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Developing a Decision-Support Process for Landscape Conservation Design

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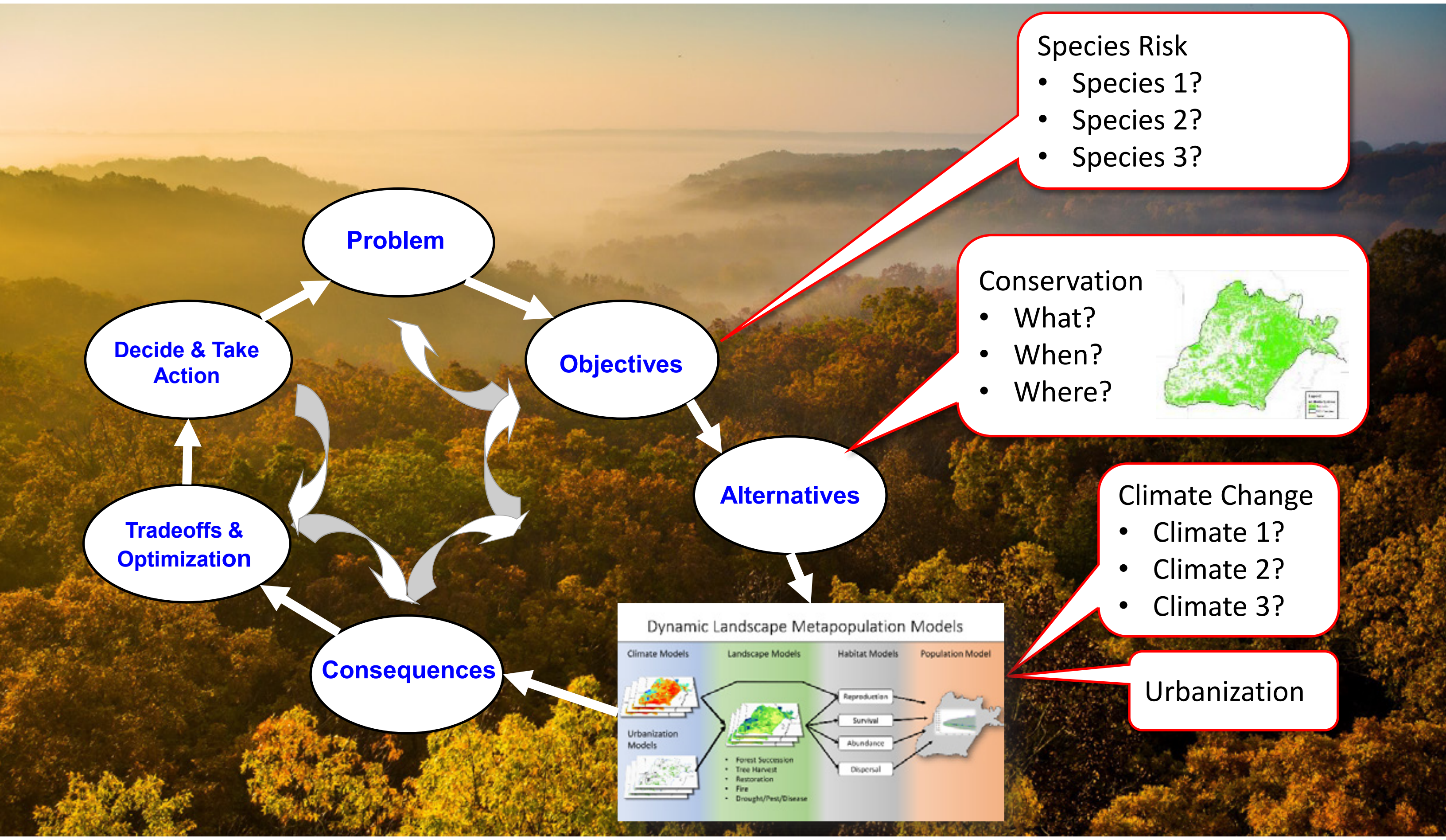

Forest Service Northern Research Station General Technical Report NRS-190 July 2019

Abstract

Planning for sustainable landscapes is hampered by uncertainty in how species will respond to conservation actions amidst impacts from landscape and climate change. Planning decisions, including tradeoffs among competing species objectives, are complex. We developed a decision-support framework that integrates dynamic-landscape metapopulation models (DLMPs) and structured decision making (SDM) to help guide landscape conservation design. With this framework, we demonstrated that planning for viable populations across broad scales can be achieved under global change. Furthermore, the integration of DLMPs with SDM enabled decisions to be more objective and transparent, and thus, more defensible.

Cover Art

Illustration of integrating dynamic-landscape metapopulation models with structured decision making. Background photo of Ozark forest at the Baskett Wildlife Research and Education Center by Kyle Spradley, MU College of Agriculture, Food & Natural Resources, Flickr Creative Commons.

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SUMMARY

The Gulf Coastal Plains and Ozarks landscape conservation cooperative (GCPO) is a regional collaboration across agency and ownership boundaries to conserve sustainable landscapes in the face of global change. Planning for sustainable landscapes is hampered by uncertainty in how species will respond to conservation actions amidst impacts from landscape and climate change, especially when those impacts are also uncertain. Conservation is also made difficult by the complexities of the planning decisions, including tradeoffs among competing species objectives. We developed a decision-support framework that integrates dynamic-landscape metapopulation models (DLMPs) and structured decision making (SDM) to help guide landscape conservation design. This framework allowed a team of partners to choose among scenarios for habitat restoration that best met desired endpoints for focal wildlife species in the GCPO's Ozark Highlands region under climate change and urbanization.

Through the framework, the Ozark Highlands team identified focal species to represent priority ecosystems, designed alternative scenarios, and used DLMPs to model the consequences of each, given concurrent impacts of climate and landscape change. The overall modeled impact of restoration on focal species was positive and presented evidence to support landscape conservation design. Despite the general effectiveness of restoration, species-specific responses to individual scenarios varied through interactions with landscape change processes such as urbanization, climate change, the demographic processes affecting each species. The planning team identified a scenario that targeted 1.2 million ha of restoration across private and protected lands, based on predicted future landscape conditions that best reduced the average risk across species. With the development of this framework, we demonstrated that planning for viable populations across broad scales can be achieved under global change. The integration of DLMPs with SDM enabled decisions to be more objective and transparent, and thus, more defensible. This framework has the potential to overcome many of the uncertainties and complexities that are inherent in the process of long-term, large-scale conservation planning.

CONSERVATION UNDER GLOBAL CHANGE

Global change poses serious issues for biodiversity conservation. Many species are now at a far greater risk of extinction than in the recent geological past (Fischlin et. al. 2007). Although landscape change and habitat loss remain the main drivers of present-day species extinction (Sodhi et al. 2009), climate change is projected to become equally or more important in the coming decades as it interacts synergistically with these threats (Brook et al. 2008). Threats such as climate change and habitat loss operate at regional to global scales (Fahrig and Merriam 1994, Lambeck and Hobbs 2002). Similarly, persistent populations depend on demographic processes that occur over a range of scales (Bonnot et al. 2011, Faaborg et al. 2010, Pulliam 1988). As a result, conservation under global change will need to proactively address future threats to a range of biological and ecological processes, across large landscapes, amidst great uncertainty (Aycrigg et al. 2016, Parrot and Meyer 2012).

RISE OF LARGE-SCALE PARTNERSHIPS AND LANDSCAPE CONSERVATION DESIGN

The natural resource field has increasingly focused on a new approach to conservation to address landscape-scale issues and global change—one that emphasizes coordination among partners across large scales. Migratory bird joint ventures (U.S. North American Bird Conservation Initiative Committee 2000) and landscape conservation cooperatives (LCCs) (Baldwin et al. 2018) have become important examples of collaborations across agency and ownership boundaries to conserve sustainable landscapes in the face of global change (also see Dinerstein et al. 2017, Groves et al. 2009). For example, the U.S. Fish and Wildlife Service developed Strategic Habitat Conservation (SHC) planning across ecoregions in a way that capitalizes on the ability of states and other partners to establish and manage conservation reserves and protected areas (Chape et al. 2005, Turner and Pressey 2009) and access to programs that support habitat conservation on private lands (Gordon et al. 2011, Rey Benayas et al. 2009). With this approach, these partnerships extend beyond the reach or resources of any one organization (Fitzgerald et al. 2009, Will et al. 2005).

Across any landscape or region, a partnership's ability to develop conservation strategies can be limited by contrasting priorities, uncertainty in the condition of the landscape, or a lack of data on the biodiversity, habitats, and ecological processes that might be vulnerable. For this reason, LCCs rely on a process formalized by the Fish and Wildlife Service known as landscape conservation design to help guide partners in their planning. Landscape conservation design is an iterative, collaborative, and holistic process that produces information, analytical tools, maps, and strategies to achieve landscape goals collectively held among partners (USFWS 2017). This process owes much of its characteristics to the larger body of landscape conservation planning work and how it has evolved over the last half century. Landscape conservation design has roots that date back to the 1980s and 1990s when planning focused on the systematic and spatial selection of nature reserves or protected areas (Margules and Pressey 2000). Therefore, a critical stage of landscape conservation design is the assessment of landscapes and prioritization of areas for conservation or protection (Baldwin et al. 2018, Knight et al. 2006).

Redford et al. (2003) broadened conservation planning's scope to consider how or what activities will best achieve conservation in the selected areas (Groves and Gaines 2016). As a result, landscape conservation design now focuses equally on the strategy and implementation of conservation designs (Knight et al. 2006). Landscape conservation design is also partner driven, drawing on more recent revelations where robust decisions must consider stakeholder values (Baldwin et al. 2018, Sinclair et al. 2016). For example, LCCs focus heavily on State wildlife action plans as a means to understand the conservation needs of state partners. See Groves and Gaines (2016) for an in-depth background on the history of landscape planning.

Landscape Models and Data

Important components of landscape conservation design include assessments of current and projected future conditions of the landscape. Conservation of large landscapes has benefited from the increased availability of various types of landscape models (Millspaugh and Thompson 2009). Models of land use such as the USGS's Gap Analysis Program (GAP), National Landcover Database (NLCD), and LANDFIRE that are derived from remotely sensed data provide estimates of current forest and habitat compositions across regions (Fry et al. 2011, USGS 2012). Similar models are being developed to map forest structure such as canopy cover and basal area (Wilson et al. 2012). The U.S. Geological Survey has compiled the National Hydrography Database that maps surface water across landscapes (USGS 2010). Spatial data on landform type, elevation, and soils are also available (Theobald et al. 2015).

More recently, advances in modeling and computation have allowed scientists to project landscape conditions based on driving factors of change (e.g., climate change and land use). By applying trends in urban growth and land-use conversion, SLEUTH and FORE-SCE models can project land cover through the next century (Jantz et al. 2010, Sohl et al. 2016; see Box 1). Progress in down-scaling future climate projections to more local scales provides the ability to identify areas that might offer refuge to species as the climate warms (Morelli et al. 2016). Down-scaled climate data have also been integrated into existing landscape models to project future changes in forest and water resources across landscapes (LaFontaine et al. 2017). LANDIS PRO is a spatially explicit forest landscape model that integrates climate data to simulate forest landscape change over large spatial and temporal scales under changing climates (Wang et al. 2015, Wang et al. 2016; see Box 2).

In their planning facilitation role, LCCs coordinate efforts to produce and compile the spatial data, models, and projections, and the LCCs make these available to partners. For example, data for online conservation blueprints have been created and distributed by various LCCs, where these data are made public and freely available. In some cases, additional data are developed such as wildlife habitat models. Maps of ecological (or environmental) site potential are used to represent the vegetation that could be supported at a given site based on the biophysical environment (CHJV 2010, Tirpak et al. 2009). With these products, the partnerships can assess conditions and vulnerabilities across landscapes (e.g., species' populations and habitats, forest communities, or ecosystem processes) and identify priorities (GCPO 2016).

Contradiction, Complexity, and Uncertainty in Deciding Strategies

Despite the information products generated by the early stages of landscape conservation assessment, deciding on the best strategies to achieve regional goals remains challenging for multiple reasons. First, choosing the best strategy requires, but may lack, some estimate of the effectiveness of competing strategies in reaching the regional goals. As an example, supporting sustainable populations of priority species is often a goal in regional conservation typically addressed through the conservation or restoration of habitat. Projections of spatial data such as landscape and habitat conditions may be available to guide strategies of habitat restoration over time. However, it is difficult to evaluate how each strategy fundamentally affects species' population growth and viability because the effects of landscape changes on population growth also depend on metapopulation processes, species interactions, and interactions between demographic and landscape dynamics (Akçakaya and Brook 2009, Keith et al. 2008). Because

BOX 1. PROJECTING URBAN GROWTH: SLEUTH MODEL

SLEUTH, named for the model input datasets (slope, land use, excluded, urban, transportation and hillshade) is the evolutionary product of the Clarke Urban Growth Model that uses cellular automata, terrain mapping, and land cover change modeling to address urban growth (Jantz et al. 2010). SLEUTH provides urban growth projections that are useful across a range of applications, including wildlife habitat analysis, conservation planning, and land cover dynamics analysis. SLEUTH incorporates four growth rules (spontaneous growth, new spreading centers, edge growth, and road-influenced growth) to model the rate and pattern of urbanization. The model simulates not only outward growth of existing urban areas, but also growth along transportation corridors and new centers of urbanization. SLEUTH incorporates dispersion, breed, spread, slope, and road gravity into the growth rules that project future urbanization. Natural and social land-use controls, such as topographic barriers or regulatory restrictions in sensitive environmental areas, are specified in the model parameterization and through resistance layers that reduce the likelihood of urbanization.

Urbanization in the Southeast

Urban growth projections used in the Ozark Highlands decision-support process extended from efforts to model urban sprawl across the rapidly growing Southeast United States (Terando et al. 2014). The modeling inputs data on slope, elevation, land cover/land use, and transportation or road networks. Urbanization was precluded for particular landcover and ownerships in the Protected Areas Database of the United States (USGS 2012).

Results published in Terando et al. (2014) point to a future where urban areas occupy a much greater portion of the landscape of the Southeast United States (Fig. 1). The projected region-wide increase in urban area would constitute a doubling or tripling of land devoted to urban and suburban uses. The tremendous growth in urbanization will come at the expense of natural areas as well as agricultural and silvicultural landscapes. Furthermore, the growth will be uneven and focused in areas that have few geographic and socioeconomic constraints, or in areas with high aesthetic value that act as strong attractants for development.

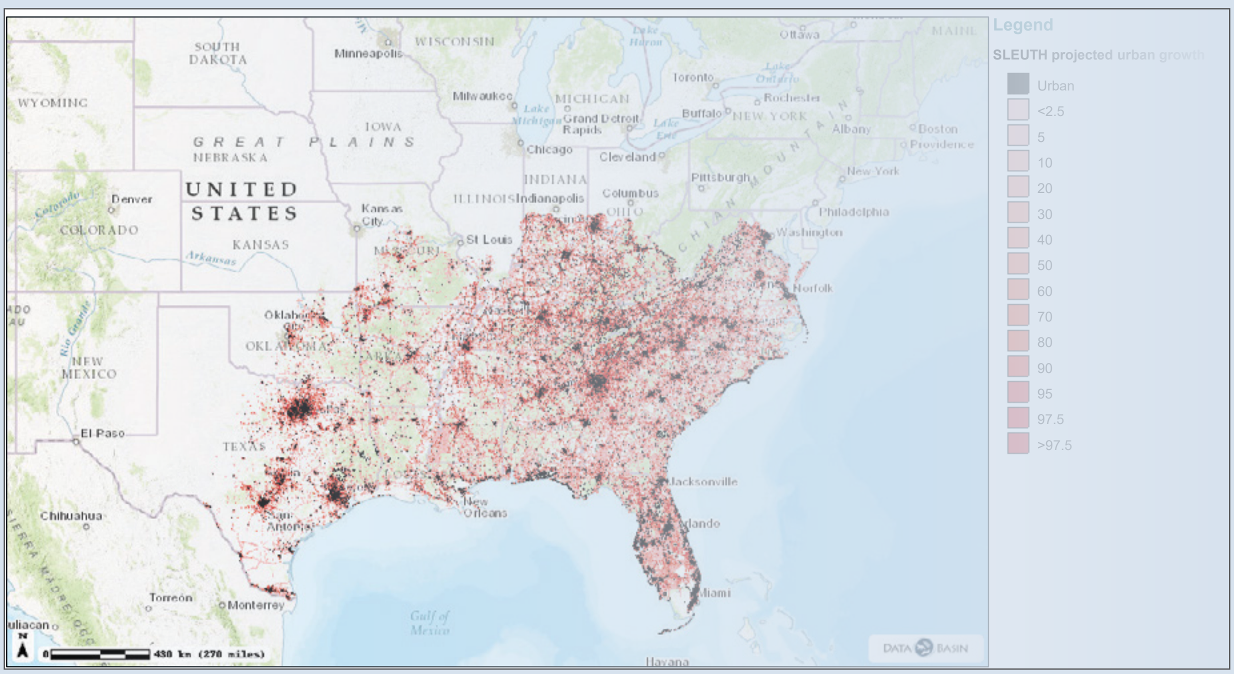

Figure 1.—Projections of urbanization through 2100 using the SLEUTH urban growth model (Terrando et al. 2014). Image obtained from the Gulf Coastal Plains and Ozarks landscape conservation cooperative via Data Basin.

BOX 2. PROJECTING FOREST LANDSCAPES UNDER CLIMATE CHANGE: THE FOREST LANDSCAPE MODEL

Landis Pro

LANDIS PRO is a spatially explicit forest landscape model (FLM) that simulates forest landscape change over large spatial and temporal scales (Wang et al. 2014). LANDIS PRO 7.0 simulates the dynamics of forest succession, seed dispersal, wind, fire, biological disturbance (insects and diseases), harvesting, fuel accumulation and decomposition, and fuel management. LANDIS integrates succession dynamics at the tree species level with landscape processes.

In LANDIS PRO, a landscape is modeled as a grid of cells (or sites) with vegetation information stored as attributes for each cell (Fig. 2). At each cell, the model tracks ages and species of individual trees. LANDIS PRO accommodates landscapes millions of square kilometers in size depending on simulation cell size and models forest change at various time steps (e.g., annual or decadal), making the model suitable for simulating short- and long-term dynamics (tens to thousands of years).

Within LANDIS PRO, forest and landscapes processes interact across scales to simulate changes in forests. Species-level processes include tree growth, seedling establishment, stem resprouting, and mortality and are simulated using species' vital attributes and empirical relationships for growth. These processes are regulated by stand-scale processes that include resource competition (e.g., light, water, and nutrients), which considers the amount of growing space occupied within each cell and incorporates self-thinning species shade tolerance and species establishment probability. Resource availability varies among different stand development stages due to the dynamics of establishment and mortality. Landscape-scale processes simulated in LANDIS PRO include seed dispersal (exotic species invasion), fire, wind, insect and disease spread, forest harvesting, fuel treatments, and silviculture treatments. These disturbance processes release growing space on one or more stands on the landscape; thereby, disturbance often resets the development stage of affected stands. Species establishment, maximum growing space, and disturbance and management patterns can vary across land type, soil type, and land use (Fig. 2).

