Chapter 11. Wildlife, Fish, and Biodiversity

The 2010 Resources Planning Act (RPA) Assessment (2010 RPA) reviewed recent trends in wildlife, fish, and biodiversity, showing varied responses, depending on the resource, suggesting varied conditions that depend on region, species group, or habitat type. For this RPA Update, we focused on four topics that were motivated by questions stemming from 2010 RPA findings or that were designed to improve on the resource analysis capability originally reported in the 2010 RPA. First, we extended the work that documented elevated housing growth in the amenity-rich areas near protected areas. Although the 2010 RPA speculated that other natural resources may be impacted by development near protected lands, for this RPA Update, we specifically tested whether biodiversity (as reflected by bird communities) on the boundary of and internal to protected areas was affected by housing development near public lands. Second, we provided a more detailed case study of wildlife habitat stress attributable to climate change across the RPA Rocky Mountain Region. This work improved on the

Highlights

- Increasing housing development near protected areas has affected bird communities at the boundary of and within protected areas. Bird species that thrive under human settlement (many of which are exotic) benefited from such development at the expense of bird species of conservation concern.
- Wildfire management affects our assessment of climate stress to wildlife habitat. Suppression of fires in semidesert systems increased stress because of turnover in the historical vegetation types. An absence of suppression in more mesic systems increased stress because of changes in habitat productivity. Geographic variability in habitat stress and its sensitivity to fire management suggest that wildlife-oriented strategies for adapting to climate change will have to be tailored to the circumstances specific to each landscape.
- Areas supporting high counts of federally listed threatened or endangered species have remained consistent over time. Our new data, however, indicate the emergence of new areas of concentration in the interior highlands and plateau regions of the Central United States.
- Residents of watersheds that support a high density of at-risk aquatic species also often share an interest in drinking-water protection. Watersheds with high rates of urbanization and low percentages of protected lands could serve as criteria to target watersheds where shared funding can jointly benefit seemingly competitive stakeholder groups.

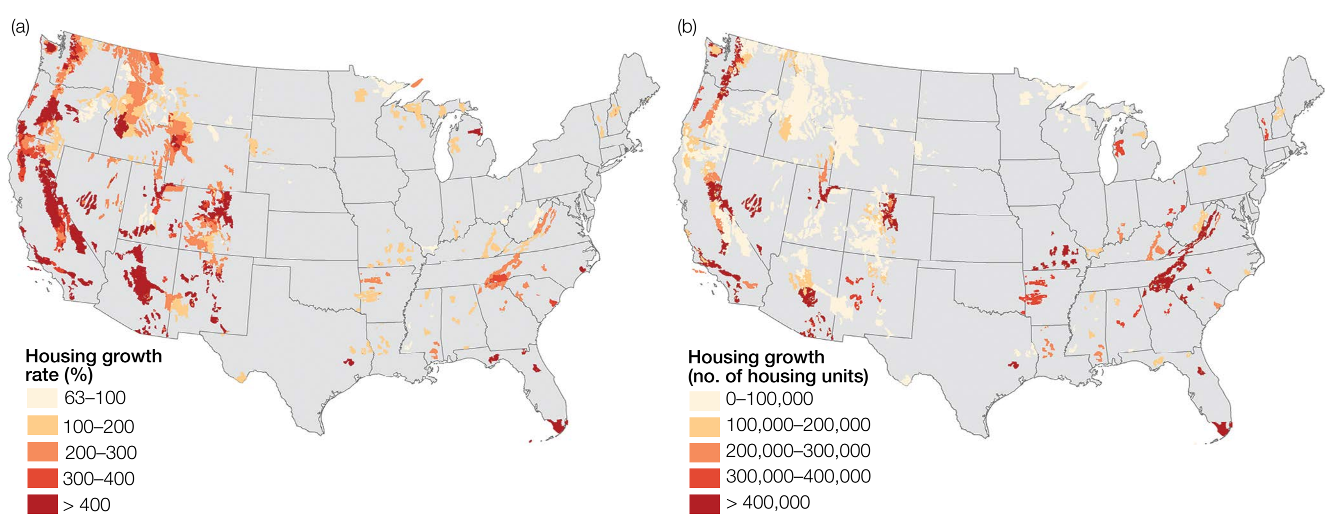
analyses presented in the 2010 RPA by increasing the spatial resolution of the analysis grid, incorporating the results from a new dynamic vegetation model, and examining the effects of wildfire management on evaluations of wildlife habitat stress. Third, we updated the status of imperiled species, using a new approach for assessing the distribution of formally listed and imperiled species that was based on an equal-area grid. Finally, the 2010 RPA noted that taxonomic groups associated with aquatic habitats had higher proportions of imperiled species than other kinds of species. Because of the high degree of imperilment among aquatic taxa, we analyzed at-risk aquatic species occurrence and drinking-water protection as an example of the potential joint benefits that can accrue to both drinking-water quality and species conservation.

Bird Diversity at the Boundary: How Are Birds Responding to Housing Development in and Near National Forests and Other Protected Areas?

- ❖ Bird communities varied, depending on whether surveys were conducted within protected areas, on the boundary of those areas, or outside the areas.
    - Species that thrive in the presence of humans were a more abundant component of bird communities outside and on the boundary of protected areas.
- ❖ The density of housing outside protected areas can affect the composition of bird communities within the areas.
    - Surveys conducted in protected areas with higher home densities outside their boundaries tended to support lower proportions of species of conservation concern.
- ❖ Housing near protected areas can strain the biodiversity conservation benefits within the areas.
    - Without effective measures to manage the rates and locations of exurban development, the conservation benefit of protected areas will likely diminish.

Privately owned forest, grassland, and shrubland habitats are being used more intensively as housing and road development increases to support growing human populations. As reviewed in the 2010 RPA, such development has been particularly strong in proximity to national forests, wilderness areas, and national parks (Radeloff et al. 2010) because of the high natural amenity values that attract housing development on inholdings and adjacent private lands. This work found housing growth rates were greater in the West on a relative scale (figure 11-1a) and greater in the East on an absolute scale (figure 11-1b). Areas that showed both high relative and absolute rates of housing growth included the southern Appalachians; the foothills and front ranges near major metropolitan areas in Colorado, Utah, and Washington; montane habitats in the arid Southwest; and southern California.

Figure 11-1. (a) Relative and (b) absolute housing growth rates within a 50-kilometer (~31-mile) buffer around the outer boundary of each national forest, wilderness area, and national park during the period 1940 to 2000.



Source: Radeloff et al. (2010).

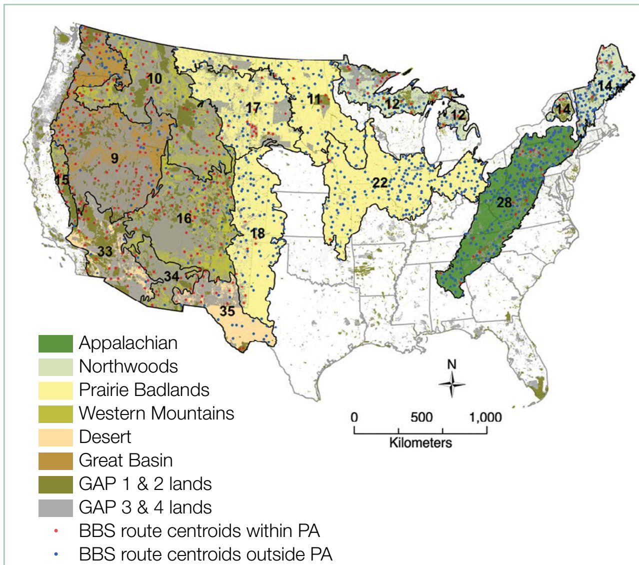
Expanding human populations and attendant land use changes have long been known to be primary factors driving changes in biological diversity (Foley et al. 2005; Sala et al. 2000). As private lands bear the growing burden of human-associated ecosystem stresses, public lands become increasingly important to the conservation of biological resources (Flather et al. 2009; Robles et al. 2008). Despite a common perception that public lands are critical to sustaining our biological heritage, little effort has been directed at understanding the proximity effects of private land use intensification on forest and rangeland biodiversity. In light of the perceived role that protected lands play in conserving biodiversity, Wood et al. (2014) investigated whether increased housing development in and near these lands had detectable effects on bird diversity.

Four primary data sources were used to address this question. First, measures of bird diversity were derived from the North American Breeding Bird Survey (BBS), an annual roadside survey that has been conducted since 1966 (Sauer et al. 2008). Second, protected lands data were obtained from the U.S. Geological Survey National Gap Analysis Program Protected Area Database (version 1.2, released in April 2011), which accounts for private inholdings within the administrative boundaries of public lands (http://gapanalysis.usgs.gov/padus/). Third, housing data were derived from the U.S. decennial census and processed at the partial block group level as described in Hammer et al. (2004). Finally, land use and land cover data were based on the 2001 National Land Cover Database data at 30-meter resolution (Homer et al. 2004). These data sources were linked via the digital BBS route paths, using a buffer set at the bird-detection distance specifications of the BBS—400 meters on either side of the route.

