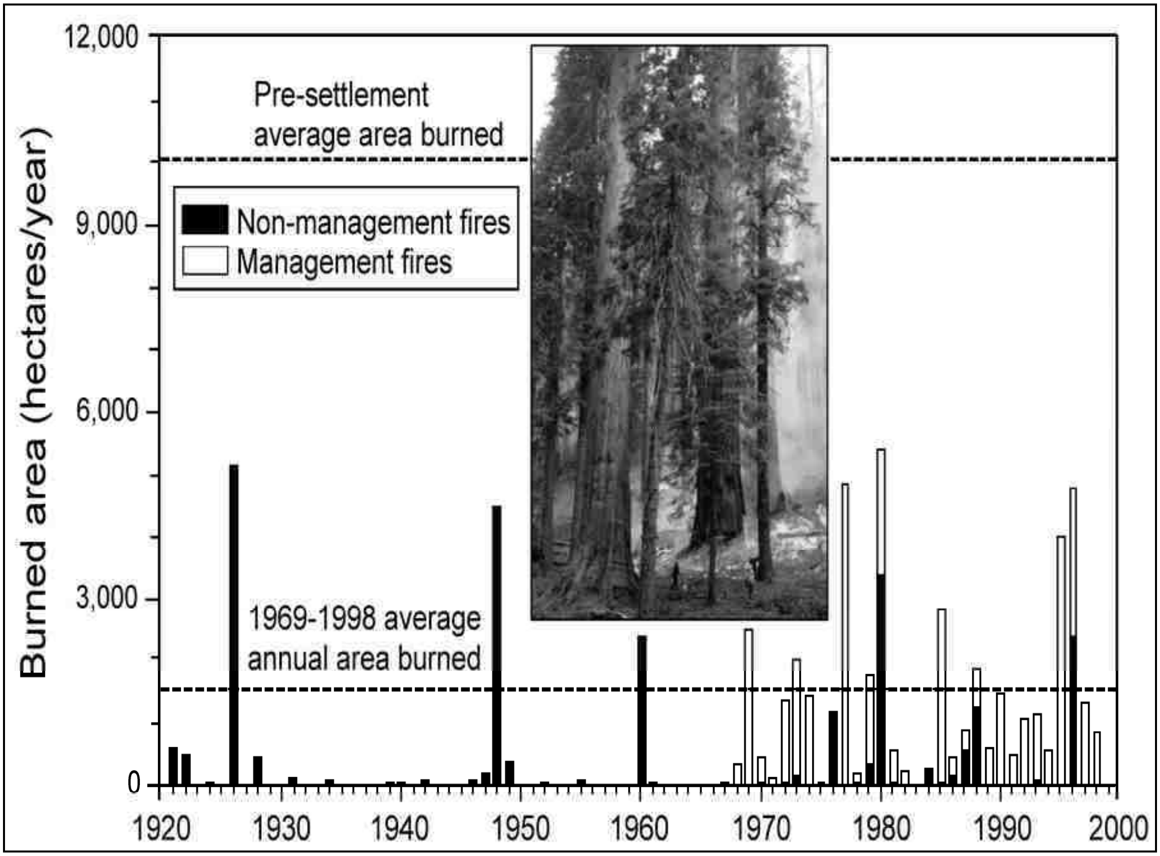


Fig. 3.11. Area burned annually by prescribed or lightning fires allowed to burn (collectively management fires, in white) and by non-management fires (wildfires suppressed as soon as possible, in black) in Sequoia and Kings Canyon National Parks. The lower broken horizontal line is the average area burned (management plus non-management fires) after the prescribed fire program began. This is compared to the upper horizontal line showing the estimated area burned annually during the pre-fire-exclusion era (before 1860). Data from Caprio and Graber (2000).[5] The photo shows a 2001 prescribed fire in the Giant Forest sequoia grove. Photo by A.C. Caprio (National Park Service) and in Swetnam et al. (2009).[34]

Yellowstone's 1988 Stand-Replacing Fires

Naturally severe fires in lodgepole pine (*Pinus contorta*) forests of Yellowstone National Park contrast with the frequent, low-severity fires characterizing forests of many parks such as Grand Canyon and Giant Sequoia. In summer 1988, a series of simultaneous wildfires burned 321,000 hectares (793,000 acres), or 36% of the entire area of Yellowstone National Park. The dry summer of 1988 coincided with numerous ignitions, and with a revised National Park Service policy of allowing some lightning fires to burn, at least early in the summer when the fires were still small. As the fires progressed, the National Park Service was increasingly criticized for its "let burn" strategy. The fires were further portrayed in media reports as "holocaustic" and destructive.[26]

But were they? Probably not. By analyzing the ages of forests in the park to reconstruct past stand-replacing fires, fires similar to those in 1988 had apparently also occurred in the late 1600s and early 1700s (Fig. 3.12). It often takes lodgepole pine forests 300 years to accumulate fuel sufficient to support widespread crown fire. Moreover, at least a portion of the lodgepole pine cones are serotinous; the cones only open to release their seed when exposed to high temperature.[27] Apparently, serotiny is an adaption signaling that it is time to germinate and reestablish the forest after a stand-replacing fire.

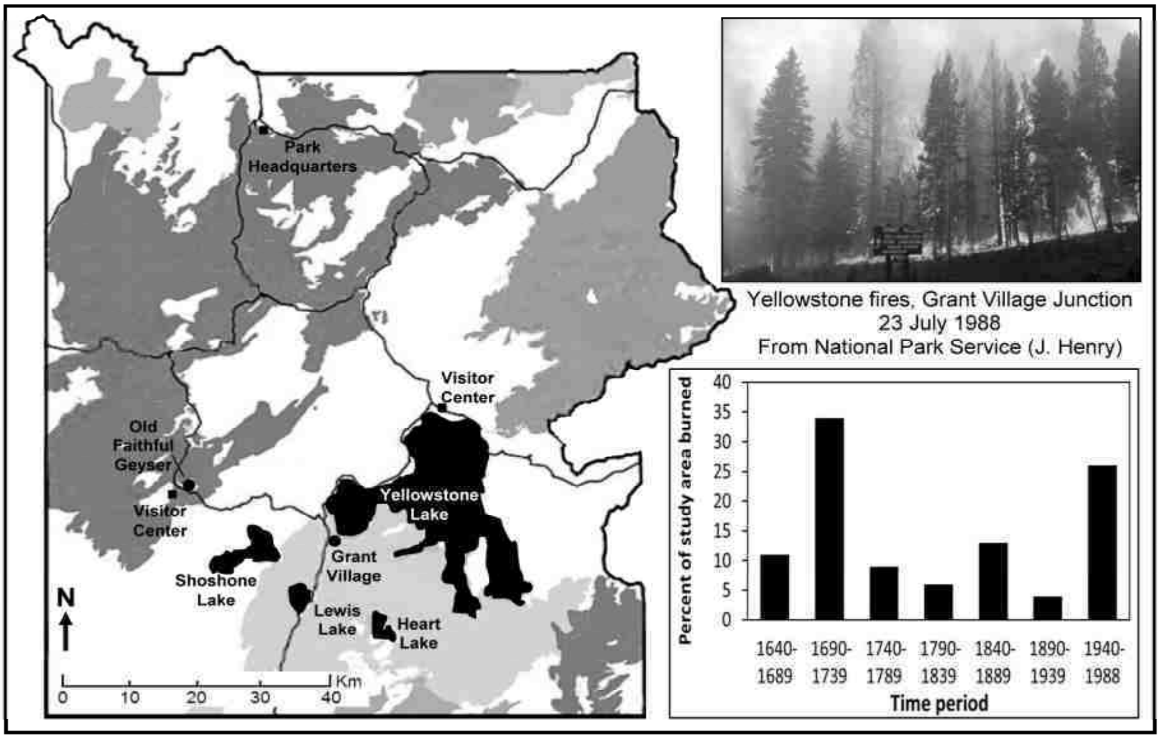


Fig. 3.12. Map showing the 1988 wildfires (shaded in gray) that burned a third of Yellowstone National Park. Graph on the bottom right is area burned since 1640, based on Romme and Despain (1989).[26]

Following the 1988 fires, wildflowers rapidly became established.[27] So did lodgepole pine trees (Fig. 3.13). Elk (*Cervus elaphus*) populations rebounded to pre-fire levels by 1993.[27]

Yellowstone's fires provide a good example for how we should use care to avoid misrepresenting fire as a natural process or not. The *type* of fire occurring matters. The 1988 Yellowstone fires – and subsequent recovery of the ecosystem – are considered to have been natural and consistent with evolutionary processes in that ecosystem. These are precisely the processes we want unfolding in national parks.

Fig. 3.13. Left: lodgepole pine forest in October 2012, 24 years after the 1988 fires, near Norris Geyser Basin, Yellowstone National Park (S.R. Abella). Bottom: lodgepole pine cone opened after the 1988 fires (J. Peaco).

Too Much Fire in the Santa Monica Mountains

Abundant and contiguous fuels, dry summers, fall Santa Ana winds, and numerous ignition sources make southwestern California one of the most fire-prone regions in the country.[20] Northwest of Los Angeles, Santa Monica Mountains National Recreation Area contains two general types of shrubland: coastal scrub nearest the coast and chaparral on inland slopes.[38] The park's natural mature shrublands are often so dense they are nearly impenetrable to walk through, attesting to the continuity of their fuels. These shrublands are thought to have historically burned via stand-replacing fire every 40 to 100 years.

In contrast with many other parks, a main concern in the park today is too much fire. Owing to pervasive ignitions by humans, non-native plants that augment fuel, and possibly changing climate, park managers are concerned that fires are now unnaturally frequent. Shifting from a historical regime of relatively infrequent stand-replacing fire, to a regime of frequent fire, does not allow enough time for the natural shrublands to re-grow. Fire too frequent might preclude reestablishment of shrubland altogether. Instead, shorter-lived non-native plants can dominate. These non-native plants are often annuals that live only one year and provide different habitat than mature shrublands. In such a case when one vegetation type completely replaces another, it is called type conversion (Fig. 3.14).

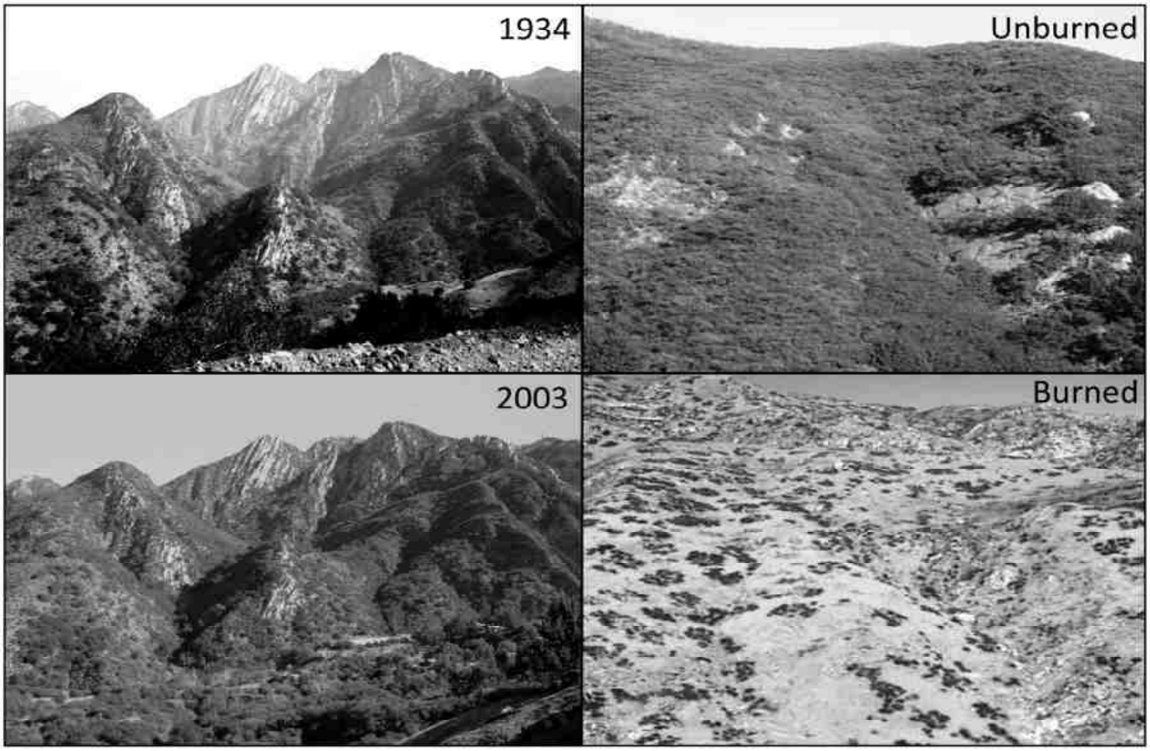


Fig. 3.14. Left side: minimal change in a mature chaparral landscape under relatively long fire intervals exceeding 23 years. Right side: type conversion from shrubland to non-native annual grasses under high fire frequency. Left side 1934 photo by A.E. Wieslander and 2003 by R.S. Taylor, with both provided by the National Park Service. Right side photos by S. Davis and adapted from Witter et al. (2007).[38]

Vegetation recovery is being closely watched on the 2013 Springs Fire, which burned 12% of the entire park. This fire was unusual in its timing – it started in early May during spring, rather than later in fall when most large chaparral fires occur. The winter of 2013 was exceptionally dry, causing extremely low fuel moisture. This situation coincided with an ignition source thought to have been a passing vehicle along the 101 freeway. In only a day and a half, the fast-moving fire burned 9,809 hectares (24,238 acres) within an earlier footprint of the 1993 Green Meadow Fire.

Fire suppression is increasingly important for conserving the park's mature shrublands, which is challenging because fire suppression is difficult in southwestern California. In fact, scientists have debated that the fire suppression policy of the 1900s has *not* resulted in unnatural fuel buildup in chaparral because widespread fires continued despite the policy.[20] Limiting unwanted ignitions created by humans, controlling non-native plants, and vigilant fire suppression are key elements of an overall strategy for lengthening intervals between fires to enable mature shrubland to develop (Fig. 3.15). If trends of the 1900s and 2000s continue, there will be no shortage of stand-replacing fire.

Fig. 3.15. Contiguous shrubland fuels in the foreground and covering the slopes, Sandstone Peak Trail, near the Springs Fire (northwest of this scene), Santa Monica Mountains, California. By S.R. Abella, 2013.

Burning Tundra in Alaska

Arctic tundra is a vast treeless region with soils underlain by a permanently frozen layer termed permafrost. World-wide, tundra occupies 16% of land area. Some researchers have estimated that tundra may store almost half of the entire amount of carbon in Earth's soil.[35] Cold temperatures limit decomposition of organic material in permafrost, while the tundra's upper soil can be surprisingly productive during short summer growing seasons. Arctic tundra occupies one-third of Alaska, in areas too cold for boreal forest.

In the central Alaska Brooks Range, Gates of the Arctic National Park and Noatak National Preserve are two parks containing extensive tundra. Seemingly paradoxically, given that tundra is underlain by permafrost, fire has long been part of the tundra ecosystem in these parks. By analyzing pollen and charcoal buried in lake sediments in and near Gates of the Arctic, vegetation and fire history were reconstructed for the last 14,000 years.[17]

Charcoal deposition was related to vegetation type, which changed through time (Fig. 3.16). Genesis of shrub tundra 13,000 years ago correlated with increased charcoal. Charcoal then declined under deciduous woodland, before increasing again 5,000 years ago, when boreal forest developed. Fires burned shrub tundra an average of every 140 years, not too different from the every 170 years subsequently in the boreal forest. To the west in Noatak National Preserve, fire intervals were similar. Fires burned herbaceous or shrub tundra every 135 to 310 years during the last 2,000 years.[18]

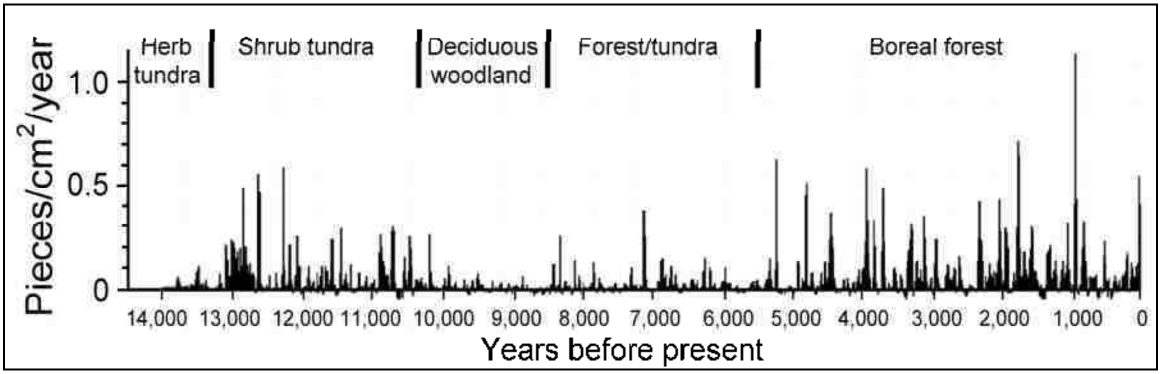


Fig. 3.16. History of vegetation types and fire, as shown by analyzing charcoal accumulation in sediment of Ruppert Lake, Gates of the Arctic National Park, Alaska. Adapted from Higuera et al. (2009).[17]

Our current understanding is that across Alaskan tundra, historical fire intervals were variable. Some tundra burned frequently like in Gates of the Arctic, whereas other tundra went 5,000 years without fire.[18] Possible changes to these fire regimes are of keen interest. Recent expansion of trees into tundra is widespread and could affect fires. In Noatak National Preserve, white spruce (*Picea glauca*) trees have encroached 100 meters (330 feet) into the tundra during the last 200 years.[32] This might be related to warming temperatures. Warming could directly affect fire, as could shifts in tundra vegetation from herbaceous to woody plants. In 2010, 37 fires burned 43,000 hectares (106,000 acres) in Noatak National Preserve (Fig. 3.17).[18] This was the most fire activity during a single year since record keeping began in 1950. A massive release of carbon into the atmosphere during tundra fires – contributing to greenhouse gases – could be an outcome of potential changes in tundra fire regimes. A changing role of fire could also affect wildlife habitat, such as tundra plants eaten by reindeer (*Rangifer tarandus*).

Fig. 3.17. Uvgoon Creek Fire burning tundra in July 2012, Noatak National Preserve, Alaska. The dark areas in the foreground are lakes (National Park Service photo).

Proactive and Reactive Approach to Fire in National Parks

Fire management invokes questions regarding which evolutionary processes and features of parks we wish to conserve. If we seek to maintain open oak ecosystems such as in Indiana Dunes National Lakeshore, then fire is a useful tool. If exceptionally frequent fire can be prevented, then conserving mature shrublands in Santa Monica Mountains National Recreation Area is feasible. Otherwise, it is unlikely. Current old-growth ponderosa pine forests cannot be conserved in many parks if stand-replacing fires continue superseding historically frequent, low-severity fires. Thus, fire management strategies across the national parks and across different ecosystems within parks will likely need to include a mixture of: 1) reintroducing or maintaining natural fires; 2) suppressing fire; and 3) using non-fire treatments, like tree thinning or non-native plant control, to manage fuels. Understanding the historical fire regime that has shaped the evolution of park ecosystems, and whether that fire regime has changed, is fundamental to conservation.

Finding ways to become more proactive with fire management is a pressing need. The U.S. Department of Agriculture reported that the Forest Service, within the department, spent $1 billion on fighting fires each year from 2000 to 2014 out of its annual $5 billion budget. In 2014, the Forest Service spent $2 billion, or 40% of its entire allocated budget, just on fighting fires. This has resulted in a vicious cycle, where overruns in firefighting expenditures result in taking money out of other programs. These other programs include proactive fuels treatment, vegetation management, maintaining recreation sites, and fire research. National forests, managed by the Forest Service, benefit society on their own. From a national park perspective, it is also important to keep national forests in good shape, because they frequently surround national parks and buffer parks from other land uses.

From 2001 to 2014 for wildfire activities, the National Park Service received an appropriated budget averaging $108 million each year. Suppressing fires required 40% of the budget, leaving 28% for fuel reduction and 32% for preparedness. The funds for fuel reduction also included educational programs for homeowners to potentially make their property more fire safe and helping communities develop fire plans. So, not all the 28% went to actually treating fuels in parks. Preparedness included prevention and educational programs to reduce unwanted ignitions of fires by humans, developing fire management plans for parks, and readiness activities like purchasing fire engines. Suppression remains an important tool, but should not be the only tool in most parks. Additionally, treating hazardous fuels in parks through prescribed fire, restoration tree thinning, or non-native plant control can have ecological benefits other than just fire management.

Fire management is intertwined with other conservation issues in nearly all land-based parks. For example, Chapter 5 discusses relationships of fire with non-native plants that alter fuel conditions. Further integrating fire management activities with other conservation actions would benefit many parks. Integration should include comparing the influences of different fire management strategies on conserving priority features of parks, and monitoring the effects of fires on wildlife habitat and other park values.

4 PREDATOR AND PREY

Predators kill and eat animals as all or part of their diet. A predator species that only eats meat is an obligate carnivore. Predators that eat both meat and plants are omnivores. The prey for predators commonly are herbivores – animals that eat only plants. However, the tables can turn for some predators, which can also be prey for other predators. Panthers, for example, kill smaller carnivores, such as bobcats.

Predators come in many shapes and sizes. They encompass predatory organisms in the soil (such as centipedes that eat other soil organisms), insects that eat insects, and large animals. Some predators kill from the air. The predatory raptor golden eagle (*Aquila chrysaetos*) can move 240 to 320 km/hour (150 to 200 miles/hour).[20] This is faster than cruising speeds of most small planes. Some plants are even predators. From stationary positions, over 600 plant species world-wide use passive or active traps (including motion-triggered snap traps) to catch and consume insects.[8]

The interactions among plants, herbivores, and carnivores create an ecosystem's food web – who eats who. Because the transfer of energy from one level to the next within a food web is not completely efficient, as energy is used for basic living functions like metabolism, fewer organisms exist at the top of food webs than at the bottom. A given ecosystem might contain millions of plants, but only a few carnivorous animals at the top of food webs. Although top carnivores are not abundant, they greatly affect ecosystems.[27] Biologists have termed such carnivores keystone species, which have a disproportionate effect relative to their abundance. Removing keystone predators can have cascading impacts to species below them in the food web, all the way down to plants.[9] Examples of predatory animal species, ranging from birds to bears, are summarized in Table 4.1.

Table 4.1. Examples of air and land predator species of the United States.

Species	Scientific name	Example diets
Golden eagle	*Aquila chrysaetos*	Rabbits, squirrels, other rodents
Red-tailed hawk	*Buteo jamaicensis*	Mice, chipmunks, squirrels, rabbits, bats
American alligator	*Alligator mississippiensis*	Fish, amphibians, reptiles, birds, mammals
Red fox	*Vulpes vulpes*	Mice, squirrels, voles, birds, berries
Bobcat	*Lynx rufus*	Rabbits, rats, skunks, young deer
Panther, Mtn. lion	*Puma concolor*	Deer, elk, moose, rabbits, rodents
Wolverine	*Gulo gulo*	Squirrels, mice, beaver, deer, elk, seeds
Coyote	*Canis latrans*	Rabbits, deer, elk, rodents, snakes, berries
Gray or timber wolf	*Canis lupus*	Deer, elk, bison, foxes, rodents, insects
American black bear	*Ursus americanus*	Plants, berries, nuts, insects, fish, deer
Grizzly bear	*Ursus arctos*	Elk, deer, squirrels, fish, plants, berries

A key function of predators is keeping in check the abundance of prey species, both by killing prey animals and altering the activities of prey (Fig. 4.1). In their role, animal predators are part of many interrelated factors that affect herbivores and the overall food web. For example, events before park designation or fire management, as discussed in Chapters 2 and 3, influence vegetation and hence forage availability to herbivores. This, in turn, affects predators via availability of prey and also because berries, nuts, and plants form a major or supplementary part of diets for some predators.

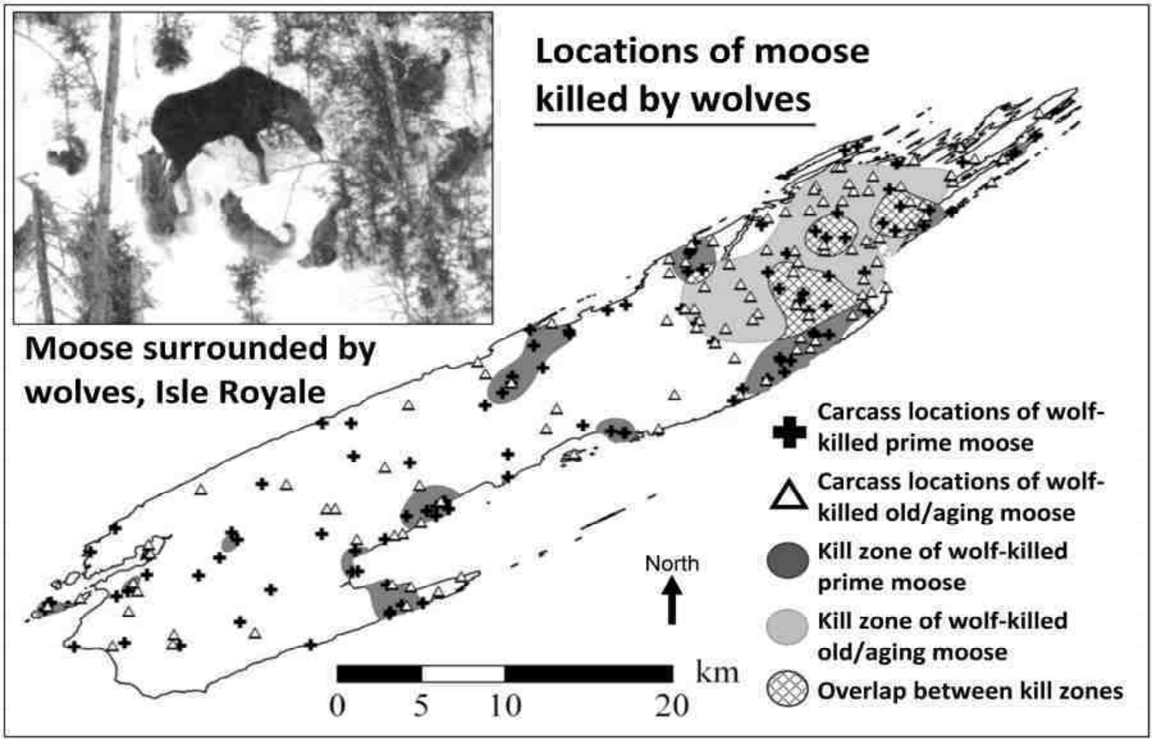


Fig. 4.1. Locations of documented kills by wolves between 2000 and 2008 in Isle Royale National Park, Lake Superior, Michigan. Map adapted from Montgomery et al. (2014).[28] Photo from Nelson et al. (2011),[29] used with permission from Elsevier.

Isle Royale National Park, Michigan, exemplifies these complex interrelationships within predator-prey systems. Described as the longest predator-prey study in the world, wolf-moose interactions on Isle Royale have given ecologists surprises.[29] When the park was established in 1940, no wolves occupied the island. Gray wolves (*Canis lupus*) arrived in the late 1940s, apparently by crossing 25 km (15 miles) of Lake Superior ice from the mainland. One can imagine how delighted wolves were that the island was inhabited by moose (*Alces alces*), a major prey species. How would predator and prey respond? A theoretical predator-prey population dynamic would resemble how a thermostat regulates temperature. Temperature fluctuates within a range, as temperatures too cold or hot trigger heating or cooling, resulting in oscillations around the set temperature. Likewise, increases in prey would trigger increases in predators, followed by declines in predators as prey become depleted. The decline in predators then allows prey to increase, and so on. Nature has been more complex than this at Isle Royale.

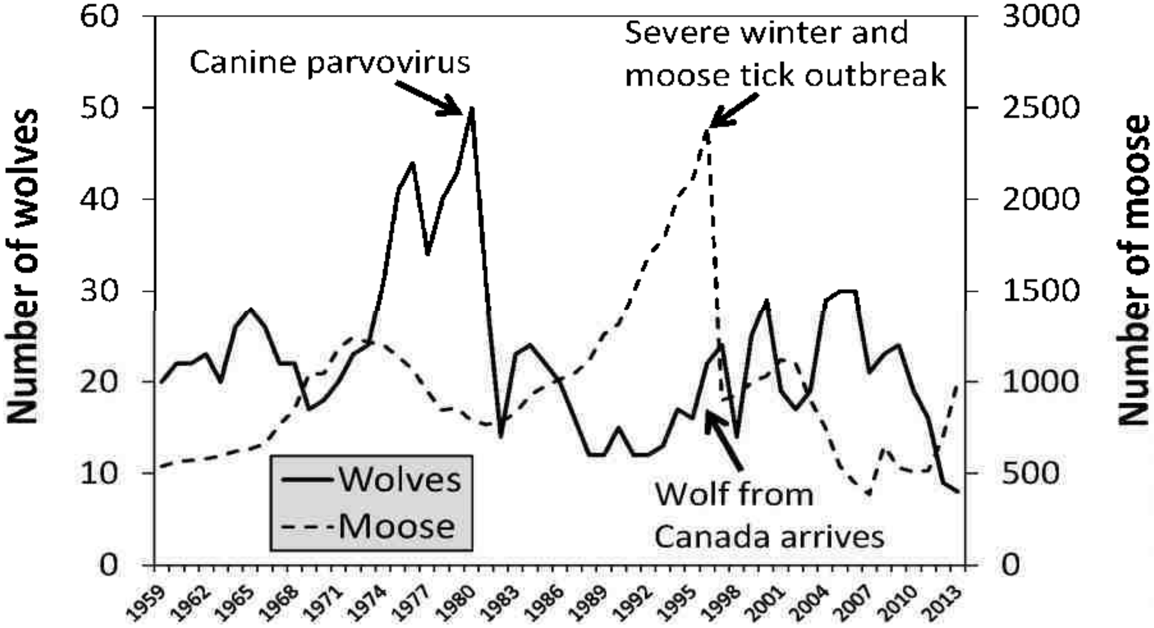


Fig. 4.2. Population dynamics of wolves and moose in Isle Royale National Park. Data from Nelson et al. (2011)[29] *and J.A. Vucetich and R.O. Peterson of the Wolves and Moose of Isle Royale project.*

In 1958, ecologists began monitoring Isle Royale's wolves and moose each year. These surveys were continued by researchers for over 50 years (Fig. 4.2). Populations of both wolves and moose initially increased. Wolf populations crashed following a 1980 inadvertent human introduction of canine parvovirus. As expected, moose populations increased for the next 16 years, until 1996. Then, unpredictably, a severe winter and outbreak of moose ticks decimated moose. Even with fewer moose to hunt, wolves sharply increased after 1997 following immigration of a wolf from Canada that alleviated inbreeding in the wolf population. As of 2015, however, sustainability of the wolf population is uncertain, with only three wolves remaining on Isle Royale. This has created an interesting, still-unfolding conservation dilemma for the National Park Service. Should humans intervene and augment the park's wolf population? Thus far, the park's remarkable study demonstrated that predator and prey both persisted for over five decades on a confined island. Furthermore, the predator-prey relationship played out in an ecological arena involving many players, including diseases, insects, climate, plants, and humans.

Conserving food webs is fundamental to conserving biodiversity and ecological processes. As with fire, people have had a mercurial relationship with predator animals in the United States, including in national parks. Humans have disrupted natural food webs in many national parks, often through purposely removing carnivores like wolves and panthers. In recent decades, however, people have also enacted new protections for predators, reversing sometimes centuries of persecution. Overcoming great obstacles, predators have even been reintroduced into some national parks. This chapter focuses on large land animals as top predators, including their role in ecosystems and conservation status, by highlighting examples of predators and their prey in the following sections.

Wolverines

The wolverine (*Gulo gulo*) is one of the most mysterious and least-studied mammals in North America. Wolverines are stocky and muscular, with a reputation for ferocity. The species is in the weasel family, but has the appearance of a small bear. This, coupled with pungent musk glands like skunks, earned wolverines the nickname "skunk bears." Adult male wolverines typically weigh 16 kilograms (35 pounds).[32] Adult females weigh 10 kilograms (23 pounds). Their body sizes approximate those of medium-sized domestic dogs. Wolverines can be solitary animals, except for mating and caring for young.

Wolverines are both scavengers and hunters.[32] To assist scavenging, wolverines have strong jaws and well-developed teeth, enabling eating frozen meat and crushing bones.[15] Although scavenging constitutes much of their diet, wolverines can make their own kills of a variety of small and large animals. Wolverines travel widely for food. Researchers placed radio transmitters on 20 wolverines to track their movements in an area east of Flathead Lake, northwestern Montana, just south of Glacier National Park.[15] Individual wolverines lived within home ranges averaging 400 km^2 (150 square miles), an area equivalent in size to the entire city of Denver, Colorado. The maximum distance traveled in a three-day period was 64 km (40 miles) for males and 38 km (24 miles) for females. There was an average of 1 wolverine/65 km^2 (1 wolverine/25 square miles).

Areas with snow cover much of the year – tall mountains and northern regions of the United States and Canada – are prime habitat for wolverines.[2] The area occupied by wolverines declined substantially in the contiguous United States after the 1800s (Fig. 4.3). During the 1800s, wolverine occurrences were documented in at least 19 states (or then territories). The inhabited area included Alaska, the Northeast, Great Lakes region, northern Rocky Mountains and Pacific coastal ranges, and the Sierra Nevada Mountains in California. As of 2005, no wolverines were documented after 1920 in the Northeast or in Colorado. In Great Lakes states, few occurrences were documented after the 1800s. California's most recent verifiable record was in 1922 as of 2005. During the 2000s, sustained populations of wolverines inhabited just five states: Washington, Idaho, Montana, Wyoming, and Alaska.

Because wolverines raided trap lines (a great source of meat for scavenging) and were considered dangerous animals, early settlers, trappers, and ranchers often killed wolverines.[2] People laid out poisoned baits, to which wolverines, with their scavenging habits, were vulnerable. Intentional poisoning of wolverines even occurred in Yellowstone National Park in the late 1800s. Killing of wolverines by humans, combined with human settlement and habitat alteration, are believed to have caused the range loss by wolverines. Uncertainty remains, however, regarding the relative influences of human factors and long-term climatic changes in wolverine range dynamics. For example, the Little Ice Age, from approximately the 1400s to mid-1800s, could have benefited wolverines by promoting more persistent snow cover at that time. It is reasonable to say, though, that settlement and killing wolverines by humans is unlikely to have helped wolverine populations.

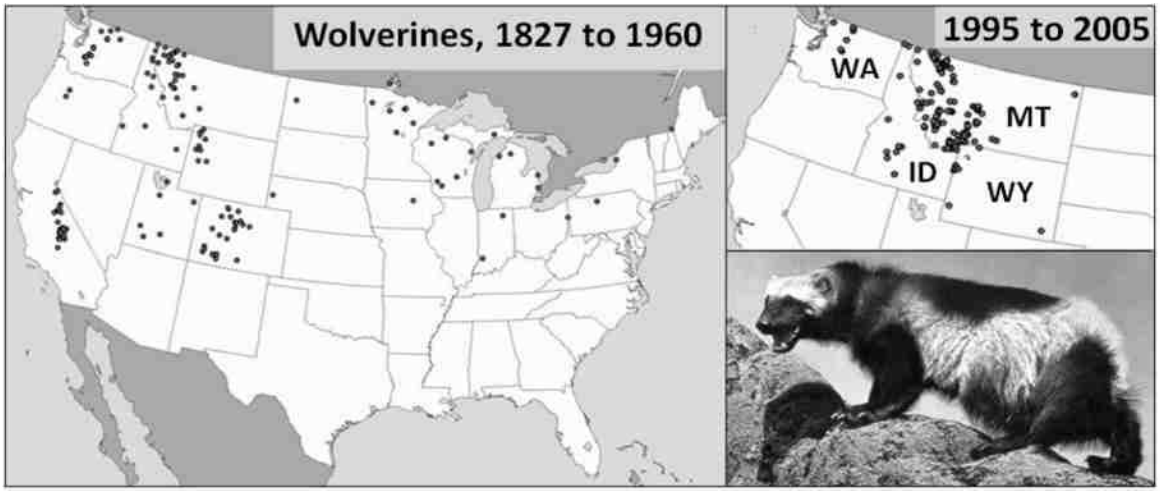


Fig. 4.3. Verified locations of wolverines showing contraction in range from historical to modern times. Data from Aubrey et al. (2007).[2] *Wolverine photo provided by Yellowstone National Park.*

What is the current status of wolverines? In 2010, the U.S. Fish and Wildlife Service determined that wolverines warranted protection under the Endangered Species Act, reversing an earlier 2008 determination. Only 300 wolverines were believed to still exist in the contiguous United States. Although warranted for listing, the backlog of other species also under consideration for protection meant that listing wolverines was not undertaken. A 2013 proposal for listing was then, again, reversed in August 2014. At that time, the Fish and Wildlife Service withdrew proposing to list wolverines as threatened, based on the reasoning that wolverine populations were increasing.

Recently, several remarkably long-distance journeys by wolverines have been documented. For example, in 2009, the first recorded wolverine in over 90 years in Colorado was sighted in Rocky Mountain National Park. According to the Colorado Parks and Wildlife Commission, this male wolverine had been previously radiotracked in Grand Teton National Park, Wyoming, over 725 km (450 miles) away. Reestablishing viable populations also requires females, so some long movements represent only transient individuals and not necessarily resident, sustained populations. Because wolverines require large habitat areas usually remote from human settlement, large national parks are likely important to future wolverine populations.

Bobcats

Bobcats (*Lynx rufus*) are obligate carnivores usually eating from their own kills.[21] Mature bobcats are twice the size of domestic cats and have short and soft fur, generally yellowish to reddish brown with black streaks (Fig. 4.4). Maximum ages that bobcats attain in the wild are around 15 years. Home ranges where bobcats conduct their daily activities vary from less than a square mile to tens of square miles. Mainly nocturnal, bobcats are also active near dawn and dusk and sometimes during the day. Bobcats are typically solitary hunters using two tactics. The first tactic is a stealthy approach followed by a pounce and strike. The second tactic is crouching on a log or vantage point, and waiting until prey passes nearby.

Fig. 4.4. Bobcat, from Grand Teton National Park, Wyoming.

Based on tracking bobcats in Golden Gate National Recreation Area around San Francisco, California, bobcats may have intermediate tolerance for living near urban developments, as long as semi-natural habitat is nearby.[33] Bobcats occupy most states and appear to be increasing since a 1981 inventory.[36] This increase possibly results from increases in prey, changing agricultural practices, and regulating killing of bobcats by humans since the 1970s. As of 2008, 41 states monitor bobcat population sizes. An estimated 2.4 to 3.6 million bobcats inhabited the United States in 2008.

A project reintroducing bobcats to Cumberland Island National Seashore, 130 km (80 miles) south of Savannah, Georgia, illustrated potential effects of bobcats in ecosystems.[7] Cumberland Island is 25 km long (16 miles) and 1 to 6 km (0.6 to 4 miles) wide, separated from the mainland by 3 km (2 miles) of salt marsh and open Atlantic Ocean. The island contains a mixture of beach dunes and forests of live oak (*Quercus virginianus*) and pine, with interspersed marshes. Bobcats went extinct on the island after the early 1900s. Despite the National Park Service allowing public hunts, populations of herbivores, such as white-tailed deer (*Odocoileus virginianus*) and non-native feral pigs (*Sus scrofa*), increased to the point that they harmed plant growth and reduced habitat quality.

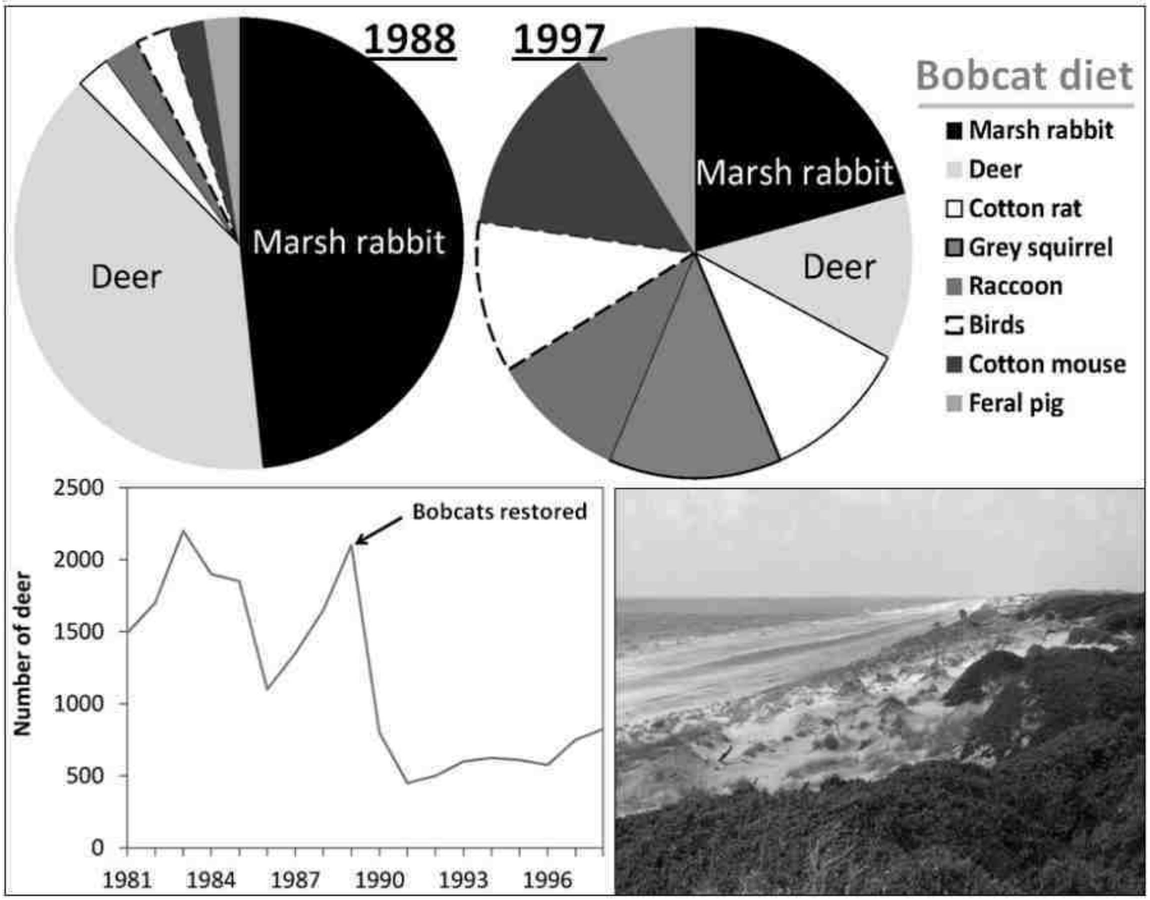


Fig. 4.5. Top: bobcat diets during the first year of bobcat reintroduction (1988) and 10 years later (1997), Cumberland Island National Seashore, Georgia. Bottom left: white-tailed deer populations decreased after bobcat reintroduction. Data from Diefenbach et al. (2009). [7] *Photo of the Cumberland Island shore provided by the National Park Service.*

Planning for reintroducing bobcats as predators began in the 1980s. Later, the organizers perceived they made a mistake by initially proposing the project for public comment as one to reduce overpopulated herbivores.[7] Environmental organizations and individuals questioned whether herbivores should be managed in national parks. Hunters were concerned that bobcats might prey on game animals, as limited hunting is permitted and managed in the seashore. When the project was recast as reinstating a key native species for restoring biological diversity, public support increased and the project proceeded.

Partly using hunting dogs, adult bobcats were captured on the mainland. Thirty-two bobcats were then translocated to Cumberland Island in 1988-1989. In the first three years of the project, 93% of bobcats survived in their new habitat. During that time, 14 bobcat kittens were also born on the island.

The reintroduced bobcats quickly began hunting and changed the ecosystem. Diets of bobcats shortly after their reintroduction were dominated by marsh rabbits (*Sylvilagus palustris*) and white-tailed deer (Fig. 4.5). By 10 years after bobcat reintroduction in 1997, different prey animals were more evenly distributed in bobcat diets. Eight prey species comprised bobcat diets. The population of white-tailed deer decreased after bobcats were reintroduced (Fig. 4.5). But, the average weight of individual deer – indicative of animal health – increased by 11 kilograms (24 pounds). Also, the number of oak seedlings increased after bobcat reintroduction, probably because fewer herbivores were eating them.

Panthers

Fig. 4.6. Panther photographed at night by the National Park Service, Everglades National Park, Florida.

The panther (*Puma concolor*) has among the most names of any animal, also being called the puma, cougar, mountain lion, and catamount. Ranging from northern Canada to South America, the panther as a species has the largest range of any land mammal in North America.[31] Individual panthers have enormous home ranges where they conduct their hunting and mating activities. Home ranges commonly exceed 150 km^2 (60 square miles) for adult males and can encompass 800 km^2 (300 square miles).[31] Usually only one to seven panthers exist within 100 km^2 (40 square miles). Panthers are solitary and secretive, preferring to avoid human settlements. They are usually silent animals but can purr and hiss, similar to domestic cats. Panthers are mainly active at night, but sometimes also at dawn and dusk and during the day (Fig. 4.6).

Panthers are slender, agile, and well-equipped for their role as carnivores. Including their tails, adult males are 2.5 meters long (8 feet) and females 2 meters (7 feet) long.[6,31] Typical weights are 50 to 100 kilograms (115 to 220 pounds) for adult males and 30 to 64 kilograms (67 to 140 pounds) for females.[6] Panthers are the second heaviest cats in the Americas, after jaguars (*Panthera onca*). Panthers have large muscles and paws, enabling leaping over 3 meters (10 feet) vertically and 12 meters (45 feet) horizontally.[31] They can run 80 km/hour (50 miles/hour), climb adeptly, and swim if necessary. Although they can sprint, panthers are typically ambush predators, preferring to stalk prey and leap in for kills with suffocating neck bites. Predation rates vary with many factors, but generally a panther kills one large animal (such as a deer) and one smaller animal every week or two.[31]

Intensive hunting and habitat alteration by humans relegated the panther extinct in parts of the West and most areas of the eastern United States by 1900, except for an isolated Florida panther population that is endangered.[23] After panthers were classified as a regulated wildlife species in the 1960s and 1970s, panther populations in the West began increasing.

The increasing western populations are source populations for recolonization of the East. Between 1990 and 2008, there were 178 confirmed panther sightings in Midwestern states, including Nebraska, Oklahoma, Iowa, Arkansas, and others.[23] Most of these sightings were within 20 km (12 miles) of highly suitable habitat. Suitable habitat was defined as forested, with relatively steep terrain and low densities of roads and humans. In 2011, a panther from the Black Hills, South Dakota, dispersed a remarkable 3,000 km (1,900 miles). It traveled through at least Minnesota, Wisconsin, and northeastern states before being killed by a vehicle in Connecticut only 110 km (70 miles) from New York City.[23] This is among the longest dispersals of a land mammal ever recorded. Panthers are clearly recolonizing portions of the United States, spurring discussion of coexistence of panthers and human society and how panthers affect the ecology of national parks.

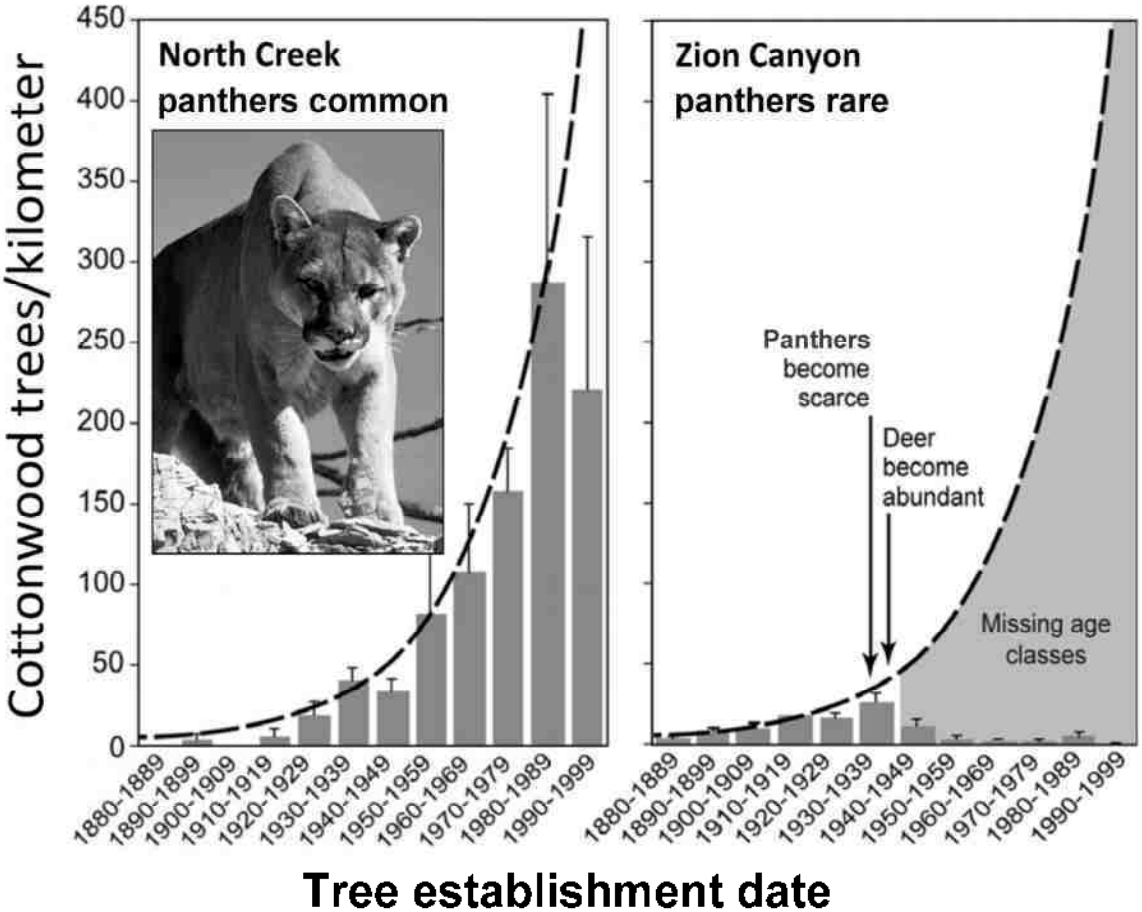


Fig. 4.7. New cottonwood trees in Zion National Park, Utah, became established during the 1900s where panthers were common but not where panthers were rare. Graphs from Ripple and Beschta (2006)[35] *and used with permission from Elsevier. Photo provided by Yellowstone National Park.*

How might panthers affect park ecosystems? A study in Zion National Park, Utah, compared ecological conditions in two canyons, one inhabited by panthers and one where panthers were rare or absent.[35] Since the park's creation in 1918, Zion Canyon has been among the park's most heavily visited sites by humans. As early as the 1930s, park managers noted that panthers were avoiding Zion Canyon. Concurrently with the vanishing of panthers, the population of mule deer (*Odocoileus hemionus*) irrupted from 80 in 1930, to over 600 in 1942. With ecological problems evident, the National Park Service killed or removed 780 deer between 1938 and 1947. Since the 1940s, deer populations have stabilized at still quite high levels of around 200 animals, or 10 deer/km^2 (25 deer/square mile). In contrast, the less-heavily visited North Creek Canyon, to the west of Zion Canyon, remained inhabited by panthers. An estimated 15 panthers occupy Zion National Park as a whole, with the North Creek area being part of the home range panthers use. With panthers at North Creek, mule deer populations remained low, around 2 deer/km^2 (5 deer/square mile).

Ecological differences measured in 2005 between the panther-free Zion Canyon and the panther-inhabited North Fork Canyon were striking. New cottonwood trees (*Populus fremontii*) became established continuously after the 1930s at panther-inhabited North Fork (Fig. 4.7). In contrast, cottonwood regeneration abruptly declined after the 1940s in Zion Canyon, correlating with the disappearance of panthers and the increase of mule deer. The few new cottonwoods in Zion Canyon mostly occurred in areas inaccessible to deer. Other ecological differences were also evident. Wildflowers were scarce in Zion Canyon but abundant with panthers at North Fork. Frogs, toads, lizards, and butterflies were also abundant where there were panthers.

While some difference between the two canyons could result from human use (such as trampling of plants), the contrasting ecological conditions are consistent with one area being influenced by deer eating much vegetation and the other area less so. If even just one panther was killing a deer every few weeks, this could have reduced Zion Canyon's deer population. When presence or absence of a top predator has a reverberating influence on herbivores, in turn affecting vegetation and other organisms, it is termed a trophic cascade.

Bears

Of North America's three species of bear, the American black bear (*Ursus americanus*) and brown bear (*Ursus arctos*, which includes the grizzly bear) inhabit the lower 48 states. The third species, the polar bear (*Ursus maritimus*), inhabits Alaska and Canada.

Black bears are the smallest and most abundant bear in the lower 48 states. Adult males weigh 80 kilograms (175 pounds); females 55 kilograms (120 pounds).[22] Black bears live up to 25 years in the wild. Walking with a shuffle-like amble, black bears can be deceptively fast, as they run 56 km/hour (35 miles/hour). This exceeds the fastest footspeed recorded for a human, which was 45 km/hour (27 miles/hour). Black bears are omnivores, eating both plants and animals. Their diet varies among parks and includes berries, nuts, insects, fish, and rodents, though they can kill larger animals like deer.[11]

Black bears are thought to have historically inhabited all states except Hawaii. Habitat alteration and killing by humans reduced the area occupied and population size of black bear during the 1800s to mid-1900s.[16] Concurrent with regulation of shooting bears and increased forest cover in some areas (such as discussed in Chapter 2 for Cape Cod National Seashore), black bear populations have rebounded in many states since the 1970s. From the 1980s through a 2001 inventory, black bear populations increased by 2% per year.[16] An inventory suggested that 320,000 black bears inhabited the lower 48 states in 2001 (Fig. 4.8). Another 80,000 black bears were estimated to inhabit Alaska.

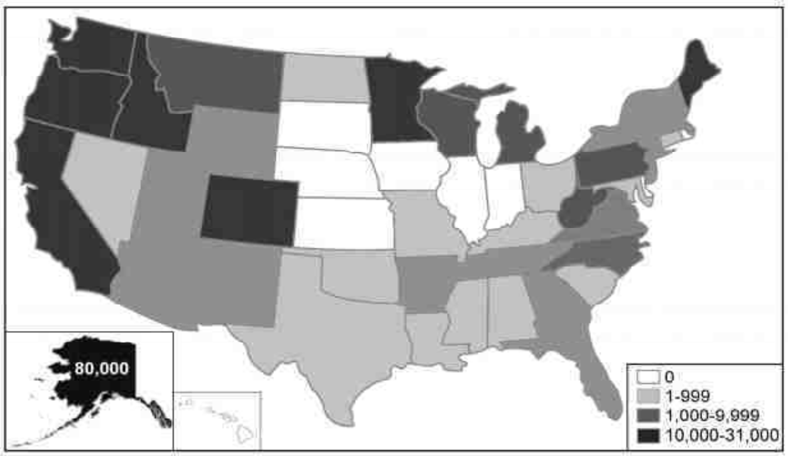


Fig. 4.8. Population sizes in 2001 of American black bear, mapped by state from mid-points of data in Hristienko and McDonald (2007). [16]

Big Bend National Park in western Texas exemplifies rebounding black bear populations. When the park was designated in 1944, bears were absent from the region through hunting and predator removal programs associated with ranching and other human activities.[30] But in the 1980s, apparently by traveling through desert from Mexico and crossing the Rio Grande River, bears returned to the Chisos Mountains within the park. About two dozen bears inhabited the Chisos by the late 1990s and early 2000s. Big Bend now reports a resident population of at least 10 bears and numerous sightings are reported each year by park staff and visitors (Fig. 4.9).

Fig. 4.9. Black bear cubs in Big Bend National Park, Texas (National Park Service photo).

The situation with grizzly bears has been different than for black bears. Weighing twice as much as black bears and among North America's largest predators, grizzly bears were more feared and vigorously persecuted by humans. Between 1850 and 1920, grizzly bears were eliminated from 95% of their former range (Fig. 4.10). Hudson Bay Company trapping records, for example, noted that 3,800 grizzly bear hides were shipped from trading posts in the North Cascade

Mountains (now including North Cascades National Park in Washington State) between 1827 and 1859. Unregulated killing of grizzly bears, likely combined with habitat alterations, further reduced the area occupied by grizzly bears through the mid-1900s. Demonstrating one of many benefits of maintaining large wilderness tracts, viable grizzly bear populations survived only in large areas exceeding 2,500 km^2 (1,000 square miles). Some parks, such as Yellowstone, were critical to sustaining the species in the lower 48 states. In 1975, the grizzly bear was listed as threatened under the Endangered Species Act.

A 2002 assessment concluded that without reductions in human lethality to bears after 1970, core grizzly bear range would not be anywhere near as extensive as it is now.[25] Grizzly bears naturally occur at low population densities, so not many animals need to be eliminated to jeopardize persistence of populations. As of 2011, Yellowstone National Park reported approximately 150 grizzly bears having home ranges within the park. In 2000, Glacier National Park and adjacent areas contained 240 grizzly bears, corresponding with a density of 30 grizzly bears/1,000 km^2 (75 grizzly bears/1,000 square miles).[19] North Cascades National Park also has some grizzlies. Denali National Park, in Alaska, reported a population of 550 grizzly bears in 2014.

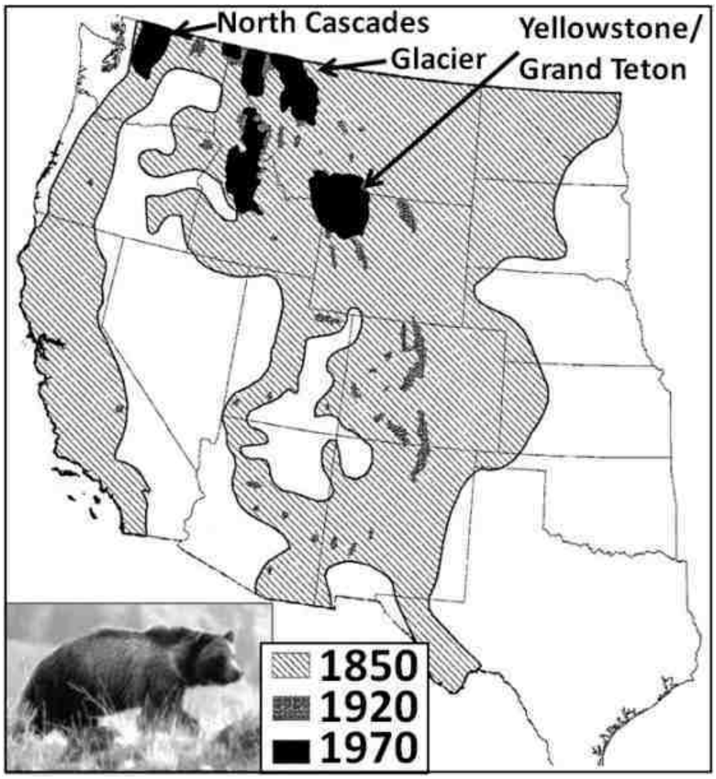


Fig. 4.10. Changing grizzly bear distribution through time in the western United States. Adapted from Mattson and Merrill (2002).[25] Photo from Grand Teton National Park.

Wolves

Wolves are a quintessential predator in North America (Fig. 4.11). They kill the continent's largest animals, much larger than individual wolves. Wolves are social animals, forming packs of two to 20 individuals, typically led by a dominant male.[26] Wolves often hunt as a pack, a formidable killing machine. Wolves travel widely, with daily movements within a home range exceeding 70 km (45 miles).

Humans have long feared wolves and sought to protect livestock from them. Written records in North America of persecution of wolves by humans appear 400 years ago. In 1637, legislation by the Colony of Massachusetts ordered a bounty of 10 shillings per wolf killed.[27] As late as the 1960s, Minnesota still offered a bounty of $50/wolf.[26] While some subspecies (representing variation in populations within a species) of wolves may have gone extinct, it is remarkable that any wolves still exist given 400 years of torment by humans.

Fig. 4.11. Gray wolf (National Park Service photo from Grand Teton National Park).

As among the world's iconic animals, it may seem we should have a clear consensus on the number of wolf species that existed, and currently exist, in the United States. But, when we consider that most eastern populations were extinct prior to 1900, before modern genetic analyses, and that small remnant populations have hybridized with expanding coyote populations, it is not surprising that taxonomy of wolves is uncertain. Some recent genetic analyses have suggested that three or four species or subspecies of wolves could have historically existed in the United States.[27] Gray wolves (*Canis lupus*) inhabited Alaska, the West, and the Great Lakes region. These animals were and are the largest wolves. Adult females weigh 18 to 55 kilograms (40 to 120 pounds); males 20 to 80 kilograms (45 to 175 pounds).[26] Current populations exist in the upper Great Lakes (including Isle Royale National Park), portions of the West, and Alaska. A smaller species or subspecies, the red wolf (*Canis rufus*), could have inhabited the Midwest and southeast and currently has a small population in North Carolina. The eastern timber wolf (*Canis lycaon*) might also have occupied the East, but is believed extinct or hybridized to the point of being genetically indistinct, assuming it had been a distinct species. The Mexican wolf (*Canis lupus baileyi*) is considered a subspecies of gray wolf. According to the U.S. Fish and Wildlife Service, 100 Mexican wolves inhabited Arizona and New Mexico in 2015.

