
11 Climate Change and Wildlife in the Southern United States

Potential Effects and Management Options

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In the southeastern United States, climate models project a temperature increase of 2–10°C by 2100 (Intergovernmental Panel on Climate Change 2007). Climate change is already evident. Since the 1970s, average temperature has risen by about 1°C, with the greatest seasonal temperature increase during winter. Average precipitation during autumn has increased by 30% since 1901, but summer precipitation has decreased (U.S. Global Change Research Program 2012). Correspondingly, drought has affected a larger portion of the Southeast over the past three decades. The patterns and

severity of storms are also changing, with more heavy downpours in many areas and the power of Atlantic hurricanes increasing (U.S. Global Change Research Program 2012).

Climate change is driven in part by activities associated with human population and economic growth that increase carbon dioxide (CO₂) emissions (Wear and Greis in press). Effects of climate change on wildlife are confounded by land-use changes associated with population and economic expansion that reduce and compromise the amount and continuity of habitat, and thus may limit their ability to respond. Strategic planning for wildlife conservation is hindered by uncertainty regarding levels of human population and economic growth, how and where climatic variables will change, and how wildlife species having widely differing life histories and habitat requirements are likely to respond. In this chapter, we explore potential impacts of climate change (2000–2060) on terrestrial vertebrates and butterflies within the five major subregions of the southeastern United States: the Mid-South, Coastal Plain, Mississippi Alluvial Valley, Piedmont, and Appalachian-Cumberland highlands (Wear and Greis in press), and discuss management options for mitigating impacts.

Studies of Pleistocene pollen and fossilized animals show correlations between glacial–interglacial fluctuations and latitudinal shifts in many species throughout much of the Southeast (Cooperative Holocene Mapping Project 1988; Hibbard et al. 1965; Root and Schneider 2002). During the transition from the last Ice Age to the present interglacial Holocene period, most plant species moved northward, but the rate of movement by individual species varied; this resulted in reassembly of forest communities different from those familiar to us today (Graham and Grimm 1990). In addition, habitat changes associated with warming and cooling periods likely forced the distribution of many animal species to shift in latitude or elevation, whereas others went extinct. Many species that are now extinct, including the American mastodon (*Mammuthus americanum*), Columbian mammoth (*Mammuthus columbi*), sabertooth cat (*Smilodon* spp), bison (*Bison antiquus*), giant ground sloth (*Eremotherium mirabile*), dire wolves (*Canis dirus*), and large salamanders lived in Florida and other southern states as recently as 10,000 years ago (Root and Schneider 2002; Webb 1974). Like plant communities, wildlife communities have reassembled as species responded differently to changing climate and associated changes in vegetation composition and structure (Graham and Grimm 1990).

As they did in the past, currently extant wildlife species will respond to predicted rapid climate change by: (1) going extinct, (2) exhibiting phenotypic or behavioral plasticity, or (3) undergoing evolutionary adaptive response (Austin et al. 2012; Holt 1990). Evolutionary adaptive response (natural selection) of a given species depends on rates of mutation, rates of gene flow, amount of genetic variation, level and consistency of selective pressure exerted on a particular trait, and generation time and age structure of the population (Austin et al. 2012). Because any of these factors can differ among populations and among locations, predicting and managing evolutionary change will be context-specific (Austin et al. 2012).

Today 1027 species of native terrestrial vertebrates occur in the Southeast, including 178 amphibians, 504 birds, 158 mammals, and 187 reptiles (Griep and Collins in press). The number of species and the species composition vary widely among subregions and ecosystems. Approximately 15% (152) of native terrestrial vertebrates are species of conservation concern, including 81 that are listed as endangered under the Endangered Species Act of 1973. Major factors contributing to population declines of these species include habitat destruction and fragmentation, isolation, small population size, low genetic diversity, diversion of waterways, introduction of nonnative invasive species, acid rain and other environmental pollutants, commercial development, human disturbance, and exploitation for trade. Climate change is an additional stress to wildlife that could be exacerbated by these factors (Griep and Collins in press).

Vulnerability of species to climate change depends on their exposure, sensitivity, and adaptive capacity (Glick et al. 2011; Intergovernmental Panel on Climate Change 2007). Species adapted to warmer or drier conditions could expand their current ranges. Conversely, the direct effects of altered temperature or rainfall may adversely affect animal species that are especially sensitive to air or water temperature, have specific moisture requirements, or rely on specific weather patterns

for survival or successful breeding. Species that are sensitive to climate will not likely be affected by climate change if they occur in little-affected or unaffected geographic areas (exposure), or if they have the capacity to respond behaviorally (such as local movement among microhabitats for thermoregulation) or physiologically (such as becoming dormant during dry periods) (Glick et al. 2011). Generalist species may be less sensitive and adapt more easily to changes in climate than specialists. Similarly, wildlife species that occur in multiple ecosystems may be more resilient, as climate-driven alterations will likely differ among ecosystems. Even among species that are sensitive to climate, not all would be affected negatively. For example, the Florida scrub-jay (*Aphelocoma coerulescens*) and sand skink (*Neoseps reynoldsi*)—federally listed species of conservation concern that primarily inhabit young, recently disturbed stands of the Florida scrub ecosystem—could benefit from more frequent fires potentially associated with climate change (Chapter 5).

Differences in impacts across subregions and ecosystems are expected because predicted changes in temperature and precipitation are unlikely to be the same across the Southeast (Chapter 2), and because some ecosystems, rare habitats, and individual species are more vulnerable to direct and indirect effects of climate change than others. For example, in the Coastal Plain, changing fire regimes (Chapter 5) may affect inland ecosystems, whereas a rise in sea level could reduce or alter mangrove (*Avicennia* spp), coastal live oak (*Q. virginiana*), beach and dune, salt marsh, and freshwater coastal wetland habitats that are important to many plants and animals. Salinization of ground and soil water from rising sea levels may affect forests and freshwater wetlands in the Coastal Plain, along with their associated wildlife (Devall and Parresol 1998). In the Southern Appalachian Mountains, high elevation spruce–fir (*Picea* spp–*Abies* spp) forests and associated wildlife may decline in response to higher temperatures. Wildlife associated with vulnerable or rare habitats such as high-elevation bogs, may also be more susceptible to changes in temperature and precipitation than species that occur in more extensive habitat types. The Interior Highlands of Arkansas, Oklahoma, and Missouri and the Southern Appalachians may serve as important refugia for plants and animals in a changing climate, as they did during the Pleistocene (Devall and Parresol 1998; Dowling 1956).

Higher temperatures and altered weather patterns are likely to have major direct and indirect effects on biological diversity; these effects may differ substantially among animal populations or communities (at a broad scale) and species whose life histories and physiologies differ. For example, the ability of birds to move long distances may make them less vulnerable to climate change than reptiles and amphibians that generally have limited mobility (Root and Schneider 2002) and a narrow tolerance limit for temperature (particularly for amphibians) or moisture. Similarly, small mammals with a narrow range of habitats and small home-range sizes may be more affected by microclimate than larger carnivores or herbivores (Hibbard et al. 1965). Species that respond by moving longer distances may be able to survive, but range shifts could result in reassembly of wildlife communities in some locations and local extirpation of species in others. Thus, simple counts of species (richness) within subregions or landscapes may not change substantially in response to climate change because some species may be replaced.

Extreme weather events such as hurricanes, tornados, or storms, and associated changes in seasonal extremes of temperature and precipitation, may also significantly affect wildlife species. For example, in 1989 Hurricane Hugo destroyed 80% of the nesting trees of the endangered red-cockaded woodpecker (*Picoides borealis*) in the Francis Marion National Forest, which had supported the largest known population of the woodpeckers (Watson et al. 1995). Increased frequency or intensity of hurricanes would likely have a major impact on the remnants of the longleaf pine (*Pinus palustris*) habitat that remain on the Coastal Plain. Prolonged droughts or altered hydroperiods (the length of time water is retained) in ephemeral wetlands could result in local extinctions of amphibian species that rely on these wetlands for breeding. Increased frequency or intensity of fires resulting from drought could alter habitat features required by some wildlife species, such as wiregrass (*Aristida stricta*) cover for Bachman's sparrows (*Peucaea aestivalis*) in the Coastal Plain or intact leaf litter required by ground nesting worm-eating warblers (*Helmitheros vermivorous*)

in the central hardwood forest of the Cumberland-Appalachian highlands (Greenberg et al. 2007). Conversely, species such as eastern wood-pewees (*Contopus virens*) could benefit from more frequent or more intense fires (Greenberg et al. 2007).

Indirect and “cascading” impacts to wildlife are likely to be much greater than direct effects, but predictions are equally uncertain (Bagne et al. 2011). For example, shifts in vegetation and habitat structure caused by gradual climate change or altered natural disturbances (frequency, duration, or intensity of drought, wind, fire, or flooding) would likely have dramatic effects on many animal species. Changing amount, pattern, and composition of forests and other land uses both drive, and are driven by, climate change, and have a major impact on the ability of animals to disperse in response to climate change. Other indirect impacts could be changes in the amount or timing of food availability (caused by changes in synchrony between flowers and pollinators) and shifts in complex competitive or predator–prey interactions. Differences in the rate and direction of range shifts among species with symbiotic relationships, such as pollinators and host plants, can lead to extinctions (Schowalter et al. 1986).

The degree of exposure, sensitivity, and capacity to adapt to climate change are important in assessing vulnerability of species to climate change. Several methods for wildlife vulnerability assessment have been developed, such as the System for Assessing Vulnerability of Species, which uses a questionnaire based on habitat, physiology, phenology (timing of life-cycle events), and biotic interactions to develop vulnerability scores in the Western United States (Bagne et al. 2011). Criteria used to predict species vulnerability to climate change generally fall into four broad categories: habitat, physiology, phenology, and biotic interactions (Bagne et al. 2011; Glick et al. 2011). Another vulnerability assessment method, the NatureServe Climate Change Vulnerability Index (NatureServe 2012) uses available information about natural history, distribution, and landscape circumstances to help identify species that may be vulnerable to climate change.

Despite these criteria for identifying vulnerable species, vulnerability assessment for any given species is fraught with uncertainty. First, great uncertainty surrounds the accuracy of predicting changes in temperature, precipitation, or natural disturbance regimes at local levels (Vose et al. in press). Second, information on the basic natural history and requirements of many vertebrate species is incomplete. Thus, the accuracy of vulnerability assessments is limited by incomplete knowledge of (1) the level of exposure given the number of and the uncertainty surrounding climate change projections; (2) the basic natural history, physiology, and range of tolerance; (3) the behavioral or genetic adaptive capacity of most species; and (4) existing or potential competitive or predator–prey interactions. Generally, vulnerability to climate change may be increased for a species that is:

- At the edge of its range
- Restricted to high elevations
- Restricted to a specific vegetation association that could be altered by changes in temperature, precipitation, or natural disturbances
- A habitat specialist
- Dependent on another species to create specific habitats, such as burrows dug by gopher tortoise
- Narrow in its range of abiotic or biotic tolerance
- Low in genetic variation and ability to adapt evolutionarily
- Restricted in behavioral or phenotypic plasticity
- Wetland dependent, especially if dependent on a specific hydroperiod, water temperature, and water quality parameters
- Dependent on specific weather-related cues or “triggers” for breeding
- Unable to colonize rapidly due to low capacity for movement or dispersal
- Located in a habitat that is fragmented, an “island,” or surrounded by inhospitable habitat
- Restricted to a specific temperature range for successful embryonic development, overwintering survival, or maintenance of its energy budget

- Dependent on specific timing of host plants or insects for important life history requirements
- Of conservation concern by virtue of low population levels, the threat of disease, or a shrinking habitat base

In this chapter, we combine literature synthesis, expert opinion, and data-based and expert-based case studies to address the vulnerability of species and possible responses to climate change in the southeastern United States using some or all of the four climate scenarios developed in the Southern Forest Futures Project (SFFP, Chapter 2; McNulty et al. in press). We focus on representative species or suites of species that have similar ecological requirements and thus may respond similarly to changes in climate or climate-driven changes to habitats. We include but do not focus solely on species of conservation concern, recognizing that some species of conservation concern may not be severely impacted, whereas other species that are not currently of conservation concern may be greatly impacted by climate change (Byers and Norris 2011). We organize our discussion broadly into taxonomic categories of vertebrates, including birds, amphibians, reptiles, and mammals. We also include a discussion on butterflies because of the important roles they serve in ecosystems as pollinators, prey, and herbivores. In addition, their visibility makes them easily monitored, early indicators of ecosystem health. Finally, we explore management options for wildlife conservation and mitigation management for a changing climate (Byers and Norris 2011).

Our case studies were selected to illustrate some of the many ways that climate change could affect wildlife in the southeastern United States through shifts in range, reproductive output, and critical habitat parameters (such as wetland hydroperiod); and by causing mismatched phenologies that disrupt biotic processes (such as the timing of anuran egg deposition and toxic leaf drop, or the timing of pollinators and flowering of host plants).

Subjects for case studies were in part selected to illustrate potential responses by broader groups of species that are likely to respond similarly to the same suite of environmental drivers, or to illustrate the very different ways that climate change might affect southeastern wildlife. For example, a case study of potential changes to hydroperiod in isolated, ephemeral ponds of the Coastal Plain that result from changes in amounts and seasonal patterns of precipitation, has implications for many pond-breeding amphibians in the Coastal Plain and other areas. Similarly, a case study on potential response to climate change by black-throated blue warblers (*Setophaga caerulescens*) in the Appalachian-Cumberland highlands illustrates potential effects on a suite of high-elevation bird species having similar requirements, such as the golden-crowned kinglet (*Regulus satrapa*), black-throated green warbler (*Setophaga virens*), and Canada warbler (*Wilsonia canadensis*). A case study that predicts Carolina northern flying squirrel (*Glaucomys sabrinus coloratus*) response to climate change scenarios illustrates the possible response of other species to potential reductions or elimination of high elevation, northern hardwood–spruce–fir forest.

MAMMALS

Because of their ability to regulate their body temperatures, mammals generally respond to climate indirectly through interactions with their food supply, predators, parasites, and habitat associations (Berteaux and Stenseth 2006). Most mammals are adapted to a particular vegetation community, the distribution of which is roughly determined by temperature and precipitation (Chapter 10). Significant changes in temperature and precipitation could alter habitats or reduce food resources and result in extirpation of some mammal species from their current range. For example, precipitation in areas that currently maintain forest could become highly variable or reduced to levels that support more drought-tolerant ecosystems such as shrublands or grasslands, resulting in a gradual shift away from forest- and woodland-associated mammals. Historically, North American forest–grassland ecotones (the boundary between different ecotypes) have retreated in a northeasterly direction and are expected to do so again if climate change continues its current trajectory (Frelich and Reich 2010).