For each BBS route, we calculated the proportional abundance and the proportional richness of several bird groups as our response variables. These bird groups included synanthropes, land cover affiliates, and species of greatest conservation need. Synanthropes are native and nonnative species that are associated with human-modified environments during the breeding season. Land cover affiliates are species that are associated with the dominant natural land cover type of a BBS route, which included forest and woodland, grassland, or shrubland breeders. The species of greatest conservation need bird group was defined by combining the bird species identified as species of greatest conservation need in each individual State Wildlife Action Plan (Association of Fish and Wildlife Agencies 2011). Bird survey routes were classified as within protected areas (defined as more than 50 percent of the BBS route occurring within protected lands), at the boundary of protected areas (defined as between 0.1 percent and 49.5 percent of the BBS route occurring within protected lands), or on private lands outside protected areas (defined as 0 percent of the BBS route occurring within protected lands); these three categories were used as treatments for analyses. To address our main question, we quantified the effect of housing density within (i.e., associated with private inholdings), at the boundary, and outside protected areas on the proportional abundance and richness of synanthropes, land cover affiliates, and species of greatest conservation need.

Within the conterminous United States, we selected six regional study areas based on Bird Conservation Regions (BCRs) (figure 11-2) that are ecologically unique regions with similar climate, vegetation, land use, and avian communities and that were developed by the U.S. North American Bird Conservation Initiative Committee (2000). We excluded BCRs from our analysis when (1) a region had few protected areas, which caused an insufficient sample of routes within protected areas, and (2) a region had steep elevation gradients, so that routes within protected areas could not be matched ecologically with routes outside protected areas.

Figure 11-2. Distribution of 1,225 Breeding Bird Survey (BBS) centroids within and outside protected areas (PA) throughout six broad regions of the United States that were made up of aggregations of Bird Conservation Regions (BCRs) as follows: Appalachian Region (BCR 28); Northwoods Region (BCRs 12 and 14); Prairie Badlands Region (BCRs 11, 17, 18, and 22); Western Mountains Region (BCRs 10, 15, 16, and 34); Desert Region (BCRs 33 and 35); and Great Basin Region (BCR 9). The U.S. Geological Survey National Gap Analysis Program (GAP) Protected Area Database was used to define lands having permanent protection from conversion of natural land cover and being mandated to maintain a natural state (GAP 1); lands having permanent protection from conversion of natural land cover and being mandated to maintain a primarily natural state but permitting management practices that may degrade the quality of existing natural communities (GAP 2); an area having permanent protection from conversion of natural land cover for most of the area but permitting extractive uses (GAP 3); and lands having no known mandate to prevent conversion of natural habitat types to anthropogenic habitat types (GAP 4).



Sources: U.S. North American Bird Conservation Initiative Committee (2000); Wood et al. (2014).

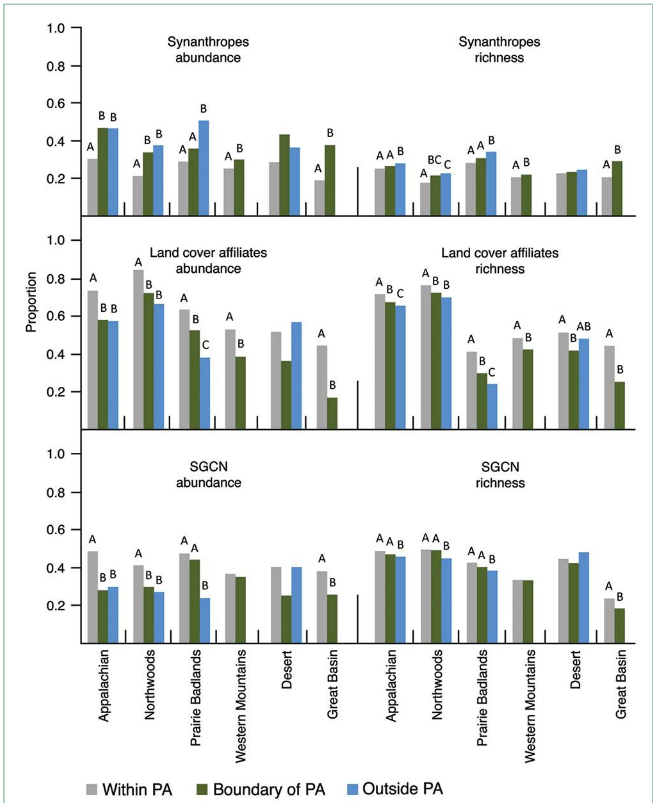
Results

Bird communities varied considerably among treatments in all BCRs except for the Desert and Western Mountains, where they were similar. The avian communities within protected areas were largely different from communities on private lands outside of protected areas and to a lesser extent from those communities along the protected-area boundaries, where the differences were less pronounced. The proportional abundance and proportional richness of synanthropes were significantly higher outside protected areas or at the boundaries than within the areas (figure 11-3). The only exception to this pattern was the Desert Region, where both the proportional abundance and proportional richness of synanthropes were similar among treatments (figure 11-3). Although the differences were qualitatively apparent, sample sizes were small for the Desert Region and, thus, affected our ability to detect differences among treatments.

Conversely, the proportional abundance and proportional richness of land cover affiliates and species of greatest conservation need were significantly higher within protected areas than either at the boundaries of those areas or outside the areas (figure 11-3). This pattern was true for all bird conservation regions, except in the Western Mountains and Desert Regions, where the proportional abundance and richness of these bird groups were similar among our housing treatments, particularly among species of greatest conservation need (figure 11-3).

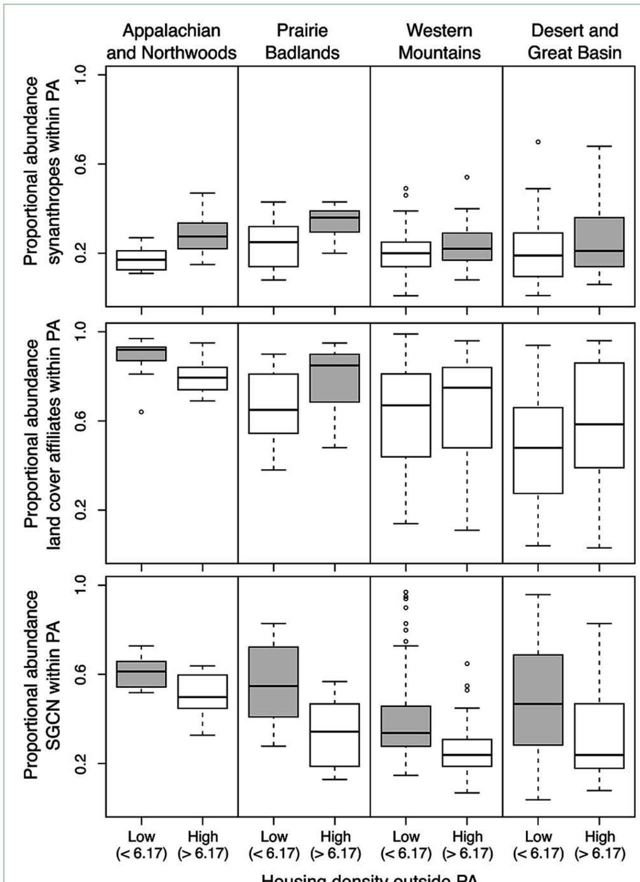
Although housing within protected areas, in general, was the best supported variable influencing protected-area bird groups, in all regions, high-intensity housing outside protected areas was associated with higher proportional abundance of synanthropes within protected areas in all regions (figure 11-4). In contrast, high-intensity housing outside protected areas resulted in lower proportional abundance of species of greatest conservation need within protected areas (figure 11-4). We observed similar relationships for land cover affiliates in the Appalachian and Northwoods Regions (figure 11-4). In the Prairie Badlands

Figure 11-3. Mean proportional abundance and richness response of synanthropes, natural land cover affiliates, and species of greatest conservation need (SGCN). The three treatment types are (1) within protected areas (PA), (2) on the boundary of PA, and (3) outside PA. Histograms sharing the same letter (A, B, C) indicate the response does not differ significantly among treatments. Histogram bar sets with no letters indicate no significant response difference among any treatments.



Source: Wood et al. (2014).

Figure 11-4. Box plot summaries of the proportional abundance of synanthropes, natural land cover affiliates, and species of greatest conservation need (SGCN) within protected areas (PA) of four regional areas. Housing outside PA was categorized as low (< 6.17) or high (> 6.17) based on the definition of the wildland urban interface (6.17 houses per square kilometer or 1 house per 40 acres). Boxes shaded in gray indicate that the proportional abundance of a protected-area bird group was significantly different between housing-intensity levels.



Source: Wood et al. (2014).

Region, however, land cover affiliates within protected areas unexpectedly were proportionally more abundant when housing intensity was high on private lands outside the areas. We detected similar patterns for the association between the proportional richness within protected areas and high-intensity housing outside the areas, but the differences were weaker than proportional abundance.

The positive benefits observed among land cover affiliates within protected areas of higher density housing outside protected areas may be an artifact of our sample; however, the results also may be a real biological response. This pattern has been described as an "oasis effect" caused by the presence of novel habitats and food/water supplementation associated with housing development (Lerman and Warren 2011; Robb et al. 2008), particularly when it occurs in harsh environments (Bock et al. 2008). Our findings of the positive association between richness of land cover affiliates within protected areas and housing outside protected areas in the central prairies and western mountains of the United States are consistent with this oasis effect. We should also note, however, that this effect may be short lived with continued increases in housing eventually having a negative effect on bird communities, suggesting that there may be housing density thresholds that, when exceeded, can cause avian community structure and abundance to suffer (Pidgeon et al. 2014; Suarez-Rubio et al. 2013).