Regardless of how many genetically distinct species or subspecies had existed, by the 1970s wolves were eliminated from 95% of their habitat in the lower 48 states. With listing of certain populations of wolves under the newly created Endangered Species Act in the 1970s, attention turned to reintroducing wolves to part of their former range. This was the first serious effort at reversing wolf persecution in three centuries.

The first wolf reintroduction in a national park was not in Yellowstone, but in a perhaps less likely location: Great Smoky Mountains National Park in North Carolina/Tennessee.[14] This was a reintroduction of the red wolf, which, depending on the taxonomy used, had formerly ranged from the Atlantic coast to northern Pennsylvania and Ohio, and southwest including all of Florida to central Texas. The red wolf was extirpated from almost its entire range by 1900. During the 1970s, only 100 red wolves still existed, in eastern Texas and western Louisiana. In 1980, the U.S. Fish and Wildlife Service captured the last of these animals and placed them in a captive-breeding program. In 1991, 37 red wolves were released in Great Smoky Mountains National Park.

The reintroduction was not successful. Most wolves did not establish territories within the park and left for surrounding lower elevation mixed agricultural/forest land. The few wolves maintaining territories in the park had low pup survival due to parvovirus, malnutrition, and parasites. Repeated attempts to reintroduce more wolves also had low pup survival. The program was terminated in 1998. Remaining wolves in and near the park were relocated to the Albemarle Peninsula, northeastern North Carolina. Although hampered by hybridization with expanding coyote populations, unauthorized shooting by humans, and inbreeding, 75 red wolves in 15 packs persist there.

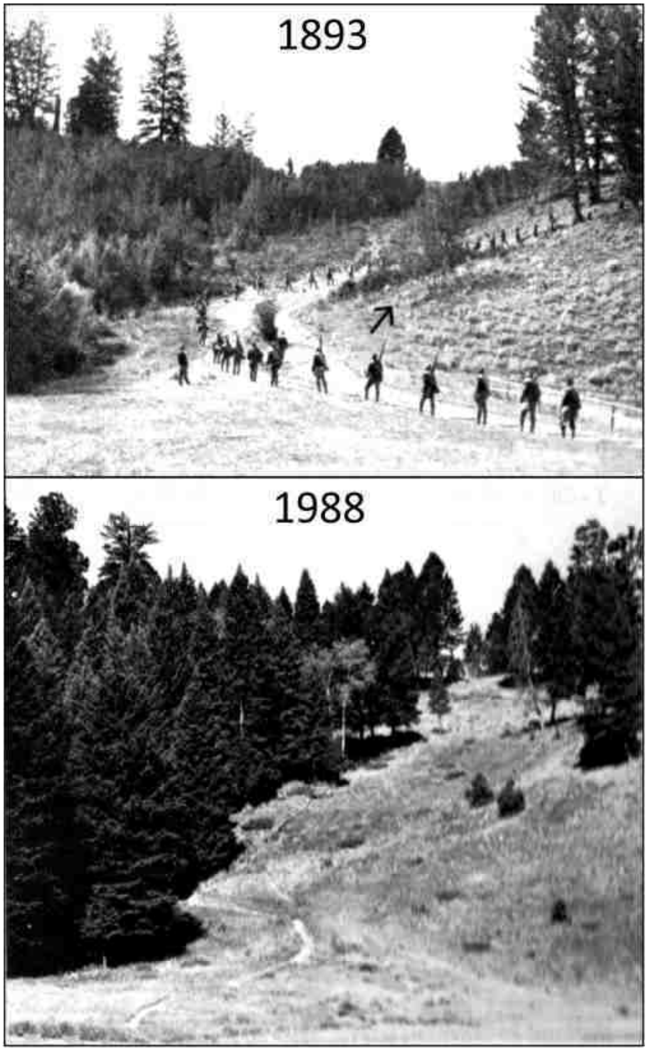


Fig. 4.12. Arrow points to aspen and chokecherry in the top photo that are gone 95 years later, Yellowstone National Park. In 1893, the military administered the park, with soldiers in the photo from the Minnesota National Guard. From Kay (1995).[18]

The better-known reintroduction of the gray wolf to Yellowstone National Park in 1995 had a different outcome and became a world-renowned example of reestablishing a top carnivore in an ecosystem. After years of debate and legislative work, history was made in January 1995 when 14 wolves from Alberta were released into Yellowstone. Seventeen more wolves were released in January 1996. The reintroduction was so successful that planned additional releases were not undertaken. As of 2013, 95 wolves in 10 packs inhabit the park.[37] These wolves made or likely made 269 kills in 2013, including 193 elk (72%), 16 bison (6%), and 13 mule deer (5%). The remaining 47 kills included coyote, pronghorn, bighorn sheep, moose, red fox, porcupine, muskrat, and others.[37]

Ecologists continue studying the unfolding predator-prey dynamic and changes within the Yellowstone ecosystem. Production of berries by shrubs are among many ecological changes of interest. Berries and seeds sustain populations of plants, and they are foods for birds and other wildlife. Based on analyzing historical photographs and fenced exclosures where large herbivores were excluded, shrubs are thought to have declined during wolf absence and the concurrent increase in herbivore populations (Fig. 4.12). In 1987, serviceberry (*Amelanchier alnifolia*) shrubs outside exclosures produced a meager 0.1 berries per plant on average.[18] Shrubs inside exclosures, protected from elk and other large herbivores, produced 1,300 berries per plant. Similarly, chokecherry (*Prunus virginiana*) outside exclosures produced no berries in 1987, compared to 2,100 berries per plant inside exclosures. Wolves can potentially affect berry production by influencing herbivores.

Along streams, willow (*Salix* species) shrubs have recently expanded and increased in height in many areas (Fig. 4.13.). In a 2007 inventory, the number of bird species also was highest around streams where willows were tall or increasing in height.[3] Moreover, yellow warbler (*Setophaga petechia*), warbling vireo (*Vireo gilvus*), willow flycatcher (*Empidonax traillii*), and song sparrow (*Melospiza melodii*) occurred only in areas with the tallest willows.

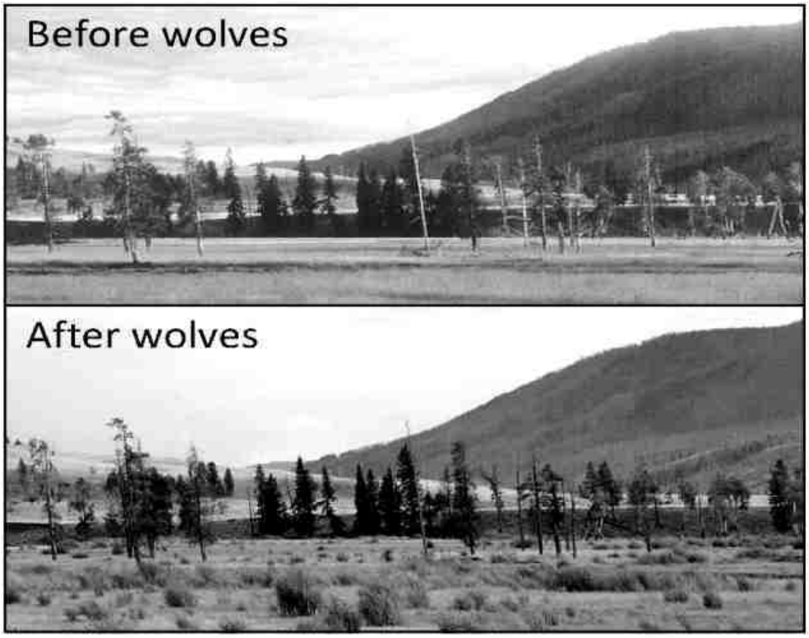


Fig. 4.13. Increasing willow shrubs over four years, at the confluence of Soda Butte Creek and Lamar River, Yellowstone National Park. Photos taken in summer (W.J. Ripple) and from Ripple and Beschta (2003),[34] permission from Elsevier.

These studies, and those assessing other ecological phenomena in Yellowstone, have found that changes have transpired coincident with wolf reintroduction. Pinpointing which of the changes wolves have caused, and how wolves cause those changes, is harder. For example, one theory holds that not only do wolves kill herbivores like elk, they also impact how elk behave through the "ecology of fear." This theory purports that elk avoid areas where risk of predation is high, such as exposed areas where escape routes for elk are limited.[4] This "ecology of fear" in turn affects how much vegetation is eaten by elk, which creates a trophic cascade affecting many other species and even stream characteristics. Other ecologists disagree, arguing that inventories of where willow and other plants are increasing do not necessarily correspond with perceived areas of predation risk.[40] Some ecologists have further noted that willow recovery in part hinges upon: 1) soil factors such as nutrients and depth to groundwater that varies from place to place; and 2) beaver (*Castor canadensis*), which construct dams and manipulate streams in ways that can promote willow.

Researchers are implementing creative experiments to determine how wolves interact with other species like beaver to affect willow recovery. In one experiment, researchers built fenced exclosures to exclude elk, then built dams to simulate how beaver increase the depth of water.[24] Over the 10-year experiment, willows grew best inside the exclosures (protected from elk) and where dams were constructed. This suggested that *both* reduced damage by herbivores and manipulation of water by beaver were needed for willows to grow their best. Most likely, wolves are interacting with other predators (such as grizzly bears), beaver, variation in soil and climate, and vegetation within an interconnected food web that continues evolving. Wolves have re-inhabited Yellowstone for only 20 years, and more remains to be learned about changes in Yellowstone's wolf-inhabited ecosystems.

Prey Irruption: White-Tailed Deer

White-tailed deer (*Odocoileus virginianus*) exemplify an irruption of a prey species under relaxed predation, partly because of the removal of top predators except humans. Deer are simply doing what organisms do – increasing their population through reproduction of individuals. But, this population increase can be detrimental to the health of individual deer and to other species in the ecosystem. In the eastern United States, white-tailed deer populations are several times larger than before Euro-American settlement.[5] This has created a conservation dilemma in national parks. On one hand, the overall public can be understandably uncomfortable with reducing deer numbers, such as via shooting or allowing public hunts. On the other hand, national parks are mandated to conserve native biodiversity and ecosystems – not just one species. Examples from Sleeping Bear Dunes National Lakeshore, Great Smoky Mountains National Park, and Valley Forge National Historical Park illustrate ecological conservation challenges created by deer in eastern parks.

Two islands in Lake Michigan were added to Sleeping Bear Dunes National Lakeshore in 1970. Before being incorporated into the park and as a private hunting reserve, nine deer were introduced in 1926 to the 6,070-hectare (15,000-acre) North Manitou Island. By 1981, the deer population ballooned to 2,000 animals (30 deer/km^2, 75 deer/square mile). Deer were not introduced to the 2,020-hectare (4,990-acre) South Manitou Island. Beech-maple forest predominated on both islands. The islands were also similar in other attributes, except for the presence/absence of deer, providing a convenient comparison for evaluating potential effects of deer. In a 2004 inventory, 30 of 33 (90%) summer wildflower species were most abundant, or only found, on the deer-free South Manitou Island.[17] Plants most palatable to deer, such as Jack-in-the-pulpit (*Arisaema triphyllum*) and sweet cicely (*Osmorhiza claytonii*), were sparse on deer-inhabited North Manitou Island (Fig. 4.14).

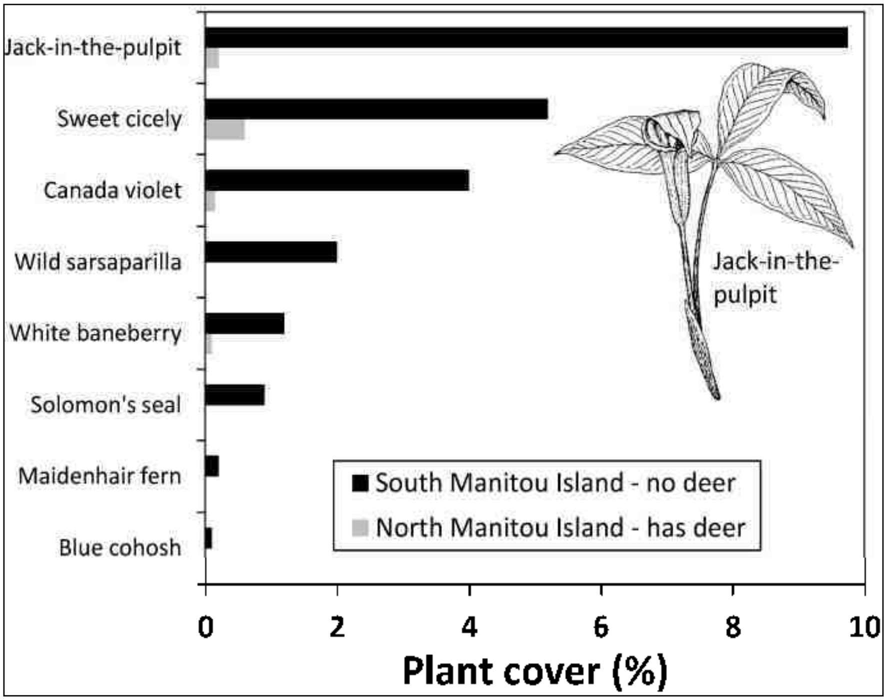


Fig. 4.14. Comparison of plant species abundance on islands with and without deer, Sleeping Bear Dunes National Lakeshore, Michigan. Data from Hurley and Flaspohler (2005). [17] *The plant sketch is from the Natural Resources Conservation Service.*

Reduced diversity of plants in the forest understory is a major loss, because the understory harbors much of the biodiversity of entire forests. To alleviate destruction of the island's diversity, the National Park Service has allowed public deer hunts since 1984. It would be interesting what would have happened if, as Isle Royale was colonized, wolves had arrived to North Manitou Island and found it populated with 2,000 deer.

To the south in Great Smoky Mountains National Park, the 2,400-hectare (5,900-acre) Cades Cove valley has contained elevated deer densities since the 1950s. Density peaked at 43 deer/km^2 (110 deer/square mile) in 1978 and remained high, despite removals by the National Park Service (Fig. 4.15). To examine effects of deer on vegetation, 15 fenced exclosures, each 12 meters by 12 meters (40 feet by 40 feet), were established in Cades Cove in 1997.[12] For comparison, 15 areas of the same size were not fenced. Plants were re-inventoried inside and outside the exclosures eight years later in 2004. Areas with deer outside the exclosures had the appearance of a "tidy forest floor," with little vegetation on the ground or within reach of deer (Fig. 4.15 photo). After eight years of protection from deer, the amount of ground covered by native plants inside the exclosures was twice that of areas open to deer. Tree seedlings were 12 times more abundant inside the exclosures. Despite these increases, native plants most sensitive to deer were recovering only slowly or were still absent inside the exclosures.[12] This is probably not surprising given that seed sources, critical to expanding plant populations, were likely reduced or eliminated from Cades Cove. One way to determine if availability of seeds limits vegetation recovery would be to plant species inside and outside the exclosures and monitor seedling establishment.

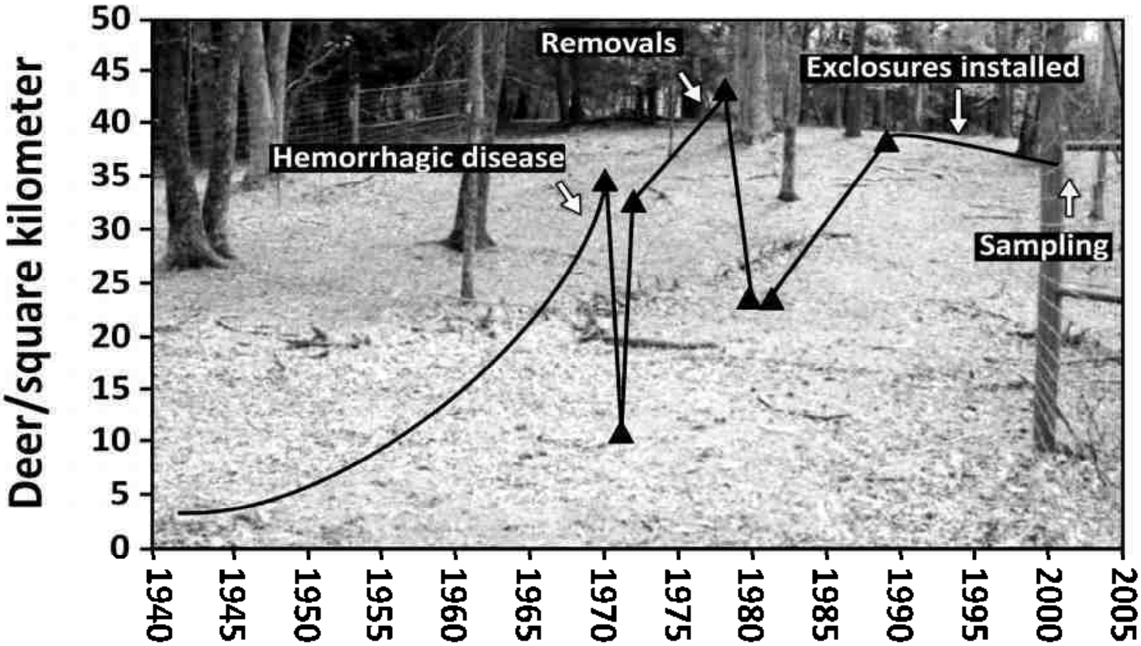


Fig. 4.15. Population size of white-tailed deer through time at Cades Cove, Great Smoky Mountains National Park. Data from Griggs et al. (2006). [12] *Photo, from Thiemann et al. (2009),*[38] *shows sparse vegetation anywhere within reach of deer (used with permission from American Midland Naturalist).*

Small historical parks have been susceptible to deer impacts, creating ecological problems as well as impeding restoring or maintaining historical authenticity of human cultural resources. The 1777-1778 Revolutionary War winter encampment of the continental army near Philadelphia, Pennsylvania, in Valley Forge National Historical Park, contained an astounding 80 deer/km^2 (200 deer/square mile) by the 2000s.[1] A 2010 inventory of deer exclosures (2 meters by 2 meters, or 7 feet by 7 feet in size), fenced 18 years earlier in 1992, revealed striking differences (Fig. 4.16). The interiors of the exclosures were small oases where tree saplings could actually grow. Areas accessible to deer were barren or dominated by "lawns" of the non-native Japanese stiltgrass (*Microstegium vimineum*), a plant deer prefer not to eat. This lawn was a monoculture compared to the more diverse vegetation inside the exclosures, which contained up to five times as many plant species. This provides an example of factors other than climate exerting control over a park. The adjacent fenced/unfenced areas shown in Fig. 4.16 were both exposed to the same climate over the 18 years, but one area was transformed by chronic impacts of deer.

As a management strategy, the National Park Service hired sharpshooters to remove 977 deer from the park between 2010 and 2012. A total of 13,527 kilograms (29,822 pounds) of venison meat was donated to the Central Pennsylvania Food Bank and provided to soup kitchens across 21 counties in Pennsylvania. To put on 29,822 pounds of meat, the deer had removed an amazing amount of vegetation from the park.

Fig. 4.16. Deer and their effects in Valley Forge National Historical Park, Pennsylvania. Top left: buck. Top right: oak forest with browse line where vegetation within reach of deer is gone. Bottom left: monoculture of non-native Japanese stiltgrass, which deer prefer not to eat. Bottom right: exclosure protected from deer enabling native plant growth. Top left is a National Park Service photo; other photos provided by M.D. Abrams.

The Future of Food Webs

Centuries of predator removal in the United States eradicated from national parks keystone species and important interactions within food webs. These are the interactions national parks are supposed to conserve. Fortunately for many predator species, recent decades have been more favorable than the preceding decades. Help from humans for the first time in literally centuries assisted the return of wolves to Yellowstone National Park. Some predator species, such as panthers, are re-colonizing parts of their former range on their own. Interestingly, and possibly in part because of the absence of wolves, the coyote has also been expanding in the eastern United States during the last century (Fig. 4.17).[10] Coyotes apparently did not occur historically in the East. The 1804 Lewis and Clark Expedition did not record coyotes until as far west as Nebraska. This raises ecological questions, like to what extent a smaller predator differs in an ecosystem compared to wolves as the former larger predator (coyotes weigh only half that of wolves).

Realistic conservation strategies for predators in today's society must manage human-wildlife interactions. This is particularly true in parks, where a specific goal is to provide people with access to nature, which includes predators. How often do predatory animals attack humans? Using news archives and other sources, one analysis found that there were 130 people reported bitten by coyotes in the United States between 1960 and 2006.[39] This translates to three people bitten by coyotes per year. In comparison, 4.5 million people are bitten by domestic dogs per year in the United States, according to the Center for Disease Control. Fatal attacks by coyotes are extremely rare, as coyotes usually retreat after initial attacks. In contrast, attacks by domestic dogs kill 25 Americans annually, and 27,000 of the attacks necessitate surgical repair to humans. Attacks to humans by other predator animals such as panthers, wolves, and bears, are also rare. This does not mean that risk is non-existent, but rather that domestic dogs pose a far greater threat to most people.

Fig. 4.17. Coyote, Chattahoochee River National Recreation Area, Georgia (National Park Service).

Additionally, not having predators poses a risk to humans. Overpopulation of deer is an example. State Farm Insurance Company (Bloomington, Illinois) reported in 2013 that the chance of an American motorist being in a deer-vehicle collision was 1 in 174 over the next 12 months. This risk is serious. If that same annual risk occurs each year, it means that one-third of drivers will be in a deer-vehicle collision during a typical 58 years of lifetime driving. This risk is also just for deer, and does not include risk of hitting other animals such as elk or moose. While risk of attack by predators is miniscule compared to other risks most humans are exposed to, reducing risk of predator attack even further can be accomplished partly through preventing wildlife from becoming accustomed to receiving food from, or being around, humans. This is why many national parks commonly post signs prohibiting feeding or approaching wildlife, to both protect humans and keep wildlife wild.

Despite some cause for optimism with some predators increasing recently, predators are so important to park ecosystems that effective conservation must be vigilant about having predators in the food web. For example, when protections are relaxed by de-listing species such as wolves from the Endangered Species Act, management responsibility often returns to states. Some states have quickly acted to establish hunting seasons to reduce wolves. After de-listing the gray wolf in Idaho and Montana, for instance, hunters immediately killed 207 wolves in these states in the first year of hunting in 2009.[27] In 2012, hunting of de-listed wolves resulted in 413 killed in Minnesota and 117 in Wisconsin. In today's society, hunting in certain cases can actually help conserve some species by ensuring enough are available to hunt. But cautious strategies are clearly needed to avoid undermining recent progress in predator conservation. Other factors, such as lead poisoning via lead bullets and fishing tackle in water, remain significant concerns for some predatory bird species.[13] Furthermore, predator conservation is linked to managing fire, non-native species, forests, freshwater, climate change, and other topics discussed in this book. As the world's top predator, humans make the conservation decisions that reverberate all the way to the bottom of the food web, affecting countless species along the way.

5 NON-NATIVE PLANTS

In 1620, the Mayflower set sail from England to North America, carrying 130 people and supplies. The ship's cargo is believed to have held crop seeds of oats, rye, barley, and other grains, typically with "weed" seeds mixed in. After the ship landed at what was to become Plymouth Colony in present-day Massachusetts, passenger William Bradford's journal also mentions the colonists planted various "garden seeds" around the settlement.[10] This transport of seeds across the ocean to North America was probably not the first. Early voyages by the Vikings in the 11th and 12th centuries, followed by Christopher Columbus's 1492 voyage, all could have introduced plants. In fact, in his second voyage from Spain to North America in 1493, Columbus brought an entire fleet (17 ships), 1,200 people, and seeds and cuttings to produce a variety of plants.[16]

Three centuries after Columbus, third U.S. president Thomas Jefferson stated in 1800 that: "The greatest service which can be rendered any country is to add a useful plant to its culture…" Jefferson had even smuggled rice seeds out of Europe – in his pockets and an offense punishable by death – to make them available to American farmers.[17] He introduced several species to the United States, including now-common broccoli, eggplant, and cauliflower. Jefferson's activities epitomize tradeoffs of interconnected global commerce and travel still unfolding today. On one hand, some plant introductions have benefited agriculture and horticulture.[25] On the other hand, intentional and unintentional plant introductions have damaged agriculture, ranching, forestry, landscaping, waterways, native species, and national parks.

Non-native species are defined as those transported by human activities to new areas (typically new continents) outside of the habitat in which the species evolved. The key point is that human activities enable a species to overcome some barrier, such as an ocean for land plants, that had prevented a species from reaching an area. Haleakala National Park (Hawaii), for example, estimates that plants in the modern era arrive at 2 million times the natural rate of introduction. The park now contains over 400 non-native plant species.

Non-native species are also commonly referred to as exotic, introduced, or invasive species, meaning they can readily invade and spread within new habitats. Because not all non-native species are invasive and some native species are, it is useful to separately define the origin of a species (native or non-native) and its invasiveness. In the original habitat in which a species evolved, other organisms commonly also have evolved to keep the species in check. The new habitat to which a species is introduced lacks these organisms or other conditions that limit a species the same way it is limited in its evolutionary habitat. As a result, non-natives have an "unfair advantage." This is why many non-native species function differently from native species and are so damaging to the new habitat.

In the lower 48 states and Alaska, introductions of plant species are generally considered to have started with European voyages to North America, such as Spanish explorations and the Mayflower, around the 1500s-1600s. In the Hawaiian Islands, the estimated 30 plant species introduced by the Polynesians starting around 1,400 years ago, plus the over 800 introduced in the modern era, are considered non-native.[29] While behavior of non-native species in the introduced habitat can differ from in their original habitat, the habitat a species most readily invades on the new continent typically is similar to its original habitat. For example, species that evolved in the deserts of Asia and introduced to North America are now most invasive in the deserts of North America.

Once a plant species is introduced, either intentionally or unintentionally, it may or may not survive in its new habitat. Many introductions fail because the species simply dies out. For species that do become established, populations often expand only slowly at first. But once "critical mass" is reached, populations can spread rapidly. This delayed expansion is termed a lag phase.[38] Lag phases complicate predicting which non-native species will become invasive and destructive, versus those that will simply die out or remain innocuous.

Not all non-native species create major problems for native species in the introduced habitat. For example, common dandelion (*Taraxacum officinale*), present in all 50 states, is thought to have arrived in North America from Europe/Asia. Its timing and mode of introduction are uncertain, with some researchers claiming it arrived with the Vikings around the year 1000 AD.[36] The first recorded observation was in 1672 in New England. While in certain areas dandelion may harm growth of native species, generally dandelion remains at low abundance and minimally influences ecosystems. Since dandelion has been in North America for at least four centuries and is not highly invasive, it is sometimes called naturalized, meaning it largely functions as a native species.

Other non-native plants seemingly provide certain benefits. Saltcedar (*Tamarix ramosissima*), originally from Europe/Asia, is a well-chronicled example. It was intentionally introduced by 1823 (it was offered for sale then by New York City nurseries)[34] as an ornamental and with the idea it would stabilize soil along riverbanks. Saltcedar subsequently spread along rivers in the semi-arid western United States.[35] Some bird species do use saltcedar as habitat, which could be considered a benefit. However, just because saltcedar provides some habitat value does not mean its positives outweigh its negatives. Forming monocultures of minimal plant diversity and fueling unnatural fires are some of saltcedar's many negatives. Given the opportunity, native trees provide bird habitat as good as or better than saltcedar. When considering effects of non-native plants on ecosystems, comprehensive perspectives including effects on numerous native species, ecological processes like fire, and water resources provide the most accurate appraisal.[31]

Some introduced species have greater impacts than others and funding for control measures is limited, so park managers commonly need to prioritize which non-native species to manage. Generally, it is better economically and ecologically to detect non-native species early in their invasion and implement control treatments when infestations are small. In fact, some parks, such as in the Great Lakes region, have developed mobile-phone applications for park visitors to report new non-native plant infestations.[14] Based on analyzing outcomes of managing 53 plant infestations of various sizes in California, one study found that infestations smaller than 0.1 hectares (0.25 acres) could be completely eradicated using 60 labor hours.[33] But once infestations covered tens of acres, financial costs escalated and success decreased. For large infestations, attention switches to simply containing them. Even if already abundant, though, non-native species that are the most damaging to a park should be prioritized to at least lessen their impact.

In addition to prioritizing species, park managers must prioritize particular areas within parks for treatment. Keeping non-native plants away from habitats rich in native species, for instance, is commonly an objective.[5] Habitats that are disturbed – such as where soil is bulldozed to remove existing vegetation or where a forest is cut – are typically readily invaded by non-native plants. As a result, roadsides and other disturbed areas are frequently prioritized for surveying for new invaders. The following sections describe the status of non-native plant invasion in parks, examples of effects non-native plants are having, and management being implemented to protect parks from non-native plants.

Increasing Numbers of Non-Native Species in Yellowstone and Glacier

Invasion by non-native plants of national parks started early and continues today. By 1886, only 14 years after designation of Yellowstone National Park, at least one non-native plant species occurred there.[40] It is possible that additional non-native species infested the park then, because finding individual plants within large parks like Yellowstone is not easy. Modes of introduction of the early invaders are poorly understood, but likely included seeds in feed of horses and livestock, attachment to human clothes and vehicles, spreading via wind and water, or intentional plantings. Some non-native landscaping plantings are conserved as cultural features within parks, but in general, planting non-native vegetation is now inconsistent with National Park Service policy.

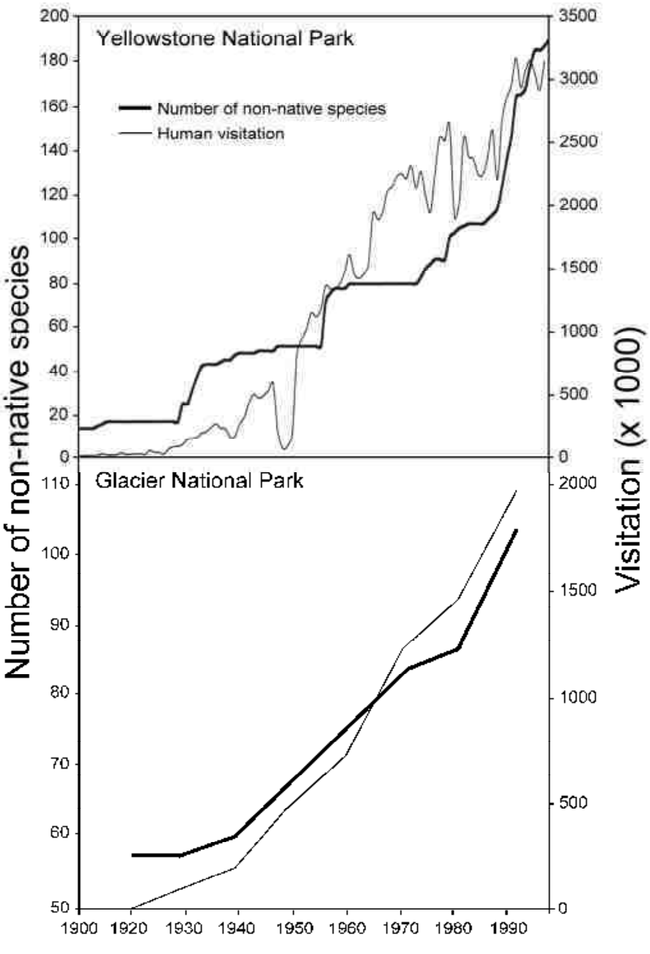


Fig. 5.1. Non-native plants increase as park visitation increases. Data from Whipple (2001)[40] and Lesica et al. (1993).[22]

Further north in Montana, Glacier National Park already contained at least 57 non-native plant species by 1920.[22] This was only 10 years after the park was established. Most of the species originated from Europe and Asia.

Repeated botanical inventories indicate that the number of invading species has steadily increased through time in both parks (Fig. 5.1). The increase has correlated with increasing human visitation. Botanical inventories have become more thorough over time, so some of the measured non-native increase might result from that. Nevertheless, it is not surprising that with millions of people visiting from all states and internationally, visitation is related to introductions of non-native plants. Today, over 200 non-native plant species inhabit Yellowstone and at least 127 inhabit Glacier National Park.

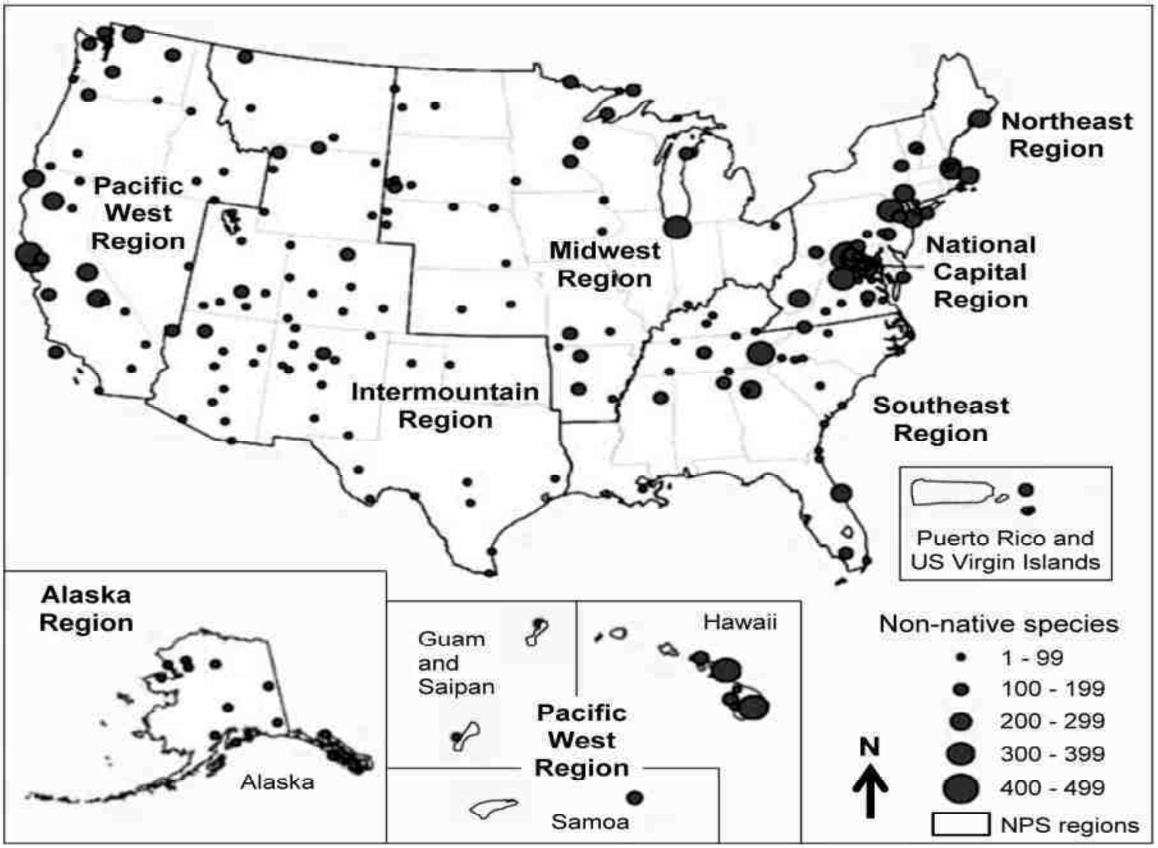


Fig. 5.2. Number of non-native plant species in 216 of the national parks. Each circle, scaled to the number of non-native plant species, is a national park unit. Adapted from Allen et al. (2009).[9]

How Many Non-Native Plant Species Have Invaded National Parks?

A 2002 analysis of 216 parks reported that a total of 3,756 non-native plant species had invaded them (Fig. 5.2). The maximum number of non-native plant species within a park was an astounding 483 species, in Chesapeake and Ohio Canal National Historical Park (Maryland/West Virginia). This park is heavily invaded via its long history as a transportation corridor, serving as a vector for seeds, and its "waterfront" location favorable to plants. Many other parks are also invaded by at least 300 species. Some of the most heavily invaded parks are Hawaii Volcanoes and Haleakala National Parks in Hawaii, Golden Gate National Recreation Area and Point Reyes National Seashore in California, Indiana Dunes National Lakeshore, Great Smoky Mountains National Park in North Carolina/Tennessee, and several small parks around Washington, D.C.

A key point is that *every one* of the 216 parks analyzed contained non-native plant species. While the analysis called for further surveying to verify non-native plant occurrences (such as whether certain species disappeared, or appeared, over time), several other park-specific surveys have independently found similar high levels of invasion.[8,32]

There is little doubt that non-native plants have invaded all or essentially all 408 parks in the national park system. If the non-native species simply remained at low abundance and just took up a little space without displacing native species, there may be less cause for concern. Unfortunately, many non-native plant species have not remained docile. In addition to increasing numbers of non-native species, two examples below illustrate increasing dominance of total park vegetation by non-native plants.

Ongoing Plant Invasion of Chickamauga Military Park and Knife River

On September 19-20, 1863, the Battle of Chickamauga raged between Union and Confederate soldiers during the American Civil War and resulted in the second most casualties of the war. The battle occurred near Chickamauga Creek, in northwestern Georgia and southeastern Tennessee. Part of the battlefield was in prairie and open eastern red-cedar (*Juniperus virginiana*) woodland on rocky, limestone soils. Today, these unique limestone habitats support rare plant species within the 3,626-hectare (8,960-acre) Chickamauga and Chattanooga National Military Park.

Fig. 5.3. Cedar woodland on limestone soil, Chickamauga and Chattanooga National Military Park, Georgia. Photo from a National Park Service-sponsored vegetation inventory by T.E. Govus and R.D. White.

In 1993, and again in 2006 and 2008, researchers measured vegetation on 11 limestone sites throughout the park.[37] During the 15-year period, the cover of woody species nearly doubled, from 36% to 64%. One of the increasers was Chinese privet (*Ligustrum sinense*), expanding from 1% cover in 1993 to 5% cover 15 years later. This non-native shrub was intentionally introduced as an ornamental to the United States in the mid-1800s.[21] The increase in shrub species apparently occurred at the expense of herbaceous (wildflower and grass) plants, which declined by 70%. Several of these declining species, such as glade violet (*Viola egglestonii*), are rare and priorities for conservation. While many factors can influence changes in native species over time, expanding dominance of the non-native shrub Chinese privet has correlated with declines in rare native plants.

Knife River Indian Villages National Historic Site is another park exemplifying increasing non-native plant invasion concurrent with declining native species. Established in 1974, the 711-hectare (1,758-acre) park is at the confluence of the Missouri and Knife Rivers in central North Dakota. The park conserves earthlodge remains that are among the best examples of Native American villages in the northern Great Plains (Fig. 5.4). Awatixa Village, in the southern part of the present-day park, was home to 16-year-old Sacagawea in 1804. She was to become well known for her role in the Lewis and Clark expedition, which arrived in the village in 1804.

Fig. 5.4. Knife River (left) and earthlodge (right), Knife River Indian Villages National Historic Site, North Dakota. Photos from the National Park Service (right photo by C. Hansen).

About 180 years later as a national park unit, vegetation was inventoried at 27 sites in the park's grassland and forest. This 1984 inventory was repeated, using the same methods, in 2007.[15] Non-native perennial grasses increased, while natives decreased, between the inventories. The two non-native grasses – smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*) – were sparse in 1984 but dominated the vegetation 23 years later. Meanwhile, native grasses went from the dominant plant cover, to only a third of plant cover in 2007. Native wildflowers similarly declined by two-thirds.

Exactly how non-native plants might be reducing native species in the park is not known. Non-natives can impact native plants by altering the soil such as through exuding chemicals; competitively usurping resources like water, nutrients, and light to "starve" native plants; occupying space to displace natives; increasing disturbance like soil erosion or fire; and altering insect pollination of native plants.[27]

Reduced Animal Use of Invaded Habitat in Theodore Roosevelt National Park

One way non-native plants alter ecosystems is through changing forage availability or vegetation structure, which affects favorability of habitat for animals. For example, different species of forage plants vary in nutrition content, chemical composition, and digestibility. To understand how non-native plant invasion potentially changed wildlife habitat, researchers examined bison (*Bos bison*) and elk (*Cervus elaphus*) in Theodore Roosevelt National Park.[39] In the Great Plains of North Dakota, the park encompasses 28,517 hectares (70,467 acres) of mostly grassland. Native grasslands have become heavily invaded by the non-native leafy spurge (*Euphorbia esula*), smooth brome (*Bromus inermis*), Japanese brome (*Bromus japonicus*), and cheatgrass (*Bromus tectorum*). Using abundance of fecal pellets as an indicator of habitat use, researchers compared animal use of four habitat areas invaded by non-native plants and four areas comparable but dominated by native vegetation (Fig. 5.5).

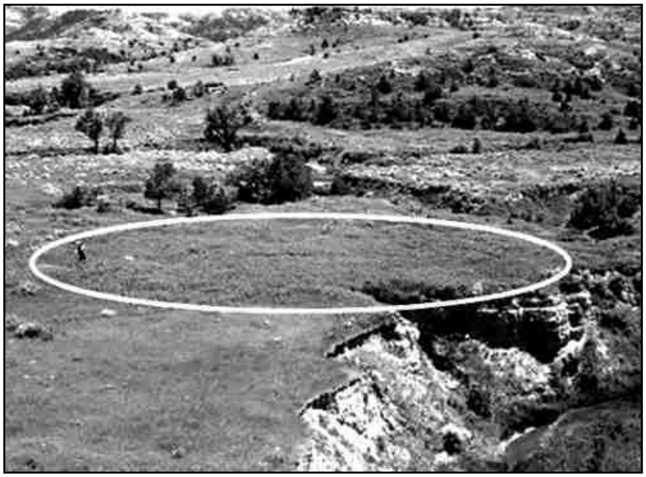


Fig. 5.5. Leafy spurge infestation, Theodore Roosevelt National Park, North Dakota (National Park Service photo).

Habitat use by bison and elk varied depending on the year and the particular non-native plant species infesting the habitat. Overall, animal use was lower where non-native plants predominated. Bison use of two leafy spurge-infested habitats was 83% less than use of non-infested sites. Similarly, elk used a leafy spurge-infested habitat 81% less than a native habitat. Leafy spurge is often avoided by herbivores because it contains a milky, latex substance. Differing from the leafy spurge areas, bison and elk still used the non-native grass areas, perhaps because they had to.

These observations illustrate the importance of having a comprehensive view of the effects of non-native plants to entire park ecosystems. For example, even if bison and elk use of the non-native grass areas did not differ from native areas, this does not mean that quality of forage was equal and that other components of the ecosystem, such as native plants, avoided being harmed by non-native plants. Indeed, native plants were 70% less abundant when the cover of non-native plants exceeded 50%.[39]

As of 2014, the park reported that leafy spurge infests 1,600 hectares (4,000 acres), or 10%, of the southern unit of the park. Given the cumulative negative impacts to park ecosystems, treatments are ongoing to reduce leafy spurge via herbicide and the biocontrol insect flea beetle (*Aphthona* species, which feed on roots). The park further noted that leafy spurge is estimated to cost North Dakota farmers and ranchers over $27 million annually in control treatments and lost revenue. This is a good example where reducing non-native plants benefits both national parks and surrounding agricultural and ranching landscapes.

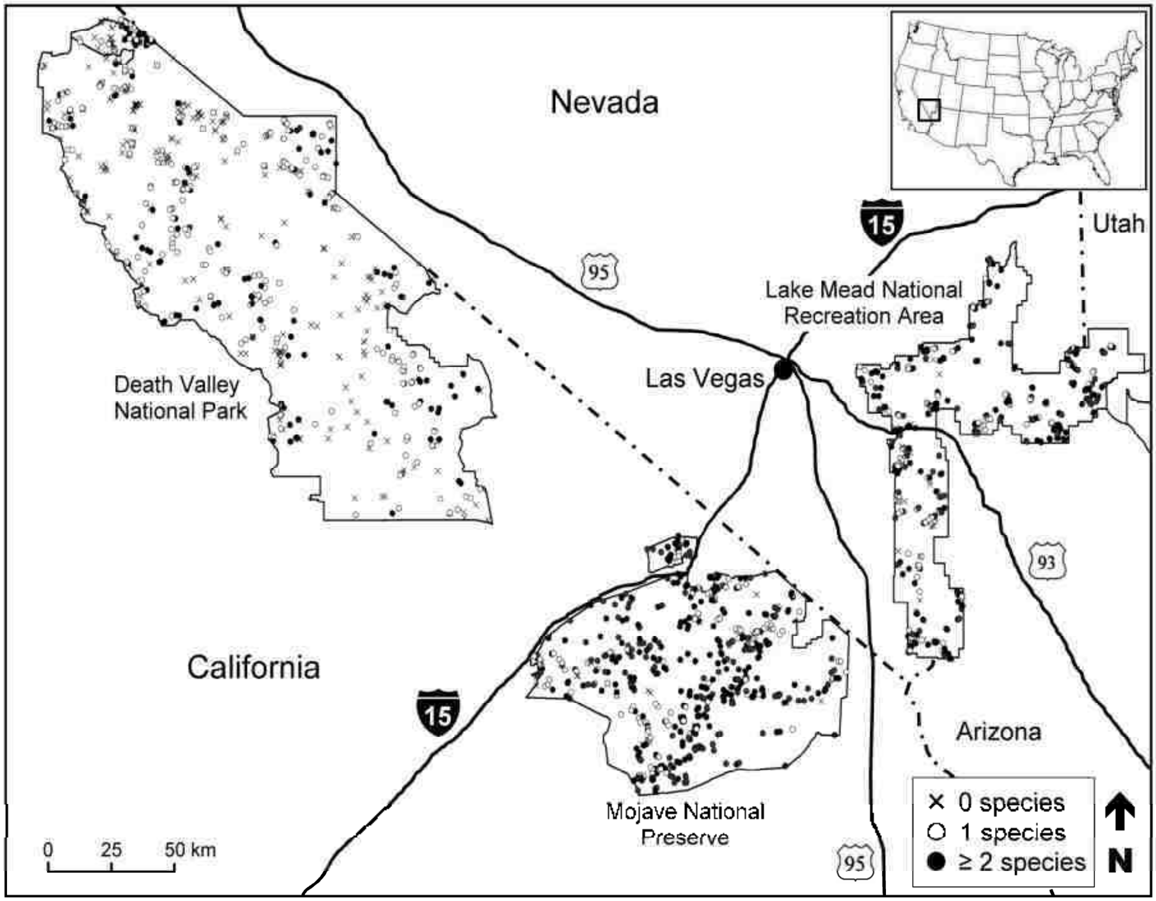


Fig. 5.6. Inventory of non-native plants at 1,662 locations in three national parks in the Mojave Desert. Each location is marked as having 0, 1, or 2 or more non-native plant species. From Abella et al. (2015).[8]

Non-Native Plants, Fire, and the Desert Tortoise in Southwestern Desert Parks

The desert Southwest is particularly significant because it is one of the few remaining regions in the lower 48 states still containing large and relatively unfragmented landscapes. The Mojave Desert, covering parts of California, Nevada, Utah, and Arizona, contains the largest national park outside of Alaska: Death Valley National Park, at 1.3 million hectares (3.3 million acres). Two of the other four largest national parks in the lower 48 states are also in the Mojave Desert: Mojave National Preserve (643,000 hectares, 1.6 million acres), and Lake Mead National Recreation Area (563,000 hectares, 1.4 million acres). In fact, the only park in the top four not in the Mojave Desert is Yellowstone National Park (900,000 hectares, 2.2 million acres). Collectively, the 2.5 million hectares (6 million acres) of these three Mojave Desert parks comprise 23% of the entire national park land in the lower 48 states. To put in perspective how large these parks are, the 14,473-hectare (35,763-acre) Acadia National Park in Maine is itself a moderately large park. By area, 93 Acadia National Parks would fit inside just Death Valley National Park.

Wilderness character is a major value of these desert parks. Death Valley contains the most congressionally designated wilderness (1.2 million hectares, 3 million acres, or 91% of the park) of any park outside Alaska. Fifty-one percent of Mojave National Preserve and 13% of Lake Mead National Recreation Area is designated wilderness. These massive parks uniquely conserve entire portions of regional native ecosystems, but they are threatened by non-native plant invasion and other disturbances.

Two of the major non-native plants invading the Mojave Desert are red brome (*Bromus rubens*) and Mediterranean grass (*Schismus* species). These are annual grasses that complete their life cycle within one year, by germinating following winter rains (November through March) and rapidly growing and producing seed in spring (February through April). The arid environment of the Mojave Desert is little barrier to them, because the species evolved in the dry regions of Africa, Europe, and Asia. In an example of the "unfair" advantage the non-native plants have over native plants, animals such as feral burros (*Equus asinus*) do eat the non-natives but preferentially eat native plants when available.[1]

Just how widespread are these non-native plants? An inventory of 1,662 locations, each 0.1 hectares (0.25 acres) in size, was done in 2009-2010 throughout Death Valley National Park, Mojave National Preserve, and Lake Mead National Recreation Area. Red brome infested 60% of the locations and Mediterranean grass 28%. Over 81% (1,358 out of 1,662) of the locations contained at least one non-native plant species (Fig. 5.6).

One way the invasion has affected parks is by changing fuels. Fire is thought to have been rare in the Mojave Desert owing to sparse, discontinuous fuels. But non-native grasses produce copious, persistent fuel in the spaces between and below perennial plants, augmenting native fuels and enabling fire to spread across the landscape. Between 1992 and 2011, 786 fires burned 56,000 hectares (140,000 acres) of national park land in the Mojave Desert (Fig. 5.7). Burning 25,672 hectares (63,436 acres), the largest was the Wildhorse Fire in Mojave National Preserve. This fire occurred in the record fire year of 2005, when a particularly wet winter stimulated plant growth and was followed by a dry summer.

With little evolutionary history of fire, most native Mojave Desert plants have not evolved adaptations to fire such as thick bark, prolific re-sprouting ability, or seeds triggered by fire to germinate.[3] As a result, fires devastate mature desert shrublands. For example, Joshua trees (*Yucca brevifolia*) in Joshua Tree National Park, California, were reduced or eliminated after seven of eight fires.[26] A paramount concern is that fires and climate change are interacting to both reduce the number of existing Joshua trees and limit recruitment of new trees. If fires continue, few Joshua trees may be around to even experience a future climate. Joshua trees are noticeably depauperate in burned areas of the park today (Fig. 5.8).

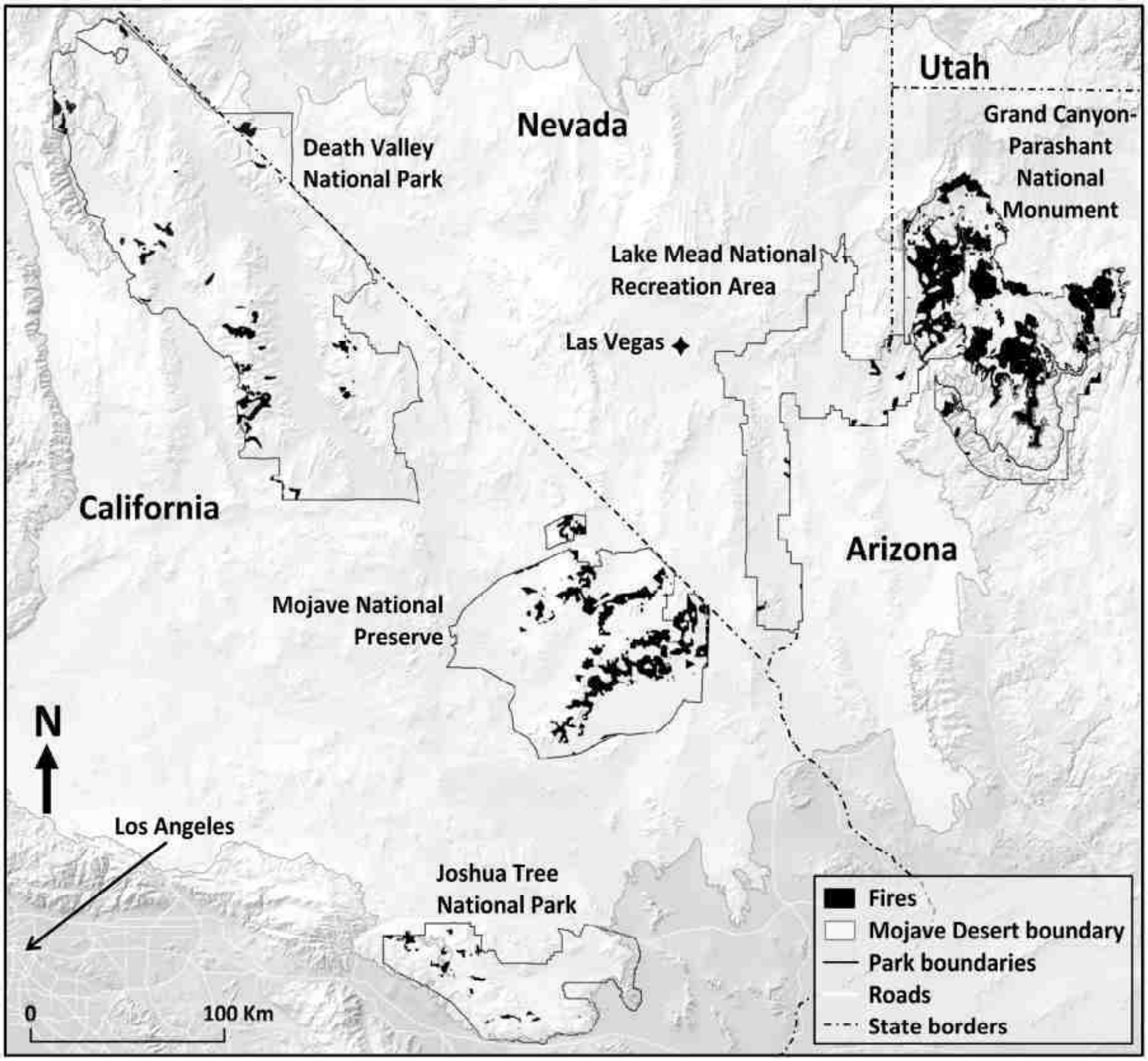


Fig. 5.7. Wildfires within Mojave Desert parks between 1992 and 2011. Adapted from a map provided by B.G. Dickson (Conservation Science Partners) using data from Hegeman et al. (2014).[19]

The desert tortoise (*Gopherus agassizii*) is another iconic species likely being affected by non-native plant invasion in the Mojave Desert. In addition to fire destroying the shrubs that tortoises construct burrows beneath, non-native plants have altered available forage. Quality forage during the spring growing season is essential to the health of desert tortoises, as they procure essential nutrients from forage plants. One study compared the forage available to what tortoises actually ate, to identify plants tortoises prefer for meeting their dietary needs. Of the 239,000 non-native grass plants they encountered, tortoises ate only 42 of them (0.02%).[30] In comparison, tortoises ate 120 of the 346 (35%) plants of the native wildflower desert plantain (*Plantago ovata*) they encountered. Remarkably, the native species was three times as important in tortoise diets even though the native was 700 times less abundant on the landscape. The desert tortoise is currently listed as threatened under the Endangered Species Act. Sustainability of the species probably partly depends on whether non-native plants can be reduced (Fig. 5.9).

Fig. 5.8. Unburned (top) and burned (bottom) Joshua tree habitat along Keys View Road, Joshua Tree National Park, California. Photos by S.R. Abella.

Fig. 5.9. Desert tortoise near Mojave National Preserve. Native annual plants are key forage for tortoises. Photo by S.R. Abella.

Solutions: Projects Successfully Managing Non-Native Plants

The National Park Service conducts non-native plant management in two main ways: through the Exotic Plant Management Team (EPMT) program and individual park-based projects. The EPMT has 16 teams that each have their own region of parks they serve. Each team spends days to weeks treating non-native plants within particular parks in efforts typically coordinated with park managers. This uniquely designed program operates on a base budget of about $5 million annually and is essentially the only resource that some parks, especially the smaller ones, have for managing non-native plants. Along with the EPMT, individual park projects, initiated by local park managers, comprise most of the non-native plant management by the National Park Service. These projects are often funded through grants or done with citizen volunteers and are rarely funded as part of base allocations.

A 2014 status assessment of documented outcomes of non-native plant management projects between 1984 and 2013 found that well-designed projects were extremely effective at reducing non-native plants at particular sites in national parks.[7] The assessment included 56 projects, performed in 35 parks in 20 states, and one in a U.S. territory (National Park of American Samoa). The 157 non-native species targeted in these projects ranged from the annual grass red brome in Zion National Park (Utah), to maidenhair vine (*Muehlenbeckia complexa*) in Golden Gate National Recreation Area (California) and the tree tamiligi (*Falcataria moluccana*) in tropical forests of National Park of American Samoa. Treatments included traditional ones such as herbicide, pulling by hand, cutting, mowing, and prescribed fire. Less-common treatments were also implemented, such as solarization (covering the ground with materials to allow the sun to heat and kill seeds), adding carbon like mulches to the soil (thus feeding soil microbes so they use nutrients then unavailable to plants), and planting native species to compete with non-native plants.[4]

At least one treatment reduced the targeted non-native plant species in 87% of the projects.[7] While few overall failures occurred, extensive experimentation was often needed to identify treatments that reduced non-natives while minimizing damage to native species. Of the 30 projects assessing treatment effects on native plants, only two (7%) reduced native plants after treatment, while 40% found no change and 53% increased natives. When a goal was to conserve existing mature native communities, such as Saguaro cactus (*Carnegiea gigantea*) landscapes of Saguaro National Park, increasing native species after treatments was not always desirable.[6] In other cases, the short-term assessment period after treatment of a few years simply might not have been long enough for native species to recover, especially if these were dry years. Slight non-target impacts to native plants may even be worth it, compared to impacts that would occur with unabated non-native plant invasion.

Fig. 5.10. Monocultures of non-native ripgut brome (bottom left) and black mustard (top right) in Cheeseboro Canyon, Santa Monica Mountains National Recreation Area, California. Before prescribed fire, both areas were ripgut brome like the bottom left. After the top-right area was burned, it converted from one non-native monoculture to another. From Moyes et al. (2005),[28] with permission from John Wiley and Sons.

Multiple non-native plant species on a site complicated management. For example, different species that grow at different times can necessitate multiple treatments appropriately timed throughout a year. Among 16 projects that evaluated responses of non-native plants other than those targeted for treatment, half found that other non-native plants increased after the focal non-native species was treated. A dramatic example was in Santa Monica Mountains National Recreation Area of California (Fig. 5.10). Treatments reduced a non-native grassland of ripgut brome (*Bromus diandrus*), only to have it replaced by the non-native wildflower black mustard (*Brassica nigra*). This frustrating aspect of non-native plant management is termed secondary invasion, and highlights the utility of early detection for preventing new invaders from becoming established in the first place.

A project at Sleeping Bear Dunes National Lakeshore (Michigan) illustrated how managing non-native plants conserved native plant pollination processes. The non-native wildflower baby's breath (*Gypsophila paniculata*) invaded sand dunes along the Lake Michigan shoreline. The dunes were habitat for sand dune thistles (*Cirsium pitcheri*), rare native plants listed as threatened under the Endangered Species Act. Using shovels, baby's breath plants were removed in 2008 on some dunes and were not removed on others. Removing baby's breath tripled visits by pollinator insects to the native sand dune thistles (Fig. 5.11). Researchers suspected that the numerous flowers of baby's breath had been "detouring" pollinators away from the native plants.[12]

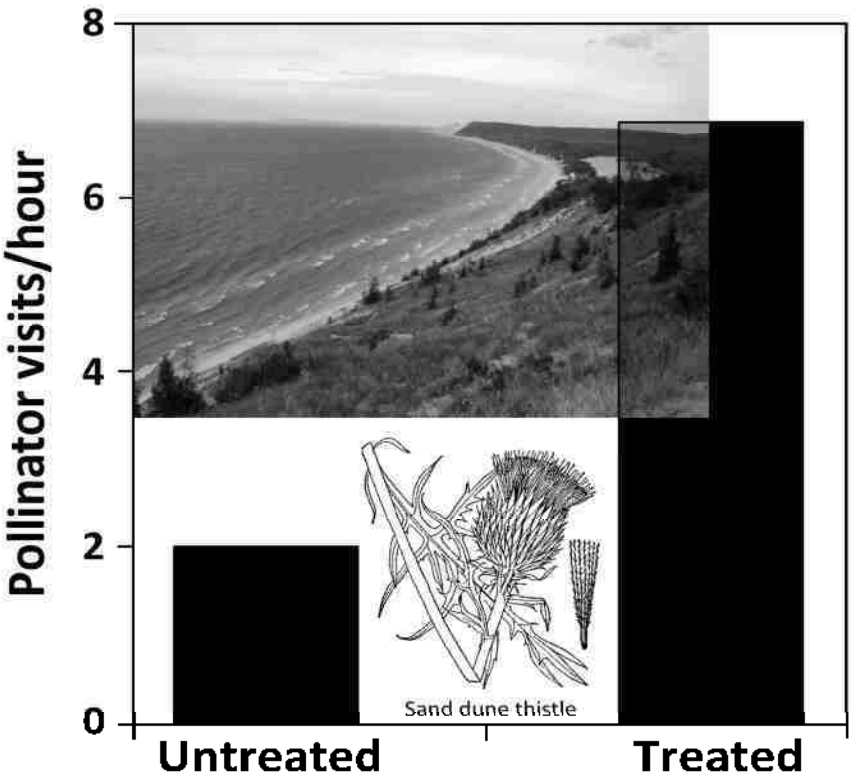


Fig. 5.11. Removing non-native baby's breath plants increased insect pollinator visits to sand dune thistle, a rare native species in Sleeping Bear Dunes National Lakeshore, Michigan. Data from Baskett et al. (2011).[12] The photo is along the Empire Bluff Trail, Lake Michigan shoreline dunes (National Park Service). The line drawing of sand dune thistle is from the PLANTS Database.