Based on the processes of extinction and colonization and their ability to disperse, species are expected to shift their distributions or move to higher altitudes in response to a warming climate (Walther et al. 2002). Species that are isolated on small islands or island-type habitats with limited dispersal abilities, such as Carolina flying squirrels, and those in small, highly vulnerable habitats will be the most at risk. Although mammals are generally very mobile, human-modified landscapes may affect their movements to more favorable climates. Agriculture, urbanization, major highways, or other inhospitable environments may impede movements of terrestrial forest mammals to more favorable locations (Francl et al. 2010). Alternatively, human activity and modified landscapes could also contribute to faster colonization of new areas. For example, nine-banded armadillos (*Dasypus novemcinctus*) have expanded their range across the southern United States by approximately 7.8 km/year over the last 115 years as a result of both natural colonization and human-facilitated movements, such as using bridges to cross rivers and hitchhiking (Taulman and Robbins 1996). Responses by individual species to climate change may disrupt their interactions with other species (Walther et al. 2002), resulting in unforeseen consequences such as die-offs, disease spread, overpopulation, and becoming invasive when moving into a new area or when resident species vacate an area.

Studies on the effects of climate change on mammals are relatively rare (Barteaux and Stenseth 2006); thus, the potential effects of climate change on many southeastern mammals are unclear. However, studies on historical climate warming, such as the end of the Pleistocene, suggest that climate change can cause extinctions of mammals, changes in communities, and reductions in richness or diversity (Barnosky et al. 2003; Blois et al. 2010). Studies in mountainous areas suggest that mammals associated with low elevations may expand their ranges, whereas the ranges of high-elevation species may contract (Moritz et al. 2008).

Bats play an important role in forest ecosystems and suppress populations of night-flying insects, including forest and agricultural pests (Boyles et al. 2011; Kalka et al. 2008; Williams-Guillén et al. 2008). Eighteen species of bats inhabit the forested portions of the southeastern United States (Trani et al. 2007), and many of these species are federally endangered or considered species of concern. Bat species vary considerably in their habitat associations, roosting and foraging habits, and strategies for coping with harsh winter conditions such as cold temperatures and reduced food supply. Thus, responses to climate change will likely differ among species. Several characteristics of bats suggest that they may be quite sensitive to predicted changes in temperature and precipitation. Because reproduction, growth, and hibernation of temperate-zone bats are highly dependent on temperature, they are one group of mammals that may respond directly to changes in temperature regimes (Jones et al. 2009). Warm spring and summer temperatures can have a positive effect on reproductive success, resulting in a higher proportion of females that give birth, earlier fledging dates, and higher growth rates of young (Burles et al. 2009; Hoying and Kunz 1998). However, excessively high temperatures in roosts may result in increased energy expenditures, particularly when coupled with low humidity (Licht and Leitner 1967).

Climate change may result in changes in the distribution of bats during both summer and winter. For example, bat distributions in Europe are forecasted to change considerably over the next century based on various climate scenarios, with species in the Boreal Zone experiencing the greatest change and risk of extinction (Rebello et al. 2010). One species may have already expanded its range within the United Kingdom (Lundy et al. 2010). In the Eastern United States, preferred hibernation temperatures of little brown bats (*Myotis lucifugus*) suggest that their winter distribution may show a pronounced northward movement (Humphries et al. 2002), whereas niche models suggest that both the little brown bat and the northern long-eared bat (*Myotis septentrionalis*) may expand farther southward in response to global climate change (Kalcounis-Ruepell et al. 2010). Brazilian free-tailed bats (*Tadarida brasiliensis*) in the Eastern United States have been expanding their range over the past several decades, which may be in response to a warming climate (Lee and Marsh 1978). Anecdotal evidence suggests that other species such as the Seminole bat (*Lasiurus seminolus*) may also be expanding their range northward (Bradley 2010; Wilhite et al. 1998). Climate

change may also alter the timing of spring and autumn migrations of migratory bats (Newson et al. 2009). Autumn migratory activity is strongly affected by weather (Arnett et al. 2008; Baerwald and Barclay 2011); thus, extreme weather events could impact migratory patterns. Further, if changes in timing of insect emergence or activity are not synchronized with movement to winter and summer ranges, bats may arrive at summer habitats before food supplies are sufficient, a phenomenon that has been documented for birds (Inouye et al. 2000). Aside from studies by Cryan and Brown (2007), Perry et al. (2010), and Walters et al. (2006), little is known about the timing of bat migration in the southeastern United States or elsewhere, which complicates efforts to document changes in migratory timing that may result from climate change.

Small terrestrial mammals, including rodents and insectivores (shrews), typically comprise the largest group of mammals in most southeastern ecosystems (Trani et al. 2007). Of these, rodents are expected to experience most of the changes in mammal distributions resulting from climate change because they represent the most abundant group of mammal species found in many ecosystems. For example, a projected doubling in carbon dioxide (CO₂) may result in the loss of eight mammal species and a gain of 29 mammal species in the Great Smoky Mountains National Park, with the majority of this turnover being rodents (Burns et al. 2003). Small mammals play an important ecological role in forests. They are the primary prey for many raptors, snakes, and furbearers. They are also dispersers of mycorrhizal fungi spores (Johnson 1996; Trappe and Maser 1977) and seeds of important trees, such as oaks, hickories (*Carya* spp), and pines (Pank 1974; Smith and Aldous 1947; Steele et al. 1993; Vander Wall 1990). Predation of tree seeds by small mammals can affect forest regeneration and the distribution of forest ecosystems (Goheen and Swihart 2003). Underground tunneling by fossorial species may affect hydrological processes on forested watersheds (Ursic and Esher 1988), change soil properties (Huntly and Inouye 1988), and influence the composition of vegetation communities (Hobbs and Mooney 1985). Small mammals also consume the larvae and pupae of forest insect pests, which may reduce the severity of insect outbreaks (Hanski 1987). Consequently, changes in abundance of small mammals have important implications for forest ecosystems in the Southeast.

Rises in sea level from melting polar and glacial ice combined with increased intensity of hurricanes or a prolonged hurricane season associated with a warming climate (Haarsma et al. 1993) could negatively affect southeastern coastal ecosystems that support unique mammals (Michener et al. 1997). For example, along the northern Gulf of Mexico, five unique subspecies of beach mice (*Peromyscus polionotus*), four of which are federally endangered (Choctawhatchee, Alabama, St. Andrew, and Perdido Key), occur along a narrow strip of white sand dunes on the Alabama and Florida Gulf Coast (U.S. Department of the Interior, Fish and Wildlife Service 1987, 2010). Rising sea levels and beach erosion may lead to loss of habitat for these species, which currently occupy only about 80 km of coastal dunes (U.S. Department of the Interior, Fish and Wildlife Service 1987, 2010).

Past hurricanes have caused substantial mortality to a number of mammals across coastal areas of the Southeast. Muskrat (*Ondatra zibethicus*), raccoon (*Procyon lotor*), rabbit (*Sylvilagus* spp), and white-tailed deer (*Odocoileus virginianus*) were reduced by 60% in coastal Louisiana following Hurricane Audrey (Ensminger and Nichols 1958). Hurricane Hugo eliminated approximately 50% of the white-tailed deer, 65% of the squirrels, and most of the rabbits on Bulls Island in South Carolina (Cely 1991). Further, populations of island-dwelling species may be especially susceptible to reductions in habitat caused by rising sea levels. For example, a number of unique animals occur only in the Florida Keys where the maximum elevation is no more than about 6 m above sea level. These include federally endangered mammals such as the Key deer (*Odocoileus virginianus clavium*), Key Largo cotton mouse (*Peromyscus gossypinus allapaticola*), silver rice rat (*Oryzomys palustris natator*), Key Largo woodrat (*Neotoma floridana smalli*), and the Lower Keys marsh rabbit (*Sylvilagus palustris hefneri*). Models of climate-change induced rises in sea level for Big Key Island, Florida, indicate that a rise of 18 cm would inundate 34% of the island by the twenty-second century under a best-case scenario, whereas a rise of 1.4 m in sea level would inundate 96% of the island under a worst-case scenario (The Nature Conservancy 2011).

Carnivores, which are important top predators in many ecosystems, may be threatened by climate change as well. For example, rise in sea level could have significant negative impacts on large carnivores in peninsular Florida. Black bear (*Ursus americanus*) habitat would be reduced 9% by a 1-m sea level rise and 31% by a 5-m rise; and the endangered Florida panther (*Puma concolor coryi*) habitat would be reduced 29% by a 1-m rise and 90% by a 5-m rise (Whittle et al. 2008). Consequently, these and other species may need suitable habitats inland and farther north, along with forested corridors linking current and potential future ranges.

Changes in precipitation may present some of the most profound effects on mammals and ecosystems of the Southeast. Aside from the potential changes in vegetation that may result from long-term changes in precipitation (Chapter 12), increased variability or reductions in precipitation may affect disease outbreaks and mammal survival, physiology, and nutritional state. Decreased precipitation may directly affect bats. Because of their naked wings and exceptionally large lungs, bats have high rates of evaporative water loss (Neuweiler 2000); and insectivorous bats must rely on accessible water to maintain water balance (Kurta et al. 1989, 1990). For example, lactating fringed myotis (*Myotis thysanodes*) in Colorado drink 13 times as often as nonreproductive females (Adams and Hayes 2008) and reproduction in females of six bat species is adversely affected by unusually low precipitation in the Western United States (Adams 2010).

Changes in temperature or precipitation could affect food resources required by many groups of mammals. All bats of southeastern forests are insectivorous; therefore, climate change may indirectly affect bats by changing availability of their arthropod prey. Increased drought will likely influence abundance and diversity of prey for species, such as the little brown bat, that rely on emergent aquatic invertebrates (Frick et al. 2010; Rodenhouse et al. 2009). Drought may also affect bats that rely on insects associated with agriculture and forests as witnessed by daytime feeding of Brazilian free-tailed bats during the 2011 Texas drought (Mylea Bayless, personal communication, 2011, Conservation Programs Manager, Bat Conservation International, PO Box 162603, Austin, TX 78716). Although some bat species specialize on particular orders or families of insects (Agosta 2002; Lacki and Dodd 2011; Whitaker 2004), most show high spatial and temporal variation in food habits (Lee and McCracken 2005; Murray and Kurta 2002; Moosman et al. 2007). Thus, it is not clear how climate-induced changes in phenology or shifts in the distribution of insect prey would affect bats.

Changes in climate could lead to increased outbreaks of animal-transmitted diseases in humans (Harvell et al. 2002). Rodents carry or serve as reservoir hosts for many tick- and flea-borne diseases that infect humans, including Lyme disease, Rocky Mountain spotted tick fever, Tularemia, plague, and Ehrlichiosis (Gubler et al. 2001). Rodent-borne diseases frequently depend on rodent population levels, and disease outbreaks are often associated with small mammal population increases (Kuenzi et al. 2007; Mills et al. 1999), which in turn often depend on environmental conditions and food availability (Gubler et al. 2001). Climate variability, such as long wet periods followed by long dry periods, may create boom and bust cycles in rodent populations. For example, Hantavirus pulmonary syndrome is transmitted by rodents and often fatal to humans (Childs et al. 1994). Increased rainfall, which led to increased food abundance for rodents in the Four Corners Region of the Western United States, created a deadly Hantavirus outbreak in 1993 (Engelthaler et al. 1999). Hantavirus outbreaks in Europe are also associated with increased rodent populations that result from elevated average temperatures and increased rodent food (Klempa 2009; Piechotowski et al. 2008). Long periods of drought may also reduce rodent predators, such as hawks, owls, and snakes; thus, when rains follow, rodent populations may rebound faster than their predators can reproduce (Epstein 2001). Thus, increased variability in precipitation could produce similar boom and bust cycles in small mammal populations in the Southeast leading to more disease outbreaks.

It is clear that climate change has the potential to greatly affect mammal communities throughout the southeastern United States but predicting the magnitude and direction of those changes is difficult due to lack of targeted research in this area. Various modeling approaches are available that will allow managers to predict or forecast mammalian responses to climate change such as

ecological niche models (e.g., Loehle 2012; Phillips et al. 2006; Wiens et al. 2009). The outcomes of these models can then be used to identify species that may be most at risk of extinction or are likely to be of management concern (e.g., become invasive).

At the end of this chapter, we present three examples that illustrate the potential effects of climate change on mammals in the southeastern United States. The first case study is for a relatively common and wide-ranging species, the eastern woodrat (*Neotoma floridana*); we examined the possible effects of climate change on the distribution of this species in the Cross Timbers section of the Mid-South, which is the western edge of its range. The second case study is on the Carolina northern flying squirrel, an endangered subspecies that depends on high elevation forests; we examined effects of climate change on persistence of the high-elevation habitats in the Blue Ridge section of the Appalachian-Cumberland highlands. In the third case study, we forecast changes in the summer maternity distribution of the Indiana bat (*Myotis sodalis*) across its entire range in response to four climate scenarios, and we estimate the relative importance of southeastern forests in the conservation and recovery of this endangered species under the four scenarios. The results of these models suggest that monitoring changes in mammal distributions, demography, and behavior in response to climate is critical for effective management.

BIRDS

Birds are especially appropriate for studying potential responses to climate change for two reasons: (1) long-term data sets on their abundance and distribution are available, and (2) their ranges are strongly associated with temperature (Root 1988b). Favorable weather conditions typically help to enhance avian reproductive success (Jarvinen 1982). For species that breed near their ecological or physiological limits, such as those near the boundaries of their geographical ranges, even small deviations from optimal climatic conditions can lead to considerable declines in survival or reproductive success (Martin 1987). The result could be declines in net recruitment (the process whereby surviving juveniles are added to a population), with potential profound effects on population size. Even small changes in net recruitment may have dramatic negative impacts on some populations of migrant landbirds that are presently declining (Askins et al. 1990; Robbins et al. 1989).

Climate modification also can indirectly affect birds by changing the amount, distribution, structure, and condition of habitats. Climate-driven changes in the amount or distribution of forest types, such as longleaf pine-wiregrass sandhills or spruce–fir forests, could affect species that are narrowly associated with those ecosystems such as red-cockaded woodpeckers and red crossbills (*Loxia curvirostra*), respectively. Similarly, increases or range extensions of exotic invasive plants due to changing climatic conditions could indirectly affect birds by changing plant food availability or vegetation structure. Climate-driven changes in fire frequency or severity may adversely affect some species while benefiting others. For example, frequent burning controls midstory hardwood vegetation, thus helping to maintain the open stand structure of pine woodlands, the primary habitat of endangered red-cockaded woodpeckers. This species is found primarily in the Florida Coastal Plain, the Piedmont, and the Mid-South. Frequent fire may also maintain young oak-scrub vegetation that is the optimal habitat for Florida scrub jays, a federally threatened species native to Florida. In contrast, more-frequent fires in the upland hardwood forests of the Appalachian-Cumberland highlands could reduce shrub cover, which is important habitat for species such as hooded warblers (*Wilsonia citrina*) (Greenberg et al. 2007). The eventual direct and indirect effects of climate change depend on a variety of factors, which make developing general predictions difficult and prone to uncertainty.