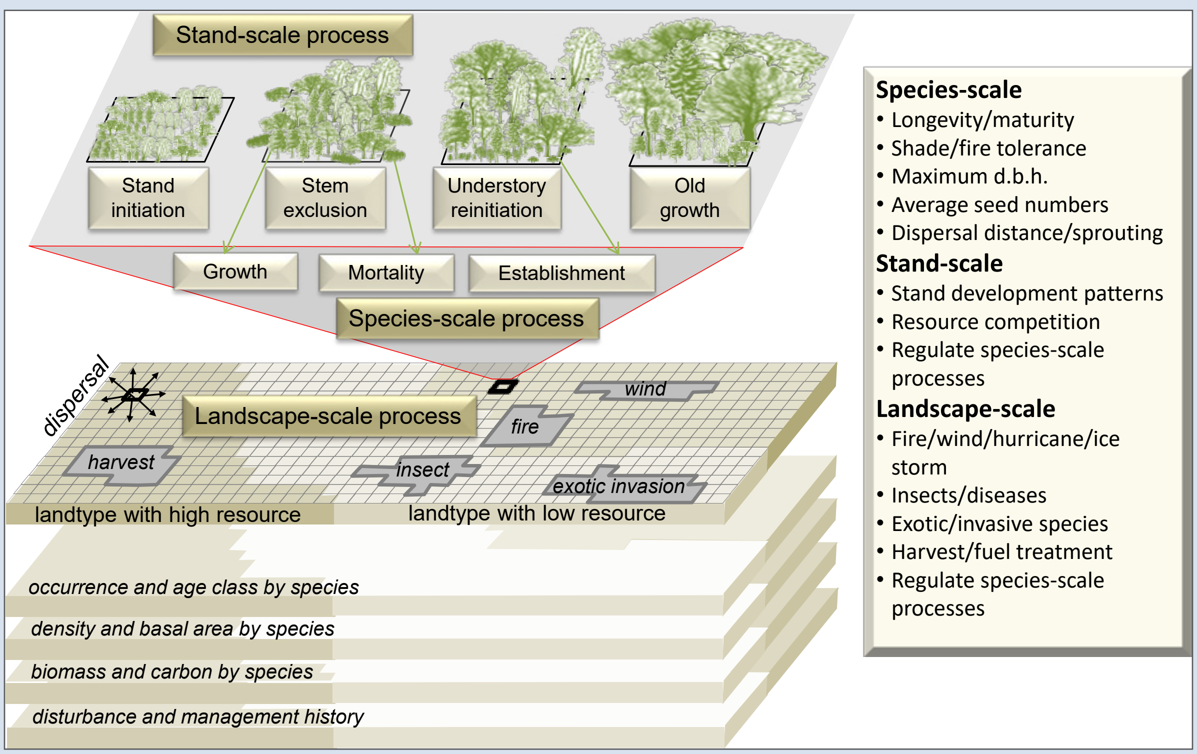

Figure 2.—The conceptual design of LANDIS PRO (Wang et al. 2014).

Box 2 continued.

LANDIS PRO and Climate Change

Wang et al. (2015) combined LANDIS PRO with the ecosystem model LINKAGES 3.0 (Dijak et al. 2016) to account for climate change. LINKAGES integrates temperature and precipitation data with nitrogen availability and soil moisture to model individual tree species growth and mortality at a site. LINKAGES provided estimates of the early growth and establishment of different tree species and the maximum allowable tree biomass over time under different climate change scenarios, which were then used in LANDIS PRO. Linking these parameters to climate with LINKAGES allows LANDIS PRO to model the impacts of climate change on forests in addition to the other forest and landscape processes.

LANDIS PRO Outputs

LANDIS PRO estimates a variety of key stand parameters such as species composition, basal area, density, stocking, and importance values for individual species and sizes/cohorts. From these basic forest metrics, biomass and carbon can be approximated. Habitat and landcover variables for wildlife species can also be derived through empirical relationships and further processing (Fig. 3).

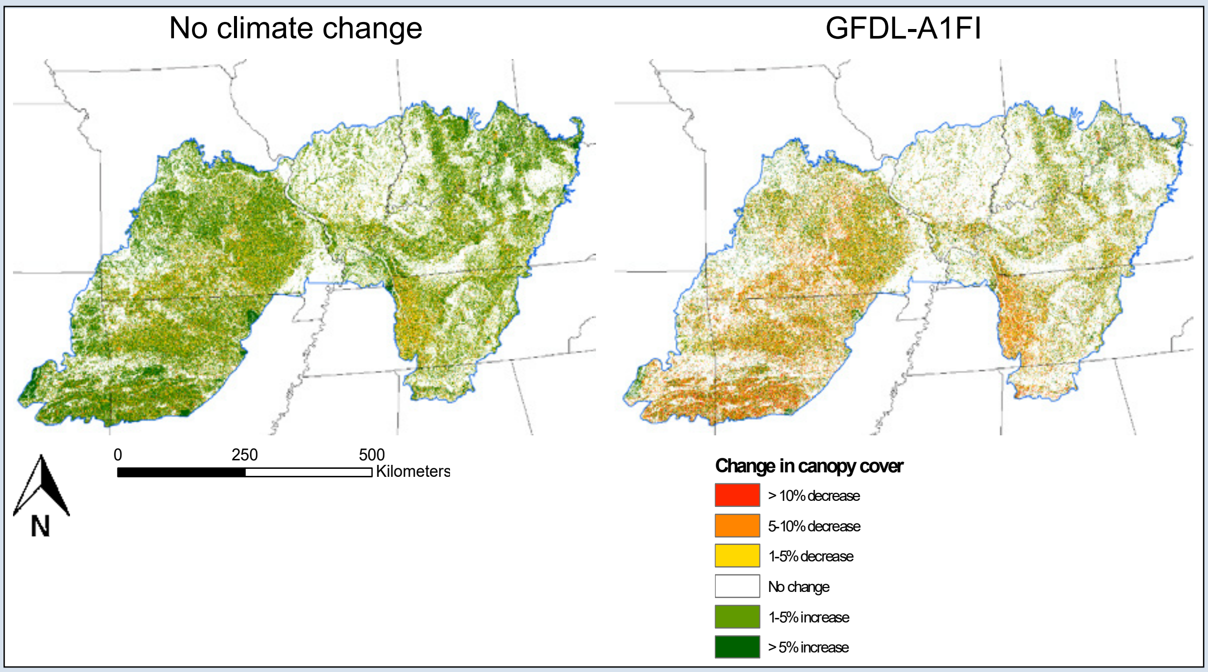

Figure 3.—Example of habitat variable derived from LANDIS PRO outputs. After calculating total stocking from LANDIS PRO, an empirical estimated relation between stocking and canopy cover was applied to estimate canopy cover across the Central Hardwoods under different climate change scenarios. The relative change in canopy cover indicated in the maps illustrates possible impacts of climate change on the region's forests in the next century. Image by the authors.

of these interactions, responses of regional wildlife populations to landscape processes such as climate change and urbanization and the conservation strategies that address them are actually quite complex and sometimes counterintuitive (Bonnot et al. 2013, Bonnot et al. 2017).

Secondly, design decisions are complicated by uncertainty of future impacts from landscape and climate change. Climate change is expected to be a key driver of landscape change, but the degree, rate, and nature of projected climate change is highly uncertain (IPCC 2007). So too are the diversity of habitat responses to climate and its interactions with other stressors such as land-use change (urbanization), invasive species, pathogens, or pollutants (Parmesan 2006, Wang et al. 2015). Where comparisons among design strategies are conditional on these threats, their uncertainty is further compounded and tools or techniques are needed to reach informed decisions (Baldwin et al. 2018, Groves and Gaines 2016).

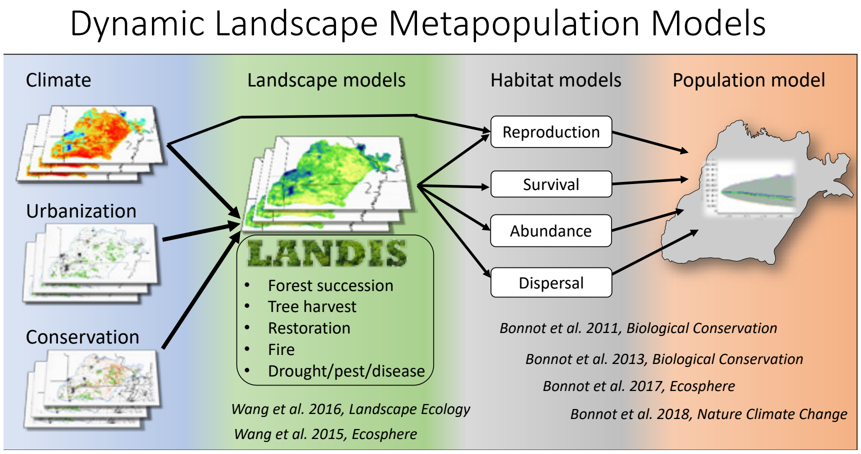

Figure 4.—Dynamic-landscape metapopulation modeling approach linking local habitat to regional species' population growth. This approach combines climate, landscape, habitat, and population models to project the responses of wildlife populations to climate and management driven changes to the landscape. Image by the authors.

Finally, SHC is impeded by the complexity of the decisions that planners must make. Partners can consider various management actions, such as restoring degraded habitat or protecting current habitat. For some species, reintroduction, captive breeding, or efforts to reduce anthropogenic mortality (e.g., communication tower alterations) may be needed in addition to habitat conservation (Bonnot et al. 2013). Identifying the best approach is difficult given the uncertainty discussed above and it is further complicated by additional strategic considerations such as the acreage, configuration, and condition of habitat needed to achieve species goals. This is particularly true as the burden of conserving biodiversity will fall increasingly on other sectors such as agriculture, water management, and urban planning (Pierce et al. 2005).

A major source of complexity is the need for designs to fulfill multiple objectives. Costs, partner needs, public policy, and private and public lands managers' interest all factor into landscape decisions. Even within environmental considerations, planners may need to balance objectives related to biodiversity, ecological processes, and ecosystem services (Groves and Gaines 2016). For example, deciding on a conservation plan that is best for a suite of focal species is complicated when different species have conflicting responses to each scenario.

DEVELOPING A DECISION-SUPPORT FRAMEWORK

Two emerging areas of research are providing regional planners with an opportunity to overcome some of the difficulties in landscape conservation design (Dreschler and Burgman 2004).

Dynamic-Landscape MetaPopulation Models (DLMPs)

DLMP models link local habitat and landscape patterns to regional population growth, providing the means to link the impacts of landscape change with the viability of regional wildlife populations (Akçakaya 2002, Bekessy et al. 2009). DLMPs combine spatial models of landscape data, quantitative models of wildlife habitat and demographics, and stochastic population models to project abundances over time. Because of this, the development of these comprehensive tools for large landscapes has been spurred primarily by technological advances that are providing more spatial data at these scales and more efficient computer processing. The flexibility to model different habitat and demographic processes in a spatial nature allows design strategies to be simulated by changing the underlying maps. DLMPs have been used to predict how bird populations might respond to forest management and restoration actions (Bonnot et al. 2013, Wintle et al. 2005). Recently, DLMPs have incorporated the effects of climate change on future forests and landscapes to model how regional wildlife populations might respond to changes in their habitat (Bonnot et al. 2017, Wang et al. 2015; Fig. 4). By directly incorporating the effects of warming temperatures on species demographics, these models have also warned of potential declines in a common species under climate change (Bonnot et al. 2018). With these advancements, DLMPs offer planners an important tool to estimate (or project) the effectiveness of conservation strategies in sustaining wildlife under climate and land-use change.

Structured Decision Making

In addition to large-scale modeling, natural resource managers and conservation planners are increasingly employing decision analysis to solve complex conservation problems (Marcot et al. 2012, Thompson et al. 2013). SDM is a formal decision analysis framework for addressing complex decisions (Gregory et al. 2012; see Box 3). Core elements of SDM include defining objectives and measures of performance, identifying and evaluating alternatives, and making choices based on a clear understanding of uncertainties and tradeoffs (Gregory and Long 2009; Fig. 5). Along this process components are addressed with an array of analytical methods, drawn from decision theory and applied ecology (Possingham et al. 2001). For example, as species have different habitat requirements, management actions detrimental to one species may enhance the persistence of other species (Dreschler and Burgman 2004). In SDM there are a variety of ways to resolve tradeoffs between competing objectives. There are also approaches within SDM that directly consider uncertainty in the outcomes of proposed alternatives for different states of the system (Fig. 6). This aspect is useful in the situation where multiple climate change scenarios are possible and the effectiveness of different conservation strategies depends on them. In general, SDM has proven effective for the types of problems conservation planners face (Drechsler 2000, McGowan 2013, Ralls and Starfield 1995). See Box 3 for examples of SDM being used in SHC.

Combining DLMPs and SDM into a single framework could benefit the current landscape conservation design approach greatly. Numerous landscape conservation planning efforts currently exist or have been conducted either organizationally or through independent projects (Sinclair et al. 2016). And most incorporate some of the current aspects that relate to making large-scale, long-term conservation decisions in the current societal and environmental reality. U.S. Forest Service management plans, The Nature Conservancy's Conservation by Design framework, and the Northeast Association of Fish and Wildlife Agencies' Nature's Network are examples of regional level planning for habitat and biodiversity targets across multiple ecosystems and species (Botrill et al. 2012, McGarigal et al. 2018, Thompson et al. 2013). Recent examples of the use of DLMPs to inform habitat management for wildlife population viability include Conlisk et al. (2014) and Ponce-Reyes (2013). Dreschler et al. (2003) and Noss et al. (2002) combined population viability models and SDM to compare species risk across reserve selection scenarios. And Hache et al. (2016) and Fordham et al. (2013) used DLMPs to examine the effectiveness of habitat management on population viability of species under climate change. However, the opportunity still exists to incorporate all these aspects into a single framework. Integrating DLMPs and SDM could provide valuable information about the relative effectiveness of alternative conservation designs at promoting species viability over time while addressing the uncertainty of threats such as climate change and the complexity associated with multiple objectives and species.

BOX 3. STRUCTURED DECISION MAKING IN NATURAL RESOURCES MANAGEMENT

Structured decision making (SDM) is an approach for careful and organized analysis of natural resource management decisions. Arising from the field decision science, SDM encompasses a framework and a set of concepts that can address the complexity and uncertainty that plague environmental planners as they increasingly include competing objectives and values from diverse stakeholders for natural systems that will be affected by global change processes. Recognizing the importance of a framework that is transparent, defensible, and adaptable has led multiple U.S. agencies to apply SDM across a range of administrative, management, and policy decisions. Examples include:

- Reintroduction of the endangered Steller's eider (*Polysticta stelleri*) in Alaska (USFWS 2016).
- Incidental take of the threatened piping plover (*Charadrius melodus*) on the Missouri River (McGowan 2013).
- Multispecies management of the horseshoe crab (*Limulus polyphemus*) and shorebird populations in Delaware Bay (Smith et al. 2013).
- Implementation of a monitoring plan on the Tongass National Forest (USDA 2008).

With SDM decision makers, stakeholders and scientists interact throughout key stages to define the problem, decide on objectives, identify alternatives, evaluate their consequences, and make a decision (Fig. 5). During this process, SDM relies on concepts in decision analysis to explicitly deal with uncertainty, transparently respond to stakeholder values, and appropriately process risk. For example, decision trees are among the methods available to identify the value/risk associated with a particular decision given when future events or conditions are uncertain (Fig. 6). Choices among alternatives when there are competing objectives can rely on multiple-objective tradeoff analysis that provides techniques to simplify either the alternatives or objectives or resolve tradeoffs.

For more information see Gregory et al. (2012) and Thompson et al. (2013) as well as the following Websites:

[USGS Science and Decisions Center]

[U.S. Fish and Wildlife Service National Conservation Training Center]

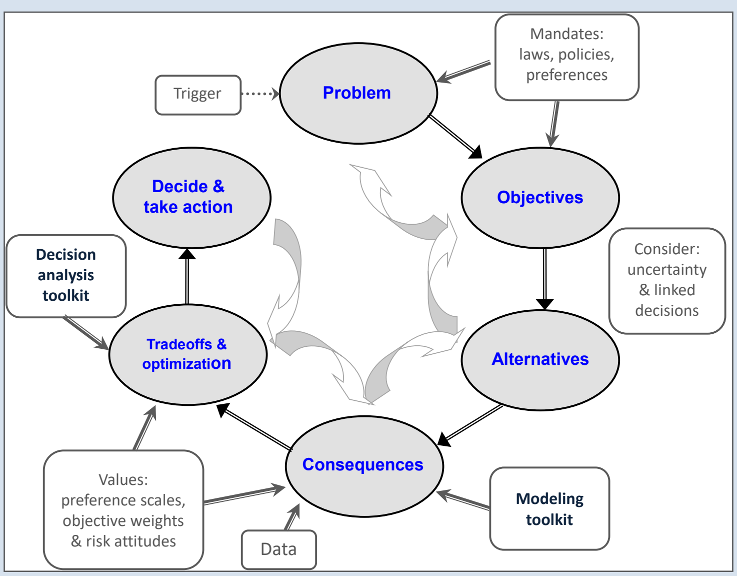

Figure 5.—The PrOACT process of Hammond et al. (2002) as implemented in structured decision making for environmental decisions. Figure adapted from Runge et al. (2011).

Box 3 continued.

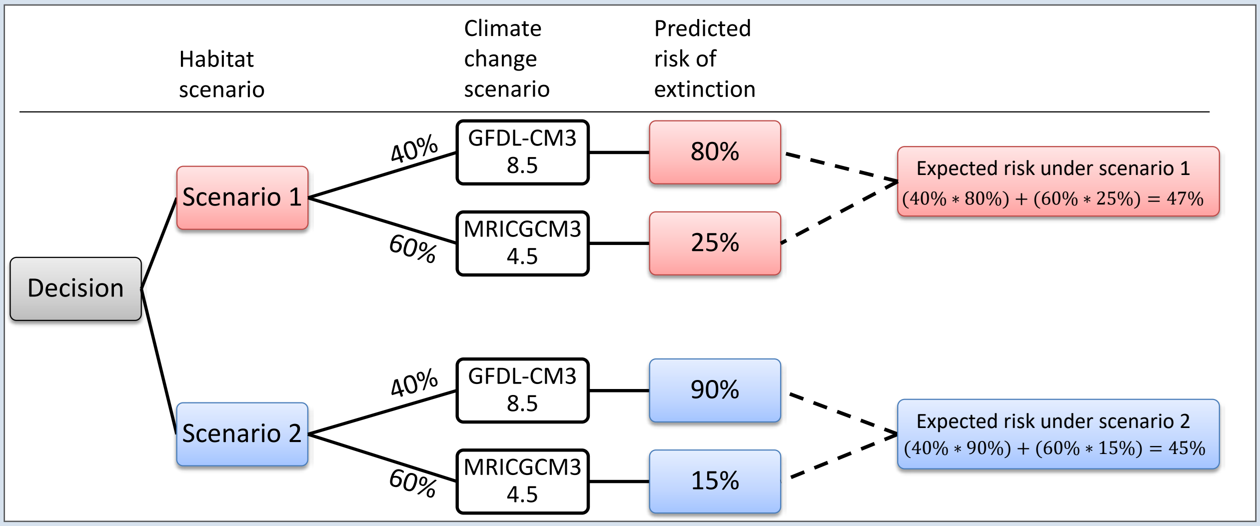

Figure 6.—Decision trees are a tool in decision analysis that help to compare choices in a decision when there is uncertainty in the final outcome. Here the expected risk of extinction to a population depends not only on one of two habitat conservation scenarios, but also how climate might change in the future. The expected risk under each scenario is estimated in a way that accounts for uncertainty in future climate change and probability of extinction in each outcome. Image by the authors.