Conserving Parks under Escalating Plant Invasion

A troubling projection with non-native plant invasion is an "extinction debt," where short-term persistence of native species masks eventual extinctions.[18] Even though thousands of non-native plant species have already invaded significant parts of national parks, it is possible that the invasion is still only in its early stages. In fact, exponential population growth at the end of the lag phase may not even have occurred for many species.

Stabilizing the situation through aggressive early detection and treatment to limit new invaders from entering parks or becoming established at new sites within parks remains a priority. For example, a 2003-2006 early detection survey of 3,300 km (2,000 miles) of roadsides within and around Lake Mead National Recreation Area resulted in removing over 37,000 individual non-native plants.[2] This potentially averted additional species invasions.

Treating tenacious, firmly established invaders will likely require more resources than the National Park Service is currently allocating. For example, the Japanese-origin Morrow's honeysuckle (*Lonicera morrowii*) occurred at a high density of 176,000 stems/hectare (71,000/acre) at Fort Necessity National Battlefield in Pennsylvania. Reducing density of this species by just half, using herbicide, required 56 labor hours/hectare (23 hours/acre) and cost $770/hectare ($312/acre).[24] When plants were dug out by hand instead of using herbicide, cost escalated to 930 labor hours/hectare (400 hours/acre) and $9,300/hectare ($3,800/acre). This also was just for a one-time treatment. Repeated treatments, followed by maintenance management, would be required. This is a difficult situation, because not performing treatments would perpetuate ecological damage and limit historical authenticity of the park. Morrow's honeysuckle was not dominant on the 1754 battlefield.

Fig. 5.12. From 16 km (10 miles) within Mojave National Preserve along Ivanpah Road, view of the Ivanpah Solar Electric Generating System. This site opened in February 2014, just north of the preserve. Expansion of the development was ongoing in 2015. Photo by S.R. Abella, November 2014.

Also troubling from a plant invasion perspective is ongoing or proposed energy development surrounding national parks. The infrastructure and roads required for natural gas extraction ("fracking") create soil disturbance. A general principle of non-native plant ecology is that disturbance often promotes non-native plant invasions. At fracking sites in Wyoming, the proportion of non-native species was 50% to 75% greater around disturbed well pads and processed water discharge areas, compared to off-site.[13]

Turning publicly held lands over to private companies to build solar energy industrial sites is proposed surrounding many southwestern desert parks. A new industrial site near Mojave National Preserve has altered the park's scenery, and transport of non-native plant species is a concern with ongoing construction of these industrial sites (Fig. 5.12).

While these observations highlight serious concerns, several tactics could enhance management of non-native plants. For example, precautions to protect the United States as a whole from harmful non-native species have been slow to develop.[23] In fact, in 2005 the federal Agricultural Research Service released a cold-hardy version of the non-native buffelgrass (*Pennisetum ciliare*), promoted as livestock forage.[20] Meanwhile, other federal agencies, including the National Park Service (like Big Bend and Saguaro National Parks), are struggling to prevent this species from damaging federal lands (Fig. 5.13).

Additionally, many non-native plants sold for landscaping have damaged wildlands.[25] Native plants are as good or better for many landscaping purposes, and avoid deleterious off-site impacts.

Fig. 5.13. Arrow marking saguaro cactus engulfed in non-native buffelgrass, Saguaro National Park, Arizona. Photo from Abella et al. (2013).[6]

To increase effectiveness with limited budgets, non-native plant management can be integrated with other management activities. For example, park staff, as well as visitors, can report new infestations when visiting sites throughout parks.[14] Further integrating fire management programs with non-native plant programs is important, because non-native plants can facilitate fire, such as in desert parks. Another reason managing non-native plants needs to be a priority is to avoid undermining other park operations. Major effort could be expended to move Joshua trees to high elevations of Joshua Tree National Park as part of managing for climate change, only to have the trees burn up in non-native grass-fueled fires.[11] Invasion by non-native plants is a pervasive threat facing every land park. The National Park Service has shown the capability to reduce this threat, at least in particular areas within parks, when tackling the threat is prioritized and resources for management allocated.[7]

6 NON-NATIVE FAUNA

During 20 weeks of heightened inspection of air cargo entering Kahului Airport, Maui, the Hawaii Department of Agriculture intercepted 279 insect species, 125 of which were not established in Hawaii.[15] Nationally, species interceptions recorded by the Port Information Network database, maintained by the U.S. Animal and Plant Health Inspection Service, also were numerous. Between 1984 and 2000, over 725,000 interceptions occurred at 42 U.S. airports, 25 maritime ports, and 33 land border sites (with Mexico and Canada).[21] Interceptions included "stowaways" in airline baggage, ship cargo, and packing material – such as in wood or soil used to package plants. Over 2,300 species were intercepted, ranging from insects to mollusks, from 200 countries. The many different ways (including in cargo or the unauthorized pet trade), vectors (ships, planes, cars, and passengers), and locations that non-native species can enter the United States seem overwhelming.

An unknown number of non-native species inhabit the United States, with one estimate of 50,000 species.[26] Regardless of the exact number of introduced species, national park boundaries have been porous to many non-native faunal species (insects and animals). Non-native faunal species currently in parks span those intentionally and unintentionally introduced to the United States. The following examples highlight non-native faunal species in parks and the challenges these species pose to conserving natural ecosystems.

Invasion of the Ants: Haleakala National Park

The Hawaiian Islands are a fascinating case of species evolution. Remoteness and isolation of the islands limited their pre-historic colonization by plants and animals. As a result, Hawaii's flora and fauna evolved from relatively few successful arrivals. Perhaps only 400 insect species successfully colonized the islands pre-historically. Evolution subsequently diversified these species to a native insect fauna exceeding 10,000 species.[15] Researchers do not believe that Hawaii has any native ant species, as ants were not among the successful colonizers. On land masses with native ants, ants strongly influence ecosystems such as through nutrient and energy flow via eating other insects, scavenging food sources, and aerating soil. Interestingly, some of these functions normally filled by ants are filled by other insects in Hawaii. Moreover, native Hawaiian insects have not needed to evolve mechanisms to co-exist with ants, which often are predators of other insects.

Human activities in recent centuries, however, have introduced around 50 species of ants to Hawaii.[15] The Argentine ant (*Linepithema humile*) is on the list of the most notorious, which also includes various fire ants (such as *Solenopsis invicta*). Originally from South America, the Argentine ant has spread to all continents except Antarctica. The species is viewed as problematic in California (including Channel Islands National Park),[27] the southeastern United States, Australia, Europe, and South Africa. In Hawaii, the argentine ant was established on Oahu by 1940 and invaded other Hawaiian Islands by the 1950s.

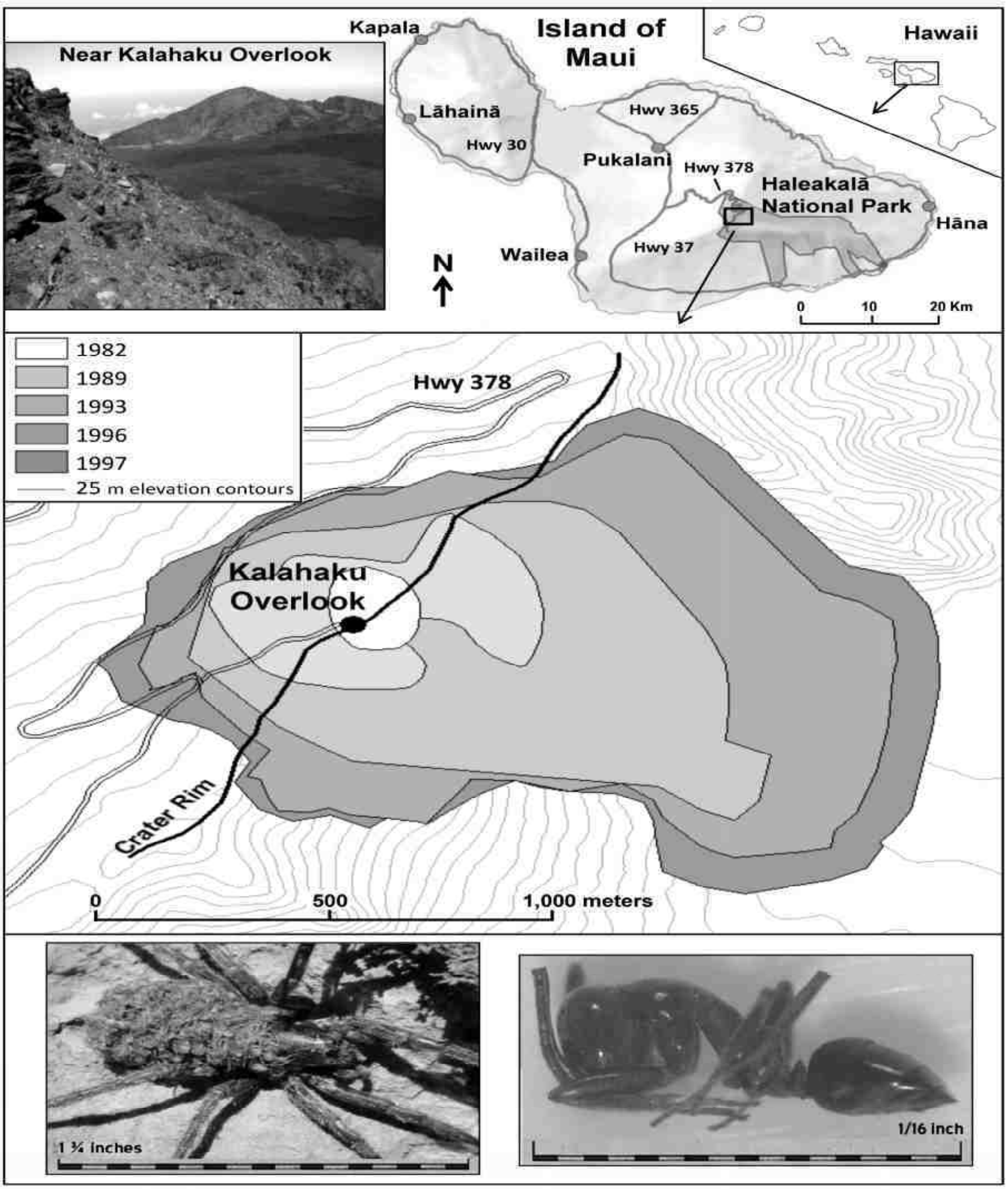


Fig. 6.1. Invasion of Argentine ants in Haleakala National Park (spread map adapted from Krushelnycky et al. 2011).[17] *Bottom left photo: Hawaiian wolf spider (with babies on its back), a native species interacting with Argentine ants. Bottom right photo: Argentine ant. Photos courtesy of F. and K. Starr.*

Argentine ants were first detected in Haleakala National Park, on Maui, at Hosmer Grove in 1967.[17] Thirteen years later, the ant population covered 165 hectares (400 acres) and overran the park headquarters. Argentine ants do not have a flying component for mating like some ant species do. Instead, Argentine ants spread on foot via queen ants moving with a group of worker ants. The ant population spread at sometimes 90 meters/year (300 feet/year). In 1982, a second population was discovered in the park's higher elevations, near Kalahaku Overlook. From 1982 to 1997, both populations expanded to cover 550 hectares (1,400 acres), shown for the Kalahaku population in Fig. 6.1.

Consistent with an insect fauna evolved without predatory ants, native Hawaiian insect assemblages are decimated in areas invaded by Argentine ants. In Haleakala National Park, researchers used pitfall traps, consisting of buried containers collecting specimens, to compare the insect community in areas invaded and not invaded by Argentine ants. Native insects including moths (*Agrotis* species), beetles (*Mecyclothorax* and *Blackburnia* species), Hawaiian wolf spiders (*Lycosa hawaiiensis*), and bees (*Hylaeus* species) were sparser in ant-invaded areas.[6] An updated assessment in the mid-2000s produced similar results: the number of native insect species was halved in ant-invaded compared to non-invaded areas.[16]

A major concern is extinction of native species, possibly before some species are even known to science. Many native Hawaiian species have restricted distributions susceptible to being overrun by mobile ant invasion fronts. There are some native insects apparently capable of co-existing with ants. Whether Argentine ants will become a dominant evolutionary force, filtering which native insect species persist, is unclear.[15]

Without management intervention, it was estimated that Argentine ants could eventually invade half the park.[17] Extensive experimentation began in the mid-1990s to develop techniques to reduce Argentine ants, while minimizing impacts to native species. A main focus was using toxic baits to lure Argentine ants. The ants are attracted to sugary foods, such as plant nectar, so some tests used sugar-water baits with a toxicant mixed in. Development of baits is tricky, because they must be attractive to ants, but contain an undetectable and slow-acting toxicant, so the ants transport it to nests for sharing with queens. Between 1994 and 2009, 30 types of baits were tested. Some were partially effective at slowing the ants' spread, but none were completely effective. Searching for other candidate baits has continued. A major goal was simply slowing the invasion to delay impacts to park resources and buy time for improvements in treatments. With Argentine ants invading the Holua campground in recent years, raising risk of spread by humans to distant parts of the park, containment strategies remain critical.

Earthworm Invasion of Great Lakes Parks

Aren't earthworms good for the soil? It depends. Over 3,500 species of earthworms are described globally, with about 100 species native to the United States.[14] However, no earthworms are considered native to the northern United States and Canada, which were covered by glacial ice during the Wisconsin Glaciation ending 11,000 years ago (Fig. 6.2 top right). Any native earthworms in these areas are thought to have been eliminated by glaciation. Recolonization of previously glaciated habitat is apparently slow, as native earthworm populations expand perhaps only 5 meters/year (16 feet/year).

In recent centuries, humans have introduced 50 species of non-native earthworms to the United States from Europe, Asia, and elsewhere.[14] Some of the main vectors of earthworm introductions are bait for fishing, composting, horticulture (such as earthworms in potting soil), and agriculture.

These introductions have created two situations: 1) areas with both native and non-native earthworms; and 2) areas that previously had no earthworms since after glaciation, but that now have non-native earthworms. Great Smoky Mountains National Park represents the first situation, as the park was not glaciated and has some native earthworms. A 2007 assessment in the western part of the park, near Highway 129 and Lake Chilhowee, found that a native earthworm-inhabited area was invaded by *Amynthas agrestis*, an Asian earthworm.[30] In invaded areas, the number of native millipede species in the soil was 75% lower. Researchers suspected that the non-native earthworms competed with native soil biota for organic matter, their food.

Forests in the northern Great Lakes region typify the second situation, where non-native earthworms are invading areas that were glaciated and without earthworms. Natural forest soils in many areas of the Great Lakes region contain a top layer of mostly undecomposed leaves or conifer needles, above a layer of partly decomposed organic material (collectively these layers are named an O horizon, for organic). This organic layer, which is typically a few to several inches thick, is on top of the A horizon. The A horizon contains mixtures of decomposed organic material and particles of sand, silt, and clay. Deeper soil layers occur below the A horizon. Different species of earthworms are active in these different layers of the soil and have different effects.[11] For example, litter-dwelling earthworms feed on the surface organic horizon, diminishing its thickness. Other earthworms are active in deeper soil layers, mixing organic material deep into the soil. The net effect of multi-species earthworm invasions is a reduction or elimination of the partly decomposed O horizon and mixing and thickening of the A horizon. This effect is not dissimilar to plowing the soil by humans.

While earthworm-free areas still exist in the Great Lakes region, a regional assessment in 2008-2010 of 125 sites in Minnesota, Wisconsin, and Michigan showed just how widespread earthworm invasion is.[9] Eighty-two percent of the sites contained some evidence of soil alteration by non-native earthworms (Fig. 6.2). In another inventory, Voyageurs National Park, Minnesota, contained non-native earthworms at 19 of 20 inventory sites.[13]

In addition to profoundly affecting soil, introduced earthworms influence plants and animals. Plant composition changes drastically after earthworm invasion.[11] In earthworm-free forests in the Great Lakes region with thick soil organic layers, plant species tolerant of shade and with large seeds are favored. Large seeds provide the energy reserves needed to allow a seedling to penetrate up through the organic layer. Once earthworms remove this organic layer, the advantage can shift toward species with small seeds or weak stems. A common plant species in earthworm-invaded forests is the weak-stemmed Pennsylvania sedge (*Carex pensylvanica*). Many other interactions between invading earthworms and the ecosystem are possible, such as the upper soil becoming more susceptible to freezing in the absence of the insulating organic layer. Additionally, fungal species in the soil – many of which help plant roots obtain nutrients – can change after earthworm invasion.

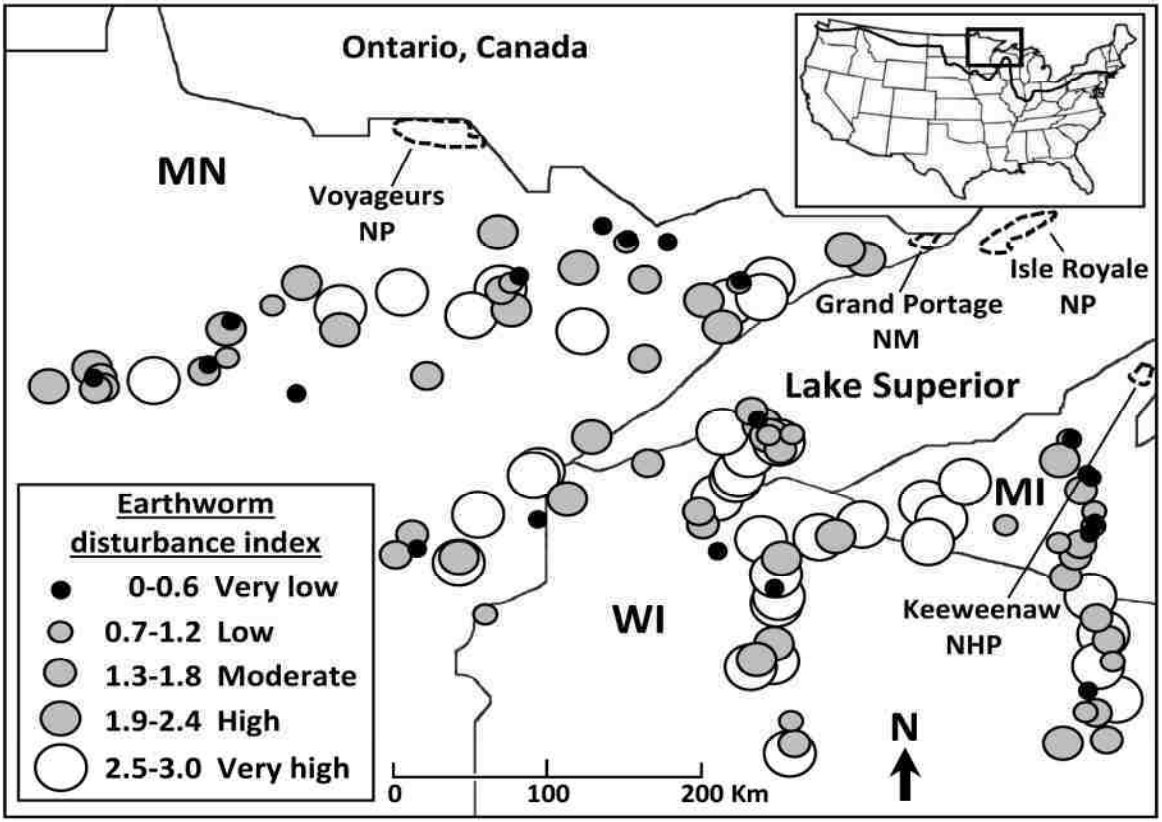


Fig. 6.2. Top right: the distribution of native earthworms, estimated by Hendrix and Bohlen (2002),[17] is generally south of the ice age limit of glaciation (black line). Bottom: inventory of 125 sites for soil disturbance by non-native earthworms near Great Lakes region parks (data from Fisichelli et al. 2013).[9]

Earthworm invasion could also alter the species composition of birds and wildlife. In northwestern Wisconsin near the Saint Croix National Scenic River, the ground-dwelling birds hermit thrushes (*Catharus guttatus*) and ovenbirds (*Seiurus aurocapilla*) were sparse in areas invaded by non-native *Lumbricus* earthworm species.[19] This may relate to reduced nest concealment for birds or lowered abundance of insects (food sources for the birds) from the removal of soil organic layers by earthworms.

Protecting parks from earthworm invasion is difficult, but several strategies can help delay the invasion or partly mitigate its effects. First, the non-profit Great Lakes Worm

Watch and other organizations are increasing public awareness about the transport and release of non-native earthworms to hopefully slow the invasion. Second, earthworms have long been chemically controlled on golf courses. Chemical control is difficult or undesirable in national parks, but exploring different types of treatment options may be valuable, especially for eradicating small earthworm infestations before they expand. Third, managing components of the ecosystem more easily managed – such as white-tailed deer (*Odocoileus virginianus*) and non-native plants – might help reduce effects of earthworms. For example, removing the non-native shrub Chinese privet (*Ligustrum sinense*) decreased abundance of non-native earthworms, while increasing native earthworms, in the Oconee National Forest, Georgia.[18] Removing Chinese privet apparently made soil pH unfavorable to the non-native earthworms. At the very least, it would be worthwhile for parks to identify priority areas to keep free of non-native earthworms, so some examples of earthworm-free soils remain.

Burmese Pythons in the Everglades

The Burmese python (*Python molurus bivittatus*) is a huge constrictor snake up to 6 meters (20 feet) long. In its native southeastern Asia habitat, the International Union for Conservation of Nature lists the species as "declining and vulnerable" partly due to overharvesting. In the United States, Burmese pythons were commonly purchased as pets. The U.S. Fish and Wildlife Service reported that 87,000 Burmese pythons were imported into the United States between 1999 and 2010. Before and during that time, unknown numbers of Burmese pythons escaped captivity or were abandoned as pets and released. First observed in the Everglades in 1979 and confirmed as breeding in 2006,[8] thousands of occurrences of the Burmese python have been documented in southern Florida (Fig. 6.3).

It is difficult to pinpoint ecological effects of the pythons in southern Florida parks, because many other factors have also changed concurrently with the python invasion. For example, the abundance of Virginia opossums (*Didelphis virginiana*), raccoons (*Procyon lotor*), red foxes (*Vulpes vulpes*), and other mammal species decreased between 1996-1997 and 2003-2011 surveys in Everglades National Park.[7] The differences in wildlife abundance between survey periods could relate to changes in procedures used to perform the surveys, shifts in climate, the python invasion itself, or other factors. With due heed of these considerations, the study identified an apparent correlation between python invasion and declines in native wildlife that warrants additional evaluation using more rigorous surveys.

Further evidence of the effect of pythons was gleaned by examining the digestive tracts of 85 Burmese pythons collected within Everglades National Park between 2003 and 2008.[8] Twenty-five species of birds were found inside the pythons. Four of the bird species are listed as of special concern by the Florida Fish and Wildlife Conservation Commission: little blue heron (*Egretta caerulea*), snowy egret (*Egretta thula*), white ibis (*Eudocimus albus*), and Limpkin (*Aramus guarauna*). Another bird species the pythons ate, the wood stork (*Mycteria americana*), is federally endangered. Burmese pythons also have eaten other conservation-priority species, such as American alligators and Key Largo woodrats (*Neotoma floridana*).

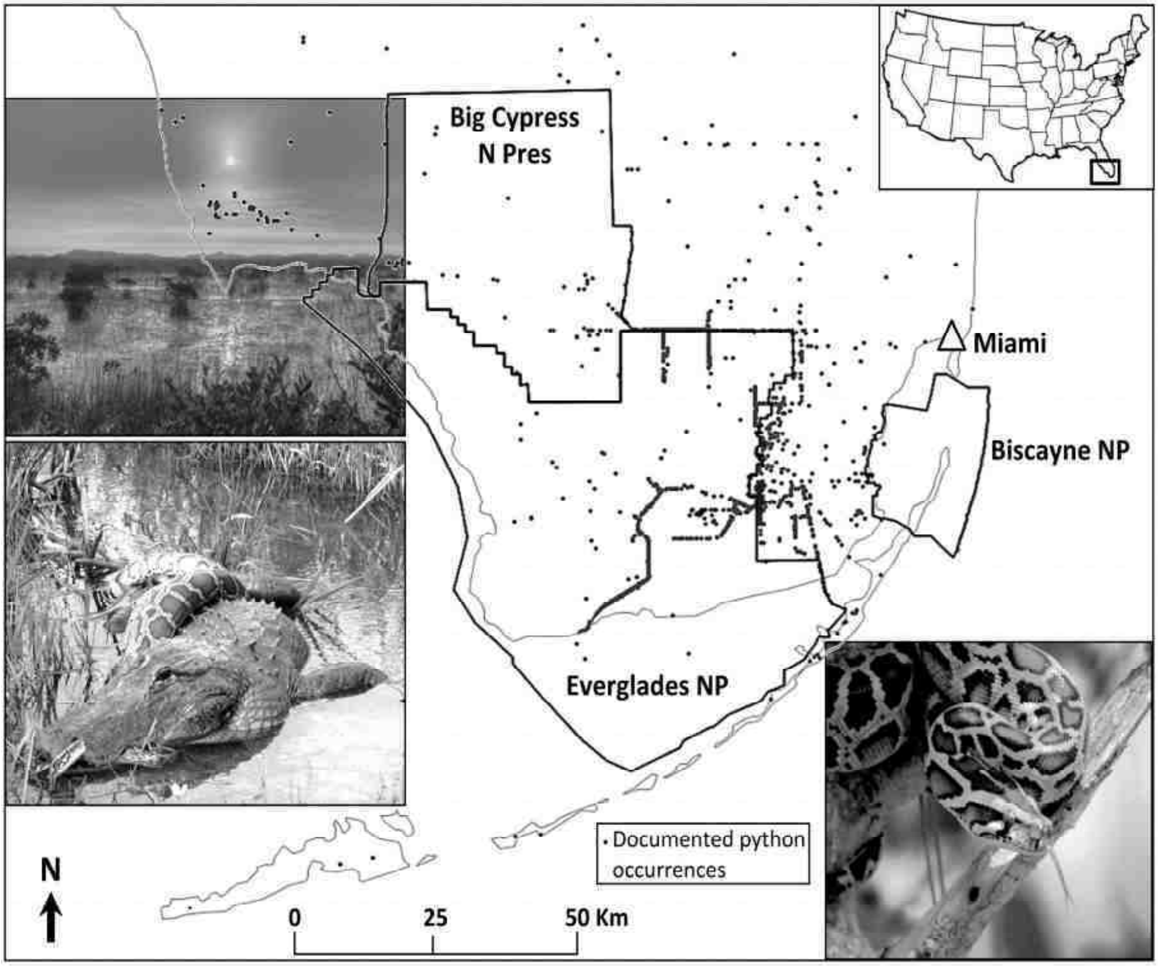


Fig. 6.3. Documented occurrences through 2014 of Burmese pythons in and around southern Florida parks reported in the Early Detection and Distribution Mapping System, Center for Invasive Species and Ecosystem Health, University of Georgia. Photos: a mangrove landscape of Everglades National Park (top left, by G. Gardner), Burmese python interacting with an American alligator (bottom left, National Park Service), and coiled python (right, by R. Cammauf).

Several management actions are being attempted to ameliorate the python invasion. Between 2002 and 2012, 1,972 Burmese pythons were removed from Everglades National Park, Big Cypress National Preserve, and nearby lands. In 2012, the U.S. Fish and Wildlife Service listed the Burmese python as an injurious wildlife species under the Lacey Act of 1900. This legal action prohibited, except by permit, the importation of Burmese pythons into the United States and limited interstate transport of the animals. While this does not mean that all Burmese python introductions will stop, attention in the national parks could focus on developing management strategies for Burmese pythons already inside.

The Unique Case of Feral Horses and Burros

Horses (*Equus ferus caballus*) and burros (*Equus asinus*, also called donkeys) represent a unique and controversial case of species generally considered non-native animals. Their history in North America has some uncertainty. Descendants of the genus *Equus*, containing modern horses, burros, and zebras, originated in North America around 4 million years ago. This is partly based on a 2013 analysis of DNA in a 700,000-year-old fossil horse bone recovered from permafrost at a gold mine near Dawson City, Yukon Territory, Canada.[24] *Equus* in North America was extinct by 11,000 years ago before the Wisconsin Glaciation ended.[2] It is unclear, however, whether the extinction related to climate change, hunting by Native Americans (or combined climate change and hunting), or other factors.

Regardless of the reason, horses and burros were absent from North America for at least 11,000 years until they were brought to North America from other continents by Spanish conquistadores in the 1500s.[2] By then, horses and burros had been domesticated in Eurasia and Africa for 5,000 years. Domesticated horses and burros continued to be used as work animals in North America for centuries, including in areas later becoming national parks. An example was the "Twenty Mule Teams" (Fig. 6.4). From 1883 to 1889, teams of 20 burros pulled 36-ton wagons hauling borax (used in soaps and other products) from the Harmony Borax Mine near Furnace Creek, within present-day Death Valley National Park, California. The arduous, 10-day route crossed 265 km (165 miles) of primitive roads to the railroad near Mojave, California, through the driest desert in North America. The hardiness of burros enabled their survival when they escaped or were abandoned after mining or other work operations ended. In the 1950s, five to 13 million feral burros inhabited the West.[22]

By 1960, ecologists had already expressed concern about damage created by feral horses and burros to natural ecosystems in national parks.[22] Contrastingly, horse and burro advocates were galvanized to write letters to political officials for conserving the feral animals. The result was the 1971 Wild and Free-Roaming Horse and Burro Act. The act required identifying herd management areas, based on the 1971 distribution of horses and burros, and sustainably managing populations in those areas.

Partly because the horses and burros were brought by humans from other continents and were not native to North America, the National Park Service was generally exempt from the 1971 act. However, horses and burros had already long inhabited national parks by 1971. And, despite some removals or relocations, the feral animals persisted in many national parks in the decades following the act. Removals and relocations were unpopular among some public constituencies, and combined with small management budgets, efforts to reduce populations of horses and burros were often limited.

Meanwhile, ecologists continued reporting effects of horses and burros in national parks in the 1970s and 1980s.[1] For example, in Death Valley National Park, burros preferentially ate native perennial grasses (Fig. 6.4). Inside fenced areas protected from burros, grasses were up to 10 times more abundant than outside with burros. Competition for forage and watering sites between burros and native bighorn sheep (*Ovis canadensis*) was also suspected.[2]

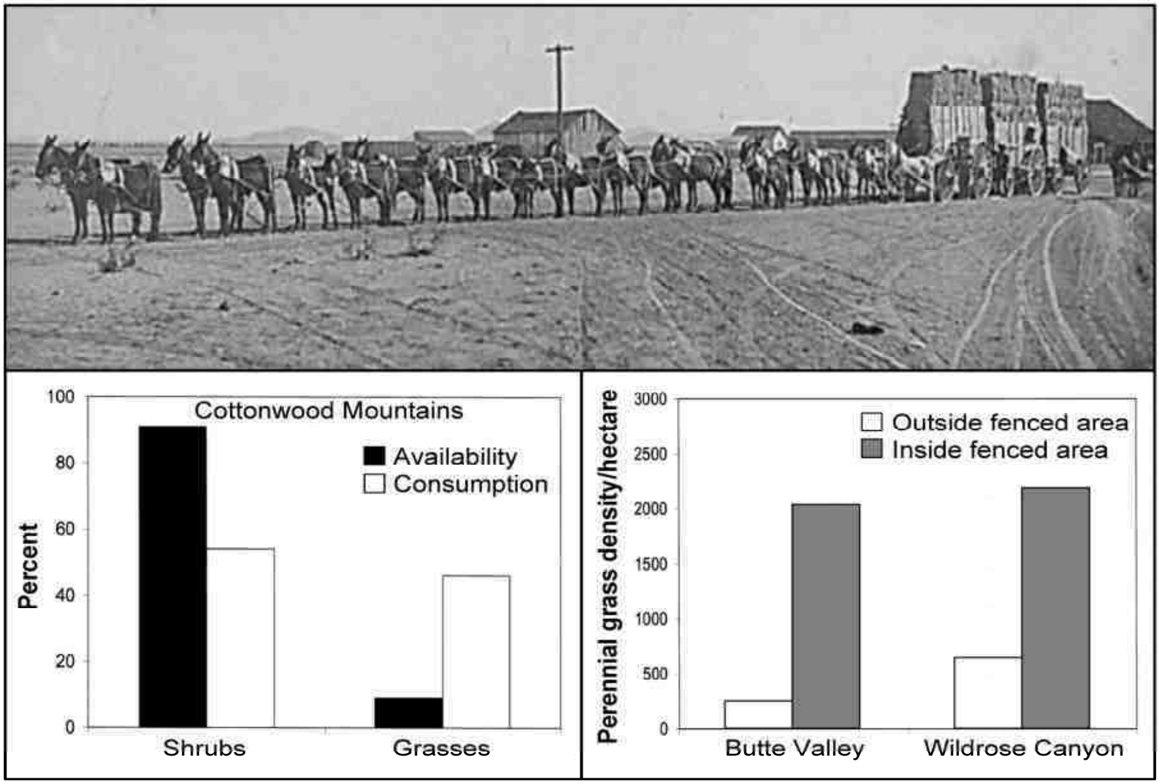


Fig. 6.4. Top: Twenty Mule Team in the Death Valley region, 1890 (photo courtesy of the National Park Service). Bottom left: feral burros disproportionally ate native perennial grasses in the Cottonwood Mountains, Death Valley National Park. Bottom right: perennial grasses were more abundant when protected from burros at two sites in Death Valley National Park. Graphs adapted from Abella (2008).[1]

Understanding how past activities of horses and burros have affected the development of park ecosystems remains important, because of possible trajectories for change initiated long ago by high populations of the feral animals. Moreover, horses and burros still inhabit some parks, via dispersal from surrounding lands where the feral animals are protected.

Presently, 179 herd management areas exist in 10 western states and cover 13 million hectares (32 million acres) of public land (Fig. 6.5). In 2014, the Bureau of Land Management reported that 41,000 horses and 8,000 burros roam on lands managed by the agency. Some of these animals freely roam onto national park lands.

Managing horses and burros on public lands continues to be challenging. Without wolves and other predators, horse and burro populations grow rapidly, exceeding designated population sizes within herd management areas. The Bureau of Land Management reported that the estimated carrying capacity of 27,000 horses and burros was exceeded by 22,000 animals in 2014. This was despite having removed 14,000 horses and burros from rangelands between 2012 and 2014. Many of these animals were offered to people for adoption, but as in previous years, many animals were held in captivity. The 47,000 horses in captivity in 2014 actually exceeded the estimated 41,000 ranging freely on public lands. The wild horse and burro program cost $77 million in 2014, with 64% of that used to maintain captive animals at a rate of $1,000/animal/year. Projected expenses for the program between 2015 and 2030 exceed $1.2 billion, with $0.7 billion for maintaining captive animals. On public lands, non-native plant invasion, wildfires, climate change, and limited management budgets are further increasing tension between balancing needs of native wildlife with sustaining herds of feral horses and burros.

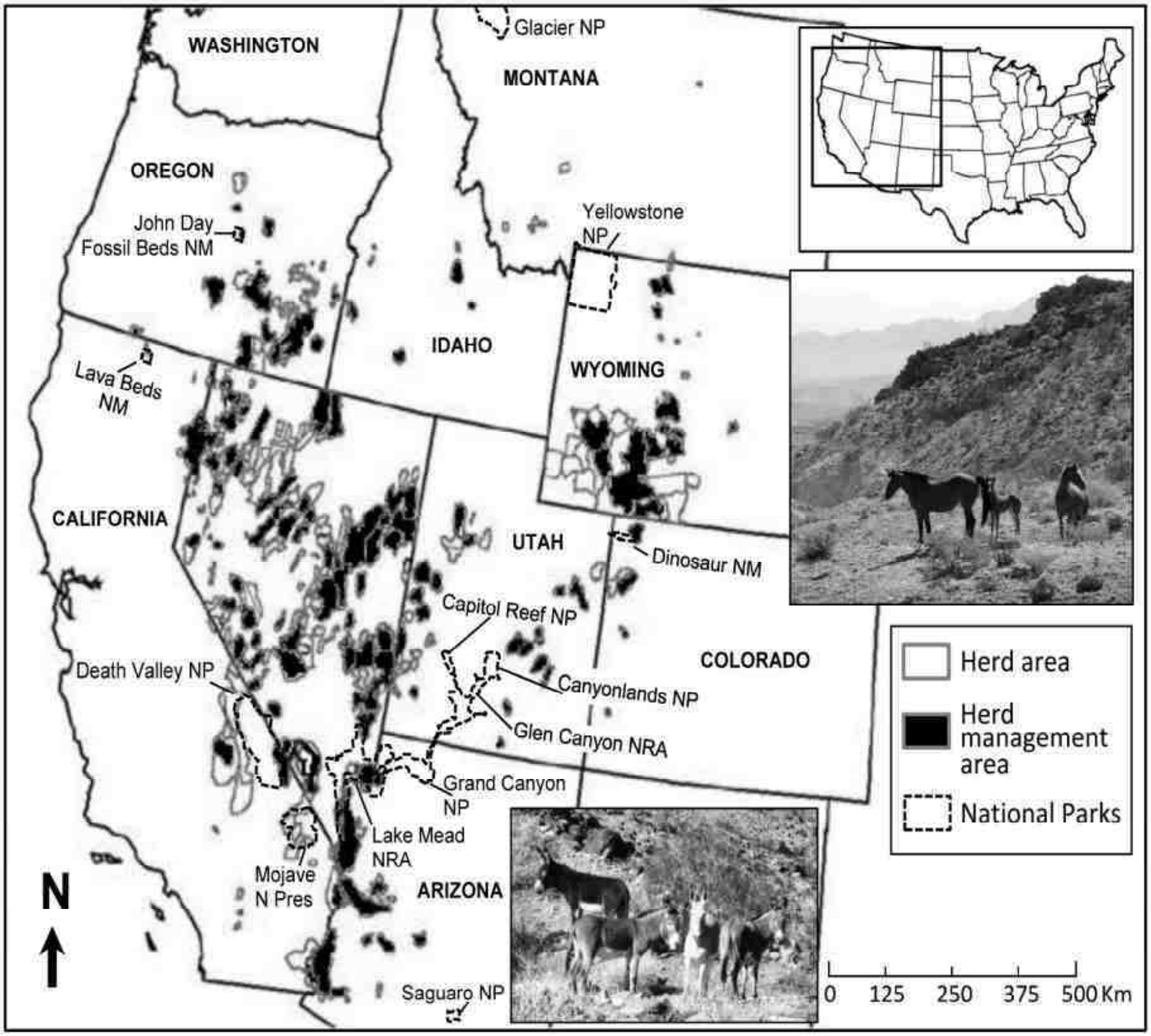


Fig. 6.5. Distribution and management areas of feral horses and burros in 2011 according to the Bureau of Land Management. Selected national parks are shown. Photos of feral horses (top, by S.R. Abella) and burros (bottom, by R.J. Abella) in Lake Mead National Recreation Area, Nevada.

Pork in the Parks

Animals collectively known as feral pigs (*Sus scrofa*) encompass feral domesticated pigs (also known as hogs or swine), European wild boars, and hybrids between domesticated pigs and boars (Fig. 6.6). The original, native range of pigs is believed to have included North Africa, Europe, and Asia.[3] However, pigs have been domesticated and moved by humans for thousands of years.[31] The Polynesians brought pigs to Hawaii over 1,000 years ago, and Europeans transported pigs to mainland North America by the 1500s.[29]

In the last three decades, pigs have continued expanding their distribution in the United States. Between 1982 and 2012, the range of feral pigs grew from 17 to 38 states (Fig. 6.7). This increase resulted from dispersal and likely the escape or deliberate transport of the animals by humans, such as for establishing pig populations for sport hunting.[3] As many as 3 million feral pigs inhabit Texas alone, with a density of 1 pig/km^2 (2 pigs/square mile) in suitable habitat. A bewildering array of different-colored pigs inhabit national parks from the Southeast to California as well as island parks such as in Hawaii.

Some of the most thorough research on how feral pigs affect park ecosystems was performed during the 1970s in Great Smoky Mountains National Park (North Carolina/Tennessee), which feral pigs have inhabited since the 1940s. Pigs eat many items in the park, such as acorns, plant roots, flowers, insects, salamanders, snails, bird eggs, small mammals, and fish.[25] Pigs also disturb the soil via trampling, wallowing, and rooting. Rooting entails churning the soil to locate roots and soil organisms to eat. At high elevations around Clingman's Dome and the Appalachian Trail, plant cover was seven times lower in pig rooting areas compared to undisturbed areas.[5] Soil disturbance by pigs also altered plant composition by favoring deep-rooted or poisonous plants.

A more recent analysis in the 1990s in Channel Islands National Park, off the California coast, suggested possibilities for how feral pigs influence native wildlife.[28] The largest native carnivore on the Channel Islands was the island fox (*Urocyon littoralis*), which predated upon a smaller carnivore, the island spotted skunk (*Spilogale gracilis amphiala*). Researchers hypothesized that availability of feral piglets as easy prey altered the natural food web, by allowing predatory golden eagles (*Aquila chrysaetos*) to colonize the islands.

Fig. 6.6. Left: feral pig with piglets (photo provided by Buffalo National River, Arkansas). Right: feral pigs in Cumberland Island National Seashore, Georgia (National Park Service photo).

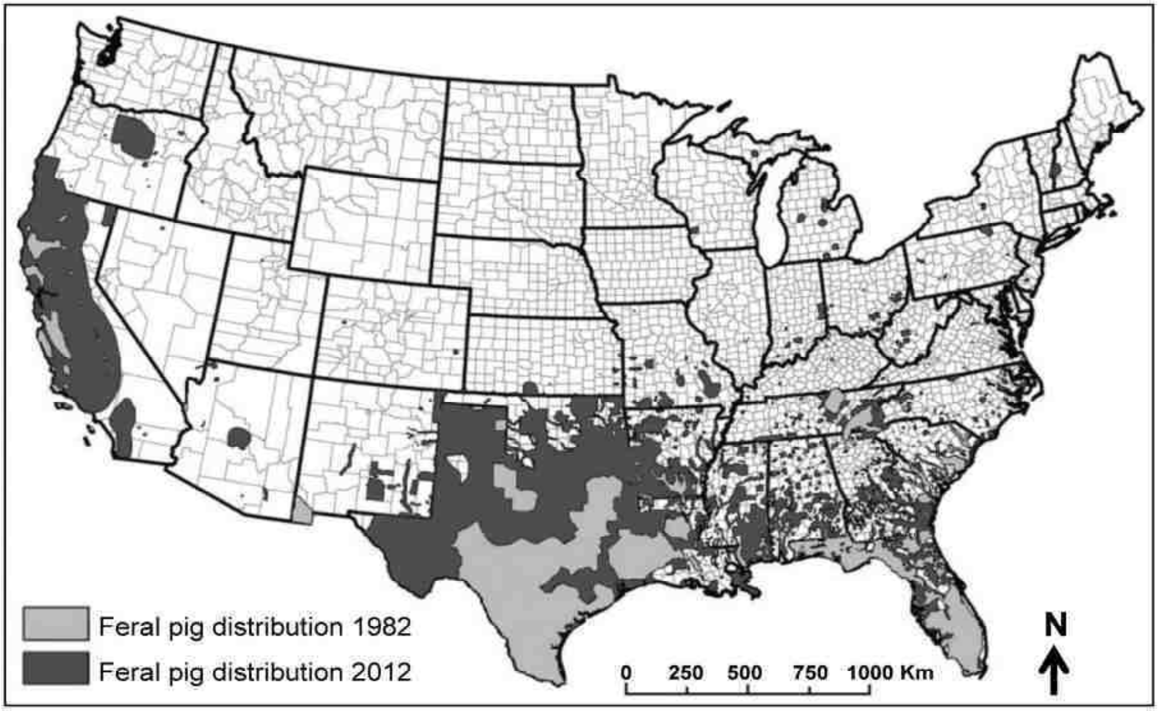


Fig. 6.7. Expanding distribution of feral pigs. Map adapted from Bevins et al. (2014).[3]

Concurrently with colonization by eagles and their predation on piglets as well as island foxes, the fox population crashed. Meanwhile, spotted skunks increased, presumably because fewer foxes existed to eat them. Researchers carefully noted that these were correlations and establishing cause-effect for the changes is difficult. It would not seem surprising, however, that introducing feral pigs to an ecosystem would alter the food web.

Managing feral pigs in parks is challenging, but a recent example in Pinnacles National Park illustrated that focused effort can be effective.[20] Established in 1908 to protect unique rock formations and near the 101 Pacific Coast Highway south of San Jose, California, the park contains chaparral and oak woodlands. With effects of feral pigs intensifying, the park decided to construct a fence to enclose 5,700 hectares (14,000 acres), about half the park. The fence cost $2 million and was completed in 2003. Between 2003 and 2006, all 200 pigs inside the fenced area were removed using hunting, capture via trapping, tracking dogs, and Judas animals (tracking one animal to lead to other animals). Animal welfare protocols were followed during the pig removals. The eradication effort, including planning, fieldwork, caring for tracking dogs, purchasing traps, administration, monitoring (such as checking the condition of fences), and other tasks, required 13,000 hours and $623,000. Each individual pig required an average of 68 hours of time and $3,100 to remove. Maintaining the fence and monitoring ecological condition, such as whether native species recover in the pig-free environment, would be required into the future.

The Unusual and Uncertain in Species Introductions

Some species introductions seem bizarre. In 1890-1891, the American Acclimatization Society released European starlings (*Sturnus vulgaris*) in New York's Central Park.[10] Some authors have claimed this release related to the society's goal of introducing every bird species mentioned in the works of William Shakespeare.[4] The original 120 birds released began a population now exceeding 100 million European starlings across North America.

In 2014, Great Smoky Mountains National Park expressed concern over invasion of park streams by the single-celled didymo (*Didymosphenia geminata*). This strange algae species, also called rock snot, forms smothering mats covering stream beds (Fig. 6.8). The transport of flora and fauna between and within continents essentially is an uncontrolled global experiment, and it is not known whether ecological outcomes will seem as bizarre as some of the species introductions.

Fig. 6.8. Rock snot. Photo by S. Spaulding, U.S. Geological Survey.

Pinpointing non-native species as causing extinctions of native species often is difficult, because so many other changes (such as in climate and disturbance) have occurred simultaneously with species introductions.[12] Native species extinctions are also not a requirement for ecological transformations to have occurred. Even the American chestnut, discussed in Chapter 2, was not driven to extinction by the non-native blight. Eastern forests were, however, transformed by elimination of chestnut as an overstory tree. The troubling possibility also exists that instead of obvious effects (Fig. 6.9), non-native species invasions can manifest subtly, such as altering genetic structure of native species or removing populations important to a native species' evolution.

Fig. 6.9. Effect of exclusion fencing and a disappointed non-native goat, Haleakala National Park, Hawaii. Photo by D. Reeser, National Park Service.

Demonstrating that non-native species affect parks is not required to justify managing non-native species in parks. This is because national parks are mandated to contain native species, per National Park Service policy, with few exceptions such as conserving landscaping vegetation for cultural reasons.[23] Given the plethora of non-native species already in parks and the fact that not all are likely to greatly alter parks, prioritizing limited management budgets to species with the greatest potential impacts, or to vital areas within parks, will remain important. This approach can be successful, as the feral pig eradication program at Pinnacles National Park showed.

7 FORESTS

William Bartram, born in 1739 in Pennsylvania, was among America's first naturalists. He made botanical and natural history observations during extensive travels in the 1770s in the Southeast. His resulting 1791 book, titled *Travels*, is a seminal account of American natural and cultural history. *Travels* describes an old-growth forest Bartram encountered in 1773 in east-central Georgia: "Leaving the pleasant town of Wrightsborough, we continued 8 or 9 miles through a fertile plain and high forest, to the north branch of Little River…crossing which, we entered an extensive fertile plain, bordering on the river, and shaded by trees of vast growth, which at once spoke its fertility. Continuing some time through these shady groves, the scene opens, and discloses to view the most magnificent forest I had ever seen. We rose gradually a sloping bank of 20 or 30 feet elevation, and immediately entered this sublime forest. The ground is perfectly a level green plain, thinly planted by nature with the most stately forest trees, such as the gigantic black oak, [tulip poplar]…and [sweetgum], whose mighty trunks, seemingly of an equal height, appeared like superb columns…In describing the magnitude and grandeur of these trees…I think I can assert, that many of the black oaks measured 8, 9, 10, and 11 feet diameter 5 feet above the ground, as we measured several that were above 30 feet [girth], and from hence they ascend perfectly straight, with a gradual taper, 40 or 50 feet to the limbs…"[4]

Few old forests such as those Bartram described remain in the United States, due to centuries of clearing and increasing efficiency of logging. The first sawmill in North America may have been built in 1623, near York, Maine.[9] By 1682, Maine had 24 sawmills. Despite some early attempts at regulating logging, cutting proceeded rapidly in eastern North America. The prized tree was eastern white pine (*Pinus strobus*), used for ship masts and lumber. By 1750, an individual sawmill cut 4,000 feet of white pine boards daily. Boards were one inch thick, 15 to 25 feet long, and a foot wide. In 1792, Massachusetts exported 28 million feet of pine boards, 350,000 feet of oak boards, and 210 white pine trees for ship masts. In 1832, the first railroad in Maine (and among the first in the United States) was built to haul timber from mills to the city of Bangor, a port near the Maine coast. By 1890, Maine had 894 sawmills and timber processing sites, employing 11,500 people. The cities of Boston and New York were largely built with lumber from Maine.

Wood has underpinned America's economy and infrastructure for four centuries and continues to do so. In 2012, the U.S. Energy Information Administration reported that the United States had 5.6 million commercial buildings, most containing wood. According to the U.S. Census Bureau, 133 million housing units (houses, mobile homes, and apartments) existed in the United States in 2013. The United States produced between 12 and 19 billion cubic feet of wood products annually from its forests between 1965 and 2011.[17] The value of wood exported by the United States was $19-45 billion/year between 2000 and 2011 (2 to 4% of all U.S. exports). Similarly, the value of wood imported to the United States from other countries (90% from Canada) was $19-44 billion/year (1 to 3% of total U.S. imports).

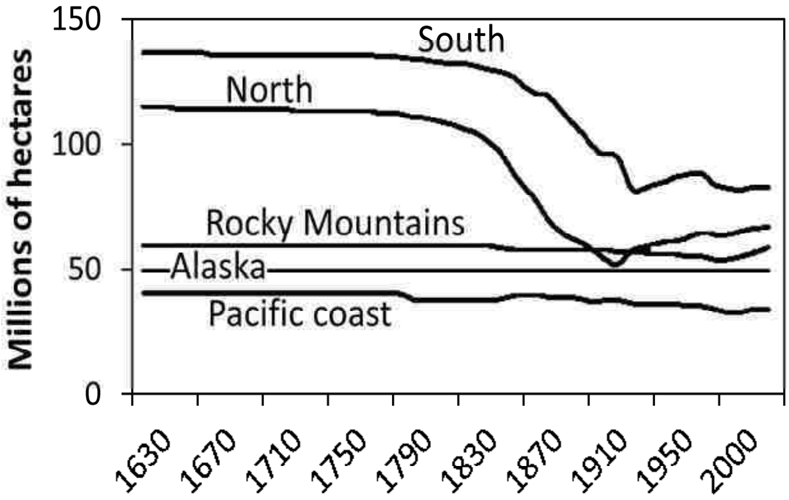


Fig. 7.1. Forest area in the United States from 1630 to 2007. Data from U.S. Forest Service (2010).[27]

Given the long history of forest clearing, it may be surprising that the total area of forest in the United States has been relatively constant during the last century (Fig. 7.1). In 1630, estimated forest area was 400 million hectares (1 billion acres). Forest area began declining during the 1600s and 1700s. It then plummeted during the 1800s from intensive clearing for agriculture and logging. Since the early 1900s, however, forest area has been constant or even increased, depending on the region. Stabilization of forest area partly resulted from the designation of 77 million hectares (190 million acres, or 8% of the U.S. land area) of national forests managed by the U.S. Forest Service, creation of national parks, and reforestation of logged or abandoned farm lands. The forest area in the United States in 2007 was 304 million hectares (751 million acres). Stability of forest area could be viewed favorably for forest conservation. However, forests are increasingly fragmented into smaller parcels, and other stressors like non-native insects threaten forests.

National parks interface with past and present forest use and management in many ways. Some parks were specifically designated to maintain living examples of managed forests, showcasing America's cultural history of forestry and conservation. In most parks, the long history of forest use, before parks were established, results in forests being relatively young (less than 150 years old) and never-harvested, old forests rare. Yet, in appropriate forest types (excluding those that are disturbed naturally before becoming old), national parks can facilitate development of old forests free from producing timber. Parks also interact with the global wood and horticulture industries, which have inadvertently introduced non-native pests that threaten park forests.

This chapter covers three topics related to the health of park forests: forest management, conservation of old forests, and introduced pests. Definitions of what constitutes a "healthy" forest may vary depending on goals of who is managing a forest, such as a private timber company compared with agencies that do not harvest timber. Generally, healthy forests have sizes of trees appropriate to the forest type, lack unnatural mortality or slowed growth of trees, and provide desired ecosystem functions. These functions could include conserving biodiversity, protecting streams and lakes, storing carbon, growing large trees for wildlife habitat, or providing for human recreation.

The Oldest Managed Forest: Marsh-Billings-Rockefeller National Historical Park

With the 225-hectare (555-acre) Marsh-Billings-Rockefeller National Historical Park in Vermont, the National Park Service is the current steward of the oldest continuously managed forest in the United States.[24] The property is a living example of the evolution of forestry and forest conservation. George Perkins Marsh, an early conservationist, was born in 1801 on the property, which remained his family farm until 1869. At that time, Frederick Billings, lawyer and railroad executive, purchased the property. Interested in conserving forests, Billings' vision was to create a model example of reforestation and sustainable forestry. This was a pioneering effort unique for the time, a significant shift in thinking from the intensive forest exploitation then occurring across the United States.

Fig. 7.2. Marsh-Billings-Rockefeller National Historical Park, Vermont (National Park Service photo).

More well-developed forestry in Europe influenced Billings. His early reforestation efforts included planting European larch (*Larix decidua*) and Norway spruce (*Picea abies*) into evenly spaced plantations. Later, in other areas of the property, Billings planted native pines and deciduous trees. Billings also created 19 km (12 miles) of carriage roads through the property, encouraging the public to view his forestry experiment. These carriage roads remain in use for recreation (Fig. 7.2).

Frederick Billings' family continued the forestry program after 1890. Mary French Rockefeller, Billings' granddaughter, and her husband, Laurance Rockefeller, oversaw the property from the 1950s to the 1990s, before donating it to the American people. Marsh-Billings-Rockefeller National Historical Park opened in 1998. The park is a mosaic of plantations of different species and ages, remnant native conifer and deciduous trees, streams, ponds, fields, and recently cut or re-planted forests (Fig. 7.3).

Fig. 7.3. Left: examples of vegetation types, including an old field surrounded by deciduous forest, and a red pine plantation established in 1926 shown in 2003. Right: demonstration of historical use of horses for transporting logs, and a portable saw mill cutting lumber (National Park Service photos).

The National Park Service completed a forest management plan in 2006, building upon 130 years of forestry on the property.[24] The plan balances conserving historical features (such as some plantations), maintaining forestry practices including thinning the forest and re-planting, and promoting natural, deciduous forests. For management purposes, the park delineated 50 forest stands, averaging 4 hectares (10 acres). Each stand is an independent unit receiving customized forestry practices, but managed within the park's overall goals.

Meeting a definition of sustainable forestry, current harvesting does not exceed the forest's capacity for re-growth. A 2006 timber inventory indicated that the park contained 7 million board feet of timber and 4,500 cords of pulpwood. A board foot of lumber is a piece 1 foot wide, 1 foot tall, and 1 inch thick. A cord is 128 cubic feet of wood, or a woodpile 8 feet long, 4 feet high, and 4 feet deep. Between 2007 and 2014, the National Park Service harvested 483 cords of wood per year from 21 different stands. This harvesting represents a balance among forest re-growth each year, implementing the forestry practices the park was designated to perpetuate, and other objectives such as using forest thinning to encourage development of more natural forests to replace plantations.

The park has received sustainable forestry certification by the Forest Stewardship Council, and sells wood on the open market and to mills certified by the Council. Wood produced from the park's forests includes high-quality logs of softwoods (conifer trees) and hardwoods (deciduous trees), and lower-quality material, such as pulpwood for making paper. As of 2014, 28 companies have purchased wood from the park's forestry program. Uses of the wood ranged from lumber for building, fences, plywood, firewood, pulpwood, and pellets for burning to produce energy. Specialty products have included larch decking, maple tables, canoes, and bird houses. The park's visitor center contains furniture built from the park's forests.

Old-Growth Forests

Never-harvested, old-growth forests are now uncommon. Also, too little time has passed for old forests to re-develop following historical logging before parks were created. Examples do exist, however, of remnant old-growth forests and individual old trees in some parks. America's national parks contain among the world's oldest trees (Table 7.1).

Table 7.1. Some old trees in national parks of the United States.[5,7,8,12,13,14,20]

Species	Age (years)	Park
White oak	344	Great Smoky Mountains National Park
Tulip poplar	385	Great Smoky Mountains National Park
Eastern hemlock	451	Great Smoky Mountains National Park
Shortleaf pine	232	Ozark National Scenic Riverways
Plains cottonwood	370	Theodore Roosevelt National Park
Pinyon pine	420	Mesa Verde National Park
Ponderosa pine	674	Jewel Cave National Monument
Douglas-fir	1,274	El Malpais National Monument
Rocky Mountain juniper	1,500+	El Malpais National Monument
Alaska-cedar	1,200	Mount Rainier National Park
Giant sequoia	2,000+	Sequoia National Park
Bristlecone pine	4,500+	Great Basin National Park

Congaree National Park, southeast of Columbia in central South Carolina along the Congaree River, has the largest old-growth bottomland forest remaining in the southeastern United States. The 11,000-hectare (27,000-acre) park supports diverse, low-lying floodplain forests of baldcypress (*Taxodium distichum*), water tupelo (*Nyssa aquatica*), sweetgum (*Liquidambar styraciflua*), water hickory (*Carya aquatica*), swamp chestnut oak (*Quercus michauxii*), persimmon (*Diospyros virginiana*), and many other tree species (Fig. 7.4).

Fig. 7.4. Bottomland forest of baldcypress and water tupelo along Cedar Creek, Congaree National Park, South Carolina. Photo by V.B. Shelburne, South Carolina Big Tree Program Coordinator.

During the 1990s, researchers re-inventoried a 1970s assessment of the park's large trees.[19] Researchers walked 450 km (275 miles), recording the location, height, and circumference of large trees. Large trees in both inventories were submitted to the American Forests Big Tree Program, which designates trees based on their size as state and national champions. Of the 30 state champion trees in 1979 (including nine national champions), only two were recorded as alive in 1995. Many apparently died before Hurricane Hugo struck the area in 1989, but some may not have been found or were damaged by the hurricane. However, new state and national champions were found during the 1990s inventory. Some of the enormous trees included: a water hickory 5 meters (17 feet) in trunk circumference and 45 meters (148 feet) tall; a water tupelo 6 meters (20 feet) in circumference and 35 meters (114 feet) tall; a swamp chestnut oak 6 meters (20 feet) in circumference and 37 meters (120 feet) tall; and a baldcypress 8 meters (26 feet) in circumference and 40 meters (131 feet) tall.

The death of previous trees, but the growth of new champion trees, illustrates the dynamic nature of forests and that human influences are not the only ones acting on old-growth forests. Natural disturbances, such as hurricanes and severe fires, often prevent forests from becoming old. A long period of relatively stable conditions, free from influences that remove the forest, is required for old-growth forests to develop.

In another example of old forests, El Malpais National Monument occupies the Zuni-Bandera volcanic field, near Grants, New Mexico. The El Malpais eruptions occurred relatively recently geologically, beginning 60,000 years ago, with the youngest lava flow 3,900 years old. The stark, volcanic landscape contains some of the oldest trees – and ironically smallest – in the Southwest. Pygmy forests of bonsai trees of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) inhabit the lava flows. Many of the trees are ancient. One Douglas-fir was 1,274 years old.[14] In other areas of the Southwest, ponderosa pine and Douglas-fir are normally large and long-lived, but do not attain the old ages like in

El Malpais. It is not fully understood how El Malpais's lava soils can support such old trees. Reasons could include lack of competition between trees due to their small size and open spacing, slow growth enabling persistence to antiquity, or features of the lava soils (Fig. 7.5).

Fig. 7.5. Dwarf conifer trees in El Malpais National Monument, New Mexico (National Park Service photo).

Exemplifying the scientific value that old forests provide, researchers developed a tree-ring chronology that was 2,129 years long (136 BC to 1992 AD) from living and dead trees in the monument.[14] The width of tree rings was correlated with precipitation, enabling long-term reconstruction of past climate. The tree rings revealed several prolonged droughts of centuries, and the 400-year Little Ice Age. This is a valuable baseline for evaluating contemporary climate change.

Fig. 7.6. Ancient bristlecone pines, Great Basin National Park, Nevada (photo by R.J. Abella).

Great Basin National Park takes tree longevity to the next level. Some of Earth's oldest trees inhabit high-elevation forested mountain ranges of the Great Basin Desert, including the park in east-central Nevada (Fig. 7.6).[1] The 3,983 meter (13,063 foot) tall Wheeler Peak, in the park's Snake Range, contains a small glacier and open forest of bristlecone pine (*Pinus longaeva*) below treeline. One tree, measured in 1964, was only 5 meters (17 feet) tall, with the living part of the canopy only 3 meters (10 feet) tall. Bark was missing from 92% of the trunk. This unassuming tree was 4,900 years old.[8] The environmental changes this tree experienced are amazing, encompassing nearly half the time period since the last major ice age ending 11,000 years ago. The United States as a country is 239 years old. The tree persisted through the equivalent of 20 complete histories of the United States.

