Chapter 10

Biodiversity: Wildlife and Aquatic Biota

Flitcroft, Rebecca L.; Bury, Gwendolynn W.; Joyce, Linda A.; Kay, Shannon L.; Knowles, Michael S.; Nelson, Mark D.; Warziniack, Travis. 2023. Biodiversity: Wildlife and Aquatic Biota. In: U.S. Department of Agriculture, Forest Service. 2023. Future of America's Forest and Rangelands: Forest Service 2020 Resources Planning Act Assessment. Gen. Tech. Rep. WO-102. Washington, DC: 10-1–10-34. Chapter 10. https://doi.org/10.2737/WO-GTR-102-Chap10.

Biological diversity, or biodiversity, measures the variety of organisms and life forms in a specific area and is a key attribute of healthy ecosystems (Elmqvist et al. 2003). Biodiversity naturally varies across the United States in response to geologic history, tectonic activity, and ecoregional characteristics, yet areas of the country with naturally lower biodiversity still support species of regional ecological, cultural, and commercial importance and provide important ecosystem services. Significant declines in biodiversity have been documented globally across taxonimic groups (Garcia-Morena et al. 2014) and within the conterminous United States (e.g., in birds; see Rosenberg et al. 2019). Declining biodiversity is strongly linked to the anthropogenic actions that have modified ecosystems leading to declines in habitat quality, quantity, and connectity in marine, freshwater, and terrestrial settings. Tracking patterns in biodiversity, therefore, is key to informing management regarding places of particular vulnerability to current and emerging threats.

In this chapter of the Resources Planning Act (RPA) Assessment, we focus on patterns and threats to native wildlife and aquatic species biodiversity, measured as species richness. We present biodiversity information at multiple spatial and taxonmic scales, comparing current conditions with those presented in previous reports and adding new approaches for better addressing biodiversity conservation in the United States, with minimal representation of distributions and analysis for Alaska and Hawaii. We report geographic patterns of biodiversity, recent

- ❖ Trends from breeding bird surveys indicate population declines in at least 20 percent of all bird species across habitat types since the 1950s/60s, and in more than 50 percent of species that occupy grasslands or are ground nesting. These declines are linked to land use modifications of habitats as well as introduced species and loss of habitat connectivity.
- ❖ Concentrations of imperiled taxa with a listing status under the Endangered Species Act are found across the country, with particular concern in Peninsular Florida and Hawaii for birds, and in the RPA North and South Regions for fishes, crayfish, and mussels.
- ❖ Watersheds of the RPA North and South Regions are most vulnerable to compounded land use stress. Regardless of RPA region, development stands out as the largest overall land use stressor for native ecosystems.
- ❖ Areas of potential high climate stress were consistently found in mountainous areas of the RPA North, Rocky Mountain, and Pacific Coast Regions, with pockets of stress identified in arid regions of the Rocky Mountain Region.
- ❖ Federal lands with a lower risk of development or land conversion, such as those managed by the U.S. Department of Agriculture Forest Service and U.S. National Park Service, are projected to be under higher climate stress compared with other lands, potentially limiting their future ability to function as climate refugia for native biota.

historical trends in populations (particularly of birds), and patterns of species at risk of extinction as defined by their listing status under the Endangered Species Act (ESA). We also provide a discussion of threats to biodiversity including invasive species, pathogens, current land uses, and projected future climate change.

Patterns of Biodiversity

- ❖ The Mississippi River basin and Madrean Sky Islands/U.S.-Mexico border areas contain the highest terrestrial and aquatic biodiversity across the conterminous United States. These centers of biodiversity are linked to geologic history and the localized isolation of species (resulting in high rates of endemism).
- ❖ Population trends in migratory game birds (webless and waterfowl groups) since the 1950s/60s exhibit inter-annual variability, with harvest generally tracking overall population numbers. Over time, populations and harvest of ducks and geese tend to be increasing, while woodcocks and mourning doves are declining.
- ❖ Long-term population trends from breeding bird surveys report statistically significant declines in all groups of breeding birds, with at least half of all species of grassland or ground-nesting species showing significant population declines.

Biodiversity is in decline globally, with freshwater aquatic biodiversity declining at rates that exceed marine or terrestrial ecosystems (Tickner et al. 2020). In the United States, concern about species and their habitats has rallied diverse partners to unite in the shared goal of ecosystem protection and enhancement. In this section, we begin by presenting spatial patterns of terrestrial and aquatic biodiversity, followed by a presentation of status and trend information for avian species. We also present patterns of imperiled species as defined by their listing status under the ESA. We acknowledge that additional species may be imperiled across the United States that are not federally listed under the ESA. We primarily present ESA-listed species to provide direct information to land managers who have a responsibility to participate in the development of conservation plans.

Native Terrestrial Biota

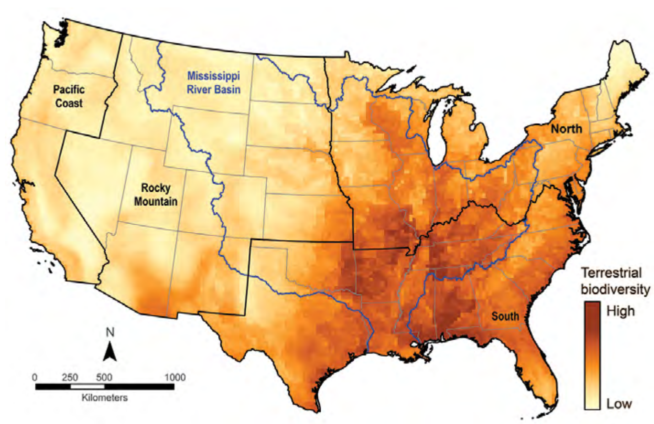
In order to assess geographic patterns of distribution, and for reference in vulnerability assessments described later in this chapter, we mapped terrestrial biodiversity across the conterminous United States. We acquired native terrestrial biodiversity datasets from NatureServe (https://www.natureserve.org), the organization that maintains the most comprehensive national data available on the taxonomy, distribution, and conservation status of native species. Individual species were counted for each 250-square-mile (647-km^2) grid from the NatureServe species dataset and displayed in map form (figure 10-1). The dataset includes 4,107 total species from across the country (inclusive of Alaska and Hawaii), capturing insects, birds, mammals, crustaceans, reptiles, and amphibians, among others. Of particular relevance for forest management are the subset of these species identified as forest-associated (see the sidebar Avian Species Associated with Forested Environments).

Across the conterminous United States, the highest biodiversity of terrestrial fauna is found in the eastern portions of the Mississippi River basin, in the Madrean Sky Islands, in the border areas between Texas and Mexico (consistent with findings of Van Devender et al. 2013), and in the south Atlantic- and Gulf-draining watersheds of Alabama, Florida, North Carolina, and South Carolina. Western States cover a geologically younger, less-dissected landscape than the eastern portion of the United States, leading to lower overall biodiversity in this area (Elkins et al. 2019).

Native Fish, Crayfish, and Freshwater Mussel Biodiversity

For the first time in an RPA Assessment, we are able to present species distribution data for native fish, crayfish, and mussels, allowing us to describe regional characteristics of aquatic species diversity. As for terrestrial biodiveristy, we acquired datasets of native aquatic species from NatureServe

Figure 10-1. Biodiversity of native terrestrial species (excluding plants) mapped at a resolution of 250 mi^2 for the conterminous United States, with RPA regional boundaries and the outline of the Mississippi River basin in blue. Invasive or introduced species are not included.

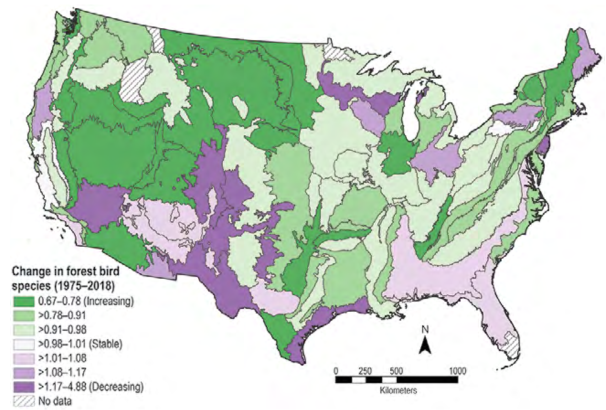


Source: NatureServe.

Avian Species Associated with Forested Environments

A closer look at long-term data available on forest-associated birds is available through the North American Breeding Bird Survey (BBS). Using BBS datasets, we found that long-term (1975 to 2018) trends in the number of bird species associated with forests varied among ecoregions of the conterminous United States (figure 10-2). The greatest increases in numbers of forest bird species were clustered in ecoregions of the northern Great Plains and Intermountain West, with additional gains scattered among the Cross Timbers and Arkansas Valley, Southern Texas Plains, Central Corn Belt Plains, Southwestern Appalachians, and Northeastern Highlands. The greatest declines were distributed among the Southern Rockies, Southwestern Tablelands, and Chihuahuan Deserts, and scattered among disjunct ecoregions of the Mojave Basin and Range, Western Gulf Coastal Plain, North Central Hardwood Forests, and Atlantic Coastal Pine Barrens (see Breeding Birds subsection of Avian Fauna section, figures 10-10, 10-11, 10-12, 10-13).

Figure 10-2. Estimated long-term change in the number of forest-associated bird species detected from 1975 to 2018. Change is measured as 1975 species divided by 2018 species number estimate, excluding exotic species. Values <1.0 indicate species numbers increasing (green shades); values >1.0 indicate species numbers decreasing (purple shades).



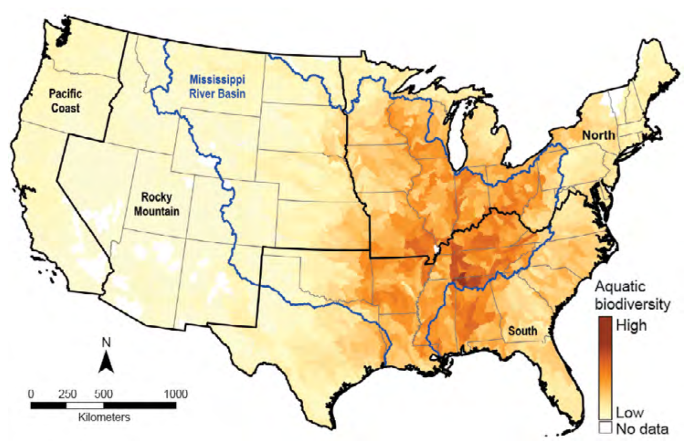
Source: USGS Breeding Bird Survey.

(https://www.natureserve.org/biodiversity-science/species-ecosystems) and mapped them at the 8-digit hydrologic unit code scale (HUC 8; https://water.usgs.gov/GIS/huc.html) across the conterminous United States (figure 10-3). We summarized individual species information per HUC 8 into counts to characterize biotic richness. A HUC 8 is intended to capture an entire subbasin from the headwaters to the mouth. In a large river system such as the Columbia River, there are multiple subbasins at the HUC 8 scale. The NatureServe aquatic biodiversity dataset contains 905 species of native fishes, 391 species of native crayfish, and 304 species of native mussels. All subspecies and populations were merged into species designations for mapping and tabular summaries. Nonnative or introduced aquatic biota are not included in this assessment. For example, introduced warmwater fish such as largemouth bass (*Micropterus salmoides*) in the Western United States are not included in the total count of native fish species in western areas.

Native aquatic species biodiversity and the distribution of individual species vary in response to a diversity of factors including geologic age, patterns of river connectivity, latitude and longitude, precipitation and thermal regimes, riparian habitat composition (see the sidebar Riparian Areas for more information regarding classification of riparian ecotone habitat for management applications), and human modifiers of the landscape such as land use and water management. The RPA South Region is a global hotspot in aquatic species biodiversity generally, and for crayfish in particular, and is the RPA region with the highest biodiversity of fish, crayfish, and mussels (table 10-1). Overall biodiversity of aquatic biota is lowest in the RPA Pacific Coast Region (table 10-1).

Biodiversity can be higher in areas with more endemic species. For this analysis, we defined endemism by RPA region: species endemic to a region occur only in that region. Because RPA regions do not follow watershed boundaries,

Figure 10-3. Aquatic biodiversity of the conterminous United States mapped at a HUC 8 watershed scale, with the Mississippi River basin outlined in blue.



HUC = hydrologic unit code.

Source: NatureServe.

Table 10-1. Native aquatic biodiversity and species endemic to each RPA region for fish, crayfish, and mussels. Numbers are species richness counts, with the number of endemic species in parentheses.

RPA region	Combined biodiversity	Fish	Crayfish	Mussels
	Total biodiversity (total endemic)			
North	560 (63)	354 (35)	91 (24)	115 (4)
South	1,353 (855)	702 (385)	360 (293)	291 (177)
Rocky Mountain	342 (59)	259 (57)	18 (2)	65 (0)
Pacific Coast	127 (67)	109 (63)	10 (2)	8 (2)

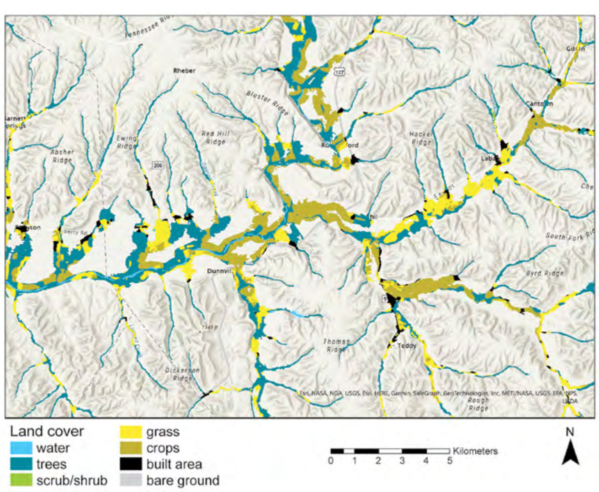
we assigned HUC 8 watersheds located along border areas to the RPA region that contained the majority of the HUC. All regions had high numbers of endemic species (table 10-1). The highest percentage of biodiversity identified as regionally endemic was found in the South Region (63 percent), followed by the Pacific Coast Region (53 percent) (calculated from table 10-1).

An accurate understanding of the distribution of species is important for both current and future management. This means having accurate maps not only for individual species, but also for abundance and diversity across the United States. Tracking patterns of overall and endemic species biodiversity can provide important information about ecosystem health broadly (Bonn et al. 2002), in addition to the potential resilience of local ecosystems (Burlakova et al. 2011). Detection of changes and patterns in biodiversity at a national scale associated with changes in the intensity and distribution of human uses may be found through examination of the entire suite of species present (e.g., Brown and Laband 2006), as well as by exploring the response of endemic species to human-mediated changes such as climate, land management, and human population growth. Endemic species have been identified as potentially more vulnerable to climate change than more widely distributed species (Malcolm et al. 2006) because many endemic species occur within a small and sometimes isolated geographic extent, leading to narrower habitat and environmental tolerances.

Riparian Areas

Riparian ecotones are an important natural resource, rich in biodiversity, ecological, and hydrological functions, supporting both aquatic and terrestrial biodiversity. Further, these ecotones contain specific vegetation and soil characteristics that play important roles in protecting water quality and stream ecosystem health and are very responsive to land management activities (Mitsch and Gosselink 1993, Naiman et al. 1993).

Figure 10-4. Percent riparian ecotone area per HUC 10 watershed in the National Riparian Areas Base Map in 2020.



HUC = hydrologic unit code.

Since the early settlement of the United States, riparian areas have experienced alterations resulting from urbanization, agricultural activities, and floodplain development that altered their extent and land cover condition (Brinson et al. 1981, Doppelt et al. 1993, Tockner et al. 2002).

The National Riparian Areas Inventory Project provides free tools and riparian datasets to facilitate riparian area delineation and quantification on multiple scales. These data support monitoring, riparian land cover classification, riparian conservation prioritization, and riparian areas management. The new National Riparian Areas Base Map uses the Riparian Buffer Delineation Model (Abood et al. 2012, www.riparian.solutions) to display riparian acreage, spatial distribution, and land cover at a national scale, where riparian areas are defined as streamside zones within the 50-year flood area of a stream. This Riparian Areas Base Map shows both the extent of riparian areas and the general riparian land cover composition, highlighting where riparian areas contribute disproportionately to biodiversity conservation compared to their abundance (figure 10-4).

Sinan Abood, USDA Forest Service, Washington Office Biological & Physical Resources; **Linda Spencer**, USDA Forest Service, Forest Management, Rangeland Management, and Vegetation Ecology; **Michael Wieczorek**, U.S. Geological Survey; and **Ann Maclean**, Michigan Technological University

Avian Fauna

Although trends in terrestrial and aquatic biodiversity over time are not available at a national scale, species-specific temporal trends for some avian fauna are available and provide insights into their patterns of population abundance. Given increasing threats to fish and wildlife, examining patterns across both time and space helps us understand the potential consequences of management decisions and locate risks as well as potential refugia. We present status and trends in selected bird species using data collected by the U.S. Fish and Wildlife Service (FWS) and U.S. Geological Survey (USGS), in collaboration with States, Tribes, private landowners, nongovernmental organizations, and other Federal partners. These groups have responsibility for the conservation or monitoring of migratory species (e.g., waterfowl and neotropical migratory songbirds), federally listed species (endangered, threatened, and proposed for listing), and species with special designations like the golden eagle (*Aquila chrysaetos*) and bald eagle (*Haliaeetus leucocephalus*). This section describes population and harvest trends in migratory game birds as well as trends in populations of breeding birds. We present these results using the organizational units associated with the surveys and trend assessments associated with data collection rather than in relation to RPA regions, to preserve the statistical accuracy of the source information.

Migratory Game Birds

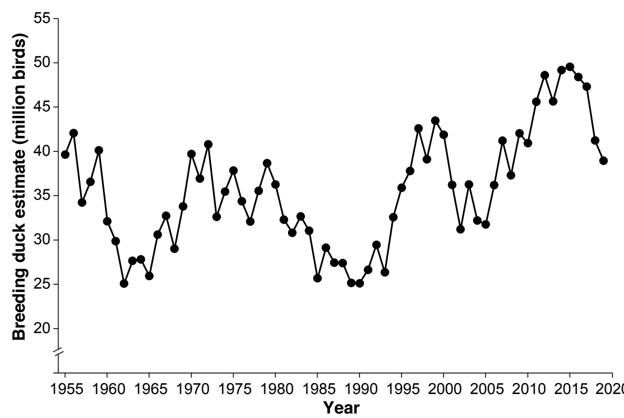
Migratory game birds reported here collectively refer to waterfowl (ducks, geese, and swans) and two additional species defined as webless migratory game birds: the mourning dove (*Zenaida macroura*) and American woodcock (*Scolopax minor*). Migratory game birds are economically important through recreational harvest and contribute to native ecosystems and biodiversity. These birds have a rigorous management history traceable to a series of international agreements to conserve them that were signed at the turn of the 20th century. This focused management led to the development of what many consider to be the leading monitoring system for conterminously distributed species (Nichols et al. 1995). Population flyways or management units have been established to achieve consistent monitoring and management of waterfowl, mourning doves, and woodcock. Waterfowl harvest regulation decisions are informed by population monitoring data (Nichols et al. 2007), so it is not surprising that harvest trends mirror breeding population trends (Flather et al. 2013).

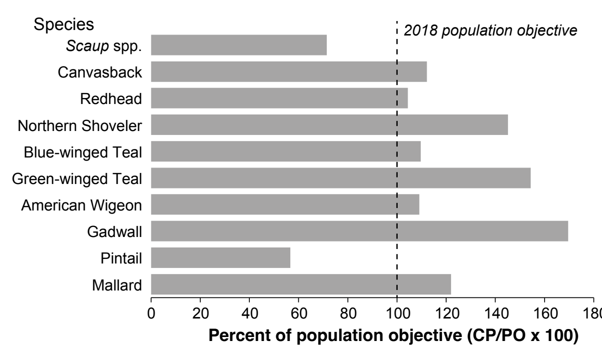
Migratory Waterfowl—Ducks, Geese, and Swans

Trends in the populations of migratory waterfowl, including ducks, geese, and swans, vary over time. Breeding duck population estimates in 2019 were 2 percent lower than in 1955 and 10 percent higher than the long-term (1955 to 2019) average (figure 10-5). After falling to record lows in 1990 (25 million birds), duck populations increased to almost 50 million birds in 2015, with a subsequent decrease to nearly 39 million birds by 2019, amounting to a 55-percent net increase since 1990. Breeding population trends among the 10 most common duck species have been variable. Relative to population objectives established in the 2018 North American Waterfowl Management Plan, 8 of the 10 most common duck species (species grouped for lesser and greater scaup, *Aythya marila* and *A. affinis*, respectively) have 2019 breeding populations that exceeded population objectives, but pintail (*Anas acuta*) and scaup (greater and lesser combined) fell below objectives by 43 and 29 percent, respectively (figure 10-5). Gadwall (*Mareca strepera*) and green-winged teal (*Anas carolinensis*) both exceeded population objectives by more than 50 percent. Mallard (*Anas platyrhynchos*), the most abundant duck (9.4 million in 2019), exceeded its long-term population objective by 22 percent.

Duck harvest numbers across the United States increased from 5.1 million in 1961 to 9.7 million in 2019 (figure 10-6). During that period, harvest rates increased to exceed 15 million in the 1970s, decreased to below 5 million in the late 1980s, peaked above 17 million in the late 1990s, and then declined gradually to the current rate. The national pattern of harvest lows during the late 1980s is repeated in each of four flyways

Figure 10-5. Trend in the duck population from 1955 to 2019 (top); the relation between current (2019) duck population estimates (CP) for the 10 principal duck species (species grouped for greater and lesser scaup) with reference to the population objectives (PO) specified in the 2018 North American Waterfowl Management Plan, measured as percent of objective (bottom).