Bird responses to climate change could include adapting, changing the temporal or phenological aspects of their ecology, changing their spatial distribution, or moving to higher elevations. Many species may be capable of adapting to changes in abiotic conditions such as temperature and precipitation, and (or) biotic conditions such as vegetation (amount, type, and structure) and food resources. A certain amount of adaptation is predicted based on their natural plasticity. However, even highly

adaptive species may reach their limits and would be forced to employ other measures to deal with climate change effects.

Temporal responses may appear as changes in migratory patterns, with birds arriving on the breeding grounds earlier than is typical. Phenology of bird migration is significantly affected by climatic conditions (Root et al. 2003; Sanz 2002; Sparks and Crick 1999), and recent shifts in migration dates are likely in response to recent climate change (Gordo 2007). Shifts in migration timing during the last few decades have typically trended toward earlier arrival when spring temperatures have increased (Gienapp et al. 2007; Root et al. 2005). These earlier occurrences of spring-like conditions may have negative impacts on nesting success if the trees, shrubs, or other vegetation have not leafed out or developed sufficiently to provide adequate concealment for eggs and hatchlings. Further, the earlier arrival may not be timed to match emergence of prey, which may impact adult as well as chick survival. Therefore, negative effects on nest success, breeding success, and population characteristics are possible.

Higher spring temperatures may also cause advances in the phenology of plants and insects (Schwartz et al. 2006). If birds arrive earlier on the breeding grounds, they will likely breed earlier. If the end of the breeding season is unchanged or if it is prolonged, more time would be available for reproduction, which could allow breeding pairs of some species to increase the number of clutches they produce. Thus, the annual reproductive rate (number of fledglings produced per pair) may actually increase. Further, there would be an extended period for nestlings to grow and for fledglings and adults to add body fat for the autumn migration and winter. These potential advantages of earlier arrival depend on possible changes in availability, distribution, and abundance of the food supply. However, whether the prey base will be sufficient to support birds and their young throughout an extended breeding period is unknown. Also unknown are additional consequences related to the presence and activities of animals that prey on birds. Weather events and climate change may impact departure dates of birds, progression and speed of migration, and stopover frequency and duration. Gordo (2007) suggested that additional research is needed to determine the actual relevance of each climatic variable on bird migratory phenology.

Some species may undergo distribution shifts, most likely by extending the northernmost boundary of their ranges. Hitch and Leberg (2007) found that the northern limits for many birds are significantly shifting northward, on average 2.35 km/year, and suggested this was a response to climate change. For some species, warming may have a beneficial effect if it results in a widening of the area where their climate requirements are met and if they can use this expanded area. The expanded area may contain food resources that differ from the current range; hence, the species must be capable of adjusting to new food resources and possibly different predators.

Some bird species may respond to climate change by shifting their elevation distributions. Because climate attributes—including temperature, windspeed, and precipitation—vary along elevational gradients, biotic variables such as vegetation composition and food resources also vary. In a study examining responses of birds along two elevation gradients in the French Alps over a 30-year period, Archaux (2004) found that 30% of bird species had significant shifts in elevation, with five species moving downward and three moving upward. Although there was a 2.3°C increase in spring temperature over the previous 25 years, most communities remained at their same elevation. He suggested that bird distributions by elevation have not yet been influenced by warming and that habitat suitability, interspecific interactions, or other site-specific factors likely accounted for the elevation shifts by some species. However, Berthold (1998) found an increase in elevation for several alpine species in Germany and Switzerland and concluded that these shifts were the result of temperature increases.

WATERFOWL

A warming climate may bring some localized benefits to ducks, geese, and swans but the overall impact on waterfowl populations is likely to be negative (Glick 2005), largely because of changes

to breeding and wintering habitats. Wetland habitats are particularly sensitive to changes in precipitation and temperature and may be degraded or eliminated if effects of climate change are not mitigated (North American Bird Conservation Initiative 2010). Expected temperature increases without accompanying increases in precipitation would affect these wetlands by reducing water inputs, reducing recharge capacity, changing the timing of wetland recharge, and increasing the frequency of droughts; and those wetlands that depend on snowmelt would be reduced or disappear. Moreover, the predicted increase in the intensity of storms and tornadoes would raise the likelihood of erosion from flash floods, reduce the length of time that water may be contained in wetlands, and compromise the predictability of changes in wetland water levels. These events could alter plant communities, prey abundance, and eventually the abundance of wetland species of birds.

In many areas of the United States, shallow wetlands known as potholes (shallow depressions that fill with water during the spring) and the wetland-dependent breeding birds that use them would be highly threatened by climate change (North American Bird Conservation Initiative 2010). This is particularly true of the prairie pothole region (spanning both sides of the U.S./Canadian border in the northern Great Plains), which is known for high productivity of waterfowl. With warming temperatures, many of these potholes are expected to disappear or be wet for shorter periods of time, making them unsuitable as breeding habitat (Glick 2005). Many of the waterfowl that travel through or overwinter in the Southeast breed in these wetlands; hence, reductions in waterfowl productivity in that area may affect populations in the five southeastern subregions.

Climate change is predicted to affect the timing and distance traveled during waterfowl migration. Wintering habitat for most species consists of freshwater lakes, river basins, deltas, coast marshes, and estuaries in the United States and Mexico. Many of the birds from the Atlantic flyway winter in locations along the mid-Atlantic coast, including the Chesapeake Bay and the Delaware Bay. Others fly further south into the Carolinas, Georgia, and Florida. Mississippi flyway and Central flyway species mostly winter in the Platte River Basin, the Mississippi Alluvial Valley, the lower Mississippi River Delta, the Playa Lakes Region, and in Coastal Plain marshes and flooded fields along the Gulf of Mexico. Thus, birds that breed within the Southeast or elsewhere, such as in the prairie pothole region, may be affected by conditions in several of the subregions in the Southeast.

Changes in sea level, precipitation patterns, and other climate conditions may influence the availability and quality of food and habitat for waterfowl. Glick (2005) suggested that a rise in sea level may reduce viable winter habitat in coastal wetlands by 17–43% in areas without structured protection, and by 20–45% in areas where sea walls and other protective structures are in place. She predicted an 8- to 86-cm rise in average sea level by 2100, which could eliminate up to 45% of coastal wetlands in the conterminous United States and threaten important wintering habitat including the shallow wetlands along the Gulf of Mexico and Atlantic coasts. Further, part of the Coastal Plain, Mississippi Alluvial Valley, and Mid-South would be particularly vulnerable (Glick 2005). Hence, climate change appears to have the potential for damaging breeding areas, migration patterns, and wintering areas of waterfowl, which spend at least part of their annual cycle in the Coastal Plain and Mississippi Alluvial Valley in the Southeast.

SHOREBIRDS

Shorebirds likely are similar to waterfowl in terms of the potential effects of climate change on their productivity and habitat quality. Shorebirds rely on habitat found in low-lying coastal and intertidal areas during migration and wintering. According to the North American Bird Conservation Initiative (2010):

- The Gulf of Mexico and mid-Atlantic coasts (Mid-South and parts of the Coastal Plain) have seen the highest rates of relative rise in sea level and recent wetland loss in the United States.

- Rise in sea level is anticipated to inundate or fragment existing low-lying areas, including mudflats, barrier islands, and salt marshes—all habitats that shorebirds utilize.
- In areas that receive heavier rainfall, adverse effects of runoff with excess nutrients or changes in salinity are expected in coastal areas.

Galbraith et al. (2005) modeled changes in intertidal foraging habitat for shorebirds in response to sea level rise at five sites, one of which was in the Mid-South (Texas). They found that a conservative 2°C warming over the next 100 years would result in major intertidal habitat losses at four of the sites, with the Texas site experiencing an almost complete habitat loss by 2100 (which might be mitigated by construction of a new sea wall).

The quality of shorebird breeding habitat in far north latitudes also is likely to be affected by climate change. Adverse habitat changes already may be evident in arctic and subarctic areas of North America (Chapin et al. 1995). Increased temperatures and thawing of permafrost in arctic areas would deepen the active soil layer, allowing trees and shrubs to expand into areas that previously supported sedges, grasses, and dwarf shrubs. According to the North American Bird Conservation Initiative (2011), this change in habitat structure and plant species composition would have multiple effects. It would affect bird distributions and abundances, especially arctic and alpine shorebirds and waterfowl, by narrowing or eliminating their breeding habitats. However, an earlier onset of spring could initially increase productivity of nesting shorebirds if they migrate earlier to take advantage of earlier insect availability. Although precipitation would likely increase, warmer temperatures in the arctic could mean higher rates of evaporation, thus reducing soil moisture and the amount of tundra wetlands in the western and central arctic. Further, it could mean more frequent fires on the tundra. Thus, shorebirds that utilize the five subregions in the South for part of their life cycle may be adversely affected by climate change on their breeding habitat, migration routes, and wintering habitat.

UPLAND GAME BIRDS

Extremes in climate are likely to impact game birds at all points of their life cycle, including the timing of breeding, selection of nest sites, and availability of food resources. Increased precipitation, which may result in unusually wet springs, could flood or wash away nests and drown chicks. Females of ground-nesting species may nest or re-nest above the high water, at sites of higher elevation than they would normally select. When the land dries, these areas may be more susceptible to predation (Wildlife Management Institute 2008). In contrast, increased drought frequency could also adversely affect upland game birds.

A warming climate may cause an accelerated breeding timetable, but how that would correspond to the emergence of insects and plants is unknown. For example, food resources may adapt to the evolving climate conditions at a different rate than game birds; thus, the presence of warm-season grasses and insects vital to chick growth may be lacking at the appropriate time. Extreme changes in climate would favor further invasion of nonnative plants and insect pests, which would degrade habitat quality.

Northern bobwhite (*Colinus virginianus*) and ruffed grouse (*Bonasa umbellus*)—two important, southeastern nonmigratory game birds with different habitat associations and range distributions—are predicted to differ in their response to climate change (Matthews et al. 2007). Bobwhite occur in all five southeastern subregions, but are least abundant in the Southern Appalachians (Matthews et al. 2007). Optimal bobwhite habitat in forested areas consists of early successional forests that are dominated by pines or hardwoods and that include both herbaceous and woody growth (Wildlife Management Institute 2008). Hot, dry summers could potentially reduce bobwhite recruitment by causing embryo mortality (Wildlife Management Institute 2008) or hyperthermia in young chicks (Evans 1997; Sumner 1935). However, increases in fire frequency associated with drier, hotter conditions (Chapter 5) will likely promote open or early successional habitat. Thus, reductions

in annual productivity could be offset by higher quality habitat that could sustain more bobwhite breeding pairs (Wildlife Management Institute 2007).

The natural range of ruffed grouse in the Southeast is limited to the Appalachian-Cumberland highlands, which is the southern limit of their range, most of which is in Canada, Alaska, the upper Midwest, and the Northeast. In the East, ruffed grouse use deciduous forest and oak-savannah woodland and prefer young forest with abundant ground cover. Because they burrow in the snow to protect themselves from freezing temperatures, reduced snowfall or accelerated rates of melting snow associated with climate change could reduce their winter survival (Wildlife Management Institute 2008).

Matthews et al. (2007) modeled potential abundance and distribution of bobwhite and ruffed grouse in the Eastern United States under eight climate change scenarios. Each of their models for bobwhite (ranked as having high reliability) predicted an increase in range and (or) abundance in the Appalachian-Cumberland highlands and the Northern United States. In contrast, their models for ruffed grouse (ranked as having low reliability) predicted a range contraction and reduced abundance in the Southeast. Results of these models illustrate potential differences in climate change effects on different species with very different habitat associations, current ranges, and life history requirements.

NEOTROPICAL MIGRATORY BIRDS

Climate change and associated habitat alterations are likely to cause significant losses in Neotropical migratory species in the conterminous United States (Price and Root 2001). For example, bioclimatic models developed by Rodenhouse et al. (2008) predicted that high-elevation bird species in the Northeast may experience a $\geq 50\%$ reduction of suitable habitat with warming as little as 1°C . They also suggested that mid-elevation species are likely to experience declines in habitat quality that could ultimately affect their future viability. However, models predicting climate effects on the distribution of 13 Southern Appalachian forest-nesting birds indicated that naturally occurring climate oscillations may have a more immediate impact on species distributions, and ultimately population viability, than a slowly warming climate (Kim et al., unpublished report).

Climatic conditions in the Neotropic Ecozone (South and Central America, the Mexican lowlands, the Caribbean Islands, and southern Florida) likely influence the winter distribution and abundance of many Neotropical migratory birds (Root 1988a,b). Winter temperatures are predicted to increase $2\text{--}3^{\circ}\text{C}$, with average increases up to 156% over current levels (Smith and Tirpak 1989). These climate changes and potential associated changes in habitat quality, distribution, and availability on wintering grounds could affect populations of Neotropical migratory birds that breed in the southeastern United States.

Difficulties in predicting outcomes of climate change for Neotropical migratory birds is exacerbated by potential changes on their breeding grounds, wintering grounds (which for many migratory birds are primarily outside of the United States), and along migration routes. The distributions of some species may shift northward, move higher in elevation, or contract as warming occurs. Species that nest at high elevations in spruce-fir habitat are especially vulnerable, as that forest type is likely to diminish or disappear as temperatures warm (Chapter 11). Species that depend on specific vegetation associations may also be vulnerable, because most U.S. forests cannot rapidly respond to climate change (Root et al. 2003).

AMPHIBIANS

Shifts in climate can have negative effects on amphibian populations (Beebee 1995; Gibbs and Breisch 2001; Parmesan and Yohe 2003). Temperature and precipitation have major influences on the life cycle of amphibians, particularly their breeding activities (Busby and Brecheisen 1997; Donnelly and Crump 1998; Gosner and Black 1955; Saenz et al. 2006). Because most North

American amphibians lay eggs in water, the amount and timing of precipitation can affect their reproductive activities and yearly reproductive output (Conant and Collins 1998; Saenz et al. 2006). Water availability also affects adult amphibians because they are vulnerable to losing water from their skin and respiratory systems (Carey and Alexander 2003).