Much of the National Park Service's extensive old forest in the lower 48 states is in the Sierra Nevada Mountains and Pacific Northwest. A 1992 inventory indicated that 496,000 hectares (1.2 million acres) of old forest inhabited national parks in California, Oregon, and Washington (Table 7.2). Most of this was in Olympic and North Cascades in Washington, and Yosemite, Kings Canyon, and Sequoia National Parks in California.

Old and young forests differ ecologically. An example is the diverse habitat provided by complexity of multi-layered tree canopies and large, fallen trees on the forest floor (Fig. 7.7).

Table 7.2. Area of old forest more than 200 years old in national parks of California, Oregon, and Washington. Data from Bolsinger and Waddell (1993).[6]

Park	Acres of old forest	% of park
California		
Golden Gate National Recreation Area	8,100	24
Lassen Volcanic National Park	27,130	25
Lava Beds National Monument	570	1
Muir Woods National Monument	240	46
Redwood National Park	15,790	20
Sequoia/Kings Canyon National Parks	202,430	23
Whiskeytown National Recreation Area	1,220	3
Yosemite National Park	225,510	30
Oregon		
Crater Lake National Park	50,000	27
Oregon Caves National Monument	480	99
Washington		
Mount Rainier National Park	91,000	39
North Cascades National Park	236,000	47
Olympic National Park	366,000	40

Fig. 7.7. Old growth in Hoh Rainforest, Olympic National Park, Washington (photo by S.R. Abella).

Fallen logs are a major part of the old-growth Hoh Rainforest, near the Pacific Ocean in Olympic National Park, Washington.[16] What happens when a tree falls in this forest? It takes 60 to 190 years for all the bark to decompose and drop off, depending on the species of tree. Bryophytes (mosses and liverworts) cover fallen logs within 20 years and continue accumulating for 150 years. Bryophytes help retain tree seeds on the logs, and initiate soil formation to provide a rooting medium. Development of partially decomposed organic soil (humus) is slow the first 10 years a log is on the forest floor. Then, humus rapidly accumulates for the next 180 years. Seeds of the trees Sitka spruce (*Picea sitchensis*), Douglas-fir, and western hemlock (*Tsuga heterophylla*) readily germinate on the decomposing logs (termed "nurse logs"). Ironically, though, the logs are not optimal locations for seedlings to survive, because continued breaking up of the bark disrupts the seedlings. Although many seedlings die, not many need to survive to perpetuate the forest.

Accelerating Old-Growth Features in Redwood National Park

There is no substitute for time in developing old forests, but active forest management can accelerate the formation of old-growth features in previously cut young forests.

Through a long effort of gaining public support for cutting trees in a national park, Redwood National Park in northern California has used restoration tree cutting to accelerate development of natural forests. For example, the park began a 700-hectare (1,700-acre) forest thinning project in 2009.[26] The project area had been clearcut (all trees removed) from 1954 to 1962, before designation of the park in 1978. In 2009, the young forest was dominated by trees other than redwood (*Sequoia sempervirens*), and their growth stagnated in the dense forest. After the National Park Service selectively thinned out half the trees, redwood was dominant in the forest and began increasing its growth. This thinning, and additional thinnings that may be required, are anticipated to accelerate development of large trees and layering of forest canopies, typifying older forests (Fig. 7.8).

Fig. 7.8. Top: old redwood forest, Lady Bird Johnson Grove, Redwood National Park, California. Bottom: thinned forest to accelerate development of large redwoods (photos by S.R. Abella).

Forest Pests

Outbreaks of native insects and diseases (many caused by fungi) that damage or kill trees have long been natural processes in forests. In fact, openings created by the death of trees are important for promoting understory plants on the forest floor and the establishment of new tree seedlings. While these forest mortality events may be nuisances on industrial forestlands, they are natural processes to maintain in national parks. Unfortunately, non-native pests, fire exclusion, and likely climate change are interacting to threaten park forests and indeed the very existence of forests in some areas.

Table 7.3. Examples of non-native forest insects and diseases in 121 eastern national parks. The table shows the number of parks (out of 121 parks) containing the host trees, and the number of parks (and percentage) within the zone of infestation of the non-native pest. Data from Fisichelli et al. (2014).[11]

Non-native tree pest	Major host trees	Parks with host trees	Parks within infested zone
Asian longhorned beetle	Maples, buckeyes, elms	121	4 (3%)
Balsam woolly adelgid	Balsam fir, Fraser fir	16	9 (56%)
Beech bark disease	American beech	98	35 (36%)
Dogwood anthracnose	Flowering dogwood	101	64 (63%)
Dutch elm disease	American elm	108	108 (100%)
Emerald ash borer	Ash species	120	30 (25%)
Gypsy moth	Oaks	121	60 (50%)
Hemlock woolly adelgid	Eastern hemlock	59	46 (78%)
White pine blister rust	Eastern white pine	79	63 (80%)
Winter moth	Black cherry, oaks, maples	121	8 (7%)

Non-native insects and diseases are threats more pressing than climate change to forests in many areas.[22] The number of non-native forest insects already introduced to the United States exceeds 450 species.[3] Not even considering projections of at least one new introduced pest likely to become established per year in coming decades,[21] a frightening array of non-native pests are already impacting U.S. forests, including in national parks (Table 7.3). Some examples of established pests are the insects emerald ash borer (*Agrilus planipennis*) and Asian longhorned beetle (*Anoplophora glabripennis*), and the fungus creating the disease white pine blister rust (*Cronartium ribicola*). White pine blister rust infects eastern white pine in eastern parks and whitebark pine (*Pinus albicaulis*) and other western pines in western parks. The next sections provide just two of the numerous examples of non-native forest pests and their effects and potential management.

Dogwood Anthracnose in Catoctin Mountain Park

The flowering dogwood (*Cornus florida*) is the state tree of Virginia and Missouri and the state flower of Virginia and North Carolina. Dogwoods have showy flowers in spring, and while their bright red berries are unsuitable for human consumption, many wildlife species eat them. A small, deciduous tree, dogwood typically grows below the canopies of taller trees in forests of the eastern United States (Fig. 7.9).

Of unknown origin and first recognized in North America in 1978, the fungus *Discula destructiva* creates the disease dogwood anthracnose.[25] The disease is decimating flowering dogwoods. Symptoms of infected trees are lesions and blotches on the leaves, twig dieback,

Fig. 7.9. Flowering dogwood, Buffalo National River, Arkansas (photo by T. Fondriest, National Park Service).

cankers on the trunk, and sprouting from the trunk. Cankers can girdle and kill the trees. Trees on cool, moist sites are the most susceptible, as are trees already weakened by other factors.

Flowering dogwoods have been hard hit in the 2,400-hectare (5,900-acre) Catoctin Mountain Park. In north-central Maryland, an hour drive from Baltimore and Washington, D.C., Catoctin contains Camp David, the U.S. presidential retreat. Camp David is within the park's boundaries but is a military installation not open to the public. The National Park Service manages the surrounding park land.

In 1984 during the early stages of dogwood anthracnose infection in the park, flowering dogwood was a major understory tree. There were 682 dogwood trees/hectare (276/acre). Only 8% of the trees remained alive 10 years later in 1994.[25] A 2013 inventory found even fewer live dogwoods and a 98% decline since 1984 (Fig. 7.10). Remaining trees had some symptoms of infection but were generally healthy. This might relate to some resistance to the disease, trees inhabiting non-susceptible sites, favorable climatic conditions for tree survival, or that the disease was not spreading because so few dogwoods were still alive.

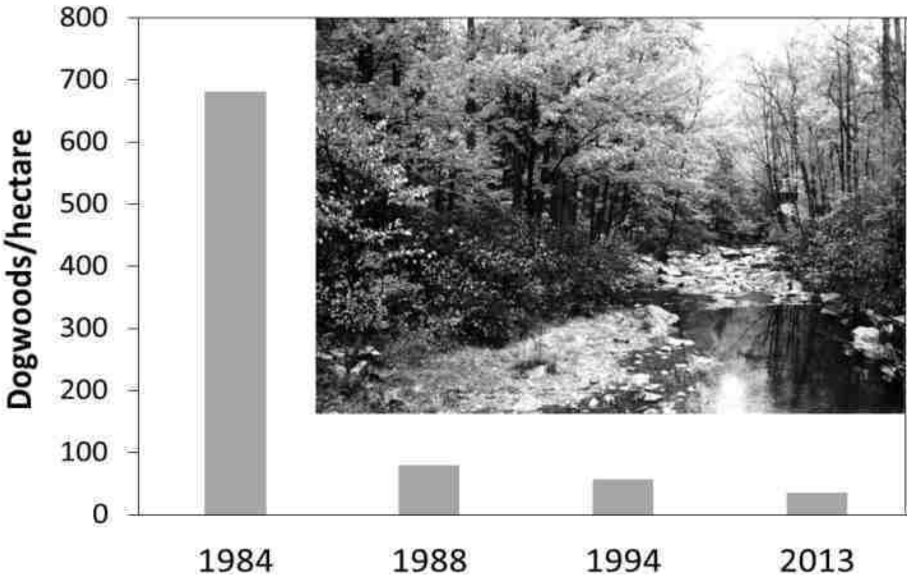


Fig. 7.10. Declining density of flowering dogwood trees in Catoctin Mountain Park, Maryland. Data from Sherald et al. (1996)[25] and D.K. Martin and W.J. Jones (U.S. Forest Service). National Park Service photo shows a forest along Big Hunting Creek.

Long-term ecological effects of dogwood's downturn are poorly understood. Changes to wildlife habitat and soil nutrient cycling might occur. Dogwood is a key part of calcium cycling in forest soils, because dogwood leaves are high in calcium and decompose rapidly. Although fire is not necessarily common at some of the moist sites flowering dogwood inhabits, Great Smoky Mountains National Park has found that burned sites exhibit reduced virulence of dogwood anthracnose.[18] The open, aerated conditions after fire may not be optimal for development of the fungus. Fire is worth exploring as a management option.

Hemlock Woolly Adelgid

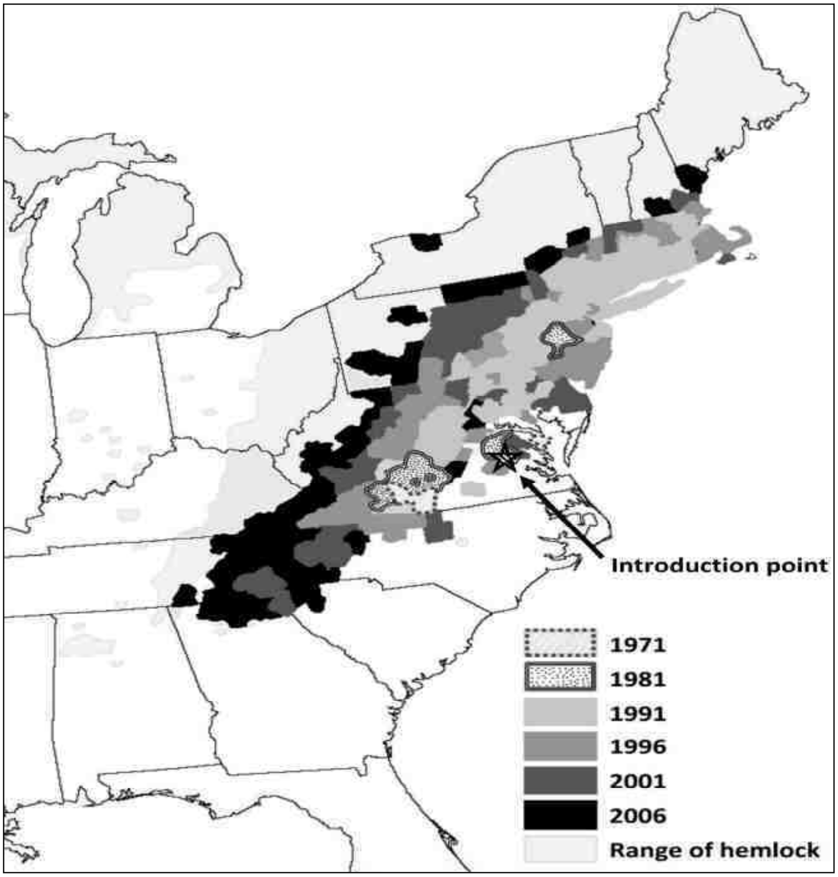


Fig. 7.11. Spread of hemlock woolly adelgid from its 1951 introduction to 2006 in eastern hemlock forests. Data from Morin et al. (2009).[23]

The majestic eastern hemlock (*Tsuga canadensis*) is a foundation tree species of eastern forests, because it provides unique ecological functions that many other species have formed relationships with. For example, the dense, evergreen foliage of hemlock trees moderates fluctuations in forest and stream temperatures, creating unique habitat. Eighty-five national parks, or 21% of the total number of parks in the national park system, are within the native range of eastern hemlock (Fig. 7.11).[2] Eastern hemlock is a component of up to 26% of the forest area of four of the largest eastern parks. These include: Delaware Water Gap National Recreation Area (New Jersey/Pennsylvania), Shenandoah National Park (Virginia), New River Gorge National River (West Virginia), and Great Smoky Mountains National Park (North Carolina/Tennessee).

Hemlock woolly adelgid (*Adelges tsugae*) is an aphid-like insect, named for the woolly white appearance it develops by producing a wool-like covering for its eggs (Fig. 7.12). The species is native to Japan, where it is a relatively innocuous component of Japanese hemlock forests.[2] Hemlock woolly adelgid was first reported in the United States in 1951, near Richmond, Virginia. From then until the mid-1980s, woolly adelgid spread slowly and was simply considered a pest to ornamental trees in urban areas. The species' behavior abruptly changed in the mid-1980s, when it spread over 20 km (13 miles) per year (Fig. 7.11).[23]

Fig. 7.12. Top left: eastern hemlock trees killed by the insect hemlock woolly adelgid within a mixed-species forest of Great Smoky Mountains National Park. Hemlock also forms a major part of forests along the park's streams (right). Bottom left: branch containing whitish sign of hemlock woolly adelgid. Top left and right photos by S.R. Abella, and bottom left courtesy of the U.S. Forest Service.

Adelgids kill trees by sucking sap and depleting the trees' starch reserves. Infested hemlock trees have died within four years, but some trees have remained alive (albeit weakened) for 20 years in northern areas such as Delaware Water Gap National Recreation Area. In many areas, however, mortality of hemlock is complete or nearly complete, and resistance of hemlock trees to the insect appears minimal. Forest changes reported in parks following woolly adelgid infestation include: increased understory plant cover (probably because of increased light reaching the forest floor under dead trees), but greater invasion of non-native plants; altered bird composition, especially of species associated with hemlock trees; likely altered native insect communities; and potential shifts in fish composition in streams.[2] Future changes could depend on which tree species replace hemlock, though many of the potential replacement species are themselves threatened by non-native pests.

Management is ongoing in several parks to reduce impacts of hemlock woolly adelgid.[2] For example, in a heroic effort, Great Smoky Mountains National Park has treated over 300,000 hemlock trees since the early 2000s in priority "hemlock conservation zones." The park's treatment method is applying a water solution of the insecticide imidacloprid to the soil around the base of hemlock trees. Treatments have kept hemlocks alive for five to eight years, before re-treatment is necessary. Park managers recognize the possibility of unintended impacts to native insects, but this must be balanced against impacts of losing hemlock. Moreover, only a fraction of the park's hemlock trees can be treated anyway.

Several additional management strategies are being implemented. Biocontrol insects (using species that are natural enemies of the species targeted for control) have been released, which does entail introducing even more non-native species. The overall ecological effectiveness and non-target impacts to the ecosystem of biocontrol insects are not yet clear. Another strategy is facilitating colonization of dead hemlock forests by other native species, such as by removing non-native plants and limiting eating of colonizing vegetation by white-tailed deer. Other options are likely to be explored in the longer term, including breeding resistance to adelgids into hemlock trees. Even if resistant trees were produced now, hundreds of years would be required for them to grow to the sizes of trees being lost. This makes the proactive treatments presently implemented at priority sites all the more significant, to help ensure that some large hemlocks remain in parks for the coming decades.

Complications with Native Insect Outbreaks

Extensive forest die offs, beginning in recent decades and often still ongoing, have occurred in western conifer forests through outbreaks of native insects. Some examples in parks include the spruce-fir forests of Cedar Breaks National Monument in Utah, pine-fir forests of Rocky Mountain National Park in Colorado, and pinyon-juniper woodlands of Mesa Verde National Park in Colorado.

In Cedar Breaks, past exclusion of fire, interacting with recent droughts, may have intensified tree mortality since a 1990s outbreak of the native spruce bark beetle (*Dendroctonus rufipennis*). Forest die off has dramatically changed the environment, such as the amount of light reaching the ground. This probably affects understory plants and wildlife (Fig. 7.13).

In Rocky Mountain National Park, researchers suspected that contemporary outbreaks of western spruce budworm (*Choristoneura occidentalis*) and Douglas-fir bark beetle (*Dendroctonus pseudotsugae*) were exacerbated by 1800s logging followed by severe fire.[15] This may have homogenized the forest landscape, resulting in an unnaturally large proportion of the landscape simultaneously reaching a forest age susceptible to insect outbreaks.

Fig. 7.13. Tree mortality in spruce-fir forests of Cedar Breaks National Monument, Utah (S.R. Abella).

Rocky Mountain National Park's lodgepole pine (*Pinus contorta*) forests, which have experienced epidemic outbreaks of the mountain pine beetle (*Dendroctonus ponderosae*) in the 2000s, illustrate the challenge of partitioning natural from human influences on native insect outbreaks. Lodgepole pine forests are naturally relatively homogenous, originating after severe fires remove a previous forest.[10] Beetle outbreaks correspond with droughts, when trees are weakened. It is unclear whether human-induced climate change has exacerbated this situation. Although most trees die during severe outbreaks, sufficient numbers of trees do withstand or avoid the outbreaks to produce seed for renewing this forest type.[10]

Old-growth woodlands of the small trees pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) in southwestern Colorado's Mesa Verde National Park have been devastated in recent decades.[12] During a 2002-2005 drought and outbreak of pinyon bark beetles (*Ips confusus*), over one-third of the park's pinyon pine trees died. In addition to droughts and insect outbreaks, severe wildfires have halved the amount of old-growth woodland in the park (Fig. 7.14). Many of these woodlands were 200 to 500 years old. Better understanding relationships of historical land uses (such as grazing and manipulation of fire) with current conditions, and how to conserve the woodlands in today's dry conditions, is a pressing need if much of the park's old woodland is to persist.

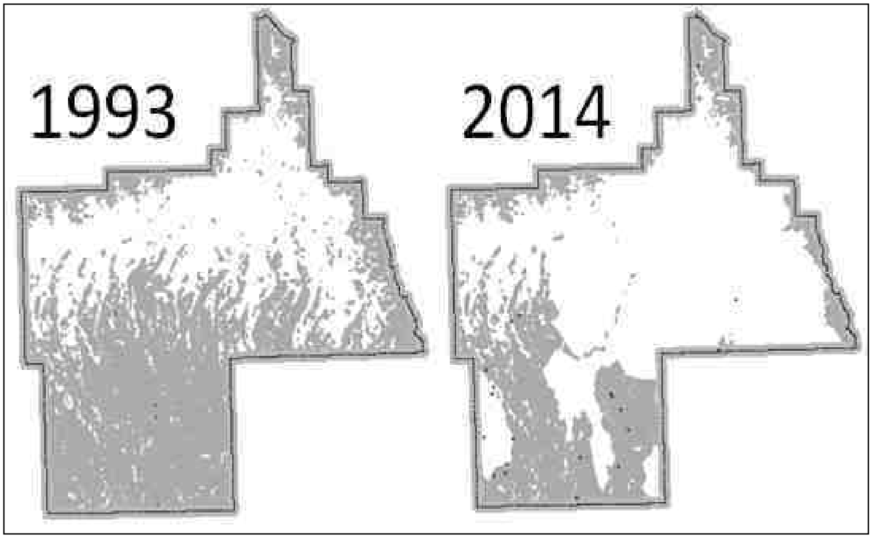


Fig. 7.14. Loss of old-growth pinyon-juniper woodland after combined effects of drought, bark beetle outbreaks, and severe wildfire in Mesa Verde National Park, Colorado. Old-growth woodlands are shown in gray on both maps. Data from Floyd et al. (2015).[12]

Outlook for Conserving National Park Forests

Forests are critical to national parks. Forests anchor aesthetic and recreation sites for park visitors, are part of the historical authenticity of historical parks, provide habitat for countless species, and contribute to global biodiversity and sequestering carbon from the atmosphere. Forest conservation relies on good ecological knowledge of variation and natural processes in forests. As Chapter 3 discussed, for example, severe fires are a natural process for renewing some types of forest. But in other forest types where humans have manipulated fuels and fire, the severe fires now occurring decimate natural forests. This underscores how crucial it is to understand what the natural processes are in a particular forest to conserve that forest. Fire management, reducing non-native species, curbing air pollution, and adapting to climate change will likely all require integration for conserving national park forests. Implementing just one of these in isolation can become meaningless. Solely focusing on helping forests adapt to future climate change, for instance, is meaningless if forests are decimated now by non-native pests or severe fires.

It is clear that forests in many national parks are changing dramatically under a "preservationist" approach that does not include any active management. This is partly because external influences, like non-native pests entering parks, continue affecting parks. Within parks, natural processes, such as fires, have been disrupted and have yet to be fully reinstated. Given these circumstances, management more active than is typical for the National Park Service may be required to conserve park forests. Actively treating non-native insects to conserve at least a fraction of eastern hemlock forests is one example. Reducing fuel to prevent severe fire where it is not natural is another example. A priority in national park forest management is identifying where, and when, active versus passive ("hands-off") management is most appropriate.

8 FRESHWATER

Freshwater has a low concentration of salts compared to the saltwater of oceans and some inland desert lakes. Freshwater is essential to humans and much of Earth's life and has no substitutes. In addition to enabling human life, the 126,000 known freshwater animal species comprise 10% of Earth's total animal species.[2] This includes 15,000 species of fish (45% of Earth's total).

Freshwater is only a small part of Earth's water and is distributed unevenly across Earth. Of the total amount of Earth's water, including in the atmosphere, 98% is saltwater and 2% is freshwater. Of the freshwater, 68.7% is perennially frozen, 30% is groundwater, and only 0.3% is on the surface (lakes, rivers, and wetlands).[2] Freshwater occupies only 0.8% of Earth's surface. Moreover, Earth's 189 largest lakes (each larger than 500 km^2, or 200 square miles) contain 68% of the planet's liquid surface freshwater.

Earth's liquid freshwater is constantly moving and interconnected through a water cycle. Water evaporates from oceans and landmasses to the atmosphere, and falls back to Earth as precipitation. Precipitation recharges rivers, lakes, and groundwater. Some of the water again evaporates to the atmosphere or flows off landmasses into oceans.

The movement and cycling of water have given rise to a watershed approach to conservation and management. A watershed is an area of land where all of the water that is under it or drains off of it goes out to the same place. The United States contains hundreds to thousands of watersheds, depending on how broadly we wish to delineate them. For example, a watershed for a small mountain could consist of the land containing groundwater and small, temporary streams carrying snowmelt, all draining into a larger river. Watershed components include groundwater, rivers, wetlands, and lakes.

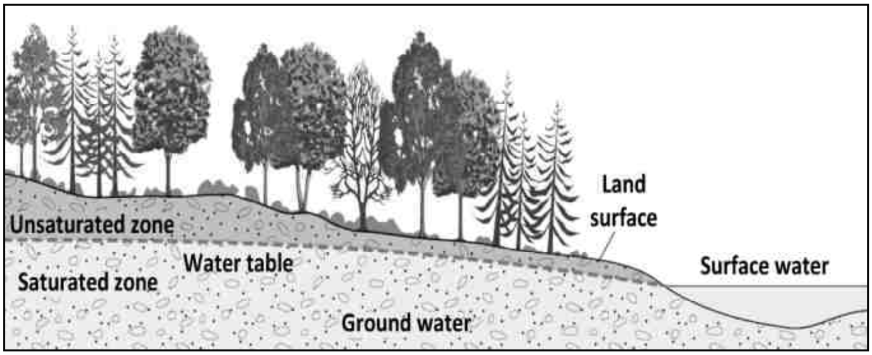


Fig. 8.1. Diagram of upper soil unsaturated with water; the water table or upper limit of water-saturated ground; and an aquifer saturated with groundwater. From U.S. Geological Survey (1999).[23]

Groundwater is water below Earth's surface (Fig. 8.1). An aquifer is defined as geologic material (rock or sediment) that is permeable and saturated with groundwater and will yield water to a well.[6] This issue of water saturation is important, because soil near the surface often contains both water and air between the soil particles and thus is not saturated with water. The water table is the upper surface of the completely water-saturated zone of an aquifer. The water table can be at the soil surface where groundwater rises to the surface, or deep in the ground requiring wells hundreds of feet deep to access water. Moving downward from a point on Earth's surface, multiple shallow and deeper aquifers can occur and be separated by impermeable rock material containing little water. Groundwater typically moves slowly through rock and thick sediment, with a flow rate of 0.3 meters (1 foot) per day considered fast.[6] In addition to supplying over a third (130 million people) of the United States' human population with drinking water, aquifers feed rivers, lakes, and wetlands. About 60 regional aquifers occur in the United States (Fig. 8.2).

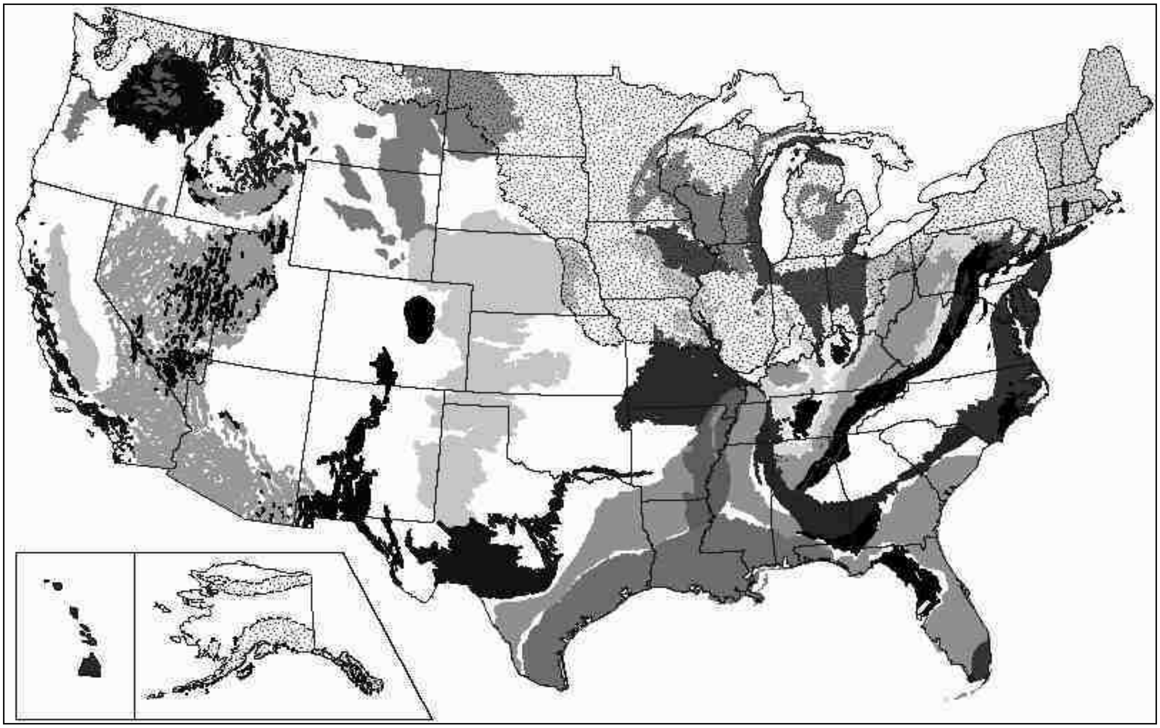


Fig. 8.2. Major aquifers shown as different shadings. Adapted from DeSimone et al. (2014).[6]

Rivers have flowing water in a channel, with small flows commonly referred to as creeks or streams. Rivers have been heavily modified by humans. A 2001 assessment reported that 5 million km (3 million miles) of rivers existed in the United States (including all 50 states).[9] Of these, only 2% were minimally affected by human activities. The rest were dammed or diverted to alter flow (79%) or drowned by reservoirs (19%).

Dams have been built to alter rivers for generating power, storing and diverting water, and creating lakes.[19] Although dams provide certain benefits, some of the negative tradeoffs of impounding rivers are altered aquatic habitat and blocking fish from moving and spawning. Dams also prevent natural flooding, affecting shorelines, and they can change the temperature and chemistry of water to further modify habitat (Fig. 8.3).

Fig. 8.3. Free-flowing Colorado River photographed in 1889 by the Stanton Expedition, and the same location after construction of the Glen Canyon Dam in contemporary Glen Canyon National Recreation Area, Utah/Arizona (2011 photo by R.H. Webb and both photos provided by the U.S. Geological Survey).

A 2013 inventory of dams by the U.S. Army Corps of Engineers reported that 87,035 dams occur in the 50 states and U.S. territories. Owners of the dams included: private entities (65%), local governments (18%), states (7%), federal government (4%), and other entities such as public utilities (6%). The U.S. Geological Survey has mapped 8,000 dams over 15 meters (50 feet) tall in the lower 48 states (Fig. 8.4).

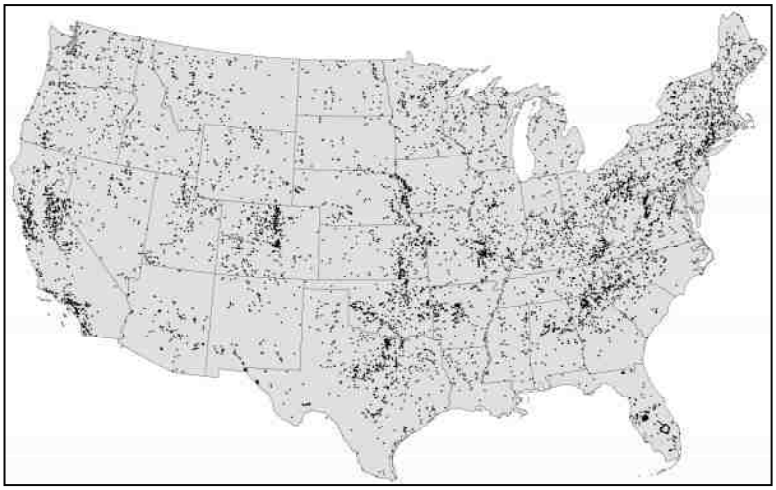


Fig. 8.4. Locations of the 8,000 largest dams in the United States. Each dam is a dot. Data from the U.S. Geological Survey.

Wetlands have water tables at or near the surface, or are covered by shallow water all or part of the year.[3] Wetlands are also commonly called swamps, marshes, and floodplains. While wetlands are often dominated by grasses and low-growing plants, many are forested. Indeed, some of the largest trees in the United States grow in wetlands, such as in Congaree National Park, South Carolina.

In the 1780s, 10% of the land that was to become the lower 48 states was covered with wetlands totaling 90 million hectares (221 million acres).[4] Euro-American settlers drained wetlands and replaced them with other land uses, such as farms, pastures, and towns. Six states lost over 85% of their wetlands: Ohio, Indiana, Illinois, Iowa, Missouri, and California. Much of northwestern Ohio, for example, was covered by the 12,000-km^2 (4,800-square mile) Great Black Swamp, nearly the size of Connecticut. The swamp was drained in the mid-1800s. By 1884, Ohio had over 32,000 km (20,000 miles) of ditches draining wetlands. Similar alterations occurred in the Everglades of Florida, Okefenokee Swamp in southern Georgia, Great Dismal Swamp in eastern Virginia and North Carolina, Horicon Marsh in Wisconsin, Mississippi River floodplains, California's Central Valley, and numerous smaller wetlands throughout the country. By the 1980s, wetland area was halved. An average loss of 25 hectares (60 acres) of wetlands occurred every hour during the 200 years between the 1780s and 1980s. Many of the remaining wetlands were altered to some extent by logging, modified drainage patterns, and other human activities (Fig. 8.5).

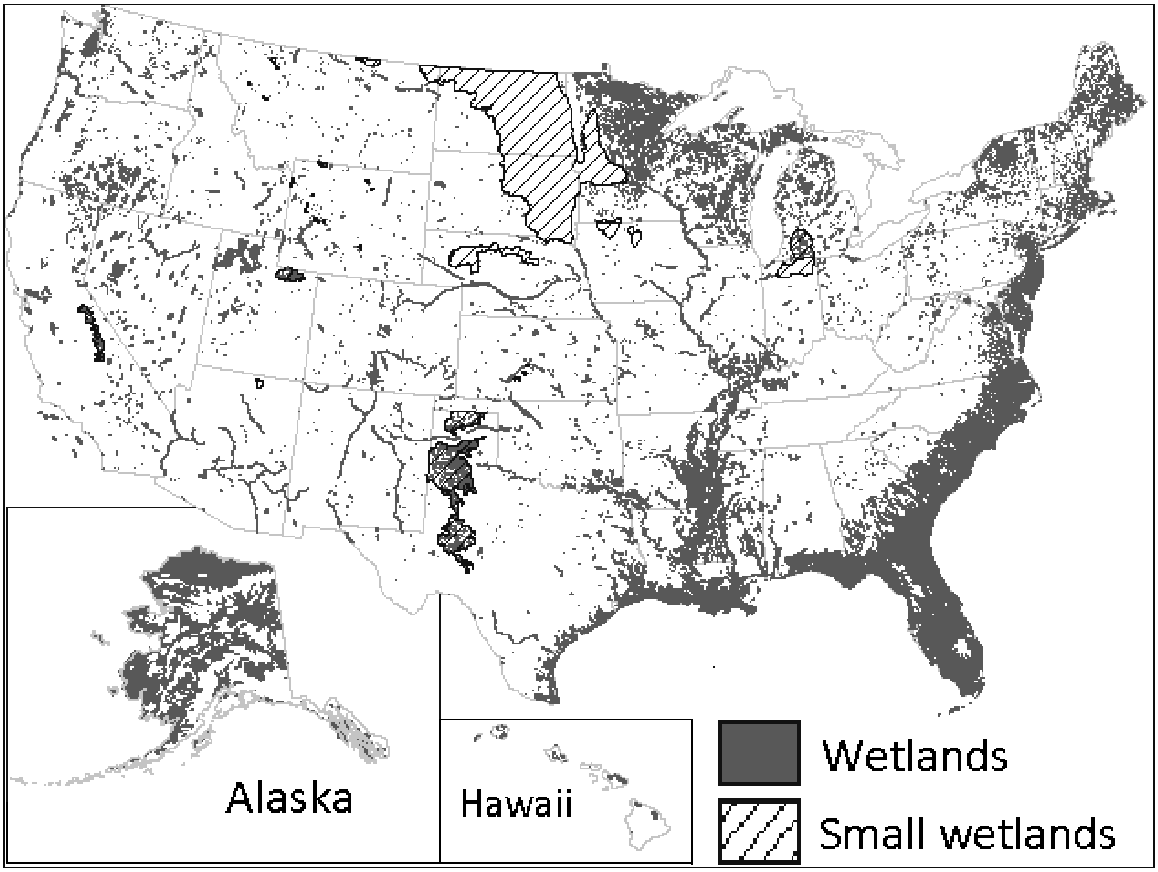


Fig. 8.5. Major wetlands of the United States in the 1980s/1990s. The hatched lines indicate areas with numerous small wetlands. Map adapted from Winter et al. (1998).[25]

Alarming losses of wetlands and growing recognition of the value of wetlands to society resulted in some protections for wetlands in the 1972 Clean Water Act and the 1986 Emergency Wetlands Resources Act. These regulations have slowed, but not stopped, the loss of wetlands. The most recent wetland inventory by the U.S. Fish and Wildlife Service reported a net loss of 25,000 hectares (60,000 acres) of wetlands between 2004 and 2009.[5]

In 2009, wetlands covered 45 million hectares (110 million acres) of the contiguous United States, or 5% of the area. Functions of wetlands are increasingly appreciated, such as improving water quality and retention (including for municipal water treatment systems), buffering coasts from damaging storms, and wildlife habitat.

National parks play a key role in conserving watersheds in the United States. National parks frequently are designated as "anchor points" for watershed conservation by containing relatively high-quality habitat and critical parts of watersheds like source waters of rivers. National parks also contain key ecosystems, including wetlands, converted to other land uses outside parks. Many parks were specifically designated to conserve water ecosystems, such as Lake Roosevelt National Recreation Area of Washington, Buffalo National River of Arkansas, Apostle Islands National Lakeshore of Wisconsin, Saint Croix National Scenic River of Minnesota, Gauley National River of West Virginia, Upper Delaware Scenic and Recreational River of Pennsylvania and New York, Everglades National Park, and Yukon-Charley Rivers National Preserve in Alaska. Several national park units are also part of the National Wild and Scenic Rivers System. Congress created this system in 1968 to conserve certain rivers in a free-flowing condition. Some national parks that are part of the system include the Obed Wild and Scenic River of Tennessee, and particular rivers within parks such as the Merced Wild and Scenic River within Yosemite National Park, California.

National parks have been locations for landmark watershed restoration projects, areas of recovery for animals like beaver that are integral to the water cycle, and research for understanding changing conditions and conservation needs of watersheds. This chapter provides examples of changing conditions and restoration of watersheds in national parks.

Removing Roads to Improve Redwood National Park Watersheds

Redwood Creek flows 107 km (67 miles) from its source waters in mountains of the Coast Range to the Pacific Ocean, near Orick, California (Fig. 8.6). Most of the creek flows through Redwood National Park, expanded in 1978 by adding previous industrial timberlands. Floods from the 1950s to the 1970s in the logged forest damaged Redwood Creek and its feeder streams through gullying, soil erosion, choking streams with sediment, and disrupting fish habitat. Over 650 km (400 miles) of logging roads crisscrossed the park in 1978 and generated sediment eroding into Redwood Creek.[14] Redwood Creek's sediment-laden water was evident where it flowed into the Pacific Ocean (Fig. 8.7).

As part of the 1978 expansion of the park, Congress directed the National Park Service to curtail human-induced erosion on the new park lands. Particular focus was on reducing input of sediment to streams from eroding logging roads. Removing a road and restoring natural forest is a major endeavor. Early techniques in the 1970s and 1980s focused on de-compacting the road surfaces (such as using heavy equipment to break up the soil), removing culverts (structures concentrating water into a channel below a road), and moving road material away from streambanks. By the 1990s, more aggressive treatments moved large volumes of soil to re-create the original topography as closely as possible.

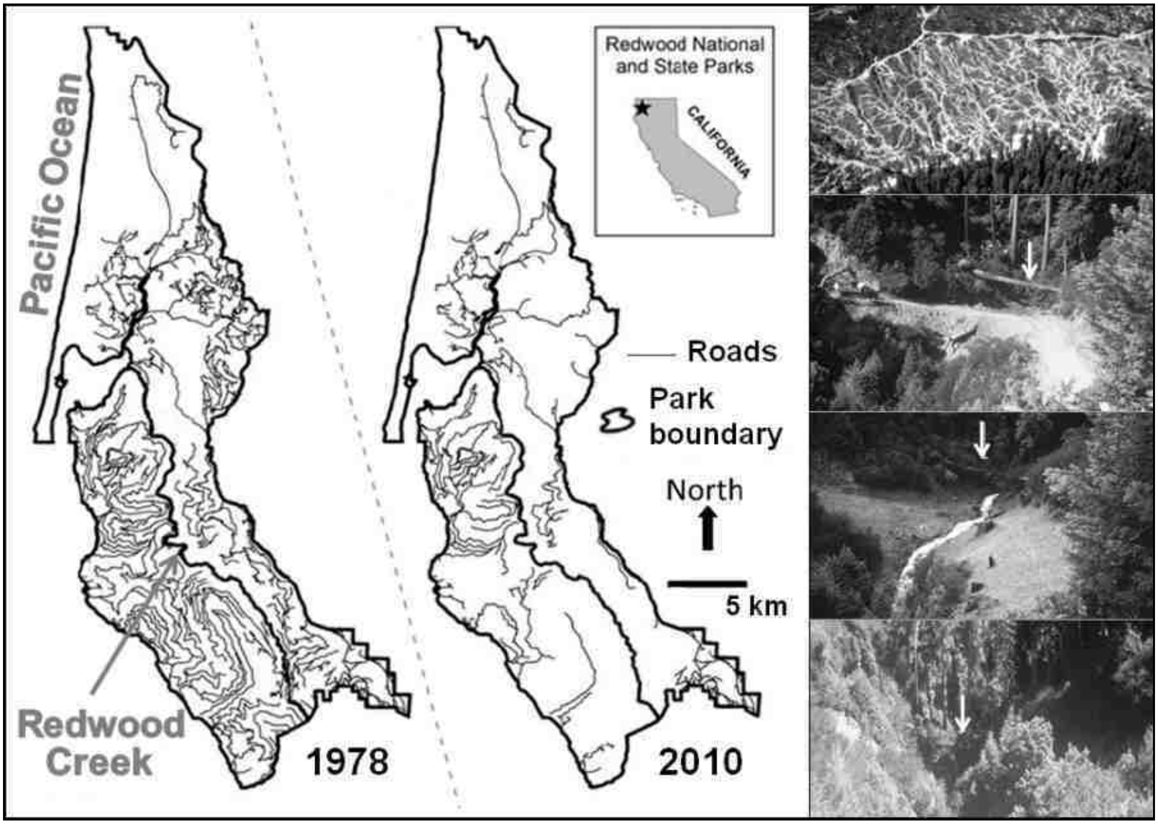


Fig. 8.6. Redwood National Park's road network in 1978 and in 2010, after 32 years of the road removal program. Maps adapted from Madej et al. (2013).[14] The top photo shows a typical view of a road network before removal. The three bottom photos show the progression of decommissioning a road and its revegetation (National Park Service photos).

After 32 years of work, the National Park Service decommissioned 425 km (265 miles) of roads.[14] Only 35% of the original road length remained in 2010 (Fig. 8.6). As the restoration program progressed, condition of the Redwood Creek watershed improved. This was reflected in a long-term reduction of the streambed elevation of Redwood Creek, indicating that the creek switched from filling with sediment, to scouring. The creek essentially became "unclogged" of its accumulated sediment (Fig. 8.7).

Disappearing Wetlands in Great Sand Dunes National Park

Pinpointing causes of changes in watersheds is not always straightforward, as an example from Great Sand Dunes National Park illustrates. The 18,000-hectare (44,000-acre) park is southwest of Colorado Springs, Colorado, in the San Luis Valley between the Sangre de Cristo and San Juan Mountains. The Valley's climate is desert-like, with only 20

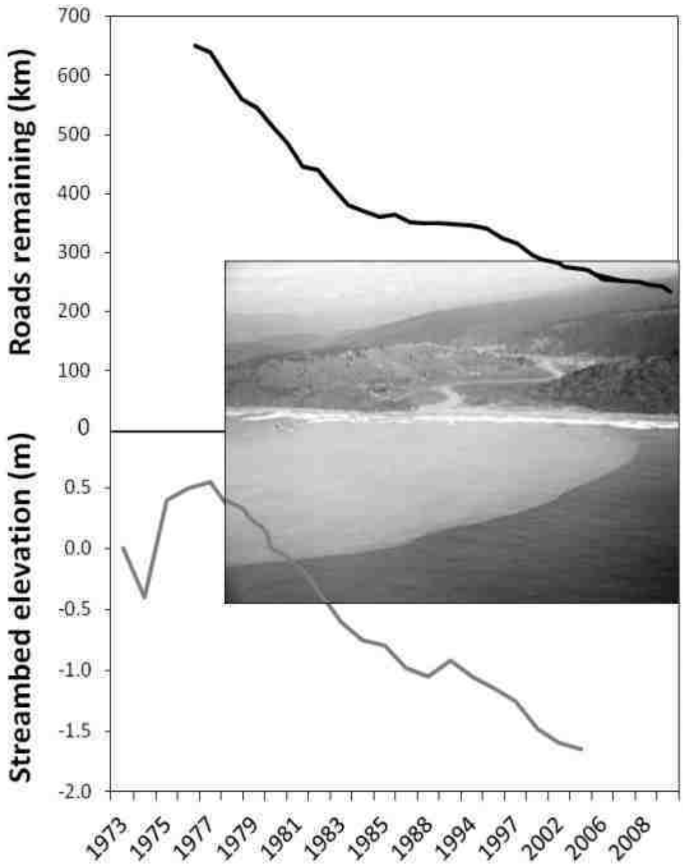


Fig. 8.7. As eroding roads were decommissioned (top graph), the depth of sediment (meters) in Redwood Creek declined from the 1970s to 2000s (bottom graph). Data from Madej et al. (2013)[14] for the top graph and from Redwood National Park for the bottom graph. Photo, courtesy of Redwood National Park, shows Redwood Creek emptying into the Pacific Ocean.

centimeters (8 inches) of annual precipitation. The park contains the tallest sand dunes in North America, with dunes 230 meters (750 feet) tall. Wetlands occupy low-lying areas between the dunes. Within the dry landscape, wetlands are oases of biological activity. Individual wetlands are typically small, less than 0.4 hectares (1 acre) in size, but together form extensive habitat for birds and other wildlife.

Researchers assessed changes to wetlands within 15 km^2 (6 square miles) in the northwestern part of the park east of Sand Creek.[26] Based on comparing aerial photographs taken from planes, wetland acreage declined by 76% between 1937 and 1995 (Fig. 8.8). The number of wetlands decreased by over half, from 114 to 51.

What caused this decline? Researchers evaluated several possibilities, such as groundwater pumping, alterations to stream channels, and climate change. As early as 1891, people had created 2,000 wells tapping the San Luis Valley's aquifer to irrigate crops and withdraw drinking water. Over 7,000 wells tapped the aquifer by 1958. Concerned with depletion of groundwater in the 1970s, a moratorium was placed on drilling new wells in 1981. However, unlike in many areas where groundwater pumping has lowered the water table to "dry up" wetlands and streams, researchers did not believe that groundwater pumping was the prime culprit for the disappearing wetlands in this area of the park. The area was largely disconnected from the part of the aquifer most affected by pumping, and much of the decline in wetlands occurred before particularly intensive pumping of the 1970s.

Instead, researchers surmised that the main cause of the disappearing wetlands was severe drought.[26] With reduced mountain snowpacks and water inputs to Sand Creek, its flow decreased, making less water available to seep out of the stream channel to recharge groundwater. This lowered the water table by 1 meter (3 feet). Although that may not seem like much, it made the difference between water flowing out to the surface, sustaining the wetlands, versus remaining below ground, causing the wetlands to dry up.

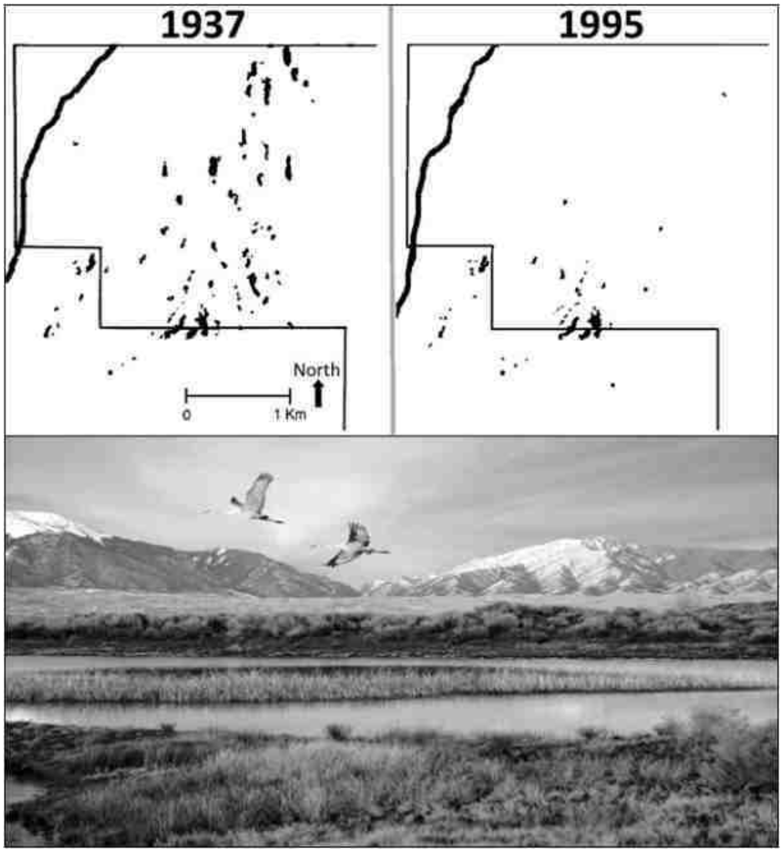


Fig. 8.8. Loss of wetlands (black areas) between 1937 and 1995 east of Sand Creek in Great Sand Dunes National Park, Colorado. Data from Wurster et al. (2003).[26] Photo shows sandhill cranes flying over current wetlands (by P. Myers, National Park Service).

Different combinations of factors than at Great Sand Dunes National Park might affect watersheds in other places differently. It consistently is important to identify long-term changes in watersheds, causes of the changes, how the changes affect habitat, and whether conservation measures are needed. Current climate change complicates matters. For instance, loss of wetlands in Great Sand Dunes National Park could actually be a natural process in response to fluctuations in dry and wet periods. Separating out whether human activities exacerbated naturally dry climate periods, via climate change, groundwater pumping, or other activities, is challenging but necessary to identify ecologically appropriate conservation strategies.

Restoring Wetlands in Lassen Volcanic National Park

Reestablishing natural water flow where it has been disrupted by human activities can produce seemingly magical ecological outcomes. An example was the 33-hectare (82-acre) Drakesbad Meadow, the largest wetland in Lassen Volcanic National Park, northern California.[17] In a basin surrounded by conifer forest at the park's high elevations, the meadow receives water emanating from springs fed by groundwater flowing out of surrounding rock ridges. Before the park was created, irrigation and drainage ditches were constructed in the meadow from the late 1800s to the mid-1900s for agriculture and ranching. These modifications continued intercepting surface and groundwater flows after inclusion of the meadow into the park in 1958. Additionally, a road was constructed on the north side of the meadow, further diverting water flow from the springs to the meadow. Instead of water flowing into the meadow and spreading out, water was conveyed past the road into a few culverts. To identify changes in water table depths, in 2002 researchers installed 87 monitoring wells by augering holes into the soil. Plastic pipes, 4 centimeters (2 inches) in diameter, were then inserted to measure water levels.

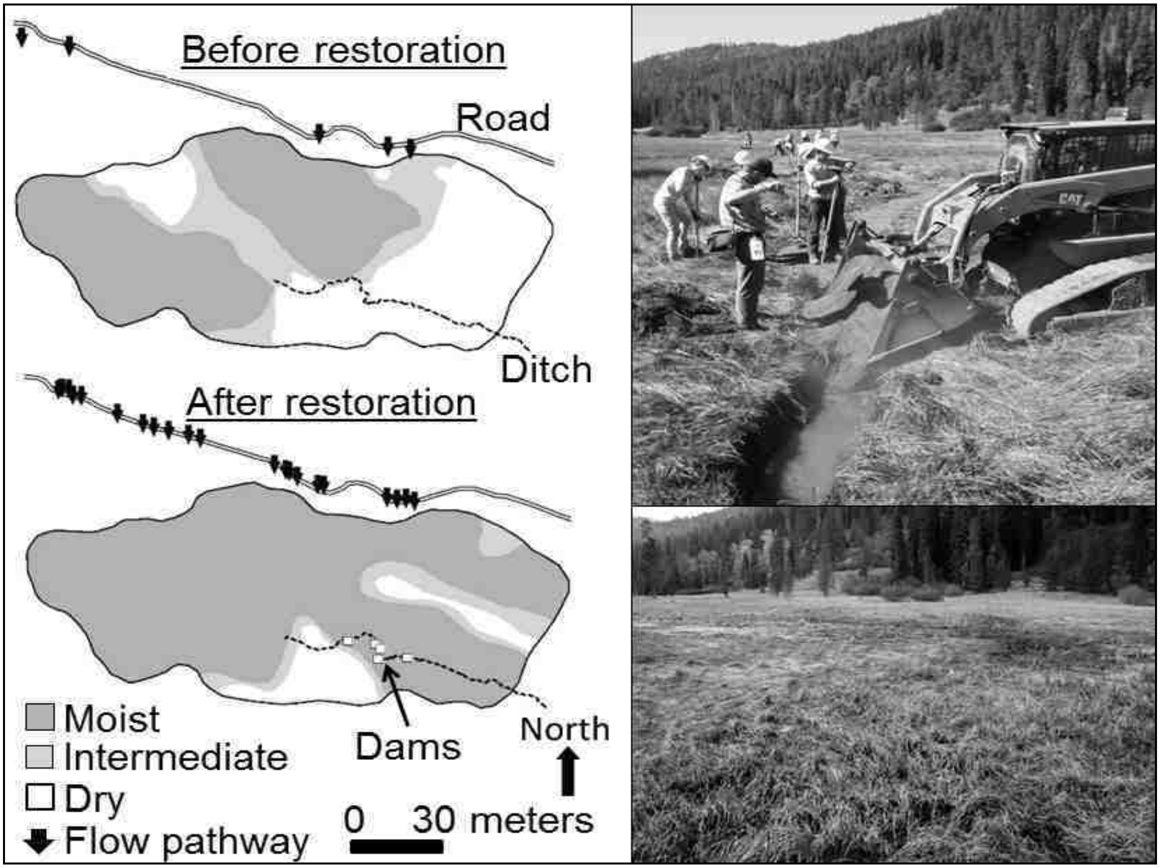


Fig. 8.9. Restoring surface water flow to Drakesbad Meadow, Lassen Volcanic National Park, California. Top diagram shows flow blocked by a road and stopped by a drainage ditch. Bottom diagram shows increasing wetness of the meadow after restoring flow paths by breaching the road and blocking the drainage ditch with small check dams. Diagrams adapted from Patterson and Cooper (2007).[17] National Park Service photos show during and after blocking the drainage ditch.

In 2003, the park began restoring natural water flow to and through the meadow. The park breached the road in 21 locations by constructing channels across the road as water flow paths. Next, five metal dams were installed to slow water flow in the largest drainage ditch within the meadow. This shifted the ditch from its original purpose of rapidly conveying water out of the meadow, to retaining water in the meadow.

The restoration treatments quickly enhanced the meadow's wetland characteristics. Constructing channels across the road increased the number of flowpaths reaching the meadow from five to 21. This added 73,000 liters/day (19,000 gallons/day) of water reaching the meadow that had been diverted. Once it received this flow, the meadow's water table rose immediately. In one monitoring well nearest the road, the water table rose within 24 hours from a depth of 74 centimeters (29 inches) to within 7 centimeters (3 inches) of the ground surface. Blocking the drainage ditch further raised groundwater levels. In 2004, one year after the restoration, 82% of the meadow had a water table within 20 centimeters (8 inches) of the surface. This was more than double the area before restoration. In response, the native wetland plants short-beaked sedge (*Carex simulata*) and panicled bulrush (*Scirpus microcarpus*) increased, while the non-native bluegrass (*Poa pratensis*) decreased.

Recolonization of Beaver and the Construction Boom in Voyageurs National Park

Before humans began building dams, beaver (*Castor canadensis*) had a monopoly on dam building in North America. Beaver are special animals with engineering skills to construct their own homes and dams for modifying their habitat in ways favorable to them. Beaver extensively affect watersheds. The native range of beaver encompasses Alaska, Canada, and the contiguous United States, including desert Southwest streams and wetlands.

As North America's largest rodent, adult beaver weigh 16 to 32 kilograms (35 to 70 pounds), are 64 to 120 centimeters (25 to 47 inches) long, and are well adapted for modifying watersheds.[1] About a third of a beaver's length is its flattened, paddle-like tail. Beaver use their tail for storing fat and communicating with other beaver, such as loudly slapping the water to signal danger from predators. Beaver have 20 teeth, including chisel-like incisors that grow continuously and thus need to be regularly worn down. Powerful facial muscles enable beaver to use their teeth for cutting large trees. Beaver's nimble forefeet can carry building materials, such as tree branches, and rotate pencil-sized twigs for gnawing off the bark. Beaver have the ability of functioning both in water and on land. They are excellent swimmers and can remain underwater for 15 minutes. They are somewhat clumsy on land, but can walk upright, being partly supported by their tails. Beaver are social animals, usually living in groups consisting of an adult female and male, along with two to six juveniles less than three years old. Beaver are vocal and can whine, hiss, and growl, depending on the situation. Whines by young kits often result in adults providing food to the kits.

Beaver build dams at favorable locations (such as where watercourses narrow) by pushing soil, rocks, or branches to form a foundation (Fig. 8.10). Beaver then fortify their dams using a variety of woody materials and debris. When their preferred food plants are in short supply, beaver often use plants of low palatability for "construction materials" to save their preferred food plants. Beaver regularly inspect their dams and repair breaches. One purpose of building dams is increasing water depth to enable ice-free water for swimming at the bottom of ponds in winter.

Beaver construct lodges of mud and wood, with the lodges in open water or near shore (Fig. 8.10). A chamber is constructed above the waterline within lodges, sheltering beaver from weather and predators. In northern regions, beaver construct food caches near lodges by sinking "rafts" of intertwined branches, which they access in winter.

Fig. 8.10. Top: beaver dam along Bright Angel Creek, Grand Canyon National Park, Arizona (S.R. Abella). Bottom: beaver lodge during winter on frozen lake, Acadia National Park, Maine (National Park Service photo).

Beaver are herbivores. Their preferred food plants include deciduous shrubs or trees, such as alder, willow, and cottonwood, and non-woody plants like sedges and cattails. Beaver eat leaves, twigs, roots, fruits, and acorns when available. An adult beaver consumes 0.5 to 2 kilograms (1 to 4 pounds) of woody material daily. This foraging, combined with woody plants cut for building material, influences an area's vegetation. For example, a group of four beaver on a pond each eating 1 kilogram (2 pounds) daily would consume 1,460 kilograms (3,220 pounds) of woody material from area forests annually, not including building materials. By altering vegetation via selectively cutting woody plants and creating ponds, beaver affect habitat for other species such as fish, insects, and birds.

An estimated 60 to 400 million beaver occupied North America at the time of Euro-American settlement.[16] Beaver fur was valued by Europeans. As a result, beaver populations were decimated by fur trapping from the 1600s through the 1800s. For instance, 1 million beaver were removed from Connecticut, Massachusetts, and western New York from 1620 to 1640 to support the fur trade. Regulation of trapping after the early 1900s allowed beaver populations to expand and increase, though the current population of 6 to 12 million is less than before Euro-American settlement. Beaver presently influence national parks as wide-ranging as Acadia National Park in Maine, Cumberland Gap National Historical Park in Kentucky, Grand Canyon National Park in Arizona, and Denali National Park in Alaska.

One example is Voyageurs National Park, in northern Minnesota, where beaver activity has increased since 1940.[11] Within the park's 250-km^2 (100-square mile) Kabetogama Peninsula, beaver increased from almost none in the 1930s, to an average of one group of adults and juveniles within a km^2 (3 groups/square mile) in the 1980s.

This stimulated a construction boom. The number of beaver dams on the landscape increased from 71 in 1940 to 835 in 1986. Correspondingly, the area covered in ponds and wetlands created by beaver dams increased from 1% to 13% (Fig. 8.11).

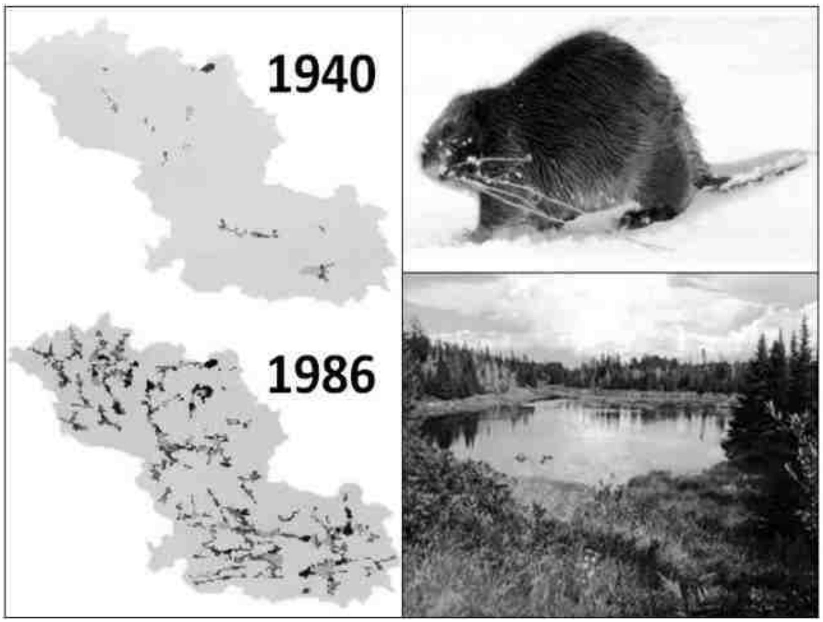


Fig. 8.11. Dark areas on the maps signify areas influenced by beaver activities, which increased between 1940 and 1986, within 45 km^2 (17 square miles) of the Kabetogama Peninsula, Voyageurs National Park, Minnesota. Adapted from Naiman et al. (1988).[16] The top photo shows a beaver carrying willow twigs (J. Peaco). The bottom photo shows a beaver pond along the Cruiser Lake Trail, Voyageurs National Park (M. Holly).