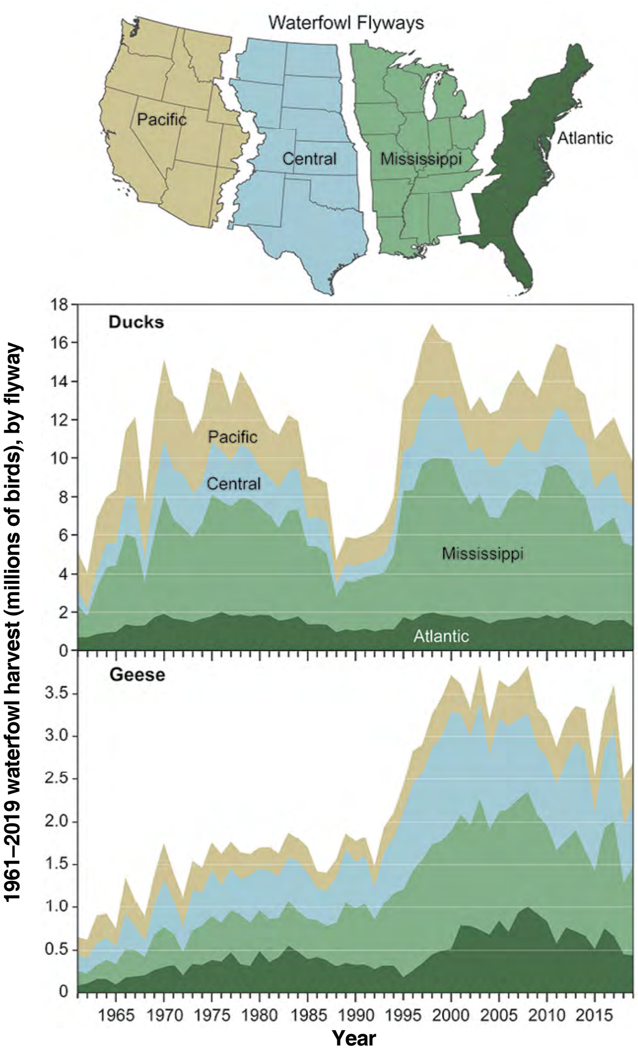


Source: U.S. Department of the Interior, U.S. Fish and Wildlife Service.

tracked by the FWS (i.e., Pacific, Central, Mississippi, and Atlantic) (figure 10-6). Similarly, goose harvest—including Canada geese (*Branta canadensis*), brant (*Branta bernicla*), snow geese (*Chen caerulescens*), Ross's geese (*Chen rossii*), emperor geese (*Anser canagicus*), and white-fronted geese (*Anser albifrons*)—increased slowly from 0.65 million in 1961 to 1.82 million in 1991, then increased substantially to 3.82 million in 2008. Since then, the harvest rate has been more variable, with a slightly decreasing trend through the 2019 harvest rate of 2.69 million birds (figure 10-6).

Swan populations (only tundra swans, *Cygnus columbianus*) are monitored by the FWS through surveys of many separate population segments of varying size. Population and harvest estimates are reported for Eastern versus Western U.S. regions. In 2019, the total swan population was estimated at 194,000 birds, nearly identical to the estimate of 193,000 birds from 1985. Fluctuations in populations ranged from a low of 158,000 in 1993 to a high of 270,000 in 2007 and 2008. Eastern and western populations exhibit similar total numbers and trends (figure 10-7). Swan harvest estimates have been variable, with increasing trends similar to populations from 1962 to 2019 (figure 10-7).

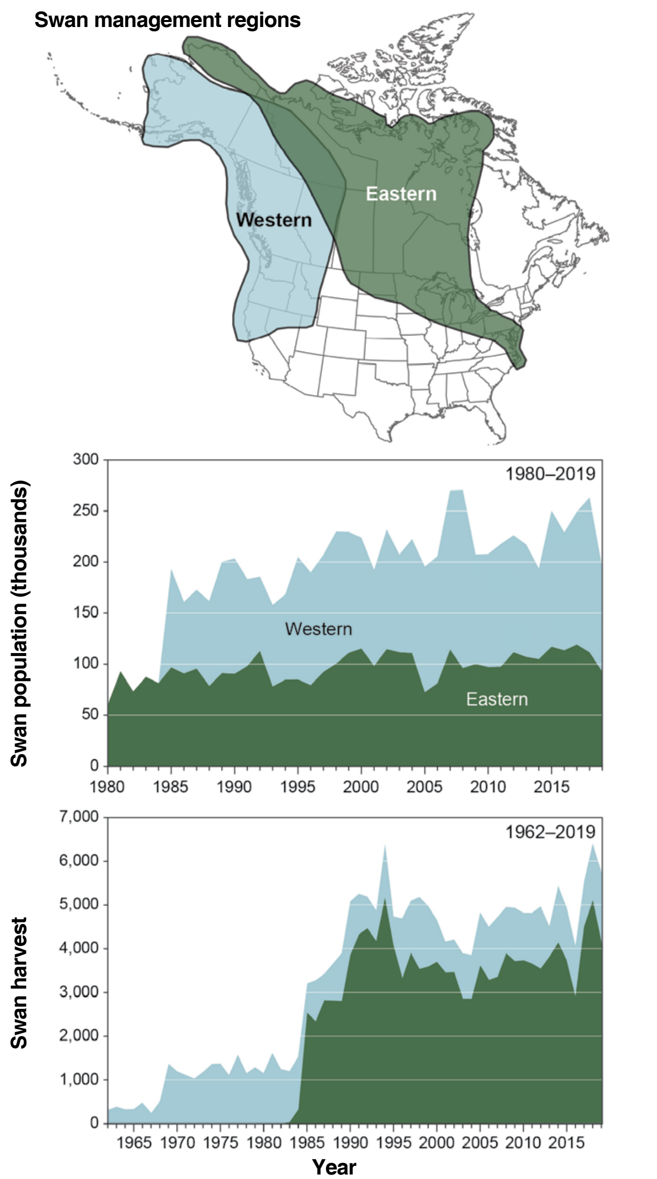
Figure 10-6. National trends across FWS administrative waterfowl flyway boundaries (top) for total duck harvest (middle) and total goose harvest (bottom), from 1961 to 2019.



FWS = U.S. Fish and Wildlife Service.

Source: U.S. Department of the Interior, U.S. Fish and Wildlife Service.

Figure 10-7. National trends for the western and eastern regions (top) for swan population from 1980 to 2019 (middle) and swan harvest from 1962 to 2019 (bottom).



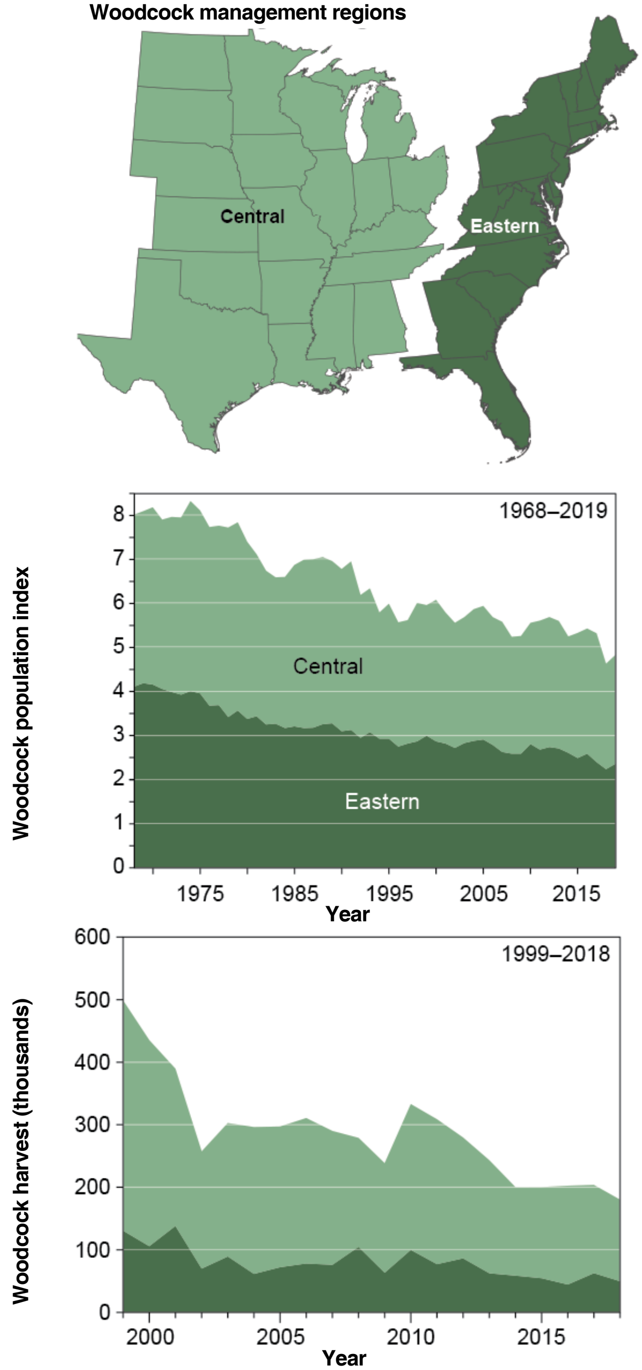
Source: U.S. Department of the Interior, U.S. Fish and Wildlife Service.

Webless Migratory Game Birds—American Woodcock and Mourning Dove

American woodcock populations and harvest have decreased over the past 50 years. American woodcock singing-ground surveys (SGS) have been conducted along permanent survey routes by the FWS each spring since 1968 (Seamans and Rau 2020), and populations are reported as an index of average numbers of birds detected per SGS route. Woodcock SGS indices have decreased consistently in both the Eastern and Central management regions (figure 10-8). On average, 4.0 woodcock were detected per SGS route in 1968, decreasing to 2.4 birds in 2019 (figure 10-8). Woodcock harvest rates have also decreased, from nearly 500,000 birds in 1999 to 180,000 birds in 2018, with harvests in the Eastern and Central management regions decreasing by similar proportions (figure 10-8). For the period during which both SGS and harvest rates are reported (1999 to 2018), the woodcock population index decreased from 3.0 to 2.3 birds per route.

Mourning doves have declined in abundance from 352 million doves in 2003 to 183 million doves in 2019, a 3.5-percent average annual rate of decline (figure 10-9) (contact chapter authors for extensive reference list of data sources). Average annual rates of change during this period were −3.3 percent in the Eastern, −0.5 percent in the Central, and −5.1 percent in the Western FWS management units (figure 10-9). Mourning dove harvests have also declined, from 24 million in 1999 to 10 million in 2019 (figure 10-9). For the period during which population estimates are reported (2003 to 2019), harvest rates changed at average annual rates of −3.5 percent in the Eastern, −2.0 percent in the Central, and −3.5 percent in the Western management units, for an overall decline of 2.8 percent per year.

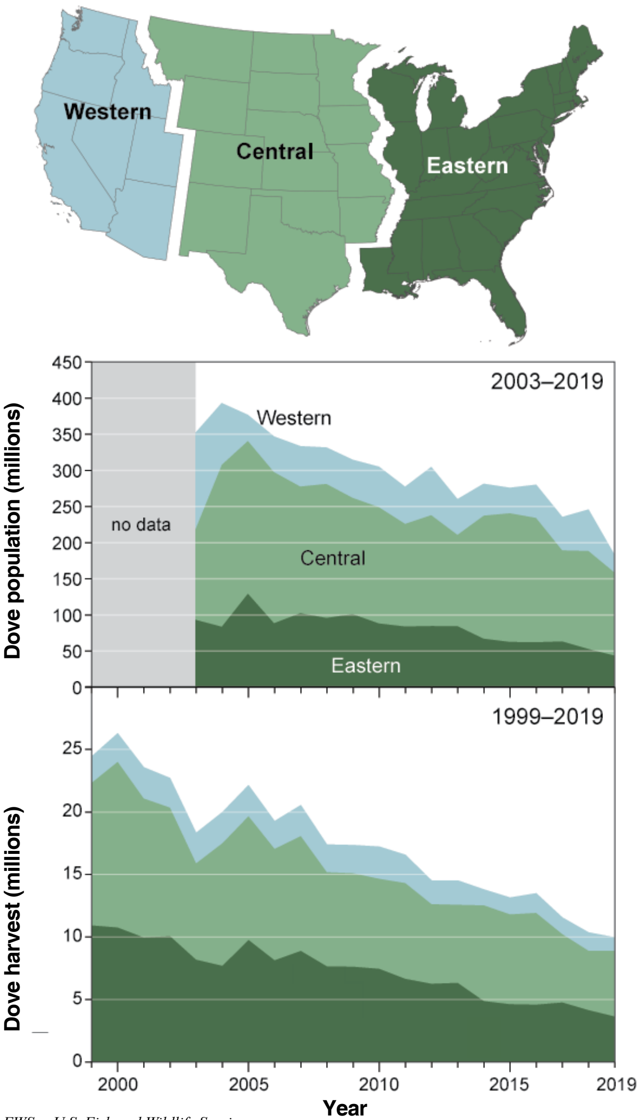
Figure 10-8. American woodcock FWS administrative management regions (top); population index from 1968 to 2019 (middle); and harvest trends from 1999 to 2018 (bottom).



FWS = U.S. Fish and Wildlife Service.

Source: singing-ground surveys (SGS) conducted by the U.S. Department of the Interior, U.S. Fish and Wildlife Service.

Figure 10-9. Mourning dove FWS administrative management units (top); population trends from 2003 to 2019 (middle); and harvest trends from 1999 to 2019 (bottom).



FWS = U.S. Fish and Wildlife Service.

Source: Population data from U.S. Department of the Interior, U.S. Fish and Wildlife Service.

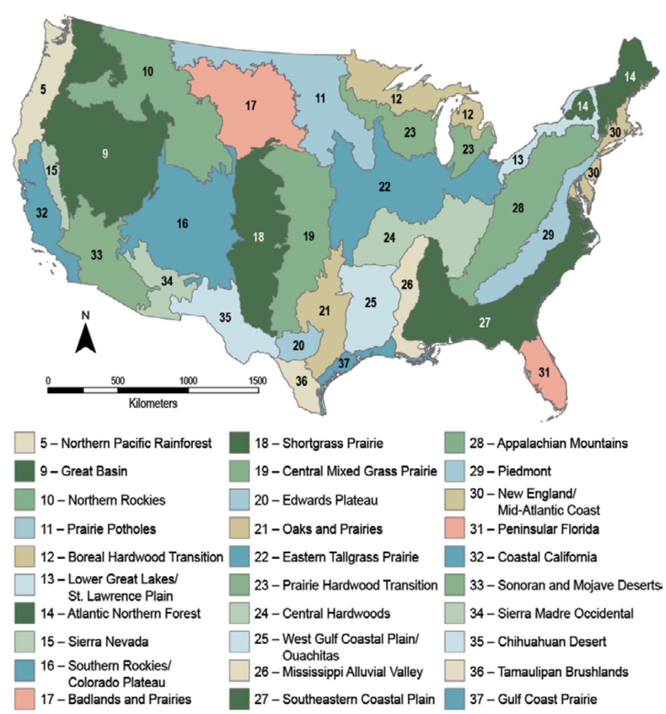
Breeding Birds

Wild bird populations have long been considered good indicators of environmental threats like landscape change because changes in habitat affect the abundance and diversity of bird species that occupy a particular region (Flather and Sauer 1996, Pidgeon et al. 2007). Given that North American bird populations have declined by 29 percent since 1970, a net loss of nearly 3 billion birds (Rosenberg et al. 2019), it is important to evaluate the status and trends among bird species throughout the country.

Reported breeding bird trends are based on the North American Breeding Bird Survey (BBS), an annual survey managed by the USGS that provides trends in the relative abundance of more than 400 bird species nationwide (Robbins et al. 1986). BBS population trends are summarized nationally and by individual Bird Conservation Regions (BCR; figure 10-10), which delineate ecologically distinct regions in North America with similar bird communities, habitats, and resource management issues. Thirty BCRs are located in the conterminous United States and are included in this report (https://nabci-us.org/resources/bird-conservation-regions/).

To obtain more detailed understanding of how birds respond to changes in their environments, we grouped bird species by breeding habitat type (grassland, successional-scrub, wetland, woodland), nest type and location (cavity, ground-low, midstory canopy, open cup), and migration status (neotropical, permanent, short distance). Details on data sources and methods are reviewed in Flather et al. (2013). Trends in abundance are reported for long-term (1966 to 2019) and short-term (1993 to 2019) time periods. Direction and statistical significance of population trends are labeled as having significant increase, nonsignificant increase, nonsignificant decrease, or significant decrease, based on a hierarchical modeling approach (Sauer and Link 2002) that provides a convenient framework for summarizing population change among regions. Long-term population trends from breeding bird surveys show declines in most categories of birds defined by habitat type.

Figure 10-10. Bird conservation regions of the United States.



Source: North American Bird Conservation Initiative (https://nabci-us.org/resources/bird-conservation-regions/).

Long-Term Abundance Trends

From 1966 to 2015, statistically significant decreases in bird populations exceeded significant increases for species in grassland and successional-scrub habitats; species that build ground-low, midstory, and open-cup nests; and species with neotropical or short-distance migration patterns (figure 10-11). Grassland bird species had the greatest declines in long-term trends, with 54 percent of species showing significant decreases while only 4 percent had significant increases. The remaining 42 percent of grassland birds experienced population changes that were not statistically significant. The cavity nesting category contained the largest percentage of species with statistically significant increasing populations (37 percent), but this was mostly offset by significantly decreasing populations (29 percent). Other categories where significantly increasing populations exceeded decreasing populations included wetland habitat users and permanent residents.

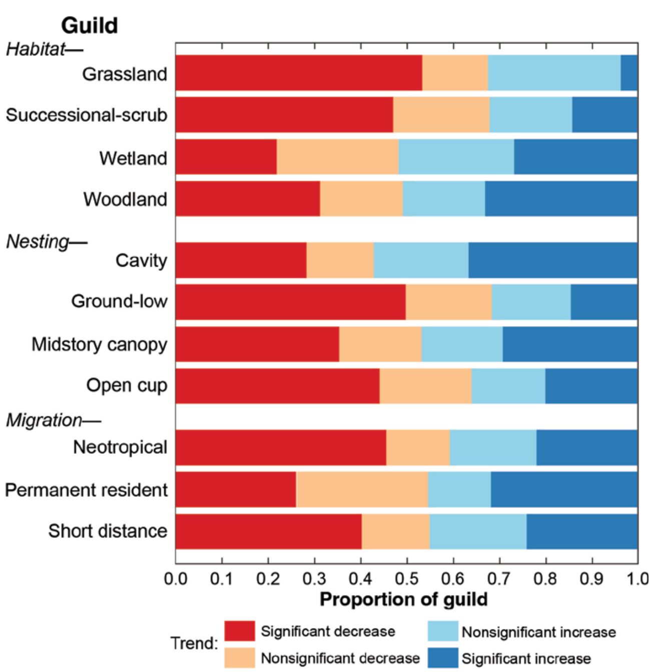
Short-Term Abundance Trends

Patterns of species populations over the 2005 to 2015 period mirror longer term trends; however, smaller proportions of species have statistically significant increasing or decreasing populations (figure 10-12). This may be partially explained by smaller sample sizes available during shorter periods. Trends for midstory canopy nesters and neotropical migrants differed from longer term trends; these groups had more species with statistically significant increasing populations in the short term (figure 10-12).

Trend Comparisons

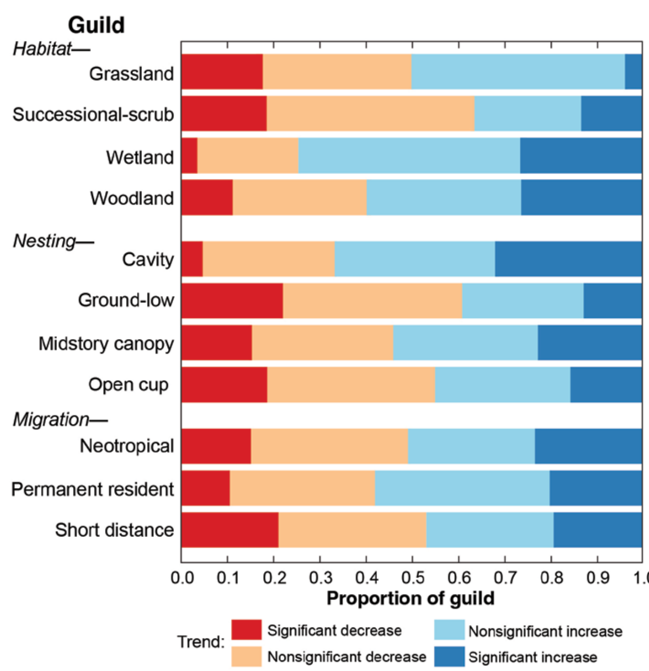
Across all the habitat affinity groupings, all BCRs had at least one bird species that showed statistically significant population gains over both the long term (1966 to 2019) and the short term (1993 to 2019) (see “All” in figure 10-13). When examining individual habitat affinity groups, patterns of increasing or decreasing populations varied. Grassland species are almost universally in decline over both long- and short-term assessments (figure 10-13). Successional-scrub-associated species in the North Region and Southeast Subregion also experienced population declines over both time periods. Some increases in wetland species in the north-central portion of the United States

Figure 10-11. Long-term increases and decreases in proportions of native bird populations in the conterminous United States, 1966 to 2015. Long-term proportion of native bird species with decreasing populations shown in red and increasing populations shown in blue. Changes that are statistically significant appear in a dark shade; nonsignificant changes appear in a light shade. Species populations are analyzed based on major habitat affinity, nesting position, and migratory status.