OBJECTIVE: DEVELOP A DECISION-SUPPORT FRAMEWORK THROUGH INTEGRATION OF STRUCTURED DECISION MAKING WITH DYNAMIC-LANDSCAPE METAPOPULATION MODELS

We conducted a pilot project with the Gulf Coastal Plains and Ozarks landscape conservation cooperative (GCPO) to investigate the use of a decision-support framework for landscape conservation design that integrated DLMPs and SDM. The GCPO is a federally sanctioned regional partnership charged with defining, designing, and delivering landscapes capable of sustaining natural and cultural resources now and into the future (see Box 4). The GCPO has been working with a team of partners (hereafter the "team") on a landscape conservation design that will achieve landscapes capable of sustaining healthy plant and animal communities throughout the Ozark Highlands portion of the GCPO region. The team prioritized conservation opportunity areas across the Ozark Highlands of Arkansas, Missouri, and Oklahoma for restoration of key habitat types (Fig. 7). Our goal was to integrate DLMPs and SDM to evaluate alternative restoration designs within a decision-support framework that best sustain focal wildlife populations in the Ozark Highlands region under climate change and urbanization and to demonstrate the effectiveness of such a framework for informing landscape conservation design.

BOX 4. THE OZARK HIGHLANDS WITHIN THE GULF COASTAL PLAINS AND OZARKS LANDSCAPE CONSERVATION COOPERATIVE

The Gulf Coastal Plains and Ozarks landscape conservation cooperative (GCPO) spans a 180-million-acre region that includes all of Arkansas and Mississippi and parts of 10 additional states spanning five major subgeographies: East Gulf Coastal Plain, West Gulf Coastal Plain, Ozark Highlands, Mississippi Alluvial Valley, and the Gulf Coast (Fig. 7). The GCPO LCC exists to define, design, and deliver landscapes capable of sustaining natural and cultural resources at desired levels now and into the future. The GCPO is focused on assessing how climate, development, and other drivers make resources more vulnerable in the future and develops proactive management recommendations, tools, and adaptation strategies in response.

The Ozark Highlands covers approximately 14.5 million ha (>140 km) of the GCPO geography and includes the Ouachita and Boston Mountains regions (Fig. 7). The landscape features rugged uplands—some peaks higher than 700 m above sea level—with exposed rock and varying soil depths and includes extensive areas of karst terrain (USGS 2009). The Ozark Highlands contains diverse topographic, geologic, soil, and hydrologic conditions that support a broad range of forest communities, including: upland oak (*Quercus* spp.)–hickory (*Carya* spp.) forests, oak-pine (*Pinus* spp.) forests, woodlands, and savannas.

Although the region includes some of the most extensive forests in the middle of the continent, widespread logging in the early part of the 20th century and fire suppression in subsequent decades resulted in conversion of many glade, barren, and pine woodland communities to oak or oak-pine forests (Fitzgerald et al. 2005).

The Ozark Highlands is characterized by extreme biological diversity and high endemism, having the largest extent of glade communities in North America and being home to nearly two-thirds of the 45 federally listed plants and animals in Missouri, Arkansas, and Oklahoma (USGS 2009). The Ozark Highlands is also an important center for neotropical bird migration and breeding grounds (Fitzgerald et al. 2005).

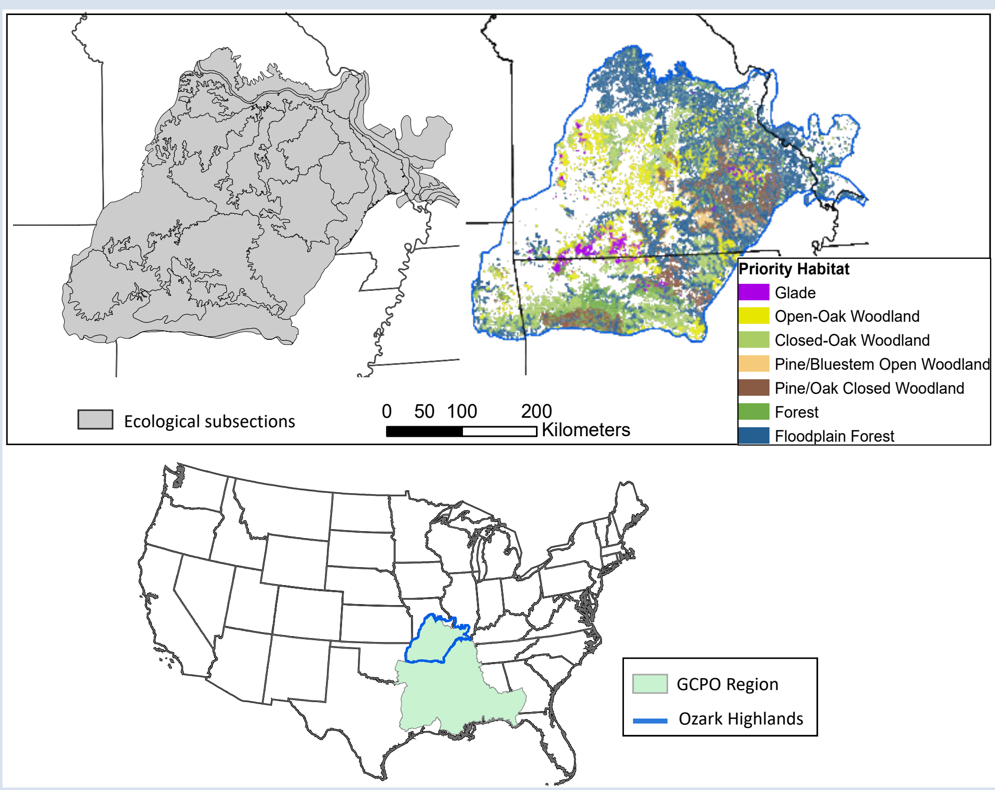

Figure 7.—Ecological subsections and lands prioritized for habitat restoration in the Ozarks Highlands, a subgeography of the Gulf Coastal Plains and Ozarks LCC. Lands selected to restore priority habitats were identified through landscape conservation design that considered current and potential landscape and habitat conditions. Image by the authors.

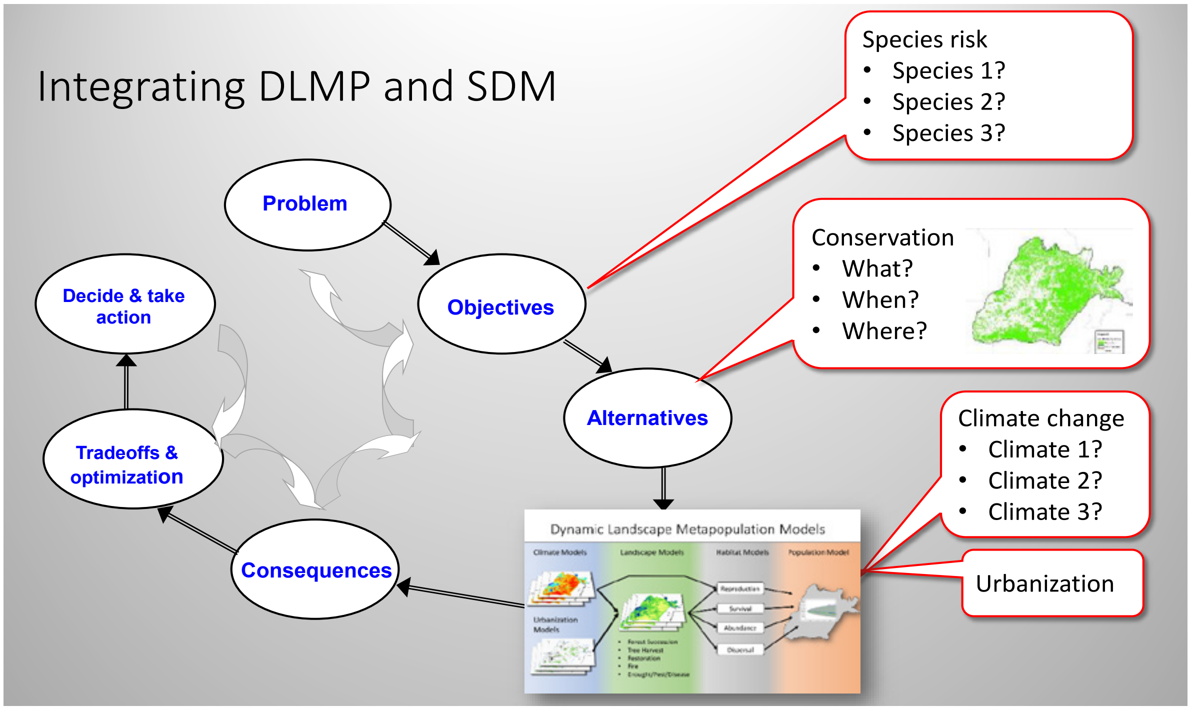

Figure 8.—Process for integrating dynamic-landscape metapopulation (DLMP) models and structured decision making (SDM) to select conservation strategies amidst uncertainties and complexities. Image by the authors.

APPROACH

We used the PrOACT approach to SDM, which is commonly used in solving natural resource management problems (Hammond et al. 2002). PrOACT breaks complex decisions down into components, which are: (Pr) articulate the problem, (O) delineate objectives with measurable attributes, (A) identify alternative actions or scenairos that address the objectives, (C) evaluate the consequences of alternatives, and (T) examine tradeoffs among competing alternatives (Hammond et al. 2002, Keeney and Gregory 2005, McGowan 2013; Fig. 8). We used a series of interactive workshops and webinars to complete the process.

Define Problem

Our team comprised ecologists and conservation planners from various agencies and organizations throughout the Ozark Highlands and included state wildlife habitat coordinators from Arkansas, Missouri, and Oklahoma and staff from the Central Hardwoods Joint Venture (CHJV), GCPO, U.S. Geological Survey, and the U.S. Fish and Wildlife Service. The team's vision was to design landscapes capable of sustaining healthy plant and animal communities throughout the Ozark Highlands through landscape conservation design.

Beginning in 2012, the team undertook an ecosystem-based approach that prioritized conservation opportunity areas across the Ozark Highlands for seven forest communities (Table 1). These habitat types included glades, variations of oak and pine woodlands, and mesic and floodplain forests. The potential distribution of these systems was based on an ecological potential model that characterizes forested or semi-forested native communities in the larger Central Hardwoods region according to land-type associations, landform positions, and assumed historic disturbance regimes (CHJV 2010). Using the National Fish Habitat Partnership 1 km catchment boundaries (USGS 2010) as planning units, the team defined the

Table 1.—Priority forest communities in the Ozark Highlands. Dynamic-landscape metapopulation models were used to predict responses of focal wildlife species (bold), selected from lists of candidate species representing each community, to alternative restoration scenarios. Acreage targets for restoration were derived from the Central Hardwoods Joint Venture bird population objectives. Restoration of forest communities was simulated in scenarios with desired forest characteristics, reflective of each community. NLCD = National Land Cover Database.

		Targeted restoration acreage		Simulated forest characteristics				
Forest vegetation community	Representative candidate and **focal** species[a]	Hectares	Acres	Canopy cover (%)	Small-stem density[b]	NLCD class	Stocking (%)	Hardwood basal area[c]
Glade	**Prairie warbler**, **eastern collared lizard**, Diana fritillary, wood rat	77,938	192,589	5	8,000	41	10	5
Open-oak woodland		479,193	1,184,110	40	10,000	41	30	8
Closed-oak woodland	Ozark big-eared bat, ***Myotis* spp.**	895,095	2,211,824	70	4,000	41	50	13
Open-pine woodland	**Brown-headed nuthatch**, eastern tiger salamander	56,979	140,798	40	6,000	42	30	3
Pine-oak woodland	Chuck-will's-widow, whip-poor-will, ringed salamander	471,530	1,165,174	70	4,000	43	50	8
Mesic forest	**Wood thrush**, wood frog, Ozark salamander, **cerulean warbler**	654,451	1,617,181	90	2,000	41	70	16
Riparian/ bottomland forest	**Prothonotary warbler**, Oklahoma salamander, Rafinesque's big-eared bat	363,363	897,888	90	2,000	90	70	19

[a]Prairie warbler (*Setophaga discolor*), eastern collared lizard (*Crotaphytus collaris*), Diana fritillary (*Speyeria diana)*, wood rat (*Neotoma floridana*), Ozark big-eared bat (*Corynorhinus townsendii ingens*), brown-headed nuthatch (*Sitta pusilla*), eastern tiger salamander (*Ambystoma tigrinum*), Chuck-wills-widow (*Antrostomus carolinensis*), whip-poor-will (*Caprimulgus vociferus*), ringed salamander (*Ambystoma annulatum*), wood thrush (*Hylocichla mustelina*), wood frog (*Lithobates sylvaticus*), Ozark salamander (*Plethodon angusticlavius*), cerulean warbler (*Setophaga cerulea*), prothonotary warbler (*Protonotaria citrea*), Oklahoma salamander (*Eurycea tynerensis*), Rafinesque's big-eared bat (*Corynorhinus rafinesquii*).

[b]Stems/ha.

[c]m^2/ha.

relative priority of catchments for each forest community through set criteria. These criteria considered the potential to conserve the habitat within the catchment and the acreage of repurposed and developed lands it holds. Catchments with known locations of focal species and existing conservation network lands also received priority (GCPO 2016).

After the previous steps were completed, the team needed to decide which catchments among those of the highest priority would be selected for restoration planning. Based on the above criteria, the team identified a network of almost 3 million ha (6 million ac) of catchments that were high priority for the seven forested community types throughout the Ozark Highlands (Fig. 7). At that point, the decision became complicated by the many possible scenarios of how much, when, and where to restore habitats. Further, how can one scenario be decided when it is likely that that species would respond differently to different scenarios? And how could the team make those comparisons when they are uncertain in how species would respond to conservation activities amidst impacts from landscape and climate change, especially when those impacts are also uncertain? Therefore, the team defined the problem as needing to develop a conservation design or strategy for restoring natural communities capable of sustaining animal populations throughout the Ozark Highlands under urbanization and climate change.

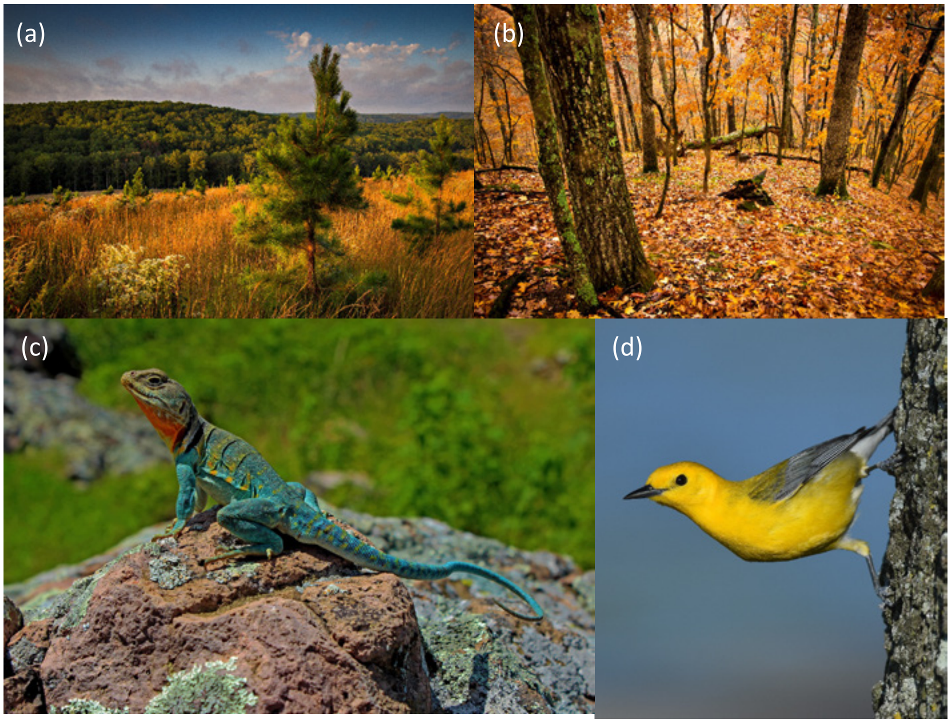

Figure 9.— (a) An open woodland/savannah with native grasses and shortleaf pines at Wurdack Research Center, Missouri. (b) An oak forest at the Baskett Wildlife Research and Education Center. (c) Collared lizards are no longer a common sight in the Ozarks, as many of the open glades they inhabit have become overgrown with red cedar trees. Without prescribed burns or natural fire regimes, glades tend to become slowly overgrown by trees. (d) Prothonotary warbler. Photos (a) and (b) by Kyle Spradley, University of Missouri College of Agriculture, Food and Natural Resources, Flickr Creative Commons, https://www.flickr.com/photos/. Photo (c) by Peter Paplanus, Flickr Creative Commons. Photo (d) by Andy Reago and Chrissy McClarren, Flickr Creative Commons.

Objectives

In 2015, we convened a workshop with the team to begin the SDM process. At this workshop, the team established the measurable objectives and identified alternative scenarios for meeting those objectives. The fundamental objective inherent in the problem statement was the restoration of natural communities that could sustain wildlife populations in the Ozark Highlands (Fig. 9). Therefore, the team established as objectives the viability of populations of focal species associated with the different communities. Focal species were selected from a set of candidates representing each community type (Table 1). We merged species for glade and open-oak woodland habitats given the similarities in these systems and the overlap in species. Species that received consideration were well-associated with the targeted communities (not habitat generalists) and had some reason for conservation concern (e.g., declining population or reduced range). We also attempted to represent multiple taxa and species with a range of movement/dispersal abilities.

We selected seven species for which sufficient habitat and demographic data existed (Table 1). Songbirds comprised five of the species: prothonotary warblers (*Protonotaria citrea*) for bottomland forests, wood thrush (*Hylocichla mustelina*) and cerulean warblers (*Setophaga cerulean*) for mesic forest, prairie warblers (*Setophaga discolor*) for glade and open-woodland habitat, and brown-headed nuthatch (*Sitta pusilla*) in open-pine woodlands. A paucity of data prohibited modeling the team's original choice of Ozark big-eared bats (*Corynorhinus townsendii ingens*) for closed-oak woodlands. Therefore, we developed a model for a generic *Myotis* spp. based on data for Indiana bats (*Myotis sodalist*) and northern long-eared bats (*Myotis septentrionalis*). The team also selected the eastern collared lizard (*Crotaphytus collaris*) to represent herpetofauna in glade systems. An imbalance of data and information across taxa meant that the majority of focal species were birds, which might influence the decision in ways that relate to differences in life history or dispersal between birds and other species. However, we simulated restoration of natural communities based on definitive characteristics of each type and independent of wildlife habitat needs. We did not target restoration of habitat to birds. Therefore, we feel the set of focal species offers an objective evaluation of activities across communities.

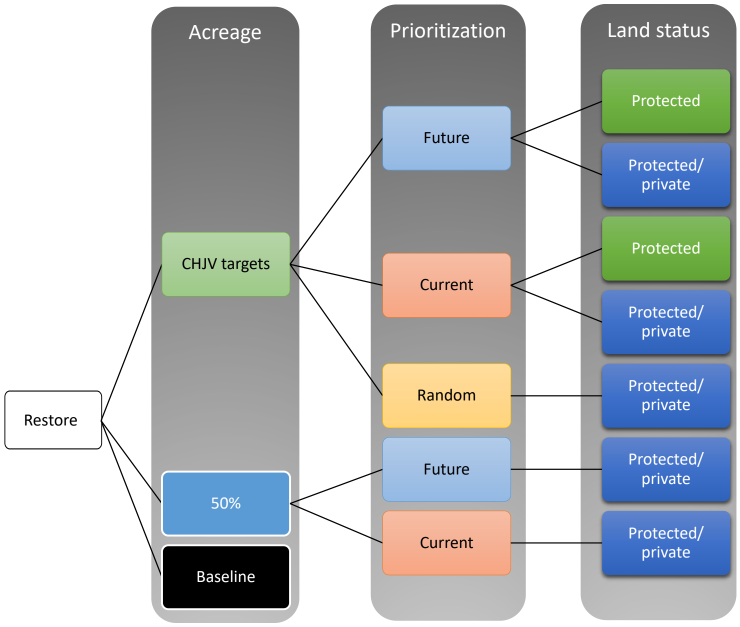

Figure 10.—The planning team developed alternative restoration scenarios around three considerations: the acreage of habitat to be restored; the basis for prioritizing areas to restore (current landscape, projected future landscape, or random); and the protection status of lands to be restored. Image by the authors.