Implications

Our results suggest that housing development both within and at the boundary of protected areas impacts avian community structure. Housing on private inholdings or outside protected areas was often positively associated with the proportional abundance and proportional richness of synanthropic species and negatively associated with the proportional abundance and proportional richness of land cover affiliates and species identified by State agencies to be of conservation concern. Our results suggest that protected areas of the United States are successful at maintaining natural land cover and harboring avian communities of conservation concern compared with surrounding private lands. Thus, protected areas of the United States may serve as sources for regional avian metapopulations (Robinson et al. 1995).

Our results, however, also support findings that housing development near protected areas has created strains on protected areas themselves. A novel finding from our study was that, in addition to the pressure of private land development on biodiversity outside protected areas, this same development pressure threatens biodiversity within protected areas. Synanthropic species were proportionately more abundant (and more species rich) within protected areas when housing densities on private lands at the protected-area boundary were higher. In a similar way, the proportional abundance of species of conservation concern was consistently lower when housing density was higher at the boundary of protected areas. Although housing growth has recently slowed compared with the rapid growth in the 1970s (Radeloff et al. 2010), our findings suggest that even marginal increases of housing growth on the boundary of protected areas could degrade avian communities within the protected areas. Without effective measures to manage the rates and locations of exurban development, the conservation benefit of protected areas will likely diminish.

Climate Change Effects on Wildlife Habitat in the RPA Rocky Mountain Region

- ❖ An index of climate-induced stress to terrestrial wildlife habitat indicates that stress in the RPA Rocky Mountain Region is geographically varied.
  - Areas of low climate stress to wildlife habitat occur in southern Arizona and New Mexico, and areas of high climate stress are associated with the desert and semidesert ecoregions of Arizona, Idaho, and Nevada and the temperate steppe ecoregion of Colorado and Wyoming.
- ❖ Stress to wildlife habitats is affected by wildfire and fire management.
  - Strategies to actively suppress fires result in higher stress among the Intermountain semidesert and desert ecoregions, whereas strategies directed at not suppressing fires result in higher stress among the steppe-coniferous and open woodland ecoregions.
- ❖ These varied patterns indicate that there will be no general strategy for addressing climate change stress to terrestrial wildlife habitats.
  - Strategies will need to be tailored to the circumstances, including fire history and changing climate, that characterize various landscapes.

The RPA Rocky Mountain Region (see figure 2-1 in chapter 2) is a mosaic of public and private land. Federal (national forests, national parks and monuments, wildlife refuges), State, and local governments manage 75 percent of this region. Any management or policy response to climate change directed at conserving wildlife resources will need to be integrated across administrative boundaries if landscape-scale phenomena like wildlife species movement via corridors are to be adequately addressed. A regionally consistent information base facilitates the collaboration necessary for managing wildlife species and their habitat. In the 2010 RPA, we used a nationally consistent approach to assess the impact of climate change on wildlife, the Terrestrial Climate Stress Index (TCSI) (Joyce et al. 2008). That analysis identified areas of relatively high and low habitat stress across the conterminous United States and concluded that locations where current conservation issues were most pronounced tended not to overlap with the location of high future stress associated with climate change. This situation potentially complicates efforts of managers to prioritize wildlife conservation actions. That analysis did not explicitly incorporate wildfire as an important driver of change to wildlife habitat. Here, we focus on climate change and wildfire effects on the RPA Rocky Mountain Region, a landscape on which fire greatly influences the distribution of vegetation types.

Within the RPA Rocky Mountain Region are found the driest areas (Southwest) and some of the coldest areas (high-elevation Rocky Mountains) of the United States. Only 20 percent of the region is forest land, and the remaining habitat ranges from woodland and shrubland types to grasslands grading into desert vegetation. These habitats are influenced by climate and the elevational gradients in mountainous terrain. Climate change will affect wildlife directly through temperature and moisture changes and indirectly by altering habitat availability and quality as driven by natural disturbances and land use change. This region is also noted for the influence of fire on vegetation distribution; fire regimes range from frequent surface fires to crown fires that occur infrequently. Wildfire area has increased in size annually and is projected to continue to increase further. This disturbance is of concern because it can change a landscape in a short period of time, by comparison with gradual changes in mean climate. The United States has a long history of suppressing wildfires. We explore how changing wildfire regimes and climate change may interact to affect habitat.

Terrestrial Climate Stress Index

The TCSI is the sum of three separate metrics that reflect changes in (1) climate regime (temperature and precipitation), (2) biomass production, and (3) vegetation type area. We used a historical period of 1950 to 1999 to define baseline conditions that, when compared with measures during a future period of 2050 to 2099, defined the magnitude of change for each metric during a 100-year period. The geographic extent of our analysis is the 12 States within the RPA Rocky Mountain Region. In the 2010 RPA, the spatial scale of analysis was a grid cell of nearly 2,500 square kilometers (km^2) (~954 square miles [mi^2]]. In this analysis, the scale is finer—an equal-area hexagon grid with each cell approximating an area of 69 km^2 (~27 mi^2).

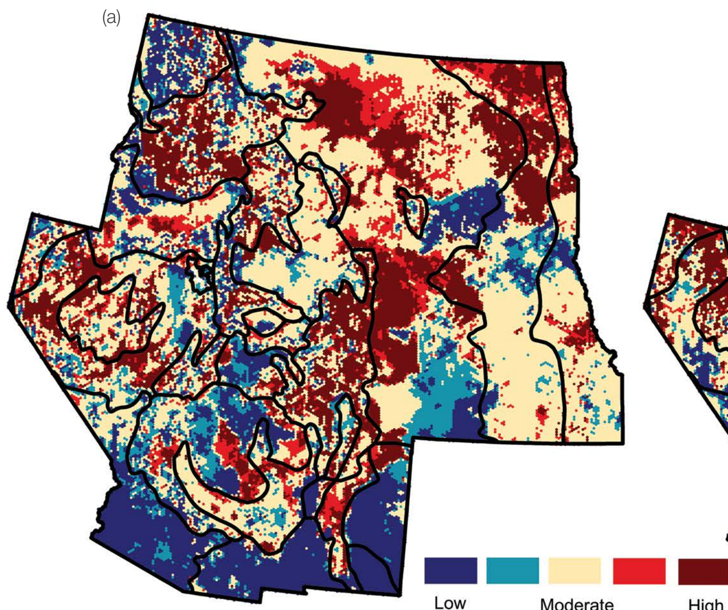
We used climate projections from three general circulation models (GCMs) (MIROC2medres, UKMO HadCM3, and CSIRO-MK3.0) forced by three 4th Intergovernmental Panel on Climate Change Assessment scenarios: A1B, A2, and B1 (Bachelet et al. 2014). Future projections of biomass and shifts in vegetation types were obtained from the dynamic global vegetation model MC2—a model that projects vegetation response to climate change and disturbance (fire, drought) based on biogeographic and biogeochemical processes in ecosystems (Bachelet et al. 2014). The MC2 model, which is a revision of the MC1 model that was used in the 2010 RPA, is designed to provide a more accurate representation of fire and vegetation dynamics (Bachelet et al. 2014). Within the MC2 model, the balance between trees and grass is determined by simulated competition between these two life forms, as mediated by fire. At the prairie-forest boundary of the eastern Great Plains, frequent surface wildfires historically have maintained grassland types. Under fire suppression and climate change, MC2 projects woody expansion onto the Great Plains (Bachelet et al. 2014). Fire suppression is assumed to be highly effective and only catastrophic fires can escape (Bachelet et al. 2014). Examples of such large fires can occur in western mountainous areas when warmer, drier conditions result in the drying of abundant fuels.

Regional High- and Low-Stress Areas

The TCSI can be used to identify areas of high and low habitat stress across the Rocky Mountain region (figure 11-5), where high stress is defined as the top 20 percent highest scoring grids and low stress is the 20 percent grids with the lowest TCSI scores. We show the ensemble TCSI score for the A2 scenario (estimated as the mean across the three climate models) and for each of the two fire scenarios (suppression and no suppression of wildfire). The areas of highest stress, as estimated by the index, are represented in red in figure 11-5. The influence of topography can be seen across the mountainous West as high-stress areas transition to low-stress areas in a relatively short distance (a matter of a few grid cells) in response to steep-elevation gradients characterizing these montane systems. Across the plains, areas of high or low stress are more expansive, reflecting a more homogenous landscape.

Common areas of low stress across both fire scenarios are projected to occur in southern Arizona and New Mexico, the plateau region of northeastern Arizona and southeastern Utah, and the high plains of southeastern Colorado and western Kansas. Future shifts in climate (temperature and precipitation) are greatest in the southwest part of the region (not shown), because temperature is expected to rise and precipitation is expected to decline. In the northern part of the Rocky Mountain Region, the projection is for temperature to rise and precipitation to increase slightly. Even though the greatest changes in

Figure 11-5. Terrestrial Climate Stress Index (TCSI) for the RPA Rocky Mountain Region under the A2 scenario and three climate models in which (a) fire is not suppressed and (b) fire is suppressed. High stress is defined as those grid cells with TCSI scores in the top 20 percent; low stress is defined as scores in the lower 20 percent. Ecoregion provinces (thick black lines) after Bailey (1995).



RPA = Resources Planning Act.