Source: USGS Breeding Bird Survey.

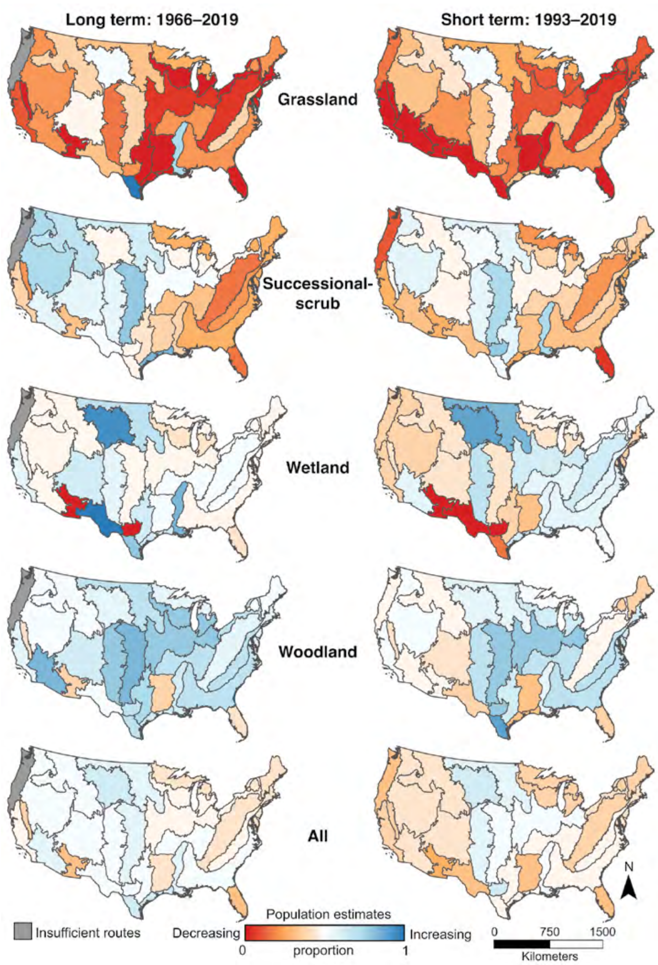
Figure 10-12. Short-term increases and decreases in proportions of native bird populations in the conterminous United States, 2005 to 2015. Short-term proportion of native bird species with decreasing populations shown in red and increasing populations shown in blue. Changes that are statistically significant appear in a dark shade; nonsignificant changes appear in a light shade. Species populations are analyzed based on major habitat affinity, nesting position, and migratory status.



Source: USGS Breeding Bird Survey.

are noted (figure 10-13). Wetland species in the Texas borderlands region experienced significantly decreasing populations in the short term, alongside increasing populations over the long term (figure 10-13).

Figure 10-13. Decreasing or increasing native bird populations in the conterminous United States, by Bird Conservation Region. Proportion of native bird species with decreasing (red) or increasing (blue) populations estimates, by major habitat affinity, over the long term (1966 to 2019; left) and short term (1993 to 2019; right). Gray areas denote where there were insufficient routes for observing species.



Source: USGS Breeding Bird Survey and North American Bird Conservation Initiative.

Imperiled Animal Species

- ❖ Concentrations of ESA-listed birds are documented in Peninsular Florida and Hawaii, whereas listed mammals and fish are widely distributed, with regional concentrations across multiple areas.
- ❖ Regionally constrained ESA-listed distributions were identified for amphibians (Coastal California), crustaceans (Coastal Mountains and Dry Steppe), mollusks (Upper Midwest, Southern Appalachia, and Interior Highland Hills and Plateau), and reptiles (Gulf Coast and Peninsular Florida).

The protection of native biota to avoid decline and the conservation of species already in decline are both primary goals of State and Federal agencies. This important work addresses changes in habitat and environmental conditions resulting from a variety of causes that affect some of our most iconic and culturally important species (see the sidebar Pacific Trout in the Conterminous United States). In this section, we describe patterns in the distribution of federally identified imperiled species listed under the ESA because their conservation status is linked to applied management actions within the United States. We also present hotspots for taxonomic groups and describe taxa identified as Species of Greatest Conservation Need from across the set of federally mandated State Wildlife Action Plans, which guide biodiversity management and conservation actions at the State level.

Federally Listed Imperiled Species

Federal listing demonstrates both empirical evidence of species-specific long-term population-scale trends, as well as political will in support of listing decisions. We focus on ESA-listed species, specifically species listed as federally endangered, threatened, proposed endangered or threatened, candidate, species of concern, or listed threatened because of similarity in appearance to another species (definition of these species codes can be found at: https://www.fws.gov/endangered/about/listing-status-codes.html). We refer to this broad assemblage of species as imperiled throughout this chapter. Species assigned G1 or G2 conservation status by NatureServe are not included in this analysis because they are not federally listed under the ESA, but these species are included in the sidebar Forest-Associated Species at Risk of Decline.

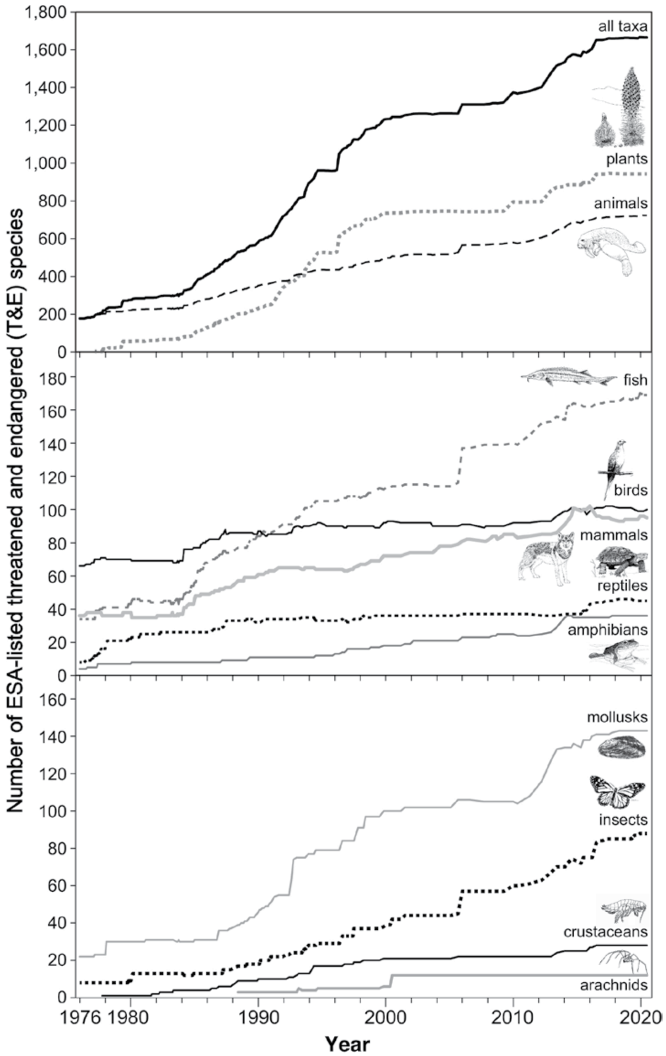
Distributions of ESA-listed taxa vary geographically, with some portions of the country containing higher numbers of listed species than others. Hawaii has the largest number of listed species (499; mostly flowering plants), nearly double the second highest State, California (286). States with the fewest listed species include Washington, DC (3), Vermont (6), North Dakota (8), and Alaska (8) (https://ecos.fws.gov/ecp/report/species-listings-by-state-totals?statusCategory=Listed). For this assessment, species distributions across the United States were mapped onto an equal-area grid (250 square miles [647 km^2]) to eliminate area effects of State or county size (figure 10-14). Imperiled birds are widely distributed, with notable hotspots in Hawaii and Peninsular Florida (figure 10-14). Imperiled mammal and fish species are also widely distributed, but with multiple regional areas of concentration (figure 10-14). More regionally constrained distributions are evident for imperiled amphibians (Coastal California), crustaceans (Coastal Mountains and Dry Steppe), mollusks (Upper Midwest, Southern Appalachia, and Interior Highland Hills and Plateau), and reptiles (Gulf Coast and Peninsular Florida) (figure 10-14).

The total number of ESA-listed species are tracked by the FWS, Environmental Conservation Online System (https://ecos.fws.gov/ecp/report/boxscore). We plotted this information from July 1976 through October 2021 and saw a steady rise in the overall number of federally listed imperiled species, with sharp increases in aquatic taxa (e.g., fish, mussels) and insects (figure 10-15). With few species being delisted, current patterns of distribution reflect cumulative counts of species that have been federally listed over time.

Figure 10-14. Geographic distributions of plant, mollusk, coral, bird, crustacean, insect, arachnid, mammal, fish, amphibian, and reptile species formally listed under the Endangered Species Act as endangered, threatened, proposed endangered or threatened, candidate, species of concern, or listed threatened because of similarity in appearance to another species. Alaska and Hawaii are displayed at a different scale for presentation purposes.

Source: NatureServe.

Figure 10-15. Cumulative number of species listed as endangered or threatened under the Endangered Species Act (accounting for delistings) from 1 July 1976 through 4 October 2021 for plants and animals (top), vertebrate groups (middle), and invertebrate groups (bottom). Increases in a given year are additional species added, making the total number of species in a given year cumulative over time.



ESA = Endangered Species Act.

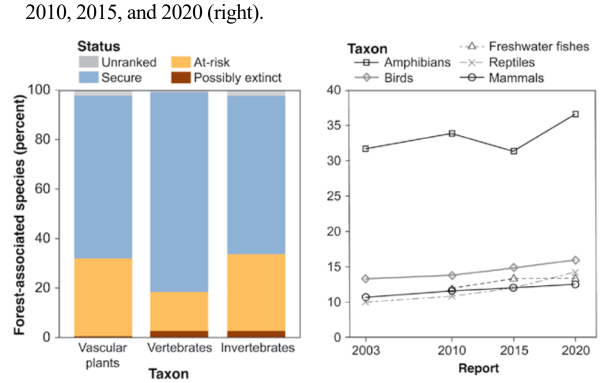
Source: FWS Environmental Conservation Online System (https://ecos.fws.gov/ecp/report/boxscore).

Forest-Associated Species at Risk of Decline

Forest-associated species are tracked by NatureServe through a habitat matrix domain table developed for each species (NatureServe Central Databases, metadata on file with Michael S. Knowles, Rocky Mountain Research Station). We examined forest-associated species that were classified by NatureServe with conservation status G1 (critically imperiled), G2 (imperiled), or G3 (vulnerable). This list includes, but is not limited to, federally listed ESA species. Among forest-associated species of vascular plants, vertebrates, and select invertebrates, 111 (~1 percent) were determined to be presumed or possibly extinct; 5,328 (31 percent) were determined to be at risk of extinction (includes species classified as critically imperiled, imperiled, or vulnerable to extinction); and 12,025 (69 percent) were determined to be apparently secure or were unranked.

At-risk species associated with forest habitats are concentrated geographically in Hawaii, the arid montane habitats of the Southwest, the chaparral and sage habitats of Mediterranean California, and in the coastal and inland forests of northern and central California. The number of possibly extinct and at-risk species is proportionately greatest among vascular plants (32 percent) and select invertebrates (34 percent)—nearly double the percentage observed among vertebrates (19 percent) (figure 10-16, left). Among forest-associated vertebrates, the greatest proportion of possibly extinct and at-risk species is found among amphibians (37 percent). Birds (16 percent), reptiles (14 percent), freshwater fishes (13 percent), and mammals (12 percent) show substantially lower percentages of forest-associated species considered to be at risk (figure 10-16, right).

Figure 10-16. The percent of vascular plant, vertebrate, and select invertebrate species associated with forest habitats determined to be possibly extinct, at risk of extinction, secure, or unranked (left). Change in the percent of forest-associated amphibian, bird, freshwater fish, reptile, and mammal species classified as at-risk using NatureServe global conservation status (G1, G2, G3) as described in the National Report on Sustainable Forests from 2003, 2010, 2015, and 2020 (right).



Source: NatureServe and multiple sustainability reports.

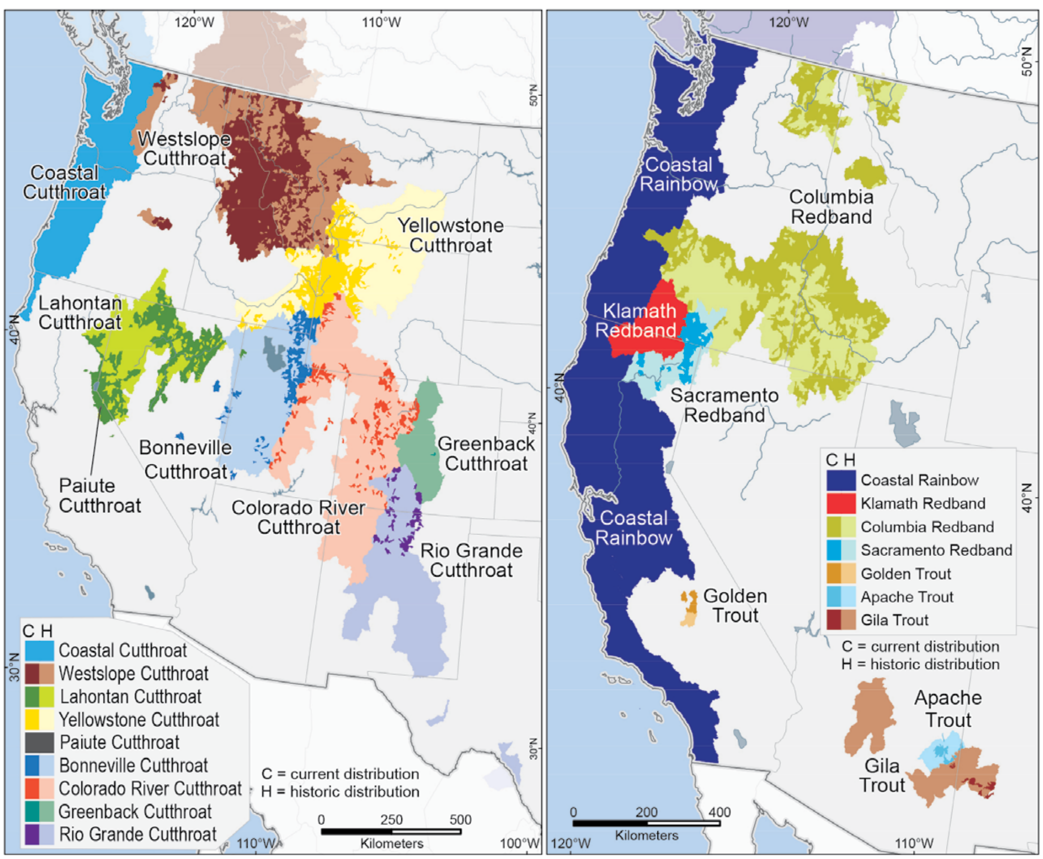
Pacific Trout in the Conterminous United States

Pacific trout (*Oncorhynchus* spp.) are ecologically, socioeconomically, and culturally important. Substantial declines in abundance and contractions in distribution across Pacific trout species and subspecies have led to substantial research and conservation efforts (figure 10-17). Population declines of at least two-thirds from historical levels for some populations have led to elevated Federal protection (under the ESA) and by State management agencies in all or part of their range. Two cutthroat trout subspecies, the Alvord cutthroat trout (*O. clarkii alvordensis*) and the yellowfin cutthroat trout (*O. clarkii macdonaldi*), are considered extinct.

Human influences leading to declines in Pacific trout began with Euro-American colonization of North America (Penaluna et al. 2016). The decline of Pacific trout over recent decades and, in some cases, the last century, reflects the challenges of balancing societal values with natural resources and wild places under a changing climate. Legacies of past overfishing and land-use activities have led to habitat degradation (e.g., from forest harvest, agriculture, cattle grazing, mining, migration barriers, nonnative species, climate change, land development, water withdrawal, etc.). Fortunately, Pacific trout have evolved characteristics including genetic, phenotypic, and life-history diversity, along with long-distance migration to be resilient to large-scale natural disturbances (e.g., wildfire, flood). These characteristics may be the keys to their future persistence. Scientists and managers can work together to consider social pressures that increase vulnerability of Pacific trout and find opportunities to restore species diversity through flexible management.

Brooke Penaluna, USDA Forest Service, Pacific Northwest Research Station

Figure 10-17. Historical and current distributions of Pacific trout in the conterminous United States, with distributions of *Oncorhynchus mykiss* spp. and other Pacific trout (left), and *O. clarkii* spp. (right). Historical distributions (faded colors), represent areas that are no longer occupied.



Source: Modified from Penaluna et al. 2016.

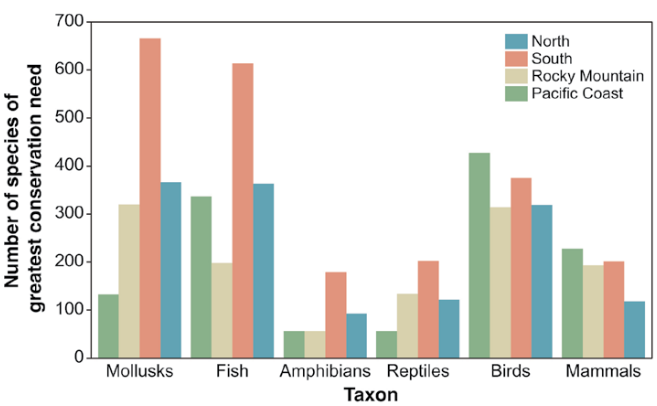
State-Level Species of Concern

In addition to Federal agencies, individual States and Tribal governments also conduct wildlife and fish management conservation. Maintaining an approved State Wildlife Action Plan (SWAP; https://www.fishwildlife.org/afwa-informs/state-wildlife-action-plans) is a prerequisite to receiving FWS State and Tribal Wildlife Grant Program funding. All States completed initial SWAPs in 2005 and updated SWAPs in 2015. Each SWAP identifies Species of Greatest Conservation Need (SGCN) within a State—many of which are not federally listed—as well as their key habitats and threats, and actions needed to conserve them. To illustrate the diversity of species of importance to each State, we report the numbers of SGCNs by taxonomic group, State, and RPA region, based on a USGS compilation of SWAP data.

Not all States considered all taxonomic groups, nor did they consider all species within taxonomic groups when designating SGCN. For example, few States included plant species. Therefore, we present information for amphibians, fish, mollusks, reptiles, birds, and mammals for more consistent comparisons among States.

A compilation of SWAPs across all 50 States and 5 territories revealed 4,723 SGCN in 2005 and 4,484 SGCN in 2015, resulting in 5,525 distinct SGCN. The 2015 SWAP taxonomic breakdown included 289 amphibians, 865 birds, 1,180 fish, 518 mammals, 1,253 mollusks, and 379 reptiles. The RPA South Region had nearly twice as many SGCN compared to each of the other three major RPA regions (figure 10-18). The South Region also had the most SGCN for amphibians, fish, mollusks, and reptiles; bird and mammal SGCNs were most numerous in the Pacific Coast Region (figure 10-18).

Figure 10-18. Count of species of greatest conservation need listed in State wildlife action plans by RPA region, 2015.



Data collected by U.S. Fish and Wildlife Service and compiled by U.S. Geological Survey.

Threats to Biodiversity

- ❖ The aggregate index of land use stress shows the stark difference in pressure in eastern compared with western watersheds. Eastern watersheds face compounded pressures from mining, energy development, nitrogen deposition, and roads, whereas high-risk watersheds in the Western United States face pressure mainly from development in large metropolitan regions.
- ❖ Although the Pacific Coast and Rocky Mountain Regions exhibit low levels of aggregate land use stress, pockets of high stress occur in the human population and agricultural centers of Washington, Idaho, and California, and areas of the Rocky Mountains that are experiencing rapid population growth.
- ❖ Areas of high climate stress were found in all RPA regions. A majority of our climate change models project future high climate stress in the mountains of the Pacific Coast, Rocky Mountain, and South Regions, as well as in large areas from New York to Maine in the North Region. Lower elevation lands in southern New Mexico, southern Arizona, Oklahoma, and Texas are also projected to experience high climate stress.
- ❖ USDA Forest Service and U.S. National Park Service lands are projected to experience higher future climate stress than all other lands, likely due to their location in vulnerable higher elevation areas. These results suggest that the ability of these lands to serve as climate refugia for native biota and ecosystems may be limited.
- ❖ Overlays of climate stress and terrestrial or aquatic biodiversity indicate that places of high climate stress and high biodiversity are commonly found in the North and South RPA Regions, although pockets of high stress and higher biodiversity are also found in the Pacific Coast Region.

Native species have experienced significant losses in habitat owing to a wide variety of threats derived from human-driven land use and management (Foley et al. 2005) that will likely be compounded in the future as the stress on native ecosystems increases under a changing climate (Manytka-Pringle et al. 2015). In this section, we describe ongoing stressors affecting native ecosystems, including invasive species, pathogens, and land use alteration—with a focus on urban areas, agriculture, mining, pipelines, and energy development. We also include a section examining ecosystem stress related to climate projections under future scenarios. Finally, we overlay existing land use stress with modeled climate stress and consider climate stress relative to distributions of native terrestrial and aquatic biota. These comparisons highlight the different stressors currently affecting specific regions, and places of future concern, particularly considering biodiversity distributions.