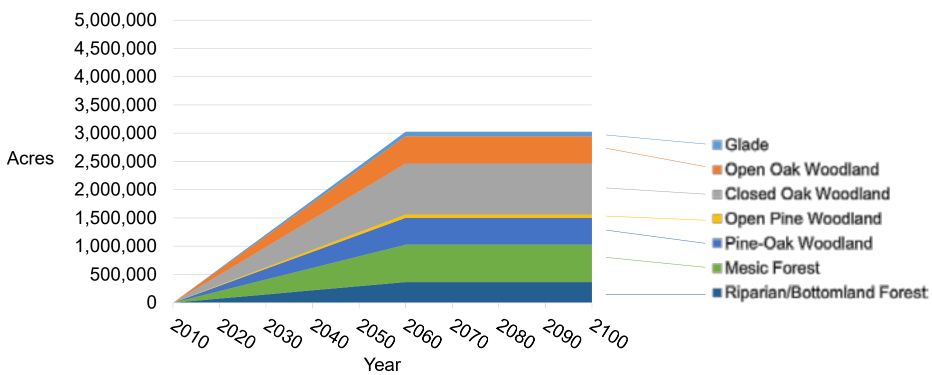

Figure 11.—Simulations of implemented restoration accumulated over a 50-year period. Under the full habitat acreage scenario (shown), approximately 1.2 million ha of habitat were targeted for restoration in the Ozark Highlands. Image by the authors.

Alternatives

At the first workshop we began developing alternative scenarios for restoring communities across priority areas to achieve viability objectives. Any kind of scenario can be considered in SDM, given it can be evaluated with respect to the objectives (Keeney and Gregory 2005). The team developed alternative scenarios around key questions faced when planning long-term conservation across large scales. The questions related to the amount of acreage needed to: benefit populations; processes used to conservation opportunity areas; and the relative impacts of restoring only public or otherwise protected lands versus restoring habitat on both private and protected lands (Fig. 10). By structuring the scenarios around these considerations, the team intended to explore an effective strategy for habitat conservation in the Ozark Highlands.

Acreage. The team outlined scenarios by the acreage of habitat restored. Realizing that restoring the entire 3 million ha network of priority areas was unrealistic, the team established acreage goals on a biological basis. We relied on previous work by the Central Hardwoods Joint Venture that estimated acreages needed for different habitats to meet population objectives of bird species associated with those habitats (CHJV 2012). This scenario targeted a total 1.2 million ha of restoration with individual community goals ranging from 23,059 ha of open-pine woodland to 362,232 ha of closed-oak woodland (Fig. 11). The team compared this scenario with one in which half of the 1.2 million ha was targeted and a baseline scenario where no restoration occurred so the team could assess the projected impact of no future restoration efforts.

Prioritization. The team also considered scenarios that considered different habitat prioritization schemes. We included scenarios that restored habitats in the highest priority catchments according to the team's original rankings of the current landscape. We also considered scenarios that restored habitat randomly across catchments, regardless of priority (Fig. 10). This comparison examined strategic approaches to regional conservation activities.

We next considered scenarios that modified the prioritization to incorporate predictions of the landscape in the future. We originally assigned higher priority to catchments that currently had less developed land. To develop a scenario that incorporated projections of the future landscape, we prioritized catchments based on projected landcover data for 2100. Future landcover was derived by combining data from recent climate, landscape, and urbanization models, which we describe in more detail below (Bonnot et al. 2017, Terando et al. 2014, Wang et al. 2015, Wang et al. 2016,). The future prioritization also incorporated projected impacts of climate change on forest conditions that complemented restoration efforts. Restoring woodland and glade communities across much of the Ozark Highlands involves thinning and burning to reduce tree density to open up canopies of closed forests. However, LANDIS projections indicate that climate change will cause similar shifts in structure in forests across parts of the region (Wang et al. 2015; Fig. 3). Therefore, the team compared current canopy cover to projections for 2100 and increased priority for woodland and glade habitats with reduced canopy cover. This future-based prioritization explored the effects of using model projections to develop conservation strategies.

Protection status. We also structured scenarios around the ownership or protection status of lands. Although the Ozark Highlands contains large amounts of publicly owned land compared to other Central United States regions, the approximately 80 percent of forests lie in private ownership. Federal and state programs incentivizing habitat restoration on private lands have increased in the region in recent decades. Despite this trend, the permanence and efficacy of private lands conservation in sustaining species of conservation concern is still questionable. Therefore, we compared scenarios that focused restoration only on protected lands (i.e., public lands or lands with protected status) against "combined" scenarios that targeted both protected and private forest lands (Fig. 10). We identified protected lands using classes I-IV in the Protected Area Database 1.3 (USGS 2012). Because the acreage of habitat targeted under a full acreage scenario exceeded the amount of protected lands, the protected-only and combined scenarios were not comparable from an acreage standpoint. However, we maintained this comparison to capture the limitation of focusing only on protected lands. We also omitted scenarios restoring half of the acreage on the protected lands given similarities in the amount of restoration (Fig. 12).

Simulations. We simulated scenarios by selecting lands for restoration and characterizing the habitat of those lands based on the desired conditions. We designed the process of selecting lands to be dynamic and interactive to reflect the reality of planning and carrying out restoration across the Ozark Highlands over time. Restoration occurred at a forest patch scale that we delineated through the intersection of landcover, catchment units, and protected areas. This scale was relevant to the level of forest management, while enabling distinction between catchment priorities and protected status. Multiple patches comprised each catchment. Because the Ecological Potential Model provided the distribution of desired habitats across the region, patches could contain multiple potential habitats. Therefore, we identified patches based on the highest scoring (priority) habitat occurring within the patch.

We implemented restoration goals for all scenarios over 50 years and selected individual patches in each decade to fulfill the acreage goals for each habitat (Fig. 13). Thus, we targeted approximately 244,901 ha and 122,668 ha of combined habitats each decade from 2010–2050 for the full and half-acreage scenarios, respectively (Fig. 11). We considered this rate of activity

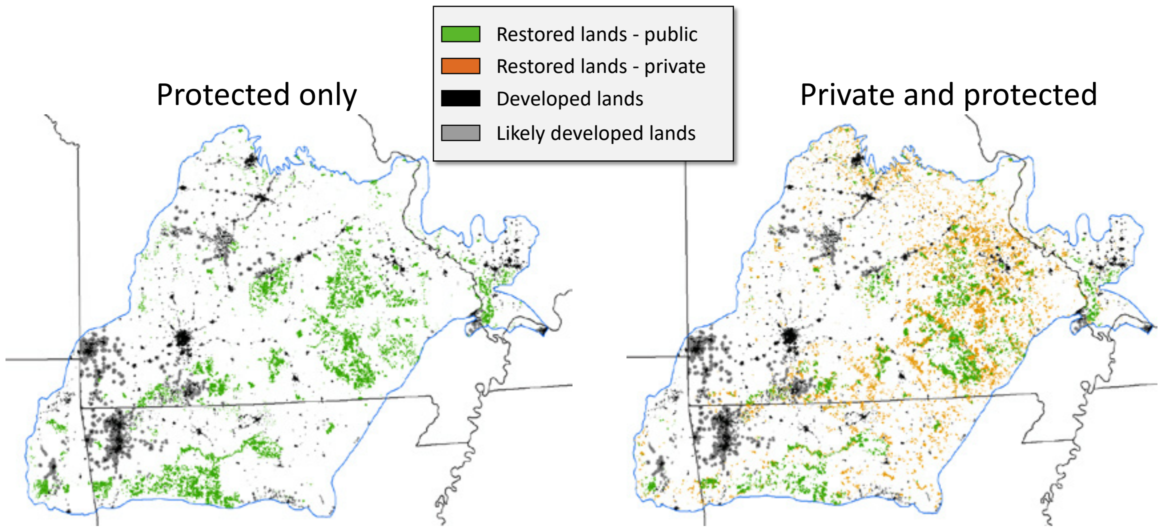

Figure 12.—Example of lands selected for habitat restoration accumulated from 2010 through 2100 under two different conservation scenarios. Scenarios simulated restoration on protected lands only (public lands or lands under easement) or on a mixture of private and protected lands. Restoration scenarios integrated urbanization projections to account for loss of lands to development over time. Image by the authors.

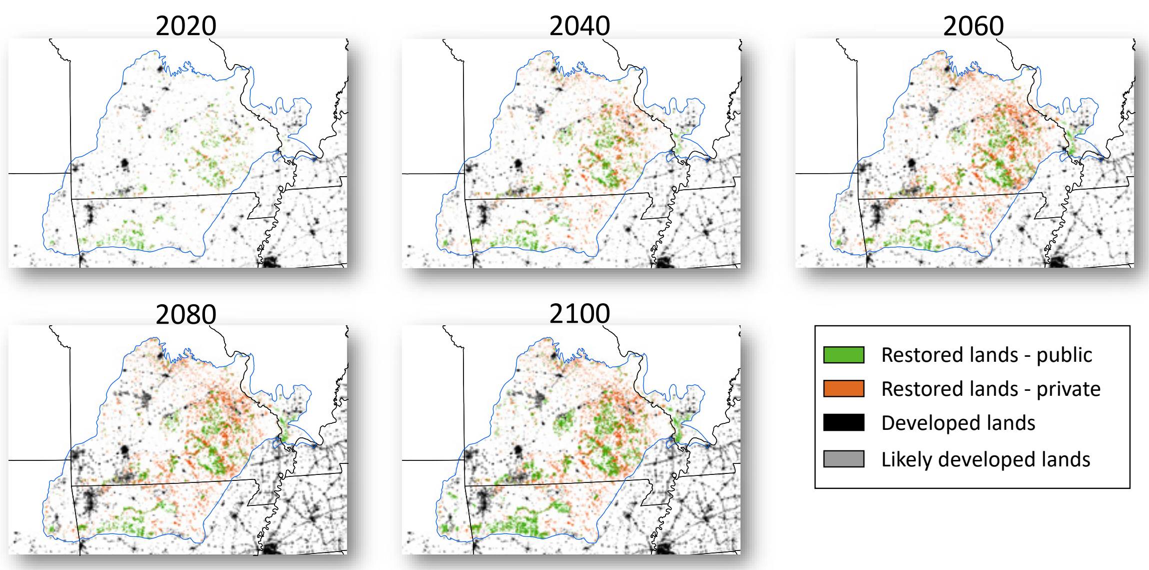

Figure 13.—Simulated location and accumulation of restored habitat under a scenario targeting 1.2 million restored ha on both private and public lands prioritized by current landscape conditions. Image by the authors.

reasonable given current efforts across the region. We selected patches in priority order of their catchment and its protected status as defined by individual scenarios. For example, to restore glade habitat we selected patches with the highest priority for glades. Only when we exhausted all available patches of a given ranking did we begin restoring patches of the next highest priority. We randomly selected patches within a priority ranking to reflect the opportunistic nature of both public- and private-land restoration. All patch processing was conducted using custom scripts in ArcGIS 10.4 (ESRI 2013).

The selection process was also interactive. Landscape projections and the patches delineated from them in each decade were a function of the previous decade's landcover, the restoration simulated during the previous decade, and the urbanization projected for that decade. This combination allowed us to incorporate interactions between private and protected lands and

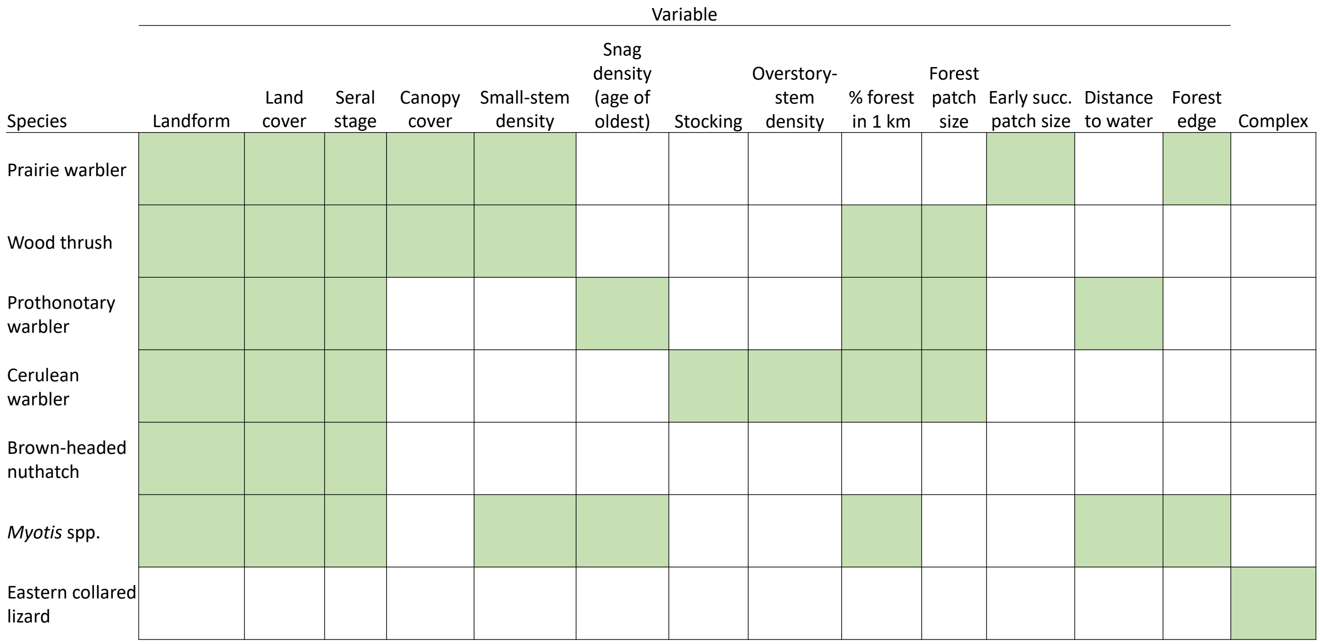

Figure 14.—Landscape and habitat variables included in habitat suitability models for focal wildlife species in the Ozark Highlands. Image by the authors.

projected urbanization (Fig. 12). We exempted protected lands from urbanization and prevented private restored patches from being developed for 20 years to reflect the nature of most private conservation agreements. Following that period, private restored patches were again susceptible to development per SLEUTH projections. We selected additional patches throughout the entire simulation to maintain restored acreage targets given losses to urbanization. All urbanization effects that removed forest were permanent and precluded restoration.

Once future habitat restoration was planned, we characterized forests on restored and unrestored lands to represent the effects of urbanization, climate change, and restoration on focal species habitat over time under alternative scenarios. We focused on characteristics related to habitat for the focal species, including variables such as landcover and landform, while other variables described aspects of forest structure and landscape configuration (Fig. 14).

Climate change and urbanization. We characterized habitat on unrestored lands by combining urbanization projections with data on forests projected under climate change. Data on forest composition and structure was a subset of 2000–2300 Central Hardwood projections from Wang et al. (2015, 2016). Wang et al. (2015, 2016) used the forest landscape model LANDIS PRO to project forest changes due to succession, harvest, and climate change (Box 2). Tree harvest reflected current management patterns of the region's forests.

We expanded on the methods by Bonnot et al. (2017) to derive habitat variables from LANDIS PRO output through geoprocessing in ArcGIS 10.4 (ESRI 2013). We characterized NLCD land cover classes by comparing the relative importance values estimated by LANDIS PRO for deciduous versus coniferous species and incorporating probabilities of urbanization (Table 2). We grouped land cover to characterize overall forest composition and configuration across the region. We derived additional forest characteristics from LANDIS PRO data: seral stage classifications, basal area, and stocking, from which we estimated canopy cover. We approximated densities of snags, small stems, and overstory trees based on LANDIS density projections for specific aged cohorts (Table 2). We used a landform classification derived from

Table 2.—Processes for deriving variables used to model habitat of focal wildlife species in the Ozark Highlands. Raw output from SLEUTH and LANDIS PRO sources was transformed to estimate habitat variables based on ecological relationships found in literature.

Model source	Source output	Habitat variable	Processing
SLEUTH[a]	Probability of development	Landcover	Classified cells as deciduous forest if the combined importance of deciduous species >65% and coniferous forest if such species comprised >47% of the cell's importance. Forested cells not classified as either deciduous or coniferous were assigned to the mixed forest type. Further classified deciduous forest cells as woody wetlands for any cells with this original NLCD class. Incorporated changes from urbanization by assigning cells with >50% probability of being urbanized as developed lands.
LANDIS PRO[b]	Importance value		
	Tree age	Snag index	Used maximum tree age of cell to approximate the density of snags (Larson et al. 2003).
		Large snag index	Applied relationship with maximum tree age derived by Rittenhouse et al. 2007. The function is based on the estimates of snag density by stand rotation age, weighted for larger snags.
	Tree density	Small-stem density	Approximated density of stems (<2.54 cm d.b.h.) by the density of all tree species in the 0–10 age cohort. Based on assumption that most hardwood species take approximately 10 years to reach 2.54 cm d.b.h. (Johnson et al. 2009).
		Overstory stem density	Approximated by the density of potential overstory tree species in the 70-300 age cohort.
	Basal area and tree density	Seral stage	Calculated quadratic mean diameter from cell basal area and tree density. Classified QMD into five seral stages: grass-forb (<2.5 cm d.b.h.), shrub-seedling (2.5 to 7.5 cm), sapling (7.5 to 12.5 cm), pole (12.5 to 37.5 cm), and sawtimber (>37.5 cm) (Tirpak et al. 2009).
		Stocking	Calculated total stocking from QMD and tree density using allometric equations for coniferous and deciduous species.
		Canopy cover	Estimated based on empirical relationship with stocking identified for hardwood forests in the region (Canopy cover = [41.83 * Log_{10}(1.21 + STOCKING]) (Blizzard et al. 2013).
	Basal area	Hardwood basal area	Extracted basal area for hardwood species.

[a]Jantzen et al. 2010.
[b]Wang et al. 2015.

DEM (Jenness 2013) and measured distance to water based on the National Hydrography Database (USGS 2010). All landscapes were processes with a cell size of 30 m.

We accounted for uncertainty in climate change projections by characterizing landscapes under three future climate scenarios. Wang et al. (2015, 2016) modeled landscapes under a current climate scenario and two climate change scenarios, based on combinations of general circulation models (GCMs) and emission scenarios from the IPCC (2007). The current climate scenario used temperature, precipitation, and wind speed data for the 30-year period from 1980 to 2009 observed throughout the region (Wang et al. 2015a). The two IPCC-derived climate change scenarios CGCM.T47-A2 and GFDL-A1FI represented alternative degrees of climate change. The GFDL-A1FI scenario combined a more substantial and immediate increase in greenhouse gas emissions (A1FI) with a model that is more sensitive to that increase (GFDL; IPCC 2007). Thus, the GFDL-A1FI scenario presented more severe changes in climate relative to the CGCM. T47-A2 scenario. These different characterizations provided contrasting background impacts against which we could capture uncertainty in the effectiveness of conservation scenarios.

We incorporated estimates of urbanization from the SLEUTH model of the Southeastern United States. (Terando et al. 2014; Box 1). The SLEUTH model is the evolutionary product of the Clarke Urban Growth Model that uses cellular automata, terrain mapping, and land cover change modeling to estimate the probability of urban development (Jantz et al. 2010). This approach incorporates growth rules to model the rate and pattern of outward growth of existing urban areas and growth along transportation corridors and new centers of urbanization (Terando et al. 2014).