Warming climates influence the body condition of some amphibians (Corn 2005; Reading 1998; 2007; Walther et al. 2002). For example, Reading (2007) found that increased air temperatures are correlated with reductions in body condition indices of female European toads (*Bufo bufo*), rendering them less able to assimilate energy reserves during spring and summer and subjecting them to rapid depletion of stored energy reserves during mild winters compared to cold ones. Accelerated tadpole development caused by warmer water temperature can result in smaller body mass of young frogs at metamorphosis (Harkey and Semlitsch 1988).

Emissions and pollutants associated with climate change may also cause thinning of the stratospheric ozone, leading to increased atmospheric ultraviolet-B radiation (UV-B) that has been suggested as a cause of amphibian declines (Blaustein et al. 2003). Although experimental studies indicate that UV-B exposure may cause mortality or deformities in some amphibian species, no studies show a linkage between these findings and actual changes in abundance or distribution, or show that amphibians are being exposed to higher levels of UV-B in the wild (Corn 2005). Climate change also could alter the spread or virulence of emerging infectious diseases associated with amphibian population declines (Daszak et al. 2001), but responses would differ according to the ecology of the particular pathogen involved. For example, chytrid frog fungus (*Batrachochytrium dendrobatidis*), the cause of mass die-offs in some amphibian populations, thrives at moderate temperatures (23°C) but dies at 30°C; therefore, warmer, drier climates seem unlikely to promote the spread of this pathogenic fungus (Corn 2005).

Other consequences of a warming climate are changes in range and distribution of species and changes in interactions among species (Badeck et al. 2004; Blaustein et al. 2001; Collatz et al. 1998; Schierenbeck 2004; Walther et al. 2002). Parmesan and Yohe (2003) estimated that shifts in climate over the past 20–140 years caused more than half (59%) of the 1598 plant and animal species documented to experience measurable changes in phenology or distribution or both. Responses of amphibian phenology to climate change was double that of trees, birds, and butterflies and nearly eight times that of herbs, grasses, and shrubs (Parmesan 2007). In temperate areas, increases in temperature can trigger early amphibian emergence from hibernation and influence reproductive activities (Carey and Alexander 2003; Oseen and Wassersug 2002; Saenz et al. 2006).

It is well known that amphibian phenology in North America is influenced by weather. Gibbs and Breisch (2001) showed that breeding by wood frogs (*Lithobates sylvaticus*), spring peepers (*Pseudacris crucifer*), and bullfrogs (*Lithobates catesbeianus*) shifted to earlier dates during winter and spring when temperatures were higher than the previous year near Ithaca, New York. Saenz et al. (2006) found that several species in eastern Texas are sensitive to temperature and that temperature influences breeding activity in winter breeding species. Higher temperatures could alter the timing of breeding activity for several species, possibly increasing the risk of asynchronous timing with typical rainfall periods and availability of breeding sites or altering competitive interactions among tadpole species that would otherwise be temporally segregated.

Higher water temperatures also could affect the reproductive success of some amphibians, eventually altering their geographic distribution. Water temperature affects developmental rates of many amphibians, and larva of different species differ in their tolerance to minimum and maximum temperatures. For example, maturation was advanced and body mass increased for ornate chorus frog (*Pseudacris ornata*) tadpoles at day 32 with increasing temperature up to 25°C, only to reverse at 30°C; no tadpoles metamorphosed from water kept at 10°C after 111 days (Harkey and Semlitsch 1988). Geographic distribution and the timing of breeding and egg laying by some aquatic-breeding amphibians generally correspond with limits of larval temperature tolerance (Moore 1939). Compared to species that cannot tolerate very cold water temperatures, amphibians whose larvae tolerate low temperatures but are sensitive to higher temperatures are more northerly

in their distributions or breed earlier in the spring or both (Moore 1939). For example, wood frogs are early breeders (March, in New York City) and range as far north as Canada, but also occur in the Southern Appalachians. Under experimental conditions, wood frog larvae successfully developed in waters ranging from 2°C to 24°C but died in waters about >25°C. In contrast, green frogs (*Lithobates clamitans*) are later breeders (June, in New York City), but do not range as far north as wood frogs and occur as far south as Florida. Under experimental conditions, green frog tadpoles successfully developed in waters ranging from 12°C to 33°C, but died at temperatures that were below (about 10°C) or above (about 36°C) this range (Moore 1939). These studies suggest that higher temperatures in the Southeast could cause some amphibians to shift their distributions northward because of tadpole temperature tolerance limits.

Impacts are expected to be stronger for species that occur farther north, where climate (particularly temperature) has changed more than at lower latitudes (Parmesan 2007). However, studies focused on southeastern amphibians suggest threats from climate change as well. Milanovich et al. (2010) predicted that increased temperatures projected in long-term weather models would cause a loss of salamander diversity in the Appalachians. Many mountain-top species may be near their thermal maxima and have limited dispersal ability. Increases in temperature would result in lost habitat for many species, and those with small geographic ranges will be at greater risk of extinction. McCallum (2010) predicted catastrophe for a subspecies of cricket frogs (*Acris crepitans blanchardi*) in Arkansas, based on long-term projections of a warming climate. He suggests that predicted climate change would significantly reduce the inclination of frogs to reproduce, which could induce population declines.

Change in precipitation, drought in particular, has been suggested as a threat to southeastern amphibians. Drier soil and leaf litter may create suboptimal conditions for terrestrial salamanders and the terrestrial stages of aquatic-breeding amphibians. Mole salamanders (*Ambystoma talpoideum*) are primarily burrowers (Ashton and Ashton 1988b) and likely are sensitive to soil moisture levels. Terrestrial salamanders (Plethodontidae) may decrease surface foraging in response to dry forest floor conditions; the resulting loss of energy could lead to poor body condition or delayed breeding (Petranka 1998). Similarly, some adult stream-breeding salamanders use upland habitats for foraging and overwintering or both (Ashton and Ashton 1978; Crawford and Semlitsch 2007). Studies have shown that abundance of adult stream salamanders, such as the Blue Ridge two-lined salamander (*Eurycea wilderae*), may be reduced in sites with reduced leaf litter depth, soil moisture, and overstory cover (Crawford and Semlitsch 2008; Moorman et al. 2011), suggesting that they are sensitive to moisture and temperature.

Amphibians that are primarily aquatic as adults may be adversely affected if changes in temperature or precipitation reduce the availability of permanent ponds or streams. Stream-dwelling salamanders rely on stable stream flows for larval development, which can take from several months to many years (Rodenhouse et al. 2009). Increased water temperatures may also be detrimental to larval salamanders in streams (Peterman and Semlitsch 2009; Semlitsch 2000). Long-term drying of wetlands could cause populations of species such as cricket frogs (*Acris* spp) or pig frogs (*Lithobates grylio*) to become more isolated and less widely dispersed across landscapes. Daszak et al. (2005) suggested that population declines of four species at the U.S. Department of Energy Savannah River Site in South Carolina are linked to a drying trend in the 1990s and shortened hydroperiods of breeding sites. In Arkansas, Trauth et al. (2006) suggested that drought and land leveling have caused population declines in Illinois chorus-frogs (*Pseudacris streckeri illinoensis*).

Changes in precipitation patterns could affect amphibian species that have distinct breeding seasons and depend on seasonal rainfall patterns and consequent hydroperiods in breeding ponds. For example, oak toads (*Bufo quercicus*) in Florida breed from May to September (Greenberg and Tanner 2005a); spring and summer droughts that result in dry breeding ponds could reduce juvenile recruitment. Widespread long-lasting drought could extend beyond the oak toad's expected lifespan of 2–4 years, producing dramatic or even catastrophic impacts to the species. Changing weather patterns also could alter amphibian-breeding cues (events that trigger breeding activity).

For example, explosive breeding by eastern spadefoot toads (*Scaphiopus holbrookii*) is triggered nearly exclusively by heavy rainfall events that fill previously dry ephemeral ponds (Greenberg and Tanner 2004, 2005b). In the absence of heavy rainfall events or in situations where ponds are already full during heavy rainfall, spadefoot toads would be unlikely to breed. Hydroperiod lengths are likely to affect amphibian populations differently because developmental rates of aquatic larvae differ among species. For example, spadefoot toad tadpoles develop and metamorphose in as little as two weeks, whereas most frog tadpoles require much longer, from three months for the gopher frog (*Lithobates capito*) to more than a year for bullfrogs (Ashton and Ashton 1988a).

Increases in hurricane activity in the Atlantic Ocean have been linked to higher sea surface temperatures in the North Atlantic (Goldenberg et al. 2001). More frequent storms are likely to result in higher storm surges and more wind damage in coastal areas. Schriever et al. (2009) found a drastic overall decrease in amphibian abundance in Louisiana following hurricanes Ivan, Katrina, and Rita. However, Gunzburger et al. (2010) found lasting changes in water chemistry in wetlands overwashed by storm surge during hurricanes in northwestern Florida, but they did not see a lasting impact on amphibian communities. They suggest that amphibian communities adjacent to marine habitats in the Southeast are resistant to the effects of storm surge overwash.

Climate-induced changes in the phenology of amphibian breeding and larval development could introduce unexpected interactions and results as well. For example, Fucik (2011) demonstrated that earlier breeding by winter-breeding frogs in Texas would increase the likelihood of adverse interactions with nonnative invasive Chinese tallow tree (*Triadica sebifera*) leaf litter, which can be lethal to developing tadpoles. Thus, a warming climate in the southeastern United States may lead to declines in some amphibian species as the result of multiple direct or indirect effects.

REPTILES

As a group, reptiles may be less vulnerable to climate change than amphibians because their scale-covered skin makes them less vulnerable to desiccation and better able to tolerate the drier, warmer conditions that are predicted (Pough et al. 2001). Their eggs are also protected from desiccation by calcareous shells. Further, their life cycles do not involve an aquatic egg or larval stage, which limits many amphibians to wetland habitats for reproduction. Many reptile species are highly mobile and capable of evading thermal stress; they also can travel long distances and have very large home ranges compared to amphibians (Brown 1993). Despite having several characteristics that should decrease vulnerability to climate change, reptiles could be affected by changes to primary habitats, temperature-driven energetic shortfalls, temperature-dependent sex determination, and changes in food availability.

Numerous reptile species are semi-aquatic, using wetlands or rivers as their primary habitat, and terrestrial habitats for egg-laying (Gibbons et al. 2000). In the Southeast, black swamp snakes (*Seminatrix pygaea*), water snakes (*Nerodia* spp), common mud turtles (*Kinosternon subrubrum*), American musk turtles (*Sternotherus* spp), and chicken turtles (*Deirochelys reticularia*), among others, are closely associated with ephemeral wetlands. Others, such as riverine map turtles (*Graptemys* spp), live in streams or rivers. Bog turtles (*Glyptemys mühlenbergii*) are restricted to bogs, seeps, and similar habitats. Increased frequency or duration of drought that affects these habitats could have dire effects on the reptile species that depend on them.

Reptile species with limited range distributions, such as the Louisiana pine snake (*Pituophis ruthveni*), may be particularly vulnerable to extinctions driven by climate change. Louisiana pine snakes are rare and declining throughout their range, which is limited to sandy soils in eastern Texas and west-central Louisiana (Rudolph et al. 2002). Baird's Pocket Gophers (*Geomys breviceps*), their primary prey (Rudolph et al. 2002), depend on a well-developed herbaceous layer maintained by frequent fires. The presence of major river barriers further impedes potential population migration of Louisiana pine snakes. Climate change or changes to habitat quality resulting from changes in

fire frequency could potentially eliminate this species because of its limited and restricted distribution and barriers to movement.

The higher temperatures predicted over the next several decades could have major effects on the population characteristics and persistence of reptiles whose sex is determined by the temperature during incubation (Gibbons et al. 2000; Hawkes et al. 2007). This has been shown to occur in crocodylians (which include alligators and crocodiles) and in some turtles, with a greater proportion of females hatching with temperature increases $\geq 1^{\circ}\text{C}$ (Janzen 1994). Hawkes et al. (2007) used sand temperatures and historical air temperatures at Bald Head Island in North Carolina to develop predictive models of climate change effects on hatchling production and gender of loggerhead sea turtles (*Caretta caretta*). They found that an increase of just 1°C tips the gender balance of hatchlings to female, which could lead to unfertilized clutches, lost cohorts, and eventual extinction, and that temperatures $>3^{\circ}\text{C}$ produce high mortality.

Higher temperatures associated with climate change could also have other implications for productivity in some reptile species. For example, Hawkes et al. (2007) found that earlier and prolonged nesting seasons for loggerhead sea turtles were correlated with warmer sea surface temperatures. In addition, several studies indicate that pond turtles exhibit increased juvenile growth rates and reached sexual maturity at an earlier age in warmer temperatures and longer growing seasons (Frazer et al. 1993). Gibbons et al. (1981) found that in a pond receiving thermal effluent at the Savannah River Site, male and female sliders (*Trachemys scripta*) grew faster than controls in nearby natural habitats; the males matured at an earlier age, but at the same size as the controls, while the females matured at the same age but at a larger size.

High temperatures that restrict foraging activity by reptiles could lead to energy shortfalls (Huey et al. 2010). Sinervo et al. (2010) correlated extinction probability of blue spiny lizard (*Sceloporus serrifer*) populations in Mexico with the number of hours per day that spring temperatures exceeded the body temperature at which the lizards are still capable of activity. They found that lizard populations persisted when temperatures acceptable for activity were over 3.85 h, but did not persist when the time available for foraging was shorter because net energy gain became insufficient for reproduction (Sinervo et al. 2010). Dunham's model for canyon lizards (*Sceloporus merriami*) in Texas corroborates the idea that restricted activity time may cause extinction, finding that even a 2°C increase in air temperature reduces activity time, causing a reduction in energy gain and population growth (see Gibbons et al. 2000). These studies demonstrate ecophysiological mechanisms that incorporate adaptive evasion (retreat to avoid high air temperatures) into evaluations of vulnerability to climate change and likelihood of extinction.

Reptile species with specialized diets also could be vulnerable to changes in climate that affect their food sources. For example, populations of eastern hog-nosed snakes (*Heterodon platirhinos*) and southern hog-nosed snakes (*Heterodon simus*) could decline in response to drought-induced population declines of frogs and toads, their primary prey (Ashton and Ashton 1988a).

BUTTERFLIES

Butterflies play a significant and critical role in ecosystem function as pollinators (Withgott 1999). Climate change, to the extent that it alters butterflies and other pollinator populations, would have cascading effects on entire ecosystems (Kremen and Ricketts 2000). As a consequence of their complex life cycles, diverse larval hosts, and frequent dependence on a particular suite of nectar resources, butterflies would respond to climate change in complex ways. Butterfly larvae are also important herbivores in terrestrial ecosystems and critical prey for birds and other predators. Therefore, the effects of climate change on butterfly populations would have consequences for both host plant and predator species (Schowalter et al. 1986).