In addition to changing vegetation and habitat for fish, birds, and other wildlife, beaver have influenced patterns of nutrient cycling and carbon retention.[16] For example, beaver dams and the resulting ponds trap sediment containing particles of organic matter composed of carbon, nitrogen, and other elements. Beaver ponds become storage areas for carbon by retaining this sediment, supporting growth of aquatic plants, and receiving plant material transported to the ponds by beaver.

Changing Water Levels and Lake Mead's Shoreline Habitat

While lakes created by human-built dams in national parks are not natural features, they provide some aquatic habitat and are subject to changes that influence land habitats. Lake Mead, just east of Las Vegas, Nevada, was created by impounding the Colorado River with completion of the Hoover Dam in 1935. By volume, Lake Mead is the largest reservoir in the United States.[21] It provides drinking water to 25 million people in three western states (California, Nevada, and Arizona), including the cities of Los Angeles, San Diego, Las Vegas, and Phoenix. It also provides irrigation to 1 million hectares (2.5 million acres) of cropland. At maximum capacity, Lake Mead covers 637 km^2 (246 square miles), is 162 meters (532 feet) deep, and has 1,221 km (759 miles) of shoreline. Water flowing out of Lake Mead at the Hoover Dam runs 17 turbines, 15 of which are rated at 178,000 horsepower each. These turbines generate 4 billion kilowatt hours of electricity annually, powering 1.4 million homes.

Fig. 8.12. Receding Lake Mead, shown in 2015 at Boulder Basin (S.R. Abella).

Lake Mead is surrounded by the 563,000-hectare (1.4-million-acre) Lake Mead National Recreation Area, established in 1964 as the first national recreation area in the United States. While the recreation area may be best known for Lake Mead, it also contains some of the highest-quality land habitat in the Mojave Desert.

Although the recent lowering water level of Lake Mead may be detrimental to aquatic resources and alarming from a water supply perspective, it has exposed formerly submerged soil that now is land habitat. The last time Lake Mead was near full capacity was 1999, following wet years with deep snowpacks in the Rocky Mountains feeding the Colorado River. Dry conditions in the 2000s prevailed, and Lake Mead's depth decreased by 41 meters (133 feet). By 2010, the lake level was its lowest since the 1930s during construction of the Hoover Dam. A white "bathtub ring" around the lake attests to the drop in water levels (Fig. 8.12). This drawdown of the lake exposed 25,000 hectares (60,000 acres) of desert land that had been submerged underwater as long as 75 years (Fig. 8.13).

Would this "new" land be colonized by plants, or would the soil be so altered by submersion it could not support plant growth? Somewhat surprisingly, there were few differences in 2011 between soil on lands formerly submerged and never submerged.[8] At Boulder Beach, Stewarts Point, and Overton Beach, concentrations of salt were slightly greater in soils recently exposed (less than three years). The concentrations dropped by 10 years of exposure, to levels similar to never-submerged soil. Soil texture (proportion of sand, silt, and clay) and other properties differed little among soils varying in exposure time.

As may be expected for a disturbed environment, non-native plants, especially saltcedar trees (*Tamarix ramosissima*), were the main colonizers of recently exposed shoreline.[8] Once a land surface had been exposed for six years, however, saltcedar diminished as water levels continued receding. At least 10 years of exposure was typically required for native perennial plants to colonize the shoreline. At Stewarts Point, which contains soils with gypsum, a special native colonizer was the rare wildflower Las Vegas bearpoppy (*Arctomecon californica*).

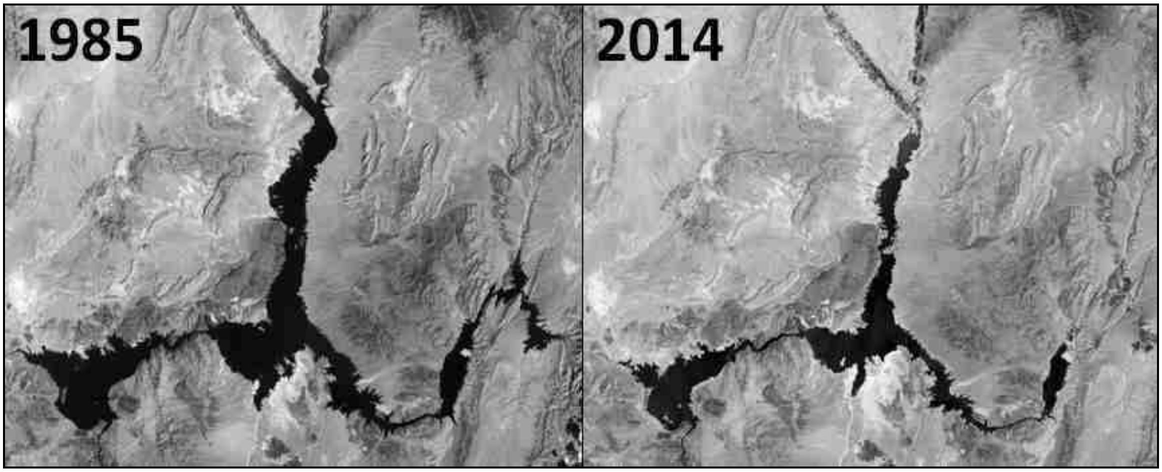


Fig. 8.13. NASA satellite images illustrating contraction of Lake Mead and re-exposure of land habitat.

Managing recently exposed shoreline is challenging. Although the overall trend is receding water, the lake levels fluctuate from winter to summer and among different years. Why treat non-native plants along the shoreline if they will simply be inundated by rising water? Treating them risks wasted effort, while not treating them risks seed production resulting in other infestations. Likewise, planting native trees along the shoreline could be desirable for accelerating native plant colonization, but only if water levels remain low.

To deal with this uncertainty, the park plans to designate high-elevation portions of the shoreline as unlikely to be re-submerged soon. These areas could be managed as essentially new, semi-permanent land habitat. Shoreline closest to the water, with the greatest chance of being re-submerged, could be left alone or strategically treated for non-native plants. The mid-elevation area is especially uncertain. There, the park is interested in identifying high points of land (such as small islands) likely to remain above water even if the lake rises. Efforts to plant native trees could focus on these high points, and potentially serve as seed sources for plant colonization of surrounding freshly exposed land. Similarly, bulldozers or other equipment could be used to create these high points, given that the shoreline already is an artificial, manipulated environment. Priority areas for this type of management may include where the Muddy and Virgin Rivers empty into Lake Mead. Although wreaking havoc for water recreation and water storage, the receding lake has created an opportunity to restore habitat for birds and other land species.

An Explosive End to River Impoundment in Olympic National Park

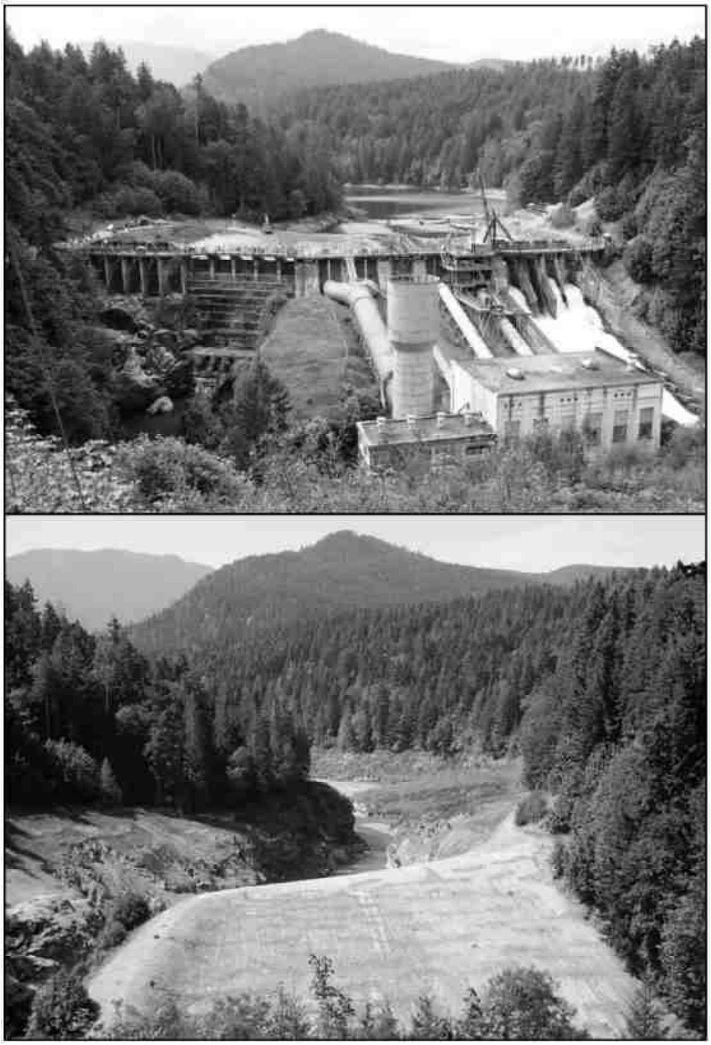


Fig. 8.14. Top: Elwha Dam photographed by Olympic National Park webcam at the start of removal operations in 2011. Bottom: August 2014, after dam removal, with the river flowing through on the bottom left of the photo (S.R. Abella).

In Washington's Olympic National Park, a recipe for dam removal on the Elwha River occurred when negative ecological tradeoffs of dams coincided with supporters of removal and uncertain economic viability of continuing to operate the dams (Fig. 8.14). One of the removal advocates was the Lower Elwha Klallam Tribe, whose traditional salmon fisheries were disrupted by the dams.

Originating near Mount Olympus in the Olympic Mountains, the Elwha River flows 72 km (45 miles) to the Pacific Ocean near Port Angeles, Washington. The river drains a watershed of 833 km^2 (320 square miles), 20% of Olympic National Park. Before the park was established in 1938, the 33 meter (108 feet) tall Elwha Dam was completed in 1913 and the 64 meter (210 feet) tall Glines Canyon Dam in 1927. The dams converted the river to reservoirs.[7] Lake Aldwell formed behind the Elwha Dam and Lake Mills behind the Glines Canyon Dam.

The Elwha River historically had a large fishery, including spawning runs of 500,000 fish of salmon species such as steelhead (*Oncorhynchus mykiss*), coho salmon (*Oncorhynchus kisutch*), and chinook salmon (*Oncorhynchus tshawytscha*).[18] This ended with completion of the dams. The dams blocked salmon from moving upriver to their spawning habitat. In the post-dam era, only a few miles of the river between the Pacific Ocean and the Elwha Dam were navigable by salmon, reducing the river's salmon population to a few thousand.

In 1992, Congress enacted the Elwha River Ecosystem and Fisheries Restoration Act, directing restoration of natural ecosystems and salmon fisheries. In 2000, the federal government purchased both dams for $30 million, setting the stage for their removal. The ensuing removal that began in 2011 of the two dams is considered the largest project of removing dams in U.S. history.

Fig. 8.15. Top: Elwha Dam mostly eliminated in 2012, but before the diversion channel on the right side was filled in (National Park Service). Star shows the location where the bottom photo was taken. Bottom: restored Elwha River channel, August 2014 (S.R. Abella).

Removing large dams is complex. Before removal could even begin, new water intake structures and water treatment plants had to be built for Port Angeles, which draws its water from the Elwha River. Additionally, the Lower Elwha Klallam Tribe needed to modify flood-control structures for its reservation, near the mouth of the Elwha River. It was estimated the dams contained 35,000 cubic yards of concrete – half the amount used to construct the Empire State Building.[20] A "blow-and-go" approach of dynamiting a dam all at once and allowing water, sediment, and concrete to pour out can work for small dams, but was unsuitable for the Elwha's large dams.[13] To avoid catastrophic flooding that could endanger human infrastructure and destroy habitat, removing the dams was done in phases.

Abolishing both dams involved explosives, re-contouring topography to restore the former river channel, and removing concrete and other building materials. Removal of the Elwha Dam commenced in June 2011 by lowering Lake Aldwell's water level behind the dam using the dam's spillways. Next, a temporary diversion channel was excavated through the left spillway for additional draining of the reservoir.

Beginning in September 2011, explosives and heavy equipment, including a hydraulic hammer positioned on a floating barge, removed concrete and metal from the Elwha Dam over a period of months. Extensive moving of soil also was required to approximately reconstruct the river's former channel, which had been destroyed by the dam (Fig. 8.15).

On March 16, 2012, the Elwha River freely flowed through its channel for the first time in 100 years. Contractors continued removing concrete and gravel, re-contoured the land, and filled remaining spillway channels. On August 24, 2012, adult Chinook salmon were observed throughout the first 20 km (13 miles) of the river, from the Pacific Ocean upstream of the former Elwha dam to the Altair Bridge.

Removing the Glines Canyon Dam also started in September 2011 and continued through 2014. During the first two years of removal of both dams from September 2011 to September 2013, 10 million metric tons (11 U.S. tons) of sediment that had accumulated for decades in the reservoirs were released by free-flowing water.[24] The river may continue flushing accumulated sediment out to the ocean. Through 2017, the National Park Service anticipates planting 400,000 seedlings of native plants to accelerate revegetation of land formerly below the lakes. Over the next 30 years, the Elwha River's salmon population is forecasted to reach 400,000 fish, similar to pre-dam levels. Reestablishing the free-flowing Elwha River is thrilling. It is anticipated to provide unprecedented opportunities for understanding ecological responses to removing large dams.

Conserving Freshwater in a Changing World

Many issues that challenge conservation in parks at least partly originate outside of parks. Conserving rivers and other watershed features, altered before they reach parks, is no exception. Water, food production, and energy use are interrelated in broadly affecting U.S. watersheds. Shifts in water use by humans, or altered availability of water such as through climate change or contamination, will likely influence watersheds including in parks. A 2010 inventory of water use estimated that the United States used 1.2 trillion liters (306 billion gallons) of water daily.[15] This is equivalent to 464,000 Olympic-size swimming pools. Of this, 75% was from surface water sources and 25% was from groundwater. About 38% each was used for irrigation (crops, pastures, golf courses, and large lawns) and for producing thermoelectric power. This is done by heating water to produce steam to turn turbines in power plants. The public supply used 14%, for domestic purposes in homes and businesses. Industries obtaining their own water used another 4%. The remainder was used for mining, aquaculture (including fish farms), ranching, and self-supplied domestic typically in rural areas with groundwater wells.

Energy production can influence watersheds even without directly consuming water. Fossil fuels provided 82% of the 98 quadrillion BTU energy consumption of the United States in 2014.[22] Natural gas was a primary fossil fuel, supplying 28% of total U.S. energy use. The process of extracting natural gas, popularly known as "fracking," involves drilling through aquifers and pumping chemicals deep into the Earth. Much of the processed water remains in the ground, while some flows back up to the surface and must be effectively dealt with to avoid contaminating surface waters and shallow aquifers. While potential effects of fracking are controversial and being vigorously debated, care is needed to avoid major risk to long-term freshwater resources while supplying short-term energy.[12]

Solar power provided 0.4% of energy used by the United States in 2014, and initiatives such as the California Desert Renewable Energy Conservation Plan aim to increase that percentage. This may seem good. Unfortunately, building solar farms thousands of acres in size on public lands, including around national parks, is prioritized over rooftop solar and other technologies that would avoid creating new land-surface disturbances. Watersheds in deserts are particularly sensitive. Severing drainages and flow of surface water by bulldozing for solar farms can impact desert watersheds, at a time when water is in short supply in California and other southwestern states.

Another renewable energy source, dams, has a major ecological impact but produces proportionally little energy. While not all dams were built to generate energy, the 87,000 dams in the United States produced only 2.5% of the energy used by the United States in 2014.[22] The Federal Emergency Management Agency is increasingly concerned about the age, deterioration, and sediment accumulation of many dams. Owing to gradual deterioration, the life expectancy of most dams is 50 to 100 years. Thereafter, maintenance costs and risk of catastrophic failure grow. By 2020, over 70% of the existing dams in the United States will exceed 50 years old. Public safety, economic, and ecological considerations may coincide to incentivize decommissioning certain dams.

Indeed, there is an increasing trend to remove dams in the United States, which may benefit river ecosystems within and around parks.[10] The organization American Rivers reported that 72 dams were removed in the United States in 2014, restoring free-flowing condition to 1,200 km (700 miles) of rivers. In addition to the Olympic National Park dams, other recent removals included: the Upper Swepsonville Dam, built in 1790 to power gristmills and sawmills, on the Haw River, North Carolina; Idylwilde Dam, built in 1925 on the Big Thompson River, east of Rocky Mountain National Park near Loveland, Colorado, damaged by flooding in 2013 and with its material recycled to rebuild U.S. Highway 34 leading into the park; five earthen dams 2 to 6 meters (15 to 20 feet) tall on a tributary of the Cuyahoga River flowing through Cuyahoga Valley National Park, Ohio; and the Newbold Diversion Dam along the Gros Ventre River in Grand Teton National Park, Wyoming.

Effects of climate change on watersheds are uncertain, because climate change manifests differently in different areas of the country. For example, in areas where precipitation increases, increased groundwater recharge and enhanced river flow may be anticipated. However, this may not be the case if the type and timing of precipitation shifts. A slowly melting snowpack in spring influences watersheds differently than torrential summer rains. Similarly, forest die offs from wildfire or non-native pests can affect the amount and quality of water running off the land into streams. These observations underscore a need to manage land and water as connected systems. Moreover, they show the value of watershed restoration projects within parks. For example, restoring natural water flow into wetlands of California's Lassen Volcanic National Park likely buffers the ecosystem from possible reductions in precipitation during drought cycles.

9 AIR

Many people are aware of the influence of air quality on their health and daily activities. Indeed, measures of air quality, visibility, and airborne allergens are commonly provided with daily weather reports. Poor air quality can aggravate asthma, create difficulty breathing, increase susceptibility to respiratory infections, reduce brain functioning, and contribute to cancers.[15] When air pollutants are deposited onto land and water, they also harm quality of food and natural ecosystems. Mercury accumulation in fish is just one example.

Air quality is an important measure of the condition of national parks for several reasons. First, air quality can affect the experience people have during visits to parks. National parks of all places should be areas to experience clean air and good visibility. In fact, 90% of people noted that scenic views and the “smells of nature” were important to their visit to parks.[35] Both of these values are compromised by air pollution. Second, air pollutants can affect plant growth, fish and wildlife health, and cultural resources parks are mandated to conserve. Human-built structures made of limestone, for example, are decomposed by acid-rain forming air pollutants. Third, by being distributed both near cities and in some of the most remote parts of the country, national parks are key locations for understanding the transport of pollutants across the country and the world. Fourth, air pollution can change Earth’s climate, with many implications for parks.

Sources of air pollution are diverse. Many pollutants originate outside of parks and blow in, but some, such as pollutants from vehicles, are produced inside parks. Stationary pollution sources include industrial sites, power plants, paper mills, oil and natural gas extractions, and livestock operations. Mobile sources, which can move through parks to pollute along the way or generate pollutants that blow in, are vehicles, airplanes, trains, and ships. Volcanoes and wildfires are natural sources, though some extreme western fires related to human alteration of fuels probably emit unnaturally large amounts of pollutants.

Table 9.1. Examples of air pollutants and their effects. The list is not exhaustive, as additional effects of these pollutants occur and many more pollutants exist.

Pollutant	Symbol	Effects on humans	Effects on environment
Sulfur dioxide	SO_2	Aggravate asthma	Contributes to acid rain
Nitrogen dioxide	NO_2	Aggravate lung diseases	Acid rain and nutrient enrichment
Ammonia	NH_3	Lung, heart disease	Nutrient enrichment
Particulate matter	PM	Lung, heart disease	Visibility, can alter rainfall pattern
Ozone	O_3	Asthma, lung disease	Damages vegetation
Lead	Pb	Brain, cardiovascular	Accumulates in soil, water, biota
Mercury	Hg	Liver, kidney, brain	Accumulates in soil, water, biota
Volatile organics	VOCs	Cancer	Forms greenhouse gases

Some major air pollutants are ozone, sulfur and nitrogen oxides, particulate matter, metals (such as mercury), and volatile organic chemicals (Table 9.1).[15] Ozone in the upper atmosphere is good, as it creates a layer shielding Earth from the sun's radiation. Ozone in the lower atmosphere, near Earth's surface, is not good. Ozone forms through chemical reactions primarily between nitrogen oxides and volatile organic chemicals (like fumes from cars and gasoline) in the presence of sunlight. As a result, ozone pollution is worst on warm, sunny days. In addition to affecting the respiratory system of humans, ozone damages plant tissues and is a potent greenhouse gas contributing to climate change.

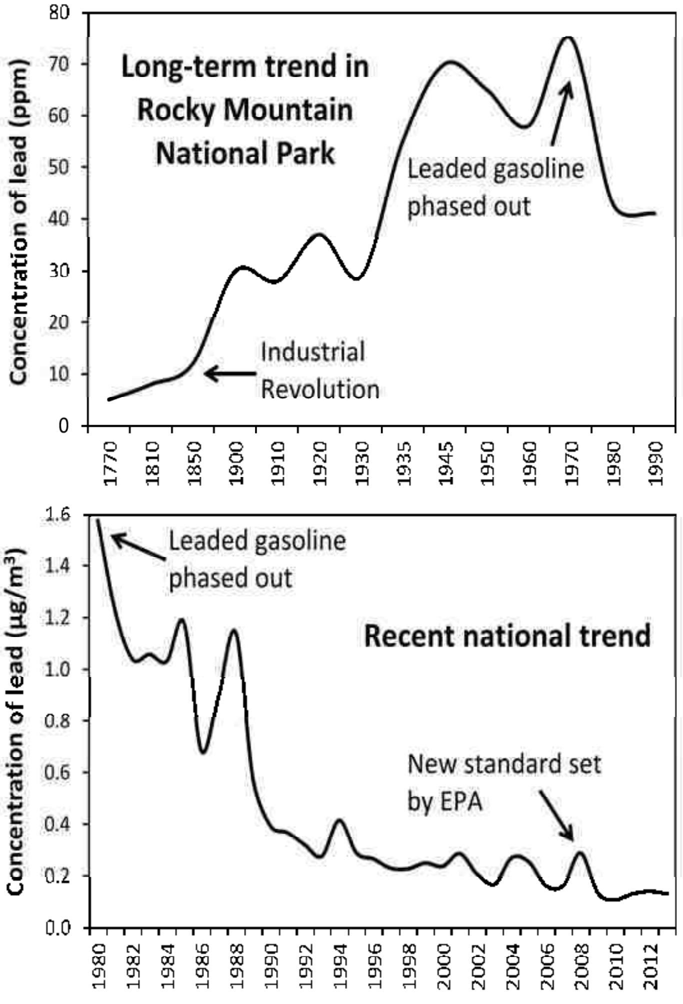


Fig. 9.1. Top: concentration of lead in sediment of Sky Pond, Rocky Mountain National Park, Colorado. Data from Wolfe et al. (2001).[39] Bottom: recent national trend in atmospheric lead. Data from the U.S. Environmental Protection Agency.

Burning fossil fuel is the primary emitter of sulfur and nitrogen oxides, while agriculture emits ammonia. These chemicals react with oxygen and hydrogen in the air to form sulfuric acid, nitric acid, and ammonium – the constituents of acid rain.[9] Land and water are exposed to the acids in a dry form (by the gases remaining in air and touching land and water) and a wet form (in rain, snow, and fog) collectively known as acid rain. In addition to contributing to acid rain, nitrogen pollutants enrich ecosystem nutrient content unnaturally, triggering algal blooms in lakes and various effects on land.[19]

Metals, such as lead, mercury, and cadmium, are worrisome as air pollutants because they accumulate in soil, water, and biota including humans. Sources of lead pollution include mining, metal processing, and manufacturing. Mercury is emitted via combusting coal, incinerating municipal and medical waste, and mining.

These diverse pollutants can interact to worsen air quality. Sulfur reacts with mercury to increase its toxicity, for example.

The United States has curbed several major air pollutants since the 1970s. Passage of the 1970 Clean Air Act was landmark legislation that established national standards for air quality and set limits on allowable concentrations of pollutants in air. A good example is lead. Concentrations of lead in the air rose during the Industrial Revolution in the 1800s, exemplified by lead deposited in lake sediments in Rocky Mountain National Park, Colorado (Fig. 9.1). Lead concentration further increased during the 1920s, with use of cars that burned leaded gasoline to enhance engine performance. By the late 1970s, concentration of lead in the blood of U.S. children was dangerously high. In 1978, the Clean Air Act set the ambient air standard for lead to be 1.5 micrograms of lead in a cubic meter of air. Phasing out leaded gasoline reduced atmospheric lead (Fig. 9.1). Because no level of lead in humans (especially children) is considered safe, in 2008 the U.S. Environmental Protection Agency strengthened the standard to 0.15 micrograms of lead per cubic meter of air.

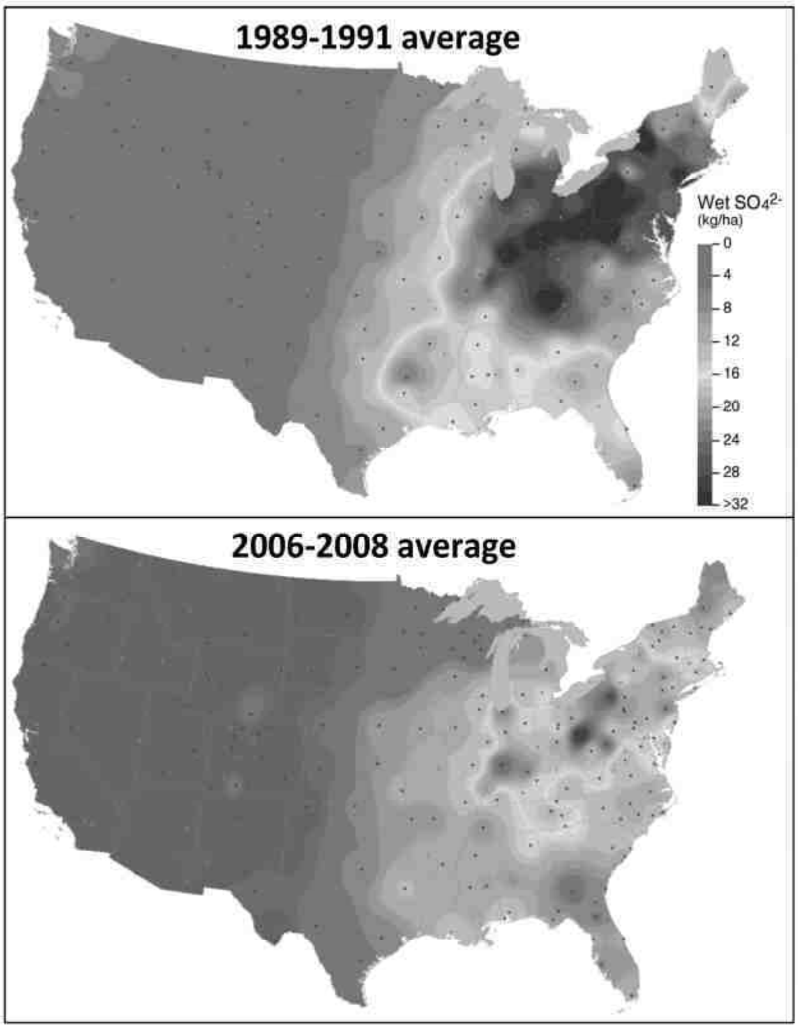


Fig. 9.2. Decreasing deposition of sulfate, a component of acid rain, in the United States. From the U.S. Environmental Protection Agency (2010).[15]

Similarly, acid rain-forming pollutants have decreased. Deposition of sulfur from the air declined 60% in the past two decades (Fig. 9.2). Much of the improvement was from tighter controls on power plant emissions.

Trends in air quality give cause for optimism in the ability to tackle air quality problems, but they also indicate cause for concern. In 2010, the U.S. Environmental Protection Agency noted that 127 million people in the United States live in counties that do not meet air quality standards.[15] Although air quality has improved since the 1970s, this does not necessarily mean that air quality is currently good. Moreover, continued release of pollutants at lower levels still results in accumulation in soil, water, and organisms for toxic chemicals like mercury. Acid rain remains a problem in certain areas, and recovery of streams and lakes from past severe acid rain is frequently slow. Additionally, detection of less-well-known contaminants in relatively remote Alaskan and Sierra Nevadan parks raises questions as to their transport and adverse effects in ecosystems.

This chapter discusses recent trends in air quality across national parks and provides examples of the ecological effects of ozone, acid rain, and metals in air pollution. Lichens are indicators of air quality, and their status in national parks is also discussed. The chapter concludes with emerging issues anticipated to affect air quality in parks.

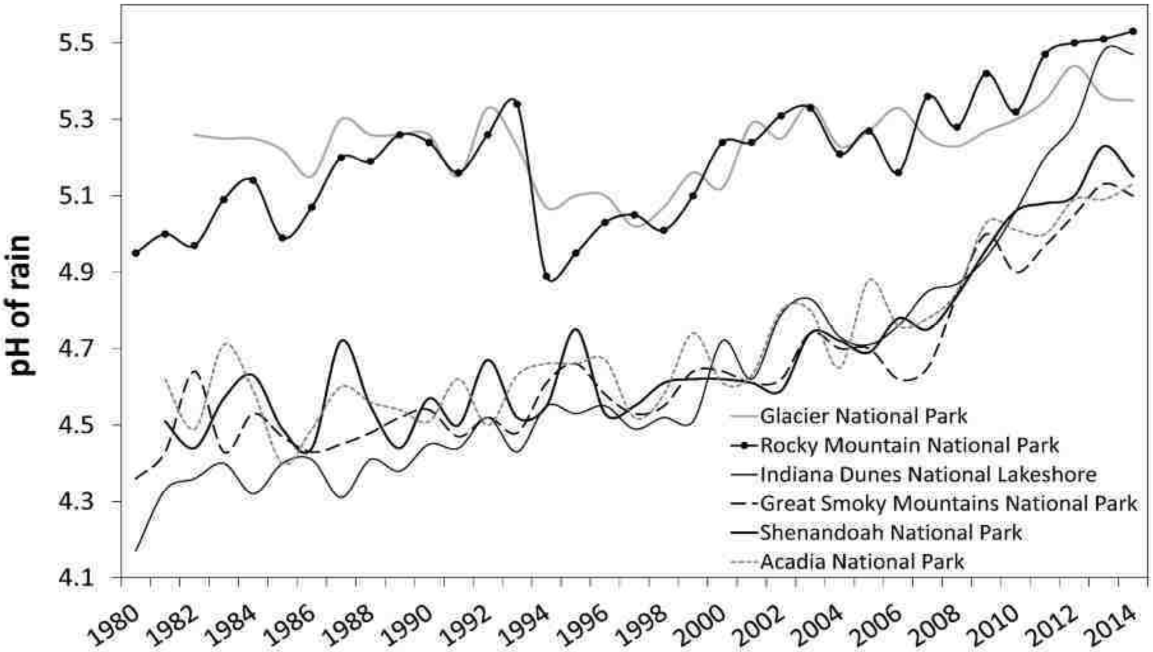


Fig. 9.3. Examples of increasing pH of rain falling in six parks across the country. Increasing pH signifies reduced acid rain. Data from the National Atmospheric Deposition Program.

Trends in Air Quality across Parks

A 2013 assessment by the Air Resources Division of the National Park Service evaluated trends in air quality between 2000 and 2009 in 342 parks.[3] Air quality in parks generally mirrored nation-wide trends. Overall, air quality improved across parks during the nine-year period. There were four general types of trends between 2000 and 2009, including air pollutants that: 1) generally decreased across parks; 2) decreased but that remained at levels damaging to natural features; 3) changed little, or even worsened slightly; and 4) were variable among parks by improving in some and worsening in others.

A notable improvement, consistent with national trends, was a reduction in acid rain, driven by reduced sulfur, and to a certain extent, nitrogen oxide pollution. This is illustrated for six parks with the longest-term records showing rainwater becoming less acidic at monitoring stations between 1980 and 2014 (Fig. 9.3). Despite improvement, acid rain remains at concerning levels in parks like Shenandoah, as discussed later in the chapter.

Ozone is characterized by elements of all four trend types. Ozone concentration in air decreased by 10% in the 2000s nationally. However, ozone did not change or increased in certain parks and is considered still damaging in some parks (Fig. 9.4). Reducing ozone is challenging because it requires curbing several other pollutant contributors to ozone formation (such as methane), and warming temperatures increase ozone production.

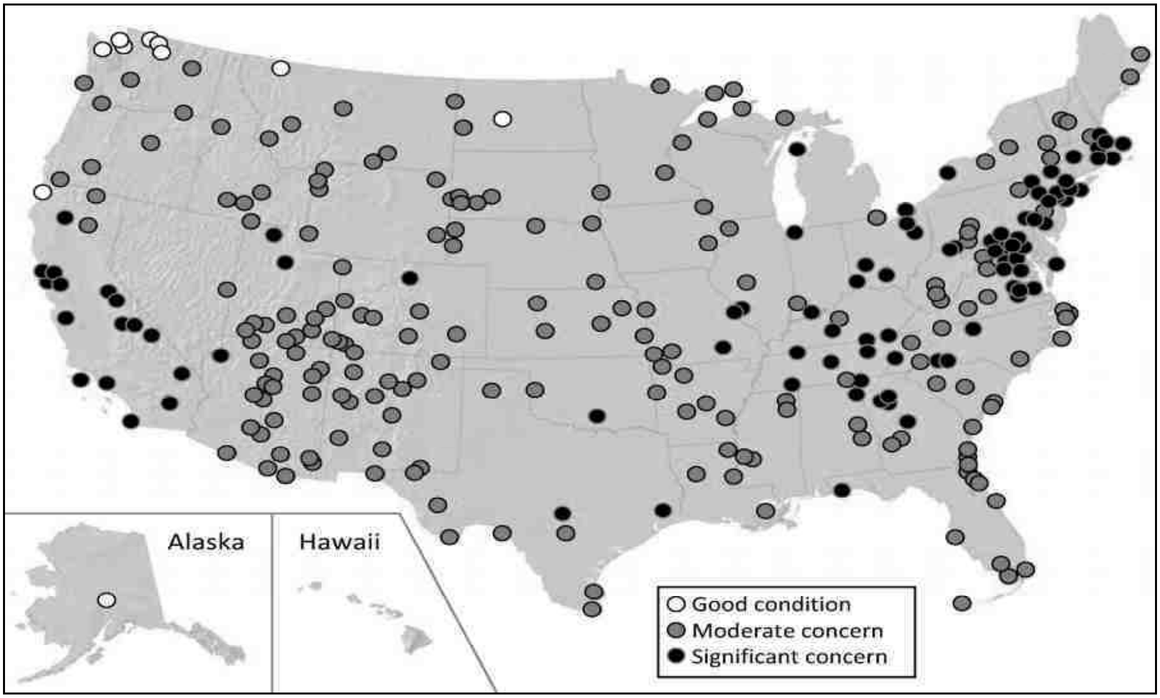


Fig. 9.4. Status of ozone air pollution in national parks in 2009, with each oval representing a park. A park is classified as "significant concern" if ozone in air exceeds the national standard; "moderate concern" if within 20% of the standard; and "good condition" if at least 20% lower than the standard. Parks in good condition were mainly in Alaska and the Northwest. Adapted from Air Resources Division (2013).[3]

Ozone and Forest Growth in Acadia National Park

One effect of ozone in ecosystems is damaging plant tissues, reducing capacity for photosynthesis and potentially plant growth.[13] Sensitive plant species exhibit various symptoms of ozone exposure, such as reddening or blotching of leaves (Fig. 9.5). Other plants may not exhibit symptoms but still be affected.

Fig. 9.5. Top: healthy tulip poplar leaf. Bottom: leaf showing symptoms of exposure to ozone (photos provided by the U.S. Forest Service).

In Acadia National Park, Maine, researchers suspected that foliar damage in eastern white pine trees (*Pinus strobus*) observed in the 1980s resulted from exposure to elevated ozone concentrations. Masses of polluted air from the urban and industrial corridor of the Northeast move through the park, often containing high concentrations of ozone.

A detailed study ensued in 1993, involving collecting cores from 102 white pine trees to measure the width of the annual growth rings of the wood.[6] Tree growth each year was compared to ozone concentrations measured at a monitoring station near the park headquarters. From 1983 to 1992, the concentration of ozone in August and the amount of rainfall accounted for 80% of the variation in tree growth from year to year. In years with high August concentrations of ozone, trees grew slowly (Fig. 9.6). Conversely, trees grew rapidly when ozone concentrations were low. White pine may be most vulnerable to ozone in August, because growth of pine needles is completed and the mature needles are photosynthesizing at maximum capacity. The needles are thus maximally outputting oxygen while taking in carbon dioxide from the air. This makes the wide open needle tissues susceptible to also taking in ozone. As of 2015, the park continues monitoring ambient ozone and remains concerned about effects of ozone on park resources.

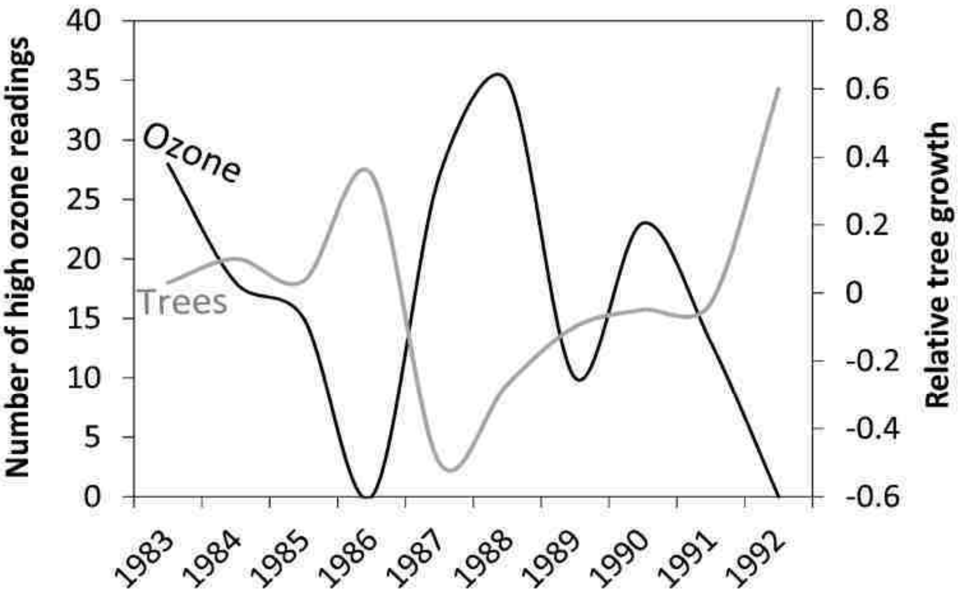


Fig. 9.6. When concentration of ozone in the air in August was high, growth of white pine trees that year was slow in Acadia National Park, Maine. Data from Bartholomay et al. (1997).[6]

Correlations between ozone and damage to foliage or reduced plant growth have also been reported in other parks, such as in the Sierra Nevada Mountains (Yosemite and Sequoia National Parks) downwind from southern California urban areas.[31] In Great Smoky Mountains National Park of North Carolina/Tennessee, certain forest understory plants appear more sensitive to ozone than others.[11] This variation could result from many factors, including the abilities of different plants to produce ascorbic acid and other antioxidants that protect plants from ozone. Evaluating how sensitivity to ozone may interact with the genetic makeup and evolution of species is currently of interest.[26]

Acid Rain and Streams of Shenandoah National Park

Shenandoah National Park has been at the forefront of ecological effects of acid rain since the 1970s.[14,25] In northern Virginia, 120 km (75 miles) west of Washington, D.C., the park receives acid rain produced by upwind sulfur and nitrogen emissions. Acidity is measured on a pH scale approximately from 0 (acidic) to 14 (basic), with 7 being neutral. For reference, vinegar is acidic, with a pH of 2. Baking soda is basic, with a pH of 9. Normal rain has a pH of about 5.6, from dissolving carbon dioxide in the air forming carbonic acid. Typical rains in the 2000s in Shenandoah National Park were more acidic than normal rain. Only slight improvements have occurred over time. The pH of rain falling in the park averaged 4.53 in the 1980s, 4.59 in 1995, and 4.62 in 2005.

Of particular concern are effects of acid rain on the park's streams and fisheries, among the most significant of eastern national parks. Shenandoah is in the central Appalachian Mountains, containing small streams that flow from the mountaintops to the lowlands. The park has 231 headwater streams and over 1,000 km (620 miles) of streams in total (Fig. 9.7). These streams eventually drain into Chesapeake Bay, by feeding major rivers such as the Potomac River flowing through Washington, D.C.

Fig. 9.7. White Oak Run, Shenandoah National Park, Virginia (K.E. Hyer, U.S. Geological Survey).

The type of bedrock a stream flows through affects its susceptibility to acidification. Bedrock rich in bases (such as calcium) can partly neutralize acid rain. Shenandoah contains three main types of bedrock, each of which covers a third of the park.

Bedrocks of silica rocks, such as sandstone and quartzite, have the least ability to neutralize acid rain. Granite bedrock has an intermediate ability. Basalt has a pH over 7 and has the greatest ability to buffer streams from acid rain.

The distribution of bedrock has created variability in the severity of, and recovery from, effects of acid rain in the park's streams. Between 1980 and 2010, a subtle improvement in stream water quality (indicated by increasing pH of the water) has occurred overall, but streams on acid-susceptible silica have not improved.[25] This is perhaps not surprising, because even with pollution controls, air pollution remains greater than naturally occurring levels and impacts can continue cumulatively. Some streams are also naturally more acidic.

In general, the diversity of fish in the park's streams has been unchanged or increased slightly through time. Fish abundance remains related to stream pH, influenced by the ability of different bedrocks to buffer acidity and the acidity tolerances of different fish species. For example, blacknose dace (*Rhinichthys atratulus*) die when stream pH is lower than 6.2 to 5.6. Common shiner (*Luxilus cornutus*) can live in slightly more acidic streams, dying when pH lowers below 6.0 to 5.4. Brook trout (*Salvelinus fontinalis*) have greater tolerance before dying at pH 5.2 to 4.7, but young fish are less tolerant of acidic water than are adults.

Brook trout are conservation-priority species that indicate stream condition. As of 2010, brook trout were least abundant in streams on acid-susceptible silica and have not exhibited consistent trends through time among the bedrock types. In fact, brook trout populations might be declining slightly in certain areas. It is not fully understood what might be limiting brook trout populations, as interactions could occur among aquatic insects (food for the fish), stream temperatures, and forest defoliations by non-native insects that can affect stream chemistry. Stream temperatures have increased by 1 degree Celsius (2 degrees Fahrenheit) since 1980, and gypsy moth defoliations and loss of shade-providing eastern hemlock trees may exacerbate warming temperatures.

While continued curtailing of acid rain since 2005 is encouraging (Fig. 9.3), the acid rain issue has not gone away. Intensive local mitigations, such as liming streams to increase pH, can be difficult or inappropriate to implement. Future changes might be influenced by regional air quality trends and managing forests and streams as connected ecosystems.

Lichens as Indicators of Air Quality

A lichen is a unique "2-for-1" organism consisting of a fungus and green algae or blue-green bacteria.[33] The fungus and algae or bacteria have a symbiotic relationship, where the fungus provides structure and the algae or bacteria photosynthesize. Lichens have many growth forms, and some resemble tiny wildflowers or miniature shrubs. Certain lichens grow upright from the ground, are hair-like hanging off tree bark, or are relatively flat and appear "painted" onto rocks (Fig. 9.8). Lichens are of particular interest in air quality because many are highly sensitive to pollution (especially sulfur) and store elements taken in from the air to indicate air chemistry.[10] In general, upright lichens that look like miniature shrubs (called fruticose lichens) are the most sensitive to air pollution. Thus, presence of these lichens is an indicator of good air quality. Leafy lichens (foliose) have intermediate sensitivity. Flat, crust-like lichens (crustose) are the least sensitive to pollution. Highly polluted areas typically only contain crustose lichens, if any lichens are present at all.

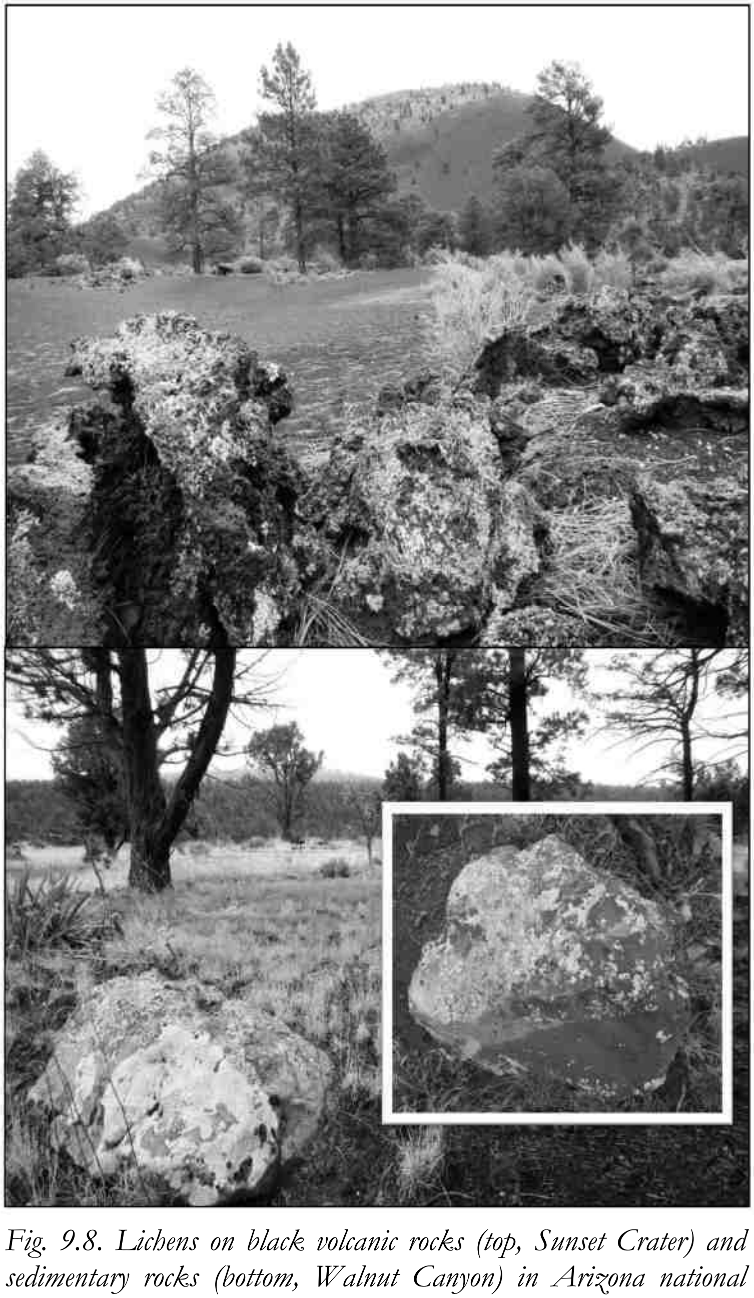


Fig. 9.8. Lichens on black volcanic rocks (top, Sunset Crater) and sedimentary rocks (bottom, Walnut Canyon) in Arizona national monuments. Photos by S.R. Abella.

The 7,700-hectare (19,000-acre) Cuyahoga Valley National Park, established in 1975 between Cleveland and Akron, Ohio, is representative of several parks where long-term declines in lichens were reported between the early 1900s and the peak of sulfur pollution in the 1970s-1980s.[37] The park contains deciduous forests on rolling hills around Tinkers Creek and the Cuyahoga River. An early lichenologist recorded 151 lichen species as present in the area between 1895 and 1917. In 1985, the area was re-inventoried, including over 30 sites in the park. The 1985 inventory found only 31 of the earlier 151 species. No lichens most sensitive to sulfur pollution were found in 1985. Air quality monitoring indicated that concentrations of sulfur were above known lethal levels for most lichens. Chemical analysis further found high levels of sulfur and lead within remaining lichens near Furnace Run Creek and O'Neil Woods in the park. Air pollution may have destroyed 80% of the park's lichen flora.

What may have happened when air quality began improving? Between 1975 and 1995, emissions of sulfur dioxide in Ohio were halved. In southeastern Ohio, periodic inventories between 1973 and 1996 found almost no lichens during the 1970s and then steady increases during the 1980s and 1990s, as sulfur pollution decreased (Fig. 9.9). One sulfur-sensitive species, greenshield lichen (*Flavoparmelia caperata*), was absent in 1973 but inhabited 96% of inventory sites in 1996. Repeating the lichen inventories specifically in Cuyahoga Valley and similar parks may reveal whether recoveries of lichen communities have occurred in the parks too. It should also be kept in mind that today's pollution levels still exceed pre-industrial levels, so certain sensitive organisms might remain absent.

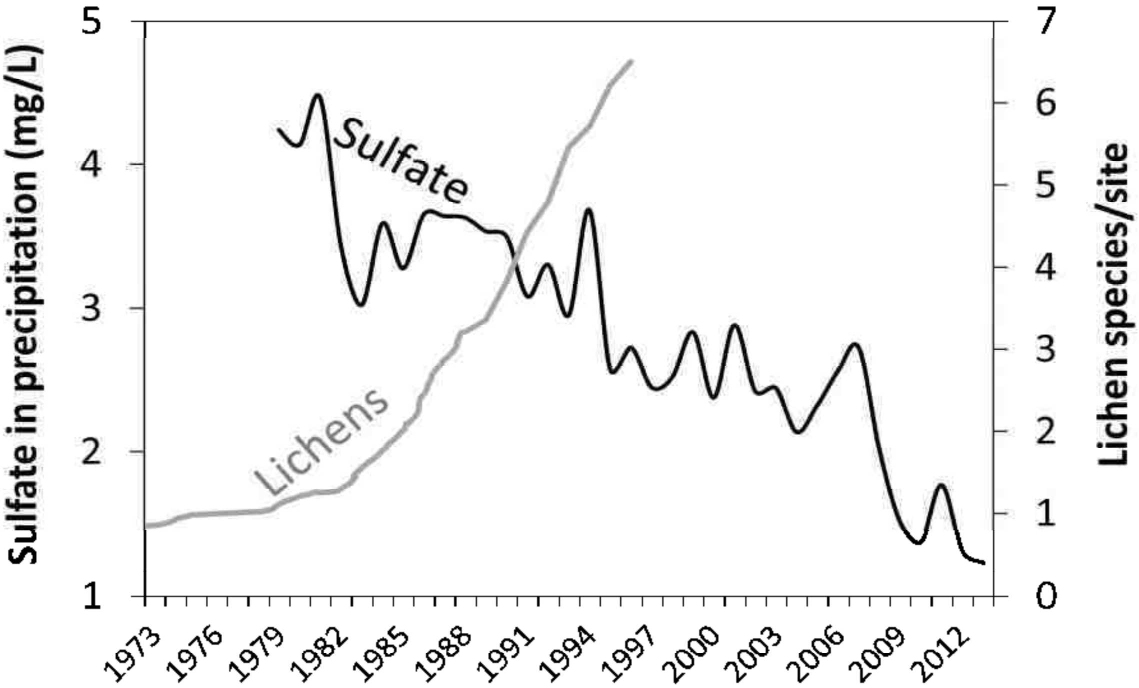


Fig. 9.9. Recovery of lichens with decreasing sulfur air pollution, Ohio. Lichen data from Showman (1997)[34] and available through 1996. Sulfate data (in milligrams per liter) from the National Atmospheric Deposition Program (Noble, Ohio station).

A recent compilation revealed the diversity of lichens found in national parks.[7] As of 2010, over 2,600 species of lichens were reported from 153 parks. That is a lot of species, but many more lichen species probably exist in the full 408 parks and new lichen species (including those previously unknown to science) are regularly documented. An individual park averages 100 species of lichens, with some parks with fewer species and other parks with many more. Parks especially rich in lichens are generally in cold, northern regions.

The park with the current distinction of having the most recorded lichen species is Klondike Gold Rush National Historical Park, with 674 species.[36] Near Skagway, Alaska, north of Juneau, the park is not particularly large at 1,384 hectares (3,420 acres). It was designated to conserve historical trails (Chilkoot and White Pass Trails) leading to goldfields near Dawson City, in Canada's Yukon Territory (Fig. 9.10). Goldseekers used these trails during the 1897-1898 Klondike Gold Rush. Ironically, few of the 100,000 goldseekers made money, and the park's "gold mine" of lichens proved more sustainable.

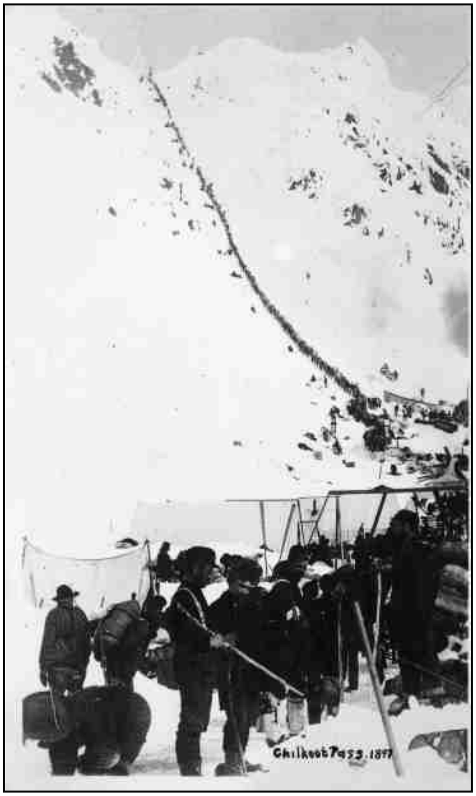


Fig. 9.10. Left: Chilkoot Pass, with a line of people hauling gear up the slope in 1897 during the Klondike Gold Rush (photo courtesy of the National Park Service). Right: view of the contemporary Skagway River and route of the historic White Pass Trail, north of Skagway, Alaska. Nearby Klondike Gold Rush National Historical Park has the most lichen species of any park (photo by R.J. and S.R. Abella).

In addition to being useful barometers of air quality, lichens provide important functions within ecosystems. Disappearance of lichens due to poor air quality can have ecological effects. Lichens are eaten by numerous animals, are used by birds for building nests, convert nitrogen in air to forms in soil available to plants, contribute to decomposition of rocks to form soil, and stabilize soil to limit erosion.[28] Recolonization by lichens during improving air quality could have ecological influences within parks.

Mercury in Great Lakes Fish and Birds

Mercury is responsible for 80% of the fish consumption advisories for humans in the United States.[18] It is highly toxic to humans and wildlife.[5] While mercury is a naturally occurring element, much of its release into the environment is now associated with human activities like burning coal, mining, smelting, and industrial processes. Deposition from the atmosphere is a main way that mercury enters lakes and rivers, where it accumulates in sediment, fish, and other aquatic organisms. Mercury affects wildlife by impairing their performance, such as compromising foraging ability or reproduction, at low but dangerous levels of mercury. Death occurs at high doses. Mercury "biomagnifies," as its concentration increases while being "passed on" up the food web. For example, mercury can biomagnify in birds that eat fish already containing elevated levels of mercury.

Mercury has displayed an inconsistent trend in the environment in recent decades. An example is the Great Lakes region. One assessment analyzed mercury in fillets of 5,807 fish of lake trout (*Salvelinus namaycush*) and walleye (*Sander vitreus*) caught in the Great Lakes from the 1970s to 2007.[8] Concentrations of mercury in lake trout generally declined. Mercury in walleye decreased in Lakes Superior and Huron, but increased slightly in Lakes Erie and Ontario. In Lake Erie, mercury in walleye decreased from the 1970s to the 1980s, but then trended upward after the 1990s back toward 1970s levels. It is not fully understood if these trends relate to new pollution or to other recent factors affecting the Great Lakes. One factor is the invasion of non-native mussels, which alter food webs and may affect mercury accumulation in fish.[18]

Mercury also is a concern in smaller, inland lakes in the Great Lakes region. In Minnesota, mercury in fish from 845 lakes decreased from the early 1980s to the mid-1990s, but increased thereafter.[29] Alarmingly, northern pike (*Esox lucius*) from some lakes in Voyageurs National Park in northern Minnesota contained the highest concentrations of mercury reported in the state.[38] Similarly, mercury concentrations in the park's bald eagles (*Haliaeetus leucocephalus*) were decreasing by the late 1990s but have recently been increasing.[32] Mercury concentrations as of 2010 were not deemed high enough to adversely affect the bald eagles, but further increases would be worrisome.

Concern with mercury deposition is not just isolated to the Great Lakes region. In the Northeast, for example, a huge analysis of 4,000 birds of 38 species found that every single bird contained at least traces of mercury.[17] Common loons (*Gavia immer*), which eat fish and thus would be expected to bioaccumulate mercury, had among the highest concentrations of mercury in their tissues. National parks were not immune to elevated levels of mercury in birds. The concentrations of mercury in bird feathers and food being fed to hatchlings by adult tree swallows (*Tachycineta bicolor*) were 1.5 times greater at Aunt Betty Pond in Acadia National Park, Maine, than they were at the toxic dump of a superfund site in Massachusetts used for comparison.[27] These types of high mercury concentrations also raise questions about whether mercury can be introduced to food webs in different areas during migrations and movements of contaminated animals.

Both the U.S. and Canadian governments have acted to reduce mercury emissions. However, continued emission of some mercury to perpetuate its accumulation, cycling of already deposited mercury from past air and water pollution, and global transport of mercury released in other countries complicate efforts to reduce mercury.

Fig. 9.11. Lake Irene, Rocky Mountain National Park, Colorado, typifying the types of high-elevation lakes receiving pollutants from distant sources. Photo by S.R. Abella.

Emerging Issues in Air Quality

Several issues are likely to influence future air quality of parks. First, park air quality will probably continue to be linked with regional, national, and world-wide trends in air quality. A good example is Joshua Tree National Park, in the Mojave Desert downwind of urbanized southern California. A gradient of nitrogen deposition occurs across the park with increasing distance from pollution sources. This has raised questions regarding whether the pollution increases nitrogen-loving, non-native plants at the expense of native species, and whether pollution-related changes in soil communities occur in different areas of the park.[4]

Second, legacy effects of past pollution warrant consideration even if better pollution controls have curbed current emissions. A good example is lead. While regulations have drastically cut emissions since the 1970s, lead accumulated in soil and lakes during centuries of elevated emissions. Lead concentrations in forage plants eaten by mule deer (*Odocoileus hemionus*) along roadsides were 40 times greater than in roadless areas in Rocky Mountain National Park, Colorado.[23] This probably resulted from past use of leaded gasoline in cars.

Third, we do not fully understand the ecology of the bewildering array of chemicals introduced to the environment, nor the "byproduct" chemicals produced by their breakdown or combination with other chemicals. Cocktails of pesticides, polychlorinated biphenyls (PCBs), and other chemicals have been detected in relatively remote national parks such as Denali (Alaska), Mount Rainier (Washington), and Sequoia (California).[1,20,22] While concentrations in the remote parks are typically low, they are not always low, and raise concern about future accumulation of the chemicals (Fig. 9.11).

Fourth, conservation of priority natural features in parks may be at least partly related to air quality. Amphibians (frogs, toads, and salamanders) and reptiles (lizards, snakes, crocodiles, turtles, and tortoises), are examples (Fig. 9.12). There is global concern about declines in amphibians and reptiles, including in the United States.[21] Between 2002 and 2011, amphibian populations declined by 4% annually at 34 U.S. monitoring sites, several of which were in national parks.[2] Reasons for the declines continue to be actively studied, and air pollution is among several factors often proposed.

Fig. 9.12. Short-horned lizard (Phrynosoma hernandesi), an example of reptiles of conservation concern. Photo by S.R. Abella.

Fifth, an emerging concern is energy developments already around or pervasively proposed to be around national parks. Natural gas extraction, or "fracking," has expanded, creating a massive infrastructure of extraction facilities, roads, pipelines, and vehicles.[12,16] Disagreement surrounds the amount of air pollution created by the extraction facilities. However, the facilities and infrastructure (including vehicles) do represent new potential sources of air pollution around many national parks.[12,30]

Similarly, industrial solar energy collecting facilities are extensively proposed around desert national parks. While it may seem that solar facilities should not emit air pollution, building and maintaining the facilities requires burning fossil fuel and the facilities themselves have undocumented relationships with air quality. Extensively bulldozing desert soil to build the industrial sites can release hazardous dust and pollutants.[24]

The United States has shown the ability to reduce certain air pollutants through national standards and local action, creating cause for optimism for improving air quality in parks. However, legacy effects, continued accumulation of pollutants, and trends for worsening in certain measures of air quality underscore that air pollution is not just a problem of the past.

10 COASTS

Coasts are critical features ecologically and for human societies. The United States has 142,640 km (88,633 miles) of ocean coastline, according to the National Oceanic and Atmospheric Administration. Alaska has the most coastline with 54,563 km (33,904 miles), followed by Florida, Louisiana, Maine, California, and 19 states with the remainder. Coastal counties in the lower 48 states comprise only 9% of the land area.[21] Yet, they contain 40% of the U.S. population and 56% of the income. These observations, combined with the importance of coasts for fisheries and protecting inland areas from storms, underscore that conserving coasts is economically strategic in addition to ecologically beneficial. Unfortunately, coasts and their associated ecosystems, such as wetlands and coral reefs, are damaged and continually threatened by human activities.[2]

Parks of diverse designations conserve coasts. National parks like Olympic, Redwood, Channel Islands, and Hawaii Volcanoes include Pacific Ocean coast. Virgin Islands, Dry Tortugas, Biscayne, Everglades, and Acadia include Atlantic Ocean coast. Other types of parks with coasts are national recreation areas (such as Santa Monica Mountains along the Pacific and Gateway along the Atlantic), national preserves (such as Glacier Bay in the Gulf of Alaska), monuments (like Buck Island Reef in the Virgin Islands), and several historical parks. Parks specifically designated as national seashores are Point Reyes (California), Padre Island (Texas), Gulf Islands (Mississippi/Florida), Canaveral (Florida), Cumberland Island (Georgia), Cape Lookout and Cape Hatteras (North Carolina), Assateague Island (Maryland/Virginia), Fire Island (New York), and Cape Cod (Massachusetts).