Restoration. For each decade, we characterized habitat data on restored lands to reflect the conditions desired through restoration of the natural communities. The GCPO and CHJV have established desired endpoints for the different habitats that describe landcover and vegetation structural characteristics intended through restoration and continued management (Table 1). These values are based on ecological literature and the ecological potential model underlying the natural communities (Blaney et al. 2016, CHJV 2010, Kabrick et al. 2014). We assigned these endpoints and any habitat variables derived from them to restored cells. Although selection occurred at the patch level, restoration of individual cells within a patch was based on its potential community type. Thus, when a forest patch was selected for a given habitat (e.g., open-oak woodland), all potential forested cells within that patch would be restored to their potential forest type. As a result, the amount of area restored for the most common community types exceeded their targets while attempting to meet goals for less common types. We considered this a realistic assumption given management activities, such as prescribed burning, would affect entire patches.

Translocation simulations. Concerns about dispersal limitations that could prevent brown-headed nuthatch from colonizing suitable habitat outside of its current distribution prompted the team to also consider translocation of this species in addition to habitat restoration. We simulated translocation directly in the population model by introducing 100 breeding females into the Current River Hills subsection in each of the first 5 years of the simulation. This simulated reintroduction intended to provide a source population to colonize the newly created habitat.

Consequences

To predict the consequences of the alternative conservation scenarios on viability objectives, we developed DLMP models for focal species based on the landscapes simulated by each scenario. Following the approach developed by Bonnot et al. (2017), we began with landscape data projected under alternative restoration and climate change scenarios and used habitat models to translate these projections into species' habitat and demographics at each time step. Then, we incorporated these spatially and temporally varying demographics into metapopulation models that included stochasticity and parametric uncertainty. The resulting models provided spatially and temporally explicit representations of focal species habitat and population growth throughout the region.

Habitat models. We employed habitat models to link landscapes to demographic processes such as distribution, reproduction, and dispersal. We modeled distributional abundance and carrying capacity using multiscale Habitat Suitability Index (HSI) models. These models predict suitability of habitat in cells based on their attributes and the surrounding landscape (Dijak and Rittenhouse 2009). In some cases, these models have been verified and validated with empirical data (Tirpak et al. 2009). For focal bird species we relied on HSI models developed for the Ozark Highlands and surrounding landscapes (Larson et al. 2003, Tirpak et al. 2009). We modeled *Myotis* spp. habitat by combining an HSI model for Indiana bats (*Myotis sodalis*) from Rittenhouse et al. (2007) with habitat relationships for northern long-eared bats (*Myotis septentrionalis*) (Larson et al. 2003, Starbuck et al. 2014). We identified Eastern collared lizard habitat as glades interconnected by adjacent woodlands. See Boxes 5-12 for detail on the habitat and population models used for each species.

Table 3.—Species-specific demographic parameters used in dynamic-landscape metapopulation models of focal wildlife species in the Ozark Highlands. The models were stage-based models that incorporated published estimates of survival and fertility rates. Carrying capacity and initial abundance were estimated by modeling empirical densities.

		Species					
		Brown-headed nuthatch	Prairie warbler	Wood thrush	Prothonotary warbler	Cerulean warbler	*Myotis* spp.[a]
Survival	Adult	0.67	0.6	0.61	0.675	0.6	0.9
	juvenile	0.3	0.32	0.29	0.18	0.28	0.57
	pup						0.898
Fertility (females/females/year)	Adult	1.3	1.55	1.45	1.95	1.5	0.42
	juvenile						0.279
Density (females/ha)	Carrying capacity	0.225	1	0.5	0.6	2	0.1
	Initial	0.169 (75%)[b]	0.5 (50%)	0.06 (12%)	0.03 (5%)	0.14 (7%)	0.025 (25%)
Density dependence		Modified ceiling	Modified ceiling	Modified ceiling	Modified ceiling	Modified ceiling	Ricker (contest)
Parasitism		None	RPI[c]	RPI	RPI	RPI	None
Dispersal distance[d] (km)		1.27	70	70	70	70	None

[a]*Myotis* spp. = *Myotis sodalist* and *Myotis septentrionalis.*
[b]Percentage of carrying capacity.
[c]Relative Productivity Index is applied to fertility rate to account for parasitism processes in reproduction of open-nesting birds.
[d]Assumed mean distance.

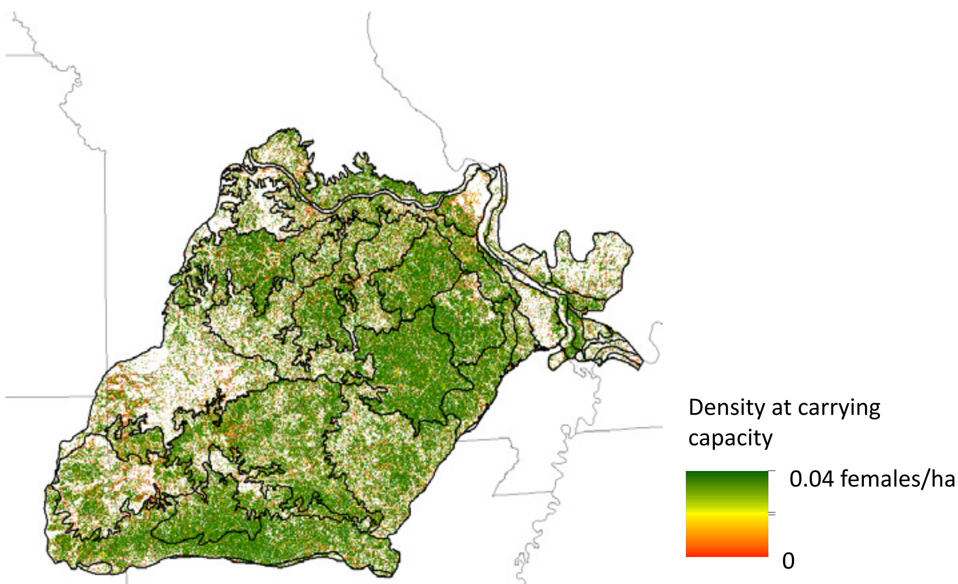

Figure 15.—Example of carrying capacity of wood thrush estimated across the Ozark Highlands in 2010. Carrying capacity is a function of habitat suitability where more suitable habitats are able to support higher densities. Image by the authors.

We followed Bonnot et al.'s (2013) approach to estimating K for cells by assuming a linear relationship between HSI and maximum densities found in the literature (Table 3). We then scaled density by the area of cells and filtered areas of the landscape that could not support at least one territory or colony (for *Myotis* spp.), constrained by a maximum territory or home range (for *Myotis* spp.) size found in the literature. This process more realistically captured the interaction between spatial and resource limitations inherent in estimating K (Fig. 15). Shifts in distribution over time due to the climate's effect on habitat were captured by subsequent changes in K.

We used a relative productivity index (Bonnot et al. 2011) model to link breeding productivity of birds affected by parasitism to habitat fragmentation. The index (0–1) modifies reproductive success of birds breeding in fragmented landscapes and proximate to edge based on the amount of forest cover in a 10-km radius and edge within a 200-m radius.

Finally, we applied dispersal models to estimate cell-based movements of dispersing individuals to the surrounding landscape based on species-specific functions of distance between cells, weighted by K of the destination cell (Bonnot et al. 2017). Thus, future changes in dispersal patterns reflected shifts in the distribution of focal species in the region.

Population modeling. We linked the regional populations of focal species to landscapes by treating ecological subsections as subpopulations and summarizing their demographics for each subsection. The region contains 32 subsections (Cleland et al. 2007), which we delineated into 45 subpopulations. For each subpopulation, we summarized results of the habitat models to obtain estimates of initial abundance (year 2010) and K at each decade. We averaged cell reproductive indices in each subpopulation, weighted by their K in each decade, so that productivity estimates reflected any changes in distribution of birds over time. For each decade, dispersal movements were summarized by subpopulation and standardized to obtain relative rates of dispersal from each subpopulation to surrounding ones (Bonnot et al. 2011). Because we modeled population dynamics annually, we calculated yearly values of spatial demographics by linearly interpolating between decadal estimates and stored them in spatio-temporal arrays.

We programed the population models in Program R (R Core Team 2013). We used female only, Lefkovitch matrix models comprising various life stages, dependent on species life history. We specified stage-specific survival and fertility rates from the literature and assumed a post-breeding census (Table 3). For species affected by parasitism, we multiplied relative productivity indices by published estimates of maternity to obtain population-specific fertility estimates in each decade. We specified breeding and natal dispersal rates for relevant stages that determined the proportion dispersing each year and redistributed them among the subpopulations according to multinomial distributions with probabilities equal to the relative dispersal rates for that year. We used a Ricker contest and modified ceiling density dependence model to incorporate density dependence. The modification to the commonly referred to ceiling density dependence (Akçakaya and Brook 2009) prohibits individuals over K in a population from breeding but allows them to remain in the population or disperse.

We used Monte Carlo simulations to induce parameter uncertainty and stochasticity in our population dynamics. We simulated parameter uncertainty by sampling a different survival and fertility rate in each of the 1,000 iterations from beta and gamma distributions, respectively, with means equal to their overall estimates and corresponding error, derived from the literature (McGowan et al. 2011). In each iteration, the rates drawn were used to construct beta and lognormal distributions, from which annual survival and fertility rates could be drawn. Patterns in annual survival rates were correlated among subpopulations based on a negative exponential relationship with the distances among them (Bonnot et al. 2017). We based variances for these distributions on the amount of temporal variation empirically observed for survival or reproduction in the literature. In each year, we modeled demographic stochasticity by drawing the number of survivors and the number of young produced in each stage each year from binomial and Poisson distributions, respectively.

Tradeoffs

Performance metrics. We examined the consequences of each scenario using three types of metrics of species viability produced by the DLMPs: the carrying capacity of species, estimated abundance, and species persistence or, conversely, species risk (Fig. 16). Each metric conveys important aspects of conservation planning (see Box 5). Carrying capacity provided a straightforward assessment of the amount and quality of habitat available to each species under habitat restoration scenarios. Population abundance provided insight into how each species responded to the scenarios. However, ultimately we relied only on the estimated risk of a 25 percent decline for the Ozark Highland's populations of each species to measure viability performance. With the DLMPs, risk captures the effects of habitat and demographic processes on population dynamics over time and under variation to measure viability in a manner more aligned with the goals of landscape conservation design (Box 5). Estimated as the probability of surpassing a specified threshold in population size, risk simultaneously conveys responses of wildlife populations to conservation scenarios and the uncertainty in those responses, providing managers with a more intuitive and defensible way of comparing choices and sound decisions (Millspaugh et al. 2009). The lack of demographic data for eastern collared lizards precluded us from modeling their population and estimating risk. Instead, we considered only estimates of habitat acreage available to that species under each scenario.

When quantifying risk, we incorporated the uncertainties inherent in DLMP projections and multiple climate futures. We pooled projections across climate models. We estimated the risk of species decline under each restoration scenario by calculating the proportion of model iterations for which each focal population declined by various percentages from the initial size through 2100 (Ackakaya 2002). We identified a 25 percent decline as the threshold to compare risk because most species were not at risk of major declines (based on the demographics in our models) and it provided greater contrasts across scenarios and species.

Resolving tradeoffs. We used multicriteria decision analysis to resolve the tradeoffs between competing objectives (i.e., disparate species responses) and facilitate decision making. With this approach, various methods are available to rank scenarios based on their performance (Gregory et al. 2012; also see Box 5). Given our use of risk-based metrics, we relied on multi-attribute utility theory to combine single species estimates into a single multispecies risk function (Beissinger et al. 2009, Nicholson and Possingham 2007). Nicholson and Possingham (2006) review the implications of various risk functions that we discussed with the team. For example, we considered a mini-max strategy that minimized the maximum risk across species (Drechsler and Burgman 2004, Polasky et al. 2011, Table 4). Ultimately, we identified the expected risk of extinction/decline as a useful metric that balances the needs of species facing low and high in an equitable manner. We calculated the expected risk as the average risk across species under each scenario. We also considered, but did not implement, a weighted average based on species conservation status or the responsibility of the Ozark Highlands in conserving each species relative to other regions (e.g., see Table 4).

RESULTS

Overall effects of restoration on focal species were positive and presented evidence to support landscape conservation design. Restoration scenarios increased habitat for most species relative to the baseline scenario. Growth of focal populations either responded favorably to restoration scenarios or were unaffected (Fig. 17). As a result, restoration scenarios generally reduced risk of declines for focal wildlife under climate change and urbanization, compared to baseline projections. Despite the general effectiveness of restoration, species-specific responses to scenarios varied in complex ways through interactions with landscape change processes such as urbanization and climate change and their individual demographic processes.

BOX 5. TRADEOFFS IN PERFORMANCE METRICS

Performance Metrics

In structured decision making, performance metrics (also called measurable attributes or evaluation criteria) are the quantities that measure consequences of scenarios related to the objectives (Thompson et al. 2013). A variety of metrics have been used to address objectives related to viability or biodiversity. Early systematic conservation planning examples often sought to design networks of protected reserves focused on habitat area, landscape composition or fragmentation, or representativeness (Margules and Pressey 2000). Species persistence could also be approximated using models based on patch occupancy and metapopulation theory (Beissinger and Westphal 1998). These metrics lent themselves to quantitative algorithms and software that would produce optimal designs (e.g., Marxan, Ball and Possingham 2000, and Zonation, Moilanen 2007). But, as LCCs and other partnerships increasingly consider a broader array of conservation approaches that span multiple sectors (e.g., habitat restoration on public and private lands or species reintroductions). Conservation decisions will become less structured and amenable to optimization and rely more on comparing and learning from the relative performance of scenarios (Seppelt et al. 2013). In these cases, DLMPs provide the flexibility to evaluate a variety of conservation actions that target landscapes and populations.

The use of DLMPs here allowed quantification of three types of metrics to assess how each scenario performed regarding species viability objectives: the amount and quality of habitat (quantified by carrying capacity), population abundance, and species persistence or, conversely, species risk. While all are useful in understanding the benefits and shortfalls of scenarios, more enlightened decisions are possible as one metric builds upon the information of the other (Fig. 16).

Resolving Tradeoffs

Decisions regarding landscape conservation design must not only consider competing objectives stemming from stakeholder interests, policy goals, and various costs, but also in the form of species impacts if individual species respond differently to scenarios (Drechsler and Burgman 2004). In these cases, multicriteria decision analysis, commonly used to resolve tradeoffs across competing objectives in the overall decisions, can be applied to objectively identify scenarios that best resolve tradeoffs in the viability of multiple species (Beissinger et al. 2009).

Multicriteria decision analysis provides a set of systematic methods to combine risk and other performance or cost/benefit information to choose alternatives (Gregory et al. 2012). Multiple methods fall under the umbrella of this approach and the utility of each will depend on the decision process.

Outranking is similar to a voting system, where an overall ranking can be derived by finding the scenario that ranks highest for the most species (Beissinger et al. 2009). When identifying the best decisions for the persistence of all species, the use of rankings has been proven to result in decisions that are robust to uncertainty (Nicholson and Possingham 2007).

Habitat

- Estimated from habitat model in the DLMP
- Incorporates the amount of area and quality of habitat
- Often most straightforward metric to calculate for alternative scenarios
- Easily monitored over time
- Does not consider demographic processes affecting population growth
- Could mislead decisions when other threats are affecting population growth (Bonnot et al. 2013)

Abundance

- Estimated from population model in the DLMP
- Integrates changes in habitat over time with demographic process such as survival and reproduction
- Most common and direct characteristic of populations targeted by managers
- Often estimated with substantial error or uncertainty that precludes decision
- Does not capture the sustainability of populations (Morris and Doak 2000)

Persistence (risk)

- Estimated from population model in DLMP
- Calculated from distribution of projected abundances from stochastic model runs
- Describes the probability of meeting or surpassing specified thresholds of population size
- Simultaneously conveys responses of populations to scenarios and the uncertainty in those responses (Marcot et al. 2015)
- Fundamental to viability and therefore, a central tenet of conservation (Beissinger et al. 2009)
- Risk is an intuitive metric for making decisions (Millspaugh et al. 2009)

Figure 16.—The use of dynamic-landscape metapopulation models provides three performance metrics for which to compare the consequences of landscape conservation design scenarios. Each successive metric addresses potential limitations of the previous to better inform decisions. The appropriateness of each metric will depend on the context of the decision and the data available to modelers and decision makers. Image by the authors.

Box 5 continued.

Analytic hierarchy processes use pairwise comparisons of criteria that ask how much more important one scenario is than the other. It is generally thought to be simple, and it can be flexible when multiple stakeholders are involved (Huang et al. 2011).

Multi-attribute utility theory describes the process of combining scores across objectives to form a single utility function for comparison (Gregory et al. 2012). This method is well suited for combining multiple species risk because of their common currency and probabilistic nature. Table 4 demonstrates examples of multispecies risk metrics presented in Nicholson and Possingham (2006). Comparisons between the scenario rankings for individual risk for the six focal species under the seven conservation design scenarios in the Ozark Highlands (highly varied) and the multispecies metrics (more consistent) illustrate the resolution of tradeoffs that can be achieved with these approaches. Each metric is the mathematical representation of an approach to conservation and emphasizes different levels of risk; therefore, Nicholson and Possingham (2006) recommend careful consideration to choose metric in line with conservation goals.

Table 4.—Identifying a landscape conservation design scenario that best supports the viability of focal species can present tradeoffs among scenarios when individual species' risk varies among scenarios. In the Ozark Highlands, dynamic-landscape metapopulation models were used to evaluate the effectiveness of seven potential designs for habitat restoration to minimize the risk of decline of six focal species. Multi-attribute utility theory can be used to combine individual risks under each scenario into various multispecies metrics that emphasize different aspects of risk depending on decision makers, priorities, and concerns. Presented here is a selection of metrics from Nicholson and Possingham (2006). Within each set, scenarios are color coded by risk (%) of a 25 percent decline in abundance (red to green, red being highest).

				Future landscape			Current landscape			Random
				Combined lands		Protected	Combined lands		Protected	Combined lands
			Base	Full acreage	Half acreage	Full acreage	Full acreage	Half acreage	Full acreage	Full acreage
	Species	Percent of SWAP plans enlisted[a]								
Individual risks	Prairie warbler	14	59	26	45	32	28	40	37	50
	Wood thrush	18	39	39	37	37	39	38	40	37
	Prothonotary warbler	17	34	34	35	33	31	32	34	37
	Cerulean warbler	22	53	30	33	28	29	32	28	46
	Brown-headed nuthatch	6	7	5	3	5	13	4	7	3
	Myotis spp.	23	36	35	35	36	35	34	36	35
	Metric	Description								
Multispecies risk	Mini-max	Minimizes the maximum risk	59	39	45	37	39	40	40	50
	Joint risk of ≥ 1 decline	Minimizes the risk of one or more declines	95	87	90	87	88	89	89	93
	Joint risk of all declines	Minimizes the risk of all species declining	0.11	0.02	0.02	0.02	0.04	0.02	0.04	0.03
	Expected risk of decline	Minimizes the expected number of declines	38	28	31	28	29	30	30	35
	Expected risk of decline (weighted)	Minimizes the expected number of declines, weighted by the number of SWAP plans in which species is listed	42	31	34	31	31	33	33	38
	Relative increase in risk	Minimizes the increase in risk from the best case scenario for each species	12	2	5	2	3	4	4	8

[a] Relative weighting measure of conservation concern based on the number of State Wildlife Action Plans (SWAPs) for which each species is listed in. Species with higher weights receive more concern.