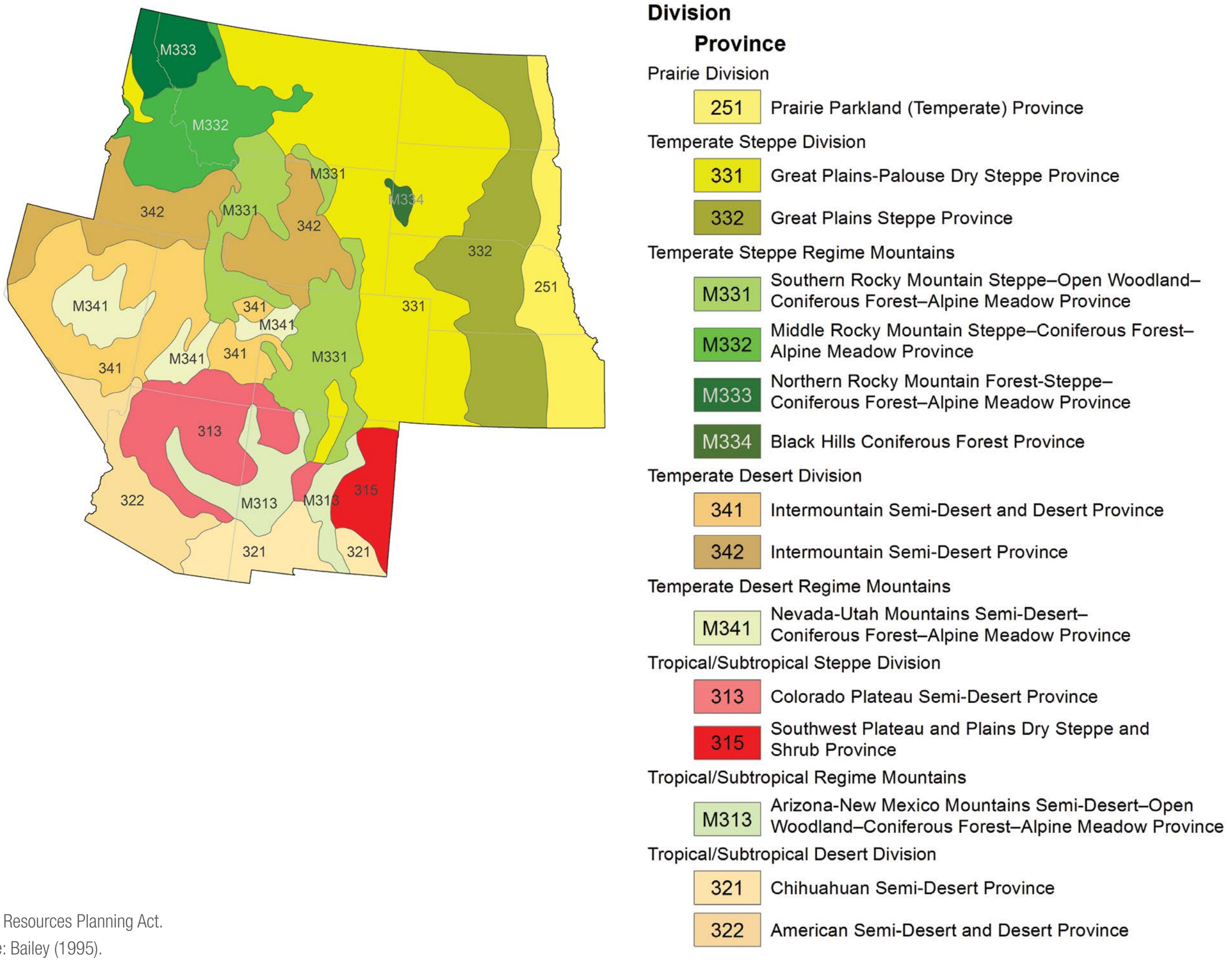
climate are projected to occur in the Southwest, the occurrence of high-stress areas is also influenced by future changes in biomass production and vegetation types as mediated by disturbances such as fire. Future biomass production changes and vegetation type shifts are greater in central to northern parts of the region than in the Southwest; as a consequence, these central to northern areas are projected to have higher stress (figure 11-5).

Ecoregional Pattern of High-Stress Areas in the RPA Rocky Mountain Region

The regional expectation of high stress is defined by the high-stress cut point used to quantify the degree of relative climate stress—20 percent. We explored the percent of high-stress areas within each ecoregion province in the RPA Rocky Mountain Region (figure 11-6). Only one province (the Great Plains-Palouse Dry Steppe) shows a pattern of high-stress area that approximates (within ± 5 percent) the regional 20-percent expectation under both suppression and no-suppression fire scenarios (figure 11-7). For all other provinces, departure from the regional expected value is greatly influenced by the ecosystem geography of the province.

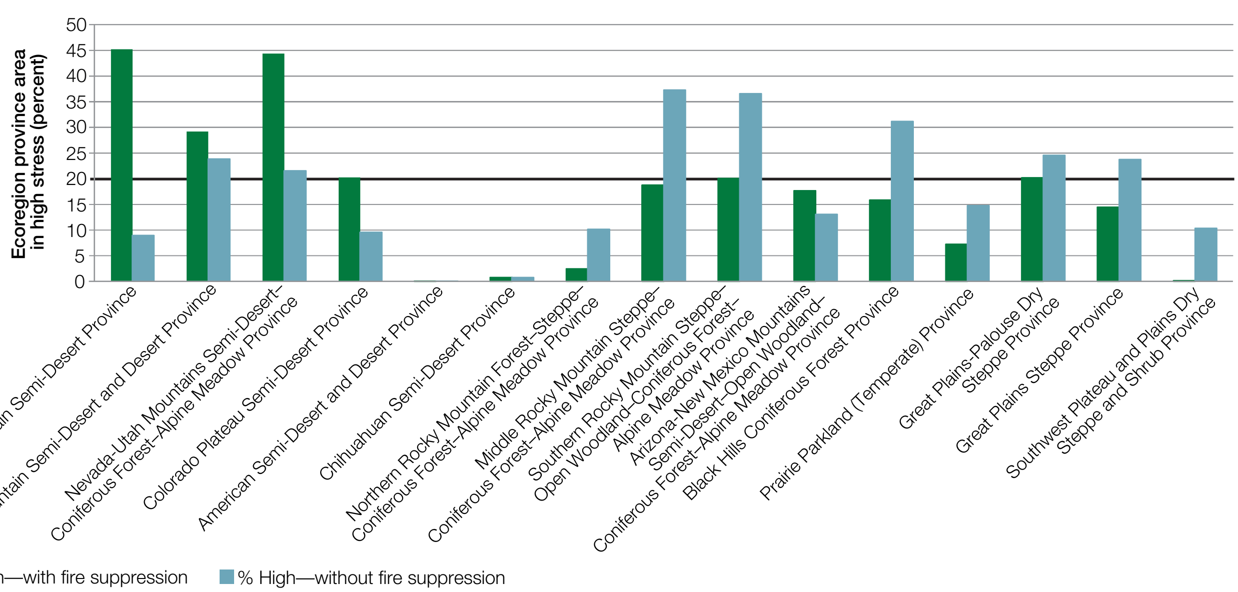
One measure of how sensitive our estimates of climate stress are to the wildfire scenarios is captured by the difference in the percent of high-stress areas within a province under suppression and no-suppression fire scenarios (figure 11-7). Under this definition of sensitivity, the Intermountain Semi-Desert, Intermountain Semi-Desert and Desert, and the Nevada-Utah Mountains Semi-Desert–Coniferous Forest–Alpine Meadow provinces all showed notably (greater than ± 15-percent difference between fire scenarios) greater high-stress areas with fire suppression when compared with the no-suppression scenario. This higher stress associated with fire suppression was caused primarily by complete turnover in historical vegetation types when temperate forest and woodlands replaced temperate shrublands. Conversely, forested provinces in the mountainous temperate steppe (i.e., Middle Rocky Mountain Steppe–Coniferous Forest–Alpine Meadow, Southern Rocky Mountain Steppe–Open Woodland–Coniferous Forest–Alpine Meadow, and Black Hills Coniferous Forest) showed that high climate stress was more prominent on the landscape without fire suppression when compared with the suppression scenario. Although some shifts occurred among vegetation types, when fire is not suppressed, changes in climate and increased wildfire result in biomass declines in the mountain provinces. Fire suppression affects habitat quality, particularly when woody encroachment occurs. Bachelet et al. (2014) report that fire suppression in MC2 results tends to increase the woody component of arid western ecosystems.

Figure 11-6. Baailey ecoregion divisions and provinces in the RPA Rocky Mountain Region.



RPA = Resources Planning Act.
Source: Bailey (1995).

Figure 11-7. Percent of Bailey ecoregion provinces classified as high climate stress according to Terrestrial Climate Stress Index based on the A2 emissions scenario. Paired bars compare estimates with and without fire suppression. Bar heights reflect the ensemble mean across three climate models. Bolded horizontal line defines the regionwide 20 percent expectation.



Source: Ecoregions defined by Bailey (1995).