Like land-based parks, coastal parks are subject to external influences that challenge conservation within parks. Examples are off-shore alterations for storm protection, sea level rise, overfishing or inadvertent take of non-target species, oil spills, and climate change, all beyond the control of local park managers. National parks are at the forefront of these coastal conservation issues, which are challenges globally. This chapter provides examples of interactions of parks with coastal storms, rising sea levels, fisheries, and conservation of iconic ecosystems such as kelp forests, coral reefs, and their associated sea animals.

Hurricane Katrina Hits Jean Lafitte National Historical Park

With over $100 billion in damage, the 2005 Hurricane Katrina was the most costly hurricane in U.S. history. Katrina affected 233,000 km^2 (90,000 square miles), an area larger than Great Britain.[25] It had a major impact on human well-being still evident today. As summarized by the National Oceanic and Atmospheric Administration, Hurricane Katrina originated near the Bahamas in the Atlantic Ocean. Katrina hit southern Florida on August 25, 2005, with maximum sustained winds of 130 km/hour (80 miles/hour). During the next few days, the hurricane moved across the Everglades and re-intensified over the Gulf of Mexico. Hurricane-force winds extended 145 km (90 miles) from the eye of the storm. On

August 29, Katrina again hit land, near Buras, Louisiana, with maximum sustained winds of 200 km/hour (127 miles/hour). In addition to the winds, Katrina created a storm surge 3 to 9 meters (10 to 30 feet) high that pushed ocean water onto land. The storm surge extended inland as far as elevation of the land allowed. Eastern New Orleans experienced a surge of 6 meters (19 feet), and western Lake Pontchartrain 2 to 3 meters (5 to 10 feet). The storm surge and heavy rainfall resulted in severe flooding. Katrina continued northeastward across the United States before finally dissipating as a low-intensity storm over Ohio on August 31.

Understanding effects of hurricanes on coastal ecosystems is important ecologically and for human interactions with hurricanes, because coastal ecosystems buffer inland areas. Upon making landfall on the Louisiana coast, Katrina hit Jean Lafitte National Historical Park and Preserve, named for the mercurial French-born Jean Lafitte. Sometimes considered a pirate, Lafitte and his Baratarian employees purportedly operated a land and sea smuggling ring, extending from the Gulf of Mexico through southern Louisiana's swamps to New Orleans. In the War of 1812 between the United States and Britain, Lafitte's Baratarians – well-armed with cannons and ammunition of questionable origin – helped the U.S. military win the Battle of New Orleans. As a result, President James Madison offered pardons to the Baratarians for any crimes committed. Today, the 6,780-hectare (16,750-acre) park is in southern Louisiana, between New Orleans and the Gulf of Mexico.

Fig. 10.1. Baldcypress forest near the Bayour Coquille Trail, Jean Lafitte National Historical Park and Preserve, Louisiana (National Park Service photo).

Hurricane Katrina passed through the park on August 29, with winds of at least 112 km/hour (70 miles/hour). Damage to the park's baldcypress (*Taxodium distichum*) forest was assessed in 2006, one year after Hurricane Katrina (Fig. 10.1).[15] Although the forest sustained some damage from the hurricane, the portion of dead wood in the forest (7%) was still minimal compared to the portion of live wood (93%). The amount of ground covered by live tree canopies (83%) also remained high even after the hurricane. Many individual shrubs of wax myrtle (*Morella cerifera*) and blueberry (*Vaccinium* species) were dead after the hurricane, by being pushed or pulled over by storm winds or uprooted by flooding.

In general, however, the park's baldcypress forest fared better than buildings constructed by humans. The buttressed architecture of the lower trunks and large root systems of baldcypress anchor the trees in the ground to withstand hurricanes. Additionally, baldcypress leaves fall off quickly during storms, reducing wind friction and potentially protecting trees from wind damage. Despite Hurricane Katrina striking the park fairly directly, its baldcypress coastal swamp forest was remarkably little affected.

Hurricane Katrina stimulated re-evaluations of the importance of coastal ecosystems for tempering effects of storms. Economists evaluated 34 major hurricanes that struck the U.S. coastline since 1980 and compared economic damage with the amount of coastal wetland available to buffer the coast.[5] For each loss of 1 hectare (2.5 acres) of wetland, damage to human infrastructure increased by $33,000. Losing 100 hectares (250 acres) of wetland increased damage by $3 million. The economists estimated that U.S. coastal wetlands provided $230 billion worth of storm protection services each decade.

During Hurricane Katrina, earthen levees unprotected by wetlands were destroyed, but levees fronted by extensive wetlands escaped substantial damage.[7] Forested wetlands, such as at Jean Lafitte, diminish penetration of wind inland, reducing wave action and storm surge. Also, the shallow water of wetlands generates friction against the underlying land, dissipating wave energy. Combining structural engineering with conservation and restoration of natural coastal ecosystems is likely to best protect coastlines.

Rising Seas, New York City, and Gateway National Recreation Area

The Atlantic Ocean has been rising since the end of the last major ice age 11,000 years ago. Recent acceleration of sea level rise is consistent with ocean warming related to climate change.[8] Regardless of the cause, rising seas did not previously coincide with massive human infrastructure along coastlines as they do today.

With 840 km (520 miles) of coastline, New York City in the northeastern United States illustrates challenges that rising oceans pose. Average sea level along New York's coast is projected to rise 0.2 to 0.6 meters (1 to 2 feet) above a year 2000 baseline by 2050.[8] By 2080, sea levels may rise 0.3 to 1.1 meters (1.5 to 3.5 feet). In addition to inundating land lying below the expanded ocean, frequency of severe storms may increase. Severe storms that occur only once every 100 years under present conditions could occur every 19 to 68 years by 2050 and every 4 to 60 years by 2080.

Sea level rise and severe storms can have effects that vary from place to place among coastlines with different characteristics. For example, coasts receiving sediment from inland waterways to replenish beaches may respond differently to sea level rise than a beach actively eroding and thus more susceptible to inundation. To artificially replenish sand on beaches where humans had disrupted natural sand replenishment, New York City spent $500 million from the 1920s to the 1990s. With continued sea level rise, sand replenishment would need to increase above today's already high cost, making it practical only for a few high-priority areas. Combining several strategies to adapt to rising sea levels has become a priority.

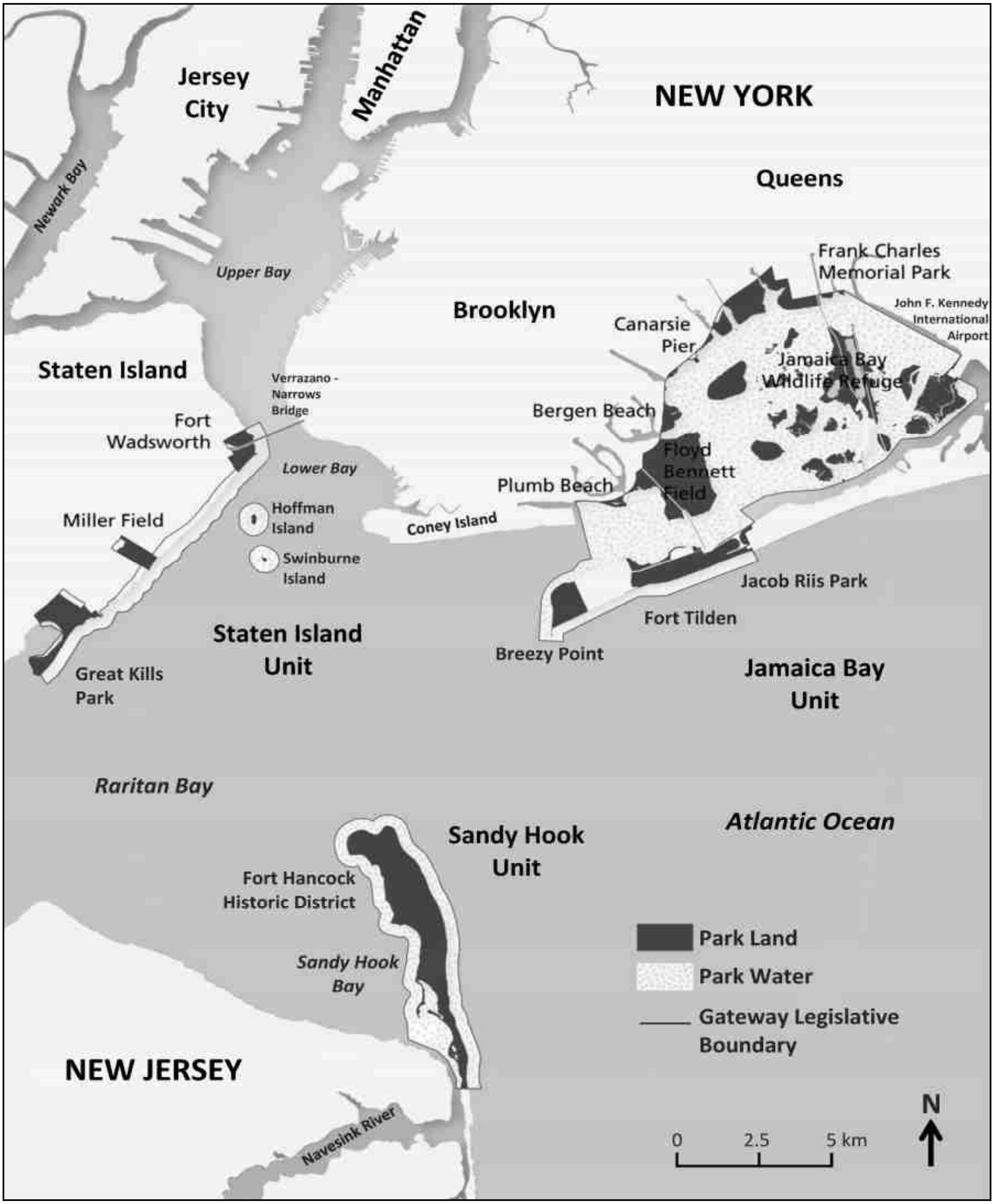


Fig. 10.2. Location of Jamaica Bay and other units of Gateway National Recreation Area, New York. Adapted from a National Park Service map.

Fig. 10.3. Coastal marsh and osprey nest along the New York City coastline, Jamaica Bay, Gateway National Recreation Area (National Park Service photo).

One strategy around New York City is restoring coastal marshes to help buffer coastlines, including in Gateway National Recreation Area.[27] Gateway's 8,160 hectares (20,164 acres) are in several parcels along the New York and New Jersey coasts (Fig. 10.2). Not only does Gateway provide recreation opportunities for its 6 million visitors annually (the seventh most visited national park in 2014), it is a key area protecting the most populated U.S. coastline (Fig. 10.3).

One of the park's units, Jamaica Bay, is on the southwestern end of Long Island, New York, near John F. Kennedy International Airport. Jamaica Bay was historically known for its marsh islands, tidal creeks, mudflats, and abundant shellfish and birds.[10] The bay's coastal marshes have long been manipulated by humans and degrading. Loss of marshes is attributed to a combination of coastline erosion and inundation by rising seas. Channelizing streams emptying into the bay curtailed sediment that historically maintained the bay's sand bars and beaches. Around 1900, developers dredged a channel into the bay, in an effort to create a commercial port. The port did not materialize, but the depression created by the dredged channel continued trapping the little sediment reaching the bay.

Concurrently with reduced sediment reaching the bay, sea levels have risen, to further accelerate erosion and inundate low-lying islands. At Sandy Hook, a peninsula across from the bay, sea levels have risen 0.4 meters (1.3 feet) since 1934 (Fig. 10.4). This may not seem like much, but when islands and beaches are only slightly above sea level, a small rise of the ocean readily inundates them.

Between 1924 and 1999, Jamaica Bay's salt marshes shrank to half their former size as low-lying land converted to open water or barren mud flats. In 1924, 1,004 hectares (2,480 acres) of marsh occupied the bay.[10] By 1999, only 495 hectares (1,220 acres) remained.

In 2006, a multi-agency effort, funded by mitigation requirements of the New York Harbor Deepening Project, began restoring salt marsh.[20] An initial restoration site was a 16-hectare (39-acre) area at Elders Point, in the northern part of Jamaica Bay near Brooklyn and Queens. Restoration involved transporting new sand to areas historically supporting salt marsh but that had been inundated through erosion and rising seas. Then, a characteristic grass of Atlantic Ocean salt marshes was planted – smooth cordgrass (*Spartina alterniflora*). By 2009, three years after restoration began, cordgrass covered 80% of the site.

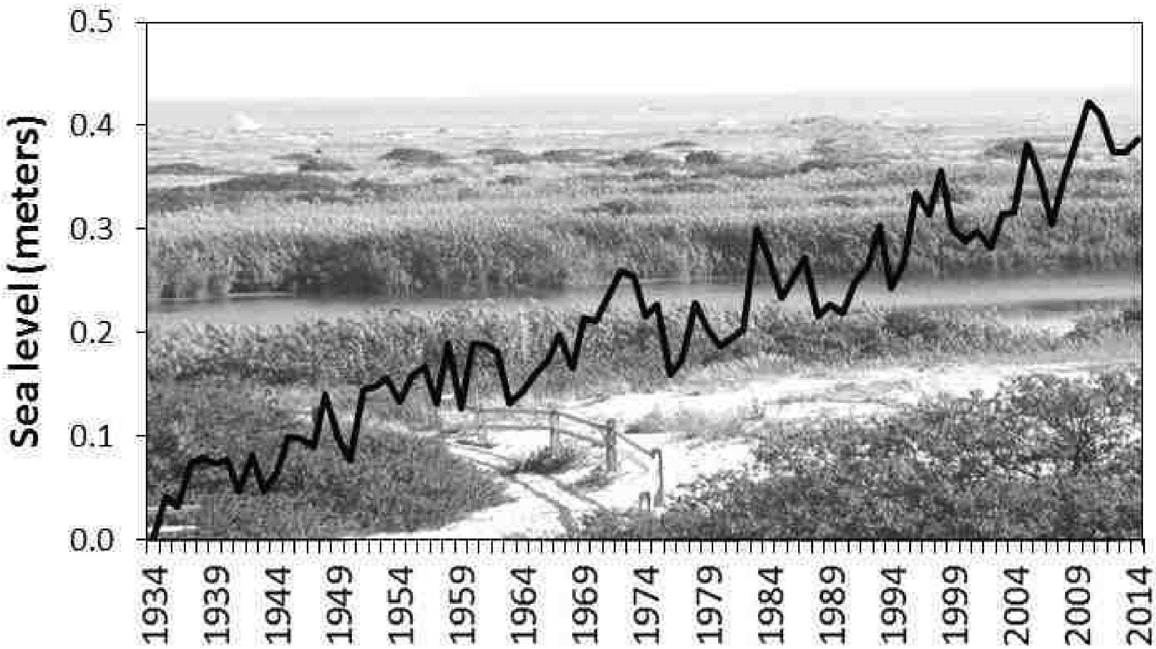


Fig. 10.4. Graph shows rising sea level, relative to a 1934 baseline, off Sandy Hook monitored by the National Oceanic and Atmospheric Administration. National Park Service photo shows coastal marsh on Sandy Hook, with the Atlantic Ocean in the background, Gateway National Recreation Area, New York.

It is not yet clear whether the restored salt marsh islands can become self-sustaining, given inadequate sediment supply and sea level rise. Sand bars and beaches are a balance between erosion and deposition of sand, with a net loss of sand resulting in erosion of beaches. Rising sea levels may necessitate continued inputs of sand, similar to how humans have maintained some New York beaches for a century. Restoring marshes may block sea water and reduce energy of waves, thereby buffering the coast from rising seas and storms.

In addition to benefiting humans, another species anticipated to benefit from Gateway's coastal restoration is the horseshoe crab (*Limulus polyphemus*). This species is one of Earth's oldest creatures, originating over 200 million years ago and existing during the time of dinosaurs to the present. In addition to horseshoe crabs influencing coastline ecology and the fishing industry, their blood is used by the medical industry to test for presence of bacterial toxins in humans. The crab is shaped like a helmet, and has gills capable of breathing both on land and in the ocean. It has 12 sight sensors on its shell, which provide different capabilities for detecting light and seeing. Horseshoe crabs can live 20 years, and grow new shells (molting) up to 17 times by age 10. Adult females are 0.5 meters (2 feet) long and weigh 5 kilograms (10 pounds).[29] Males are only a third of that size. Female horseshoe crabs ascend beaches at high tide during the May to July spawning season. Females dig nests on the sandy beaches and can lay 90,000 eggs. Thousands of eggs are eaten by birds and other animals, with some undiscovered eggs hatching within two weeks. Egg-laying is limited by availability of suitable sandy substrates (Fig. 10.5).[3] These substrates increase through restoration of Gateway's beaches and coastal marsh islands.

Fig. 10.5. Horseshoe crabs spawning on Plumb Beach, Gateway National Recreation Area (National Park Service photo).

Kelp Forests of Channel Islands National Park

Kelp are brown algae of shallow waters near coastlines in cold oceans. Kelp form underwater forests. In 1834, naturalist Charles Darwin remarked: "I can only compare these great aquatic forests…with the [land forests] in the tropical regions. Yet if in any country a forest was destroyed, I do not believe nearly so many species of animals would perish as would here, from the destruction of the kelp. Amidst the leaves of this plant numerous species of fish live, which nowhere else could find food or shelter; with their destruction the many cormorants and other fishing birds, the otters, seals, and porpoise, would soon perish also."[26]

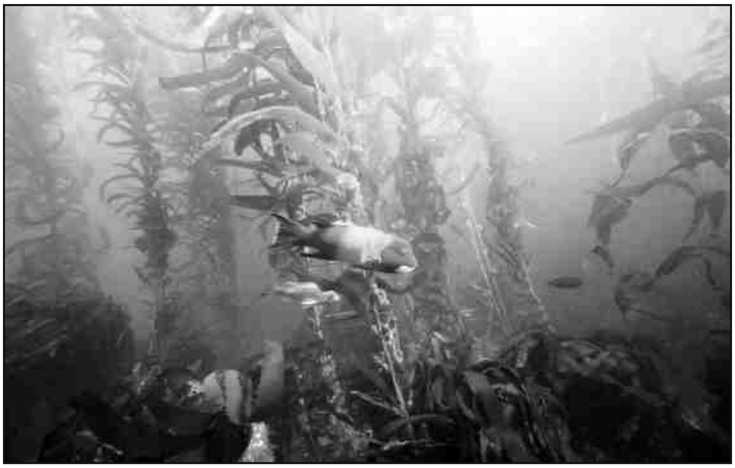


Fig. 10.6. Underwater kelp forest with sheephead fish, near Santa Cruz Island, Channel Islands National Park, California (R. Schwemmer, National Oceanic and Atmospheric Administration).

In the Pacific Ocean, along the California coast, the giant kelp (*Macrocystis pyrifera*) is the dominant kelp (Fig. 10.6). This species grows 45 meters (150 feet) tall, which is about the maximum ocean depth at which kelp occur. Remarkably, kelp can grow 15 meters (50 feet) in a year. Kelp do not have roots, but rather root-like holdfasts that anchor them to the ocean floor. Kelp canopies reduce light and provide protection, creating different habitats that promote species diversity. A variety of sea animals also eat kelp. Despite their large size, kelp forests can be ephemeral. Extensive kelp deforestation occurs in years with abundant herbivores eating kelp, warm ocean temperatures, and an absence of upwelling of nutrient-rich waters.

Kelp forests in Channel Islands National Park off the California coast, west of Los Angeles, provide a dynamic ocean version of a food web described for land habitats in Chapter 4. At the time of early European contact (1540s to 1770s), the kelp-laden coastlines of the Channel Islands supported one of the highest concentrations of Native American hunter-gatherer societies in human history.[4] Maritime Native Americans had colonized the Channel Islands by 12,000 years ago and had seaworthy boats and the first hook and line fishery in the Americas. These early inhabitants harvested fish from kelp forests and hunted sea otters (*Enhydra lutris*), a top predator in the coastal waters (Fig. 10.7). Intensified hunting of sea otters by the Euro-American fur trade decimated them by the early 1800s. They were eliminated from the Channel Islands by the late 1800s. Otters had been a main predator of sea urchins (such as *Strongylocentrotus franciscanus* and *Lytechinus anamesus*), which eat kelp forests. In the absence of otters, spiny lobsters (*Panulirus interruptus*) and the carnivorous fish sheephead (*Semicossyphus pulcher*) may have "picked up the slack" of predating upon sea urchins. Abalone (*Haliotis* species), sea snails that feed on algae, compete with sea urchins for habitat and food, and may also have kept sea urchin populations in check. However, through the 1800s and 1900s, several "boom and bust" cycles of intensive commercial fishing for different species transpired. When one species, such as the spiny lobster, was intensively harvested, its population would crash and often remain low. Then, fishing would target other species, which subsequently had population crashes. Assisted by the removal of predators via commercial fishing, sea urchin populations expanded in the 1950s and 1960s, heavily grazing kelp and causing extensive deforestation. Commercial fishing of sea urchins began in the 1970s, alleviating grazing and allowing kelp recovery in some areas.

Fig. 10.7. Southern sea otter (U.S. Fish and Wildlife Service).

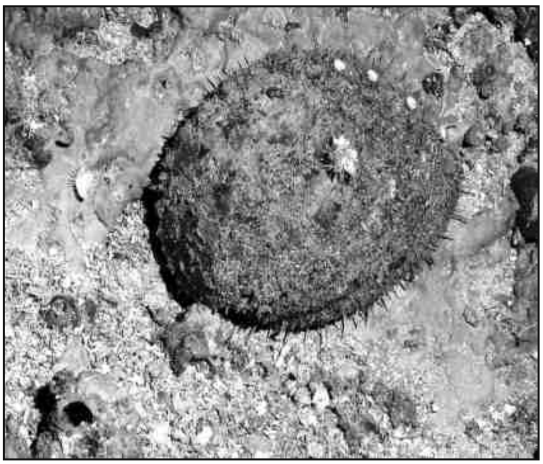


Fig. 10.8. White abalone (National Oceanic and Atmospheric Administration).

Because of these changes, the current food web of kelp forests is unnaturally simplified. Key predators are missing and the species composition of herbivores is altered. This simplification has likely increased the susceptibility of kelp forests to major shifts in condition under changing environments, such as climate change. This illustrates a principle that ecological "redundancy" can help an ecosystem continue functioning when one or a few species are removed if some similar species remain.[28] But with further removal of species, redundancy is lost and ecological collapse (such as the collapse of fisheries) can ensue. This is one of many reasons why eliminating native species from ecosystems is often undesirable.

Declines in several species of the kelp forest have triggered their listing under the U.S. Endangered Species Act. One endangered species is white abalone (*Haliotis sorenseni*), which became the first marine invertebrate listed by the Act. White abalone are single-shelled, stationary snails (Fig. 10.8). They inhabit rocky reefs with kelp and eat drifting algae.[6] This, and the seven other abalone species of western North American coastal waters, once supported a commercial and recreational fishery.

Around Channel Islands National Park, there were 2,000 to 10,000 white abalones in a hectare of ocean area in the early 1970s. Commercial fishing began targeting white abalone, and populations crashed. By the 1990s, the white abalone density had plummeted to only one per hectare. The commercial fishery for white abalone remained open until 1996, even though virtually none were left to catch after the 1970s. Conserving this species in the wild is challenging because population densities are so low that natural reproduction is infrequent and genetic structure is likely disrupted. Additionally, a disease continued removing individuals. This unfortunate example illustrates that national parks are not immune from catastrophic crashes in species populations.

Coral Reefs

Coral reefs are the largest biological structures in the sea. A coral reef is a ridge or mound built up over thousands of years from limestone (calcium carbonate) deposited from skeletons of coral animals.[13] Earth's oceans contain over 800 species of corals. The term coral reef often refers to the entire ecosystem: the coral; the limestone substrate built by the coral; and other organisms, such as fish, that live around the reef. Coral reefs gradually increase in size through the accumulation of coral skeletal debris and the growth of living coral tissue. Reefs increase in height by 9 to 15 meters (30 to 50 feet) in a thousand years.

Coral reefs occur in warm, well-lit (thus relatively shallow) waters at depths typically within 70 meters (230 feet) of the ocean surface. Reefs are generally in tropical waters within 30 degrees latitude of either side of the equator (Fig. 10.9). Ocean temperatures become too cold for coral at distances greater than this from the equator. An estimated 284,000 km^2 (110,000 square miles) of coral reefs occur in the world's oceans.[24] This is only 1% of the area of the ocean, making reefs a rare habitat. Despite the small area occupied by reefs, over 4,000 species of fish are associated with them.[24] This is almost a quarter of all known fish species in the world's oceans.

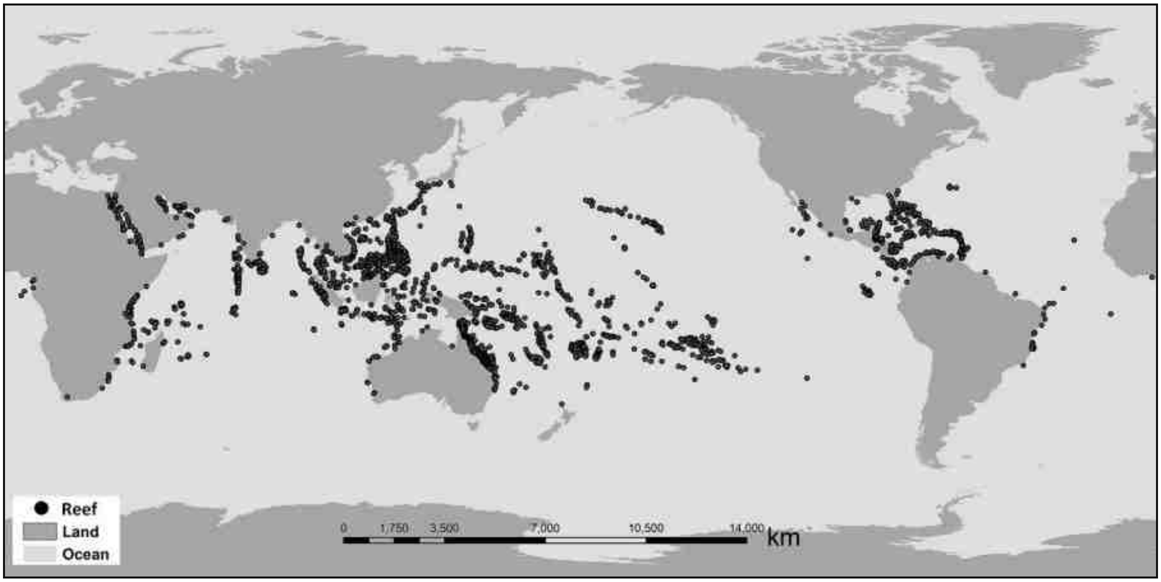


Fig. 10.9. Distribution of coral reefs in Earth's oceans. Adapted from a National Oceanic and Atmospheric Administration map.

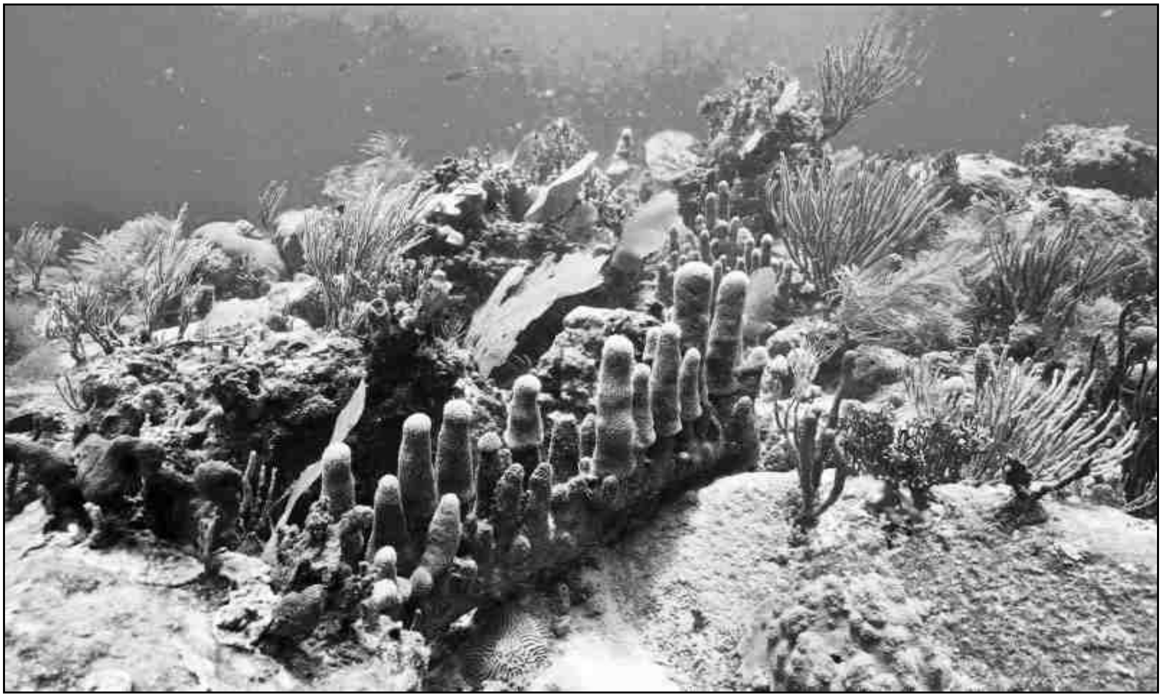


Fig. 10.10. Coral reef in Virgin Islands Coral Reef National Monument. Photo by S. Pershern, Submerged Resources Center, National Park Service.

Reef-forming corals are classified as animals, but they really defy classification as they have characteristics of animals and plants (Fig. 10.10).[11] Like plants, most corals are stationary. Corals have three basic tissue layers: an outer epidermis, inner cells lining a digestive cavity, and a layer in between. Corals have tentacles encircling their mouths, which have stinging cells for capturing prey and defense. Algae grow inside the coral's tissues, through a mutually beneficial relationship. The algae perform photosynthesis within the cells of the coral animal. Corals benefit from this by receiving some of the energy (carbon compounds) produced by the photosynthesizing algae. As another energy source, corals feed on zooplankton (tiny, single-celled, floating animals in sea water). These energy sources, combined with manufacturing calcium carbonate from seawater, enable the corals to form their skeletons. The living tissue of the corals then grows around these skeletons. Multiple corals can grow near each other, even becoming directly attached to each other to form colonies. Some individual corals grow to diameters of 10 meters (33 feet) and live for millennia. The shapes of corals range from branching (similar in shape to small trees) to sheets or mounds (Fig. 10.11). Corals regenerate from fragments that break off and begin growing, and by producing larvae, which drift to new locations.

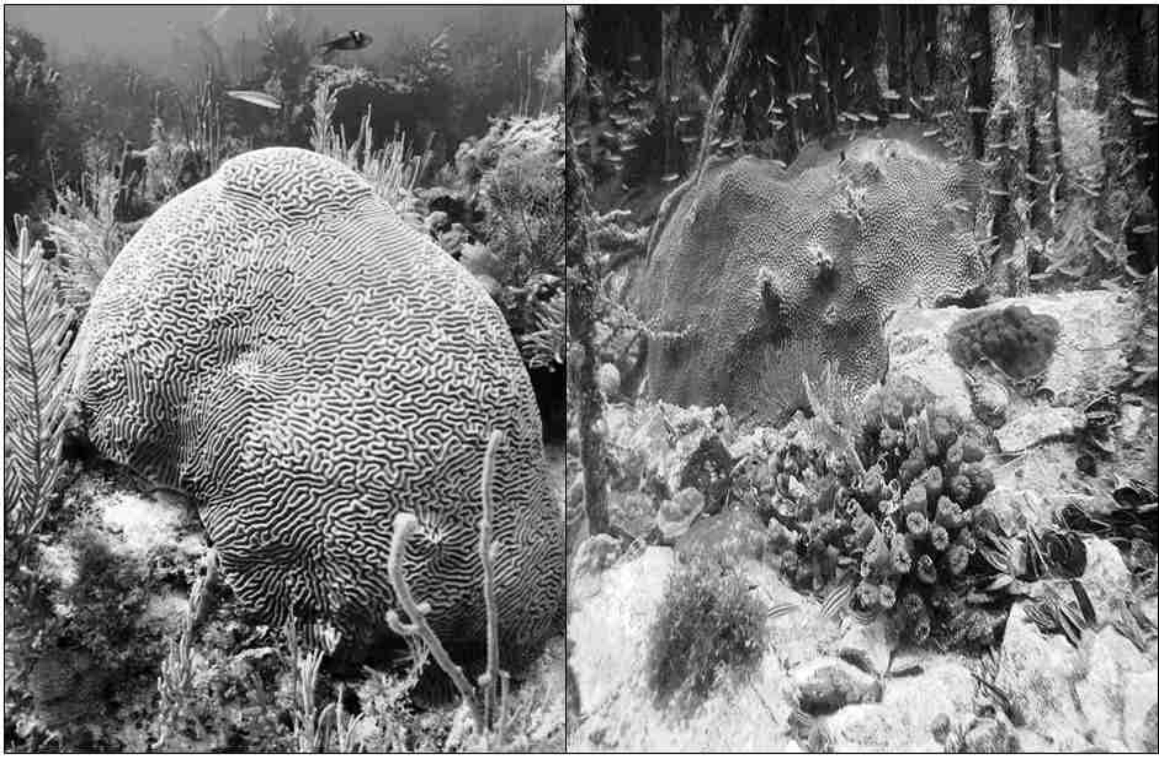


Fig. 10.11. Left: brain coral, Biscayne National Park (by S. Pershern, Submerged Resources Center, National Park Service). Right: different shapes of coral and associated fish in Virgin Islands National Park (by A.B. Tihansky, U.S. Geological Survey).

The high biodiversity and ecological functions of coral reefs are increasingly threatened by degradation of coral reef ecosystems. This is exemplified around the Virgin Islands in the Caribbean Sea, where three U.S. national park units contain coral reefs. Virgin Islands National Park and Virgin Islands Coral Reef National Monument are on or near the island of St. John. Buck Island Reef National Monument is near the island of St. Croix. Since the 1970s, long-term monitoring of reefs has found that live coral cover has declined, diseases of coral have increased, and fish of several species are rarer and smaller.[22]

Condition of coral reefs significantly worsened in 2005, which had some of the warmest temperatures ever recorded in the Caribbean Sea.[1] More than 90% of coral bleached that year followed by effects of a white plague disease.[16] The process of coral bleaching occurs when corals under environmental stress expel the algae inside their tissues, exposing the white carbonate skeleton of corals. The usually brilliantly colored corals become whitish. Ocean temperatures of 1 to 2 degrees Celsius (2 to 4 degrees Fahrenheit) above normal for four to eight weeks can trigger bleaching. Increased temperatures cause bleaching by reducing the ability of algae to process light. When temperatures exceed certain thresholds, incoming light overwhelms the algae's photosynthetic systems, causing them to produce reactive oxygen chemicals. These chemicals damage cells and cannot be tolerated by corals in high doses. As the chemicals accumulate, corals are forced to expel the algae. Bleached corals essentially begin starving, due to the loss of their energy-producing algae. Like many animals, including humans, healthy corals have stored energy and can survive brief periods without food. If corals have been chronically stressed, however, and if the bleaching is severe, their ability to survive bleaching is compromised. Also like humans, weakened coral have greater susceptibility to disease, as some coral that survive bleaching then succumb to disease. Because corals grow slowly, recovery of coral from severe bleaching can require decades, assuming that future conditions are conducive to recovery.[1]

The decline in corals triggered the first coral species to be listed under the U.S. Endangered Species Act. In 2006, staghorn coral (*Acropora cervicornis*) and elkhorn coral (*Acropora palmata*) of the Caribbean Sea around the Virgin Islands were listed as threatened. As of 2014, 20 more coral species of the Caribbean and Pacific Ocean have been listed under the Endangered Species Act. The National Oceanic and Atmospheric Administration, the agency sponsoring the listings, based the listings on the continued threats to coral reefs of warming ocean temperatures, ocean acidification, disease, effects of commercial fishing, and effects from land uses (such as pollution). Ocean acidification is a concern, because higher carbon dioxide in the atmosphere forms carbonic acid in sea water.[14] The acid reduces the amount of carbonate in the water needed by corals to form their skeletons.

In 2014, an assessment was completed of the trend during the 2000s in the condition of coral reefs of Caribbean national parks.[19] Trends varied among the parks and for different components of the coral reef ecosystem. The cover of live coral declined overall between 2003 and 2010 across parks, but there was a slight increase in cover after 2008 (Fig. 10.12). The total amount of fish around coral reefs displayed no clear trend through time. Commercial fishing had been precluded from the parks only starting in 2003, and reefs were still slowly recovering from the severe 2005 bleaching event. Longer term monitoring is likely to be useful for understanding changing conditions of coral reefs in response to stressors as well as conservation measures.

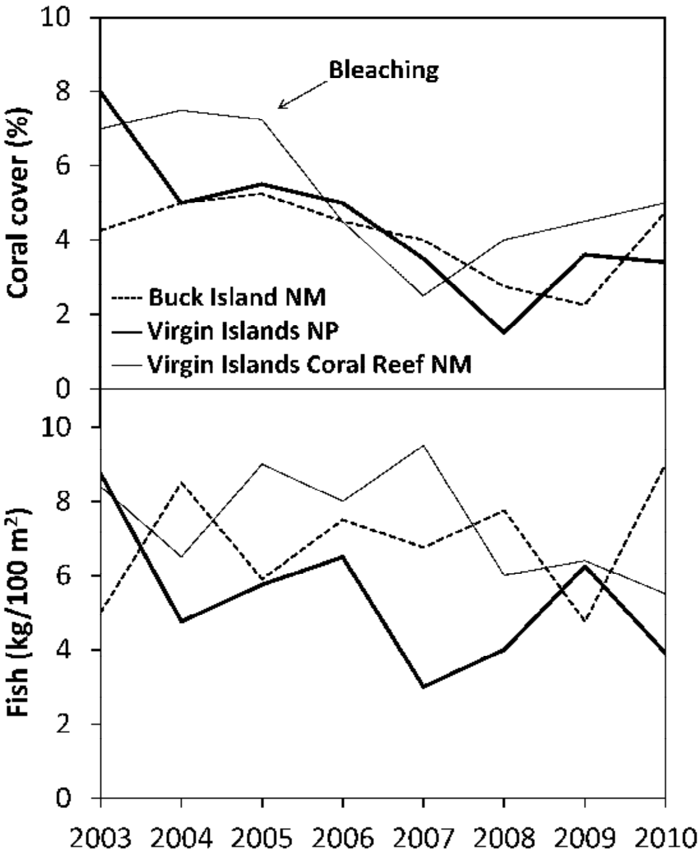


Fig. 10.12. Trends in coral cover and the amount of fish (as kilograms of fish per 100 square meters) in three Caribbean national parks. Data from Pittman et al. (2014).[19]

General approaches to conserving coral reef ecosystems include reducing stressors to the extent possible (such as curtailing pollution, overfishing, and damage to reefs from boats) and determining which reefs are resilient or that may require resoration.[17] Actively restoring coral animals to damaged reefs has not been extensively implemented but may have potential, if the causes of coral declines have been ameliorated. One approach to restoring reefs is rearing small coral fragments in underwater "nurseries" and transporting the animals onto damaged reefs.[23] For instance, coral fragments have been propagated by attaching them to floating nets or ropes anchored to the ocean floor within protected nurseries. Over 86 species of coral worldwide have been "farmed" in underwater nurseries, at least in small quantities. Colonies are then moved from nurseries and attached to reef structures desired for restoration by drilling small holes into reefs and securing the colonies with pegs or underwater glue. Intensive restoration of corals is expensive and difficult to apply over large areas, but could help initiate recovery on damaged reefs not recovering naturally.

Challenges to Conserving Coastal Parks in Large Oceans

Conservation in coastal parks is challenging because threats come from both land and sea, parks are tiny relative to the size of oceans, and the condition of parks depends on the condition of nearby international waters. These challenges are exemplified at Dry Tortugas

National Park, 115 km (70 miles) west of Key West, Florida. The park is at the western end of an ancient coral reef extending 350 km (220 miles) from Key West to near Miami.

Some of the park's conservation-priority species are sea turtles (Fig. 10.13). All six species of sea turtles of U.S. waters are protected under the Endangered Species Act. Five species inhabit southern Florida waters around Dry Tortugas: loggerhead (*Caretta caretta*), green turtle (*Chelonia mydas*), leatherback (*Dermochelys coriacea*), Kemp's ridley (*Lepidochelys kempi*), and hawksbill (*Eretmochelys imbricata*). Sea turtles swim in open ocean, and emerge from the ocean to crawl up beaches for nesting. Female sea turtles excavate a nest in the sand with their rear flippers to deposit a clutch of eggs. After covering the eggs with sand, females swim away, allowing the eggs to incubate and hatch on their own. An egg is the size of a ping-pong ball and requires two months to hatch.

Fig. 10.13. Hawksbill sea turtle (courtesy of Virgin Islands National Park).

Sea turtles face several threats from land and sea. Euro-American settlement introduced the tree Australian pine (*Casuarina equisetifolia*) to coastlines of what would become Dry Tortugas National Park. The invading pines reduced debris-free beaches required by sea turtles for nesting. In response, the park has been clearing non-native trees from coastlines.[18] Artificial lighting is a more insidious threat. Sea turtles have long been traveling on and offshore, but artificial lighting of coastlines by humans may affect the turtles' navigation. Threats in the sea include capture in commercial and recreational fishery equipment, entanglement in trash or debris, and collisions with watercraft.[12]

A tracking program of the park's sea turtles demonstrated just how far sea turtles travel and how important international waters are to the species' survival.[9] One of the turtles that was tracked was an adult female loggerhead named by researchers as "Courtney." She was affixed with a tag for tracking via satellite after nesting in East Key in Dry Tortugas National Park in July 2013. Over the next four months, Courtney journeyed 6,332 km (3,935 miles), or the equivalent of 1.5 times the maximum width of the contiguous United States. Her route was north out of the park into the Gulf of Mexico, then south to the shores of Cuba, along the Yucatan Peninsula, back out across open ocean to the Cayman Islands, and then along the coasts of Honduras and Nicaragua. As of November 2013, she was near Miskito Cays Biological Reserve, a Nicaraguan protected area off the Nicaragua coast. Not unlike a situation with migratory birds, conserving sea turtles in Dry Tortugas National Park is contingent upon safety within international habitat that the turtles occupy for significant periods of time.

11 CLIMATE CHANGE

Climate change has captured society's attention, perhaps because weather is noticeable to people daily and climate strongly influences society and ecosystems. But what exactly is climate change? After all, Earth's climate has long changed through natural fluctuations in the Sun's output, Earth's tilt, concentrations of greenhouse gases in the atmosphere that warm Earth, and other gases, such as from volcanoes, that reflect sunlight to cool Earth. Between the 1950s and 2000s, Earth's average temperature warmed by 0.6 degrees Celsius (1.1 degrees Fahrenheit).[14] This warming cannot solely be explained by natural fluctuation. By partitioning natural and human influences on climate, natural variation accounted for 25% and human influences 75% of the warming (Fig. 11.1). In other words, if climate was just controlled by natural factors, only slight warming would have occurred after 1950. But actual warming was substantial.

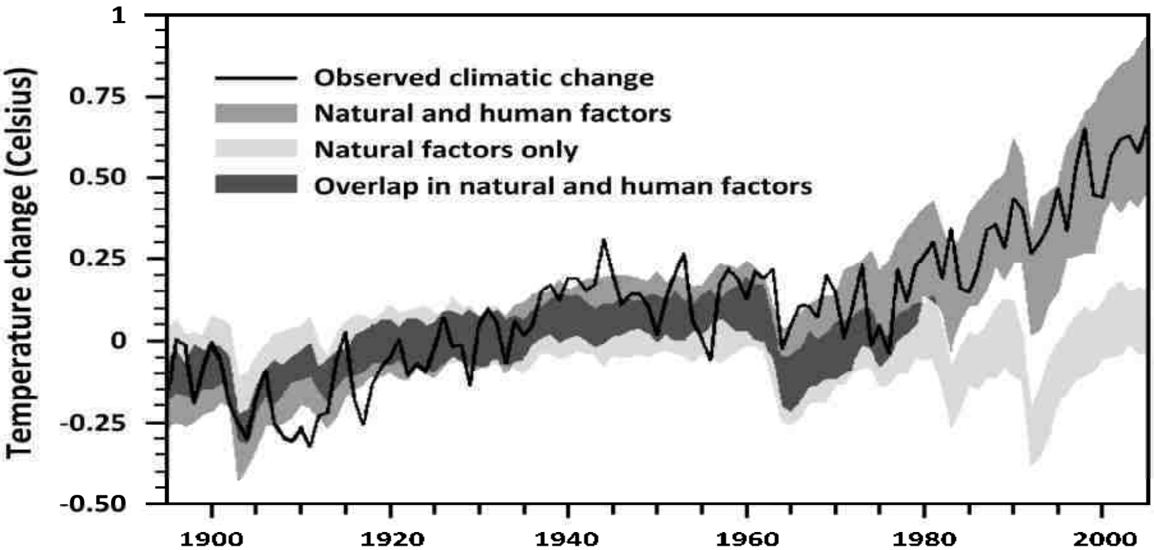


Fig. 11.1. Simulation of how global temperature would have changed during the last century if it was influenced only by natural variation, compared to how temperature did change. Particularly after the 1970s, human factors had to be included to account for the observed temperature change. Adapted from Melillo et al. (2014)[18] using data from Huber and Knutti (2012).[14]

A main way humans influence climate is by producing greenhouse gases through burning fossil fuels. Carbon dioxide, methane, and other gases are byproducts of burning organic material such as fossil fuels. In the atmosphere, human-produced emissions augment the naturally occurring carbon dioxide and greenhouse gases produced by wildfires and other natural processes. Changes in land use, such as reducing forests that take in carbon dioxide from the atmosphere, also raise concentrations of greenhouse gases. In

March 2015, the National Oceanic and Atmospheric Administration reported that the monthly global concentration of carbon dioxide in the atmosphere exceeded 400 particles per million particles of air (ppm) for the first time in the modern era. This concentration had not occurred in Earth's atmosphere for at least a million years.[17]

Temperatures are projected to rise another 1 to 2 degrees Celsius (2 to 4 degrees Fahrenheit) in most areas of the United States by the year 2050.[18] Thereafter, projections vary from an additional 2 to 4 degrees Celsius (4 to 8 degrees Fahrenheit) or more increase by the year 2100. These projections partly depend on whether emissions of greenhouse gases by humans are reduced. Temperature increases of this magnitude are anticipated to produce major ecological changes on their own, but other manifestations of climate change, like shifting rainfall patterns, might be just as important.

Three keys for understanding climate change and its effects are remembering that: 1) Human influences on climate are superimposed on an already naturally varying climate. If a few cold years occur, for example, it does not mean that a trend for climate warming has reversed. Natural fluctuations in cold spells, droughts, and storms are anticipated to continue being a major part of climate. 2) Climate change and its effects are likely to keep varying from place to place and through time. For instance, precipitation is projected to increase in the northern United States but decrease in the Southwest.[18] 3) Climate change is anticipated to have large effects on ecosystems, but so are many other factors like non-native species. Climate change is most accurately placed within a comprehensive perspective of numerous factors simultaneously affecting ecosystems.[23]

Also important to consider are the uncertainties in climate changes and their effects. Generalized projections of overall increases or decreases in temperature and precipitation can be useful, but the specifics might drive the ecological effects of climate change.[21] Just as daily weather forecasts often struggle to pinpoint the exact locations or amounts of localized rainstorms, projecting things like specific shifts in the timing of seasonal rainfall years to decades in advance has uncertainty.

Even a scenario of no net change in annual precipitation could induce major ecological changes, depending on how that scenario unfolds. If more of the precipitation falls as rain rather than snow, winter snowpacks will probably decline. Unless this is perhaps compensated for by regular, light rains in spring and summer, the ecosystem has lost a reliable source of slowly released water that had come from melting of snowpacks in spring and summer. Similarly, in southwestern deserts, different annual plant communities develop depending on whether an area receives more of its rainfall in the cool, winter growing season versus in the summer monsoonal season.[15] A change in the timing of rainfall, even if the total yearly rainfall does not change, can alter the vegetation. A key part of understanding changes in climate is identifying which changes matter the most in particular ecosystems.

Compensating or counteracting changes and the ability of species to adapt are additional uncertainties. For example, greater rainfall predicted in certain areas could be expected to benefit plants, but not if the benefit is counteracted by increased storm severity that damages plants.[27] Warming temperatures are anticipated to increase the frequency and severity of extreme events, including storms, floods, and droughts. These types of extreme events – difficult to predict – can counteract other changes and dominate the development of an ecosystem for a long time. When confronted with these changing environments, species can adapt in place to the new environment, move to locations that have a semblance of the species' old environment, or incur population decline or extinction. Much current research seeks to understand which of these situations characterizes different species, so we can anticipate which species and ecosystems might be most affected by climate change.

For as profound as climate change could be, we must not make the mistake of dismissing the importance of other factors on their own and their interaction with climate. Some examples illustrate this. Stopping natural low-severity fires has resulted in severe fires extensively de-foresting western parks with dry forests (Chapter 3). Similarly, introduction of non-native pests has nearly eradicated certain forests from parks (Chapter 7). How can we say climate change drives these alterations, when factors other than climate have already removed the forests? Climate change could influence these factors, however. Warming temperatures can exacerbate the severity of fires primarily driven by increased fuel loads from past manipulation of forests by humans.[1] Likewise, climate change might influence the reestablishment of tree species in forests killed by non-native pests.

This chapter provides a long-term perspective of changing climates in national parks, examples of how climate change may be influencing parks, and conservation strategies in a changing climate. Responses of ecosystems to past climate fluctuations during Earth's history provide valuable context for contemporary change. U.S. national parks have had international significance for their fossil assemblages and long-term reconstruction of Earth's climate history. Examples of changing natural features (such as glaciers) and species distributions are also discussed in the context of conserving park namesakes, rare species, and key ecological processes such as pollination.

Fig. 11.2. Fossil redwood tree stump originating from a previously milder climate, surrounded by cold-climate contemporary ponderosa pine forest, Florissant Fossil Beds National Monument, Colorado (S.R. Abella).

Florissant Fossil Beds Reveals an Ancient Humid Forest in Colorado

Envisioning how ecosystems have changed through Earth's history can be difficult. National parks containing fossils intermingled with modern biota are helpful in this regard. Examples of parks known for fossils include Petrified Forest National Park (Arizona), and the national monuments Dinosaur (Utah/Colorado), Fossil Butte (Wyoming), John Day Fossil Beds (Oregon), Hagerman Fossil Beds (Idaho), and Tule Springs Fossil Beds (Nevada).

Another park, Florissant Fossil Beds National Monument, conserves within its 2,425 hectares (5,992 acres) one of the richest fossil assemblages in the world (Fig. 11.2). The park is 55 km (35 miles) from Colorado Springs, Colorado, northwest of Pikes Peak, at an elevation of 2,560 meters (8,400 feet). The modern climate is a cold Rocky Mountain climate that supports forests of ponderosa pine (*Pinus ponderosa*) and deciduous aspen (*Populus tremuloides*). But 34 million years ago, the area's climate was humid and warmer, perhaps not unlike the mild climate of modern San Francisco, California.[20]

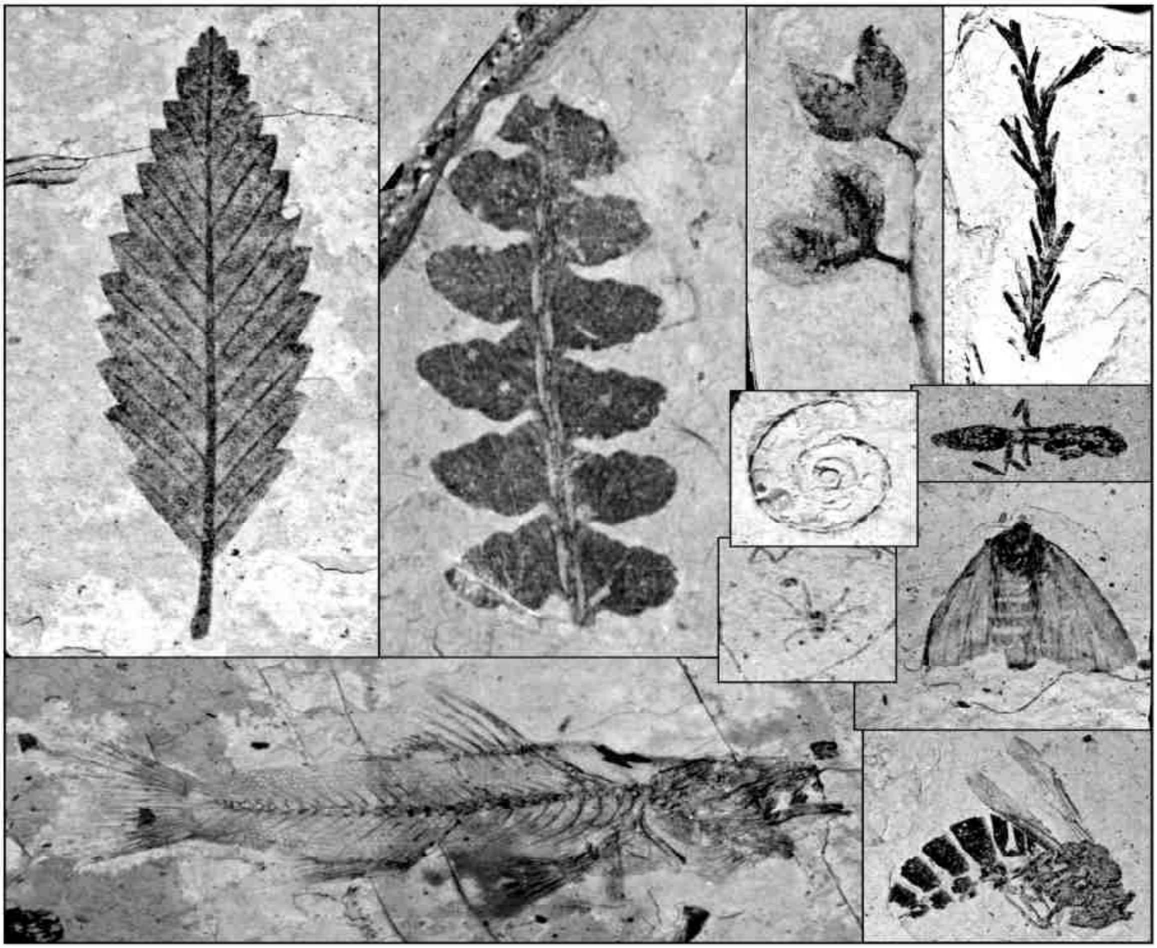


Fig. 11.3. Examples of rich assemblages of fossil plants, snails, spiders, ants, butterflies, wasps, and fish, Florissant Fossil Beds National Monument, Colorado (National Park Service photos).

A nearby volcano eruption 34 million years ago buried the Florissant Valley, "freezing it in time." Rapidly moving volcanic flows of mud and ash sped through the valley, faster than animals could move and leaving immobile plants no chance to escape.[20] Quickly, the forest was buried in volcanic material up to 5 meters (16 feet) deep, encasing the lower trunks of trees. The trees died, and the unburied tops broke off and rotted away. While the still-buried stumps were entombed in the silica-rich volcanic mud, dissolving minerals seeped into the wood and hardened over time, turning the wood into stone. The fossilized stumps, termed petrified wood, have been exposed at the surface via erosion of the ancient volcanic mud. Paleontologists have identified the fossil stumps as a now-extinct redwood (*Sequoia affinis*), apparently similar to contemporary redwood trees (*Sequoia sempervirens*) of the California coast.

In addition to the sequoia trees, Florissant's rich fossil assemblage includes 1,700 species, representing plants, insects, birds, mammals, and even fish that inhabited water bodies present long ago (Fig. 11.3). The fossilized semi-tropical forest ecosystem of Florissant, surrounded by a much different cold-climate modern forest, provides visible evidence for the variability of Earth's climate and the species different climates supported.

600,000 Years of Climate Change in the Death Valley Region

Earth's climate is subject to long- and short-term cycles of change, with ice ages dominating the past 600,000 years. Researchers reconstructed climate history at Devils Hole, a fissure in a fault zone, in eastern Death Valley National Park.[37] The site was 115 km (70 miles) northwest of Las Vegas, Nevada, in the Mojave Desert. Devils Hole offered unique insight into climate history by containing veins of calcite preserved underwater inside the fissure. A 36 centimeter (15 inch) long core from a calcite vein was extracted by divers. The core was chemically analyzed for different forms of oxygen, which are related to temperature and can be dated. A 600,000-year record revealed cold periods lasting 80,000 to 130,000 years, with brief intervening warm periods lasting 20,000 years (Fig. 11.4).

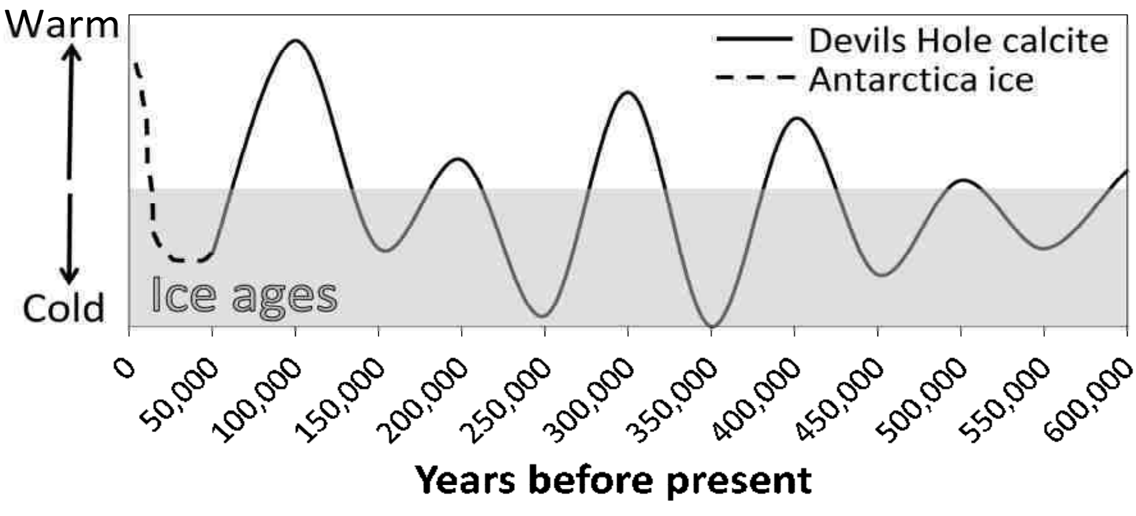


Fig. 11.4. Reconstructed temperature for the last 600,000 years using a calcite vein in Death Valley National Park and Antarctica ice. Ice ages dominated climate. Data from Winograd et al. (1992).[37]

What about more recent climate? West of Death Valley, in Sequoia National Park, California, climate of the last 2,000 years was reconstructed from tree rings (Fig. 11.5). A 150-year drought occurred from the year 850 to 1000 AD. After a brief, 50-year moderately wet period, another drought ensued, lasting 150 years from 1050 to 1200 AD. A generally warm and dry 550-year span between about 850 and 1400 AD is termed the Medieval Warm Period. While the modern drought in California may seem severe, its duration thus far is nothing compared to the centuries-long droughts of the recent past.[32,36]

Following the Medieval Warm Period, the Little Ice Age began. The cold and relatively wet Little Ice Age lasted from about 1400 to 1850 AD.

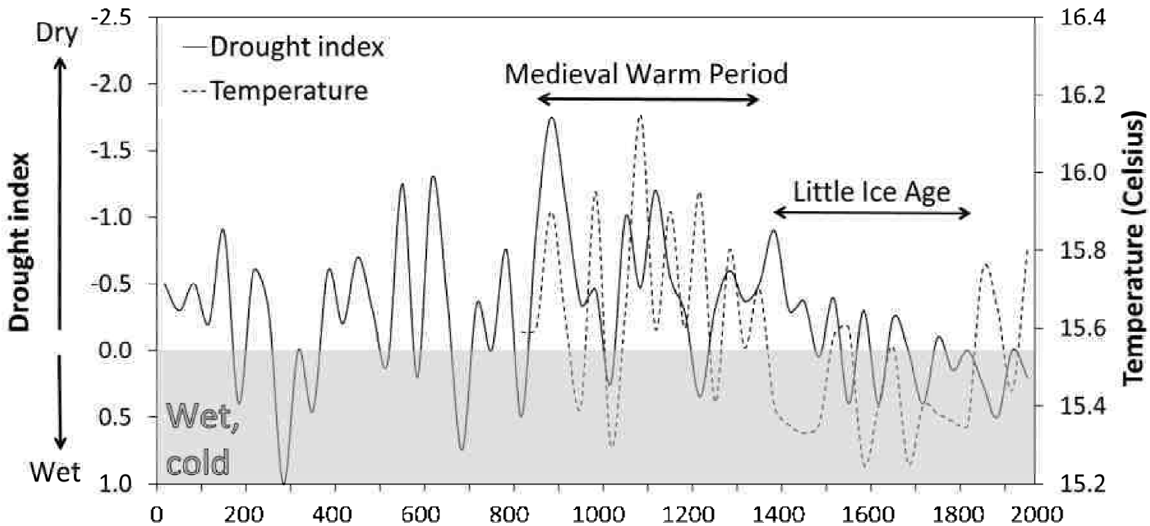


Fig. 11.5. Climate in Sequoia National Park, California, for the last 2,000 years. Drought is measured as the Palmer Drought Severity Index, scaled on the left side, and shown as the solid black line. Temperature is graphed on the right side and shown as the broken line. Data from Swetnam et al. (2009).[32]

The climate of the last 115 years since 1900 has been moderate, compared to the long-term extremes of ice ages and mega-droughts, but it too has experienced fluctuations.[13,29] Back in Death Valley National Park, a weather station at Furnace Creek, at an elevation of 58 meters (190 feet) below sea level on the Death Valley floor, has relatively complete climate records since 1926 for one of Earth's most extreme climates. The average rainfall recorded at the station is only 5 centimeters (2 inches) per year between 1926 and 2014. Some of the extreme events (relative to within the recent 90 years) included: 1) a mid-1920s to mid-1930s drought, during which 393 consecutive days occurred without any measurable precipitation between 1928 and 1930; 2) a mid-century drought from 1954 to 1963; 3) a relatively wet period during the 1970s; 4) a brief but severe drought from 1989 to 1991; and 5) a record-breaking 2004-2005 rainfall total that stimulated extraordinary wildflower blooms (Fig. 11.6).