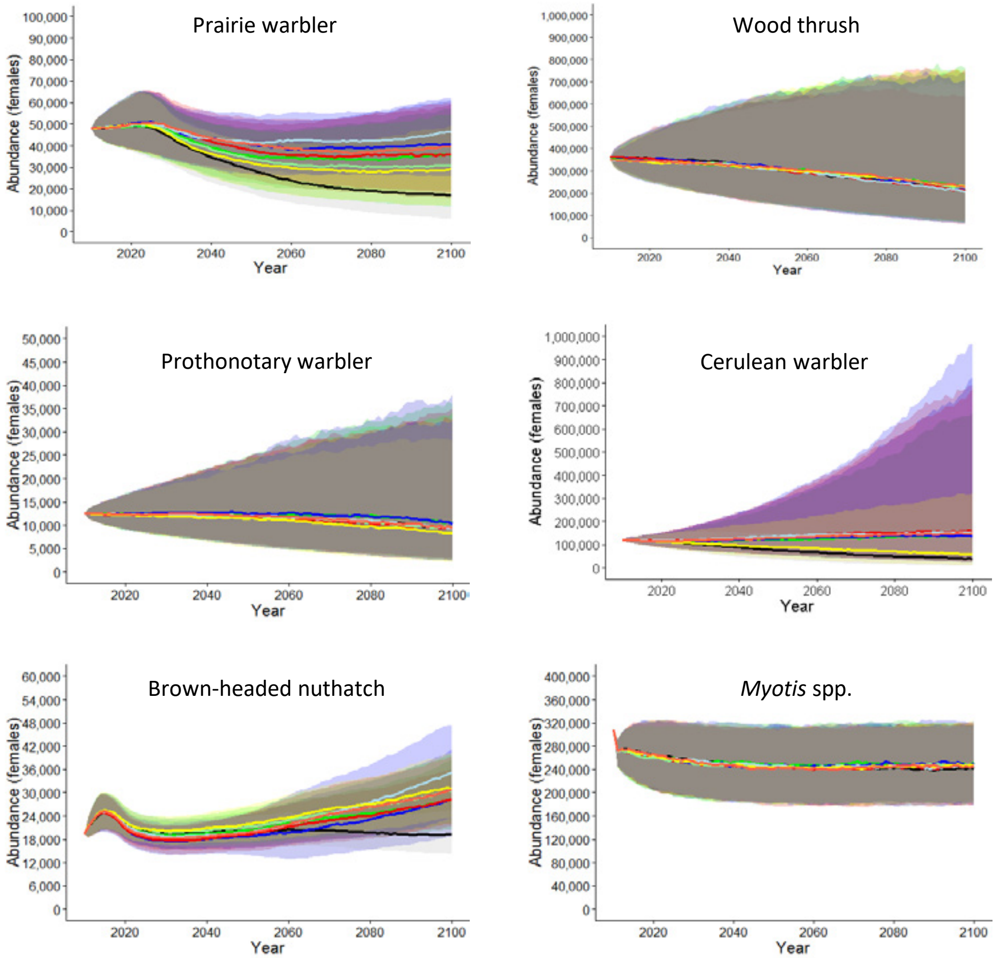

Figure 17.—Projected population growth of focal wildlife species in the Ozark Highlands under alternative habitat restoration scenarios, proposed by a team of partners. Predictions are based on dynamic-landscape metapopulation models applied to landscapes projected under climate change and urbanization. Projections are pooled across model of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Results for brown-headed nuthatch assume added translocation efforts. Image by the authors.

Species Responses to Baseline Scenario

In the absence of restoration, landscape changes due to climate and forest succession and management produced substantial changes in habitat for the focal species. For example, habitat for prairie warblers, cerulean warblers, and brown-headed nuthatch involves particular aspects of forest structure such as open canopies, increased small stems, or mid-range stocking rates. Following historic losses in these habitats, our initial carrying capacity estimates indicated that the Ozark Highlands could support >96,000 breeding individuals for each of these species (Table 5). However, with an inability of current forest management to maintain that structure, natural forest succession under the current climate resulted in dense, mature, closed forests, causing 34 percent, 85 percent, and 71 percent decreases in carrying capacities of these species

Table 5.—Cumulative effects of restoration scenarios and climate change on amount of habitat for focal wildlife species in the Ozark Highlands by 2100. Final carrying capacities for the baseline scenario under current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change models, are listed. The relative impacts of each restoration scenario on final carrying capacity relative to the baseline are presented as the percent difference. Scenarios that resulted in lower capacity compared to the baseline scenario are highlighted in red.

			Prioritized by current landscape condition			Prioritized by future landscape condition			Random restoration
			Combined lands[a]		Protected	Combined lands		Protected	Combined lands
		Baseline	Full acreage (%)	Half acreage (%)	Full acreage (%)	Full acreage (%)	Half acreage (%)	Full acreage (%)	Full acreage (%)
Prairie warbler (96,008)[b]									
	Current	63,603	4	4	4	14	6	6	-15
	CGCM.T47-A2	69,185	-22	-17	9	11	6	7	-15
	GFDL-A1FI	82,841	-3%	-1%	1%	1%	1%	4%	-16
Wood thrush (3,046,160)									
	Current	3,450,832	-4	-2	-2	-8	-6	-7	-7
	CGCM.T47-A2	3,395,167	-10	-9	-3	-8	-6	-7	-7
	GFDL-A1FI	3,345,546	-4	-2	-2	-8	-6	-6	-7
Prothonotary warbler (255,200)									
	Current	394,119	11	5	4	10	5	4	10
	CGCM.T47-A2	396,326	31	27	4	10	5	4	10
	GFDL-A1FI	398,656	11	5	4	10	5	4	10
Cerulean warbler (120,870)									
	Current	238,021	602	335	370	771	385	427	496
	CGCM.T47-A2	622,367	257	144	141	287	144	159	186
	GFDL-A1FI	2,505,305	44	25	25	57	29	30	34
Brown-headed nuthatch (106,191)									
	Current	31,008	33	16	18	45	20	30	10
	CGCM.T47-A2	38,105	-18	-9	5	31	14	17	6
	GFDL-A1FI	47,410	14	8	5	24	10	10	-2
Myotis spp.[c] (597,608)									
	Current	666,782	0	0	0	0	0	0	0
	CGCM.T47-A2	667,744	2	2	0	0	0	0	0
	GFDL-A1FI	678,216	0	0	0	-1	-1	-1	-1

[a]Combined lands comprise both privately owned and protected lands.
[b]Initial carrying capacities (breeding individuals) in year 2010.
[c]*Myotis* spp. = *Myotis sodalist* and *Myotis septentrionalis.*

by 2100 (Figs. 19, 31, 39). The baseline scenario for these species under the two climate change models initially projected similar habitat losses but by the latter half of the century reduced precipitation and elevated temperatures began to alter forest structure, reducing tree stocking and resulting in lower canopy cover and more open forest structure that allowed prairie warbler and cerulean warbler habitat to rebound. It is also possible that slight shifts in forest composition to pine species may have played a role in the positive effects of climate change on brown-headed nuthatch carrying capacity (Table 5). In contrast, the baseline scenario projections for wood thrush, prothonotary warbler, and *Myotis* spp. (species associated with characteristics of mature forests) provided increased habitat through the first half of the century driven by forest succession (Figs. 23, 27, 35). Under the current climate, these increases plateaued later in the century as more forests reached older stages. However, wood thrush carrying capacity was generally lower under the GFDL-A1FI climate scenario, a negative response to the same processes benefiting prairie warblers.

Species responses to the baseline changes in habitat further depended on varied demographic processes affecting population growth. Except for brown-headed nuthatch, all focal populations declined ≥19 percent through 2100 in the baseline scenario (Table 6). Driven by early habitat losses, prairie warblers and cerulean warblers declined early in the simulations. By midcentury, prairie warbler began responding to the increased habitat created with the GFDL-A1FI, eventually reaching abundances twice that under the current climate model (Fig. 20). Cerulean warblers also saw increased habitat later in the century from climate change, but the population had little time to react (Fig. 32). As such their declines were similar for the baseline scenario under all climate models. Despite increases in habitat, we projected declines in wood thrush, prothonotary warblers, and *Myotis* spp. for the baseline scenario under all climate models. Declines in wood thrush and prothonotary warblers likely stemmed from the impacts of fragmentation on reproduction (Figs. 24, 28). *Myotis* spp. were unaffected by changes in habitat (Fig. 36). Projections for brown-headed nuthatch suggested they were most limited by habitat (Fig. 40). Their population growth closely tracked changes in carrying capacity, resulting in projections of a declining, stable, and growing population under the current, CGCM.T47-A2, and GFDL-A1FI climates, respectively (Table 6).

Ultimately, we projected moderate to high risk of decline for focal species across all climate models, without restoration. Brown-headed nuthatch had a 7 percent chance of a 25 percent decline by 2100 and was the only species with risk <30 percent (Table 7). Prairie warblers and cerulean warblers were most at risk with a 59 percent and 53 percent probability of a 25 percent decline, respectively (Figs. 22, 34). Despite available habitat, wood thrush, prothonotary warblers, and *Myotis* spp. had a 34-39 percent risk of a 25 percent decline (Figs. 26, 30, 38).

Species Responses to Restoration Scenarios

Restoration scenarios interacted with baseline climate-driven changes in the landscape to variably affect species habitat over time. Scenarios that restored only protected lands failed to reach acreage targets, given limited extents of these lands, but scenarios that combined private with protected lands achieved acreage goals. Wood thrush habitat (mature and closed forests) was abundant in the geography and projected to increase in the absence of restoration. But restoration of woodlands and glades opened forest structure, decreasing wood thrush K 2-10 percent below baseline projections (Table 5). Except for *Myotis* spp., habitat for the other focal species increased under restoration scenarios. Restoration increased K over the baseline scenario most for cerulean warblers (335-662 percent under current climate; Fig. 31, Table 5). These increases were smaller under the climate change models, given K was already elevated due to climate effects. Prairie warbler, brown-headed nuthatch, and prothonotary warbler K's also increased under most restoration scenarios; however, prairie warbler habitat declined under the random restoration scenario (Fig. 19). Habitat for these species increased immediately under restoration in contrast to more gradual climate-driven changes on the forest and habitat. *Myotis* spp. habitat was relatively unaffected by climate change or restoration (K changed <2 percent; Fig. 35).

We projected complex responses of focal populations to the changes in their habitat from restoration. In addition to their positive responses to climate change, prairie warblers and cerulean warblers showed the greatest responses to conservation as their abundances under some scenarios doubled that projected with no restoration (Table 6). Although prairie warbler carrying capacity increased <10 percent under the scenarios, the population responded greatly, in some cases reversing declines and cutting the risk of significant decline by half (Table 7; Fig. 21).

Table 6.—Projected population sizes of focal species in the Ozark Highlands by 2100 under restoration scenarios amidst landscape changes from urbanization and climate change. Abundance of breeding birds and bats is given for each alternative scenario under models of the current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. The impact of each scenario on populations relative to the baseline scenario is indicated by the percent difference value (italics and underlined). Scenarios that resulted in lower population sizes compared to no action scenario are highlighted in red.

		Prioritized for current Landscape						Prioritized for future landscape						Random	
		Combined lands[a]				Protected		Combined lands				Protected		Combined lands	
	Baseline	Full acreage[b]		Half acreage		Full acreage		Full acreage		Half acreage		Full acreage		Full acreage	
Prairie warbler	(48,284)[c]		%		%		%		%		%		%		%
Current	12,700	41,351	226	31,420	147	23,447	85	42,834	237	28,402	124	33,300	162	22,195	75
CGCM.T47-A2	14,263	34,386	141	33,484	135	43,507	205	44,617	213	25,592	79	37,160	161	27,665	94
GFDL-A1FI	27,763	57,227	106	49,696	79	45,142	63	55,633	100	41,112	48	52,099	88	45,165	63
Wood thrush	(363,946)														
Current	238,677	246,051	3	216,293	-9	205,302	-14	233,926	-2	254,564	7	248,158	4	222,855	-7
CGCM.T47-A2	201,696	191,735	-5	244,467	21	204,123	1	191,048	-5	231,364	15	250,522	24	256,087	27
GFDL-A1FI	207,960	219,760	6	204,340	-2	217,254	4	199,564	-4	224,334	8	234,948	13	240,218	16
Prothonotary warbler	(12,533)														
Current	10,126	10,516	4	10,690	6	9,393	-7	10,061	-1	9,764	-4	9,790	-3	7,598	-25
CGCM.T47-A2	9,158	9,123	0	10,252	12	9,099	-1	8,958	-2	9,063	-1	8,920	-3	8,726	-5
GFDL-A1FI	9,497	11,949	26	10,986	16	9,388	-1	9,440	-1	9,462	0	9,783	3	8,530	-10
Cerulean warbler	(120,870)														
Current	32,661	138,023	323	125,135	283	161,945	396	134,234	311	151,750	365	205,055	528	56,573	73
CGCM.T47-A2	45,227	121,658	169	158,989	252	142,828	216	139,422	208	127,985	-100	166,779	269	56,325	25
GFDL-A1FI	40,784	161,390	296	118,445	190	162,003	297	149,483	267	123,344	202	136,227	234	57,406	41
Brown-headed nuthatch[d]	(19,428)														
Current	15,672	29,053	85	25,550	63	26,420	69	32,255	106	27,293	74	28,677	83	28,891	84
CGCM.T47-A2	19,584	22,213	13	26,421	35	26,466	35	34,666	77	30,202	54	30,238	54	32,017	63
GFDL-A1FI	22,600	37,109	64	35,178	56	32,584	44	39,297	74	34,722	54	33,575	49	33,438	48
Myotis spp.[e]	(307,237)														
Current	237,475	242,973	2	255,057	7	242,241	2	251,085	6	248,363	5	244,575	3	251,458	6
CGCM.T47-A2	245,671	254,538	4	252,205	3	250,324	2	243,760	-1	248,128	1	247,914	1	246,443	0
GFDL-A1FI	244,397	250,901	3	247,283	1	246,959	1	244,590	0	248,751	2	246,533	1	244,231	0

[a]Combined lands comprise both privately owned and protected lands.
[b]Acreage targets from Central Hardwoods Joint Venture. Full acreage = 1.2 million ha. Half acreage = 0.6 million ha.
[c]Initial estimate of abundance (breeding individuals) in year 2010.
[d]Results for brown-headed nuthatch assume added translocation efforts.
[e]*Myotis* spp. = *Myotis sodalist* and *Myotis septentrionalis*.

Table 7.—Estimated risk of population decline for focal wildlife species in the Ozark Highlands under proposed restoration scenarios amidst landscape change from urbanization and climate change. Values represent the probability (%) of the population declining by 25 percent from its 2010 abundance through 2100. Estimates are pooled across future climates modeled under the current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change.

		Prioritized by current landscape condition			Prioritized by future landscape condition			Random restoration
		Combined lands[a]		Protected	Combined lands		Protected	Combined lands
Species	Base	Full acreage	Half acreage	Full acreage	Full acreage	Half acreage	Full acreage	Full acreage
Prairie warbler	59	26	45	32	28	40	37	50
Wood thrush	39	39	37	37	39	38	40	37
Prothonotary warbler	34	34	35	33	31	32	34	37
Cerulean warbler	53	30	33	28	29	32	28	46
Brown-headed nuthatch[b]	7	5	3	5	13	4	7	3
Myotis spp.[c]	36	35	35	36	35	34	36	35
Average	38	28	31	28	29	30	30	35

[a]Combined lands comprise both privately owned and protected lands.
[b]Results for brown-headed nuthatch assume added translocation efforts.
[c]*Myotis* spp. = *Myotis sodalist* and *Myotis septentrionalis*.

In contrast to the late habitat increases cerulean warblers experienced under the GFDL-A1FI model, the population had the time to respond to habitat provided by restoration earlier in the simulations (Fig. 33). Prothonotary warbler, and *Myotis* spp. population growth was largely unaffected by restoration (Figs. 29, 37). Because the wood thrush population is primarily influenced by fragmentation and not habitat availability, their population growth was largely unaffected by restoration. Projected wood thrush abundance under different restoration scenarios ranged from 14 percent below to 27 percent above base projections (Fig. 25). However, the substantial uncertainty surrounding these projections suggests a lack of an overall response. Indeed, risk of declining 25 percent varied <2 points across scenarios when pooling climate models (Table 7). Risk of decline for prothonotary warblers and *Myotis* spp. also remained relatively unchanged across scenarios (Figs. 30, 38).

Despite being initially limited by habitat and restoration increasing their habitat region-wide, brown-headed nuthatch populations declined with restoration alone (Fig. 41a). Restoration of open-pine forests that provide their habitat mainly took place in the Current River Hills and Black River Border subsections, outside of the Boston Hills and adjacent sections where this bird's distribution is currently limited. Instead, the majority of restoration that occurred where the population currently resides targeted oak woodlands, thus removing their habitat that decreased K and caused population declines. Scenarios that incorporated translocation of brown-headed nuthatch to the Current River Hills subsection in addition to restoration overcame the dispersal limitations and allowed these birds to utilize the new habitat, leading to population growth (Fig. 41b). With added translocation, risk of decline for brown-headed nuthatch under restoration scenarios fell below 5 percent (Table 7).

The various complex processes that affected how each species responded to restoration ultimately resulted in a different ideal scenario for each species. The scenario that targeted the full acreage objectives on both private and protected lands and prioritized based on future landscapes best reduced risk, on average, across all species (Fig. 18). This scenario also provided the most habitat for Eastern collared lizards and thus was identified as the best scenario by the team (Fig. 45).

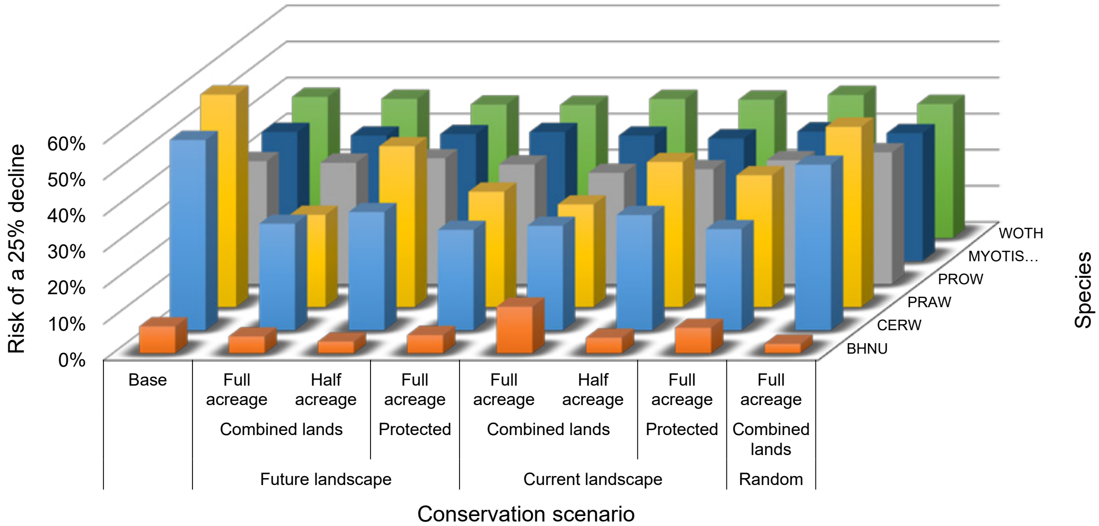

Figure 18.—Estimated risk of populations declining by 25 percent under alternative habitat restoration scenarios, proposed by the Ozark Highlands team. Risk is pooled across projections under urbanization and three future climate models that range from continuation of the current climate, moderate climate change, and extreme climate change. Focal species include brown-headed nuthatch-BHNU, cerulean warblers-CERW, prairie warblers-PRAW, prothonotary warblers-PROW, *Myotis* spp. bats, and wood thrush-WOTH. Results for brown-headed nuthatch assume added translocation efforts. Image by the authors.