Implications

Identification of areas of relatively high and low habitat stress can provide managers and planners with information on the potential for climate-induced stress to wildlife habitats within regions and States. Spatially explicit information on habitat stress attributed to climate change can be integrated with the location of current conservation issues to evaluate the coincidence of future climate change threats with important wildlife conservation issues. These results suggest that the historic influence of fire on vegetation types is important to consider in projecting the future effects of climate change on wildlife habitat. For situations in which a particular fire regime has been a major influence in sustaining a vegetation type, such as surface fires and grasslands, the effect of a changing climate and a changing fire regime will have different responses depending on the management of wildfire. These findings indicate no general strategy will be effective for addressing climate change stress to terrestrial wildlife habitats. Rather, strategies will need to be tailored to the circumstances that characterize various landscapes (as reflected in ecoregional provinces here). As in our previous analysis, these results also indicate that locations where current conservation issues were most pronounced (e.g., concentrations of at-risk biodiversity; see figure 11-9) tend not to overlap with the location of high future stress associated with climate change. If our model is an accurate reflection of the geography of stress to wildlife communities, then these differences will further complicate the efforts of managers to prioritize wildlife conservation actions.

We highlight four limitations that are important to keep in mind when interpreting the implications of our analysis. First, our index was designed to capture the projected shifts in natural vegetation in response to climate change, but it did not incorporate land use as a factor affecting wildlife habitat availability. Second, invasive plant species are expected to affect broad areas under climate change (Walther et al. 2009), but we do not yet know how to incorporate their effect on wildlife habitat quality. Third, the vegetation dynamics model used here does not simulate wetland vegetation types that serve as an important terrestrial-aquatic transition, that support a high biological diversity, and that may be particularly vulnerable to climate change effects in drier ecosystems (Johnson et al. 2005; Perry et al. 2012). Fourth, our analysis bracketed potential future climates based on three scenarios: (1) B1, with the lowest future emissions; (2) A2, with the second highest future emissions used in the IPCC (2007); and (3) A1B, with a middle future projection. In addition, for each scenario, we used three GCMs with different sensitivities to atmospheric carbon dioxide and implemented two management scenarios (fire suppression or no fire suppression). Additional scenarios with different emission levels would fill in the range of our analysis and provide additional detail on the behavior of the TCSI.

Trends and Geography of At-Risk Biodiversity

- ❖ The rate of listing Federal threatened and endangered species is accelerating in response to efforts to process listing decisions on the backlog of candidate and proposed species by 2018.
- ❖ Areas of listed species concentration have become refined, but they still emphasize similar areas of geographic concentration.
    - New occurrence data for listed species revealed that areas of concentration in Hawaii, the southern Appalachians, peninsular Florida, coastal areas, and the arid Southwest have remained largely unchanged; an emerging area of concentration is indicated in the interior highlands and plateau region of southern Illinois, Indiana, and Missouri; northern Arkansas; and western Kentucky.

As the number of species considered to be rare increases, the likelihood of species extinction also increases. Demographic and environmental events, such as failure to find a mate, disease, disturbance, habitat loss, and climate change, interact to increase extinction risk as populations become smaller. Because important ecosystem functions (e.g., productivity, nutrient cycling, or resilience) can be degraded with the loss of species, there is concern that the goods and services humans derive from ecological systems will become diminished as more species become rare. For this reason, the RPA Assessment has long tracked national trends in the number of species that are formally listed as threatened and endangered and has also tracked the geographic patterns associated with where listed species occurrence is concentrated as indicators of ecosystem well-being.

A change in the rate of listing and a new analysis of listed species occurrence are the focus of this section. First, a recent settlement agreement was reached between the U.S. Fish and Wildlife Service (FWS) and the Center for Biological Diversity and Wild Earth Guardians (U.S. Court of Appeals for the District of Columbia 2013) to process the backlog of species awaiting listing decisions under the Endangered Species Act (ESA). This agreement will result in a more rapid pace of species additions to the list of those determined to be threatened or endangered than has been observed in the recent past. Upwards of 750 species will be considered for listing by 2018. Second, past RPA Assessments have geographically depicted areas of species concentration based on county-level occurrence data. Because counties vary greatly in size, identification of species concentration hotspots has been confounded by area effects. It has long been known that species accumulate with area nonlinearly (Flather 1996) and the accumulation rate varies among ecosystems (Rosenzweig 1995). Therefore, it is difficult to control for these area effects in a manner that leads to a general, unambiguous county ranking of increasing species concentration (Joppa et al. 2013). For this RPA Update, we analyzed the occurrence records of formally listed species on an equal-area grid across the United States, thus removing the area effects that have confounded past efforts.

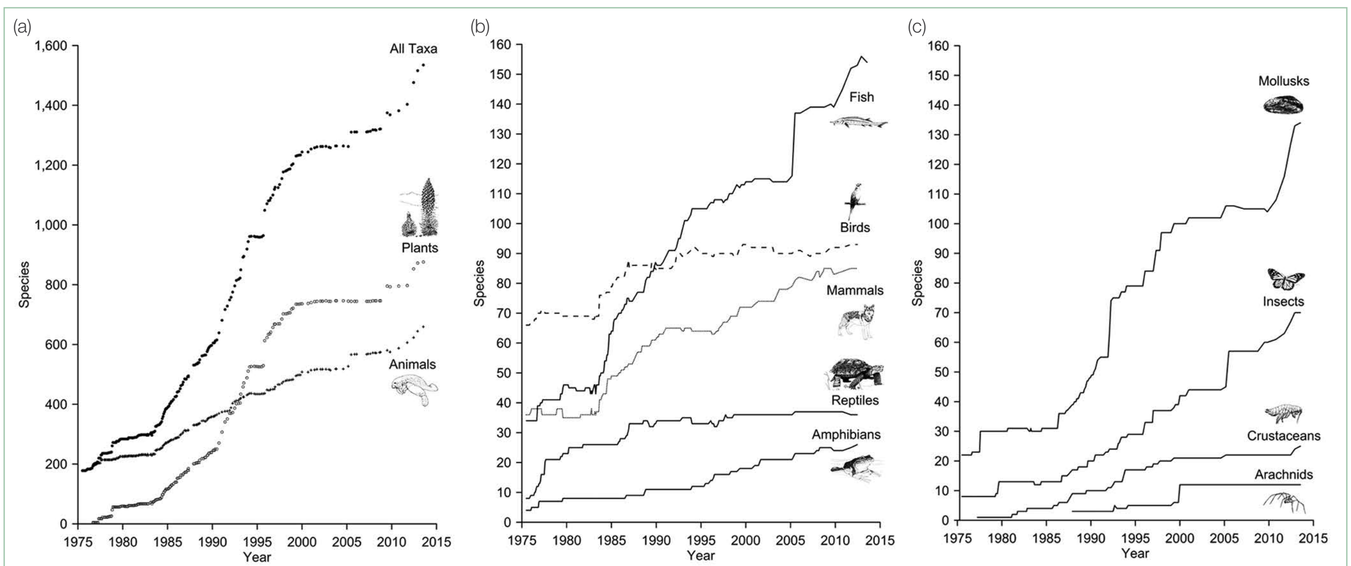
Recent trends in species listings and their geographic occurrence were derived from two existing data sources. First, trends in the number of species by broad taxonomic groups were compiled from the FWS's *Endangered Species Bulletin* and online resources that keep the current count of U.S. threatened and endangered species, subspecies, and distinct population segments (http://ecos.fws.gov/ecp0/reports/box-score-report). These data enabled us to document the cumulative number of listed species by major taxonomic groups from July 1, 1976, through July 14, 2014. Second, NatureServe's Central databases (NatureServe 2014) were used to compile lists of threatened and endangered species and species considered to be imperiled, according to NatureServe's global conservation status ranks (http://www.natureserve.org/explorer/ranking.htm). These lists were compiled on a systematic equal-area grid of 647.5 km^2 (250 mi^2) using contemporary (post-1970) occurrence records. Imperiled species include those determined to be critically imperiled (G1) or imperiled (G2)—a set that has been shown to be a less biased reflection of the true number of species of conservation concern than the number formally listed under ESA—the latter being affected by budget constraints, bureaucratic processes, and variation in listing policy over time (Master et al. 2000). Imperiled species are also thought to define the pool from which new ESA listings will be drawn and their geographic occurrence is thought to presage where new hotspots may emerge in the future (Evans et al. 2016). Maps of ESA-listed species or imperiled species were generated by classifying species counts within grid cells into approximate percentile classes to highlight those cells with relatively high species counts.

Results

As of July 14, 2014, 1,535 species were formally listed as threatened or endangered within the United States. That total represents a net gain of 167 species since the 2010 RPA (figure 11-8). The increase was distributed equally among animals (+84) and plants (+83). Among animal groups, there were notable increases in mollusks (+30), fish (+15), and insects (+10). The listing rate has increased to an average of 45 species per year—nearly doubling the recent average listing rate reported in the 2010 RPA (Flather et al. 2013). This increase is attributed, in large part, to the focused efforts of the FWS to make listing decisions about the more than 700 species named in the settlement agreement. Processing this pool of species by 2018 will likely result in even greater listing rate increases in the near term.

The distribution of ESA-listed species based on county occurrence data has been shown to vary geographically, with prominent hotspots of threatened and endangered species occurring in Hawaii, the southern Appalachians, peninsular Florida, coastal areas, and the arid Southwest that have remained largely unchanged since the late 1990s (Flather et al. 2013). The

Figure 11-8. Cumulative number of species listed as threatened or endangered (accounting for delistings) from July 1, 1976, through July 14, 2014, for (a) plants and animals, (b) vertebrate groups, and (c) invertebrate groups.