Biological Drivers of Change

Invasive species and pathogens are significant drivers of change in biodiversity across the United States and are anticipated to increase in influence as individual species respond to anthropogenic drivers of land use and climate change. Neither of these topics have been addressed in prior RPA reports. In this report, we introduce these two drivers of ecological change, and describe some of the mechanisms through which they alter ecosystems. We did not model either invasive species spread or pathogens but acknowledge that such efforts would be highly informative for local, regional, or national management planning and implementation.

Invasive Species

Invasive species in the United States are highly diverse and represent every taxon. More than 6,500 invasive species have been documented as currently established (see USGS, https://www.usgs.gov/programs/invasive-species-program; OTA 1993), and each species' level of impact is as varied as the organism. Invasive species are defined here as plentiful, nonnative organisms that negatively affect the area they inhabit (Beck et al. 2008, Colautti and MacIsaac 2004). Invasive individuals and species compete for resources and predate native species (Doherty et al. 2016, Doody et al. 2009, Dugger et al. 2011, Salo et al. 2007), and may also interbreed and hybridize with related native organisms (Huxel 1999, Muhlfeld et al. 2017). Invasive species contribute to habitat change and destruction, which can lead to changes in trophic structure or disturbance regimes (Johnson et al. 2009, Sousa et al. 2009, Mack and D'Antonio 1998, Vitousek 1990). In the most severe cases, introduced

Herpetofauna Diversity and Threats

Aquatic-dependent herpetofauna enrich freshwater, riparian, and moisture-rich ecosystems. There are more than 300 amphibian species in the United States, with 70 percent being endemic. Notably, the United States is the global hotspot for salamander biodiversity (198 species), with Appalachian and Pacific Northwest forests having particularly unique communities. Among the world's largest salamanders are the awe-inspiring hellbenders (*Cryptobranchus alleganiensis*) (to ~29 inches) and common mudpuppy (*Necturus maculosus*) (to ~17 inches) of Eastern U.S. waters, and the northwest stream-breeding Pacific giant salamanders (*Dicamptodon* spp.), the largest terrestrial-occurring salamander (to ~12 inches). Of the more than 300 U.S. reptiles, freshwater-dependent species include turtles, snakes, and alligators. The United States is second in the world in freshwater turtle biodiversity, with most of its 57 species occurring in the Southeast. Both amphibians and reptiles are centrally positioned as both predators and prey in food webs, being critical cogs of complex trophic systems, cycling energy and nutrients between water and land. Some amphibians play a role in carbon sequestration from the atmosphere (Best and Welsh 2014, Semlitsch et al. 2014), and both amphibians and reptiles have been pivotal for biomedical research. Their ecosystem services span a variety of aesthetic, cultural, educational, recreational, food, medicine, and other product categories.

Herpetofauna epitomize the ongoing sixth mass-extinction event on Earth (Wake and Vredenburg 2008), with 40 percent of world amphibians (IUCN 2020a) and over half of world turtles and tortoises (Stanford et al. 2020) threatened with extinction. Although U.S. herpetofauna are faring better, about 27 percent of freshwater turtles, 21 percent of salamanders, and 13 percent of frogs and toads are threatened (IUCN 2020b). Although habitat loss is pervasive, diseases and climate change are emerging threats that are gaining conservation concern for management action (Bletz et al. 2022, Olson and Pilliod 2022, Wogan et al. 2022). Snake fungal disease, snake lung parasites, turtle shell disease (fungal pathogen), turtle aural abscesses, turtle and amphibian ranavirus infections, amphibian chytridiomycosis, and trematode infections are among leading concerns (NWDC 2021, PARC 2021). Forestalling human-mediated disease transmission is a top biosecurity priority, especially for aquatic pathogens that are also invasive species such as the chytrid fungi (Julian et al. 2020; NWCG 2017, 2020; Olson 2022). For aquatic herpetofauna in the United States, climate change is projected to alter amphibian, reptile, and pathogen habitat macro- and micro-refugia, generally moving optimal conditions northward and higher in elevation, raising concerns for rare species already threatened by habitat loss and fragmentation, and requiring proactive retention of predicted habitat strongholds and corridors for dispersal (Olson 2022).

Deanna H. Olson, USDA Forest Service, Pacific Northwest Research Station

species can lead to local extirpation or extinction of native species (Clavero and Garcia-Berthou 2005, Huxel 1999): Impacts of invasive species were important in 68 percent of the extinctions of North American freshwater fishes in the past 100 years (Miller et al. 1989).

The damage invasive species inflict on economic and ecosystem services humans use is also extensive (Born et al. 2005, Charles and Dukes 2008, Poland et al. 2021). The monetary cost of invasive species management is estimated at approximately $120 billion each year (Pimentel et al. 2005), with the crop damage from European starlings alone estimated at $800 million in the year 2000 (Linz et al. 2007). Understanding of invasive species is both incomplete and important for management of natural systems of the United States (Rahel et al. 2008). Invasive species are anticipated to increase in the future, as will their compounded interactions with climate change, increased human pressures on natural systems, and other forces (Muhlfeld et al. 2014, Rahel and Olden 2008).

Pathogens

Pathogens are a serious risk to fish, game, and wildlife in the Nation's forests (Wobeser 2007, Woo et al. 2006). In addition to direct impacts on individuals and species, disease can interact with human-caused environmental changes and alter ecosystems (Brearley et al. 2013, Daszak et al. 2001, Hilker and Schmitz 2008, Tompkins et al. 2011). Disease pathogens are defined here as any fungal, bacterial, or viral microorganism causing dysfunction in structure or function of the body (Balloux and van Dorp 2017, Scholthof 2007) as well as trematode parasites that can cause limb malformations in anurans (Johnson et al. 2002).

There are a variety of ways that pathogens interact with wildlife communities. Endemic or preexisting diseases may change in rate or intensity (Price et al. 2019, Rachowicz et al. 2005). Diseases that have not previously been seen in a population or species may arrive via transmission from another group, or by evolving into a new pathogen (Daszak et al. 2000, Kock et al. 2010, Rachowicz et al. 2005). These new pathogens vary in the rate of transmission and spread within and between wildlife populations (Gallana et al. 2013, Van Hemert et al. 2014, see the sidebar Herpetofauna Diversity and Threats). These patterns are influenced by human activities such as land management and the interactions of domesticated and wild animals (Bradley and Altizer 2007, Brearley et al. 2013, Daszak et al. 2001) because of the potential for disease transmission between these groups (Daszak et al. 2000, Miller et al. 2017, Pedersen et al. 2007). The existence of multiple hosts, parasites, and life stages, however, can complicate transmission (Rachowicz et al. 2005, Van Hemert et al. 2014).

The effects of disease on wildlife can range from mild to catastrophic. Although some individuals may reproduce and live long lives with few symptoms of a disease, these carriers can be responsible for extensive disease transmission (Artois et al. 2009, Garwood et al. 2020). On the other end of the spectrum, some pathogens have led to entire species becoming endangered or even extinct (Pedersen et al. 2007, Skerratt et al. 2007, Smith et al. 2006). For example, White Nose Syndrome, caused by the fungal pathogen *Pseudogymnoascus destructans*, has decimated populations of bats in the conterminous United States (Cheng et al. 2021, Hoyt et al. 2021). Severe disease outbreaks may influence the workings of entire ecosystems or influence disturbance regimes (Hilker and Schmitz 2008, Tompkins et al. 2011).

The severity of disease is predicted to increase in the future as introduced species bring novel pathogens into native populations, climate change influences species ranges (including those of pathogens), and human alterations to landscapes increase disease transmission (Brearley et al. 2013, Hemert et al. 2014, Price et al. 2019, Wilkinson et al. 2018, Young et al. 2017). The wildland-urban interface and other areas of increased interaction between humans, domestic animals, and wildlife are especially likely to lead to increased transmission of disease (Bradley and Altizer 2007, Deem et al. 2001, Miller et al. 2017, Wilkinson et al. 2018). Given these compounding factors, research is currently lacking on the full complexity of these interactions (Ryser-Degiorgis 2013, Stallknecht 2007) that could help managers prepare for projected increases in outbreaks and severity (Buttke et al. 2021).

Anthropogenic Drivers of Change

Connection and interconnection within and among habitats are crucial for the long-term persistence of native biota. Native species are adapted to the disturbance processes most prominent on a landscape (such as wildfire), assuming they occur within the natural range of variability (Johnstone et al. 2016). Disturbances associated with human land use and development (including wildfire suppression), however, have extensively altered the availability and patterns of habitats at landscape scales, leading to extirpation of some species. Vulnerability to extinction currently exists for hundreds of species, far beyond those listed under the ESA (Harris et al. 2012). The existing landscape will likely be further stressed in complex ways as the effects of climate change provide an additional layer of stress on native ecosystems (Mantyka-Pringle et al. 2015). This section provides modeling work addressing the effects of land use and climate change on native biodiversity, individually and together.

Land Use

Human use and development of landscapes directly affect both terrestrial and aquatic biodiversity. Drivers of biodiversity loss associated with human activities include long-term land cover change, degraded and homogenized wildlife habitats, and the creation of pathways for introduction of nonnative species (Allan 2004, Flather et al. 1998, Howard et al. 2020, Regetz 2003, Wilcove et al. 1998). To assess where future development and expansion may cause increasing stress on native ecosystems, we explored changing patterns of three key themes related to land use change: human population growth and urban development; agriculture; and energy development and mining.

For each of the themes, we created stress indices for HUC 10 watersheds throughout the conterminous United States using principal component analysis (PCA) of variables shown in table 10-2, using the *prcomp* function in the *stats* package in R (R Core Team 2021). PCA is a data-reduction technique that simplifies datasets with many, sometimes-correlated variables, into fewer dimensions (called principal components) to draw out trends and patterns—in this case, to more clearly show variation in the combination of stressors among watersheds. We performed PCA on variables associated with each of the themes individually and for all the variables together (the aggregate index), providing stress indices for each HUC 10 watershed, shown in figure 10-19. Results allow more refined discussion of watershed threats, examining the major drivers and how they vary by region.

Population Growth and Urban Development

The U.S. population is projected to grow between 24 and 44 percent from 2010 through 2070 under intermediate scenarios (Shared Socioeconomic Pathways 1, 2, and 4; Wear and Prestemon 2019). Although much of that growth is expected to be centered on metropolitan regions, the fastest areas of development are on the urban fringe, as large metropolitan regions expand their exurban and wildland interface areas (see the Land Resources Chapter, also Beale and Johnson 1998, Hjerpe et al. 2020, Kirk et al. 2012, Radeloff et al. 2018). Such expansion threatens native ecosystems with habitat fragmentation and decreases in native species diversity. Indirect effects of development include increased amounts of impervious surfaces, roads, noise, and light. Impervious surfaces and roads increase sedimentation and pollutant runoff into streams and modify regional hydrology. Roads also lead to direct mortality of animals, disrupt dispersal and migration, and serve as pathways for invasive species (Bennett 2017, Regetz 2003, Siemers and Schaub 2011). Anthropogenic noise and light pollution have also been linked to declines in wildlife and alterations in behavior (Barber et al. 2010, Carral-Murrieta et al. 2020, Kerth and Melber 2009, Shannon et al. 2016, Siemers and Schaub 2011).

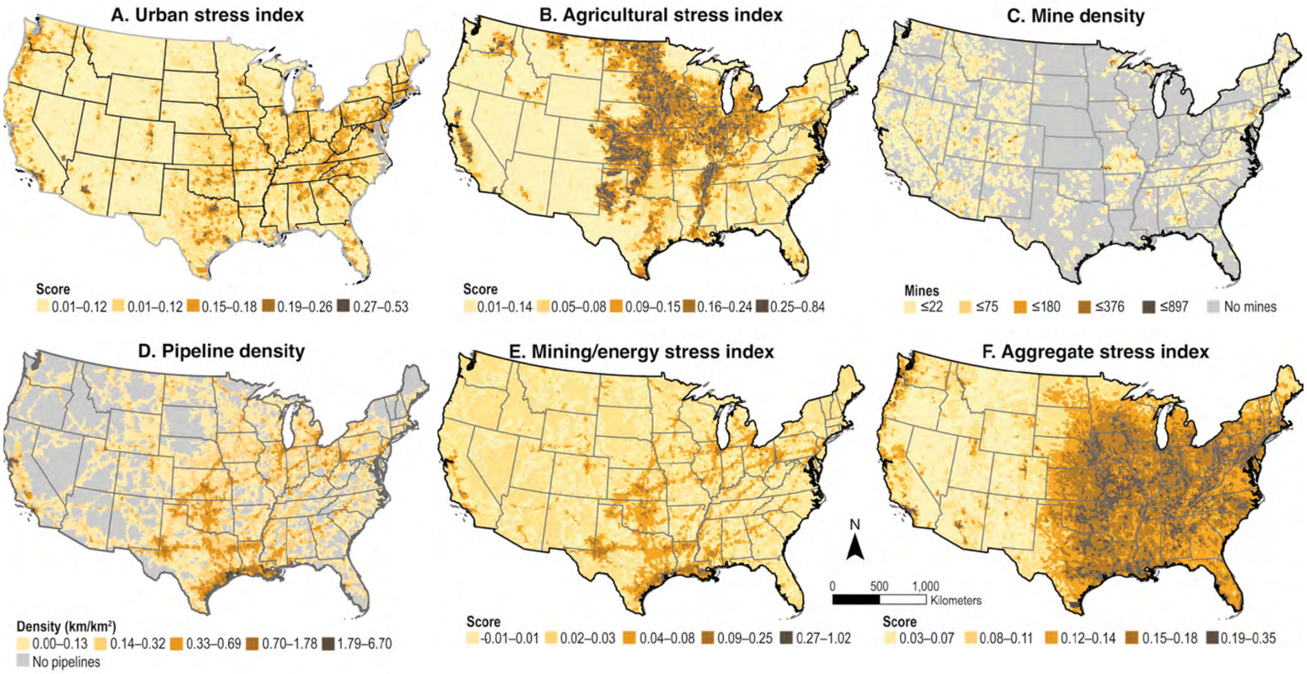
The stress index used for population growth and urban development focuses on housing density, population, and roads (figure 10-19a). PCA for these variables reduced the data to three principal components. The first principal component, accounting for 47 percent of the variation among watersheds, is heavily weighted toward population density and roads, identifying watersheds near cities and large metropolitan regions. Watersheds affected by human activity outside of cities, namely roads in steep areas and crossing streams, explain 18 percent of the variation. Roads fragment habitat by creating barriers to aquatic organism passage, and lead to increased sedimentation of waterways. These high-stress watersheds occur frequently in the southern Great Plains, central Arizona, and mountainous areas across the United States, with particularly high stress in the Appalachians stretching from western North Carolina to Pennsylvania.

Table 10-2. Variables and data sources used in stress indices.

Variables	Data sources
Urban development and population growth	
Developed land use	30-m NLCD 2011 Land Cover (2011 Edition, amended 2014); National Geospatial Data Asset Land Use Land Cover
Housing density (population per km²) **Population change** **Population density**	SILVIS Lab, Dept. of Forest & Wildlife Ecology, University of Wisconsin-Madison (2017)
Road density **Road stream crossings** **Roads on steep slopes**	U.S. Census TIGER 2015 Roads National Geodatabase; USGS 2018 Quarterly National Hydrography Dataset high-resolution (1-to-24,000 or better) stream coverage
Agriculture	
Cultivation on gentle slopes	30-m NLCD 2011 Land Cover (2011 Edition, amended 2014); National Geospatial Data Asset Land Use Land Cover USGS 7.5-minute (~30-m) National Elevation Dataset
Cultivation on highly erosive soils	National Cooperative Soil Survey Web Soil Survey
Mean deposition of N, dry	~4-km National Atmospheric Deposition Program (2018). Total Deposition Maps, v2018.01.
Energy development and mining	
Oil wells	Homeland Infrastructure Foundation-Level Data (2018). Retrieved from https://hifld-geoplatform.opendata.arcgis.com/datasets/oil-and-natural-gas-wells/data. Downloaded 3 Dec 2020.
Pipeline density	National Pipeline Mapping System (2004). U.S. Dept. of Transportation. Pipeline and Hazardous Materials Safety Administration. Washington, DC

NLCD = National Land Cover Database
USGS = U.S. Geological Survey

Figure 10-19. Land use stress at watershed scales from (a) population growth and urban development; (b) agricultural expansion; (c) density of mines; (d) density of pipelines; (e) mining/energy; and (f) aggregate stress across all sectors (at HUC 10 scale). Higher scores indicate greater stress.



The third principal component, explaining 14 percent of the variation in the data, represents watersheds with new residential developments, reflecting trends in urbanization around cities, and for rapidly growing urban areas (e.g., southern California, western Washington and Oregon, central Arizona, Colorado Front Range, central Texas). This factor also captures growth in rural areas, such as parts of the Wasatch Range, Rocky Mountains, and Appalachian Mountains, where amenity-driven growth has led to expansion of housing in the wildland-urban interface, with significant impacts on wildlife habitat.

Agriculture

In human-dominated ecosystems, established agricultural areas can host an abundance of species, including pollinators (Devictor et al. 2008, Krauss et al. 2003, Westphal et al. 2003), but agricultural expansion also converts complex landscapes into simple managed ecosystems, intensifies resource use, and increases pollutants via fertilizer and pesticide application (Donald et al. 2001, Dudley and Alexander 2017, Robinson and Sutherland 2002, Tscharntke et al. 2005). The stress index for agriculture uses variables related to cultivation, erosion potential, and nitrogen deposition (table 10-2). The PCA results for agriculture do not differentiate watershed types as clearly as the development PCA, perhaps because there is more variation in degrees of development than agriculture. The first principal component captures 74 percent of the variation among watersheds and mostly distinguishes watersheds with heavy concentrations of agriculture from those without heavy concentrations. The second principal component, which is almost entirely composed of watersheds without cultivated crops but with high levels of nitrogen deposition, explains 21 percent of the variation in the data and identifies the downstream impacts of agriculture on watersheds. The two principal components together explain 95 percent of the variation in the data and provide a fuller description of risk to watersheds associated with agriculture (figure 10-19b), namely those with agriculture in the watershed and those that experience the indirect effects of runoff from upstream watersheds. Agriculture-related stress is high throughout the Great Plains, Mississippi River basin, eastern Washington, and central California.

Energy Development and Mining

Stress on watersheds from mining activity derives from acid rock drainage, increased movement of pollutants such as metal sulfides following heavy rainfall and floods, and impacts on wildlife populations and movements. Watersheds containing active or abandoned mines are affected by complex interactions of surface and subsurface flows that introduce acidity and metals to the receiving stream. These influences may also emanate from natural sources in the underlying bedrock. Even in areas without active mining, some ecosystems have not completely recovered from mining that took place in the 19th century and continues to impact stream channels, sedimentation, and release of toxic chemicals (Schmidt et al. 2012, Wohl 2006, Wohl et al. 2015). Short-term impacts related to the construction of

Southeastern Crayfish Diversity and Threats

Crayfish are found in a wide range of habitats, including permanent and seasonal riverine and lacustrine habitats, freshwater caves and springs, and terrestrial burrows. Crayfish play a significant role in these ecosystems by processing detritus and macrophytes, which increases the availability of nutrients and organic matter to other organisms; digging burrows, which manipulates and mobilizes substrate, making nutrients and habitat available to other stream organisms; and serving as prey for, or predators on, numerous aquatic animal species, especially some game fishes such as bass and catfish (Holdich 2002, Reynolds et al. 2013). Thus, crayfish influence multiple aspects of ecosystem structure and function. They also serve as a profitable and popular food resource (Larson and Olden 2011).

Globally, crayfish reach their highest level of diversity in the United States (~400 species and subspecies versus 500+ in the world [65 percent]) (Taylor et al. 2019). In the United States, crayfish diversity reaches its pinnacle in the Southeast (~200 species; notably Tennessee, Alabama, and Mississippi) (Richman et al. 2015). Crayfish have a high level of endemism, with over two-thirds of Cambaridae species (the family of 99 percent of North American crayfishes) endemic to the Southeastern United States (Simon 2011). Crayfish are imperiled over much of their range, with 50 percent of all U.S. species at some level of conservation concern (Taylor et al. 1996, 2007) and 22.5 percent listed in threatened or higher concern categories (Taylor et al. 2019). Although interest in crayfish conservation is rapidly growing, basic biological and ecological information is lacking, severely hindering our ability to assess crayfish status and manage crayfish populations (Barnett 2017, Loughman and Fetzner 2015, Moore et al. 2013).

Crayfish imperilment is often attributed to small range sizes and degradation of habitats through pollution, urban development, and dams/water management (Crandall and Buhay 2008, Richman et al. 2015); however, there is a paucity of studies that directly assess these threats to southeastern crayfishes. Existing studies assessing dams/water management show that small dams shift the relative abundance of stream crayfishes (Adams 2013, Barnett and Adams 2021, Barnett et al. 2022) and large dams decrease the density and diversity of stream crayfishes when compared to streams without dams (Barnett 2019, Barnett and Adams 2021, Barnett et al. 2022). Dams have also caused genetic fragmentation of stream crayfish populations (Barnett et al. 2020, Barnett and Adams 2021). Similarly few studies have assessed the impacts of threats to crayfish—the Comprehensive Everglades Restoration Plan is the only well-known major restoration initiative that focuses on crayfishes (Taylor et al. 2019). Because of their important role as ecosystem engineers and both predator and prey in aquatic and terrestrial ecosystems, crayfish conservation also protects other organisms of conservation concern, as well as critical ecosystem functions (Boyle et al. 2014, Reynolds et al. 2013, Wolff et al. 2015).