Considering just February, an important month at Death Valley because it is when annual plants are germinating or already growing from rains the previous fall, patterns of temperature fluctuations during the last 10 years have not been particularly unusual relative to the last 90 years (Fig. 11.6). However, the average high daily temperature in February has been rising since the 1940s. For example, the February average daily high temperature was 22.2 degrees Celsius (71.9 degrees Fahrenheit) in the 1940s, 23.0 degrees Celsius (73.3 degrees Fahrenheit) in the 1950s, and 23.2 degrees Celsius (73.7 degrees Fahrenheit) in the 2000s. The average daily low temperature in February has risen more. It was 6.4 degrees Celsius (43.6 degrees Fahrenheit) in the 1940s, 6.8 degrees Celsius (44.3 degrees Fahrenheit) in the 1950s, and 8.1 degrees Celsius (46.5 degrees Fahrenheit) in the 2000s.

These temperature increases might seem modest, but they are sufficient to reduce the number of years with below-freezing temperatures. In the 70 years between 1925 and 1994, 19 years had at least one day in February below freezing. Only a single year (2002) of the last 21 years from 1995 to 2015 had a freeze in February. This is a change from a February freeze every 3.5 years up to 1994, to a freeze only once every 21 years from 1995 to 2015.

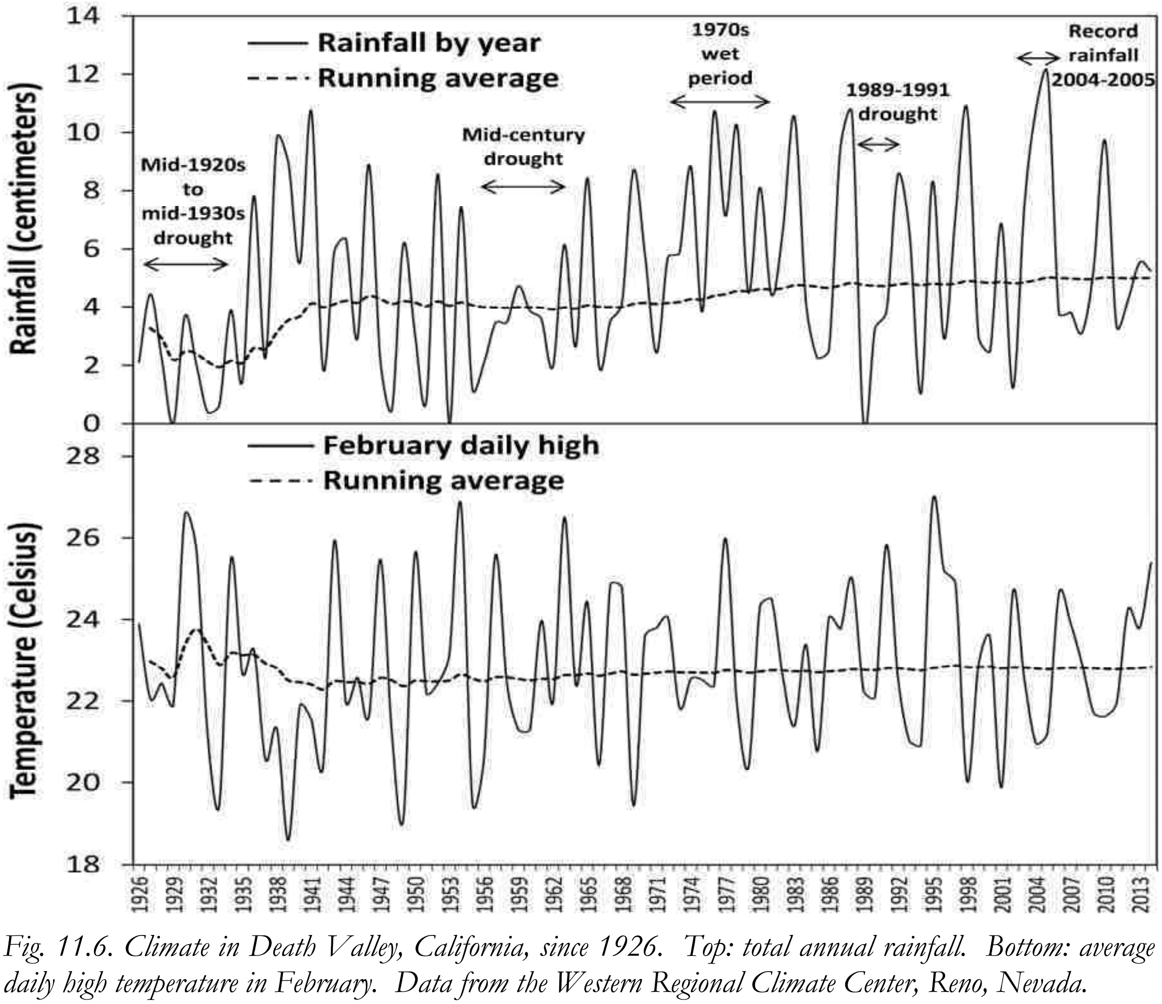


Fig. 11.6. Climate in Death Valley, California, since 1926. Top: total annual rainfall. Bottom: average daily high temperature in February. Data from the Western Regional Climate Center, Reno, Nevada.

What are some of the ecological implications of these climate changes? We know that germination of Death Valley's annual wildflowers, so valued ecologically and by park visitors, is sensitive to slight variation in temperature and rainfall from year to year.[34] But we do not have a good understanding for specifically how the warming temperatures might alter the amount and timing of germination by different species. Death Valley's average temperature changes also illustrate that even slight changes of only a degree can compound into major effects related to freezing and thawing. Freeze-thaw cycles are linked with plant growth, whether it be in a national park or a horticultural and agricultural setting.

Past Climates as Analogs for the Future

Can ecological responses to past climate changes serve as "analogs" for forecasting future ecological responses? During the most recent major ice age lasting from about 60,000 to 11,000 years ago, Death Valley was not covered in ice like northern regions, but its climate was colder and moister than the modern climate. Some of the same species present today had already originated and were in the area, but they were distributed differently across the landscape. Packrat middens provide valuable clues to reconstruct these types of shifts. Packrats do what their name implies – they forage for plant materials, which they bring back to protected locations (such as caves) and hoard. The plant materials become cemented together and are preserved for thousands of years. When analyzed and dated, the preserved material provides an estimate of the local plant composition at a particular time.

Based on analyzing packrat middens in the Amargosa and Panamint Mountains, which rise abruptly from the east and west sides of the Death Valley floor, 13,000 years ago juniper trees (*Juniperus osteosperma*) grew 1,200 meters (3,900 feet) lower in elevation than they do today.[33] Similarly, Joshua trees (*Yucca brevifolia*) occurred just above the Death Valley floor during the ice age before 11,000 years ago, lower than presently. Vegetation now restricted to middle and high elevations could be similar to the Joshua tree and juniper woodlands more widespread during the ice age, but some species present during the ice age are now absent. One example is chaparral yucca (*Hesperoyucca whipplei*), which occurred in ice age woodlands but is not found around Death Valley today. In response to warming at the end of the last major ice age, some species were gained, some lost, and some changed their distribution. The present vegetation still includes "remnant" species from the ice age, which have persisted in favorable locations (Fig. 11.7).

More recently, between 1435 and 1795 AD during the Little Ice Age, the shrub blackbrush (*Coleogyne ramosissima*) shifted its distribution downward 75 meters (250 feet) in elevation.[7] This shift is consistent with a cooler and moister climate, enabling blackbrush to grow at a lower elevation than it could during the warmer and drier Medieval Warm Period.

These past changes are valuable for anticipating how habitat may shift in response to possible future climates, but several factors should be kept in mind.[8] Atmospheric chemistry differs today than during the recent past, such as the amount of carbon dioxide in the air which can affect plant growth. Non-native plants and animals, altered fire regimes, missing predators, and many other factors are also now different and may shape how species respond to a changing climate. Furthermore, fragmentation of habitat by human-built structures, such as freeways and cities, could influence movements of species.

Fig. 11.7. View from a high-elevation bristlecone pine forest in the Panamint Mountains, Death Valley National Park, California. The view is looking east downslope to the Death Valley floor and the Amargosa Mountains rising from the eastern side of the floor. Species have shifted up and down these slopes during ice ages and intervening warm periods. Photo taken during Abella et al. (2015).[3]

Shifting Animal Distributions in Yosemite National Park

In addition to plants, animals can shift their distributions in response to changes in locations of favorable habitat. To evaluate this possibility, an early inventory of small mammals was repeated in the Sierra Nevada Mountains of Yosemite National Park, California.[22] Small mammals were inventoried from 1914 to 1920 along lower to upper mountain slopes. The inventory was repeated using similar methods in the 2000s. In the 90 years between inventories, average minimum monthly temperatures increased by 3 degrees Celsius (6 degrees Fahrenheit). Of 28 mammal species inventoried, 12 (43%) had significant changes in the lower limits of their distributions on the mountain slopes. Of the 12 species that changed, 10 shifted upward in elevation, by an average of 430 meters (1,400 feet). Because these species moved upslope from their lower elevation limit, but did not or could not (by already being at the mountaintop) shift their upper limit higher on the mountain, the species had a net loss of habitat area. Some of the species losing the most habitat were the long-tailed vole (*Microtus longicaudus*), shadow chipmunk (*Neotamias senex*), alpine chipmunk (*Tamias alpinus*), and bushy-tailed packrat (*Neotoma cinerea*). The remaining two species that changed significantly, both shrews (*Sorex ornatus* and *monticolus*), were exceptions to the typical pattern, by moving down in elevation and gaining habitat.

The inventories indicated that many, but not all, species are found in different parts of the landscape today than they were in the early 1900s. Species responded individualistically, as not all species changed their distribution and some even shifted in the opposite direction (by moving downslope) of what would be expected in a warming climate. While it may be tempting to exclusively attribute changes in species distributions to climate change, this is a mistake until the many other factors that affect species distributions are accounted for. For example, exclusion of natural forest fires by humans has corresponded with increases in the number of trees and fire-intolerant species moving downslope during the 1900s in many western mountains.[2] Because different tree species provide different food sources and habitat for animals, changes in the forest resulting from altered fire would be expected to affect animal inhabitants. The observed changes in animal distributions during a period of climate change could result from a combination of several factors.

Tradeoffs in Bird Habitat in Wupatki's Shifting Grasslands and Woodlands

Park managers are tasked with conserving numerous species interacting with each other. Earlier chapters have discussed factors that already make this task challenging, so how might overlaying climate change complicate conservation strategies? One interaction of paramount interest is the relationship between vegetation and wildlife. This is a two-way street where changes in vegetation alter habitat suitability for wildlife, but in turn, shifting wildlife communities change vegetation through selectively eating plants and dispersing seeds.

The shifting grasslands and woodlands of Wupatki National Monument, northeast of Flagstaff, Arizona, illustrate potential changes we should look for in vegetation-wildlife relationships. During the last 100 years, the small trees pinyon pine and juniper have expanded into historical grasslands across vast areas of the West, including at Wupatki. Causes of this expansion remain poorly understood but may entail combinations of past grazing, changes in fire, and climate change. At Wupatki, the expansion of one-seed juniper (*Juniperus monosperma*) into grasslands has created three main vegetation types: former grassland now mostly wooded with juniper, grassland in the early stages of becoming wooded, and grassland still without trees (Fig. 11.8).

A 1998 assessment found that bird diversity was highest in the most diverse vegetation habitat: grassland in the early stages of becoming wooded.[30] This habitat had features of both woodlands and grasslands. Bird diversity was lowest in grassland without trees. In addition to the differences in the total number of birds, certain species occupied particular habitats. Ground-nesting birds, such as meadowlarks (*Sturnella neglecta*), predominated in grasslands and declined when trees occupied the grasslands. As expected, tree- and cavity-nesting birds, such as pinyon jays (*Gymnorhinus cyanocephalus*), favored woodlands.

This example illustrates the types of tradeoffs that may occur with climate-related shifts in vegetation habitats and associated tradeoffs with accompanying management decisions. Maximizing Wupatki's total bird diversity might be accomplished by maintaining the savanna-like vegetation of trees interspersed in grasslands. Thus, if climate change is related to persistence of trees in grasslands, it may serve to actually increase the capacity of the landscape for supporting certain wildlife. On the other hand, conserving birds dependent on grasslands might require reducing juniper on a portion of the landscape, if the trend for conversion of grasslands into woodlands continues in the future climate.

Fig. 11.8. Juniper trees in the grasslands of Wupatki National Monument, Arizona. Photo taken from the Citadel Pueblo (S.R. Abella).

Disappearing Park Namesakes: Glacier National Park

Losing the namesake of a national park might seem unthinkable, yet continuation of current trends could do just that in Glacier National Park, Montana. A glacier is a body of snow and ice that moves. Glaciers form when snow and ice accumulation exceeds melting, and they shrink when melting outpaces accumulation. Around 150 large glaciers, over 10 hectares in size (25 acres), existed in Glacier National Park in 1910 when it was established.[11] Today, the U.S. Geological Survey identifies only 25 large glaciers remaining in the park. Based on analyzing aerial photographs taken between 1966 and 2005, the 25 remaining large glaciers lost an average of 26% of their area. Twelve smaller glaciers (less than 10 hectares in size) shrunk even more, decreasing by 60%. There were no exceptions, as all 37 glaciers shrank (Fig. 11.9). One projection is that by 2030, the small glaciers will disappear completely and so will some of the large ones.[11] Since 1900, the average annual temperature of the park has increased 1.3 degrees Celsius (2.5 degrees Fahrenheit). Additionally, snow melt has been occurring earlier in the year, and rain (rather than snow) has proportionally increased. None of these changes are conducive to sustaining glaciers.

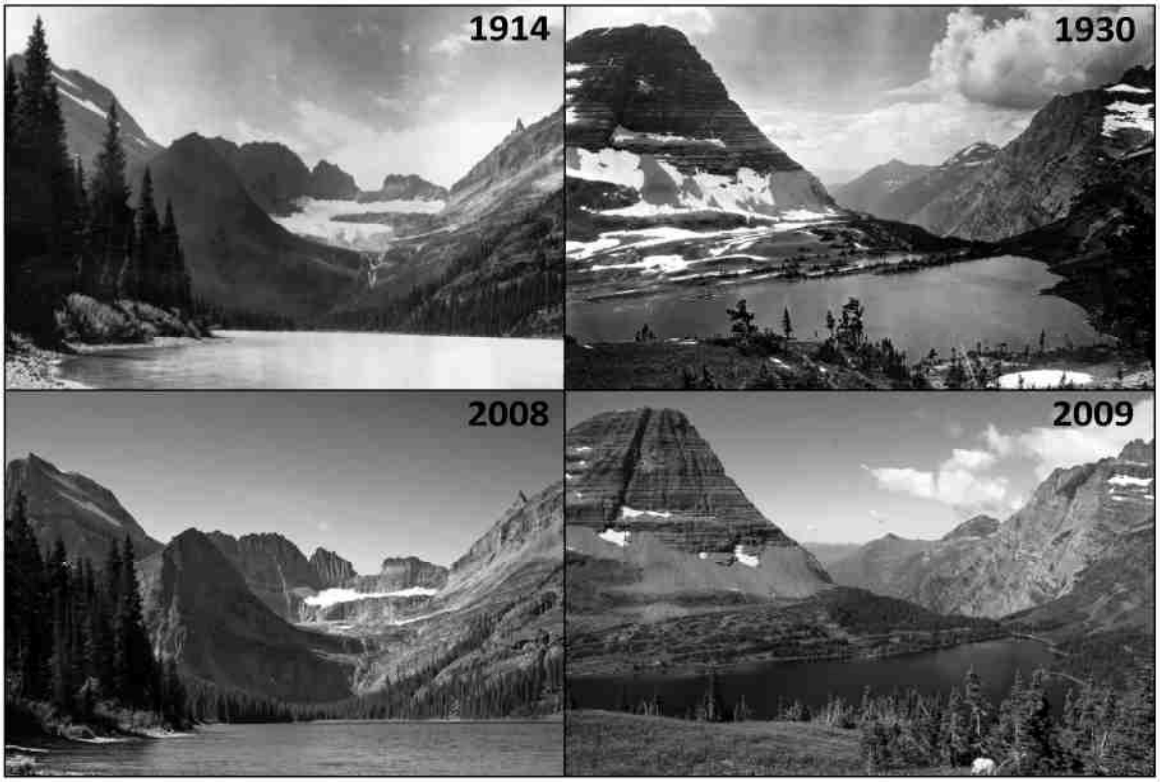


Fig. 11.9. Left: retreat of Grinnell Glacier, viewed from the shore of Lake Josephine, Glacier National Park, Montana. Right: vegetation colonizing ground exposed by retreating glaciers, Hidden Lake. Photos courtesy of Glacier National Park (top left), T.J. Hileman (top right), and L. McKeon (bottom).

The glacial melting needs to be viewed in context. Relative to the last few thousand years, the sizes of glaciers in 1910 when the park was designated were likely unusually large, owing to a cold, snowy period during the Little Ice Age. Glacial melting could be an example where human changes to climate are superimposed on natural fluctuations. Human activities have likely accentuated natural warming following the Little Ice Age. While continuation of climate trends does not bode well for maintaining glaciers, some years in the future might still support re-forming of glaciers and future changes are hard to predict.

Furthermore, while losing the ecological functions of glaciers (such as slow release of water and moderation of temperature swings) is likely to have some negative effects, areas formerly containing glaciers have been colonized by plants and wildlife (Fig. 11.9). This is not unlike the situation in Lake Mead National Recreation Area, where receding of Lake Mead has enabled colonization by land species, including by conservation-priority rare plants (Chapter 8). With or without glaciers, Glacier National Park supports an incredible landscape inhabited by a variety of species. Still, if they disappear, glaciers are likely to be missed as the park's namesake and for their ecological functions.

Conserving Iconic Species: Haleakala Silversword

Similar to other stressors, a concern with climate change is how it may affect rare species with small populations or that occur only in small areas. One example is a unique plant species – Haleakala silversword (*Argyroxyphium sandwicense*) – only found near the summit of Haleakala Volcano on the island of Maui, Hawaii, in Haleakala National Park. Silverswords are estimated to grow for 20 to 90 years as non-flowering, basal leafy plants, before flowering only once at the end of their lives.[16] When flowering, the plants produce a stalk 2 meters (7 feet) tall bearing up to 600 flowers. These characteristics give the silversword a striking appearance in its barren volcanic habitat and iconic status within Hawaii's unique evolutionary heritage. As a result, silverswords are prime attractions for Haleakala's 1 million visitors annually (Fig. 11.10).

Fig. 11.10. Silverswords flowering along the Sliding Sands Trail, Haleakala National Park, Hawaii (National Park Service).

But with its habitat around the high-elevation volcanic summit, how will the species fare in a changing climate? The silversword population size has fluctuated during the last century.[16] The population was a critically low 4,000 plants in the 1920s, a decline attributed to feral livestock and vandalism of the plants by humans. After conservation measures were implemented, the population jumped to 65,000 plants in 1991. Following this increase, the 1990s and 2000s were warmer and drier. A census in 2010 detected 28,492 silversword plants, less than half the number present in 1991. It is difficult to unambiguously attribute the recent 20-year decline to climate change, but climate is the main factor that has changed (for example, feral livestock are still kept away from the plants). Also, populations at the lowest elevations are displaying the greatest mortality, consistent with an expectation that climate warming is pushing the sites outside of the tolerance limits of silverswords.

The future of the species is uncertain, because favorable years with good silversword reproduction could reverse the downward trend. Indeed, in July 2014, Haleakala National Park reported that the silversword bloom that summer was among the best in recent decades. Especially with such a valued park feature, a cautious approach is warranted including continuing to reduce controllable stressors (such as continuing to keep feral livestock away from the plants) and maintaining long-term monitoring. Monitoring is crucial to detect meaningful population trends and to identify if further management is needed. Keeping an eye out for effects of non-native argentine ants, discussed in Chapter 6, is also important because the ants may interact with native insects that pollinate silverswords.

Pollinators in a Changing Climate

Pollination is the transfer of pollen grains from the male part of a flower to the female part of a flower, triggering the development of a seed.[35] Pollination is fundamental to life on Earth as we know it. Of 352,000 flowering plant species, 88% completely or partly (along with wind or water) rely on animals for pollination.[24] A major concern with climate change is that the timing of flowering by plants becomes mismatched with periods of activity of animal pollinators.[19] Some plant species have advanced their spring flowering by a month in the contemporary climate, compared to before the 1970s.[6]

Many animals are pollinators, including bees, butterflies, moths, wasps, flies, beetles, ants, bats, birds, and some small mammals. Different plant species "cater" to different pollinators. For example, plants reliant on bees for pollination generally have bright flowers, such as yellow or blue; a pleasant odor; and sweet nectar. Bee-pollinated plants also have shallow flowers, with landing platforms, to entice bees through convenient parking. Additionally, the centers of many flowers pollinated by bees reflect ultraviolet light, readily seen by bees but not by humans (Fig. 11.11). Plants pollinated by bats open their flowers at night when bats are active, and the flowers are typically white with a musty odor. These are fascinating cases of co-evolution of plants and animals, where the plants benefit via pollination and the animals are "paid" for this work by receiving energy from the nectar and some pollen they eat. Some plant species are highly specialized by being only pollinated by a single animal species. Most plant species are more general. They might be predominately pollinated by a particular type of pollinator (such as bees), but are capable of being pollinated by several pollinators.

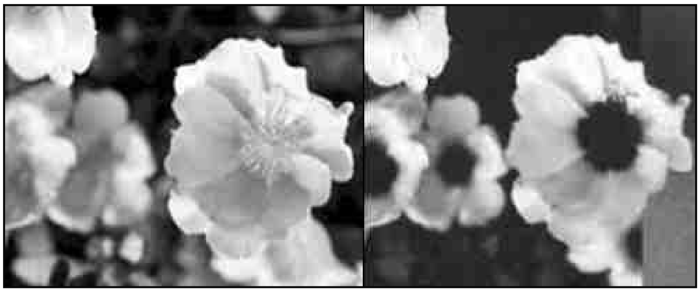


Fig. 11.11. How humans see flowers (left) compared to how bees see them (right). Photos courtesy of the U.S. Forest Service.

Recent inventories have illustrated the diversity of pollinators in national parks. For example, a late 1990s inventory found 66 species of bees in Everglades National Park, Florida.[25] The bees were mostly native species, except for two widespread non-native species, the European honey bee (*Apis mellifera*, perhaps some as Africanized hybrids), and the leaf-cutting bee (*Megachile concinna*). A total of 143 insect species other than bees, plus hummingbirds, were observed visiting flowers of 178 Everglades plant species (Fig. 11.12).

While not all insects visiting flowers are pollinators, the inventory provided a glimpse into how diverse the park's plant-pollinator network could be. Just one plant species, the palmetto (*Sabal palmetto*), was visited by 29 species of wasps, 25 species of flies, seven species of beetles, four species of butterflies, and several species of bees.[26]

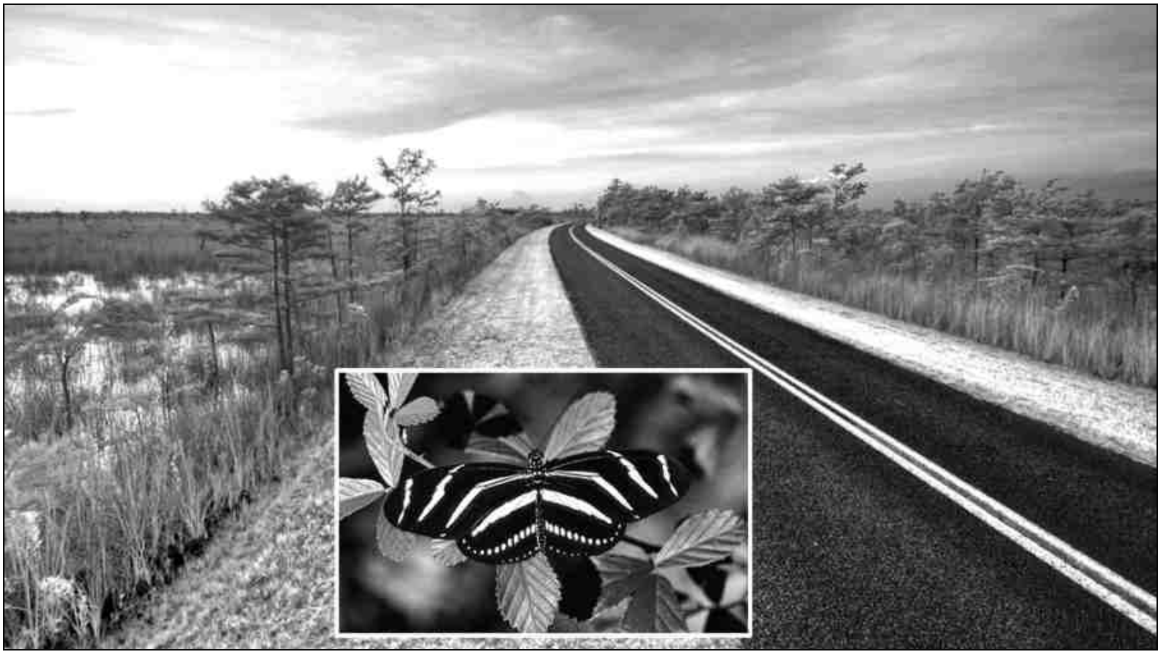


Fig. 11.12. About 200 species of insects visited flowering plants during an inventory of diverse habitats of Everglades National Park, Florida.[25,26] Main photo by G. Gardner (National Park Service). Inset photo, courtesy of the National Park Service, shows the zebra longwing butterfly (Heliconius charithonia).

In another inventory, the 4,266-hectare (10,541-acre) Indiana Dunes National Lakeshore contained 204 species of bees.[10] The lakeshore's location – northwestern Indiana along the Lake Michigan shoreline – and its mixture of inland oak savanna, forest, and prairie, may have promoted the exceptionally high bee diversity. An average of six bee species visited a given flowering plant species during the 2003 to 2010 inventory.

Plant pollination epitomizes the type of natural process desirable to conserve within national parks. Currently, there is global concern about declines in pollinator abundance and what the implications are for growing food for humans. Changes during the last century of fragmenting habitat into small areas, growing crop monocultures that do not support diverse pollinators, and competition between non-native insects (such as European honeybees) and native pollinators are some of the many factors thought to be affecting plant pollination. Providing refuges for the diversity of native bees and other pollinators is a key function of national parks. Indeed, one synthesis of 23 research studies world-wide found that pollinator diversity was highest on farms that had a natural area such as a park within a kilometer (0.6 miles).[28] Climate change is an added stressor to plant pollination, though some insects shift their activity to correspond with shifting timing of plant flowering, so the plants and pollinators remain in sync. Also, if some current pollinators do become out of sync with plant flowering, the diversity of pollinator animals could help buffer certain plant species from pollination failure. Clearly, there is a need to monitor changes in plant pollination in park ecosystems and other systems dependent on pollination.

Conservation and Adaptation Strategies in a Changing Climate

While political and social debates might continue on whether current climate change is human-caused, a changing climate, regardless of the cause, poses challenges to societies. Changing climates have apparently created difficulties for societies in the past. For example, drought periods likely affected the Maya civilization and other Native American cultures, such as the Anasazi in the Southwest and the Mississippian Cahokian in Illinois.[5] Most human infrastructure, including parks, is not designed to be "mobile" if climate changes.

One challenge to developing proactive conservation strategies in parks is uncertainty in specific future climate projections and potential future ecological effects of climate change. There are cases where a threat is clearly known, such as the rising sea levels along the New York coast discussed in Chapter 10. But often it is hard to accurately predict the specific aspects of climate change (such as extreme events like storms) that can matter the most to ecosystems. It should also not be assumed that all effects of climate change will be negative for everything. For example, fewer freezing days may benefit certain cacti species in Organ Pipe Cactus National Monument, Arizona, that are sensitive to cold.[4] Other climate changes might seem like they should have effects, but might be "cancelled out" by compensating factors. Greater atmospheric carbon dioxide, for example, could be expected to enhance plant growth, but not if greater aridity prevents plants from taking advantage of the carbon dioxide.[31] There probably will be some positive and negative effects of climate change, which are likely to vary in different ecosystems and with different species.

Many strategies for helping ecosystems adapt to a changing climate are under discussion. One strategy often discussed for national parks is actively moving species within a park to areas anticipated to be favorable in a future climate. Such a strategy is called assisted migration or assisted translocation.[12] This strategy is partly based on an assumption that species cannot migrate fast enough on their own to keep pace with a rapidly changing climate. An example would be actively planting a tree species further up a mountainside to track cooler and moister climates still found at higher elevations. A problem, though, is what happens to the forest already at those high locations? This type of project could be described one way as "assisting the migration of species for forest adaptation." But it could be described another way as "removing a natural forest to create openings to plant trees that could grow there in the future." In practice, planting lower-elevation tree species into a higher area might be most applicable where an existing high-elevation forest is unnaturally lost anyway, such as through an unnatural fire or a non-native pest.

Fig. 11.13. The Blue Ridge Parkway, here near Asheville, North Carolina, traverses some of the highest elevations in the eastern United States (photo by S.R. Abella).

Currently, no scientific consensus exists on the appropriateness of drastic measures like assisted migration in parks, because there are so many unknowns. What if the climate projections are wrong in some way (such as with complex changes in rainfall seasonality), so that an existing forest on a site actually is adapted to the current and future climate? Could translocating species around the landscape affect their ability to migrate naturally or limit their adaptation in place? And in the case of a forest already at the mountaintop, there may be no other location to move it to (Fig. 11.13). Actual examples of climate adaptation strategies in parks are few. Possibly having some test examples could provide an impression for what these strategies actually "look like" on the ground rather than just in theory.

An option for managing for climate change that seems robust is to ameliorate other stressors, controllable by local park managers, to promote adaptive capacity of ecosystems to any number of possible future climates. For instance, managing non-native animals could produce several benefits under a variety of climate scenarios.[9] Forage could be freed up for native animals to lessen at least one stress native wildlife may experience in a future climate. Similarly, western dry forests unnaturally choked with fuels by a century of fire exclusion are not well adapted to any climate supporting fire, because they are so susceptible to destruction by severe fires. Thinning these forests makes them more sustainable in the current climate as well as in potentially warmer and drier future climates.[1] Reducing tree density is the best known climate change adaptation strategy for these forests.

12 ACTIVE CONSERVATION

Management more active than has been traditional is likely paramount to successfully conserve national park ecosystems between the present centennial and the future bicentennial of the National Park Service in the year 2116. Losing old-growth forests, coastal beaches, species, and clean freshwater are multi-generational deprivations that compromise the quality of natural resources inherited by future generations.

Passive management of parks mainly seeks to limit new impacts to parks (such as by enforcing regulations on off-road driving), while minimizing actions of human managers on nature. It is a "hands off" approach. Passive management can most effectively conserve nature if two conditions are met: 1) natural processes are freely operating and can continue to do so uninterrupted; and 2) external influences, like non-native species, pollution, and climate change, stay out of parks. Some areas of parks mostly (but not completely) meet these conditions. But many areas do not meet these conditions today, nonetheless in possible futures. "Allowing nature to take its course" is frequently a misnomer, because the course of nature has long since been deflected by human activities. Major ecological changes can and do occur within parks managed using a "hands off" approach.

More actively conserving nature through active conservation measures should not be taken to mean a "command and control" approach to nature by humans. Rather, the opposite outcome is desired for parks: to provide some assistance to ecosystems that have been damaged to better equip them to take care of themselves.[2] Evolution of life on Earth has required thousands to millions of years and is not designed to turn on a dime. A main purpose of active nature conservation is to reestablish or maintain conditions that species require to persist. Examples of active conservation measures are removing non-native species to promote native species, reintroducing or maintaining natural processes like floods or fire that have shaped natural ecosystems for millennia, and planting native plants on unnaturally denuded areas to protect thousands of years of soil formation. The National Park Service has been more active in conservation during certain periods of its history and for certain issues than others. For example, actively reducing non-native plants has been a focus in national parks since the 1980s, with over 56 documented projects.[1]

Many parks have a desire to implement integrated active conservation programs, but there are several constraints. A main constraint is that many parks have only a few full-time staff in natural resources, and small parks may not have any. They also have little or no budget for private contracting to perform conservation activities, such as forest thinning. Citizen volunteers are a major asset to many parks, but basic resources are still needed to coordinate and support the volunteers.

When carefully planned, active conservation projects based on local park ecology have been implemented, they have tremendously improved ecological conditions in parks. Such projects suggest the possibilities. One example was in Bandelier National Monument, northwest of Santa Fe, New Mexico. By the 1990s, the park had supported research studies revealing that the park's woodlands of the small trees pinyon pine (*Pinus edulis*) and one-seed juniper (*Juniperus monosperma*) had changed negatively since the 1800s.[3] Heavy historical livestock grazing had occurred before the park was established in 1932. Natural fires, here primarily low-severity surface fires, were also excluded by humans during the 1900s. As a result, the park's woodlands in the 1990s were choked with trees, had few plants growing on the ground, and incurred soil erosion also resulting in damage to cultural resources. The woodlands at that time supported little biodiversity other than trees. In the late 1990s, chain saws were used to cut trees and reduce the area covered by tree canopies from 35% to 10%.

Fig. 12.1. Pinyon-juniper woodland with openings supporting vigorous wildflower growth and high butterfly diversity, Bandelier National Monument, New Mexico (photo by S. King, National Park Service). Inset photo shows a butterfly (Colias species) visiting deervetch (Lotus wrightii), a host plant for butterflies that increased after trees were thinned (photo by S.R. Abella).

By four years after treatment in 2001, monitoring revealed that twice as many species of wildflowers occurred at restoration sites where woodlands had been thinned, compared to nearby areas where no active restoration was undertaken (Fig. 12.1). Wildflowers covered 13% of the ground at restoration sites, compared to only 1% at unrestored sites.[3] Butterflies responded positively to the more open woodland conditions and greater availability of nectar plants. An average of 20 butterflies occurred within 1,000 square meters (11,000 square feet) at restoration sites, compared to nine butterflies in the same size area at unrestored sites. All 13 species of butterflies found were more abundant at restoration sites. Some of the butterflies benefiting from the conservation action the most were the lupine blue (*Plebeius lupini*), painted lady (*Vanessa cardui*), checkered white (*Pontia protodice*), and the dainty sulfur (*Nathalis iole*). To help the ecosystem sustain these positive changes for biodiversity and plant pollination, the park subsequently began reintroducing fire as a natural process through prescribed burning areas that had been thinned.

In another example, decommissioned military installations became part of Golden Gate National Recreation Area, along the California coast near San Francisco and the Golden Gate Bridge (Fig. 12.2). One area, Fort Funston, contained degraded coastal sand dunes dominated by non-native plants when it was transferred to the National Park Service in

1972.[4] In 1991, the park began restoring a 9-hectare (22-acre) portion of the former military installation to native coastal dune vegetation. San Francisco coastal dune ecosystems had become rare (less than 5% remaining) with urban development. Non-native plants were laboriously removed by hand, and 30 native plant species were propagated and planted on the restoration site. Citizen volunteers maintained the restoration site through weeding and other site care after the main restoration treatments were implemented.

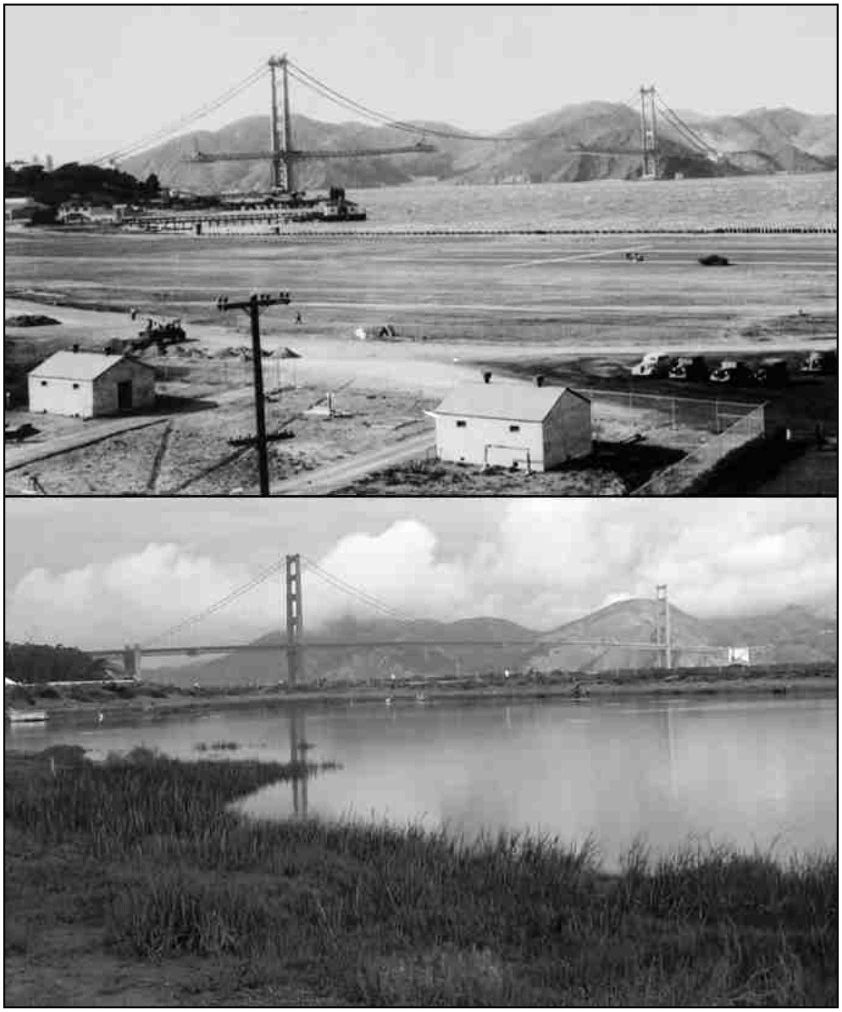


Fig. 12.2. Top: Crissy Field as a military installation in the 1930s, during construction of the Golden Gate Bridge. Bottom: contemporary view of Crissy Marsh, Golden Gate National Recreation Area, California. Photos courtesy of the U.S. Army and the National Park Service.

By the mid-2000s, the habitat condition of the restored site was improved compared to a 4-hectare (10-acre) site that had not been restored (Fig. 12.3). Native plants were three times more abundant at the restored site, while non-native plants continued to dominate the unrestored site.

Ground-dwelling vertebrates (mammals, reptiles, and amphibians) were four times more abundant at the restored site. California newts (*Taricha torosa*) and Trowbridge shrews (*Sorex trowbridgii*) were found only at the restored site. The restored site also supported three times as many native bird species. Black phoebes (*Sayornis nigricans*), Anna's hummingbirds (*Calypte anna*), and Cooper's hawks (*Accipiter cooperii*) only inhabited the restored site. While the work required to improve the heavily degraded site was substantial, it benefited the park's wildlife.

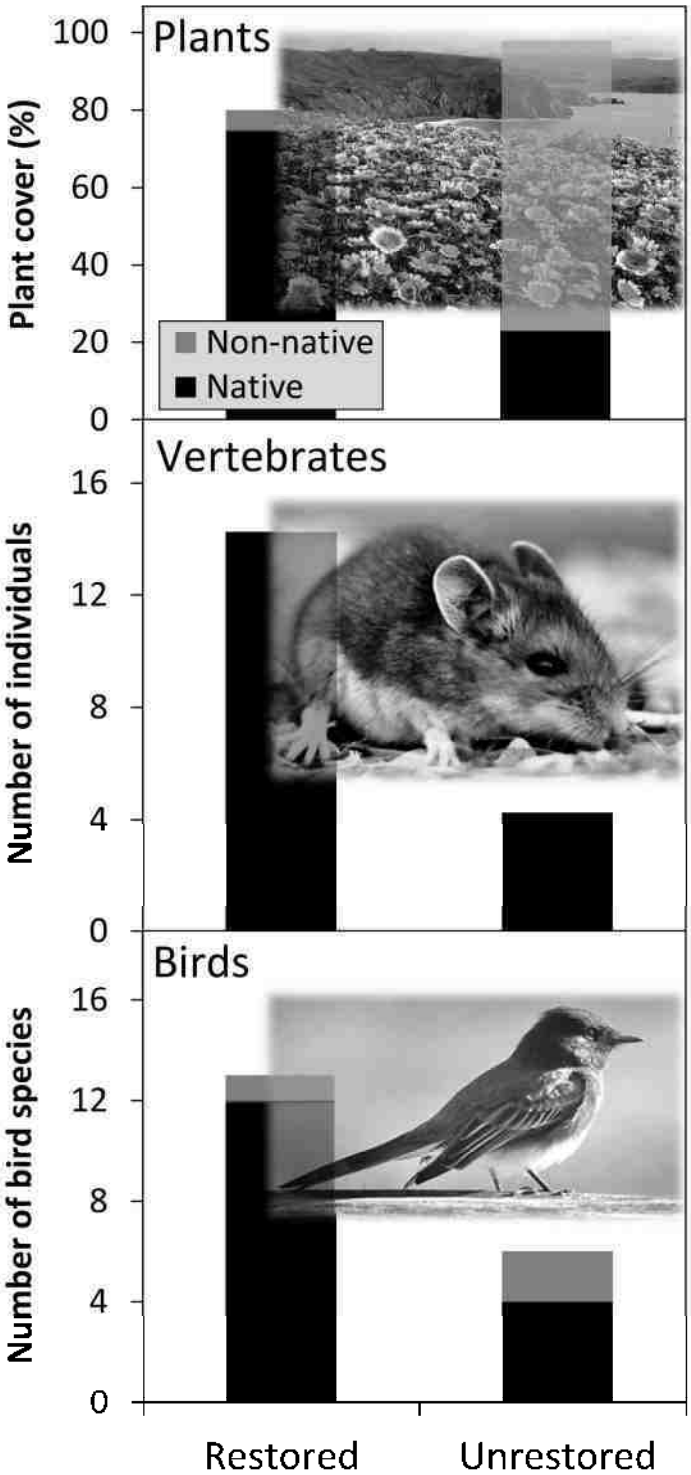


Fig. 12.3. Positive response of native plants and wildlife to active restoration in Golden Gate National Recreation Area, California. For vertebrates, the data are the average number of individuals detected per sample in May and October. For birds, the data are the total number of bird species detected in point counts during summer. Data from Russell et al. (2009).[4] *Photos of representative species include: Plants: Mori Point (courtesy of the National Park Service). Vertebrates: deer mouse (J. Good, National Park Service). Birds: black phoebe (W. Elder, National Park Service).*

There is much to appreciate in national parks and much awaiting discovery. Since 1998, 8,095 species new to Great Smoky Mountains National Park have been discovered.[5] Of these, 951 were new to science. New species continue to be discovered in other parks as well. Unfolding ecological responses to new restoration projects, such as restoring a free-flowing Elwha River in Olympic National Park, are also exciting. The current task is to conserve the existing natural riches of national parks, while ideally enhancing the condition of ecosystems degraded by past events or external influences. Meeting this challenge is likely to require a mix of preservation and active conservation, guided by understanding local park ecology as to which strategies are required.

REFERENCES BY CHAPTER

1 The legacy

[1] Abella, S.R. 2014. Effectiveness of exotic plant treatments on National Park Service lands in the United States. Invasive Plant Science and Management 7:147-163.
[2] Costa, J.T. 2009. The Darwinian revelation: tracing the origin and evolution of an idea. BioScience 59:886-894.
[3] Cullinane Thomas, C., C. Huber, and L. Koontz. 2015. 2014 national park visitor spending effects: economic contributions to local communities, states, and the nation. Natural Resource Report NPS/NRSS/EQD/NRR—2015/947. National Park Service, Fort Collins, CO. 42 pages.
[4] National Park Service. 2006. Management policies 2006. U.S. Department of the Interior, National Park Service. U.S. Government Printing Office, Washington, D.C. 168 pages.
[5] Rykken, J., A. Rodman, S. Droege, and R. Grundel. 2014. Pollinators in peril? A multipark approach to evaluating bee communities in habitats vulnerable to effects from climate change. Park Science 31:84-90.
[6] Schullery, P. 2001. What is natural? Philosophical analysis and Yellowstone practice. Western North American Naturalist 61:255-256.
[7] Secord, P.R. 2014. Images of America: Pecos. Arcadia Publishing, Charleston, SC. 128 pages.
[8] Sellars, R.W. 1997. Preserving nature in the national parks: a history. Yale University Press, New Haven, CT. 404 pages.
[9] Sharkey, M.J. 2001. The all taxa biological inventory of the Great Smoky Mountains National Park. Florida Entomologist 84:556-564 (updated in the online All Taxa Biodiversity Inventory, Discover Life in America project).
[10] Tuberville, T.D., J.D. Wilson, M.E. Dorcas, and J.W. Gibbons. 2005. Herpetofaunal species richness of southeastern national parks. Southeastern Naturalist 4:537-569.
[11] Young, C.C., L.W. Morrison, and J.L. Haack. 2009. Habitat relationships and management implications for *Lesquerella filiformis* Rollins (Missouri bladderpod) on a xeric limestone prairie. Journal of the Torrey Botanical Society 136:233-241.

2 Before they were parks

[1] Abella, S.R., J.E. Craig, S.L. McPherson, and J.E. Spencer. 2014. Watercourse-upland and elevational gradients in spring vegetation of a Mojave-Great Basin Desert landscape. Natural Areas Journal 34:79-91.
[2] Abrams, M.D., and G.J. Nowacki. 2008. Native Americans as active and passive promoters of mast and fruit trees in the eastern USA. Holocene 18:1123-1137.
[3] Barrett, L.R. 1997. Podzolization under forest and stump prairie vegetation in northern Michigan. Geoderma 78:37-58.
[4] Belair, E.D., M.R. Saunders, and B.G. Bailey. 2014. Four-year response of underplanted American chestnut (*Castanea dentata*) and three competitors to midstory removal, root trenching, and weeding treatments in an oak-hickory forest. Forest Ecology and Management 329:21-29.
[5] Brown, P.M., C.L. Wienk, and A.J. Symstad. 2008. Fire and forest history at Mount Rushmore. Ecological Applications 18:1984-1999.
[6] Bucher, E.H. 1992. The causes of extinction of the passenger pigeon. Pages 1-36 in Power, D.M. (editor). Current ornithology, volume 9. Plenum Press, New York.
[7] Cordova, F.J., J.D. Wallace, and R.D. Pieper. 1978. Forage intake by grazing livestock: a review. Journal of Range Management 31:430-438.
[8] Dalgleish, H.J., and R.K. Swihart. 2012. American chestnut past and future: implications of restoration for resource pulses and consumer populations of eastern U.S. forests. Restoration Ecology 20:490-497.
[9] Dennis, J. 2006. National Park Service management policy guidance for restoration of American chestnut to national park system units. Pages 3-11 in Steiner, K.C., and J.E. Carlson (editors). Restoration of American chestnut to forest lands – proceedings of a conference and workshop. Natural Resources Report NPS/NCR/CUE/NRR-2006/001. National Park Service, Washington, D.C. 230 pages.
[10] Eberhardt, R.W., D.R. Foster, G. Motzkin, and B. Hall. 2003. Conservation of changing landscapes: vegetation and land-use history of Cape Cod National Seashore. Ecological Applications 13:68-84.
[11] Ellsworth, J.W., and B.C. McComb. 2003. Potential effects of passenger pigeon flocks on the structure and composition of presettlement forests of eastern North America. Conservation Biology 17:1548-1558.
[12] Evans Ogden, L. 2014. Extinction is forever…or is it? BioScience 64:469-475.

[13] Floyd, M.L., T.L. Fleischner, D. Hanna, and P. Whitefield. 2003. Effects of historic livestock grazing on vegetation at Chaco Culture National Historic Park, New Mexico. Conservation Biology 17:1703-1711.

[14] Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. BioScience 53:77-88.

[15] Frederick, D.J., L. Rakestraw, R.E. Christopher, and R.A. Van Dyke. 1976. Original forest vegetation of the Pictured Rocks National Lakeshore and a comparison with present conditions. Michigan Academician 9:433-443.

[16] Guyette, R.P., R.M. Muzika, and D.C. Dey. 2002. Dynamics of an anthropogenic fire regime. Ecosystems 5:472-486.

[17] Hung, C., P.L. Shaner, R.M. Zink, W. Liu, T. Chu, W. Huang, and S. Li. 2014. Drastic population fluctuations explain the rapid extinction of the passenger pigeon. Proceedings of the National Academy of Sciences 111:10636-10641.

[18] Hyde, E.J., and T.R. Simons. 2001. Sampling plethodontid salamanders: sources of variability. Journal of Wildlife Management 65:624-632.

[19] Jacobs, D.F., H.J. Dalgleish, and C.D. Nelson. 2013. A conceptual framework for restoration of threatened plants: the effective model of American chestnut (*Castanea dentata*) reintroduction. New Phytologist 197:378-393.

[20] King, C.B., and R. Muzika. 2014. Historic fire and canopy disturbance dynamics in an oak-pine (*Quercus-Pinus*) forest of the Missouri Ozarks (1624-2010). Castanea 79:78-87.

[21] MacKenzie, M.D., and P.S. White. 1998. Vegetation of Great Smoky Mountains National Park, 1935-1938. Castanea 63:323-336.

[22] Martin, P.S., and C.R. Szuter. 1999. War zones and game sinks in Lewis and Clark's West. Conservation Biology 13:36-45.

[23] McArthur, E.D., A.C. Blauer, S.B. Monsen, and S.C. Sanderson. 1995. Plant inventory, succession, and reclamation alternatives on disturbed lands in Grand Teton National Park. Pages 343-358 in Roundy, B.A., E.D. McArthur, J.S. Haley, and D.K. Mann (compilers). Proceedings: wildland shrub and arid land restoration symposium. General Technical Report INT-GTR-315. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT. 384 pages.

[24] McEwan, R.W., C. Rhoades, and S. Beiting. 2005. American chestnut (*Castanea dentata*) in the pre-settlement vegetation of Mammoth Cave National Park, central Kentucky, USA. Natural Areas Journal 25:275-281.

[25] Morrisey, R.J. 1950. The early range cattle industry in Arizona. Agricultural History 24:151-156.

[26] Neill, C., W.A. Patterson, and D.W. Crary. 2007. Responses of soil carbon, nitrogen and cations to the frequency and seasonality of prescribed burning in a Cape Cod oak-pine forest. Forest Ecology and Management 250:234-243.

[27] Pyle, C. 1988. The type and extent of anthropogenic vegetation disturbance in the Great Smoky Mountains before National Park Service acquisition. Castanea 53:183-196.

[28] Riege, D.A., and R. del Moral. 2004. Differential tree colonization of old fields in a temperate rain forest. American Midland Naturalist 151:251-264.

[29] Scurlock, D., and D.M. Finch. 1997. A historical review. Pages 43-68 in Block, W.M., and D.M. Finch (editors). Songbird ecology in southwestern ponderosa pine forests: a literature review. General Technical Report RM-GTR-292. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. 152 pages.

[30] Silsbee, D.G., and G.L. Larson. 1983. A comparison of streams in logged and unlogged areas of Great Smoky Mountains National Park. Hydrobiologia 102:99-111.

[31] Simons, T.R., S.A. Shriner, and G.L. Farnsworth. 2006. Comparison of breeding bird and vegetation communities in primary and secondary forests of Great Smoky Mountains National Park. Biological Conservation 129:302-311.

[32] Singer, F.J. 1981. Wild pig populations in the national parks. Environmental Management 5:263-270.

[33] Webb, R.H., J.W. Steiger, and E.B. Newman. 1988. The response of vegetation to disturbance in Death Valley National Monument, California. U.S. Geological Survey Bulletin 1793. 103 pages.

[34] Webb, S.L. 1986. Potential role of passenger pigeons and other vertebrates in the rapid Holocene migration of nut trees. Quaternary Research 26:367-375.

[35] Wilhelm, G. 1973. Fire ecology in Shenandoah National Park. Proceedings of the Tall Timbers Fire Ecology Conference 12:445-488.

[36] Woods, F.W., and R.E. Shanks. 1959. Natural replacement of chestnut by other species in the Great Smoky Mountains National Park. Ecology 40:349-361.

3 Fire

[1] Abella, S.R., J.F. Jaeger, D.H. Gehring, R.G. Jacksy, K.S. Menard, and K.A. High. 2001. Restoring historic plant communities in the Oak Openings region of northwest Ohio. Ecological Restoration 19:155-160.

REFERENCES

[2] Abella, S.R., J.D. Springer, and W.W. Covington. 2007. Seed banks of an Arizona *Pinus ponderosa* landscape: responses to environmental gradients and fire cues. Canadian Journal of Forest Research 37:552-567.

[3] Abella, S.R., and J.D. Springer. 2015. Effects of tree cutting and fire on understory vegetation in mixed conifer forests. Forest Ecology and Management 335:281-299.

[4] Brown, P.M., C.L. Wienk, and A.J. Symstad. 2008. Fire and forest history at Mount Rushmore. Ecological Applications 18:1984-1999.

[5] Caprio, A.C., and D.M. Graber. 2000. Returning fire to the mountains: can we successfully restore the ecological role of pre-Euroamerican fire regimes to the Sierra Nevada? Pages 233-241 in Cole, D.N., S.F. McCool, W.T. Borrie, and J. O'Loughlin (compilers). Wilderness science in a time of change conference – volume 5: wilderness ecosystems, threats, and management. Proceedings RMRS-P-15-VOL-5. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT.

[6] Cole, K.L., K.F Klick, and N.B. Pavlovic. 1992. Fire temperature monitoring during experimental burns at Indiana Dunes National Lakeshore. Natural Areas Journal 12:177-183.

[7] Cole, K.L., and R.S. Taylor. 1995. Past and current trends of change in a dune prairie/oak savanna reconstructed through a multiple-scale history. Journal of Vegetation Science 6:399-410.

[8] Dees, C.S., J.D. Clark, and F.T. Van Manen. 2001. Florida panther habitat use in response to prescribed fire. Journal of Wildlife Management 65:141-147.

[9] Falk, D.A., E.K. Heyerdahl, P.M. Brown, C. Farris, P.Z. Fulé, D. McKenzie, T.W. Swetnam, A.H. Taylor, and M.L. Van Horne. 2011. Multi-scale controls of historical forest-fire regimes: new insights from fire-scar networks. Frontiers in Ecology and the Environment 9:446-454.

[10] Flematti, G.R., M.T. Waters, A. Scaffidi, D.J. Merritt, E.L. Ghisalberti, K.W. Dixon, and S.M. Smith. 2013. Karrikin and cyanohydrin smoke signals provide clues to new endogenous plant signaling compounds. Molecular Plant 6:29-37.

[11] Floyd, M.L., W.H. Romme, D.D. Hanna, M. Winterowd, D. Hanna, and J. Spence. 2008. Fire history of piñon-juniper woodlands on Navajo Point, Glen Canyon National Recreation Area. Natural Areas Journal 28:26-36.

[12] Fulé, P.Z., W.W. Covington, M.M. Moore, T.A. Heinlein, and A.E.M. Waltz. 2002. Natural variability in forests of the Grand Canyon, USA. Journal of Biogeography 29:31-47.

[13] Fulé, P.Z., T.A. Heinlein, W.W. Covington, and M.M. Moore. 2003. Assessing fire regimes on Grand Canyon landscapes with fire scar and fire record data. International Journal of Wildland Fire 12:129-145.

[14] Fulé, P.Z., W.W. Covington, M.T. Stoddard, and D. Bertolette. 2006. "Minimal-impact" restoration treatments have limited effects on forest structure and fuels at Grand Canyon, USA. Restoration Ecology 14:357-368.

[15] Grundel, R., and N.B. Pavlovic. 2007. Distinctiveness, use, and value of Midwestern oak savannas and woodlands as avian habitats. Auk 124:969-985.

[16] Grundel, R., R.P. Jean, K.J. Frohnapple, G.A. Glowacki, P.E. Scott, and N.B. Pavlovic. 2010. Floral and nesting resources, habitat structure, and fire influence bee distribution across an open-forest gradient. Ecological Applications 20:1678-1692.

[17] Higuera, P.E., L.B. Brubaker, P.M. Anderson, F.S. Hu, and T.A. Brown. 2009. Vegetation mediated the impacts of postglacial climate change on fire regimes in the south-central Brooks Range, Alaska. Ecological Monographs 79:201-219.

[18] Higuera, P.E., M.L. Chipman, J.L. Barnes, M.A. Urban, and F.S. Hu. 2011. Variability of tundra fire regimes in Arctic Alaska: millennial-scale patterns and ecological implications. Ecological Applications 21:3211-3226.

[19] Huisinga, K.D., D.C. Laughlin, P.Z. Fulé , J.D. Springer, and C.M. McGlone. 2005. Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. Journal of the Torrey Botanical Society 132:590-601.

[20] Keeley, J.E. 2007. Chaparral and fire. Fremontia 35:16-21.

[21] Kilgore, B.M., and H.H. Biswell. 1971. Seedling germination following fire in a giant sequoia forest. California Agriculture 25:8-10.

[22] Kilgore, B.M. 1976. Fire management in the national parks: an overview. Tall Timbers Fire Ecology Conference Proceedings 14:45-57.

[23] Nuzzo, V.A. 1986. Extent and status of Midwest oak savanna: presettlement and 1985. Natural Areas Journal 6:6-36.

[24] Plummer, F.G. 1912. Forest fires: their causes, extent and effects, with a summary of recorded destruction and loss. Bulletin 117. U.S. Department of Agriculture, Forest Service. U.S. Government Printing Office, Washington, D.C. 39 pages.

[25] Pyne, S.J. 2008. America's fires: a historical context for policy and practice. Forest History Society, Durham, NC. 93 pages.

[26] Romme, W.H., and D.G. Despain. 1989. Historical perspectives on the Yellowstone fires of 1988: a reconstruction of prehistoric fire history reveals that comparable fires occurred in the early 1700s. Bioscience 39:695-699.
[27] Romme, W.H., M.S. Boyce, R.E. Gresswell, E.H. Merrill, G.W. Minshall, C. Whitlock, and M.G. Turner. 2011. Twenty years after the 1988 Yellowstone fires: lessons about disturbance and ecosystems. Ecosystems 14:1196-1215.
[28] Schmidt, K.M., J.P. Menakis, C.C. Hardy, W.J. Hann, and D.L. Bunnell. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. General Technical Report RMRS-GTR-87. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 41 pages.
[29] Schoennagel, T., and C.R. Nelson. 2011. Restoration relevance of recent National Fire Plan treatments in forests of the western United States. Frontiers in Ecology and the Environment 9:271-277.
[30] Slocum, M.G., W.J. Platt, B. Beckage, B. Panko, and J.B. Lushine. 2007. Decoupling natural and anthropogenic fire regimes: a case study in Everglades National Park, Florida. Natural Areas Journal 27:41-55.
[31] Stephenson, N.L. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. Ecological Applications 9:1253-1265.
[32] Suarez, F., D. Binkley, M.W. Kaye, and R. Stottlemyer. 1999. Expansion of forest stands into tundra in the Noatak National Preserve, northwest Alaska. Ecoscience 6:465-470.
[33] Sugihara, N.G., J.W. van Wagtendonk, and J. Fites-Kaufman. 2007. Fire as an ecological process. Pages 58-74 in Sugihara, N.G., J.W. van Wagtendonk, K.E. Shaffer, J. Fites-Kaufman, and A.E. Thode (editors). Fire in California's ecosystems. University of California Press, Berkeley. 596 pages.
[34] Swetnam, T.W., C.H. Baisan, A.C. Caprio, P.M. Brown, R. Touchan, R.S. Anderson, and D.J. Hallett. 2009. Multi-millennial fire history of the giant forest, Sequoia National Park, California, USA. Fire Ecology 5:120-150.
[35] Tarnocai, C., J.G. Canadell, E.A.G. Schuur, P. Kuhry, G. Mazhitova, and S. Zimov. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. Global Biogeochemical Cycles 23:GB2023.
[36] Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography 27:87-104.
[37] Wade, D., J. Ewel, and R. Hofstetter. 1980. Fire in South Florida ecosystems. General Technical Report SE-17. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC. 125 pages.
[38] Witter, M., R.S. Taylor, and S. Davis. 2007. Vegetation response to wildfire and fire history in the Santa Monica Mountains, California. Pages 173-194 in Knapp, D.A. (editor). Flora and ecology of the Santa Monica Mountains: proceedings of the 32nd annual Southern California Botanists symposium. Southern California Botanists Special Publication No. 4. Southern California Botanists, Fullerton, CA.