Substantial literature documents geographical and phenological shifts in a wide range of butterfly species that are correlated with recent climate change. Briefly, during the past century, northward shifts in temperature isotherms by an average of 120 km in Europe are correlated with northward

shifts in ranges of 35–240 km of 63% of nonmigratory butterfly species (Parmesan et al. 1999). Likewise, phenological variables of butterflies in Great Britain have shifted earlier by 2–3 days per decade (Roy and Sparks 2000). In the southeastern United States, many butterfly species are already experiencing similar patterns in relation to climate change, and other species are likely to join the ranks of the affected in the future. Other ecological processes (such as fire regimes) that will likely be altered by climate change would also affect butterfly populations.

The distribution of butterfly and skipper species in the southeastern United States is reasonably well documented and provides the basis for continent-wide examination of geographic patterns (Scott 1986), offering insights into the factors that control the overall distribution of these species. In general, butterflies are more diverse at lower latitudes and topographic complexity increases local diversity (Parmesan et al. 1999). In most of the southeastern United States, little topographic complexity leads to a rather homogeneous fauna as well as a weak diversity gradient from north to south. The most prominent exception to this pattern is a marked increase in species diversity in southern Texas and peninsular Florida, the result of a mild temperature regime, especially in the winter months. In these areas, numerous tropical species, which lack the physiological mechanisms to survive freezing temperatures, can persist as residents throughout the year.

The study of most southeastern butterfly species in relation to their environments is sufficiently advanced to provide a basis for examining the potential effects of climate change (Scott 1986). Host plant relationships, nectar resources, and individual species phenology will all interact in complicated ways as species respond to climate change (McLaughlin et al. 2002; Roy and Sparks 2000). To simplify our analysis, we divide the species into two categories based on their physiological ability to survive low temperatures. The first group consists of species that can survive subfreezing temperatures during some stage of the life cycle. Consequently, these species can maintain populations in both temperate and more extreme climates. They typically diapause (effectively hibernate) at specific stages of the life cycle, which differs among individual species. With few exceptions, these species are resident at any given locality.

The second group, which is more diverse, consists of species that lack the ability to diapause during periods of subfreezing temperatures. Not surprisingly, temperatures that are lethal to some species are not lethal to others, but 0°C is a reasonable approximation for most. A subset of these species—with primarily tropical affinities—is strictly resident, primarily in southern Texas and peninsular Florida. These species persist throughout the year without undergoing diapause. A second and highly variable subset includes species that are migratory (using migratory in the broadest possible sense). At one end of this spectrum are the intergenerational migrants—monarch (*Danaus plexippus*) and painted lady (*Vanessa cardui*)—that travel hundreds to thousands of kilometers to avoid low temperatures. At the other end are the species that reside permanently in southern latitudes but expand their ranges northward, often by hundreds of kilometers, during the warmer months.

These species groups can be expected to react to climate change in different ways. In relation to temperature patterns such as a northward shift in temperature isotherms, resident species with the ability to diapause will tend to adjust their ranges accordingly. The details of those adjustments will depend on concurrent shifts in host-plant distribution and the ability of individual species to move or disperse in an increasingly fragmented and rapidly changing landscape (Parmesan et al. 1999; Roy and Sparks 2000). Sedentary species that are resident in the southern-most sector, where the ability to diapause in response to low temperatures is generally not required, will tend to expand their ranges northward.

The ability of individual species to successfully shift their ranges in response to climate change will presumably be highly variable. Species that regularly migrate and recolonize the northern portions of their ranges each year will face few obstacles beyond adjustments to the geographical details and timing that may be required by changing climatic patterns. Issues related to habitat fragmentation will presumably be minimal as these species are already moving across a highly fragmented landscape every year. The presence of suitable host plants could be an issue; however, many of these species utilize a wide range of host plants over a wide geographical range.

Resident species, both southern species that do not diapause and more northerly species that do, may face more difficulties. Many of these species are thought to be quite sedentary, although the evidence against the ability of most species to migrate is not strong. Many species will have difficulty altering their ranges in fragmented landscapes, although this response is highly species dependent. Species that are adapted to disturbed or widespread habitats will be able to adapt to climate change relatively easily, but habitat specialists may not (Hoegh-Guldberg et al. 2008). An additional complication is the availability of suitable host plants. Many species, especially those with narrow host–plant requirements will be constrained by the response of host plants to climate change. Substantial lag times may result before host species colonize otherwise suitable habitats. Both factors, low capacity for movement and host–plant requirements, will undoubtedly result in many species being unable to find suitable habitat across a fragmented landscape impacted by climate change.

The response of other ecological processes to climate change could also alter butterfly community structure. Fire regimes, in particular, play a major role in structuring the abundance and species composition of butterfly communities in the southeastern United States (Rudolph and Ely 2000; Rudolph et al. 2006; Thill et al. 2004) by altering vegetation structure with profound implications for nectar resources and host plants. Climate change would potentially alter fire regimes through changes in fuel loads, precipitation patterns, and temperature regimes (Chapter 5). Prediction of fire regimes under climate change is fraught with difficulties, particularly because current regimes are primarily driven by human factors, especially prescribed fire. The fire regime before European settlement was typified by large-scale, low-intensity fires (Frost 1998). Fire return intervals were typically short, often in the range of two to five years in forested habitats, especially those dominated by pine and oak. Five centuries of escalating ecological change following European settlement have greatly altered fire regimes of the southeastern United States. Although predicting ecological responses due to changes in fire regimes is in its infancy, less fire typically results in fewer butterflies, primarily as a result of reduced nectar resources (Rudolph et al. 2006; Thill et al. 2004).

Based on predicted estimates of climate change, butterfly communities are likely to change dramatically. Predicting the responses of individual species is extremely difficult because most species are thought to respond to weather events in addition to long-term climate averages. However, changes over the last century strongly suggest that many species will expand their ranges substantially northward, contract their ranges less dramatically in the southern portions of their ranges, and adjust their distributions in relation to altitude. Other species that lack the ability to colonize new habitats rapidly will likely be locally or globally extirpated (McLaughlin et al. 2002), with resultant loss of diversity and critical ecosystem services.

MANAGEMENT OPTIONS FOR CLIMATE CHANGE

Historically, climate change has been an important driver in range shifts, extinction, colonization, and evolution through adaptation of species. Given the limitations imposed by the physiology and plasticity of each species, many of these outcomes were mediated by the availability of dispersal corridors in a landscape relatively free of modern human influence. However, the southeastern landscape is highly fragmented, with obstacles to dispersal imposed by urban and rural development, roads and highways, dams, degraded habitats, land ownership, and land use patterns. Using the past as a model, “preservation” of biological diversity—as if it were a static and unchanging entity that should remain as we know it today—is unrealistic in the face of unprecedented rates of climate change. However, conservation of biological diversity can be achieved if we recognize and allow the same processes of wildlife movement and reassembly of communities to occur by promoting landscape features that permit movement.

For some endemic island subspecies, such as the Florida Key deer, Key Largo woodrat, and Key Largo cotton mouse, their habitats could disappear, eliminating isolated areas that have maintained their genetic uniqueness. For these subspecies, establishing populations elsewhere may not

be feasible because of the potential for interbreeding with mainland subspecies and “swamping” of their unique genetic traits.

Management for the conservation of biological and genetic diversity in a changing climate can involve multiple approaches that can be implemented by land use and highway planners, landowners (especially those with large land holdings), and state and federal land managers. Listed below are some critical steps toward mitigating adverse effects of climate change.

- Increase both amount and connectivity of wild lands and habitats through acquisitions and conservation programs. To allow wildlife to travel in response to climate change, give particular attention to interconnected habitats that run from north to south, and from higher to lower elevations (Root and Schneider 2002).
- Consider wildlife movement and road-kill mortality when planning or improving roads and highways by providing (as needed) elevated sections of highways, or by building wildlife underpasses to allow uninterrupted migration corridors (Root and Schneider 2002).
- Manage ecosystems to restore and maintain conditions that promote optimal habitat and larger populations, and thereby enhance the resilience of associated native wildlife species.
- Protect and conserve coastal habitats through strategic planning, zoning, and building codes. In areas of predicted rising sea levels, address inland migration of coastal wetlands in plans for new developments.
- Use caution when building barriers for flood control or during construction as they can result in elimination of existing wetlands. Establish practices that reduce the susceptibility of coastal habitats to sea level rise; examples include eliminating ditches to restore the hydrologic regime and limit saltwater intrusion, and assisting in the development of vegetation by planting salt-tolerant species and building oyster reefs to buffer shorelines from storm events and wave action.
- Restore riparian zones and watersheds to protect and maintain water quality, quantity, flow, hydrologic processes, and temperatures in wetlands and streams.
- Conserve species and special habitats, paying special attention to restoration and management of rare species or ecosystems.
- Aim for representation, resiliency, and redundancy by creating networks of intact habitats that represent the full range of species and ecosystems in a region, with multiple robust examples of each habitat.
- Reduce existing ecosystem stressors, such as habitat loss and alteration, pollution, ozone depletion, invasive species, and pathogens.
- Foster partnerships among agencies, organizations, scientists, and citizens to develop science-driven, landscape-scale strategies to maximize the use of scarce resources.
- Encourage policies that reduce the “carbon footprint,” support wildlife and their habitats, and reduce climate change stresses.
- Mitigate climate change by reducing CO₂ emissions and manage forests to promote carbon sequestration where appropriate.
- Monitor, model, and implement adaptive management in response to unforeseen consequences of climate change such as trophic cascades (whereby the addition or removal of top predators impacts populations of their prey and the plants they eat).
- Increase management for species in areas where they are expected to advance, such as the northern limits of their ranges.
- Consider short-distance assistance to movements of species across artificial and natural impediments to migration, such as large rivers and areas dominated by intense agriculture.

Landscape Conservation Cooperatives and Climate Science Centers were established by the U.S. Department of the Interior to provide a philosophical foundation and organizational structure for improved coordination, cooperation, and partnerships in science and conservation across federal

and state agencies, tribes, conservation organizations, and universities within geographically defined areas (Austen 2011). The goal of these cooperatives is to “develop landscape-level strategies for understanding and responding to climate change impact” by identifying and coordinating scientific research priorities, identifying conservation needs, using science to inform conservation actions on the ground, and providing a national network of resource managers, interested citizens, and private organizations (Austen 2011). This coordinated, integrated approach to climate change research and conservation provides an important benefit to the knowledge base and effective planning for climate change mitigation management.

Nonconventional management options could also be considered, but warrant serious examination from a philosophical perspective that includes land management and conservation ethical concerns. For species at risk of extinction, *ex situ* gene conservation options may be appropriate (U.S. Department of Agriculture, Forest Service 2011). These could include captive breeding programs, zoo-based population maintenance, and storing gametes for potential future use. Because translocation of species to geographic areas or habitats (assisted migration) where they do not naturally occur has been proposed as a management option for addressing climate change, Hoegh-Goldberg et al. (2008) developed a decision framework for determining the need for assisted migration that is based on a species’ risk of extinction otherwise. However, the assumption that any and all means are acceptable to avoid extinction should be questioned. Assisted migration would effectively introduce nonnative species into intact wildlife communities, potentially causing unexpected repercussions for interspecific competition, genetics and natural selection, predator–prey interactions, and disease ecology (Ricciardi and Simberloff 2009). “Maverick” or unsupervised translocations and a “laissez-faire” scenario that allows extinctions to occur through inaction are the two opposite poles on a wide spectrum of approaches. What is needed is a comprehensive policy developed by conservationists, scientists, and land managers and adhered to by all (McLachlan et al. 2007).

CASE STUDIES

CASE STUDY 1: EASTERN WOODRAT DISTRIBUTION AT THE WESTERN EDGE OF RANGE IN TEXAS

Eastern woodrats (*Neotoma floridana*) are a common species associated with forests and woodlands throughout the Eastern United States, but can also be found in swamps and marshes as well as on the Great Plains, where they are associated with woody vegetation such as shelterbelts and fence lines (Beckmann et al. 2001; Wiley 1980). In Texas (Figure 11.1), they occupy the pineywoods, crosstimbers, and post oak savanna areas that occur across the eastern third of the state (Davis and Schmidly 2004). The range of eastern woodrats adjoins that of two similar woodrat species in central Texas; the three species likely diverged approximately 155,000 years ago (Edwards et al. 2001). White-throated woodrats (*Neotoma albigula*) are associated with deserts and semi-arid shrublands of the desert southwest (Davis and Schmidly 2004), whereas southern plains woodrats (*Neotoma micropus*) occur in brushlands of the semi-arid area between forests and the arid deserts to the west (Davis and Schmidly 2004). There is broad overlap in the ranges of these latter two species but in areas where they co-occur, habitat associations are likely the major factor maintaining their genetic integrity (Edwards et al. 2001). There is only slight overlap between the range of eastern woodrats and these other two species in Texas. The limited contact between the ranges of eastern woodrats and these other two species, along with differences in habitat requirements, likely play a major role in preventing hybridization (Edwards et al. 2001).