BOX 6. PRAIRIE WARBLER

Habitat Model

Throughout most of the Eastern United States and in Southern Ontario, the prairie warbler breeds in shrubby old fields, early stage regenerating forests, and glades and woodlands (Nolan et al. 2014). Open canopy and shrubby understory are important structural components in addition to patch size and the predominance of forest in the surrounding landscape (Tirpak et al. 2009). The prairie warbler nests in shrubs and small trees away from forest edge (Nolan et al. 2014, Woodward et al. 2001).

Photo by Allan Hopkins, Flickr Creative Commons, https://www.flickr.com/photos/.

We used the Habitat Suitability Index (HSI) model by Tirpak et al. (2009) to model prairie warbler habitat in the Ozark Highlands. Components in the model include a suitability index assigned to combinations of landform, landcover, and successional age class on the basis of habitat associations. In other variables, prairie warbler's association with larger forest patches and shrubby understory is incorporated through logistic functions of patch size and increased small-stem densities. Suitability also declines with increasing canopy cover and adjacency to mature forest stands.

Population Model

We based K on the maximum observed density in the region (0.99, Fink 2003) and set K = 1.0 pairs/ha when a cell's HSI = 1.0 (Table 1). We set initial abundances of prairie warblers at 50 percent of K, which is similar to the average density (0.52 pairs/ha) observed in Missouri (Brito-Aguilar 2005, Fink 2003, Thompson et al. 1992, Wallendorf et al. 2007).

We set adult survival of prairie warblers at 0.60, which we obtained by averaging rates presented by Nolan et al. (2014) with 0.65 and Lehnen and Rodewald (2009) at 0.55. For juvenile survival of prairie warblers, we used 0.32 based on Nolan et al. (2014) estimates of post-fledging and overwinter survival.

Box 6 continued.

Results

Prairie warbler habitat declined in the first 2 decades, driven by interactions between succession and management. By 2030, declining habitats stabilized under current climate. Under climate change models, habitat increased as patterns in temperature and precipitation began opening forests. These patterns are reflected in the population projections where under the current climate and moderate climate change (CGCM. T47-A2), prairie warblers continued declining. Projections under extreme climate change (GFDL-A1FI) showed declines stopping by 2070, after which the population began growing again.

Although restoration scenarios only moderately affected habitat, these activities generally benefited prairie warblers. All scenarios, except for those with a random prioritization and two under the CGCM.T47-A2 climate model, increased habitat marginally (<10 percent). These changes, however, effectively slowed and, for scenarios restoring full acreage objectives, reversed prairie warbler declines. Under the GFDL-A1FI climate model, scenarios that targeted full acreages even resulted in greater abundances in 2100 than initial estimates. Projections pooled across climate models identified that restoring full acreage, on both private and protected lands, prioritized by future conditions posed the greatest benefit to prairie warblers. This scenario resulted in abundances twice that of the baseline scenario with no conservation and reduced the risk of losing a quarter of the population by half (baseline – 59 percent; future, combined, full – 26 percent).

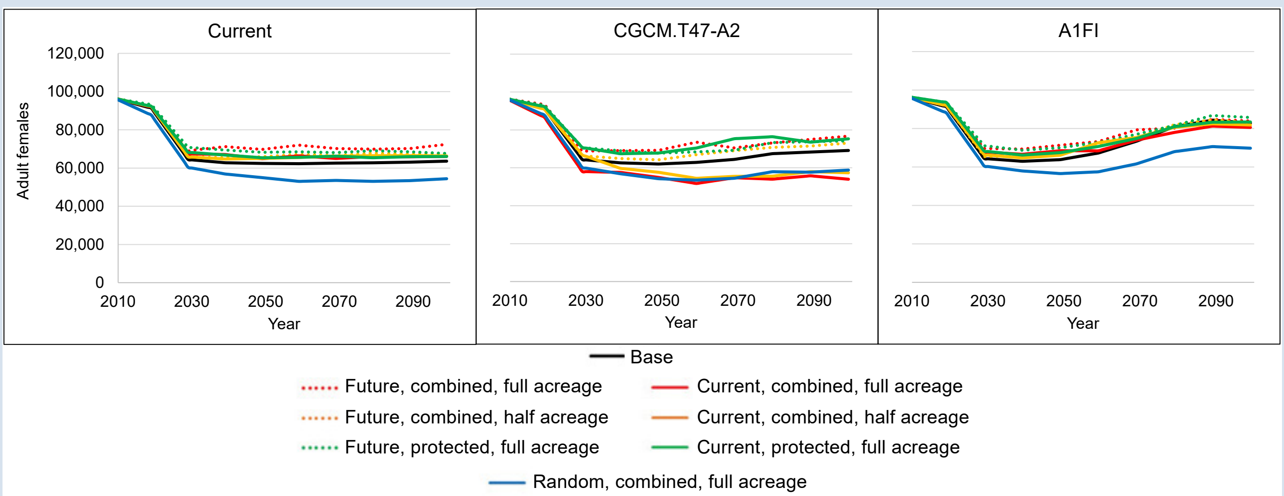

Figure 19.—Projected changes in carrying capacity of adult female prairie warblers in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

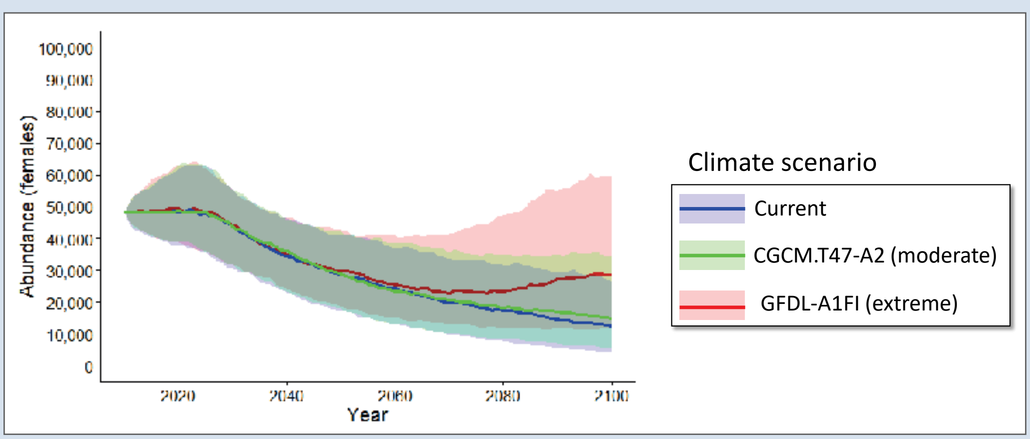

Figure 20.—Projected population growth of prairie warblers in the Ozark Highlands for a baseline scenario of no forest restoration. Predictions obtained from dynamic-landscape metapopulation models applied to landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

Box 6 continued.

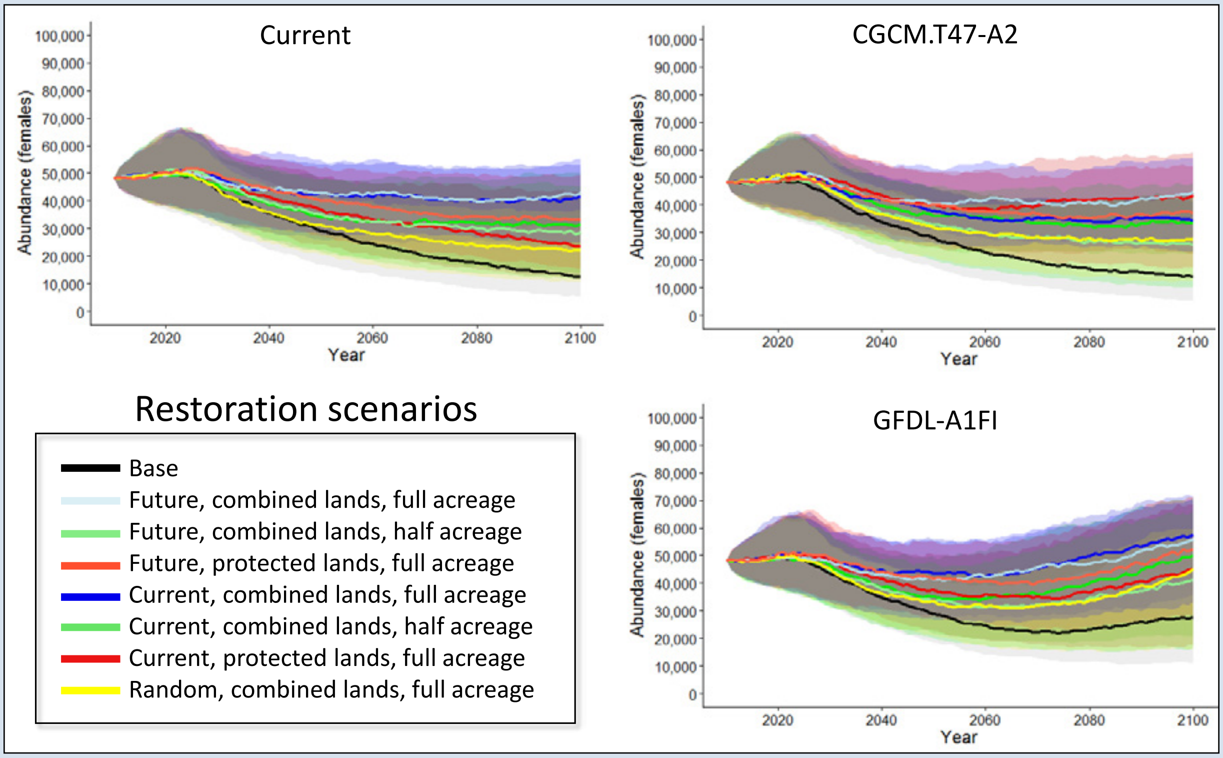

Figure 21.—Projected population growth of prairie warblers in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

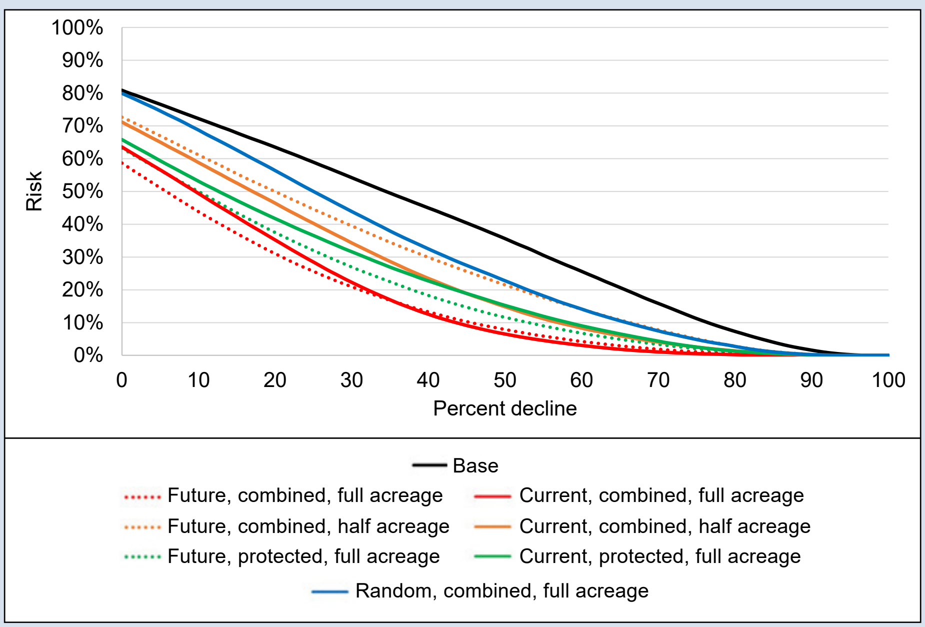

Figure 22.—Projected risk of the Ozark Highlands population of prairie warblers declining by various levels under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Image by the authors.

BOX 7. WOOD THRUSH

Habitat Model

This thrush inhabits a wide variety of deciduous and mixed forests. Key primary habitat features are a sub canopy layer of shrubs, shade, moist soil, and leaf litter, which enhance feeding and nesting (Evans et al. 2011). These features are reflected in the species' use of mature hardwood and mixed forests with relatively closed canopies (Bell and Whitmore 2000, Evans et al. 2011). Wood thrush display area sensitivity in productivity but not in occupancy of habitats (Tirpak et al. 2009).

Photo by Kelly Colgan Azar, Flickr Creative Commons, https://www.flickr.com/photos/.

We used the Tirpak et al. (2009) wood thrush HSI model. In the model, suitability of forests varied by landform, landcover, and seral stage. Higher suitability is found in forest patches >1 ha and in landscapes predominantly forested. Other variables in the model included logistic and inverse logistic relationships with canopy cover and small-stem density, respectively.

Population Model

We used demographic parameters from Bonnot et al. (2011) in the wood thrush DLMP model. We set K = 0.5 pairs/ha when HSI = 1.0. We based initial abundances of wood thrush breeding pairs on the 0.06 pairs/ha average, thus the initial abundance of wood thrush in each cell was 12 percent of K.

We specified adult wood thrush survival at 0.61, based on the average of the three estimates used in Bonnot et al. (2011). We set juvenile survival at a rate of 0.29 reported by Anders et al. (1997). We incorporated parasitism by applying mean RPI values to a maximum wood thrush maternity of 1.45 fem/fem/year, derived from three estimates from contiguous forests within the Central Hardwoods (1.21– Anders et al. 1997; 1.86 – Donovan et al. 1995; and 1.40 – Ford et al. 2001 – interior study site only) and Evans et al. (2011).

The model assumed that 90 percent of juveniles and 10 percent of adults dispersed annually. Dispersal distance was modeled as a negative exponential function with a mean distance of 70 km (Tittler et al. 2006). We accounted for density dependence using a modified ceiling model.

Results

The baseline scenarios projected increases in wood thrush habitat through 2070 due to natural forest succession. Gains in K were less substantial under climate change and by 2090 wood thrush K began declining under both climate change models. Despite these baseline increases in habitat, we projected steady declines in wood thrush through 2100, with slightly greater losses under climate change models (>40 percent decline through 2100) compared to the current climate (34 percent loss).

Of all focal species, wood thrush habitat was most affected by restoration activities. All restoration scenarios reduced K between 2-10 percent below the baseline scenarios. These reductions translated to as many as 330,000 fewer females supported under scenarios that targeted full acreage objectives.

Wood thrush population growth, however, responded little to loss of habitat from restoration scenarios. Rather, all scenarios, regardless of climate model, showed declines similar to baseline projections. Considerable uncertainty surrounded wood thrush projections. Based on 80 percent credible intervals (shaded portions of population graphs), projected final wood thrush abundances for most scenarios ranged between 100,000–800,000 females.

Box 7 continued.

Similar responses by wood thrush to restoration scenarios combined with high uncertainty resulted in marginal differences in risk to this population across scenarios. Thus, no one scenario was overwhelmingly better for wood thrush. Instead, we projected 37-40 percent risk that wood thrush numbers would decline by 25 percent by 2100, with scenarios that restored less acreage presenting slightly lower risk.

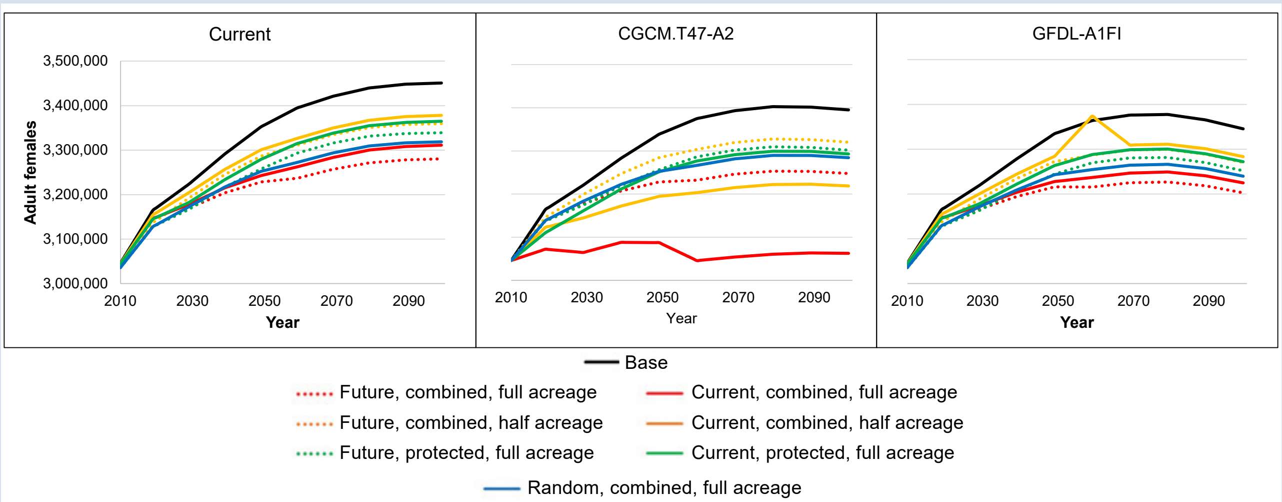

Figure 23.—Projected changes in carrying capacity of adult female wood thrush in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

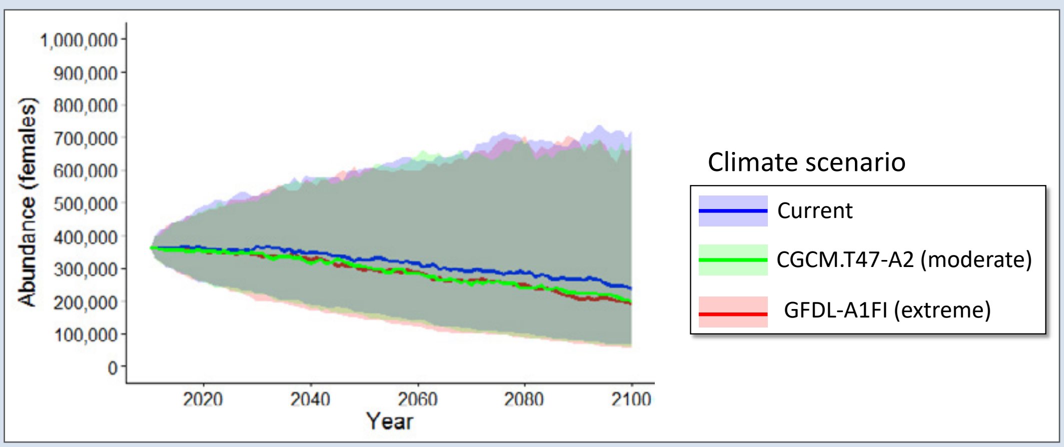

Figure 24.—Projected population growth of wood thrush in the Ozark Highlands for a baseline scenario of no forest restoration. Predictions obtained from dynamic-landscape metapopulation models applied to landscapes projected under urbanization and models of a current climate, and moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

Box 7 continued.