4 Predator and prey

[1] Abrams, M.D., and S.E. Johnson. 2012. Long-term impacts of deer exclosures on mixed-oak forest composition at the Valley Forge National Historical Park, Pennsylvania, USA. Journal of the Torrey Botanical Society 139:167-180.
[2] Aubrey, K.B., K.S. McKelvey, and J.P. Copeland. 2007. Distribution and broadscale habitat relations of the wolverine in the contiguous United States. Journal of Wildlife Management 71:2147-2158.
[3] Baril, L.M., A.J. Hansen, R. Renkin, and R. Lawrence. 2011. Songbird response to increased willow (*Salix* spp.) growth in Yellowstone's northern range. Ecological Applications 21:2283-2296.
[4] Beschta, R.L., and W.J. Ripple. 2013. Are wolves saving Yellowstone's aspen? A landscape-level test of a behaviorally mediated trophic cascade: comment. Ecology 94:1420-1425.
[5] Côte, S.D., T.P. Rooney, J. Tremblay, C. Dussault, and D.M. Waller. 2004. Ecological impacts of deer overabundance. Annual Review of Ecology and Systematics 35:113-147.
[6] Currier, M.J.P. 1983. *Felis concolor*. Mammalian Species 200:1-7.
[7] Diefenbach, D.R., L.A. Hansen, R.J. Warren, M.J. Conroy, and M.G. Nelms. 2009. Restoration of bobcats to Cumberland Island, Georgia, USA: lessons learned and evidence for the role of bobcats as keystone predators. Pages 423-435 in Vargas, A., C. Breitenmoser, and U. Breitenmoser (editors). Iberian lynx ex situ conservation: an interdisciplinary approach. Fundación Biodiversidad, Madrid, Spain. 536 pages.
[8] Ellison, A.M., and N.J. Gotelli. 2009. Energetics and the evolution of carnivorous plants – Darwin's 'most wonderful plants in the world.' Journal of Experimental Botany 60:19-42.
[9] Estes, J.A., J. Terborgh, J.S. Brashares, et al. 2011. Trophic downgrading of planet Earth. Science 333:301-306.
[10] Gompper, M.E. 2002. Top carnivores in the suburbs? Ecological and conservation issues raised by colonization of northeastern North America by coyotes. BioScience 52:185-190.

REFERENCES

[11] Graber, D.M., and M. White. 1983. Black bear food habits in Yosemite National Park. International Conference on Bear Research and Management 5:1-10.

[12] Griggs, J.A., J.H. Rock, C.R. Webster, and M.A. Jenkins. 2006. Vegetative legacy of a protected deer herd in Cades Cove, Great Smoky Mountains National Park. Natural Areas Journal 26:126-136.

[13] Haig, S.M., J. D'Elia, C. Eagles-Smith, J.M. Fair, J. Gervais, G. Herring, J.W. Rivers, and J.H. Schulz. 2014. The persistent problem of lead poisoning in birds from ammunition and fishing tackle. Condor 116:408-428.

[14] Hinton, J.W., M.J. Chamberlain, and D.R. Rabon. 2013. Red wolf (*Canis rufus*) recovery: a review with suggestions for future research. Animals 3:722-744.

[15] Hornocker, M.G., and H.S. Hash. 1981. Ecology of the wolverine in northwestern Montana. Canadian Journal of Zoology 59:1286–1301.

[16] Hristienko, H., and J.E. McDonald. 2007. Going into the 21st century: a perspective on trends and controversies in the management of the American black bear. Ursus 18:72-88.

[17] Hurley, P.M., and D. Flaspohler. 2005. An assessment of long-term biodiversity recovery from intense and sustained deer browse on North Manitou Island, Sleeping Bear Dunes National Lakeshore. Pages 85-108 in Cook, W. (editor). Proceedings: forest and wildlife – striving for balance. Society of American Foresters, St. Ignace, MI. 289 pages.

[18] Kay, C.E. 1995. Browsing by native ungulates: effects on shrub and seed production in the greater Yellowstone ecosystem. Pages 310-320 in Roundy, B.A., E.D. McArthur, J.S. Haley, and D.K. Mann (compilers). Proceedings: wildland shrub and arid land restoration symposium. General Technical Report INT-GTR-315. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT. 384 pages.

[19] Kendall, K.C., J.B. Stetz, D.A. Roon, L.P. Waits, J.B. Boulanger, and D. Paetkau. 2008. Grizzly bear density in Glacier National Park, Montana. Journal of Wildlife Management 72:1693-1705.

[20] Kochert, M.N., K. Steenhof, C.L. McIntyre, and E.H. Craig. 2002. Golden eagle (*Aquila chrysaetos*). No. 684 in Poole, A., and F. Gill (editors). The Birds of North America. The Birds of North America, Inc., Philadelphia, PA.

[21] Larivière, S., and L.R. Walton. 1997. *Lynx rufus*. Mammalian Species 563:1-8.

[22] Larivière, S. 2001. *Ursus americanus*. Mammalian Species 647:1-11.

[23] LaRue, M.A., C.K. Nielsen, M. Dowling, K. Miller, B. Wilson, H. Shaw, and C.R. Anderson. 2012. Cougars are recolonizing the Midwest: analysis of cougar confirmations during 1990-2008. Journal of Wildlife Management 76:1364-1369.

[24] Marshall, K.N., N.T. Hobbs, and D.J. Cooper. 2013. Stream hydrology limits recovery of riparian ecosystems after wolf reintroduction. Proceedings of the Royal Society B 280:article20122977.

[25] Mattson, D.J., and T. Merrill. 2002. Extirpations of grizzly bears in the contiguous United States, 1850-2000. Conservation Biology 16:1123-1136.

[26] Mech, L.D. 1974. *Canis lupus*. Mammalian Species 37:1-6.

[27] Miller, S.D., B.N. McLellan, and A.E. Derocher. 2013. Conservation and management of large carnivores in North America. International Journal of Environmental Studies 70:383-398.

[28] Montgomery, R.A., J.A. Vucetich, G.J. Roloff, J.K. Bump, and R.O. Peterson. 2014. Where wolves kill moose: the influence of prey life history dynamics on the landscape ecology of predation. PLoS ONE 9:e91414.

[29] Nelson, M.P., J.A. Vucetich, R.O. Peterson, and L.M. Vucetich. 2011. The Isle Royale wolf-moose project (1958-present) and the wonder of long-term ecological research. Endeavour 35:30-38.

[30] Onorato, D.P., E.C. Hellgren, F.S. Mitchell, and J.R. Skiles. 2003. Home range and habitat use of American black bears on a desert montane island in Texas. Ursus 14:120-129.

[31] Pierce, B.M., and V.C. Bleich. 2003. Mountain lion (*Puma concolor*). Pages 744-757 in Fieldhamer, G.A., B.C. Thompson, and J.A. Chapman (editors). Wild mammals of North America: biology, management, and conservation. John Hopkins University Press, Baltimore, MD. 1,216 pages.

[32] Rausch, R.A., and A.M. Pearson. 1972. Notes on the wolverine in Alaska and the Yukon Territory. Journal of Wildlife Management 36:249-268.

[33] Riley, S.P.D. 2006. Spatial Ecology of bobcats and gray foxes in urban and rural zones of a national park. Journal of Wildlife Management 70:1425-1435.

[34] Ripple W.J., and R.L. Beschta. 2003. Wolf reintroduction, predation risk, and cottonwood recovery in Yellowstone National Park. Forest Ecology and Management 184:299-313.

[35] Ripple, W.J., and R.L. Beschta. 2006. Linking a cougar decline, trophic cascade, and catastrophic regime shift in Zion National Park. Biological Conservation 133:397-408.

[36] Roberts, N.M., and S.M. Crimmins. 2010. Bobcat population status and management in North America: evidence of large-scale population increase. Journal of Fish and Wildlife Management 1:169-174.

[37] Smith, D., D. Stahler, E. Stahler, M. Metz, K. Quimby, R. McIntyre, C. Ruhl, and M. McDevitt. 2014. Yellowstone National Park wolf project annual report 2013. National Park Service, Yellowstone Center for Resources, Mammoth Hot Springs, WY. 23 pages.
[38] Thiemann, J.A., C.R. Webster, M.A. Jenkins, P.M. Hurley, J.H. Rock, and P.S. White. 2010. Herbaceous-layer impoverishment in a post-agricultural southern Appalachian landscape. American Midland Naturalist 162:148-168.
[39] White, L.A., and S.D. Gehrt. 2009. Coyote attacks on humans in the United States and Canada. Human Dimensions of Wildlife 14:419-432.
[40] Winnie, J. 2014. Predation risk, elk, and aspen: a reply. Ecology 95:2671-2674.

5 Non-native plants

[1] Abella, S.R. 2008. A systematic review of wild burro grazing effects on Mojave Desert vegetation, USA. Environmental Management 41:809-819.
[2] Abella, S.R., J.E. Spencer, J. Hoines, and C. Nazarchyk. 2009. Assessing an exotic plant surveying program in the Mojave Desert, Clark County, Nevada, USA. Environmental Monitoring and Assessment 151:221-230.
[3] Abella, S.R. 2010. Disturbance and plant succession in the Mojave and Sonoran Deserts of the American Southwest. International Journal of Environmental Research and Public Health 7:1248-1284.
[4] Abella, S.R., D.J. Craig, S.D. Smith, and A.C. Newton. 2012. Identifying native vegetation for reducing exotic species during the restoration of desert ecosystems. Restoration Ecology 20:781-787.
[5] Abella, S.R., and A. Tendick. 2013. Distribution of exotic plant species and relationship to vegetation type at Bryce Canyon National Park, USA. Landscape and Urban Planning 120:48-58.
[6] Abella, S.R., L.P. Chiquoine, and D.M. Backer. 2013. Soil, vegetation, and seed bank of a Sonoran Desert ecosystem along an exotic plant (*Pennisetum ciliare*) treatment gradient. Environmental Management 52:946-957.
[7] Abella, S.R. 2014. Effectiveness of exotic plant treatments on National Park Service lands in the United States. Invasive Plant Science and Management 7:147-163.
[8] Abella, S.R., N.A. Fisichelli, S.M. Schmid, T.M. Embrey, D.L. Hughson, and J. Cipra. 2015. Status and management of non-native plant invasion in three of the largest national parks in the United States. Nature Conservation 10:71-94.
[9] Allen, J.A., C.S. Brown, and T.J. Stohlgren. 2009. Non-native plant invasions of United States national parks. Biological Invasions 11:2195-2207.
[10] Ames, A. 1907. The May-flower and her log: July 15, 1620-May 6, 1621, chiefly from original sources. Houghton, Mifflin and Company, Boston, MA. 385 pages.
[11] Barrows, C.W., and M.L. Murphy-Mariscal. 2012. Modeling impacts of climate change on Joshua trees at their southern boundary: how scale impacts predictions. Biological Conservation 152:29-36.
[12] Baskett, C.A., S.M. Emery, and J.A. Rudgers. 2011. Pollinator visits to threatened species are restored following invasive plant removal. International Journal of Plant Sciences 172:411-422.
[13] Bergquist, E., P. Evangelista, T.J. Stohlgren, and N. Alley. 2007. Invasive species and coal bed methane development in the Powder River Basin, Wyoming. Environmental Monitoring and Assessment 128:381-394.
[14] Crall, A.W., M. Renz, B.J. Panke, G.J. Newman, C. Chapin, J. Graham, and C. Bargeron. 2012. Developing cost-effective early detection networks for regional invasions. Biological Invasions 14:2461-2469.
[15] DeKeyser, S., G. Clambey, K. Krabbenhoft, and J. Ostendorf. 2009. Are changes in species composition on central North Dakota rangelands due to non-use management? Rangelands 31:16-19.
[16] Foy, C.L., D.R. Forney, and W.E. Cooley. 1983. History of weed introductions. Pages 65-92 in Wilson, C.L., and C.L. Graham (editors). Exotic plant pests and North American agriculture. Academic Press, New York. 522 pages.
[17] Fulling, E.H. 1944. Thomas Jefferson: his interest in plant life as revealed in his writings. Bulletin of the Torrey Botanical Club 71:563-598.
[18] Gilbert, B., and J.M. Levine. 2013. Plant invasions and extinction debts. Proceedings of the National Academy of Sciences 110:1744-1749.
[19] Hegeman, E.E., B.G. Dickson, and L.J. Zachmann. 2014. Probabilistic models of fire occurrence across National Park Service units within the Mojave Desert Network, USA. Landscape Ecology 29:1587-1600.
[20] Hussey, M.A., and B.L. Burson. 2005. Registration of 'Frio' buffelgrass. Crop Science 45:411-412.

[21] Jenkins, M.A., and K.D. Johnson. 2008. Exotic plant species invasion and control in Great Smoky Mountains National Park, USA. Pages 295-322 in Kumar Kohli, R., S. Jose, H. Pal Singh, and D. Rani Batish (editors). Invasive plants and forest ecosystems. CRC Press, New York. 456 pages.
[22] Lesica, P., D. Ahlenslager, and J. Desanto. 1993. New vascular plant records and the increase of exotic plants in Glacier National Park, Montana. Madroño 40:126-131.
[23] Lodge, D.M., S. Williams, H.J. MacIsaac, K.R. Hayes, B. Leung, S. Reichard, R.N. Mack, P.B. Moyle, M. Smith, D.A. Andow, J.T. Carlton, and A. McMichael. 2006. Biological invasions: recommendations for US policy and management. Ecological Applications 16:2035-2054.
[24] Love, J.P., and J.T. Anderson. 2009. Seasonal effects of four control methods on the invasive Morrow's honeysuckle (*Lonicera morrowii*) and initial responses of understory plants in a southwestern Pennsylvania old field. Restoration Ecology 17:549-559.
[25] Mack, R.N., and M. Erneberg. 2002. The United States naturalized flora: largely the product of deliberate introductions. Annals of the Missouri Botanical Garden 89:176-189.
[26] Minnich, R.A. 1995. Wildland fire and early postfire succession in Joshua tree woodland and blackbrush scrub of the Mojave Desert of California. San Bernardino County Museum Association Quarterly 42:99-106.
[27] Morales, C.L., and A. Traveset. 2009. A meta-analysis of impacts of alien vs. native plants on pollinator visitation and reproductive success of co-flowering native plants. Ecology Letters 12:716-728.
[28] Moyes, A.B., M.S. Witter, and J.A. Gamon. 2005. Restoration of native perennials in a California annual grassland after prescribed spring burning and solarization. Restoration Ecology 13:659-666.
[29] Nagata, K.M. 1985. Early plant introductions in Hawaii. Hawaiian Journal of History 19:35-61.
[30] Oftedal, O.T., S. Hillard, and D.J. Morafka. 2002. Selective spring foraging by juvenile desert tortoises (*Gopherus agassizii*) in the Mojave Desert: evidence of an adaptive nutritional strategy. Chelonian Conservation and Biology 4:341-352.
[31] Pyšek, P., V. Jarošík, P.E. Hulme, J. Pergl, M. Hejda, U. Schaffner, and M. Vilà. 2012. A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment. Global Change Biology 18:1725-1737.
[32] Ransom, C.V., S.D. Christensen, K.A. Edvarchuk, and T. Naumann. 2012. A reinventory of invasive weed species in Dinosaur National Monument to determine management effectiveness. Invasive Plant Science and Management 5:300-309.
[33] Rejmánek, M., and M.J. Pitcairn. 2002. When is eradication of pest plants a realisic goal? Pages 249-253 in Veitch, C.R., and M.N. Clout (editors). Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland, and Cambridge, UK. 414 pages.
[34] Robinson, T.W. 1965. Introduction, spread and areal extent of saltcedar (*Tamarix*) in the western states. U.S. Geological Survey Professional Paper 491-A. U.S. Government Printing Office, Washington, D.C. 12 pages.
[35] Sher, A., and M.F. Quigley (editors). 2013. *Tamarix*: a case study of ecological change in the American Southwest. Oxford University Press, New York. 488 pages.
[36] Stewart-Wade, S.M., S. Neumann, L.L. Collins, and G.J. Boland. 2002. The biology of Canadian weeds. 117. *Taraxacum officinale* G.H. Weber ex Wiggers. Canadian Journal of Plant Science 82:825-853.
[37] Sutter, R.D., T.E. Govus, R. Lyons Smyth, C. Nordman, M. Pyne, and T. Hogan. 2011. Monitoring change in a central U.S. calcareous glade: resampling transects established in 1993. Natural Areas Journal 31:163-172.
[38] Theoharides, K.A., and J.S. Dukes. 2007. Plant invasion across space and time: factors affecting nonindigenous species success during four stages of invasion. New Phytologist 176:256-273.
[39] Trammell, M.A., and J.L. Butler. 1995. Effects of exotic plants on native ungulate use of habitat. Journal of Wildlife Management 59:808-816.
[40] Whipple, J.J. 2001. Annotated checklist of exotic vascular plants in Yellowstone National Park. Western North American Naturalist 61:336-346.

6 Non-native fauna

[1] Abella, S.R. 2008. A systematic review of wild burro grazing effects on Mojave Desert vegetation, USA. Environmental Management 41:809-819.
[2] Beever, E.A. 2013. *Equus caballus*. Invasive species compendium. Commonwealth Agricultural Bureaux International (CABI), Oxfordshire, United Kingdom.

REFERENCES

[3] Bevins, S.N., K. Pedersen, M.W. Lutman, T. Gidlewski, and T.J. Deliberto. 2014. Consequences associated with the recent range expansion of nonnative feral swine. BioScience 64:291-299.

[4] Blair, R.B. 2001. Creating a homogeneous avifauna. Pages 459-486 in Marzluff, J.M., R. Bowman, and R. Donnelly (editors). Avian ecology and conservation in an urbanizing world. Kluwer Academic Publishers, Norwell, MA. 585 pages.

[5] Bratton, S.P. 1974. The effect of the European wild boar (*Sus scrofa*) on high elevation vernal flora in the Great Smoky Mountains National Park. Bulletin of the Torrey Botanical Club 101:198-206.

[6] Cole, F.R., A.C. Medeiros, L.L. Loope, and W.H. Zuehlke. 1992. Effects of the Argentine ant on arthropod fauna of Hawaiian high-elevation shrubland. Ecology 73:1313-1322.

[7] Dorcas, M.E., J.D. Willson, R.N. Reed, et al. 2012. Severe mammal declines coincide with proliferation of invasive Burmese pythons in Everglades National Park. Proceedings of the National Academy of Sciences 109:2418-2422.

[8] Dove, C.J., R.W. Snow, M.R. Rochford, and F.J. Mazzotti. 2011. Birds consumed by the invasive Burmese python (*Python molurus bivittatus*) in Everglades National Park, Florida, USA. Wilson Journal of Ornithology 123:126-131.

[9] Fisichelli, N.A., L.E. Frelich, P.B. Reich, and N. Eisenhauer. 2013. Linking direct and indirect pathways mediating earthworms, deer, and understory composition in Great Lakes forests. Biological Invasions 15:1057-1066.

[10] Forbush, E.H. 1916. The starling. Circular no. 45. The Commonwealth of Massachusetts, State Board of Agriculture. Wright and Potter Printing Co., State Printers, Boston, MA. 23 pages.

[11] Frelich, L.E., C.M. Hale, S. Scheu, A.R. Holdsworth, L. Henegham, P.J. Bohlen, and P.B. Reich. 2006. Earthworm invasion into previously earthworm-free temperate and boreal forests. Biological Invasions 8:1235-1245.

[12] Gurevitch, J., and D.K. Padilla. 2004. Are invasive species a major cause of extinctions? Trends in Ecology and Evolution 19:470-474.

[13] Hale, C.M., and G.E. Host. 2005. Assessing the impacts of European earthworm invasions in beech-maple hardwood and aspen-fir boreal forests of the western Great Lakes region. Great Lakes Inventory and Monitoring Network Report GLKN/2005/11. National Park Service, Ashland, WI. 35 pages.

[14] Hendrix, P.F., and P.J. Bohlen. 2002. Exotic earthworm invasions in North America: ecological and policy implications. BioScience 52:801-811.

[15] Krushelnycky, P.D., L.L. Loope, and N.J. Reimer. 2005. The ecology, policy, and management of ants in Hawaii. Proceedings of the Hawaiian Entomological Society 37:1-25.

[16] Krushelnycky, P.D., and R.G. Gillespie. 2008. Compositional and functional stability of arthropod communities in the face of ant invasions. Ecological Applications 18:1547-1562.

[17] Krushelnycky, P.D., W. Haines, L. Loope, and E. Van Gelder. 2011. The Haleakala Argentine ant project: a synthesis of past research and prospects for the future. Pacific Cooperative Studies Unit technical report 173. University of Hawai'i at Manoa, Honolulu, HI. 127 pages.

[18] Lobe, J.W., M.A. Callaham, P.F. Hendrix, and J.L. Hanula. 2014. Removal of an invasive shrub (Chinese privet: *Ligustrum sinense* Lour) reduces exotic earthworm abundance and promotes recovery of native North American earthworms. Applied Soil Ecology 83:133-139.

[19] Loss, S.R., and R.B. Blair. 2011. Reduced density and nest survival of ground-nesting songbirds relative to earthworm invasions in northern hardwood forests. Conservation Biology 25:983-992.

[20] McCann, B.E., and D.K. Garcelon. 2008. Eradication of feral pigs from Pinnacles National Monument. Journal of Wildlife Management 72:1287-1295.

[21] McCullough, D.G., T.T. Work, J.F. Cavey, A.M. Liebhold, and D. Marshall. 2006. Interceptions of nonindigenous plant pests at US ports of entry and border crossings over a 17-year period. Biological Invasions 8:611-630.

[22] McKnight, T.L. 1958. The feral burro in the United States: distribution and problems. Journal of Wildlife Management 22:163-178.

[23] National Park Service. 2006. Management policies 2006. U.S. Government Printing Office, Washington, D.C. 168 pages.

[24] Orlando, L., A. Ginolhac, G. Zhang, et al. 2013. Recalibrating *Equus* evolution using the genome sequence of an early Middle Pleistocene horse. Nature 499:74-78.

[25] Peine, J.D., and J.A. Farmer. 1990. Wild hog management program at Great Smoky Mountains National Park. Pages 221-227 in Davis, L.R., and R.E. Marsh (editors). Proceedings of the 14th vertebrate pest conference. University of California, Davis.

[26] Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273-288.

[27] Randall, J.M., K.R. Faulkner, C. Boser, C. Cory, P. Power, L.A. Vermeer, L. Lozier, and S.A. Morrison. 2011. Argentine ants on Santa Cruz Island, California: conservation issues and management options. Pages 108-113 in Veitch, C.R., M.N. Clout, and D.R. Towns (editors). Island invasives: eradication and management. IUCN, Switzerland.
[28] Roemer, G.W., C.J. Donlan, and F. Courchamp. 2002. Golden eagles, feral pigs, and insular carnivores: how exotic species turn native predators into prey. Proceedings of the National Academy of Sciences 99:791-796.
[29] Singer, F.J. 1981. Wild pig populations in the national parks. Environmental Management 5:263-270.
[30] Snyder, B.A., M.A. Callaham, and P.F. Hendrix. 2011. Spatial variability of an invasive earthworm (*Amynthas agrestis*) population and potential impacts on soil chracteristics and millipedes in the Great Smoky Mountains National Park, USA. Biological Invasions 13:349-358.
[31] West, B.C., A.L. Cooper, and J.B. Armstrong. 2009. Managing wild pigs: a technical guide. Human-Wildlife Interactions Monograph 1:1-55.

7 Forests

[1] Abella, S.R. 2013. Great Basin montane forests. Pages 607-609 in Howarth, R.W. (editor). Biomes and ecosystems: an encyclopedia. Salem Press, Hackensack, NJ. 1440 pages.
[2] Abella, S.R. 2014. Impacts and management of hemlock woolly adelgid in national parks of the eastern United States. Southeastern Naturalist 13:16-45.
[3] Aukema, J.E., B. Leung, K. Kovacs, et al. 2011. Economic impacts of non-native forest insects in the continental United States. PLoS ONE 6:e24587.
[4] Bartram, W. 1791. Travels through North and South Carolina, Georgia, East and West Florida, the Cherokee Country, the extensive territories of the Muscogulges or Creek Confederacy, and the country of the Chactaws. Containing an account of the soil and natural productions of those regions; together with observations on the manners of the Indians. James and Johnson, Philadelphia, PA. 520 pages.
[5] Blozan, W. 1994. The importance of increment core samples and disturbance history in the evaluation of old-growth forests in Great Smoky Mountains National Park. Natural Areas Journal 14:140-142.
[6] Bolsinger, C.L., and K.L. Waddell. 1993. Area of old-growth forests in California, Oregon, and Washington. Resource Bulletin PNW-RB-197. U.S. Forest Service, Pacific Northwest Research Station, Portland, OR. 26 pages.
[7] Brown, P.M., and C.H. Sieg. 1996. Fire history in interior ponderosa pine communities of the Black Hills, South Dakota, USA. International Journal of Wildland Fire 6:97-105.
[8] Currey, D.R. 1965. An ancient bristlecone pine stand in eastern Nevada. Ecology 46:564-566.
[9] Defebaugh, J.E. 1907. History of the lumber industry of America. The American Lumberman, Chicago, IL. 655 pages.
[10] Diskin, M., M.E. Rocca, K.N. Nelson, C.F. Aoki, and W.H. Romme. 2011. Forest developmental trajectories in mountain pine beetle disturbed forests of Rocky Mountain National Park, Colorado. Canadian Journal of Forest Research 41:782-792.
[11] Fisichelli, N.A., S.R. Abella, M. Peters, and F.J. Krist. 2014. Climate, trees, pests, and weeds: change, uncertainty, and biotic stressors in eastern U.S. national park forests. Forest Ecology and Management 327:31-39.
[12] Floyd, M.L., W.H. Romme, M.E. Rocca, D.P. Hanna, and D.D. Hanna. 2015. Structural and regenerative changes in old-growth piñon-juniper woodlands following drought-induced mortality. Forest Ecology and Management 341:18-29.
[13] Franklin, J.F., W.H. Moir, M.A. Hemstrom, S.E. Greene, and B.G. Smith. 1988. The forest communities of Mount Rainier National Park. Scientific Monograph Series No. 19. National Park Service, Washington, D.C. 194 pages.
[14] Grissino-Mayer, H.D., T.W. Swetnam, and R.K. Adams. 1997. The rare, old-aged conifers of El Malpais – their role in understanding climatic change in the American Southwest. New Mexico Bureau of Mines and Mineral Resources Bulletin 156:155-161.
[15] Hadley, K.S., and T.T. Veblen. 1993. Stand response to western spruce budworm and Douglas-fir bark beetle outbreaks, Colorado Front Range. Canadian Journal of Forest Research 23:479-491.
[16] Harmon, M.E. 1989. Effects of bark fragmentation on plant succession on conifer logs in the *Picea-Tsuga* forests of Olympic National Park, Washington. American Midland Naturalist 121:112-124.
[17] Howard, J.L., and R.M. Westby. 2013. U.S. timber production, trade, consumption and price statistics 1965-2011. Research Paper FPL-RP-676. U.S. Forest Service, Forest Products Laboratory, Madison, WI. 91 pages.
[18] Jenkins, M.A., and P.S. White. 2002. *Cornus florida* L. mortality and understory composition changes in western Great Smoky Mountains National Park. Journal of the Torrey Botanical Society 129:194-206.
[19] Jones, R.H. 1997. Status and habitat of big trees in Congaree Swamp National Monument. Castanea 62:22-31.

[20] King, C.B., and R. Muzika. 2014. Historic fire and canopy disturbance dynamics in an oak-pine (*Quercus-Pinus*) forest of the Missouri Ozarks (1624-2010). Castanea 79:78-87.
[21] Koch, F.H., D. Yemshanov, M. Colunga-Garcia, R.D. Magarey, and W.D. Smith. 2011. Potential establishment of alien-invasive forest insect species in the United States: where and how many? Biological Invasions 13:969-985.
[22] Liebhold, A.M., D.G. McCullough, L.M. Blackburn, S.J. Frankel, B. Von Holle, and J.E. Aukema. 2013. A highly aggregated geographical distribution of forest pest invasions in the USA. Diversity and Distributions 19:1208-1216.
[23] Morin, R.S., A.M. Liebhold, and K.W. Gottschalk. 2009. Anisotropic spread of hemlock woolly adelgid in the eastern United States. Biological Invasions 11:2341-2350.
[24] National Park Service. 2006. Forest management plan and environmental assessment for Marsh-Billings-Rockefeller National Historical Park. Marsh-Billings-Rockefeller National Historical Park, Woodstock, VT. 234 pages.
[25] Sherald, J.L., T.M. Stidham, J.M. Hadidian, and J.E. Hoeldtke. 1996. Progression of the dogwood anthracnose epidemic and the status of flowering dogwood in Catoctin Mountain Park. Plant Disease 80:310-312.
[26] Teraoka, J.R. 2012. Forest restoration at Redwood National Park: a case study of an emerging program. Pages 561-569 in Standiford, R.B., T.J. Weller, D.D. Piirto, and J.D. Stuart (technical coordinators). Proceedings of the coast redwood forests in a changing California: a symposium for scientists and managers. General Technical Report PSW-GTR-238. U.S. Forest Service, Pacific Southwest Research Station, Albany, CA. 675 pages.
[27] U.S. Forest Service. 2011. National report on sustainable forests – 2010. U.S. Government Printing Office, Washington, D.C. 214 pages.

8 Freshwater

[1] Baker, B.W., and E.P. Hill. 2003. Beaver (*Castor canadensis*). Pages 288-310 in Feldhamer, G.A., B.C. Thompson, and J.A. Chapman (editors). Wild mammals of North America: biology, management, and conservation. Johns Hopkins University Press, Baltimore, MD. 1,232 pages.
[2] Carpenter, S.R., E.H. Stanley, and M.J. Vander Zanden. 2013. State of the world's freshwater ecosystems: physical, chemical, and biological changes. Annual Review of Environment and Resources 36:75-99.
[3] Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. Report FWS/OBS-79/31. U.S. Fish and Wildlife Service, Washington, D.C. 131 pages.
[4] Dahl, T.E., and G.J. Allord. 1996. History of wetlands in the conterminous United States. Pages 19-26 in Fretwell, J.D., J.S. Williams, and P.J. Redman (compilers). National water summary on wetland resources. Water Supply Paper 2425. U.S. Geological Survey, Reston, VA. 431 pages.
[5] Dahl, T.E. 2011. Status and trends of wetlands in the conterminous United States 2004 to 2009. U.S. Fish and Wildlife Service, Washington, D.C. 108 pages.
[6] DeSimone, L.A., P.B. McMahon, and M.R. Rosen. 2014. Water quality in principal aquifers of the United States, 1991-2010. Circular 1360. U.S. Geological Survey, Reston, VA. 150 pages.
[7] Duda, J.J., J.E. Freilich, and E.G. Schreiner. 2008. Baseline studies in the Elwha River ecosystem prior to dam removal: introduction to the special issue. Northwest Science 82(sp1):1-12.
[8] Engel, E.C., S.R. Abella, and K.L. Chittick. 2014. Plant colonization and soil properties on newly exposed shoreline during drawdown of Lake Mead, Mojave Desert. Lake and Reservoir Management 30:105-114.
[9] Graf, W.L. 2001. Damage control: restoring the physical integrity of America's rivers. Annals of the Association of American Geographers 91:1-27.
[10] Hart, D.D., T.E. Johnson, K.L. Bushaw-Newton, R.J. Horwitz, A.T. Bednarek, D.F. Charles, D.A. Kreeger, and D.J. Velinsky. 2002. Dam removal: challenges and opportunities for ecological research and river restoration. BioScience 52:669-681.
[11] Johnston, C.A., and R.J. Naiman. 1990. The use of a geographic information system to analyze long-term landscape alteration by beaver. Landscape Ecology 4:5-19.
[12] Lave, R., and B. Lutz. 2014. Hydraulic fracturing: a critical physical geography review. Geography Compass 8:739-754.
[13] Lovett, R.A. 2014. Rivers on the run. Nature 511:521-523.
[14] Madej, M.A., J. Seney, and P. van Mantgem. 2013. Effects of road decommissioning on carbon stocks, losses, and emissions in north coastal California. Restoration Ecology 21:439-446.
[15] Maupin, M.A., J.F. Kenny, S.S. Hutson, J.K. Lovelace, N.L. Barber, and K.S. Linsey. 2014. Estimated use of water in the United States in 2010. Circular 1405. U.S. Geological Survey, Reston, VA. 56 pages.

[16] Naiman, R.J., C.A. Johnston, and J.C. Kelley. 1988. Alteration of North American streams by beaver. BioScience 38:753-762.
[17] Patterson, L., and D.J. Cooper. 2007. The use of hydrological and ecological indicators for the restoration of drainage ditches and water diversions in a mountain fen, Cascade Range, California. Wetlands 27:290-304.
[18] Pess, G.R., M.L. McHenry, T.J. Beechie, and J. Davies. 2008. Biological impacts of the Elwha River dams and potential salmonid responses to dam removal. Northwest Science 82(sp1):72-90.
[19] Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47:769-784.
[20] Popular Mechanics. 2006. Tearing down the Elwha River Dam.
[21] Rosen, M.R., K. Turner, S.L. Goodbred, and J.M. Miller (editors). 2012. A synthesis of aquatic science for management of Lakes Mead and Mohave. Circular 1381. U.S. Geological Survey, Reston, VA. 168 pages.
[22] U.S. Energy Information Administration. 2015. Monthly energy review: March 2015. U.S. Department of Energy, Washington, D.C. 214 pages.
[23] U.S. Geological Survey. 1999. Ground water: general interest publication. U.S. Geological Survey, Reston, VA. 17 pages.
[24] Warrick, J.A., J.A. Bountry, A.E. East, et al. 2015. Large-scale dam removal on the Elwha River, Washington, USA: source-to-sink sediment budget and synthesis. Geomorphology (in press).
[25] Winter, T.C., J.W. Harvey, O.L. Franke, and W.M. Alley. 1998. Ground water and surface water: a single resource. Circular 1139. U.S. Geological Survey, Denver, CO. 79 pages.
[26] Wurster, F.C., D.J. Cooper, and W.E. Sanford. 2003. Stream/aquifer interactions at Great Sand Dunes National Monument, Colorado: influences on interdunal wetland disappearance. Journal of Hydrology 271:77-100.

9 Air

[1] Ackerman, L.K., A.R. Schwindt, S.L. Massey Simonich, D.C. Koch, T.F. Blett, C.B. Schreck, M.L Kent, and D.H. Landers. 2008. Atmospherically deposited PBDEs, pesticides, PCBs, and PAHs in western U.S. national park fish: concentrations and consumption guidelines. Environmental Science and Technology 42:2334-2341.
[2] Adams, M.J., D.A.W. Miller, E. Muths, et al. 2013. Trends in amphibian occupancy in the United States. PLoS ONE 8:e64347.
[3] Air Resources Division. 2013. Air quality in national parks: trends (2000-2009) and conditions (2005-2009). Natural Resource Report NPS/NRSS/ARD/NRR—2013/683. National Park Service, Denver, CO. 42 pages.
[4] Allen, E.B., L.E. Rao, R.J. Steers, A. Bytnerowicz, and M.E. Fenn. 2009. Impacts of atmospheric nitrogen deposition on vegetation and soils at Joshua Tree National Park. Pages 78-100 in Webb, R.H., L.F. Fenstermaker, J.S. Heaton, D.L. Hughson, E.V. McDonald, and D.M. Miller (editors). The Mojave Desert: ecosystem processes and sustainability. University of Nevada Press, Reno, NV. 481 pages.
[5] Bank, M.S., J.R. Burgess, D.C. Evers, and C.S. Loftin. 2007. Mercury contamination of biota from Acadia National Park, Maine: a review. Environmental Monitoring and Assessment 126:105-115.
[6] Bartholomay, G.A., R.T. Eckert, and K.T. Smith. 1997. Reductions in tree-ring widths of white pine following ozone exposure at Acadia National Park, Maine, U.S.A. Canadian Journal of Forest Research 27:361-368.
[7] Bennett, J.P., and C.M. Wetmore. 2005. Lichens of the U.S. national parks. Bryologist 108:544-553.
[8] Bhavsar, S.P., S.B. Gewurtz, D.J. McGoldrick, M.J. Keir, and S.M. Backus. 2010. Changes in mercury levels in Great Lakes fish between 1970s and 2007. Environmental Science and Technology 44:3273-3279.
[9] Binkley, D., and D. Richter. 1987. Nutrient cycles and H^+ budgets of forest ecosystems. Advances in Ecological Research 16:1-51.
[10] Blett, T., L. Geiser, and E. Porter. 2003. Air pollution-related lichen monitoring in national parks, forests, and refuges: guidelines for studies intended for regulatory and management purposes. National Park Service and U.S. Fish and Wildlife Service, Denver, CO, and U.S. Forest Service, Corvallis, OR. 25 pages.
[11] Burkey, K.O., H.S. Neufeld, L. Souza, A.H. Chappelka, and A.W. Davison. 2006. Seasonal profiles of leaf ascorbic acid content and redox state in ozone-sensitive wildflowers. Environmental Pollution 143:427-434.
[12] Burton, G.A., N. Basu, B.R. Ellis, K.E. Kapo, S. Entrekin, and K. Nadelhoffer. 2014. Hydraulic "fracking": are surface water impacts an ecological concern? Environmental Toxicology and Chemistry 33:1679-1689.
[13] Chappelka, A.H., and L.J. Samuelson. 1998. Ambient ozone effects on forest trees of the eastern United States: a review. New Phytologist 139:91-108.

REFERENCES

[14] Cosby, B.J., J.R. Webb, J.N. Galloway, and F.A. Deviney. 2006. Acidic deposition impacts on natural resources in Shenandoah National Park. Technical Report NPS/NER/NRTR-2006/066. National Park Service, Philadelphia, PA. 248 pages.

[15] Environmental Protection Agency. 2010. Our nation's air: status and trends through 2008. Publication EPA-454/R-09-002. U.S. Environmental Protection Agency, Research Triangle Park, NC. 49 pages.

[16] Environmental Protection Agency. 2015. Assessment of the potential impacts of hydraulic fracturing for oil and gas on drinking water resources: executive summary. U.S. Environmental Protection Agency, Washington, D.C.

[17] Evers, D.C., N.M. Burgess, L. Champoux, B. Hoskins, A. Major, W.M. Goodale, R.J. Taylor, R. Poppenga, and T. Daigle. 2005. Patterns and interpretation of mercury exposure in freshwater avian communities in northeastern North America. Ecotoxicology 14:193-221.

[18] Evers, D.C., J.G. Wiener, N. Basu, R.A. Bodaly, H.A. Morrison, and K.A. Williams. 2011. Mercury in the Great Lakes region: bioaccumulation, spatiotemporal patterns, ecological risks, and policy. Ecotoxicology 20:1487-1499.

[19] Fenn, M.E., J.S. Baron, E.B. Allen, et al. 2003. Ecological effects of nitrogen deposition in the western United States. BioScience 53:404-420.

[20] Flanagan Pritz, C.M., J.E. Schrlau, S.L. Massey Simonich, and T.F. Blett. 2014. Contaminants of emerging concern in fish from western U.S. and Alaskan national parks – spatial distribution and health thresholds. Journal of the American Water Resources Association 50:309-323.

[21] Gibbons, J.W., D.E. Scott, T.J. Ryan, et al. 2000. The global decline of reptiles, déjà vu amphibians. BioScience 50:653-666.

[22] Hageman, K.J., W.D. Hafner, D.H. Campbell, D.A. Jaffe, D.H. Landers, and S.L. Massey Simonich. 2010. Variability in pesticide deposition and source contributions to snowpack in western U.S. national parks. Environmental Science and Technology 44:4452-4458.

[23] Harrison, P.D., and M.I. Dyer. 1984. Lead in mule deer forage in Rocky Mountain National Park, Colorado. Journal of Wildlife Management 48:510-517.

[24] Hernandez, R.R., S.B. Easter, M.L. Murphy-Mariscal, et al. 2014. Environmental impacts of utility-scale solar energy. Renewable and Sustainable Energy Reviews 29:766-779.

[25] Jastram, J.D., C. Snyder, N. Hitt, and K.C. Rice. 2013. Synthesis and interpretation of surface water-quality and aquatic biota data collected in Shenandoah National Park, Virginia 1979-2010. Scientific Investigations Report 2013-5157. U.S. Geological Survey, Reston, VA. 77 pages.

[26] Karnosky, D.F., J.M. Skelly, K.E. Percy, and A.H. Chappelka. 2007. Perspectives regarding 50 years of research on effects of tropospheric ozone air pollution on US forests. Environmental Pollution 147:489-506.

[27] Longcore, J.R., T.A. Haines, and W.A. Halteman. 2007. Mercury in tree swallow food, eggs, bodies, and feathers at Acadia National Park, Maine, and an EPA superfund site, Ayer, Massachusetts. Environmental Monitoring and Assessment 126:129-143.

[28] McCune, B., J. Grenon, L.S. Mutch, and E.P. Martin. 2007. Lichens in relation to management issues in the Sierra Nevada national parks. Pacific Northwest Fungi 2:1-39.

[29] Monson, B.A. 2009. Trend reversal of mercury concentrations in piscivorous fish from Minnesota lakes: 1982-2006. Environmental Science and Technology 43:1750-1755.

[30] Moore, C.W., B. Zielinska, G. Pétron, and R.B. Jackson. 2014. Air impacts of increased natural gas acquisition, processing, and use: a critical review. Environmental Science and Technology 48:8349-8359.

[31] Peterson, D.L., M.J. Arbaugh, V.A. Wakefield, and P.R. Miller. 1987. Evidence of growth reduction in ozone-injured Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.) in Sequoia and Kings Canyon National Parks. Journal of the Air and Waste Management Association 37:906-912.

[32] Pittman, H.T., W.W. Bowerman, L.H. Grim, T.G. Grubb, and W.C. Bridges. 2011. Using nestling feathers to assess spatial and temporal concentrations of mercury in bald eagles at Voyageurs National Park, Minnesota, USA. Ecotoxicology 20:1626-1635.

[33] Sanders, W.B. 2001. Lichens: the interface between mycology and plant morphology. BioScience 51:1025-1035.

[34] Showman, R.E. 1997. Continuing lichen recolonization in the upper Ohio River Valley. Bryologist 100:478-481.

[35] Sotomayor, S., C. Barbieri, S. Wilhelm Stanis, F.X. Aguilar, and J.W. Smith. 2014. Motivations for recreating on farmlands, private forests, and state or national parks. Environmental Management 54:138-150.

[36] Spribille, T., S. Pérez-Ortega, T. Tønsberg, and D. Schirokauer. 2010. Lichens and lichenicolous fungi of the Klondike Gold Rush National Historic Park, Alaska, in a global biodiversity context. Bryologist 113:439-515.

[37] Wetmore, C.M. 1989. Lichens and air quality in Cuyahoga Valley National Recreation Area, Ohio. Bryologist 92:273-281.

[38] Wiener, J.G., B.C. Knights, M.B. Sandheinrich, J.D. Jeremiason, M.E. Brigham, D.R. Engstrom, L.G. Woodruff, W.F. Cannon, and S.J. Balogh. 2006. Mercury in soils, lakes, and fish in Voyageurs National Park (Minnesota): importance of atmospheric deposition and ecosystem factors. Environmental Science and Technology 40:6261-6268.

[39] Wolfe, A.P., J.S. Baron, and J.S. Cornett. 2001. Anthropogenic nitrogen deposition induces rapid ecological changes in alpine lakes of the Colorado Front Range (USA). Journal of Paleolimnology 25:1-7.

10 Coasts

[1] Baker, A.C., P.W. Glynn, and B. Riegl. 2008. Climate change and coral reef bleaching: an ecological assessment of long-term impacts, recovery trends and future outlook. Estuarine, Coastal and Shelf Science 80:435-471.

[2] Borja, Á., D.M. Dauer, M. Elliott, and C.A. Simenstad. 2010. Medium- and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuaries and Coasts 33:1249-1260.

[3] Botton, M.L., R.E. Loveland, J.T. Tanacredi, and T. Itow. 2006. Horseshoe crabs (*Limulus polyphemus*) in an urban estuary (Jamaica Bay, New York) and the potential for ecological restoration. Estuaries and Coasts 29:820-830.

[4] Braje, T.J., J.M. Erlandson, T.C. Rick, P.K. Dayton, and M.B.A. Hatch. 2009. Fishing from past to present: continuity and resilience of red abalone fisheries on the Channel Islands, California. Ecological Applications 19:906-919.

[5] Costanza, R., O. Pérez-Maqueo, M.L. Martinez, P. Sutton, S.J. Anderson, and K. Mulder. 2008. The value of coastal wetlands for hurricane protection. Ambio 37:241-248.

[6] Davis, G.E., P.L. Haaker, and D.V. Richards. 1998. The perilous condition of white abalone, *Haliotis sorenseni*. Journal of Shellfish Research 17:871–875.

[7] Day, J.W., D.F. Boesch, E.J. Clairain, et al. 2007. Restoration of the Mississippi Delta: lessons from Hurricanes Katrina and Rita. Science 315:1679-1684.

[8] Gornitz, V., S. Couch, and E.K. Hartig. 2002. Impacts of sea level rise in the New York City metropolitan area. Global and Planetary Changes 32:61-88.

[9] Hart, K.M., A.R. Sartain, and I. Fujisaki. 2015. Bahamas connection: residence areas selected by breeding female loggerheads tagged in Dry Tortugas National Park, USA. Animal Biotelemetry 3:(3).

[10] Hartig, E.K., V. Gornitz, A. Kolker, F. Mushacke, and D. Fallon. 2002. Anthropogenic and climate-change impacts on salt marshes of Jamaica Bay, New York City. Wetlands 22:71-89.

[11] Jackson, J.B.C. 1991. Adaptation and diversity of reef corals. BioScience 41:475-482.

[12] Lewison, R.L., L.B. Crowder, A.J. Read, and S.A. Freeman. 2004. Understanding impacts of fisheries bycatch on marine megafauna. Trends in Ecology and Evolution 19:598-604.

[13] Marshall, P., and H. Schuttenberg. 2006. A reef manager's guide to coral bleaching. Great Barrier Reef Marine Park Authority, Townsville, Australia. 163 pages.

[14] McClanahan, T.R. 2002. The near future of coral reefs. Environmental Conservation 29:460-483.

[15] Middleton, B.A. 2009. Effects of Hurricane Katrina on the forest structure of *Taxodium distichum* swamps of the Gulf Coast, USA. Wetlands 29:80-87.

[16] Miller, J., E. Muller, C. Rogers, R. Waara, A. Atkinson, K.R.T. Whelan, M. Patterson, and B. Witcher. 2009. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. Coral Reefs 28:925-937.

[17] Mumbry, P.J., and R.S. Steneck. 2008. Coral reef management and conservation in light of rapidly evolving ecological paradigms. Trends in Ecology and Evolution 23:555-563.

[18] Pernas, T., B. Gamble, and T.V. Armentano. 2001. Dry Tortugas National Park – Loggerhead Key exotic plant management and island restoration project. Wildland Weeds 4:13-17.

[19] Pittman, S.J. L. Bauer, S.D. Hile, C.F.G. Jeffrey, E. Davenport, and C. Caldow. 2014. Marine protected areas of the U.S. Virgin Islands: ecological performance report. NOAA Technical Memorandum NOS NCCOS 187. National Oceanic and Atmospheric Administration, Silver Spring, MD. 89 pages.

[20] Rafferty, P., J. Castagna, and D. Adamo. 2010. Building partnerships to restore an urban marsh ecosystem at Gateway National Recreation Area. Park Science 27:34-41.

[21] Rappaport, J. 2003. The United States as a coastal nation. Journal of Economic Growth 8:5-46.

[22] Riegl, B.M., and R.E. Dodge (editors). 2008. Coral reefs of the USA. Springer, New York. 803 pages.

[23] Rinkevich, B. 2014. Rebuilding coral reefs: does active reef restoration lead to sustainable reefs? Current Opinion in Environmental Sustainability 7:28-36.

[24] Spalding, M.D., C. Ravilious, and E.P. Green. 2001. World atlas of coral reefs. University of California Press, Berkeley. 424 pages.
[25] Stanturf, J.A., S.L. Goodrick, and K.W. Outcalt. 2007. Disturbance and coastal forests: a strategic approach to forest management in hurricane impact zones. Forest Ecology and Management 250:119-135.
[26] Steneck, R.S., M.H. Graham, B.J. Bourque, D. Corbett, J.M. Erlandson, J.A. Estes, and M.J. Tegner. 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. Environmental Conservation 29:436-459.
[27] Tanacredi, J.T. 1983. Coastal zone management practices at an urban national park. Environmental Management 7:143-150.
[28] Walker, B.H. 1992. Biodiversity and ecological redundancy. Conservation Biology 6:18-23.
[29] Walls, E.A., J. Berkson, and S.A. Smith. 2002. The horseshoe crab, *Limulus polyphemus*: 200 million years of existence, 100 years of study. Reviews in Fisheries Science 10:39-73.

11 Climate change

[1] Abella, S.R., and P.J. Fornwalt. 2015. Ten years of vegetation assembly after a North American mega fire. Global Change Biology 21:789-802.
[2] Abella, S.R., L.P. Chiquoine, and P.A. Sinanian. 2015. Forest change over 155 years along biophysical gradients of forest composition, environment, and anthropogenic disturbance. Forest Ecology and Management 348:196-207.
[3] Abella, S.R., N.A. Fisichelli, S.M. Schmid, T.M. Embrey, D.L. Hughson, and J. Cipra. 2015. Status and management of non-native plant invasion in three of the largest national parks in the United States. Nature Conservation 10:71-94.
[4] Anderson, G., S. Rutman, and S.M. Munson. 2010. Plant population and habitat characteristics of the endemic Sonoran Desert cactus *Peniocereus striatus* in Organ Pipe Cactus National Monument, Arizona. Madroño 57:220-228.
[5] Benson, L.V., M.S. Berry, E.A. Jolie, J.D. Spangler, D.W. Stahle, and E.M. Hattori. 2007. Possible impacts of early 11th-, middle-12th-, and late-13th-century droughts on western Native Americans and the Mississippian Cahokians. Quaternary Science Reviews 26:336-350.
[6] Bertin, R.I. 2008. Plant phenology and distribution in relation to recent climate change. Journal of the Torrey Botanical Society 135:126-146.
[7] Cole, K.L., and R.H. Webb. 1985. Late Holocene vegetation changes in Greenwater Valley, Mojave Desert, California. Quaternary Research 23:227-235.
[8] Cole, K.L. 2010. Vegetation response to early Holocene warming as an analog for current and future changes. Conservation Biology 24:29-37.
[9] Fisichelli, N.A., S.R. Abella, M. Peters, and F.J. Krist. 2014. Climate, trees, pests, and weeds: change, uncertainty, and biotic stressors in eastern U.S. national park forests. Forest Ecology and Management 327:31-39.
[10] Grundel, R., R.P. Jean, K.J. Frohnapple, J. Gibbs, G.A. Glowacki, and N.B. Pavlovic. 2011. A survey of bees (Hymenoptera: Apoidea) of the Indiana Dunes and Northwest Indiana, USA. Journal of the Kansas Entomological Society 84:105-138.
[11] Hall, M.H.P., and D.B. Fagre. 2003. Modeled climate-induced glacier change in Glacier National Park, 1850-2100. BioScience 53:131-140.
[12] Heller, N.E., and E.S. Zavaleta. 2009. Biodiversity management in the face of climate change: a review of 22 years of recommendations. Biological Conservation 142:14-32.
[13] Hereford, R., R.H. Webb, and C.I. Longpré. 2006. Precipitation history and ecosystem response to multidecadal precipitation variability in the Mojave Desert region, 1893-2001. Journal of Arid Environments 67:13-34.
[14] Huber, M., and R. Knutti. 2012. Anthropogenic and natural warming inferred from changes in Earth's energy balance. Nature Geoscience 5:31-36.
[15] Knapp, A.K., C. Beier, D.D. Briske, et al. 2008. Consequences of more extreme precipitation regimes for terrestrial ecosystems. BioScience 58:811-821.
[16] Krushelnycky, P.D., L.L. Loope, T.W. Giambelluca, F. Starr, K. Starr, D.R. Drake, A.D. Taylor, and R.H. Robichaux. 2013. Climate-associated population declines reverse recovery and threaten future of an iconic high-elevation plant. Global Change Biology 19:911-922.
[17] Lüthi, D., M. Le Floch, B. Bereiter, et al. 2008. High-resolution carbon dioxide concentration record 650,000-800,000 years before present. Nature 453:379-382.
[18] Melillo, J.M., T.C. Richmond, and G.W. Yohe (editors). 2014. Highlights of climate change impacts in the United States: the third national climate assessment. U.S. Government Printing Office, Washington, D.C. 148 pages.

[19] Menz, M.H.M., R.D. Phillips, R. Winfree, C. Kremen, M.A. Aizen, S.D. Johnson, and K.W. Dixon. 2011. Reconnecting plants and pollinators: challenges in the restoration of pollination mutualisms. Trends in Plant Science 16:4-12.
[20] Meyer, H.W., and L. Weber. 1995. Florissant Fossil Beds National Monument: preservation of an ancient ecosystem. Rocks and Minerals 70:232-239.
[21] Monahan, W.B., and N.A. Fisichelli. 2014. Climate exposure of US national parks in a new era of change. PLoS ONE 9:e101302.
[22] Moritz, C., J.L. Patton, C.J. Conroy, J.L. Parra, G.C. White, and S.R. Beissinger. 2008. Impact of a century of climate change on small-mammal communities in Yosemite National Park, USA. Science 322:261-264.
[23] Nowacki, G.J., and M.D. Abrams. 2015. Is climate an important driver of post-European vegetation change in the eastern United States? Global Change Biology 21:314-334.
[24] Ollerton, J., R. Winfree, and S. Tarrant. 2011. How many flowering plants are pollinated by animals? Oikos 120:321-326.
[25] Pascarella, J.B., K.D. Waddington, and P.R. Neal. 1999. The bee fauna (Hymenoptera: Apoidea) of Everglades National Park, Florida and adjacent areas: distribution, phenology, and biogeography. Journal of the Kansas Entomological Society 72:32-45.
[26] Pascarella, J.B., K.D. Waddington, and P.R. Neal. 2001. Non-apoid flower-visiting fauna of Everglades National Park, Florida. Biodiversity and Conservation 10:551-566.
[27] Reyer, C.P.O., S. Leuzinger, A. Rammig, et al. 2013. A plant's perspective of extremes: terrestrial plant responses to changing climatic variability. Global Change Biology 19:75-89.
[28] Ricketts, T.H., J. Regetz, I. Steffan-Dewenter, et al. 2008. Landscape effects on crop pollination services: are there general patterns? Ecology Letters 11:499-515.
[29] Roof, S., and C. Callagan. 2003. The climate of Death Valley, California. Bulletin of the American Meteorological Society 2003:1725-1739.
[30] Rosenstock, S.S., and C. van Riper. 2001. Breeding bird responses to juniper woodland expansion. Journal of Range Management 54:226-232.
[31] Smith, S.D., T.N. Charlet, S.F. Zitzer, S.R. Abella, C.H. Vanier, and T.E. Huxman. 2014. Long-term response of a Mojave Desert winter annual plant community to a whole-ecosystem atmospheric CO_2 manipulation (FACE). Global Change Biology 20:879-892.
[32] Swetnam, T.W., C.H. Baisan, A.C. Caprio, P.M. Brown, R. Touchan, R.S. Anderson, and D.J. Hallett. 2009. Multi-millennial fire history of the giant forest, Sequoia National Park, California, USA. Fire Ecology 5:120-150.
[33] Wells, P.V., and D. Woodcock. 1985. Full-glacial vegetation of Death Valley, California, juniper woodland opening to *Yucca* semidesert. Madroño 32:11-23.
[34] Went, F.W., and M. Westergaard. 1949. The ecology of desert plants. III. Development of plants in the Death Valley National Monument, California. Ecology 30:26-38.
[35] Wilcock, C., and R. Neiland. 2002. Pollination failure in plants: why it happens and when it matters. Trends in Plant Science 7:270-277.
[36] Williams, A.P., C.D. Allen, A.K. Macalady, et al. 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change 3:292-297.
[37] Winograd, I.J., T.B. Copelan, J.M. Landwehr, A.C. Riggs, K.R. Ludwig, B.J. Szabo, P.T. Kolesar, and K.M. Revesz. 1992. Continuous 500,000-year climate record from vein calcite in Devils Hole, Nevada. Science 258:255-260.

12 Active conservation

[1] Abella, S.R. 2014. Effectiveness of exotic plant treatments on National Park Service lands in the United States. Invasive Plant Science and Management 7:147-163.
[2] Abella, S.R., L.P. Chiquoine, A.C. Newton, and C.H. Vanier. 2015. Restoring a desert ecosystem using soil salvage, revegetation, and irrigation. Journal of Arid Environments 115:44-52.
[3] Kleintjes, P.K., B.F. Jacobs, and S.M. Fertig. 2004. Initial response of butterflies to an overstory reduction and slash mulching treatment of a degraded piñon-juniper woodland. Restoration Ecology 12:231-238.
[4] Russell, W., J. Shulzitski, and A. Setty. 2009. Evaluating wildlife response to coastal dune habitat restoration in San Francisco, California. Ecological Restoration 27:439-448.
[5] Sharkey, M.J. 2001. The all taxa biological inventory of the Great Smoky Mountains National Park. Florida Entomologist 84:556-564 (updated in the online All Taxa Biodiversity Inventory, Discover Life in America project).

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ABOUT THE AUTHOR

Scott Abella has a B.S. in Natural Resources Management and an Environmental Chemistry minor from Grand Valley State University, a M.S. in Forest Resources from Clemson University, and a Ph.D. in Forest Science with an emphasis in Restoration Ecology from Northern Arizona University. Since 2004, he has been qualified as a wildland firefighter. He has conducted research and conservation projects across the United States in eastern forests, Midwestern prairies and oak savannas, western forests, and southwestern arid lands. He has worked in Great Smoky Mountains National Park, Saguaro National Park, Lake Mead National Recreation Area, Death Valley National Park, Mojave National Preserve, Joshua Tree National Park, Bryce Canyon National Park, and Pecos National Historical Park. In 2012 and 2013, he worked with the Biological Resource Management Division of the Washington, D.C., office of the National Park Service. He is currently on the faculty of the School of Life Sciences, University of Nevada Las Vegas. He also owns Natural Resource Conservation LLC, an international company dedicated to providing applied science supporting conservation actions. Work for this book brought him to see conservation projects in familiar and less-familiar parks of Olympic, Crater Lake, Grand Canyon, Joshua Tree, Wupatki, Klondike Gold Rush, and many others.

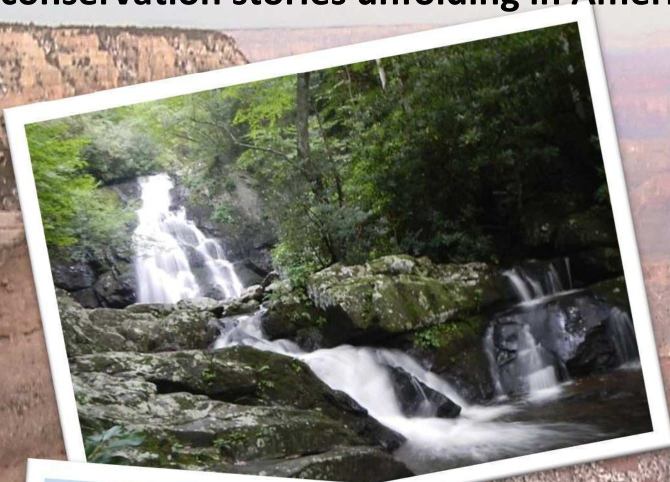
The author in the Newberry Mountains of Lake Mead National Recreation Area, Nevada (photo by A. Mayes).

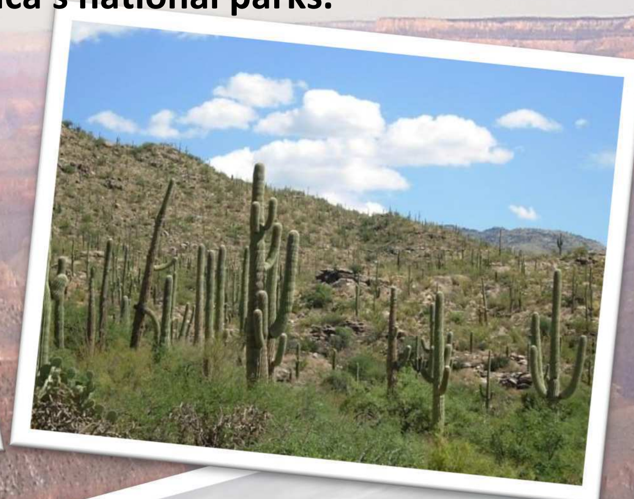
On the covers

Front: South Kaibab Trail, Grand Canyon National Park (by S.R. Abella).

Back: clockwise from top left, Great Smoky Mountains National Park, Saguaro National Park, Wupatki National Monument, Redwood National Park, and Crater Lake National Park (by S.R. Abella).

Conserving America's National Parks **tells stories of conservation challenges and successes from America's 408 national parks. Rising sea levels, loss of wildlife species, droughts, earthworm invasions, climate change, and many other challenges face parks. But inspiring conservation successes provide hope for the future of parks. Richly illustrated with 247 photos, maps, and sketches,** ***Conserving America's National Parks*** **is unprecedented in its scope of conservation stories unfolding in America's national parks.**





CONSERVING AMERICA'S NATIONAL PARKS

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