In the Mid-South, the Cross Timbers section of Oklahoma and Texas represents the historic western edge of continuous forests in the Southeast and the transition zone between eastern forests and the grasslands of the Great Plains (Chapter 10). Dominant ecotypes are woodlands and savannas of post oak (*Quercus stellata*) and blackjack oak (*Quercus marilandica*), intermixed with tallgrass prairies, all of which were historically maintained by frequent fire. Within this area, there is an east-west continuum; denser, moister forests with higher tree diversity occur in the east and drier forests with

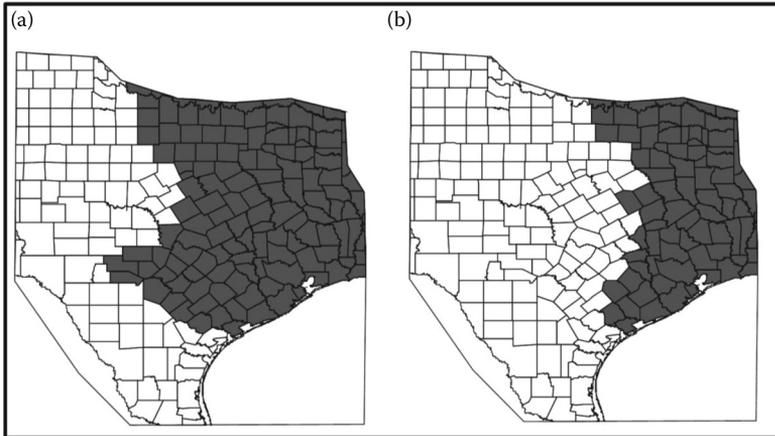


FIGURE 11.1 County-wide distribution of eastern woodrats in Texas: (a) currently, based on county records; and (b) by 2060, under the worst-case climate scenario—MIROC3.2 A1B—which predicts a mean annual reduction of 219-mm in precipitation. County records were derived from Davis and Schmidly (2004) with modifications from Birney (1976) and Edwards et al. (2001) the MIROC3.2 A1B climate scenario (McNulty et al. in press) combines the MIROC3.2 general circulation model with the A1B emissions storyline, representing moderate population growth and high-energy use.

lower tree density and lower diversity occur along the western border with the Great Plains (Rice and Penfound 1959). This east–west continuum parallels the decline in annual precipitation (Dyksterhuis 1957; Stahle and Hehr 1984), and suggests that precipitation determines the forest–grassland transition zone (Borchert 1950). Climate conditions likely become unfavorable for tree growth along the western frontier (Stahle and Hehr 1984). Reductions in precipitation and increases in temperature in forest–grassland ecotones such as the Cross Timbers could reduce dominance of forest species such as post oak in their westernmost areas. Changes in climate could affect various aspects of tree persistence, including survival, growth, and reproduction. For example, periods of drought correlate with the absence of post oak recruitment (Peppers 2004; Stahle and Cleaveland 1988). Furthermore, increased abundance of drought-tolerant eastern redcedar (*Juniperus virginiana*) or ash juniper (*Juniperus ashei*), along with altered fire regimes, may result in reductions in hard mast (acorns), which are a primary food for many forest small mammals, including eastern woodrats.

The Cross Timbers form the western boundary of the range of at least eight species of south-eastern small mammals that rely on forests, including the golden mouse (*Ochrotomys nuttalli*), woodland vole (*Microtus pinetorum*), southern flying squirrel (*Glaucomys volans*), eastern chipmunk (*Tamias striatus*), cotton mouse (*Peromyscus gossypinus*), eastern gray squirrel (*Sciurus carolinensis*), southern short-tailed shrew (*Blarina carolinensis*), and eastern woodrat. Increases in temperatures, coupled with reductions in precipitation could result in reductions in forest cover and increases in shrublands in the western reaches of these species' ranges, resulting in reductions of forest-associated small mammals in Oklahoma and Texas.

Since the end of the Pleistocene, the distribution of eastern woodrats has retreated eastward across the Southern United States as the climate became warmer and dryer in the southwest (Graham et al. 1996). During the late Pleistocene, eastern woodrats occurred in forest communities of the Southwest, including the Mexican state of Chihuahua, but that area is now a desert (Harris 1984; Van Devender et al. 1987). Consequently, increased temperature and reduced precipitation could cause more arid conditions where woodrats currently reside and associated habitat changes could be a driving factor that forces the distribution of this species to retreat farther eastward.

We compared the current climate along the western edge of the eastern woodrat's range in Texas with model projections of climate change for the area. We assumed that vegetation associated with

the current distribution of the eastern woodrat would respond to these climate changes and eastern woodrats would respond to changes in vegetation as they have done in the past. We also assumed that precipitation is the primary factor driving the vegetation associations (Chapter 10), and hence, the distribution of eastern woodrats.

We evaluated average annual temperature and precipitation for counties where the eastern woodrat occurs. We derived the current distribution of eastern woodrats from county records presented by Davis and Schmidly (2004), with modifications from Birney (1976) and Edwards et al. (2001). Using a logistic-regression model, we tested the accuracy of using temperature and precipitation to predict occurrence of eastern woodrats. Over a 10-year average (2000–2010) for precipitation and temperature, this model indicated that precipitation was highly informative for predicting distribution of eastern woodrats, and it explained 92% of the variation in the data. Including temperature in the model increased the percentage of variation explained by the model by only 2%. Therefore, to predict potential occurrence of eastern woodrats in the future based on changes in precipitation, we used a minimum 10-year average annual precipitation of 72.0 cm, which was based on the minimum annual precipitation of counties where the species has been documented. With this average annual precipitation value as the cutoff, 91% of the counties where the species is currently believed to occur were accurately classified but 11 counties along the western edge where eastern woodrats have not been documented were also included. Thus, this model overestimated the current distribution.

We predicted the future distribution of eastern woodrats under the best-case scenario (CSIROMK3.5 A1B) and the worst-case scenario (MIROC3.2 A1B) for precipitation from 2000 and 2060 (McNulty et al. in press). Based on model projections for the best-case scenario, average annual precipitation across the current Texas range of the eastern woodrat would decrease from 107.0 to 101.4 cm/year, an annual reduction of 5.6 cm of precipitation per year. Only two counties (Kerr and Gillespie) would fall below the minimum precipitation of 72.0 cm/year for potential retention of eastern woodrats under the best-case scenario.

Under the worst-case scenario, average precipitation across the current Texas range of the eastern woodrat would average 76.6 cm/year, a decrease of 30.4 cm (28%). By 2060, the range would retreat eastward approximately 160 km (Figure 11.1).

CASE STUDY 2: CAROLINA NORTHERN FLYING SQUIRREL AND HIGH ELEVATION SPRUCE–FIR FOREST IN THE SOUTHERN APPALACHIAN MOUNTAINS

The Carolina northern flying squirrel (*Glaucomys sabrinus coloratus*) is a federally endangered subspecies of the northern flying squirrel that occurs only in high-elevation forests of the Southern Appalachian Mountains. The range of this subspecies is limited to 17 counties in the mountainous areas of western North Carolina, eastern Tennessee, and southern Virginia, with potential habitat occurring in five additional counties (Trani et al. 2007). Carolina northern flying squirrels inhabit the transition zone (ecotone) between northern hardwoods and the high-elevation spruce–fir forests that are found on the highest peaks (Payne et al. 1989; Weigl et al. 1999). Elevations where they occur are usually above 1540 m (Weigl et al. 1999). They occupy cool and moist areas with cold winters, in forests that have a well-developed canopy, substantial ground cover, abundant moist downed wood, and organic substrates (Weigl 2007). Although this subspecies may use hardwood forests, spruce–fir forests and mixed spruce-fir/hardwood forests support growth of hypogeous mycorrhizal fungi (truffles), which are important to its diet (Loeb et al. 2000; Weigl et al. 1999).

Southern flying squirrels (*Glaucomys volans*) are often considered a major competitor (Weigl 2007). However, differences in habitat preferences, diets, and climatic tolerances between the two species suggest only limited competition (Bowen 1992; Bowman et al. 2005), and there is little evidence that competition is a significant factor in the conservation of Carolina northern flying squirrel (Weigl 2007). Southern flying squirrels are likely more sensitive to the cold, and they rely on stored hardwood nuts and seeds for overwinter survival (Bowman et al. 2005; Weigl 2007),

whereas Carolina northern flying squirrels are capable of surviving extremely cold and damp conditions that are lethal to southern flying squirrels and red squirrels (*Tamiasciurus hudsonicus*) (Weigl et al. 1999). However, the nematode parasite *Strongyloides robustus*, which is carried by southern flying squirrels and apparently causes few ill effects, can be detrimental to Carolina northern flying squirrels (Weigl 2007). Cold, high-elevation forests may only intermittently support *Strongyloides robustus* because of its sensitivity to cold (Weigl 2007). Therefore, effects of warming climate conditions that favor invasion of higher peaks by southern flying squirrels on the persistence of Carolina northern flying squirrels are unknown (Weigl 2007). Because of their relationship to high-elevation spruce–fir forests, we sought to determine the potential effects of rising temperatures on the persistence of Carolina northern flying squirrels in the Southern Appalachians.

Based on an analysis of the potential effects of climate change on persistence of spruce–fir forests in the Southern Appalachians, the best-case scenario (CSIROMK2 B2) predicted a 94% decrease in spruce–fir forests, and the worse-case scenario (MIROC3.2 A1B) predicted extirpation of these forests by the year 2060 (Chapter 10). Based on these estimates, the available habitat for Carolina northern flying squirrels will likely diminish significantly or may disappear by the year 2060.

Various unknowns affect estimates of spruce–fir persistence in the southern Appalachians; they include current estimates of minimal elevation where spruce–fir can persist, estimates of lapse rates (the rate of temperature change with increasing elevation), changes in precipitation, and pockets of cold air that may persist at significantly lower elevations (Chapter 10). Furthermore, the extent to which Carolina northern flying squirrels may use northern hardwoods forests is unclear. The closely related Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*) occurs in northern hardwood forests with little or no spruce, but with eastern hemlock (*Tsuga canadensis*) present (Ford et al. 2007; Menzel et al. 2006) and one population of Carolina northern flying squirrels also occurs in hemlock–hardwood habitats (Hughes 2006). However, spruce forests contain a greater abundance of hypogeous fungi than northern hardwoods, which provides greater food abundance (Loeb et al. 2000) and it is unknown if these hardwood forests are sinks or sources.

Various other factors that could be detrimental to Carolina northern flying squirrels may come into play over time as climate changes. In particular, potential interactions with southern flying squirrels may be problematic. Effects of changing climate on the ability of southern flying squirrels to colonize higher elevation areas is unknown, but potential negative interactions between these two species may occur, including transmission of *Strongyloides robustus*. Recent evidence suggests that hybridization may occur between northern and southern flying squirrels in the northeastern states and in southern areas of Canada, where southern flying squirrels have recently expanded their range in response to warming temperatures (Garroway et al. 2010).

CASE STUDY 3: SUMMER MATERNITY RANGE OF INDIANA BATS

The Indiana bat (*Myotis sodalis*) ranges throughout northeastern and midwestern states, as well as parts of the Southeast. During winter, Indiana bats hibernate in cold caves and mines in 19 states from Massachusetts to Tennessee, with the most important hibernacula ($\geq 10,000$ bats) occurring primarily in the Midwest and Southeast (U.S. Department of the Interior Fish and Wildlife Service 2007). In the spring, female bats migrate up to 575 km to their summer range (Winhold and Kurta 2006) where they form maternity colonies that usually contain < 100 individuals in the snags of various tree species (Menzel et al. 2001).

In 1967, the Indiana bat was listed as endangered by the U.S. Department of the Interior Fish and Wildlife Service based on destruction and degradation of hibernacula; disturbance during hibernation; and loss and degradation of summer maternity habitat, migratory habitat, and swarming sites (U.S. Department of the Interior Fish and Wildlife Service 2007). Despite protection and recovery efforts, populations continued to decline through the year 2000 when they began to increase. However, beginning in 2007 this trend reversed with the emergence of white-nose syndrome, a disease that affects bats during hibernation (Turner et al. 2011).

Until the late 1990s, the primary summer maternity range of the Indiana bat was assumed to be in the Midwest (U.S. Department of the Interior Fish and Wildlife Service 2007). However, with increased netting efforts and possible range shifts, a number of colonies were found in other areas including the mountainous areas of western North Carolina and eastern Tennessee (Britzke et al. 2003), Pennsylvania (Butchkoski and Hassinger 2002), New York (Britzke et al. 2006; Watrous et al. 2006), and New Jersey, Maryland, and Virginia (U.S. Department of the Interior Fish and Wildlife Service 2007). In contrast, no maternity colonies were found in Alabama even though Indiana bats have been known to hibernate there (Harvey 2002). Nor have they been found in the mountains of South Carolina despite considerable netting in these areas over the past decade (Loeb, S.C. [N.d.] Unpublished data. On file with Southern Research Station, Upland Hardwood Ecology and Management, 233 Lehotsky Hall, Clemson University, Clemson, SC 29634).

In general, female Indiana bats migrate north from their hibernacula to their summer maternity colonies while males often remain in or close to hibernacula during the summer (Gardner and Cook 2002; U.S. Department of the Interior Fish and Wildlife Service 2007). During summer, Indiana bats are not dependent on any particular tree species or forest type for roosting or foraging (Menzel et al. 2001), and suitable forest habitat is not a limiting factor for their potential range—the maximum known migratory distances from the highest priority hibernacula (Gardner and Cook 2002). This suggests that there may be some climatic factor such as temperature or precipitation that restricts Indiana bat summer distribution (Brack et al. 2002).

If temperature, precipitation, or the combination of both limits the summer maternity range of Indiana bats, the result may be a shift in the summer range in response to global climate change. To determine that likelihood, we modeled the potential maternity range of Indiana bats under four climate change scenarios—CSIROMK3.5 A1B, CSIROMK2 B2, HadCM3 B2, and MIROC3.2 A1B (McNulty et al. in press). We were especially interested in understanding whether the Southeast, particularly the Southern Appalachian Mountains, might become a more important area for Indiana bat maternity colonies in the future.

We used a species distribution modeling approach (MAXENT, Phillips et al. 2006) to test whether global climate change may influence the maternity range of Indiana bats. First, we modeled the current summer distribution of Indiana bats based on temperature and precipitation during the maternity season (May to August). Based on the outcome of this model, we modeled the distribution of Indiana bats for the period 2041–2060 using the forecasted temperature and precipitation for each of the four climate scenarios. Locations of maternity colonies were provided by the Fish and Wildlife Service and we used the county center for each occurrence record. The historical climate data were at the 5-arc minute resolution (~10 km grid size) and based on PRISM climatology from 1970 to 1999 (Coulson and Joyce 2010). Projected temperature and precipitation data for each climate scenario were obtained from Coulson et al. (2010a, 2010b). We used elevation, the average maximum summer temperature (May through August), and the precipitation for each summer month as input for our models. We included the entire Eastern United States to determine the potential importance of the Southeast for Indiana bats under the various scenarios.

The models predicted that suitable habitat for Indiana bats will decrease considerably under all climate scenarios during the 2041–2060 period (Figures 11.2 and 11.3). The western portion of the range, which is now considered to be the heart of the summer range, is forecasted to be unsuitable under all climate scenarios (Figure 11.2). Areas that are predicted to be highly suitable are in the Southern Appalachians (scenarios CSIROMK3.5 A1B, CSIROMK2 B2, and HadCM3 B2) and the Northeast. The southeastern states (Kentucky, North Carolina, South Carolina, Tennessee, Virginia, and Georgia) are forecasted to contain a greater proportion of suitable habitat under future conditions than they do now (Figure 11.3).