Zanethia C. Barnett, USDA Forest Service, Southern Research Station

wells and pipelines include increased turbidity, modification of aquatic habitats, and opportunities for fuels and chemicals to enter the system during the construction stage (Maloney et al. 2018, Reid and Anderson 1999, Reid et al. 2003). Oil and gas wells, as well as development and roads, have also been shown to alter wildlife movements and migration patterns across broad spatial scales (Jakes et al. 2020). Long-term impacts of oil and gas development include channel incision and lateral movement, along with the potential for catastrophic impacts from a future spill. Pipelines might make hundreds or thousands of stream crossings and intersect a wide variety of habitats (Levy 2009), with the potential for spills to move contaminants through large river systems. Even small releases of oil and gas have been shown to injure and kill wildlife (Ramirez 2010, Ramirez and Mosley 2015). Levy (2009) found that there were 762 pipeline failures per year on average in Alberta, Canada over a 15-year period. In the United States, there were 614 pipeline incidents of some kind reported in 2019 (FracTracker.org, accessed 26 January 2021). An analysis of about 7,000 U.S. onshore liquid pipeline incidents that occurred from 1985 to 2012 found 5.5 percent of all incidents were triggered by natural hazards, 28 percent led to releases into water bodies, and more than 20 percent resulted in fires (Girgin and Krausmann 2016).

The PCA for energy development and mining aggregates the mining, well, and pipeline data into one threat index, identifying parts of the country that have significantly more energy and mining activity than the rest of the country. Mining is widespread throughout the West, the Ozarks, parts of the Great Lakes, and the Appalachians stretching into the Northeast (figure 10-19c), whereas the density of pipelines increases toward the Gulf Coast of Texas and Louisiana, although pipelines also crisscross much of the Eastern United States (figure 10-19d). The mix of mines and pipelines leads to stress related to energy development distributed throughout the United States, with hotspots occurring in the West, Ozarks, and Southeast (figure 10-19e)—areas that have experienced impacts on terrestrial and aquatic species (Allert et al. 2009, Jakes et al. 2020). Of note are the large parts of the country where data are missing. Availability of data on mines and wells is limited, collected inconsistently by States. Lack of availability of data is a serious limitation to assessing the impacts the energy sector might have on watersheds and wildlife.

Aggregate Index of Land Use-Driven Stress

We calculated an aggregate PCA using all the variables in the individual indices. The first three principal components of this aggregate index cumulatively account for about 55 percent of the variation in watershed stressors, with the individual components accounting for 31, 14, and 10 percent of variation in the data, respectively. The aggregate index shows the stark difference in pressures faced by eastern and western watersheds. Although mining might have significant impacts on the watersheds in which it occurs, energy development is not widespread enough to score high at a large regional scale, especially compared to impacts from development and agriculture. The largest source of variation between watersheds comes from nitrogen deposition and roads; collectively, they affect most of the watersheds in the Eastern United States (figure 10-19f). High-risk watersheds in the Western United States are mainly near the large metropolitan regions. The Pacific Coast and Rocky Mountain Regions generally score low on aggregate stress, but increased stress exists in the population and agricultural centers of Washington, Idaho, and California, and pockets of the Rocky Mountains that are experiencing rapid population growth.

Climate Change

Climate change is affecting terrestrial and aquatic vertebrate species in the United States, resulting in large-scale shifts in their range and abundance (Howard et al. 2020, Lipton et al. 2018). At the scale of the conterminous United States, annual average temperature has increased 1.8 °F (1.0 °C) over the period 1901 to 2016 and is projected to rise by about 2.5 °F (1.4 °C) by 2050 under all plausible futures (USGCRP 2017). Projected changes in total annual precipitation vary geographically. In addition, climate change is projected to continue altering natural disturbances such as wildfire (see the Disturbance Chapter). The total area of the country affected by wildfire has increased annually and this increase is projected to continue under climate change (Westerling 2016). Wildlife habitat will be affected by the interaction of changes in climate and natural disturbances (Weiskopf et al. 2020).

Model Inputs

We explore the potential effects of climate change on wildlife habitat at the scale of the conterminous United States using the 10 climate projections developed for the RPA Assessment, two disturbance treatments (fire suppression and no fire suppression), and the MC2 dynamic global vegetation model. The Scenarios Chapter describes the development of the climate projections using the two climate scenarios from the Intergovernmental Panel on Climate Change (RCPs 4.5 and 8.5, representing lower and high atmospheric warming, respectively) and five climate models identified for use in the RPA Assessment. Climate projections under RCP 4.5 and 8.5 were drawn from the downscaled climate dataset for each of the five climate models, resulting in 10 unique climate projections. The core set of five climate models were selected from the available models based on their ability to capture the range of temperature and precipitation change at mid- and end-century under RCP 4.5 and 8.5 (table 10-3, also see the Scenarios Chapter for more information).

Future vegetation biomass and shifts in vegetation types were assessed by the dynamic global vegetation model MC2—a model that projects vegetation response to changes in temperature, precipitation, and disturbance (fire, drought) based on biogeographic and biogeochemical processes in ecosystems (Bachelet et al. 2016, Klemm et al. 2020). Two disturbance treatments—fire suppression and no fire suppression—were analyzed using the MC2 model under each of the 10 climate projections, resulting in a total of 20 plausible future projections. Wildfire is a disturbance of concern because it can change a landscape in a short period of time, in contrast with gradual changes in mean climate. The "no fire suppression" disturbance scenario decreases the fire-return interval and allows for a potential full range of changes in future fire dynamics, whereas

Table 10-3. Five climate models selected to reflect the range of the full set of 20 climate models in the year 2070. Each model was run under RCP 4.5 and RCP 8.5, providing a range of different U.S. climate projections.

	Least warm	Hot	Dry	Wet	Middle
Climate model	MRI-CGCM3	HadGEM2-ES	IPSL-CM5A-MR	CNRM-CM5	NorESM1-M
Institution	Meteorological Research Institute, Japan	Met Office Hadley Centre, United Kingdom	Institut Pierre Simon Laplace, France	National Centre of Meteorological Research, France	Norwegian Climate Center, Norway

RCP = Representative Concentration Pathway
Source: Joyce and Coulson 2020.

the "fire suppression" scenario can extend the mean fire-return interval in some ecosystems (Sheehan et al. 2015). The conterminous climate and MC2 base data were converted to equal-area grids at 4-km resolution.

The MC2 model assesses the effects of climate and disturbance; historical vegetation is assumed to be potential vegetation. Land use change was not included in the MC2 runs, meaning that the inevitable future land use change linked to socioeconomic changes (see the Land Resources Chapter) was not analyzed here. Areas of open water, developed, and barren land cover—as defined by the USDA Natural Resources Conservation Service (Homer et al. 2020) and approximating 11 percent of the conterminous area—were removed from the analysis. Changes in land use will likely result in added stress on terrestrial and aquatic ecosystems in the United States as urban landscapes expand and rural refuges disappear. We address the implications of this omission in our caveats.

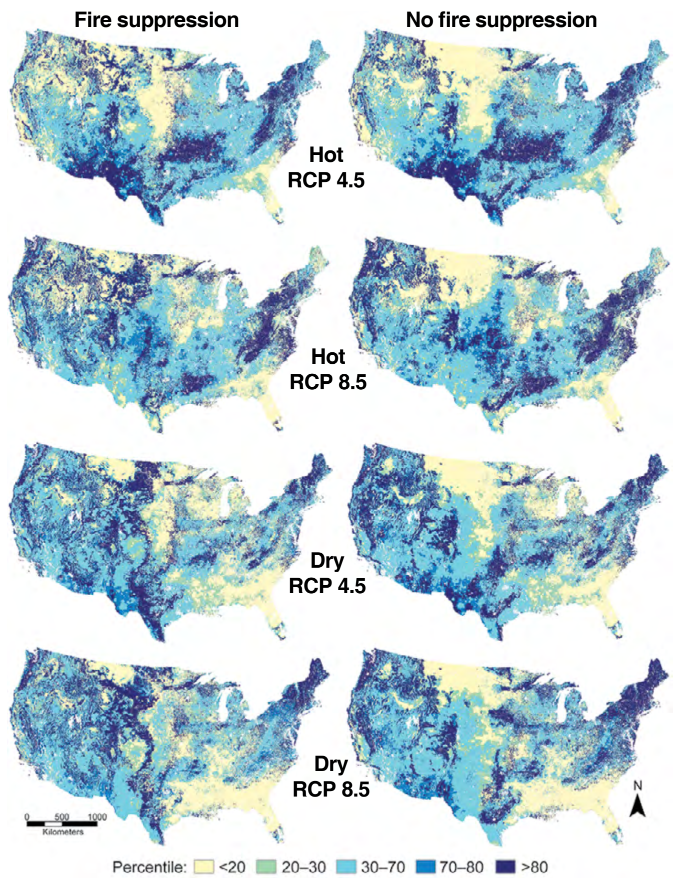
Terrestrial Climate Stress Index

The Terrestrial Climate Stress Index (TCSI) is used to simultaneously explore changes in annual mean temperature (°C), total annual precipitation (cm), and annual vegetation production (grams carbon per m^2) between historical (1951 to 2000) and future (2050 to 2099) periods for each of the 20 projections. Deviations in location and variance between historical and future distributions are captured by calculating Bhattacharyya distance (Bhattacharyya 1946), where larger distances equate to larger differences between the historic and future periods. We use the Bhattacharyya distances as the TCSI scores to rank the relative differences. We define high stress as the 20 percent of cells with the highest TCSI score (most difference between time periods) for each of the 20 projections, similar to many classifications of drought. This index therefore describes the magnitude of departures from historical conditions and identifies where future high stress is projected across the conterminous United States. Exploration of an individual projection (e.g., the hottest or the driest of the suite of projections) highlights areas where that particular stressor may result in challenges to natural resources under those plausible futures, whereas the TCSI based on all projections portrays consistency in projected future high-stress locations across the conterminous United States (see following section).

High Terrestrial Climate Stress Areas Under the Hot Projections

The hot climate model was run for RCP 4.5 and RCP 8.5, under the two disturbance treatments (fire suppression, no fire suppression) to identify areas of high stress under a hot future (figure 10-20). High stress is defined as a TCSI value greater than the 80th percentile (i.e., top 20 percent). Because these scores are relative to each projection, the shifts demonstrate how stress changes depending on the RCP scenario and disturbance treatment. Comparing the results across these four hot-model

Figure 10-20. Terrestrial Climate Stress Index (TSCI) scores ranked by percentile. High stress is a TCSI score greater than the 80th percentile (dark blue). Projections shown are under the hot model (top two rows) and dry model (bottom two rows). Projections include two RCP scenarios (RCP 4.5 and 8.5) and two disturbance treatments (fire suppression: left column, and no fire suppression: right column).



RCP = Representative Concentration Pathway.

projections suggests that although there are common areas of high stress across the four treatments (Appalachian Mountains in the South Region), individual RCPs and disturbance treatments can influence the location of high stress across the conterminous United States.

The northern area of the Rocky Mountain Region is projected to be in high stress under the fire suppression treatment (both RCPs); however, few grid cells are projected to be in high stress under no fire suppression (both RCPs). Historical fire regimes in this area have restricted the advance of woody species. Without future fire suppression, wildfire will continue to minimize woody species advances. The addition of fire-suppression treatments, however, could enable the advance of woody species and result in higher future stress (also found by Klemm et al. 2020). Wildfire management may need to consider the shifting changes in fire regimes under climate change and the future role of prescribed fire under those changes.

A noticeable difference between the RCP 4.5 and 8.5 scenarios occurs in the southwestern area of the Rocky Mountain Region. High stress is found in this southwestern area under RCP 4.5 (both disturbance treatments) but not under RCP 8.5 (both disturbance treatments), suggesting that atmospheric warming has a greater influence on stress in these areas than fire-suppression treatments.

High Terrestrial Climate Stress Areas Under the Dry Projections

The dry climate model was run for RCP 4.5 and RCP 8.5, under the two disturbance treatments (fire suppression, no fire suppression) to identify areas of high stress under a dry future (figure 10-20). Comparing the results across these four projections suggests that although common areas of high stress are visible across the four projections (northeastern area of the North Region), individual RCPs and disturbance treatments can influence the locations of high stress across the conterminous United States.

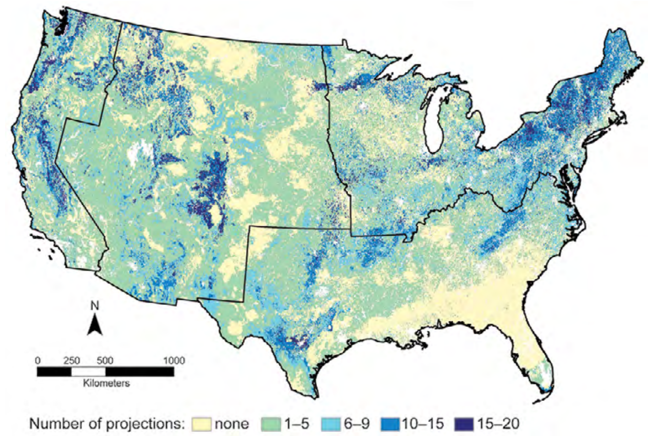
The high-stress projections under the dry model are similar to the results of the hot model for the Rocky Mountain Region: high stress in the northern part of the region is projected for both RCPs under fire suppression (covering a larger area than under the hot model) and relatively low stress is projected under no fire suppression. Changes in fire regimes under climate change may be an important consideration for resource management in these areas. All four projections predict future high stress in the northeastern area of the North Region. In contrast to the hot-model projection, where high stress was seen in the Appalachian Mountains (extending into the northeastern area but not including Maine), high stress in the dry model intensifies in the far northeastern area across all four projections. Further exploration of drought in this area may be valuable for local resource planning. Also common to all four projections is the relatively large area without high stress in the southern parts of the South Region. Stress is still evident, but when compared to other areas, this relatively wet region is projected to have few areas of high stress under the dry-model projections.

Consistent with the hot-model projections, areas of high elevation are projected to experience high stress under the dry-model projections. These common areas of high stress include the mountains throughout the Pacific Coast and Rocky Mountain Regions. Higher elevations in the eastern part of the conterminous United States appear to experience more stress under hot projections than dry projections.

High Terrestrial Climate Stress Trends Across Projections

Grid cells that are consistently ranked as high stress across the projections denote areas that are most likely to experience high stress in the future, based on this suite of plausible futures. Figure 10-21 shows the number of

Figure 10-21. The cumulative number of projections that identify future high stress for every cell, based on the set of 20 projections. Barren area, open water, and developed areas (white on the maps) are not included in the analysis.



projections that identify future high stress in each cell, out of the 20 total projections. For the conterminous United States, we define areas of concentrated stress (hotspots) as occurring where 10 or more projections identify high stress.

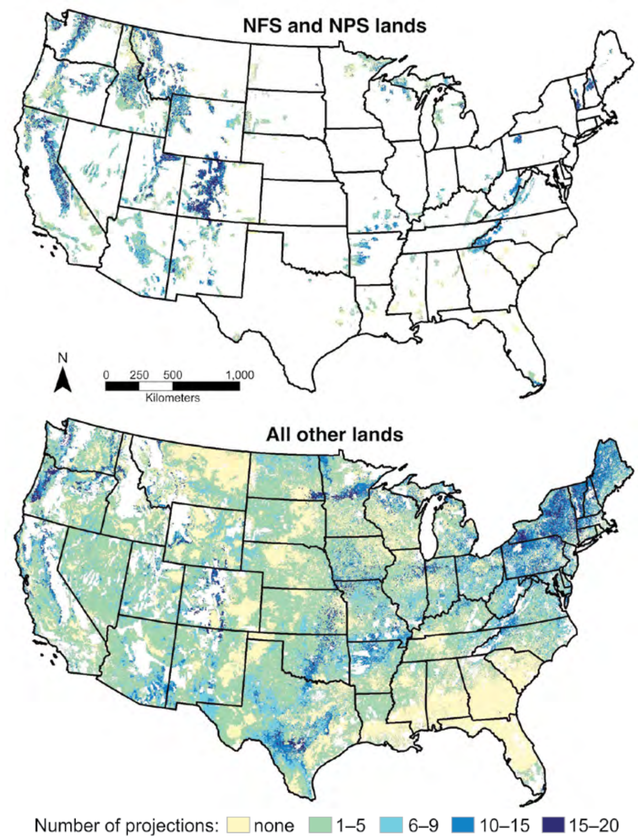
Areas of concentrated high stress occur in all RPA regions. High-elevation areas are consistently ranked as high stress, including the mountains of California, Oregon, and Washington in the Pacific Coast Region; the Rocky Mountains (Montana through Colorado) in the Rocky Mountain Region; areas in the Appalachian Mountains and Ozarks in the South Region; and northeast mountains in the North Region. Scattered arid lands in southern Arizona and New Mexico in the Rocky Mountain Region consistently show high stress. Areas in eastern Oklahoma and parts of Texas, and central Minnesota also see high stress.

Stress Projection Comparisons: National Forests and U.S. National Park Service Lands, Compared with All Other Lands Across the United States

Land management under Federal ownership includes a variety of objectives including both conservation and resource extraction, but these lands are less vulnerable to development or land conversion compared with other land ownerships. Federally owned lands therefore have the potential to play an important role as climate refugia when considering climate vulnerability. To better understand how climate change may affect Federal lands and their potential to serve as climate refugia, we compared stress projections for National Forest System (NFS) and U.S. National Park Service (NPS) lands with stress projections for all other lands (both public and private, but not necessarily set aside as protected) in the conterminous United States (figure 10-22).

Looking at the TCSI results for NFS and NPS lands shows high stress areas across these networks (figure 10-22, top). Concentrated high stress is seen in the Rocky Mountain and Pacific Coast Regions, notably the central/southern Rocky

Figure 10-22. The number of cumulative projections that identify future high stress for (top) National Forest System and U.S. National Park Service lands, and (bottom) all other lands, based on the set of 20 projections.



NFS = National Forest System; NPS = U.S. National Park Service.

Mountains, northwestern Utah, parts of Montana, and the Sierras in California. High stress on all other lands occurs as isolated areas of high stress across the country and concentrated in the northeastern area of the North Region, southern Texas, and the Coast Range in Oregon (figure 10-22, bottom).

We averaged TCSI for all 20 projections (1) across NFS and NPS lands and (2) across all other lands (not part of the NFS or NPS) and tested for differences using a paired t-test. Future stress is significantly greater for NFS and NPS lands than for all other lands, based on changes in climate and disturbance (p-value <0.0001; $t = 5.07$, $df = 19$).

Because elevation is confounded with temperature, future warming temperatures mean that higher elevation areas are more likely to be in high stress. NFS and some NPS lands largely exist in the mountainous regions of the country at higher elevations than other lands (mean elevation of 1542 m versus 709 m), a driving factor for the significantly higher stress found in these lands. However, the correlation between elevation and the number of projections identifying a cell in high stress across the conterminous United States is weak ($r = 0.15$), suggesting that other factors including wildfire suppression also contribute substantially to this result.

Implications of Terrestrial Stress

Climate change has already had negative impacts on many threatened wildlife species (Pacifici et al. 2017), and it will likely continue to adversely affect many more species as projected increases in temperature and variation in precipitation lead to changes in vegetation and fire regimes. As in our previous assessments, these results also suggest that the historical influence of fire on vegetation is an important consideration. The effect of a changing climate and a changing fire regime differs depending on the management of wildfire, particularly where fire regime has been a major influence in sustaining a specific vegetation type (e.g., grasslands). Large, high-severity fires can lead to more heterogeneous landscapes that provide a mosaic of habitat types facilitating greater biodiversity (e.g., grassland birds; Fuhlendorf et al. 2006). However, there are many feedback cycles associated with fire and confounded with climate change. For example, cheatgrass (*Bromus tectorum*), an invasive plant species of particular concern in the Great Basin owing to its negative impact on wildlife species in sagebrush ecosystems, has been shown to have a positive feedback cycle with fire (Coates et al. 2016). The mountain pine beetle (*Dendroctonus ponderosae* Hopkins) is another example in the Rocky Mountains where warming temperatures have altered the beetle's lifecycle, leading to emergence of more individuals simultaneously, which in turn leads to more successful attacks and more dead trees that are more likely to result in large crown fires (Logan and Powell 2001). These cascading effects threaten trophic interactions that have adapted to historical norms and are rapidly changing at a pace wildlife species may not be able to keep up with.

Federal lands are expected to serve as future refugia if surrounding lands are converted to other land uses. Although our study did not incorporate land use as a factor affecting vegetation, our finding that NFS and NPS lands are projected to experience higher future climate stress than all other lands suggests that the ability of these lands to serve as refugia may be limited by climate stress. Nevertheless, Federal lands may offer important landscapes through which wildlife can disperse in response to changing environmental conditions resulting from climate change.

We identified several areas where a majority of the plausible futures predict high stress: mountains in the Pacific Coast, Rocky Mountain, and South Regions; large areas from New York to Maine in the North Region; and lower elevation lands in southern New Mexico, southern Arizona, Oklahoma, and Texas. The consistency of high stress in these areas suggests that wildlife managers will likely see changes in wildlife habitat and wildlife distributions. For example, decreased ranges for wildlife that are particularly vulnerable to heat stress, such as marten (*Martes americana*) and lynx (*Lynx canadensis*), may lead to associated population declines (Carroll 2007). Other indirect effects such as increased parasitic vulnerability could also contribute to population declines in many wildlife species, such as winter ticks (*Dermacentor albipictus*) on moose (*Alces alces*) (Rodenhouse et al. 2009).