Source: Data from USFWS (2014).

higher resolution associated with the systematic grid data did refine the areas of listed species concentration (figure 11-9), but, broadly speaking, the regions supporting relatively high numbers of species have remained surprisingly consistent with earlier geographic descriptions, particularly among the Eastern States with smaller counties. Notable differences with county-based maps include a general de-emphasis of areas in the arid Southwest and the emergence of listed species concentrations associated with the interior highlands and plateau region of southern Missouri, northern Arkansas, western Kentucky, and southern Illinois and Indiana. It is also noteworthy that many regions outside the areas of concentration contain very few listed species. Overall, 54 percent of U.S. lands have no occurrence records of listed species.

Given the diversity of land form and land cover in areas supporting high numbers of listed species, it is not surprising that taxonomic composition varies among hotspots (figure 11-10). Since the early 1990s, **plants** have outnumbered animal species listed (figure 11-8a), and they are concentrated in areas characterized by high levels of endemism—species that uniquely occur in a restricted geographic area—including Hawaii, the Mediterranean climates of California, and the plant communities associated with the Florida peninsula inland scrub. Among animals, **invertebrates** dominate the pool of listed species in the interior highland ecoregions. More species of freshwater and anadromous **fish** receive ESA protections than any other vertebrate group, and they tend to concentrate in the arid Southwest, the Pacific Northwest, and the southern Appalachians. Herptiles (**amphibians** and **reptiles**) have fewer listed species than other vertebrate groups (figure 11-8b) and never attain large counts in any locale; however, they are prominently represented in coastal areas of the Southeast, as are **birds** and **mammals**.

Where are new areas of projected species concentration likely to emerge in the future? The recent settlement agreement (U.S. Court of Appeals for the District of Columbia 2013) makes this question particularly relevant. A mapping of occurrence among species identified in the settlement agreement (figure 11-11a) indicates that species likely to receive protections under ESA in the near term will both emphasize existing areas of concentration (e.g., the southern Appalachians) and also lead to the potential emergence of new areas of concentration (e.g., the southern Great Basin and the Ouachita and Boston Mountains of western Arkansas and eastern Oklahoma). We can get a sense of whether new areas of concentration will emerge in the longer term by mapping the occurrence of imperiled species that are neither formally listed under ESA nor part of the settlement agreement (figure 11-11b). Although not all species currently classified as imperiled are necessarily at risk of extinction (and therefore may not warrant protection under ESA), new listings under ESA will likely come from the pool of species considered imperiled by NatureServe, based on recent comparisons between listed and imperiled species (see Evans et al. 2016). The southwestern Basin (southern Arizona) and Range (New Mexico and western Texas) and portions of the Colorado Plateau (southeastern Utah) could emerge as new hotspots of listed species in the more distant future.

Figure 11-9. Geographic distribution of species formally listed as threatened or endangered under the Endangered Species Act. Data are derived from the National Heritage Programs as maintained by USDI FWS (2014) and mapped onto a systematic equal-area grid (647.5 square kilometers [250 square miles]) of the United States. Alaska and Hawaii are displayed on a different scale for presentation purposes.

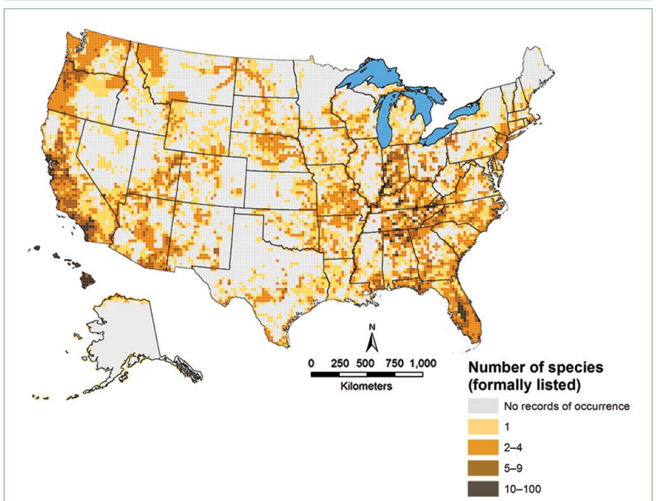


Figure 11-10. Variation in the taxonomic composition of species occurring in hotspots—areas of concentration of listed species. Hotspot boundaries are based on or are modifications of Bailey Ecoregions. Data are derived from the National Heritage Programs as maintained by NatureServe (2014) and mapped onto a systematic equal-area grid (647.5 square kilometers [250 square miles]) of the United States. Hawaii is displayed on a different scale for presentation purposes and Alaska is not displayed because it lacks hotspots.

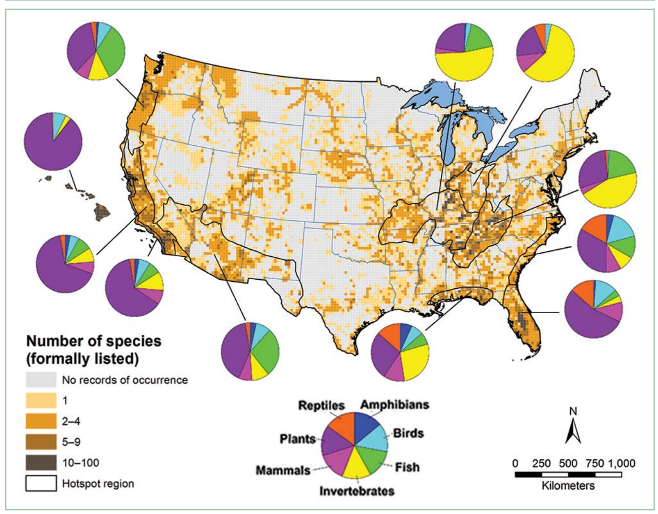
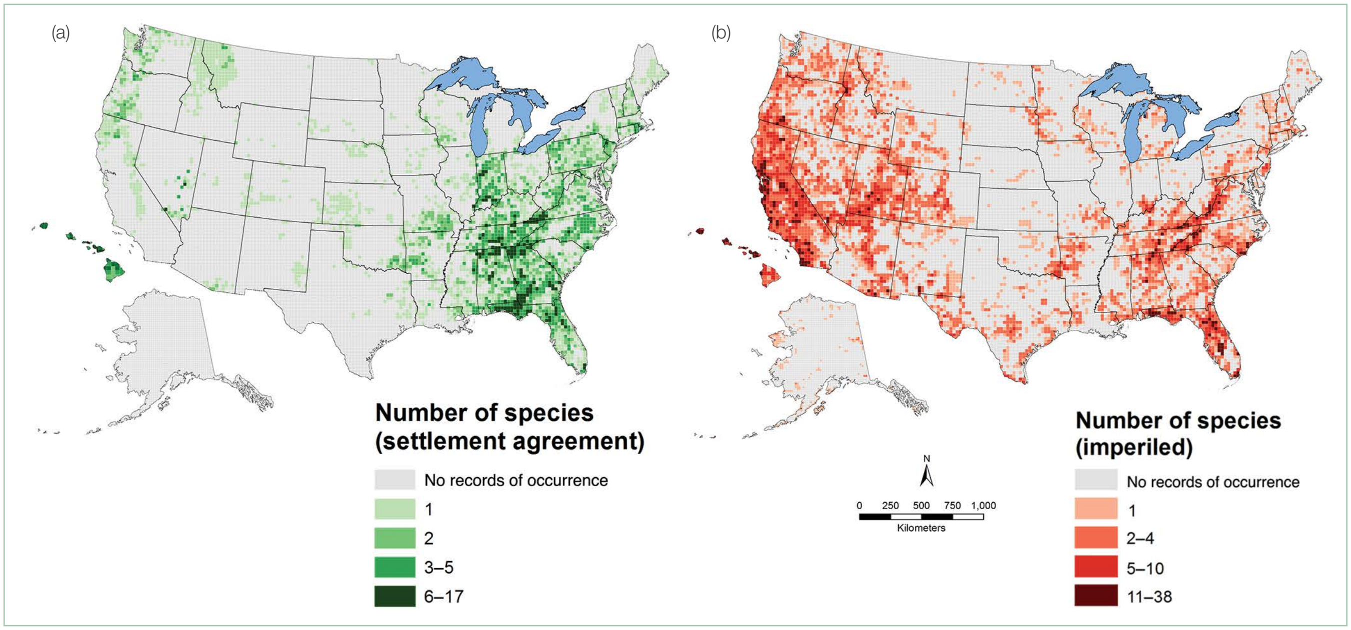


Figure 11-11. Geographic distribution of species (a) involved in the settlement agreement and (b) critically imperiled (G1 ranking) and imperiled (G2 ranking) that are not currently listed as threatened or endangered or being considered for listing under the settlement agreement. Data are derived from the National Heritage Programs as maintained by NatureServe (2014) and mapped onto a systematic equal-area grid (647.5 square kilometers [250 square miles]) of the United States. Alaska and Hawaii are displayed on a different scale for presentation purposes.



Implications

The geographic occurrence of listed and imperiled species across the United States is far from random, with available data showing that species of conservation concern concentrate in distinct regions of the country (figures 11-8 to 11-11). Although the conservation implications of these endangerment hotspots have been discussed in the past (see Dobson et al. 1997; Flather et al. 1998), conservation science has struggled with how to use this information to move beyond the species-by-species conservation strategies that have thus far dominated conservation efforts. One approach that has recently been formalized in the Forest Service Planning Rule that governs biodiversity conservation on lands it administers is based on the notion of coarse-filter conservation strategies (USDA Forest Service 2012c).