Our results suggest that warmer climates in the western portion of the Indiana bat's current summer range will force ranges to shift farther north and east as females seek suitable climatic conditions for maternity colonies. Thus, the Northeast and Southern Appalachians could become the heart of the future summer range. However, white-nose syndrome, which is associated with

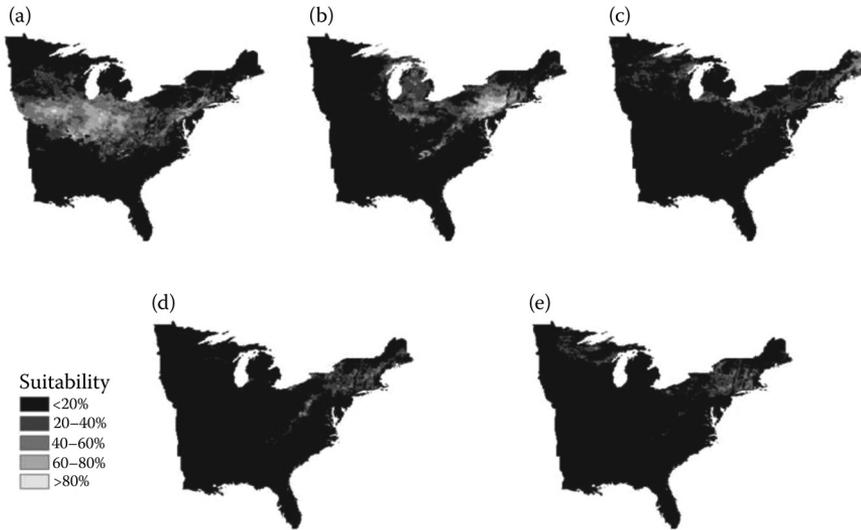


FIGURE 11.2 Suitability of summer maternity habitat of Indiana bats based on historical data, 1970–2000, and predictions from four climate change scenarios, 2040–2060: (a) historical baseline; (b) CSIROMK3.5 A1B prediction of minimal 1.15°C warming combined with moderately increasing 23-mm precipitation; (c) CSIROMK2 B2 prediction of substantial 1.68°C warming and 52-mm drying; (d) HadCM3 B2 prediction of moderate 1.35°C warming and 22-mm drying; and (e) MIROC3.2 A1B prediction of the most severe 2.35°C warming and 219-mm drying. The CSIROMK3.5 A1B, CSIROMK2 B2, HadCM3 B2, and MIROC3.2 A1B climate scenarios (McNulty et al. in press) each combines a general circulation model with an emissions storyline—the A1B storyline representing moderate population growth and high-energy use and the B2 representing lower population growth and energy use.

a fungus that grows primarily in cold temperatures within hibernacula (Gargas et al. 2009), has resulted in the death of approximately 72% of the Indiana bats that hibernate in the Northeast since 2006 (Turner et al. 2011). Studies using radiotelemetry and stable isotopes suggest that female Indiana bats hibernating in the Northeast do not migrate great distances from hibernacula to maternity colonies (Britzke et al. 2006, 2012). Therefore, even though our models forecasted that the Northeast will be important for Indiana bat maternity colonies, low numbers of bats surviving white-nose syndrome in the Northeast may retard growth of summer populations.

Although we did not model winter climate, mortality from white-nose syndrome could potentially be lower in southern hibernacula due to shorter milder winters. Consequently, the forests of the Southern Appalachians may become more important for survival of this species than our models predicted, and maintaining and restoring suitable maternity habitat for Indiana bats in the Southern Appalachians may be critical. Unlike Indiana bats in the rest of the range, Indiana bat maternity colonies in the Southern Appalachians roost almost exclusively in conifer snags in open pine–oak (*Pinus* spp–*Quercus* spp) habitats (Britzke et al. 2003; O’Keefe, J.M., Loeb, S.C. [n.d.] Unpublished data. On file with Southern Research Station, Upland Hardwood Ecology and Management, 233 Lehotsky Hall, Clemson University, Clemson, SC 29634), but there is evidence that pine–oak habitats are declining in this area (Lafon et al. 2007). Therefore, conservation and restoration of mature pine–oak habitats in the Southern Appalachians may be necessary to conserve and recover Indiana bats.

CASE STUDY 4: BLACK-THROATED BLUE WARBLERS IN THE APPALACHIAN-CUMBERLAND HIGHLANDS

Black-throated blue warblers (*Setophaga coeruleascens*) winter in Caribbean locales. During breeding season they are restricted to the Southern Appalachian Mountains, where they are most abundant at elevations over 1050 m but can occur down to 750 m (Hamel 1992). The preferred breeding habitat

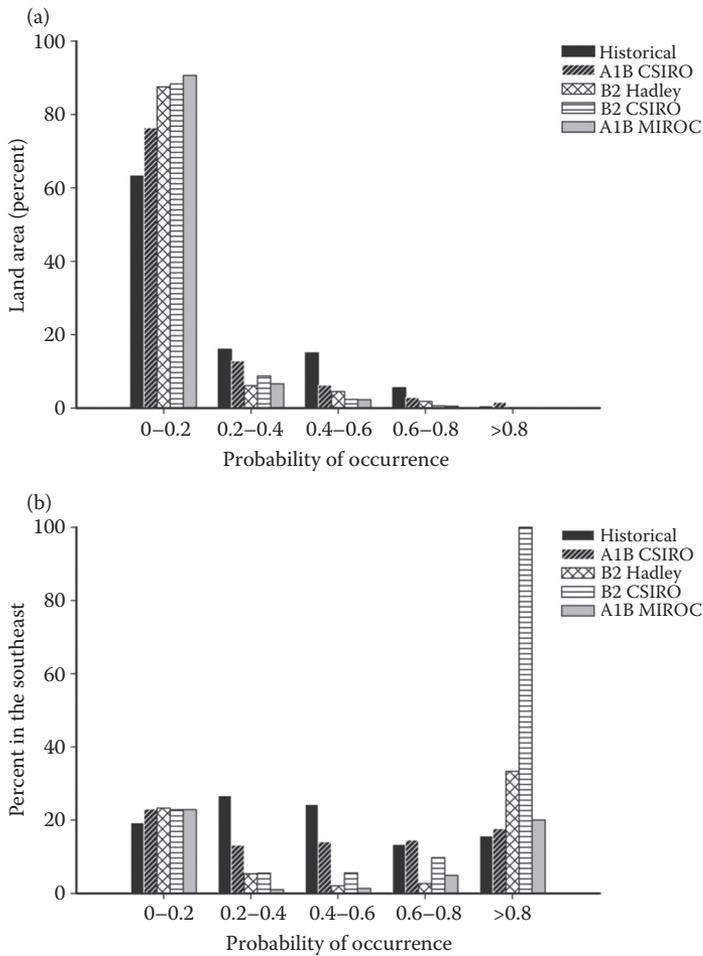


FIGURE 11.3 Land area in five habitat suitability classes for Indiana bats from 1970 to 1999 and under four climate scenarios for 2041–2060: (a) in the Eastern United States and (b) percent of the area that falls within the southeastern United States. The CSIROmk3.5 A1B, CSIROmk2 B2, HadCM3 B2, and MIROC3.2 A1B climate scenarios (McNulty et al. in press) each combines a general circulation model with an emissions storyline—the A1B storyline representing moderate population growth and high-energy use and the B2 representing lower population growth and energy use.

contains large, mature deciduous or mixed deciduous/coniferous forests or spruce–fir forests at higher elevations with a moderate or dense understory (Hamel 1992). Arthropod prey abundance can be affected by weather, which in turn influences nestling survival (Nagy and Holmes 2005; Sherry and Holmes 1992) and the likelihood of double brooding (Sillett et al. 2000). Hence, weather can indirectly affect the future viability of this species. Weather conditions can influence predator abundance and predation rates (Rodenhous and Holmes 1992), which can in turn affect black-throated blue warblers. Because long-term weather patterns also affect habitat suitability, changes in climate can influence warbler settlement rates, spatial distribution of territories, and abundance. Reproductive success, recruitment rates, and future population size may all eventually reflect changes in weather patterns.

Rodenhous (1992) suggested that increased precipitation could result in lower reproduction, which may be slightly offset by warmer temperatures with subsequent increases in insect prey and a longer breeding season. He speculated that nest predation could be reduced during periods of higher precipitation, another small potential offset for the predicted negative effects of precipitation on

warbler productivity. He also suggested that an annual increase of 2°C could lead to lower reproductive rates and reduced recruitment for this species, but warming temperatures coupled with lower precipitation (≥10%) could result in increased reproduction. Thus, both the change in temperature and in precipitation must be considered before making predictions. Further, not only must the direction of change in precipitation and temperature (increasing or decreasing) be considered, but the intensity of the change must be considered as well.

This study used predicted changes in precipitation and temperature (2000–2060) for the Appalachian-Cumberland highlands under four climate change scenarios—CSIROMK3.5 A1B, MIROC3.2 A1B, CSIROMK2 B2, and HadCM3 B2 (McNulty et al. in press) to predict changes in black-throated blue warbler abundance and range based on temperature and precipitation thresholds that Rodenhouse (1992) determined would affect black-throated blue warbler nest productivity and recruitment. After estimating the 10-year average values for precipitation and temperature for the years 2000–2010, (the baseline), the average value for precipitation and temperature in 20-year intervals was calculated (2000–2020, 2020–2040, and 2040–2060) and these averages were summed to calculate the overall estimate for the entire period.

Under the CSIROMK3.5 A1B model, average annual temperature is predicted to increase by about 1.2°C (~9%) and annual precipitation to increase by about 83.5 mm (~6%) (Table 11.1). Rodenhouse (1992) found that higher temperatures and an average increase in precipitation <10% would result in black-throated blue warblers compensating for potential reductions in reproductive output through increased insect abundance and a longer breeding season, allowing more pairs to raise second broods. Therefore, under the CSIROMK3.5 A1B climate change scenario, black-throated blue warblers would likely maintain their annual reproductive rate.

TABLE 11.1
Predicted Change in Temperature and Precipitation under Four Climate Change Scenarios (2000–2060) Compared to a 10-year Baseline Average (2000–2010), with Probable Consequences to the Black-Throated Blue Warbler in the Appalachian-Cumberland Highlands

Scenario ^a	Temperature Change		Precipitation Change		Likely Change in Warbler Reproduction
	(°C)	(%)	(mm)	(%)	
MIROC3.2 A1B ^b	3.25	23	-175.27	14	Although compensation is possible, such a high temperature increase could override the potential benefits of reduced rainfall and result in lower productivity
CSIROMK3.5 A1B ^c	1.17	9	83.46	6	This moderate temperature increase and <10% precipitation increase offers the potential for compensation and minimal change in productivity
CSIROMK2 B2 ^d	1.98	14	71.43	5	This high temperature increase is likely to result in reduced productivity
HadCM3 B2 ^c	1.71	12	17.56	13	This moderate temperature increase and >10% precipitation increase is likely to result in loss of productivity

^a The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 climate scenarios (McNulty et al. in press) combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high energy use, B2 representing lower population growth and energy use).

^b Most severe warming (+2.35°C) and drying (∓219 mm).

^c Minimal warming (+1.15°C) and moderately wetter (+23 mm).

^d Substantial warming (+1.68°C) and drying (∓52 mm).

Under the MIROC3.2 A1B scenario, average annual temperature is predicted to increase by approximately 3.3°C (~23%) and annual precipitation to decrease by 175.3 mm (~14%) (Table 11.1). Rodenhouse (1992) determined that a 10–20% precipitation decrease would result in increased productivity and a higher proportion of pairs producing second broods, but that a 2°C temperature increase would result in lower productivity and recruitment. The 3.3°C increase predicted by the MIROC3.2 A1B is over 50% higher than the harmful threshold determined by Rodenhouse et al. (2008). Although the benefits of reduced precipitation could override the harm caused by the higher temperatures in some yet-undetermined way, the magnitude of the temperature increases predicted by MIROC3.2 A1B would likely result in reduced reproduction.

The CSIROMK2 B2 model projected an average annual temperature increase of about 2.0°C (~14) and an annual precipitation decrease of approximately 71.4 mm (~5%) (Table 11.1).

Alone, the predicted decrease in precipitation could increase reproduction; however, the predicted temperature increase is likely to result in reduced productivity and lower recruitment. Because this temperature increase is substantial and the decrease in precipitation is only 5%, the CSIROMK2 B2 climate change scenario would result in reduced reproduction.

For the HadCM3 B2 model, annual temperature is estimated to increase 1.7°C (~12% over baseline), and annual precipitation to increase by 175.6 mm (~13% over baseline) from 2000 to 2060 (Table 11.1). Rodenhouse (1992) found that increased average precipitation greater than 10% over his baseline would likely result in reduced black-throated blue warbler productivity, and that second broods were unlikely to compensate for the reduction. A temperature increase of 1.7°C is higher than the increase that Rodenhouse et al. (2008) predicted would lower productivity and reduce recruitment. Hence, the wetter, hotter projections of the HadCM3 B2 climate change scenario would result in lower reproductive success.

In summary, black-throated blue warbler productivity and recruitment in the Appalachian-Cumberland highlands are unlikely to be affected by the relatively minor changes in temperature and precipitation (2000–2060) predicted under the CSIROMK3.5 A1B climate change scenario. In contrast, reductions in productivity and recruitment are likely under the MIROC3.2 A1B, CSIROMK2 B2, and HadCM3 B2 climate change scenarios, where changes in temperature or precipitation are greater (Table 11.1). These evaluations corroborate predictions of models based on other, similar climate-change scenarios that show a reduction in abundance and a range contraction of black-throated blue warblers in the Appalachian-Cumberland highlands where changes in temperature and (or) precipitation are substantial (Matthews et al. 2007).

CASE STUDY 5: SYNERGISTIC EFFECTS OF INVASIVE SPECIES AND CLIMATE CHANGE ON AQUATIC AMPHIBIANS

Leaf litter from the nonnative, invasive Chinese tallow tree (*Triadica sebifera*) is known to affect the survival of some aquatic amphibians, likely by lowering dissolved oxygen and pH (Cotten et al. 2012; Leonard 2008). Cotten et al. (2012) reported that the effects from Chinese tallow tree leaf litter may be more severe on winter breeding amphibians than on species that breed later in the spring and summer. A likely reason for this difference in effect is that impacts on water chemistry from the leaf litter diminish over time; therefore, earlier breeding amphibians may be exposed to extremely low dissolved oxygen and pH levels, whereas later breeders will encounter water conditions with more favorable oxygen and pH levels. Local weather conditions can play a significant role in the timing of amphibian breeding activity (Saenz et al. 2006). Thus, climate change may play a significant role in the interactions of amphibians and the Chinese tallow tree.