The local exploration of different individual climate projections (least warm, hot, dry, wet, middle) may assist resource managers in identifying the significant impacts on wildlife and wildlife habitat with respect to a particular plausible future (Lawrence et al. 2021). There is value in examining individual projections, as vegetation sensitivity to changes in climate varies across the conterminous United States. In our examination of the hot and dry projections, it was apparent that atmospheric warming (RCP 4.5 versus RCP 8.5) can have more influence than disturbance treatments on high-stress projections and vice versa. The notable example is fire suppression in the northern part of the Rocky Mountain Region, where woody species expanded under fire suppression but not under no fire suppression. Assessing the potential changes in wildfire regimes will be important in planning natural resource management in the future, as will tailoring strategies to the circumstances that characterize various landscapes. Given the large range of habitat types identified to be under high stress, climate change will likely be an important component of future habitat management plans, specifically addressing possible mitigation strategies that are tailored to individual conservation needs.

Combining Stressors

Interactions among individual stressors that affect ecosystem health can compound vulnerability of already altered ecosystems and the biodiversity they support. In the previous sections of this report, we described the modeling and mapping of stressors from land use and climate change. Here, we combine land use stress, future climate stress, and biodiversity data in order to visualize overall distributions of current and future risk to ecosystems across the conterminous United States. Future climate stress in these sections is the aggregate of all the climate prediction models (n = 20) that identified a specific cell as high stress, as described above in the Terrestrial Climate Stress Index section.

Anthropogenic Stressors

Our assessment of anthropogenic drivers of change (described in detail above) captures current stressors on the landscape resulting from (1) land use and associated human activities and (2) changing environmental conditions resulting from climate change. The land use stressor assessment identified the Eastern United States as highly vulnerable to several individual stressors, and with greater overall stress than the Western United States. In contrast, several areas of high stress in response to future climate conditions were identified in the Western United States. To visualize the interaction between these forms of ecosystem stress representing current and future conditions, we combined the datasets using the Plus tool in ArcGIS Pro 2.8.3. Combining these two indices allows us to see where both identify similar geographic patterns and which areas of the conterminous United States are likely to experience compounding stressors. The resulting map identifies areas of high combined stressors in the North Region, along with areas of concentrated combined stress in much of Colorado, southern Texas, and the Sierra Nevadas of California (figure 10-23).

Figure 10-23. Stress presented as an index for: future climate vulnerability—defined as the number of climate models that identified an individual cell as high stress (map in upper left); current aggregate land use impacts (map in upper right); and a combination of the two indices developed using the Plus tool in ArcGIS Pro 2.8.3 (map in lower center).

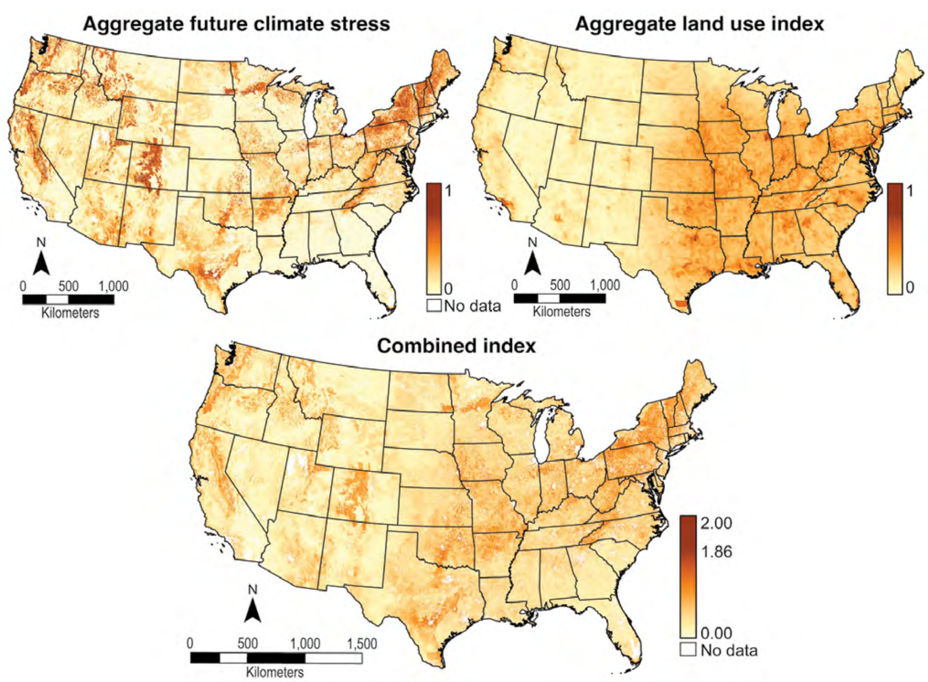


Figure 10-24. Hotspots with both high terrestrial biodiversity and a likelihood of high future stress. Overlaying an index of future climate change (map in upper left)—defined as the number of climate models that identified an individual cell as high stress—on terrestrial biodiversity (map in upper right) results in a map that identifies hotspots with both high terrestrial biodiversity and a likelihood of high future stress (map in lower center). Future climate stress and source information for biodiversity map described in prior sections of the chapter.

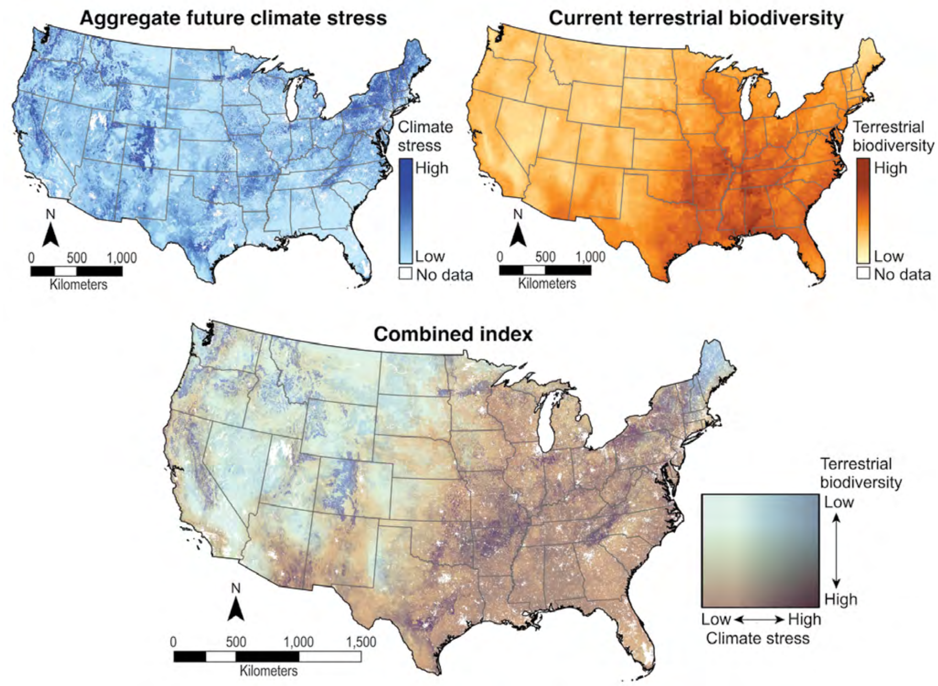
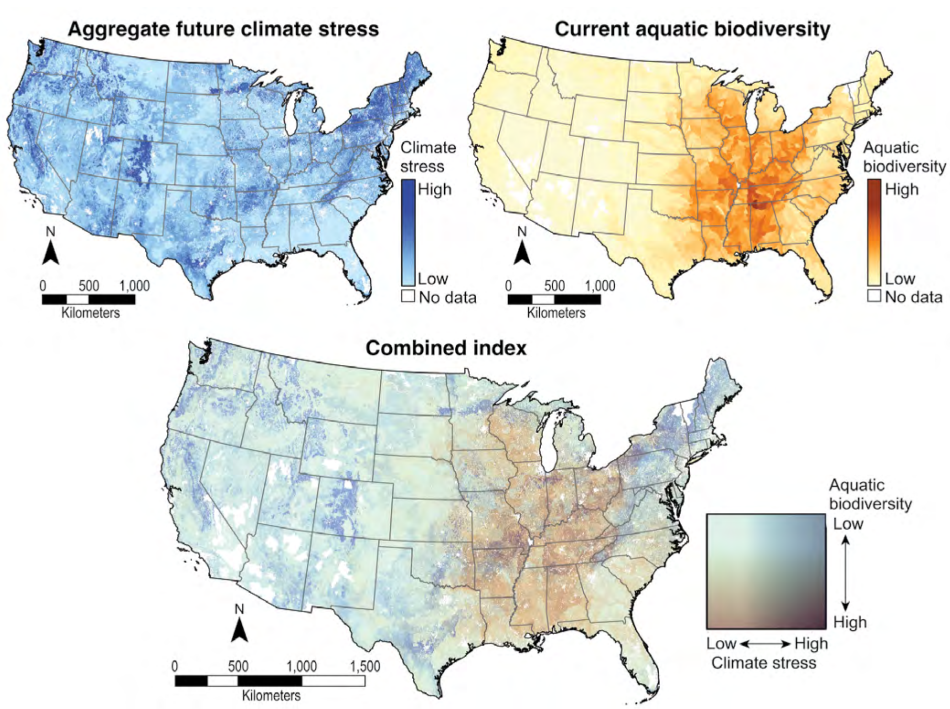


Figure 10-25. Hotspots with both high aquatic biodiversity and a likelihood of high future stress. Overlaying an index of future climate change (map in upper left)—defined as the number of climate models that identified an individual cell as high stress—on current aquatic biodiversity (map in upper right) results in a map that identifies hotspots with both high aquatic biodiversity and a likelihood of high future stress (map in lower center). Future climate stress and source information for biodiversity map described in prior sections of the chapter.



Future Climate and Biota

The potential effects of future climate on terrestrial and aquatic biota are relevant to management decisions being made today. We therefore examined how future climate stress could intersect terrestrial and aquatic biodiversity patterns by overlaying the future climate stress map onto our biodiversity maps.

The eastern portion of the country contains the areas of highest terrestrial and aquatic biodiversity. When overlaid with climate stress, we see some similar overall patterns. For terrestrial biota, the overlap with climate stress shows areas of high vulnerability in the Appalachian and Ozark Mountains, as well as in the Madrean Sky Islands. Additional areas of both high terrestrial biodiversity and high climate stress are found in southern Texas and in the Northeast (figure 10-24). Vulnerability of aquatic biodiversity to climate stress is also centered in the Appalachian and Ozarks Mountains (figure 10-25). Areas of high climate stress but low biodiversity are found throughout the Western States. Although these areas may have lower overall biodiversity, they support important ecosystems, making identification of areas of high climate stress important for management decisions.

Management Implications

The observed ongoing decline in long-term biodiversity and increases in ESA listing status across taxa signal continued biotic and anthropogenic threats to native biodiversity across the conterminous United States. The vulnerability of native biota, particularly to land use and projected climate change, highlights the importance of prioritizing habitat conservation actions in response to local and regional threats. Federal lands, including national forests and national parks, can play a role in providing long-term refugia as landscapes transition into novel future ecosystems.

Different regions of the country experience specific and uniquely interacting threats. For example, compounded land use stressors dominate the Eastern United States, while development is the primary land use stressor of habitats in the West. Climate change stress, however, is expected to be highest in higher elevation areas, but also creates a mosaic of high stress across the country where it interacts with disturbance processes such as wildfire. Because these stressors contribute to biodiversity loss, land managers in different regions will face unique combinations of stressors.

Non-Federal lands in the RPA North and South Regions are projected to be highly vulnerable to compounded land use stress in the coming decades, increasing the importance of Federal lands to serve as refugia for biodiversity. The amount of federally owned land in these regions is limited, however, and the fact that they are the most biodiverse regions in the conterminous United States highlights the difficulty for managers seeking to conserve and protect biodiversity. Collaboration across both public and private lands therefore offers the best path for biodiversity conservation and protection in the Eastern United States.

In the Western United States, stress is patchier in distribution than in the East and derives primarily from development and climate change. The higher proportion of the land in Federal ownership in the West presents an opportunity for biodiversity conservation at broad spatial scales commensurate with climate stress and wildfire. While collaboration with partners and non-Federal entities can enable the maintenance of ecological processes and habitat connectivity, there may be opportunities to track and facilitate migration of ecosystems and species into habitats more conducive to survival as climate and landscapes shift. Management could benefit from additional consideration of climate vulnerability planning that supports change as ecosystems transition in response to climate reality (West et al. 2009), including resilience to disturbances such as wildfire (e.g., Ager et al. 2020).

Conclusions

The diversity of animal life described in this chapter contributes to our well-being, our livelihoods, and our national identity. Our analysis demonstrates that ecosystems across the country are vulnerable to a variety of factors including land use change, climate change, and biological invasions. Our analysis also demonstrates that threats vary across the country, with different combinations of threats associated with different underlying topographic, climatic, and settlement patterns. These combined drivers have contributed to the current mosaic of intersecting land uses and native ecosystems that define different regions across the country.

Our analysis projects that land use pressures—including land conversion, human population growth, expansion of agricultural areas, and development of energy infrastructure and mining—will be most pronounced in the North Region, as well as areas of the South, driven in large part by population growth. Land use change has been identified as a threat to species persistence (Smith-Hicks and Morrison 2021), particularly in terrestrial systems (Sala et al. 2000), owing to a variety of impairments to habitats including reductions in quantity, quality, and connectivity (Powers and Jetz 2019). For example, endemic species that are specialized and regionally isolated are inherently vulnerable to shifts in land cover and human population pressure or climate (Malcolm et al. 2006), reflecting a potentially limited adaptive capacity to survive as conditions change. Land use changes compromise and reduce local habitat availability and/or quality and may have a more immediate effect on endemic species compared with broadly distributed species for which some portion of the population may be able to find refuge on Federal or other protected lands. Highly migratory species are also uniquely vulnerable to climate change, as their life stages are linked to specific habitat conditions at specific times (e.g., phenology). Decoupling of these linkages as climate and land use changes in different ways in different places along their migratory journey may increase their overall vulnerability (Robinson et al. 2009). The high concentration of land use stress in the RPA North and South Regions highlights the conservation role of limited Federal lands in the East, which also coincide with the areas of highest terrestrial and aquatic biodiversity in the conterminous United States.

Climate-driven stress is highest in the Pacific Coast Region and parts of the Rocky Mountain Region—areas that have a large share of Federal lands including NFS, NPS, and FWS lands—along with the North Region. Our modeling indicates that climate change may compromise the ability of federally managed lands to provide refugia to native biota. Protection of native ecosystems (that will likely morph into novel ones as they migrate in response to a changing climate) will likely require collaboration and cooperation from public and private lands. In addition, management approaches that are based on past and current climate may benefit from updates to consider different future climates and the potential for greater numbers of discrete and compound stress events (e.g., drought, heat, and wildfire) (Hagerman and Pelai 2018, IPCC 2021).

In addition to biotic and anthropogenic stressors varying across regions, biota in different parts of the country will likely also experience different types of stresses over short and long timescales. Although projections look to the future, the documented changes already occurring make climate change both a short-term and long-term issue with broad management consequences (IPCC 2014). Shifts in the distribution patterns of vegetation (e.g., Joshua Tree National Park with fewer Joshua trees (*Yucca brevifolia*), Sweet et al. 2019) and landscape features (e.g., retreating glaciers in Glacier National Park, Hall and Fagre 2003) have been occurring for decades. As the existing capacity of individual species to survive in more constricted environmental conditions declines, the threshold between survival and imperilment will likely narrow. Greater vulnerability to climate change is anticipated for long-lived species that adapt more slowly to environmental changes (Hetem et al. 2014) or for species with small range sizes who have fewer migration opportunities (Morueta-Holme et al. 2010, Schloss et al. 2012). For species with broader distributions, portions of the population may experience climate stress, rather than the entire population (i.e., coho salmon, *Oncorhynchus kisutch*, in the Pacific Northwest, Flitcroft et al. 2019), or specific seasons may be more stressful than others (i.e., seasonal temperatures and vegetation, Wang et al. 2011). The important role that protected lands can play into the future is likely to become even more critical. The ability of forest managers to maintain and enhance variability of terrestrial and aquatic habitats within their jurisdictions may provide a critical link for the persistence of native biota.

Observed patterns in the listing status of imperiled species under the ESA show some overlap between areas of high biodiversity and locations containing larger numbers of listed species. However, ESA listing bias towards large-bodied charismatic species for which long-term data often exist, suggests that current patterns of imperiled species may not necessarily reflect species-specific threats across the country (e.g., amphibians, Gratwicke et al. 2012). In recognition of the changes to the environment that are stressing native biota, it is important to look for additional vulnerable populations of biota that are not currently targeted for conservation. Additional Federal listing decisions for species of concern could occur as climate and land use continue to affect habitats across the United States.

Literature Cited

Abood, S.A.; Maclean, A.L.; Mason, L.A. 2012. Modeling riparian zones utilizing dams and flood height data. Photogrammetric Engineering and Remote Sensing. 78: 259–269.

Adams, S.B. 2013. Effects of small impoundments on downstream crayfish assemblages. Freshwater Science. 32: 1318–1332.

Ager, A.A.; Barros, A.M.G.; Houtman, R.; Seli, R.; Day, M.A. 2020. Modelling the effect of accelerated forest management on long-term wildfire activity. Ecological Modelling. 421(1): 108962.

Allan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics. 35: 257–284.

Allert, A.L.; Fairchild, J.F.; DiStefano, R.J.; Schmitt, C.J. 2009. Ecological effects of lead mining on Ozark streams: in-situ toxicity to woodland crayfish (*Orconectes hylas*). Ecotoxicology and Environmental Safety. 72(4): 1207–1219.

Artois, M.; Bengis, R.; Delahay, R.J.; Duchêne, M-J.; Duff, J.P.; Ferroglio, E.; Gortazar, C.; Hutchings, M.R.; Kock, R.A.; Leighton, F.A.; Mörner, T.; Smith, G.C. 2009. Wildlife disease surveillance and monitoring. In: Delahay, R.J.; Smith, G.C.; Hutchings, M.R., eds. Management of disease in wild mammals. Tokyo: Springer-Verlag: 187–213.

Bachelet, D.; Ferschweiler, K.; Sheehan, T.; Strittholt, J. 2016. Climate change effects on southern California deserts. Journal of Arid Environments. 127: 17–29.

Balloux, F.; van Dorp, L. 2017. Q&A: What are pathogens, and what have they done to and for us? BMC Biology. 15: 91.

Barber, J.R.; Crooks, K.R.; Fristrup, K.M. 2010. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology & Evolution. 25(3):180–189.

Barnett, Z.C. 2017. Habitat use and life history of the vernal crayfish, *Procambarus viaeviridis* (Faxon, 1914), a secondary burrowing crayfish in Mississippi, USA. Journal of Crustacean Biology. 37: 544–555.

Barnett, Z.C. 2019. Effects of impoundments on the community assemblage and gene flow of stream crayfishes. Oxford, MS: Univerisity of Mississippi. 254 p. PhD dissertation.

Barnett, Z.C.; Adams, S.B. 2021. Review of dam effects on native and invasive crayfishes illustrates complex choices for conservation planning. Frontiers in Ecology and Evolution. 8: 1–15.

Barnett, Z.C.; Adams, S.B.; Hoeksema, J.D.; Easson, G.L.; Ochs, C.A. 2022. Effects of impoundments on stream crayfish assemblages. Freshwater Science. 41(1): 125–142.

Barnett, Z.C.; Adams, S.B.; Ochs, C.A.; Garrick, R.C. 2020. Crayfish populations genetically fragmented in streams impounded for 36–104 years. Freshwater Biology. 65(4): 768–785.

Beale, C.L; Johnson, K.M. 1998. The identification of recreational counties in nonmetropolitan areas of the USA. Population Research and Policy Review. 17(1): 37–53.

Beck, K.G.; Zimmerman, K.; Schardt, J.D.; Stone, J.; Lukens, R.R.; Reichard, S.; Randall, J.; Cangelosi, A.A.; Cooper, D.; Thompson, J.P. 2008. Invasive species defined in a policy context: Recommendations from the Federal Invasive Species Advisory Committee. Invasive Plant Science and Management. 1(4): 414–421.

Bennett, V.J. 2017. Effects of road density and pattern on the conservation of species and biodiversity. Current Landscape Ecology Reports. 2(1): 1–11.

Best, M.L.; Welsh Jr., H.H. 2014. The trophic role of a forest salamander: impacts on invertebrates, leaf litter retention, and the humification process. Ecosphere. 5(2): 16. http://dx.doi.org/10.1890/ES13-00302.1.

Bhattacharyya, A. 1946. On a measure of divergence between two multinomial populations. Sankhyā: The Indian Journal of Statistics. 7(4):401–406. http://www.jstor.org/stable/25047882.

Bletz, M.C.; Gratwicke, B.; Beleason, A.M.; Catenazzi, A.; Duffus, A.L.J.; Lampo, M.; Olson, D.H.; Vasudevan, K.; and IUCN Amphibian Specialist Group. 2022 in press. Infectious Diseases. Chapter 6 in: Wren, S.; Angulo, A.; Meredith, H.; Kielgast, J.; Dos Santos, M.; Bishop, P., eds. Amphibian Conservation Action Plan, Update: Thematic Working Groups and Chairs. IUCN Species Survival Commission, Amphibian Specialist Group. https://www.iucn-amphibians.org/. (3 November 2022).

Bonn, A.; Rodrigues, A.S.L.; Gaston, K.J. 2002. Threatened and endemic species: are they good indicators of patterns of biodiversity on a national scale? Ecology Letters. 5: 733–741.

Born, W.; Rauschmayer, F.; Bräuer, I. 2005. Economic evaluation of biological invasions—a survey. Ecological Economics. 55(3): 321–336.