In general, coarse filters reflect higher level processes and patterns and are based on attributes that can be measured easily and inexpensively, relying on existing inventories (e.g., remotely sensed imagery, digital elevation models, weather station data) and describing broad characteristics of the environment. As such, coarse filters are but another kind of surrogate that attempts to identify the environmental cues —the amounts and spatial distribution of biophysical factors—that allow for inference to the state of most species inhabiting a particular ecosystem.

Although the coarse-filter methodology is intuitively appealing, the approach has been tested only rarely (Groves 2003) and with equivocal conclusions. The performance of coarse filters has been found to vary among the types of species composing the assemblage—working well for small-bodied, abundant species but failing to capture the habitat needs for large-bodied or rare species (Noon et al. 2003). This latter group often comprises the set of species that are of conservation concern, including those formally listed as threatened or endangered.

The failure of coarse filters to reliably predict the presence and persistence of rare species does not invalidate its use as a strategy to conserve at-risk species. Coarse-filter conservation strategies may be best thought of as conserving important biophysical templates that drive the overall pattern of biological diversity or concentrations of rare species (Anderson and Ferree 2010; Beier and Brost 2010) across broad landscapes. Thus, the occurrence pattern of threatened and endangered species (figure 11-9) or at-risk species in general (figure 11-11), coupled with information on their taxonomy and life histories, the portfolio of threats that triggered their listing or contributed to their rarity, and key biophysical attributes (e.g., climate, geophysical setting, vegetation, and land use) associated with their occurrence, could help define key ecosystem units that could be the focus of conservation efforts. Evidence in the literature suggests that such a strategy may have merit. The approach has been used to define "environmental domains" within broad geographic regions that represent key drivers of biodiversity and may perform particularly well defining domains that cover the occurrence of rare species (Trakhtenbrot and Kadmon 2005, 2006).

Although the idea of using data on species occurrence and the biophysical characteristics of the areas where they occur to define the geographic bounds for ecosystem planning is relatively new, aspects of this approach are starting to be implemented. The FWS has initiated an ecosystem-based approach for 48 species on the island of Kauai in the Hawaiian archipelago. These 48 species have been used to define species groups based on broad climate and physiography, shared threats, and shared critical habitat under the premise that focusing management and restoration efforts within these broad ecosystem types will be more effective at recovering species than would a species-specific strategy (USDI FWS 2010). Such thinking is also behind a push to consider species conservation across large multiple-use, multiowner landscapes under what has been called "collaborative adaptive management" (Scarlett et al. 2013). Moreover, the Forest Service has recently used the conceptual underpinnings of the coarse-filter approach to motivate landscape-scale conservation as reflected in the agency's efforts to restore the longleaf pine ecosystem and its associated species of conservation concern across the Southeastern United States (Evans et al. 2016).

Coarse filters and landscape-level conservation appear to hold much promise for the management of at-risk species (e.g., Ricketts et al. 2005; Wyman 2010). We need to acknowledge, however, that they currently represent conservation hypotheses for how species conservation may be accomplished more efficiently. Conservation practitioners and researchers alike are in need of a broader set of case studies from which to document and judge what can be gained from management with a more systems-oriented focus on the conservation of at-risk biodiversity.

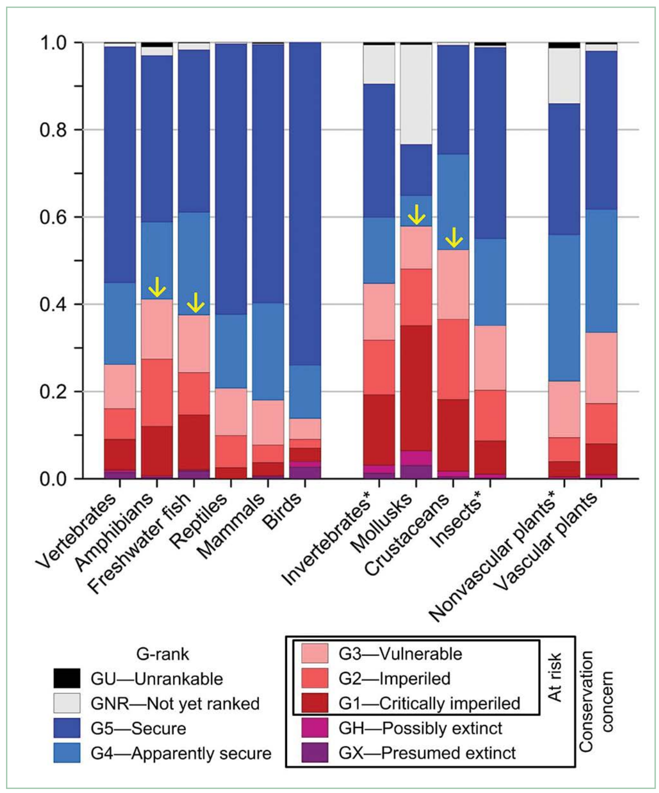
At-Risk Aquatic Species and Drinking-Water Protection

- ❖ Two-thirds of watersheds that support a high proportion of at-risk aquatic biodiversity have a collateral stake in drinking-water protection.
  - These watersheds are concentrated in the Southeast and the Mediterranean climates of California.
- ❖ Joint-benefit watersheds that also have low levels of land protection and high rates of urbanization can serve as targets for land use and conservation planning.
  - Explicit identification of watersheds with joint benefits has the potential to leverage scarce funding resources and facilitate action among traditionally contentious stakeholders.

One main finding from the 2010 RPA (Loftus and Flather 2012) and of this RPA Update is the disproportionately high number of species considered to be of conservation concern among those that are primarily associated with aquatic ecosystems. Among aquatic animal groups (amphibians, freshwater fish, mollusks, and crustaceans), the proportion of species of conservation concern always exceeds the average observed across all vertebrates and invertebrates (figure 11-12). This information, coupled with the occurrence pattern of these species, has long been used to highlight, in a geographically explicit manner, areas that should receive conservation planning focus (Flather et al. 2009; Scott et al. 1987).

Concurrent with efforts to conserve lands for biodiversity protection are renewed efforts to conserve lands for drinking-water protection (Wickham et al. 2011). These efforts are motivated by amendments to the Safe Drinking Water Act that shifts focus from monitoring to detect contaminants to protecting source water (U.S. EPA 1997). Source water protection recognizes that the quality of untreated water entering a drinking-water treatment plant could be translated into (1) risks associated with waterborne pathogens, (2) pollutant levels initially entering the

Figure 11-12. The proportion of species occurring in the United States assigned to each NatureServe conservation status rank. Asterisks (*) indicate those taxonomic groups with uncertain proportions (many species are awaiting conservation assessments). Yellow arrows indicate the proportion of species of conservation concern for species groups that are primarily associated with aquatic ecosystems. Data are derived from the National Heritage Programs as maintained by NatureServe (2014).



drinking-water supply, and, ultimately, (3) the costs associated with water treatment. As a consequence, land conservation and restoration are seen as potential mechanisms to capture the benefits associated with the ecosystem services linked to the provision of clean water (Postel and Thompson 2005).

With this renewed focus on maintaining the ecological condition associated with the uplands of source-water watersheds, land and resource management is now an overlapping concern for both biodiversity conservation and drinking-water protection. This overlapping concern sets up an opportunity in which management costs can be shared while realizing important joint benefits. Wickham and Flather (2013) undertook an analysis to determine if opportunities exist to align biodiversity and drinking-water protections goals and, if such opportunities exist, to identify where they occur geographically.

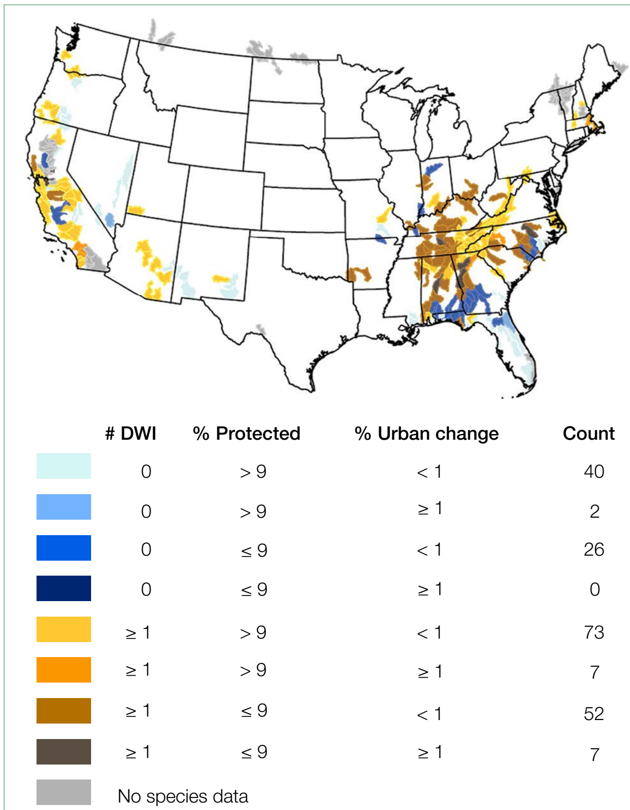
Four datasets (aquatic species occurrence, drinking-water intake locations, protected lands, and urban land cover) were used to analyze the coincidence of biodiversity and drinking-water services within eight-digit hydrologic units (watersheds). Aquatic species distributions by watershed were compiled from NatureServe's Central Databases (NatureServe 2010), data that were also used in support of the 2010 RPA (see Loftus and Flather 2012). Because watersheds capable of supporting more species also would be expected to support a greater number of rare species, we quantified the proportion of species that were potentially vulnerable to extinction (G1-G3, see figure 11-12) as our measure of at-risk aquatic biodiversity in each watershed. Drinking-water intake (surface water only) locations were provided by the U.S. Environmental Protection Agency Office of Ground Water and Drinking Water (see Wickham et al. 2011). Protected land area within watersheds was derived from the Protected Areas Database (ver. 1.2; http://gapanalysis.usgs.gov/padus/data/download). Finally, urban land cover area and urban land cover change within each watershed were derived from the National Land Cover Change Data (ca. 2001 to 2006; Fry et al. 2011).