The goal of this study was to understand the interaction between the Chinese tallow tree and southern leopard frog (*Lithobates sphenoccephalus*) larvae when influenced by changes in climate. In Texas, Chinese tallow tree leaves generally fall from November through late December (D. Saenz, research wildlife biologist, Southern Pine Ecology, Southern Research Station, U.S. Forest Service, 506 Hayter St., Nacogdoches, TX 75961, personal observation). When leaves enter a wetland, they

are quickly leached of tannins and other soluble materials, which changes the water chemistry. During drought years, leaves may remain unleached until rainfall. Southern leopard frogs breed from November through March, depending primarily on temperature (Saenz et al. 2006). This variation in the timing of leaf fall from tallow tree and the phenology of leopard frog breeding can produce a variety of scenarios, ranging from relatively simultaneous breeding and leaf fall to a four-month lapse between the two. Additionally, rainfall can play an important role in the timing of leaf leaching and breeding (Saenz et al. 2006).

To determine the relationships between invasive species, amphibian survival, and climate change, we raised leopard frog tadpoles in five *in situ* treatments, each treatment representing a different stage of tallow tree leaf decomposition in water. Water chemistry measurements were taken throughout the experiment. Tadpoles in treatments with shorter decomposition times had significantly lower survival and significantly smaller tail muscles. Treatments also had significant differences in water chemistry, supporting the hypothesis that the phenology of amphibian breeding and the timing of leaf leaching are important factors affecting tadpole survival. Because leopard frog breeding phenology and the timing of leaf leaching is regulated by precipitation and temperature (Saenz et al. 2006), climate change would have a profound and predictable impact on tadpole survival in the presence of Chinese tallow trees. Below are some general predictions of how weather may impact amphibian survival.

Hot and wet winter: Early breeding and early leaching of tallow tree leaves. This is probably the worst-case scenario because fresh leaves would fall in the water about the same time that breeding occurs. Dissolved oxygen and pH levels would be very low, causing reduced survival in tadpoles.

Hot and dry winter: Late breeding and late leaching. This scenario is not much of an improvement over hot and wet winters. Although breeding would be delayed because of a lack of water, so would leaf leaching. With the eventual onset of significant rainfall, leaching and breeding would take place about the same time. The likely result would be low dissolved oxygen and pH levels causing reduced survival in tadpoles. However, the potency of the leaves would vary because they would be exposed to the elements for some time before being inundated with water.

Cold and dry winter: Late breeding and late leaching. This scenario is similar to a hot and dry winter because leaching and breeding would be delayed. The minor difference is that both rain and temperature would delay breeding. The likely result would be low dissolved oxygen and pH levels, causing reduced survival in tadpoles.

Cold and wet winter: Late breeding and early leaching. This is the best-case scenario because breeding would take place long after the cessation of leaf leaching. In a wet winter, the leaves would leach soon after leaf drop, but in the event of low temperatures, breeding would not take place at all. Compared to the other scenarios, dissolved oxygen and pH levels would be higher when tadpoles are present, thereby increasing survival.

Shifts in the breeding phenology of many species are believed to have occurred as a result of a changing climate (Parmesan 2007). The major threat to southern leopard frogs and other winter-breeding amphibians that occur in areas invaded by Chinese tallow trees, in a warming climate, is a shift in breeding phenology. Earlier breeding by amphibians would mean greater impacts of the deleterious effects of the Chinese tallow tree.

Because shifts in amphibian breeding phenology are likely out of the control of managers, the most effective way to address negative interactions between Chinese tallow trees and early-breeding aquatic amphibians is to focus on the invasive species side of the equation. An aggressive control program for Chinese tallow tree, especially in or near wetlands, would lessen the interactions between this nonnative invasive species and native amphibians.

CASE STUDY 6: HYDROLOGICAL PATTERN IN AN EPHEMERAL POND—IMPLICATIONS FOR AMPHIBIANS

The timing and duration of wetland hydroperiod (duration of saturation), water and air temperature, and the timing and amount of precipitation have been correlated with breeding behavior,

reproductive success, and declines of some amphibian populations (Carey and Alexander 2003). Ephemeral ponds are especially vulnerable to potential changes in climate that could affect water levels, hydroperiods, and the timing of depth changes that are critical for successful amphibian reproduction. Climate change could alter weather patterns, such as the amount of rainfall and temperature, thereby affecting water quality, depth, and the timing and duration of hydroperiods in ephemeral wetlands. Life history attributes, such as dispersal ability, expected life span, breeding cues, and rates of larval development, along with landscape dynamics, would likely mediate amphibian population trends and extinction risk among species.

Small, isolated, ephemeral wetlands play a critical role in sustaining the biological diversity of entire ecosystems and landscapes (Semlitsch and Bodie 1998). For example, one study reported 75,644 juvenile amphibians of 15 species metamorphosed from a 1-ha pond in South Carolina (Pechmann et al. 1989). Several amphibian species, including species of conservation concern such as the Florida gopher frog (*Lithobates capito*) and striped newt (*Notophthalmus perstriatus*), reproduce in fish-free temporary ponds and inhabit surrounding uplands as adults. Many species rely on specific weather events and hydroperiod characteristics for breeding cues and successful metamorphosis.

We used long-term (March 1994–August 2011) monthly measurements of temperature, rainfall, and water depth of an isolated pond to develop a predictive model of pond depth and hydroperiod, and applied the model to the CSIROMK3.5 A1B (best-case) and MIROC3.2 A1B (worst-case) climate change scenarios (McNulty et al. in press) over a 60-year period. Our study area, located in the longleaf pine-wiregrass (*Pinus palustris*–*Aristida stricta*) sandhills of the Ocala National Forest, consisted of a 0.1-ha isolated, ephemeral pond that was part of a long-term study of amphibian use of ephemeral ponds.

We fit a linear model (PROC REG; SAS Institute Inc. 2000) to the time-series data with pond depth as our dependent variable. We included the total rainfall for the current month, the combined total rainfall for the current and prior month, the date (month/year), the minimum and maximum monthly temperatures, the midrange monthly temperature, and the previous number of months during which the pond was dry. We also included a first-order lag of pond depth (pond depth one month prior) as an independent variable; this accounted for autocorrelation in the data and incorporated seasonal trends.

Our model indicated a close correlation ($R^2 = 0.89$) between rainfall and pond depth the prior month (lag), which were the only significant predictors of pond depth:

$$PD = -10.2 + 0.922PDL + 1.05,$$

where PD is pond depth, PDL is pond depth lag (cm), and R is rainfall (cm). The model also calculated belowground pond depths; although we did not measure belowground water levels in the field (dry ponds were recorded as 0 cm depth), we assumed that negative pond depths calculated in our model were acceptable because the linear model was reasonably accurate.

Predictive models indicated that changes to pond hydrological patterns could have more serious impacts on amphibians under the MIROC3.2 A1B climate change scenario than under the CSIROMK3.5 A1B scenario (Figure 11.4). Under CSIROMK3.5 A1B, the number of hydroperiods (about six per decade) and their average duration (about 14 months during most decades) would not change dramatically (Table 11.2, Figure 11.4). In contrast, the MIROC3.2 A1B climate scenario predicted only five or six hydroperiods per decade for the first 30 years (2000–2029) and only one or two hydroperiods per decade for the last 30 years (2030–2059) modeled (Table 11.2, Figure 11.4). Most hydroperiods were predicted to be very shallow (<5 cm), or short-lived, or both. Hydroperiod length ranged from 1 to 18 months during the first 40 years, and from 1 to 1.5 months during the last 20 years.

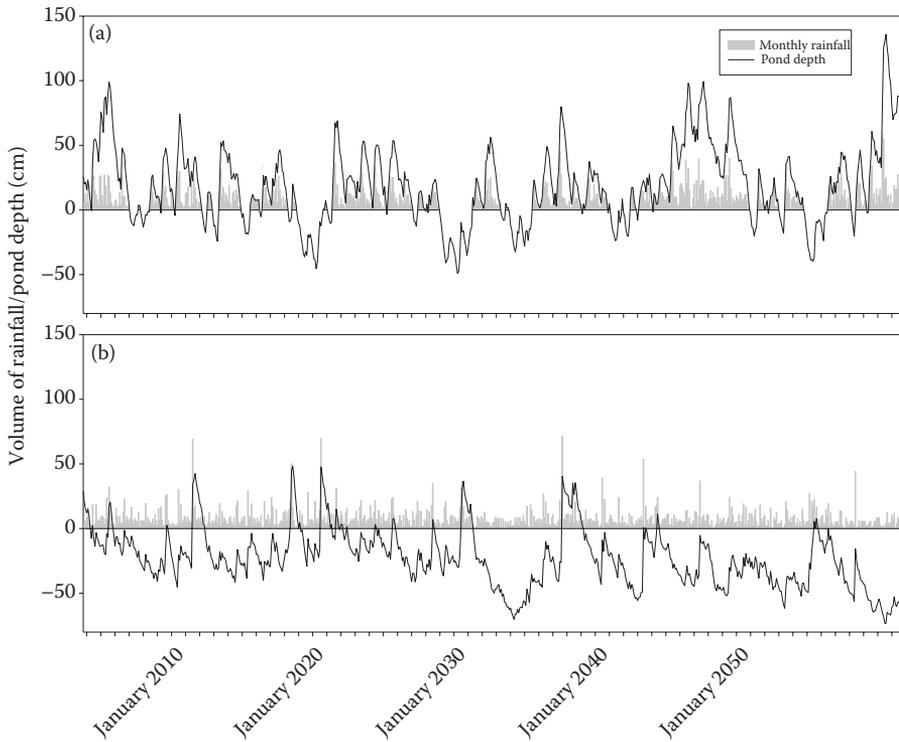


FIGURE 11.4 Predicted water depth and hydroperiod (in months) for an isolated, ephemeral pond on the Ocala National Forest in Florida, based on precipitation data for two climate change scenarios: (a) CSIROMK3.5 A1B prediction of minimal warming with moderately increasing precipitation; and (b) MIROC3.2 A1B prediction of more severe warming and drying. The CSIROMK3.5 A1B and MIROC3.2 A1B climate scenarios (McNulty et al. in press) combine the CSIROMK3.5 and MIROC3.2 general circulation model with the A1B emissions storyline, representing moderate population growth and high-energy use.

Our data represent only one of many isolated, shallow ephemeral ponds in the Ocala National Forest and throughout much of the Coastal Plain. Nonetheless, the severity of altered hydrological pattern under the MIROC3.2 A1B scenario suggests that pond breeding amphibians would be severely impacted at a much greater landscape level. Dispersion of species that are primarily aquatic, such as cricket frogs (*Acris* spp) and pig frogs (*Lithobates grylio*), would likely be severely reduced across the landscape and they could not persist at ponds that were dry most of the time. Most amphibian species that rely on ephemeral ponds for breeding would not live long enough to exploit these less frequent hydroperiods, suggesting that their populations would dramatically shrink or become locally extinct. Even if some species could breed during those infrequent hydroperiods, successful recruitment of juveniles would be unlikely because most hydroperiods would not be sufficiently long-lasting for tadpoles to complete their development to metamorphosis. Thus, Florida gopher frogs, pig frogs, southern leopard frogs (*Lithobates sphenoccephalus*) and bullfrogs (*Lithobates catesbeianus*)—all commonly captured at our study pond—would be unlikely to persist, as they have long larval development periods. Populations of summer breeders, such as oak toads, would also likely shrink or become locally extinct because summer hydroperiods under the MIROC3.2 A1B scenario would be rare. The relative abundance of species would likely shift toward species such as the spadefoot toad (*Scaphiopus holbrookii*) that live longer and have fast-developing larvae. However, even persistence of the spadefoot toad is uncertain because the interval between suitable hydroperiods could exceed their lifespan.

TABLE 11.2
Number and Duration (Average and Range) of Actual (2001–2009) and Predicted (2001–2059) Hydroperiods per Decade in an Isolated, Ephemeral Sinkhole Pond in the Ocala National Forest, Based on Two Climate Change Scenarios

Decade	Hydroperiods (Historical)			Hydroperiods (Predicted ^a)					
	Number	Average		CSIROMK3.5 A1B ^b			MIROC3.2 A1B ^c		
		Duration (months)	Range (months)	Number	Average Duration (months)	Range (months)	Number	Average Duration (months)	Range (months)
2001–2009	6	12.0	1 to 51	6	13.5	2 to 32	5	3.6	1 to 6
2010–2019	N/A	N/A	N/A	8	8.6	4 to 22	5	5.0	1 to 10
2020–2029	N/A	N/A	N/A	6	14.0	1 to 39	6	3.3	1 to 11
2030–2039	N/A	N/A	N/A	6	14.0	1 to 42	1	18.0	none
2040–2049	N/A	N/A	N/A	4	26.3	5 to 82	2	1.5	1 to 2
2050–2060	N/A	N/A	N/A	6	14.5	2 to 31	2	1.0	1 to 1

Note: N/A means no data collected for the date range.

^a The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 climate scenarios (McNulty et al. in press) combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high energy use, B2 representing lower population growth and energy use).

^b Minimally warmer (+1.15°C) and moderately wetter (+23 mm).

^c Most severe warming (+2.35°C) and drying (–219 mm).

CONCLUSIONS

Uncertainty associated with climate-change prediction limits the ability of land managers to develop specific management plans for particular species. Also daunting are knowledge gaps in the basic natural history and ranges of tolerance of many wildlife species. Further, the ecological complexities of interactions among species that would occur when vegetation and wildlife communities reassemble are unknown. Clearly, research is needed to fill these gaps, thus enabling land managers and planners to develop strategic plans with a more comprehensive understanding of climate change and likely responses of plants and animals. Proactive strategies include systematic and long-term monitoring across large areas. Programs for monitoring forest status and trends, streams, and air quality at a national level are in place (U.S. Department of Agriculture Forest Service 2011), but programs for systematic long-term monitoring of wildlife are fewer and often piecemeal. The North American Breeding Bird Survey, established in 1966, and the Christmas Bird Count are important efforts that are already proving invaluable in assessing current trends in bird populations (Rodenhous et al. 2009). A North American bat monitoring program is currently being developed. Similar long-term programs are needed for monitoring amphibians, reptiles, mammals, and other indicator species at a regional or national level. Targeted monitoring of potentially vulnerable populations in specific locales is important for early detection, assessment, and rapid response when climate change threatens population trends or population health (U.S. Department of Agriculture Forest Service 2011). Effectiveness monitoring is also essential to assess the results of management activities designed to increase resilience, reduce stressors, or otherwise benefit potentially vulnerable wildlife species (U.S. Department of Agriculture Forest Service 2011). Most importantly, monitoring can be used as an “early alert” system for adaptive management, so that land managers can be poised and flexible in implementing mitigation measures when needed. Finally, tried and true conservation practices

such as restoring land connectivity and ecosystem health would benefit wildlife and help to mitigate multiple stressors on population health regardless of the uncertainties associated with climate change predictions.

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