Boyle, R.A.; Dorn, N.J.; Cook, M.I. 2014. Importance of crayfish prey to nesting white ibis (*Eudocimus albus*). Waterbirds. 37:19–29.

Bradley, C.A.; Altizer, S. 2007. Urbanization and the ecology of wildlife diseases. Trends in Ecology and Evolution. 22(2): 95–102.

Brearley, G.; Rhodes, J.; Bradley, A.; Baxter, G.; Seabrook, L.; Lunney, D.; Liu, Y.; McAlpine, C. 2013. Wildlife disease prevalence in human-modified landscapes. Biological Reviews. 88(2): 427–442.

Brinson, M.M.; Swift, B.L.; Plantico, R.C.; Barday, J.S. 1981. Riparian ecosystems: their ecology and status. FWS/OBS81/17, Kearneysville, WV: U.S. Department of the Interior, U.S. Fish and Wildlife Service, Biological Services Program. 154 p.

Brown, R.M.; Laband, D.N. 2006. Species imperilment and spatial patterns of development in the United States. Conservation Biology. 20(1): 239–244.

Burlakova, L.E.; Karatayey, A.Y.; Karatayey, J.A.; May, M.E. 2011. Endemic species: contribution to community uniqueness, effect of habitat alteration, and conservation priorities. Biological Conservation. 144: 155–165.

Buttke, D.; Wild, M.; Monello, R.; Schuurman, G.; Hahn, M.; Jackson, K. 2021. Managing wildlife disease under climate change. EcoHealth 18: 406–410.

Carral-Murrieta, C.O.; García-Arroyo, M.; Marín-Gómez, O.H.; Sosa-Lopez, J.R. 2020. Noisy environments: untangling the role of anthropogenic noise on bird species richness in a Neotropical city. Avian Research. 11: 32. https://doi.org/10.1186/s40657-020-00218-5.

Carroll, C. 2007. Interacting effects of climate change, landscape conversion, and harvest on carnivore populations at the range margin: marten and lynx in the northern Appalachians. Conservation Biology. 21(4): 1092–1104.

Charles, H.; Dukes, J.S. 2008. Impacts of invasive species on ecosystem services. In: Nentwig, W., ed. Biological Invasions. Berlin, Heidelberg: Springer-Verlag. 217–237.

Cheng, T.L.; Reichard, J.D.; Coleman, J.T.; Weller, T.J.; Thogmartin, W.E.; Reichert, B.E.; Bennett, A.B.; Broders, H.G.; Campbell, J.; Etchison, K.; Feller, D.J.; Geboy, R.; Hemberger, T.; Herzog, C.; Hicks, A.C.; Houghton, S.; Humber, J.; Karth, J.A.; King, R.A.; Loeb, S.C.; Masse, A. Morris, K.M.; Niederriter, H.; Nordquist, G.; Perry, R.W.; Reynolds, R.J.; Sasse, D.B.; Scafini, M.R.; Stark, R.C.; Stihler, C.W.; Thomas, S.C.; Turner, G.G.; Webb, S.; Westrich, B.J.; Frick, W.F. 2021. The scope and severity of white-nose syndrome on hibernating bats in North America. Conservation Biology. 35(5): 1586–1597.

Clavero, M.; García-Berthou, E. 2005. Invasive species are a leading cause of animal extinctions. Trends in Ecology and Evolution. 20(3): 110.

Coates, P.S.; Ricca, M.A.; Prochazka, B.G.; Brooks, M.L.; Doherty, K.E.; Kroger, T.; Blomberg, E.J.; Hagen, C.A.; Casazza, M.L. 2016. Wildfire, climate, and invasive grass interactions negatively impact an indicator species by reshaping sagebrush ecosystems. Proceedings of the National Academy of Sciences of the United States of America. 113(45): 12745–12750. https://doi.org/10.1073/pnas.1606898113.

Colautti, R.I.; MacIsaac, H.J. 2004. A neutral terminology to define “invasive” species. Diversity and Distributions. 10(2): 135–141.

Crandall, K.A.; Buhay, J. E. 2008. Global diversity of crayfish (Astacidae, Cambaridae, and Parastacidae—Decapoda) in freshwater. Hydrobiologia. 595: 295–301.

Daszak, P.; Cunningham, A.A.; Hyatt, A.D. 2000. Emerging infectious diseases of wildlife—threats to biodiversity and human health. Science. 287(5452): 443–449.

Daszak, P.; Cunningham, A.A.; Hyatt, A.D. 2001. Anthropogenic environmental change and the emergence of infectious diseases in wildlife. Acta Tropica. 78(2): 103–116.

Deem, S.L.; Karesh, W.B.; Weisman, W. 2001. Putting theory into practice: wildlife health in conservation. Conservation Biology. 15(5): 1224–1233.

DeLong, D.C. Jr. 1996. Defining biodiversity. Wildlife Society Bulletin. 24(4): 738-749.

Devictor, V.; Julliard, R.; Jiguet, F. 2008. Distribution of specialist and generalist species along spatial gradients of habitat disturbance and fragmentation. Oikos. 117(4): 507–514.

Doherty, T.S.; Glen, A.S.; Nimmo, D.G.; Ritchie, E.G.; Dickman, C.R. 2016. Invasive predators and global biodiversity loss. Proceedings of the National Academy of Sciences of the United States of America. 113(40): 11261–11265.

Donald, P.F.; Green, R.E.; Heath, M.F. 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. Proceedings of the Royal Society B: Biological Sciences. 268: 25–29. https://doi.org/10.1098/rspb.2000.1325.

Doody, J. S.; Green, B.; Rhind, D. Castellano, C.M.; Sims, R.; Robinson, T. 2009. Population-level declines in Australian predators caused by an invasive species. Animal Conservation. 12(1): 46–53.

Doppelt, R.; Scurlock, M.; Frissell, C.; Karr, J.R. 1993. Entering the watershed: a new approach to save America's river ecosystems. Covelo, CA: Island Press. 504 p.

Dudley, N.; Alexander, S. 2017. Agriculture and biodiversity: a review. Biodiversity. 18(2-3): 45–49.

Dugger, K.M.; Anthony, R.G.; Andrews, L.S. 2011. Transient dynamics of invasive competition: Barred Owls, Spotted Owls, habitat, and the demons of competition present. Ecological Applications. 21(7): 2459–2468.

Elkins, D.; Sweat, S.C.; Kuhajda, B.R.; George, A.L.; Hill, K.S.; Wenger, S.J. 2019. Illuminating hotspots of imperiled aquatic biodiversity in the southeastern US. Global Ecology and Conservation. 19: e00654.

Elmqvist, T.; Folke, C.; Nyström, M.; Peterson, G.; Bengtsson, J.; Walker, B.; Norberg, J. 2003. Response diversity, ecosystem change, and resilience. Frontiers in Ecology and the Environment. 1(9): 488–494.

Flather, C.H.; Knowles, M.S.; Jones, M.F.; Schilli, C. 2013. Wildlife population and harvest trends in the United States: a technical document supporting the Forest Service 2010 RPA Assessment. General Technical Report RMRS-GTR-296. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 94 p.

Flather, C.H.; Knowles, M.S.; Kendall, I.A. 1998. Threatened and endangered species geography. BioScience. 48: 365–376.

Flather, C.H.; Sauer, J.R. 1996. Using landscape ecology to test hypotheses about large-scale abundance patterns in migratory birds. Ecology. 77(1): 28–35.

Flitcroft, R.; Lewis, S.; Arismendi, I.; Davis, C.; Giannico, G.: Penaluna, B.; Santelmann, M.; Safeeg, M.; Snyder, J. 2019. Using expressed behavior of coho salmon (*Oncorhynchus kisutch*) to evaluate the vulnerability of upriver migrants under future hydrological regimes: management implications and conservation planning. Aquatic Conservation: Marine and Freshwater Ecosystems. 29: 1083–1094.

Foley, J.A.; Defries, R.; Asner, G.P.; Barford, C.; Bonan, G.; Carpenter, S.R.; Chapin, F.S.; Coe, M.T.; Daily, G.C.; Gibbs, H.K.; Helkowski, J.H.; Holloway, T.; Howard, E.A.; Kucharik, C.J.; Monfreda, C.; Patz, J.A.; Prentice, I.C.; Ramankutty, N.; Snyder, P.K. 2005. Global consequences of land use. Science. 309 (5734): 570–574.

Fuhlendorf, S.D.; Harrell, W.C.; Engle, D.M.; Hamilton, R.G.; Davis, C.A.; Leslie, D.M.,Jr. 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecological Applications. 16: 1706-1716. https://doi.org/10.1890/1051-0761(2006)016[1706:SHBTBF]2.0.CO;2.

Gallana, M.; Ryser-Degiorgis, M.P.; Wahli, T.; Segner, H. 2013. Climate change and infectious diseases of wildlife: altered interactions between pathogens, vectors and hosts. Current Zoology. 59(3): 427–437.

Garcia-Moreno, J.; Harrison, I.J.; Dudgeon, D.; Clausnitzer, V.; Darwall, W.; Farrell, T.; Savy, C.; Tockner, K.; Tubbs, N. 2014. Sustaining freshwater biodiversity in the Anthropocene. In: Bhaduri, A.; Bogardi, J.; Leentvaar, J.; Marx, S., eds. The Global Water System in the Anthropocene. Switzerland: Springer International Publishing. 247–270.

Garwood, T.J.; Lehman, C.P.; Walsh, D.P.; Cassirer, E.F.; Besser, T.E.; Jenks, J.A. 2020. Removal of chronic *Mycoplasma ovipneumoniae* carrier ewes eliminates pneumonia in a bighorn sheep population. Ecology and Evolution. 10(7): 3491–3502.

Girgin, S.; Krausmann, E. 2016. Historical analysis of US onshore hazardous liquid pipeline accidents triggered by natural hazards. Journal of Loss Prevention in the Process Industries. 40: 578–590.

Gratwicke, B.; Lovejoy, T.E.; Wildt, D.E. 2012. Will amphibians croak under the Endangered Species Act? BioScience. 62(2): 197–202.

Hagerman, S.M.; Pelai, R. 2018. Responding to climate change in forest management: two decades of recommendations. Frontiers in Ecology and the Environment. 16(10): 579–587.

Hall, M.H.P.; Fagre, D.B. 2003. Modeled climate-induced glacier change in Glacier National Park, 1850–2100. BioScience. 53(2): 131–140.

Harris, J.B.C.; Reid, J.L; Scheffers, B.R. [et al.]. 2012. Conserving imperiled species: a comparison of the IUCN Red List and U.S. Endangered Species Act. Conservation Letters. 5(1): 64–72.

Hetem, R.S.; Fuller, A.; Maloney, S.K.; Mitchell, D. 2014. Responses of large mammals to climate change. Temperature. 1(2): 115–127.

Hilker, F.M.; Schmitz, K. 2008. Disease-induced stabilization of predator–prey oscillations. Journal of Theoretical Biology. 255(3): 299–306.

Hjerpe, E.; Hussain, A.; Holmes, T. 2020. Amenity migration and public lands: rise of the protected areas. Environmental Management. 66(1): 56–71.

Holdich, D.M., ed. 2002. Biology of freshwater crayfish.Oxford, UK: Blackwell Science. 720 p.

Homer, C.; Dewitz, J.; Jin, S.; Xian, G.; Costello, C.; Danielson, P.; Gass, L.; Funk, M.; Wickham, J.; Stehman, S.; Auch, R.; Riitters, K. 2020. Conterminous United States land cover change patterns 2001–2016 from the 2016 National Land Cover Database. ISPRS Journal of Photogrammetry and Remote Sensing. 162: 184–199.

Howard, C.; Flather, C.H.; Stephens, P.A. 2020. A global assessment of the drivers of threatened terrestrial species richness. Nature Communications. 11: art 993.

Hoyt, J.R.; Kilpatrick, A.M.; Langwig, K. E. 2021. Ecology and impacts of white-nose syndrome on bats. Nature Reviews Microbiology. 19(3): 196–210.

Huxel, G.R. 1999. Rapid displacement of native species by invasive species: effects of hybridization. Biological Conservation. 89(2): 143–152.

Intergovernmental Panel on Climate Change (IPCC). 2014. 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Core writing team, R.K. Pachauri and L.A. Meyer (eds). IPCC, Geneva, Switzerland, 151 p.

Intergovernmental Panel on Climate Change (IPCC). 2021. Summary for Policymakers. In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., Zhai, P.; Pirani, A. [et al.] (eds.)]. Cambridge University Press. In Press.

International Union for the Conservation of Nature [IUCN]. 2020a. IUCN Species Survival Commission (SSC) Amphibian Specialist Group. https://www.iucn-amphibians.org/. (1 December 2020).

International Union for the Conservation of Nature [IUCN]. 2020b. IUCN Red List of Threatened Species. Vers. 2020-3. https://www.iucnredlist.org. (November-December 2020).

Jakes, A.F.; DeCesare, N.J.; Jones, P.F.; Gates, C.C.; Story, S.J.; Olimb, S.K.; Kunkel, K.E.; Hebblewhite, M. 2020. Multi-scale habitat assessment of pronghorn migration routes. PLoS One. 15(12): e0241042.

Johnson, P.T.J.; Lunde, K.B.; Thurman, E.M.; Ritchie, E.G.; Wray, S.N.; Sutherland, D.R.; Kapfer, J.M.; Frest, T.J.; Bowerman, J.; Blaustein, A.R. 2002. Parasite (*Ribeiroia ondatrae*) infection linked to amphibian malformations in the western United States. Ecological Monographs. 72(2): 151–168.

Johnson, P.T.; Olden, J.D.; Solomon, C.T.; Vander Zanden, M.J. 2009. Interactions among invaders: community and ecosystem effects of multiple invasive species in an experimental aquatic system. Oecologia. 159(1): 161–170.

Johnstone, J.F.; Allen, C.D.; Franklin, J.F.; Frelich, L.E.; Harvey, B.J.; Higuera, P.E.; Mack, M.C.; Meetenmeyer, R.K.; Metz, M.R.; Perry, G.L.W.; Schoennagel, T.; Turner, M.G. 2016. Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment. 14(7): 369–378.

Julian, J.T.; Henry, P.F.P.; Drasher, J.M.; Mitchell, K.; Smith, S.A. 2020. Minimizing the spread of herpetofaunal pathogens in aquatic habitats by decontaminating construction equipment. Herpetological Review 51: 472–483.

Kerth, G.; Melber, M. 2009. Species-specific barrier effects of a motorway on the habitat use of two threatened forest-living bat species. Biological Conservation. 142(2): 270-279.

Kirk, R.W.; Bolstad, P.V.; Manson, S.M. 2012. Spatio-temporal trend analysis of long-term development patterns (1900–2030) in a Southern Appalachian county. Landscape and Urban Planning. 104(1): 47–58.

Klemm, T.; Briske, D.D.; Reeves, M.C. 2020. Potential natural vegetation and NPP responses to future climates in the U.S. Great Plains. Ecosphere. 11(10): e03264. https://doi.org/10.1002/ecs2.3264.

Kock, R.A.; Woodford, M.H.; Rossiter, P.B. 2010. Disease risks associated with the translocation of wildlife. Revue Scientifique et Technique. 29(2): 329.

Krauss, J.; Steffan-Dewenter, I.; Tscharntke, T. 2003. Local species immigration, extinction, and turnover of butterflies in relation to habitat area and habitat isolation. Oecologia. 137(4): 591–602.

Larson, E.R.; Olden, J.D. 2011. The state of crayfish in the Pacific northwest. Fisheries 36: 60–73.

Lawrence, D.L.; Runyon, A.N.R.; Gross, J.E.; Schuurman, G.W.; Miller, B.W. 2021. Divergent, plausible, and relevant climate future for near- and long-term planning. Climatic Change. 167: 38 https://doi.org/10.1007/s10584-021-03169-y.

Levy, D.A. 2009. Pipelines and salmon in northern British Columbia. Drayton Valley, AB: Pembina Institute. 51 pp.

Linz, G.M.; Homan, H.J.; Gaukler, S.M.; Penry, L.B.; Bleier, W.J. 2007. European starlings: a review of an invasive species with far-reaching impacts. In: Witmer, G.W.; Pitt, W.C.; Fagerstone, K.A., eds. Managing Vertebrate Invasive Species: Proceedings of an International Symposium. Fort Collins, CO: U.S. Department of Agriculture, APHIS Wildlife Services, National Wildlife Research Center.

Lipton, D.M.A.; Rubenstein, S.R.; Weiskopf, S.; Carter, S.L.; Peterson, J.; Crozier, L.; Fogarty, M.; Gaichas, S.; Hyde, K.J.W.; Morelli, T.L.; Morisette, J.; Moustahfid, H.; Munoz, R.; Poudel, R.; Staudinger, M.; Stock, C.; Thompson, L.; Waples, R.S.; Weltzin, J. 2018. Ecosystems, ecosystem services, and biodiversity. In: Reidmiller, D.; Avery, C.; Easterling, D.; Kunkel, K.E.; Lweis, K.L.M.; Maycock, T.K.; Stewarad, B.C., eds. Impacts, risks, and adaptation in the United States: fourth national climate assessment, volume II. Washington, DC: U.S. Global Change Research Program: 269–321.

Logan, J.; Powell, J. 2001. Ghost forests, global warming, and the mountain pine beetle (Coleoptera: Scolytidae). American Entomologist. 47(3): 160–173.

Loughman, Z.J.; Fetzner, J.W., Jr. 2015. Astacology and crayfish conservation in the southeastern United States: past, present, and future. Freshwater Crayfish. 21(1): 1–5.

Mack, M.C.; D'Antonio, C.M. 1998. Impacts of biological invasions on disturbance regimes. Trends in Ecology & Evolution. 13(5): 195–198.

Malcolm, J.R.; Liu, C.; Neilson, R.P.; Hansen, L.; Hannah, L. 2006. Global warming and extinctions of endemic species from biodiversity hotspots. Conservation Biology. 20(2): 538–548.

Maloney, K.O.; Young, J.A.; Faulkner, S.P.; Hailegiorgis, A.; Slonecker, E.T.; Milheim, L. 2018. A detailed risk assessment of shale gas development on headwater streams in the Pennsylvania portion of the Upper Susquehanna River Basin, USA. Science of the Total Environment. 610: 154–166.

Mantyka-Pringle, C.S.; Visconti, P; Di Marco, M.; Martin, T.G.; Rondinini, C.; Rhodes, J.R. 2015. Climate change modifies risk of global biodiversity loss due to land-cover change. Biological Conservation. 187: 103–111.

Miller, R.R.; Williams, J. D.; Williams, J.E. 1989. Extinctions of North American fishes during the past century. Fisheries. 14(6): 22–38.

Miller, R.S.; Sweeney, S.J.; Slootmaker, C.; Grear, D.A.; DiSalvo, P.A.; Kiser, D.; Shwiff, S.A. 2017. Cross-species transmission potential between wild pigs, livestock, poultry, wildlife, and humans: implications for disease risk management in North America. Scientific Reports. 7(1): 7821.

Mitsch, W.J.; Gosselink, J.G. 1993. Wetlands. New York: John Wiley and Sons. 454 p.

Moore, M.J.; Distefano, R. J.; Larson, E.R. 2013. An assessment of life-history studies for USA and Canadian crayfishes: identifying biases and knowledge gaps to improve conservation and management. Freshwater Science. 32: 1276–1287.

Morueta-Holme, N.; Fløjgaard, C.; Svenning, J.C. 2010. Climate change risks and conservation implication for a threatened small-range mammal species. PLoS One. 5(4): e10360.

Muhlfeld, C.C.; Kovach, R.P.; Al-Chokhachy, R.; Amish, S.J.; Kershner, J.L.; Leary, R.F; Lowe, W.H.; Luikart, G.; Matson, P.; Schmetterling, D.A.; Shepard, B.B.; Westley, P.A.H.; Whited, D.; Whiteley, A.; Allendorf, F.W. 2017. Legacy introductions and climatic variation explain spatiotemporal patterns of invasive hybridization in a native trout. Global Change Biology. 23(11): 4663–4674.

Muhlfeld, C.C., Kovach, R.P., Jones, L.A.; Al-Chokhachy, R.K.; Boyer, M.C.; Leary, R.F.; Lowe, W.H.; Luikart, G.; Allendorf, F.W. 2014. Invasive hybridization in a threatened species is accelerated by climate change. Nature Climate Change. 4(7): 620–624.

Naiman, R.J.; Décamps, H.; Pollok, M. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications. 3: 309–313.

NatureServe. 2014. NatureServe Central Databases. Metadata for the U.S. Forest Service – Rocky Mountain Research Station: 2014 RPA and NRSF Biodiversity Datasets. Arlington, VA: NatureServe Central Databases.79 p.

Nichols, J.D.; Johnson, F.A.; Williams, B.K. 1995. Managing North American waterfowl in the face of uncertainty. Annual Review of Ecology and Systematics 26: 177–199.

Nichols, J.D.; Runge, M.C.; Johnson, F.A.; Williams, B.K. 2007. Adaptive harvest management of North American waterfowl populations: a brief history and future prospects. Journal of Ornithology 148: S343–S349.

National Wildfire Coordinating Group [NWCG]. 2017. Guide to preventing aquatic invasive species transport by wildland fire operations. Invasive species subcommittee, Equipment Technology Committee, National Wildfire Coordinating Group, United States. PMS 444. 64 p. https://www.nwcg.gov/sites/default/files/publications/pms444.pdf. (26 August 2020).

National Wildfire Coordinating Group [NWCG]. 2020. Invasive species mitgation for ground resources. Invasive species subcommittee, Equipment Technology Committee, National Wildfire Coordinating Group, United States. Operations video. https://www.nwcg.gov/publications/training-courses/rt-130/operations/op819. (26 August 2020).