We focused our analysis on those watersheds that were in the top 10 percent (≥ 90th percentile), as measured by the proportion of aquatic species that are classified as at risk. We refer to these watersheds as hotspots of at-risk aquatic biodiversity. We then determined which of those hotspot watersheds had (1) one or more drinking-water intakes, (2) limited protected lands (proportion of protected land in the watershed was less than the national median [< 9 percent]), and (3) relatively high urban development (≥ 1-percent gain in urban land during a 5-year period).

Results

Two-thirds (139 of 207) of the watersheds that support a relatively high proportion of at-risk aquatic biodiversity have a collateral stake in drinking-water protection, as indicated by the presence of at least one drinking-water intake. These watersheds are concentrated in the Southeastern United States and the Mediterranean climates of California (figure 11-13). We further partitioned this subset of hotspot watersheds into those with limited amounts of protected land to highlight those areas that may benefit from land acquisition or conservation easement. Of the 139 watersheds, 59 (40 percent) have less than the U.S. median level of protected lands. We further ranked watersheds according to level of threat as reflected by the degree to which land was undergoing relatively rapid urbanization. This ranking resulted in a final set of seven watersheds that have shared aquatic biodiversity/drinking-water values, relatively low proportions of protected lands, and a relatively high rate of urbanization. Most of these watersheds (5 out of 7) occur in the Southeast, with a secondary concentration in California.

Figure 11-13. Watersheds that support a relatively high proportion of at-risk aquatic biodiversity (in the 90th percentile) categorized by whether drinking-water intakes (DWI) are present, whether the percentage of protected areas is limited, and whether relatively high urban development exists. # DWI is the number of drinking-water intakes. Count is the number of watersheds in each set.



Source: Wickham and Flather (2013).

Implications

This analysis identifies areas where land conservation actions are likely to support drinking-water security and aquatic species protection by applying a simple dichotomy (with or without drinking-water intakes) to those watersheds that support a high proportion of at-risk species (i.e., biodiversity hotspots). More than 65 percent of hotspot watersheds also have drinking-water intakes, and their distribution across the conterminous United States points primarily to the Southeast and, secondarily, to portions of California as areas with a shared (biodiversity and drinking-water) interest in watershed-level land management.

One potential benefit of integrating concerns for drinking-water quality with biodiversity conservation in land management planning is the increased likelihood of stakeholder buy-in. Often efforts to conserve biodiversity are seen to conflict with other societal needs because these effort are often associated with restricting land uses that can reduce the provision of other goods and services. Defining joint benefits explicitly among traditionally contentious stakeholders has the potential to leverage scarce funding resources and facilitate action that results in win-win outcomes. The focus on drinking-water intakes and aquatic biodiversity was intentionally simplistic to illustrate how land management and conservation planners could identify watersheds that benefit both human well-being and biodiversity conservation. It is certainly plausible, and perhaps expected (see Naidoo and Ricketts 2006), that additional shared benefits could be realized under a more comprehensive treatment of the full suite of services derived from ecosystem.

Future Work

If resource planners are to make informed decisions about which lands, species, or habitats should be prioritized for biodiversity conservation, we will need to further our capability to describe and model biodiversity response to human land and resource use. Future work to support the 2020 RPA Assessment will focus on furthering this capability in three areas.

First, housing growth in and near protected areas is shown in this RPA Update to affect bird communities both at the boundary of protected areas and inside those areas. An unanswered question is how future housing growth trends, along with future land use activities, will affect bird communities within and near protected lands. The focus of this effort will be on anticipating where housing growth and land use intensification are likely to have the greatest negative impact on native bird communities and those bird species that are of greatest conservation concern. By anticipating these hotspots of bird community degradation, we hope to identify areas where focusing land use and resource policy and management could buffer biodiversity loss and maintain the conservation value of those protected areas.

Second, we will extend the analysis of climate-induced stress to terrestrial wildlife habitat across the RPA Rocky Mountain Region to the entire conterminous United States. Such information will contribute to spatially explicit identification of where wildlife resources may be particularly vulnerable, or particularly resistant, to habitat stresses attributable to climate change. Such analyses will inform efforts to strategically acquire or manage lands that will help facilitate wildlife adaptation to climate change and will be particularly relevant to State wildlife agency efforts to update their State Wildlife Action Plans to better address the impacts of climate change on wildlife resources.

Finally, the depiction of at-risk biodiversity on a systematic grid reported in this RPA Update leads logically to efforts directed at understanding and anticipating where species rarity and extinction risk are expected to increase in the future. This work will focus on testing whether knowledge of how climate regime, land use intensification, land ownership, and human population change in the future can be used to anticipate where concentrations of at-risk species are likely to emerge. In addition to being rare, these species are often secretive and difficult to detect. Therefore, analyses such as those proposed here could help direct future surveys by identifying areas that "look like" they should support more at-risk species than current inventories indicate are supported. These predictions could help geographically target inventory investments to those areas that are most likely to harbor undetected at-risk biota.

All three of these efforts are geared toward expanding our capacity to model biodiversity response to changes in the environment in a spatially explicit manner. Armed with such decision-support tools, the RPA Assessment should be better positioned to inform agency planning efforts as defined in the 2012 Planning Rule (USDA Forest Service 2012c) and to address the requirement to provide those conditions that will maintain the integrity of ecological systems.

Conclusions

The status and trends of wildlife, fish, and biodiversity reported in this RPA Update do not substantively alter the conclusions of the 2010 RPA. The RPA Update does provide a refinement of wildlife, fish, and biodiversity response to human uses and management of natural resources. This refinement takes two forms: one is a simple improvement in the spatial resolution of resource response; the other involves an elaboration of the questions that were initially posed in the 2010 RPA.

Increasing spatial resolution of resource response enabled us to look more closely at the geography of at-risk species occurrence and the patterns of terrestrial wildlife habitat stress attributed to climate change. Species currently listed under the

ESA are concentrated in hotspots that occur in the southern Appalachians, peninsular Florida, coastal areas of the southeast Atlantic Ocean and eastern Gulf of Mexico, California Mediterranean climate regions, and the Hawaiian archipelago. These hotspots have long been known to house high numbers of listed species. These data, coupled with information on the location of species that will likely be added to the list in the future, reveal the emergence of new areas of concentration that include the interior highlands of Arkansas, Illinois, Indiana, Kentucky, and Missouri and the southern portions of the Great Basin.

Our assessment of climate-induced stress to terrestrial wildlife habitat also benefited from a finer analysis resolution. The 2010 RPA found that areas where habitat was thought to be particularly vulnerable to climate change were associated with the RPA Rocky Mountain Region, a region where the Forest Service and other public land management agencies have important land stewardship responsibilities. Furthermore, our focus on this region was motivated by fire—a disturbance process that is expected to increase under climate change—and its importance in shaping the ecological systems found in this region. In addition to providing a much refined picture of the TCSI that could inform agency forest plan development, we also learned that the geographic distribution of highly stressed habitats was very much affected by how wildfire was managed across this landscape.

The 2010 RPA alluded to the potential impacts associated with increasing housing development in the high-amenity areas that are associated with public lands like national forests and grasslands. A number of impacts to biodiversity that occur in these protected areas were explored through a series of case studies in the 2010 RPA, but we lacked a systematic evaluation of impacts across the country. For this RPA Update, we completed a study that was designed specifically to explore the nature of housing development impacts on bird communities both at the boundary of protected areas and within those areas. Our analysis demonstrated that not only does housing near protected lands affect bird communities locally (i.e., at the boundary), but it also affects bird communities away from the boundary within protected lands. This latter finding provides evidence that housing impacts extend well beyond their local footprint. Thus, to maintain protected areas as refugia for biodiversity, prioritizing conservation actions on private lands is necessary. The focus on private lands suggests that the agency's State and Private Forestry program should be a major participant in efforts to address these concerns. In locations where private-land housing is dense (e.g., Appalachians), land use planning and homeowner education are important. In locations where private-land housing is low, the conservation options are more diverse. Alternative strategies for preserving land near protected areas (e.g., conservation easements, cluster housing) could be pursued also with the intent to maximize unfragmented natural land cover while minimizing the footprint of housing development.

We also examined in more detail the relationship between drinking water and at-risk aquatic biodiversity. Our analysis identifying geographic coincidence between watersheds with drinking-water intakes and areas supporting high concentrations of at-risk aquatic biodiversity suggests that both biodiversity and ecosystem services can benefit simultaneously from land conservation efforts. Such opportunities point to important shared conservation costs that can be leveraged among stakeholder groups that will benefit jointly. We suspect that such shared benefits among stakeholder groups that are often viewed as confrontational are more common than past experience would suggest.