Northeast Wildlife Disease Cooperative [NWDC]. 2021. Disease Fact Sheets. https://www.northeastwildlife.org/disease-fact-sheets. (4 January 2021).

Olson, D.H. 2022. In press. Climate change adaptation-management strategies for herpetofauna. In: Walls S.C.; O'Donnell, K.M., eds. Strategies for Conservation Success in Herpetology. University Heights, OH: Society for the Study of Amphibians and Reptiles: ch. 46.

Olson, D.H.; Pilliod, D. 2022. In press. Amphibian and Reptile Conservation in the United States of America. In: Walls S.C.; O'Donnell, K.M., eds. Strategies for Conservation Success in Herpetology. University Heights, OH: Society for the Study of Amphibians and Reptiles: ch. 22.

Pacifici, M.; Visconti, P.; Butchart, S.; Watson, J.E.M.; Cassola, F.M.; Rondinini, C. 2017. Species' traits influenced their response to recent climate change. Nature Climate Change. 7: 205–208. https://doi.org/10.1038/nclimate3223.

Partners in Amphibian and Reptile Conservation [PARC]. 2021. PARC National Disease Task Team: Herpetofaunal Disease Resources. https://parcplace.org/resources/parc-disease-task-team/. (4 January 2021).

Pedersen, A.B.; Jones, K.E.; Nunn, C.L.; Altizer, S. 2007. Infectious diseases and extinction risk in wild mammals. Conservation Biology. 21(5): 1269–1279.

Penaluna, B.E.; Abadia-Cardoso, A.; Dunham, J.B.; Garcia-De-Leon, F.J.; Gresswell, R.E.; Luna, A.R.; Taylor, E.V.; Shepard, B.B.; Al-Chokhachy, R.; Muhlfield, C.C.; Bestgen, K.R.; Rogers, K.; Escalante, M.A.; Keeley, E.R.; Temple, G.M.; Williams, J.E.; Matthews, K.R.; Pierce, R.; Mayden, R.L.; Kovach, R.P.; Garza, J.C.; Fausch, K.D. 2016. Conservation of native Pacific trout diversity in western North America. Fisheries. 41: 287–300.

Penaluna, B.E.; Olson, D.H.; Flitcroft, R.L.; Weber, M.A.; Bellmore, J.R.; Wondzell, S.M.; Dunham, J.B.; Johnson, S.L.; Reeves, G.H. 2017. Aquatic biodiversity in forests: a weak link in ecosystem services resilience. Biodiversity Conservation. 26: 3125–3155.

Pidgeon, A.M.; Radeloff, V.C.; Flather, C.H.; Lepczyk, C.A.; Clayton, M.K.; Hawbaker, T.J.; Hammer, R.B. 2007. Associations of forest bird species richness with housing and landscape patterns across the USA. Ecological Applications. 17: 1989–2010.

Pimentel, D.; Zuniga, R.; Morrison, D. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics. 52(3): 273–288.

Poland, T.M.; Patel-Weynand, T.; Finch, D.M.; Miniat, C.F.; Hayes, D.C.; Lopez, V.M., eds. 2021. Invasive species in forests and rangelands of the United States: a comprehensive science synthesis for the United States forest sector. Heidelberg, Germany: Springer International Publishing. 455 p.

Powers, R.P.; W. Jetz. 2019. Global habitat loss and extinction risk of terrestrial vertebrates under future land-use-change scenarios. Nature Climate Change. 9: 323–329.

Price, S.J.; Leung, W.T.; Owen, C.J.; Puschendorf, C.S.; Cunningham, A.A.; Balloux, F.; Garner, T.W.J.; Nichols, R.A. 2019. Effects of historic and projected climate change on the range and impacts of an emerging wildlife disease. Global Change Biology. 25(8): 2648–2660.

R Core Team. 2021. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org/.

Rachowicz, L.J.; Hero, J.M.; Alford, R.A.; Taylor, J.W.; Morgan, J.A.T.; Vredenburg, V.T.; Collins, J.P.; Briggs, C.J. 2005. The novel and endemic pathogen hypotheses: competing explanations for the origin of emerging infectious diseases of wildlife. Conservation Biology. 19(5): 1441–1448.

Radeloff, V.C.; Helmers, D.P.; Kramer, H.A.; Mockrin, M.H.; Alexandre, P.M.; Bar-Massada, A.; Butsic, V.; Hawbaker, T.J.; Martinuzzi, S.; Syphard, A.D.; Stewart, S.I. 2018. Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences of the United States of America. 115: 3314–3319.

Rahel, F.J.; Bierwagen, B.; Taniguchi, Y. 2008. Managing aquatic species of conservation concern in the face of climate change and invasive species. Conservation Biology. 22(3): 551–561.

Rahel, F.J.; Olden, J.D. 2008. Assessing the effects of climate change on aquatic invasive species. Conservation Biology. 22(3): 521–533.

Ramirez, P. Jr. 2010. Bird mortality in oil field wastewater disposal facilities. Environmental Management. 46: 820–826. pmid:20844874.

Ramirez, P.; Mosley, S. 2015. Oil and gas wells and pipelines on U.S. wildlife refuges: challenges for managers, PLoS One. 10(4): e0124085. https://doi.org/10.1371/journal.pone.0124085.

Regetz, J. 2003. Landscape-level constraints on recruitment of chinook salmon (*Oncorhynchus tshawytscha*) in the Columbia River basin, USA. Aquatic Conservation: Marine and Freshwater Ecosystems. 13(1): 35–49.

Reid, S.M.; Anderson, P.G. 1999. Effects of sediment released during open-cut pipeline water crossings. Canadian Water Resources Journal. 24(3): 235–251.

Reid, S.M.; Isaac, G.; Metikosh, S.; Evans, J. 2003. Physiological response of rainbow trout to sediment released during open-cut pipeline water crossing construction. Water Quality Research Journal. 38(3): 473–481.

Reynolds, J.B.; Souty-Grosset, C.; Richardson, A.M.M. 2013. Ecological roles of crayfish in freshwater and terrestrial habitats. Freshwater Crayfish. 19:197–218.

Richman, N.I.; Bohm, M.; Adams, S.B.; Alvarez, F.; Bergey, E.A.; Bunn, J.J.S.; Burnham, Q.; Cordeiro, J.; Coughran, J.; Crandall, K.A.; Dawkins, K.L.; DiStefano, R.J.; Doran, N.E.; Edsman, L.; Eversole, A.G.; Fureder, L.; Furse, J.M.; Gherardi, F.; Hamr, P.; Holdick, D.M.; Horwitz, P.; Johnston, K.; Jones, C.M.; Jones, J.P.G.; Jones, R.L.; Jones, T.G.; Kawai, T.; Lawler, S.; Lopez-Mejia, M.; Miller, R.M.; Pedraza-Lara, C.; Reynolds, J.D.; Richardson, A.M.M.; Schultz, M.B.; Schuster, G.A.; Sibley, P.J.; Souty-Grosset, C.; Taylor, C.A.; Thoma, R.F.; Walls, J.; Walsh, T.S.; Collen, B. 2015. Multiple drivers of decling in the global status of freshwater crayfish (Decapoda: Astacidea). Philosophical Transactions of the Royal Society. B: Biological Sciences. 370: 20140060.

Riitters K.; Potter, K.M.; Iannone, B.V. III; Oswalt, C.; Guo, Q.; Fei, S. 2018. Exposure of protected and unprotected forest to plant invasions in the eastern United States. Forests. 9(11):723. https://doi.org/10.3390/f9110723.

Robbins, C.S.; Bystrak, D.; Geissler, P.H. 1986. The Breeding Bird Survey: its first fifteen years, 1965–1979. Resource Publication 157. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service. 196 p.

Robinson, R.A.; Crick, H.Q.P.; Learmonth, J.A.; Maclean, I.M.D. 2009. Travelling through a warming world: climate change and migratory species. Endangered Species Research. 7: 87–99.

Robinson, R.A.; Sutherland, W.J. 2002. Post-war changes in arable farming and biodiversity in Great Britain. Journal of Applied Ecology. 39(1): 157–176.

Rodenhouse, N.L.; Christenson, L.M.; Parry, D.; Green, L.E. 2009. Climate change effects on native fauna of northeastern forests. Canadian Journal of Forest Research. 39(2): 249–263.

Rosenberg, K.V.; Dokter, A.M.; Blancher, P.J.; Sauer, J.R.; Smith, A.C.; Smither, P.A.; Stanton, J.C.; Panjabi, A.; Helft, L.; Parr, M.; Marra, P. 2019. Decline of the North American avifauna. Science. 366(6461): 120–124.

Ryser-Degiorgis, M.P. 2013. Wildlife health investigations: needs, challenges and recommendations. BMC Veterinary Research. 9(1): 223.

Sala, O.E.; Chapin, F.S.; Armesto, J.J.; Berlow, E.; Bloomfield, J.; Dirzo, R.; Huber-Sanwald, E.; Huenneke, L.F.; Jackson, R.B.; Kinzig, A.; Leemans, R.; Lodge, D.M.; Mooney, H.A.; Oesterheld, M.; Poff, N.L.; Sykes, M.T.; Walker, B.H.; Walker, M.; Wall, D.H. 2000. Global biodiversity scenarios for the year 2100. Science. 287: 1770–1774.

Salo, P.; Korpimäki, E.; Banks, P.B.; Nordstrom, M.; Dickman, C.R. 2007. Alien predators are more dangerous than native predators to prey populations. Proceedings of the Royal Society B: Biological Sciences. 274(1615): 1237–1243.

Sauer, J.R.; Link, W.A. 2002. Hierarchical modeling of population stability and species group attributes from survey data. Ecology. 83: 1743–1751.

Schloss, C.A.; Nuñez, T.A.; Lawler, J.J. 2012. Dispersal will limit ability of mammals to track climate change in the Western Hemisphere. Proceedings of the National Academy of Sciences of the United States of America. 109(22): 8606–8611.

Schmidt, T.S.; Clements, W.H.; Wanty, R.B.; Verplanck, P.L.; Church, S.E.; San Juan, C.A.; Fey, D.L.; Rockwell, B.W.; DeWitt, E.H.; Klein, T.L.2012. Geologic processes influence the effects of mining on aquatic ecosystems. Ecological Applications. 22: 870–879.

Scholthof, K.B.G. 2007. The disease triangle: pathogens, the environment and society. Nature Reviews Microbiology. 5(2): 152–156.

Seamans, M.E.; Rau, R.D. 2020. American woodcock population status, 2020. Laurel, MD: U.S. Fish and Wildlife Service. 11. https://www.fws.gov/sites/default/files/documents/AmericanWoodcockStatusReport20.pdf. (13 July 2023).

Semlitsch, R.D.; O'Donnell, K.M.; Thompson III, F.R. 2014. Abundance, biomass production, nutrient content, and the possible role of terrestrial salamanders in Missouri Ozark forest ecosystems. Canadian Journal of Zoology. 92: 997–1004.

Shannon, G.; McKenna, M.F.; Angeloni, L.M.; Crooks, K.R.; Fristrup, K.M.; Brown, E.; Warner, K.A.; Nelson, M.D.; White, C.; Briggs, J.; McFarland, S.: Wittemyer, G. 2016. A synthesis of two decades of research documenting the effects of noise on wildlife: effects of anthropogenic noise on wildlife. Biological Reviews. 91: 982–1005. https://doi.org/10.1111/brv.12207.

Sheehan, T.; Bachelet, D.; Ferschweiler, K. 2015. Projected major fire and vegetation changes in the Pacific Northwest of the conterminous United States under selected CMIP5 climate futures. Ecological Modelling. 317: 16–29.

Siemers, B.M.; Schaub, A. 2011. Hunting at the highway: traffic noise reduces foraging efficiency in acoustic predators. Proceedings of the Royal Society B: Biological Sciences. 278(1712): 1646–1652.

Simon, T.P. 2011. Conservation status of North American freshwater crayfish (Decapoda: Cambaridae) from the southern Unites States. Proceedings of the Indiana Academy of Science. 120: 71-95.

Skerratt, L.F.; Berger, L.; Speare, R.; Cashins, S.; McDonald, K.R.; Phillott, A.D.; Hines, H.B.; Kenyon, N. 2007. Spread of chytridiomycosis has caused the rapid global decline and extinction of frogs. EcoHealth. 4(2): 125–134.

Smith, K.F.; Sax, D.F.; Lafferty, K.D. 2006. Evidence for the role of infectious disease in species extinction and endangerment. Conservation Biology. 20(5): 1349–1357.

Smith-Hicks, K.N.; Morrison, M.L. 2021. Factors associated with listing decisions under the U.S. Endangered Species Act. Environmental Management. 67: 563–573.

Sousa, R.; Gutiérrez, J.L.; Aldridge, D.C. 2009. Non-indigenous invasive bivalves as ecosystem engineers. Biological Invasions. 11(10): 2367–2385.

Stallknecht, D.E. 2007. Impediments to wildlife disease surveillance, research, and diagnostics. In: Childs, J.E.; Mackenzie, J.S.; Richt, J.A.,eds. Wildlife and emerging zoonotic diseases: the biology, circumstances and consequences of cross-species transmission. Berlin/ Heidelberg: Springer-Verlag: 445–461.

Stanford, C.B.; Iverson, J.B., Rhodin, A.G.J.; van Dijk, P.P.; Mittermeier, R.A.; Kuchling, G.; Berry, K.H.; Bertolero, A.; Bjorndal, K.A.; Blanck, T.E.G.; Buhlmann, K.A.; Burke, R.L.; Congdon, J.D.; Diagne, T.; Edwards, T.; Eisenberg, C.C.; Ennen, J.R.; Forero-Medina, G.; Walde, A.D. 2020. Turtles and tortoises are in trouble. Current Biology. 30(12): PR721–R735. https://doi.org/10.1016/j.cub.2020.04.088.

Sweet, L.C.; Green, T.; Heintz, J.G.C.; Frakes, N.; Graver, N.; Rangitsch, J.S.; Rodgers, J.E.; Heacox, S.; Barrows, C.W. 2019. Congruence between future distribution models and empirical data for an iconic species at Joshua Tree National Park. Ecosphere. 10(6): e02763.

Taylor, C.A.; DiStefano, R.J.; Larson, E.R.; Stoeckel, J. 2019. Towards a cohesive strategy for the conservation of the United States' diverse and highly endemic crayfish fauna. Hydrobiologia. 846(1): 39–58.

Taylor, C.A.; Schuster, G.A.; Cooper, J.E.; DiStefano, R.J.. 2007. A reassessment of the conservation status of crayfish of the United States and Canada after 10+ years of increased awareness. Fisheries. 32: 372–389.

Taylor, C.A.; Warren Jr., M.L.; Fitzpatrick Jr., F. 1996. Conservation status of crayfishes of the United States and Canada. Fisheries. 4: 25–38.

Tickner, D.; Opperman, J.J.; Abell, R.; Acreman, M.; Arthington, A.H.; Bunn, S.E.; Cooke, S.J.; Dalton, J.; Darwall, W.; Edwards, G.; Harrison, I.; Hughes, K.; Jones, T.; Lecleare, D.; Lynch, A.J.; Leonard, P.; McClain, M.E.; Muruven, D.; Olden, J.D.; Ormerod, S.J.; Robinson, J.; Tharme, R.E.; Thieme, M.; Tockner, K.; Wright, M.; Young, L. 2020. Bending the curve of global freshwater biodiversity loss: An emergency recovery plan. BioScience. 70(4): 330–342.

Tockner K.; Stanford J.A. 2002. Riverine floodplains: present state and future trends. Environmental Conservation. 29: 308–330.

Tompkins, D.M.; Dunn, A.M.; Smith, M.J.; Telfer, S. 2011. Wildlife diseases: from individuals to ecosystems. Journal of Animal Ecology. 80(1): 19–38.

Tscharntke, T.; Klein, A.M.; Kruess, A.; Ingolf, S-D; Thies, C. 2005. Landscape perspectives on agricultural intensification and biodiversity–ecosystem service management. Ecology Letters. 8(8): 857–874.

U.S. Global Change Research Program [USGCRP]. 2017. 2017: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., Fahey, D.W.; Hibbard, K.A. [et al.] (eds.)]. Washington, DC: U.S. Global Change Research Program. 470 pp. https://doi.org/10.7930/J0J964J6.

Van Devender, T.R.; Avila-Villegas, S.; Emerson, M.; Turner, D.; Flesch, A.D.; Deyo, N.S. 2013. Biodiversity in the Madrean Archipelago of Sonora, Mexico. Proceedings RMRS-P-67. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 10-16.

Van Hemert, C.; Pearce, J.M.; Handel, C.M. 2014. Wildlife health in a rapidly changing North: focus on avian disease. Frontiers in Ecology and the Environment. 12(10): 548–556.

Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. In Ecosystem management. New York: Springer: 183–191.

Wake, D.B.; Vredenburg, V.T. 2008. Are we in the midst of the sixth mass extinction? A view from the world of amphibians. Proceedings of the National Academy of Sciences of the United States of America. 105(Supplement 1): 11466–11473.

Wang, Z.; Piao, S.; Ciais, P.; [et al.]. 2011. Spring temperature change and its implication in the change of vegetation growth in North America from 1982–2006. Proceedings of the National Academy of Sciences of the United States of America. 108(4): 1240–1245.

Wear, D.N.; Prestemon, J.P. 2019. Spatiotemporal downscaling of global population and income scenarios for the United States. PLoS One. 14(7): e0219242.

Weiskopf, S.R.; Rubenstein, M.A.; Crozier, L.G.; Gaichas, S.; Grifffis, R.; Halofsky, J.E.; Hyde, K.J.W.; Morelli, T.L.; Morisette, J.T.; Munoz, R.C.; Pershing, A.J.; Peterson, D.L.; Poudel, R.; Staudinger, M.D.; Sutton-Grier, A.E.; Thompson, L.; Vose, J.; Weltzin, J.F.; Whyte, K.P. 2020. Climate change effects on biodiversity, ecosystems, ecosystem services, and natural resource management in the United States. Science of the Total Environment. 733: 137782

West, J.M.; Julius, S.H.; Kareiva, P.; Enquist, C.; Lawler, J.J.; Petersen, B.; Johnson, A.E.; Shaw, M.R.. 2009. U.S. natural resources and climate change: concepts and approaches for management adaptation. Environmental Management. 44: 1001–1021.

Westerling, A.L.R. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B: Biological Sciences. 371: 20150178. http://dx.doi.org/10.1098/rstb.2015.0178.

Westphal, M.I.; Field, S.A.; Tyre, A.J.; Paton, D.; Possingham, H.P. 2003. Effects of landscape pattern on bird species distribution in the Mt. Lofty Ranges, South Australia. Landscape Ecology. 18(4): 413–426.

Wilcove, D.S.; Rothstein, D.; Dubow, J.; Phillips, A.; Losos, E. 1998. Quantifying threats to imperiled species in the United States: assessing the relative importance of habitat destruction, alien species, pollution, overexploitation, and disease. BioScience. 48(8): 607–615.

Wilkinson, D.A.; Marshall, J.C.; French, N.P.; Hayman, D.T.S. 2018. Habitat fragmentation, biodiversity loss and the risk of novel infectious disease emergence. Journal of the Royal Society Interface. 15(149): 20180403.

Wobeser, G.A. 2007. Disease in wild animals: investigation and management, 2nd Ed. New York: Springer. 403 p.

Wogan, G.; Bickford, D.; Carnaval, D.A. [et al.]. 2022 in press. Amphibian ecology and climate change. In: Wren, S.; Angulo, A.; Meredith, H.; Kielgast, J.; Dos Santos, M.; Bishop, P., eds. Amphibian Conservation Action Plan, Update: Thematic Working Groups and Chairs. IUCN Species Survival Commission, Amphibian Specialist Group. https://www.iucn-amphibians.org/working-groups/acap-update-2020-thematic-working-groups-and-chairs/. (8 July 2021).

Wohl, E. 2006. Human impacts to mountain streams. In: James, L.A.; Marcus, W.A.,eds. The human role in changing fluvial systems. Proceedings of the 37th International Binghamton Geomorphology Symposium. Geomorphology. 79: 217–248. http://doi.org/10.1016/j.geomorph.2006.06.020.

Wohl, E.; Lane, S.N.; Wilcox, A.C. 2015. The science and practice of river restoration. Water Resources Research. 51(8): 5974–5997.

Wolff, P.J.; Taylor, C.A.; Heske, E.J.; Schooley, R.L. 2015. Habitat selection by American mink during summer is related to hotspots of crayfish prey. Wildlife Biology. 21: 9–17.

Woo, P.T.; Leatherland, J.F.; Bruno, D.W., eds. 2006. Fish diseases and disorders. vol. 3. London, UK: CAB International.

Young, H.S.; Parker, I.M.; Gilbert, G.S.; Guerra, A.S.; Nunn, C.L. 2017. Introduced species, disease ecology, and biodiversity–disease relationships. Trends in Ecology & Evolution. 32(1): 41–54.

Authors:

Rebecca Flitcroft, USDA Forest Service, Pacific Northwest Research Station

Gwen Bury, USDA Forest Service, Pacific Northwest Research Station through Oak Ridge Institute for Science and Education

Linda Joyce, USDA Forest Service, Rocky Mountain Research Station (emeritus)

Shannon Kay, USDA Forest Service, Rocky Mountain Research Station

Michael Knowles, USDA Forest Service, Rocky Mountain Research Station

Mark Nelson, USDA Forest Service, Northern Research Station

Travis Warziniack, USDA Forest Service, Rocky Mountain Research Station