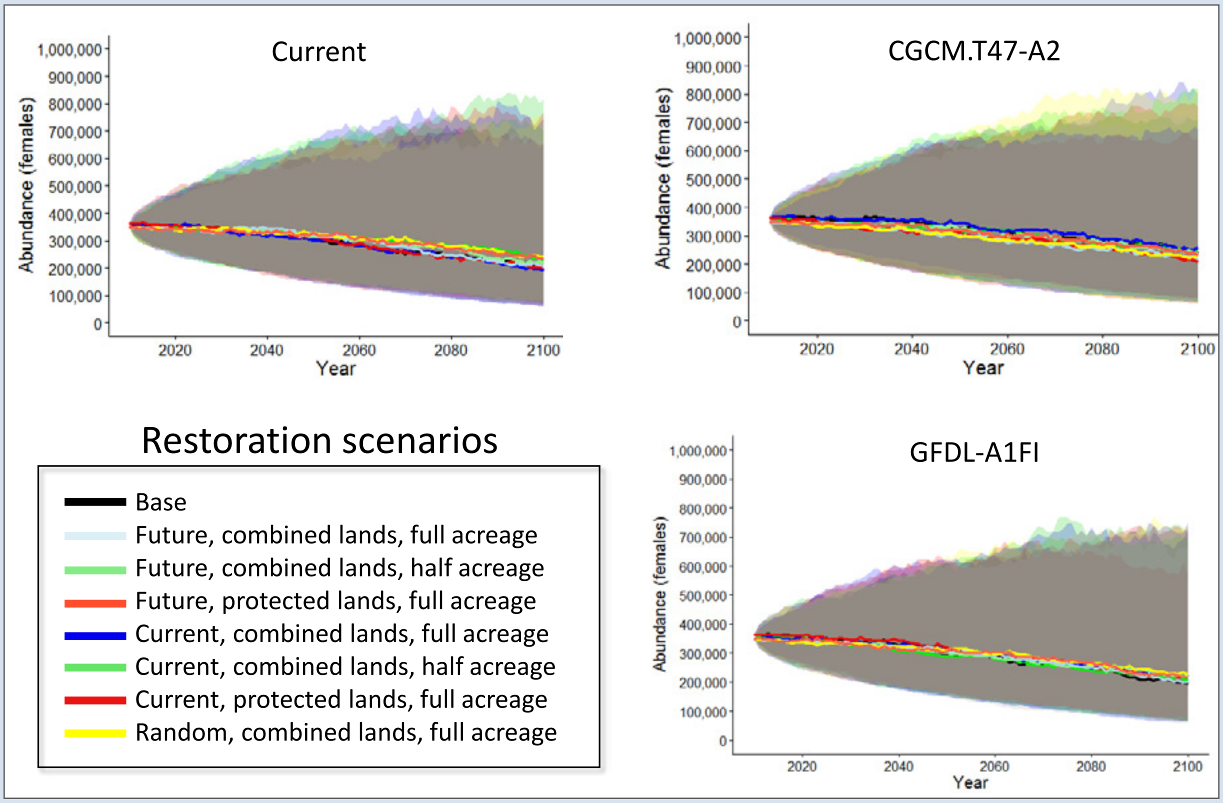

Figure 25.—Projected population growth of wood thrush in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

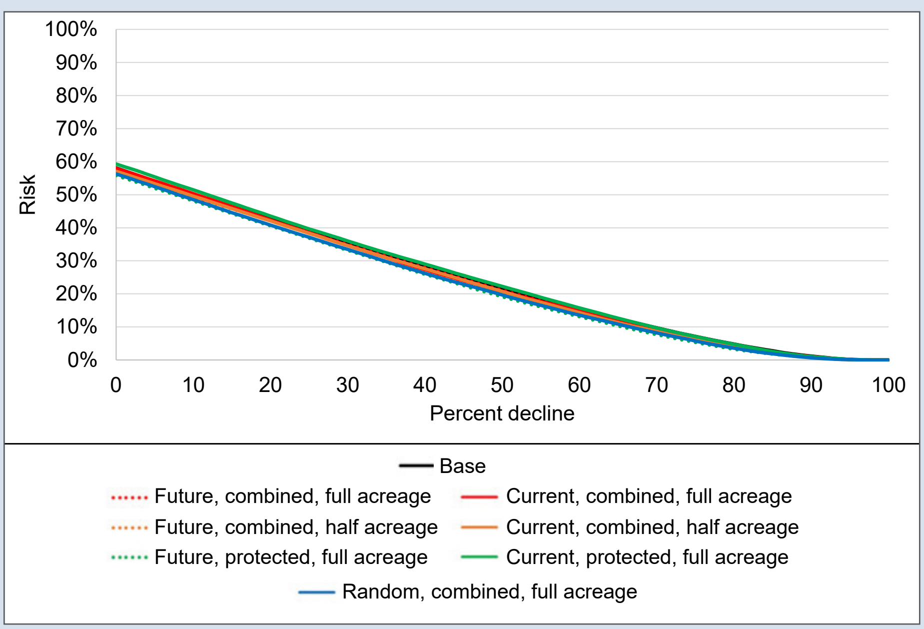

Figure 26.—Projected risk of the Ozark Highlands population of wood thrush declining by various levels under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Image by the authors.

BOX 8. PROTHONOTARY WARBLER

Habitat Model

Photo by Tom Benson, Flickr Creative Commons, https://www.flickr.com/photos/.

The prothonotary warbler is a bird of bottomland hardwood forests and other forested wetlands (Petit 1999). Key habitat features are presence of water near wooded areas with suitable cavity nest sites. Other important habitat correlates include low elevation, flat terrain, and shaded forest habitats with sparse understory (Petit 1999).

We approximated the HSI model for prothonotary warbler from Tirpak et al. (2009). In the model, suitability of forests varied by landform, landcover, and seral stage. Suitability is also based on the presence of water (identified by NLCD and national hydrology datasets) within 270 m. Two variables accounted for the interaction between forest patch size and the amount of forest within 1 km size in the area sensitivity in this species (Kahl et al. 1985). Finally, Tirpak et al. (2009) related suitability to snag density for nesting. Because LANDIS does not model snags, we approximated snag densities using the age of oldest trees projected on a site based on Rittenhouse et al. (2007).

Population Model

We estimated K and initial abundances from HSI model assuming density at K = 0.6 pairs/ha when HSI = 1.0 (Graber et al. 1983, Kleen 1973). We assumed a maximum territory size of 4 ha when using a moving window to identify habitat supporting territories (Reynolds 1997). We based initial abundances of prothonotary warblers on the 0.03 pairs/ha average or 5 percent of K (Twedt 2015).

We specified adult prothonotary warbler survival at 0.675 (Cooch et al. 1997, Hoover and Reetz 2006). We set juvenile survival at a rate of 0.18 (McKim-Louder et al. 2013). We incorporated parasitism by applying mean RPI values to a maximum maternity of 1.95 fem/fem/year (Petit and Petit 1996).

The model assumed that 90 percent of juveniles and 10 percent of adults dispersed annually. Dispersal distance was modeled as a negative exponential function with a mean distance of 70 km (Bonnot et al. 2011). We accounted for density dependence using a modified ceiling model.

Results

Similar to wood thrush, aging forests increased habitat for prothonotary warblers in baseline scenarios to support >100,000 additional birds by 2070. However, increases in K tapered off by 2100. These patterns in habitat were similar across climate models. Despite increased K, we projected >20 percent declines in the prothonotary warbler population due to forest fragmentation.

Restoration scenarios provided additional habitat, increasing K between 4-37 percent across scenarios and climate models. Alternatives that targeted full acreage objectives increased K the most. The added habitat, however, resulted in slightly larger declines. Only scenarios restoring habitat on combined lands, prioritized for the current landscape, resulted in greater population sizes compared to baseline scenarios. These were also the only scenarios that substantially reduced the risk of declines. Random restoration actually increased risk to prothonotary warblers in the Ozark Highlands.

Box 8 continued.

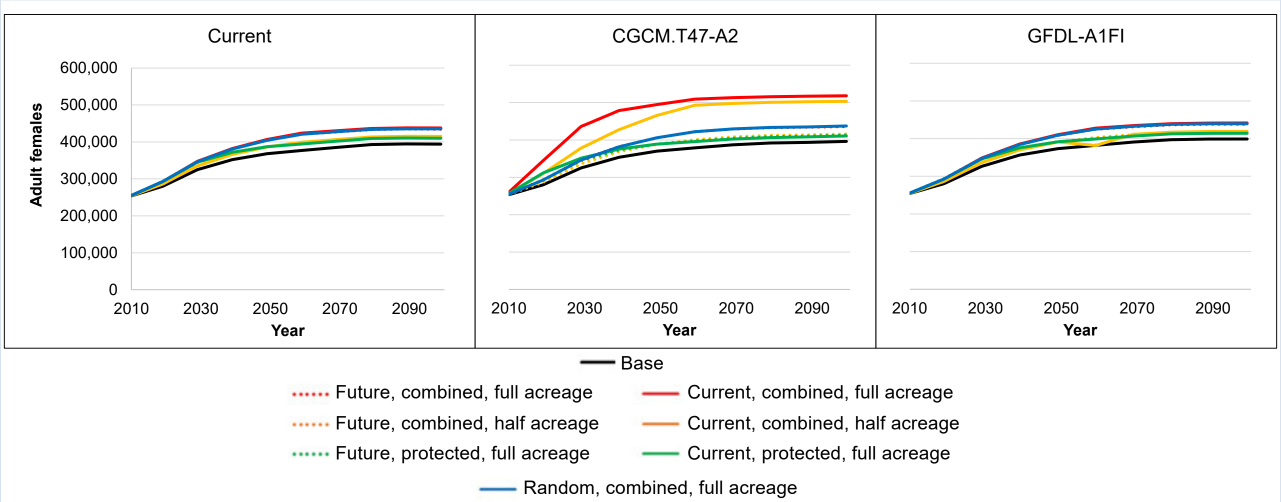

Figure 27.—Projected changes in carrying capacity of adult female prothonotary warblers in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

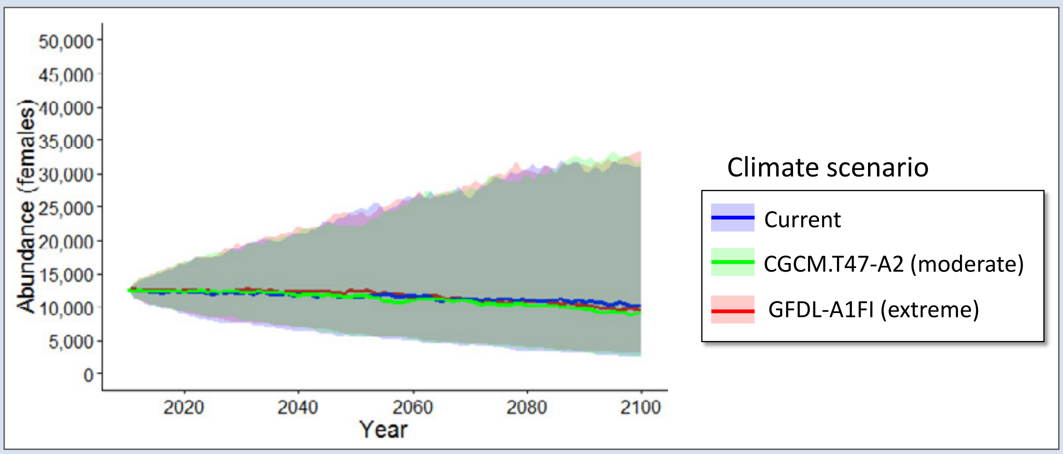

Figure 28.—Projected population growth of prothonotary warblers in the Ozark Highlands for a baseline scenario of no forest restoration. Predictions obtained from dynamic-landscape metapopulation models applied to landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

Box 8 continued.

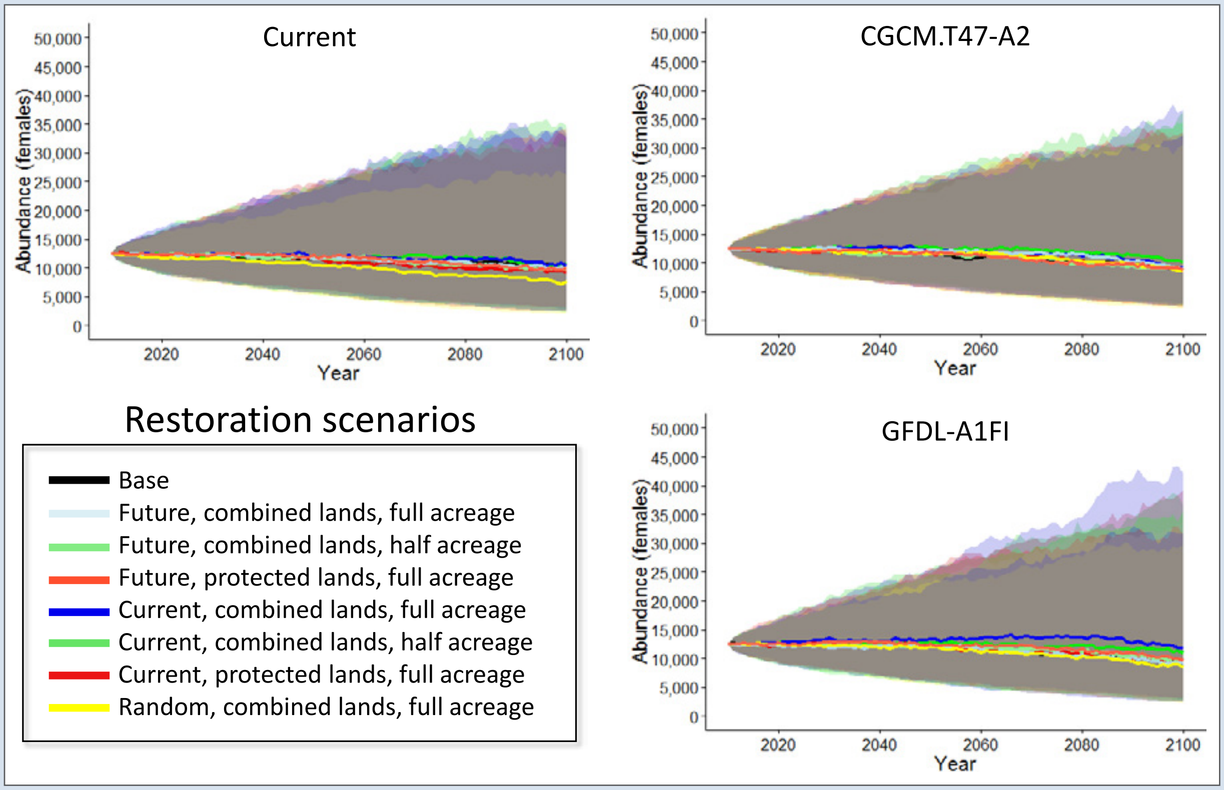

Figure 29.—Projected population growth of prothonotary warblers in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

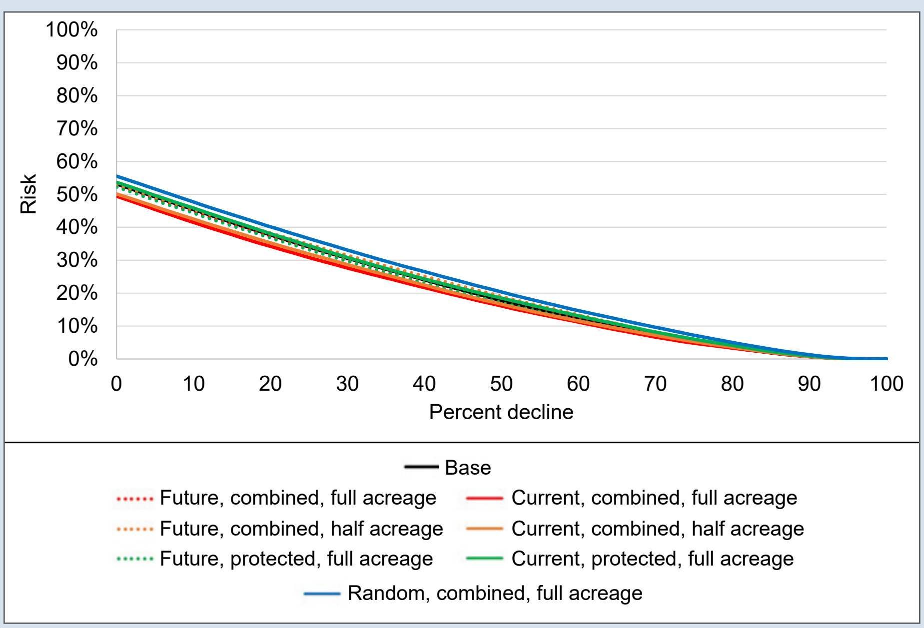

Figure 30.—Projected risk of the Ozark Highlands population of prothonotary warblers declining by various levels under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Image by the authors.

BOX 9. CERULEAN WARBLER

Habitat Model

This small, canopy-foraging insectivore breeds locally in mature and older deciduous forests with broken canopies across much of the Eastern United States (Buehler et al. 2013). Although it requires large forest tracts, the cerulean warbler establishes territories near interior forest gaps (Tirpak et al. 2009).

Photo by U.S. Department of Agriculture, Flickr Creative Commons, https://www.flickr.com/photos/.

We adapted the HSI model for cerulean warbler from Tirpak et al. (2009). The first component combines landform, landcover, and successional age class and assigns suitability scores to these combinations on the basis of habitat associations of the cerulean warbler outlined in Hamel (1992). A logistic function is used to address this species' area sensitivity to patches of forest <1,000 ha. Other variables incorporated selection for canopy cover between 75-100 percent and predominantly forested landscapes. Tirpak et al. (2009) used dominant tree density to address the vertical and horizontal canopy structure used by cerulean warblers. However, we added a component to this index that required overstory density and stocking values between 50-80 percent.

Population Model

We estimated K and initial abundances from HSI model assuming density at K = 2 pairs/ha when HSI = 1.0 (Boves et al. 2013, Buehler et al. 2013, Jones-Farrand and Bonnot 2014). We used the average of maximum territory sizes (2 ha) in Buehler et al. (2013) when using a moving window to identify habitat supporting. We based initial abundances of cerulean warblers on the 0.14 pairs/ha (7 percent of K; Reidy et al. 2011).

We specified adult survival for cerulean warblers at 0.6 through a combination of rates reported in Buehler et al. (2013): parent annual – 0.49 (Jones et al. 2004); monthly breeding – 0.98 (Jones et al. 2004); and monthly wintering – 0.97 (Bakermans et al. 2009). We set juvenile survival at a rate of 0.28 (Jones-Farrand and Bonnot 2014). We incorporated parasitism by applying mean RPI values to a maximum cerulean warbler maternity of 1.5 fem/fem/year (Boves et al. 2015).

The model assumed that 90 percent of juveniles and 10 percent of adults dispersed annually. Dispersal distance was modeled as a negative exponential function with a mean distance of 70 km (Bonnot et al. 2011). We accounted for density dependence using a modified ceiling model.

Results

Similar to prairie warblers, the importance of forest structure characteristics to cerulean warblers resulted in substantial declines (86 percent) in K due to interactions between forest succession and management the first 2 decades. Base projections indicated K decreased >1,000,000 females during this period. Under climate change models, this loss of habitat eventually began recovering by the end of the century as those structural characteristics shifted more in line with cerulean warbler habitat. The baseline scenario under the GFDL-A1FI model projected K by 2100 to be an order of magnitude than that for the current climate. The population responses to these changes were far less significant. Under all climate models, cerulean warblers experienced >1 percent annual declines throughout most of this century. As a result, we estimated the probability that the population would decline by 25 percent at 0.53.

Box 9 continued.

Model projections demonstrated dramatic responses of cerulean warblers to restoration scenarios. By 2030, habitat restored under the alternative scenarios stemmed declines in K by half. By 2050, we projected K for restoration scenarios at 4-7 times that for the baseline scenario under all climate models. The additive effects of restoring full acreage objectives in addition to increased habitat under the GFDL-A1FI scenario allowed K in 2100 to quadruple the initial estimate (initial abundance = 821,049 females; final abundance = 3,357,109 females). In contrast to climate change alone, cerulean warbler population responded positively to restoration scenarios. Regardless of climate model, all scenarios, except for those with random prioritization, prevented any significant declines. Although we projected similar responses across most alternatives, scenarios that focused on protected lands presented the lowest risk for cerulean warblers through 2100.

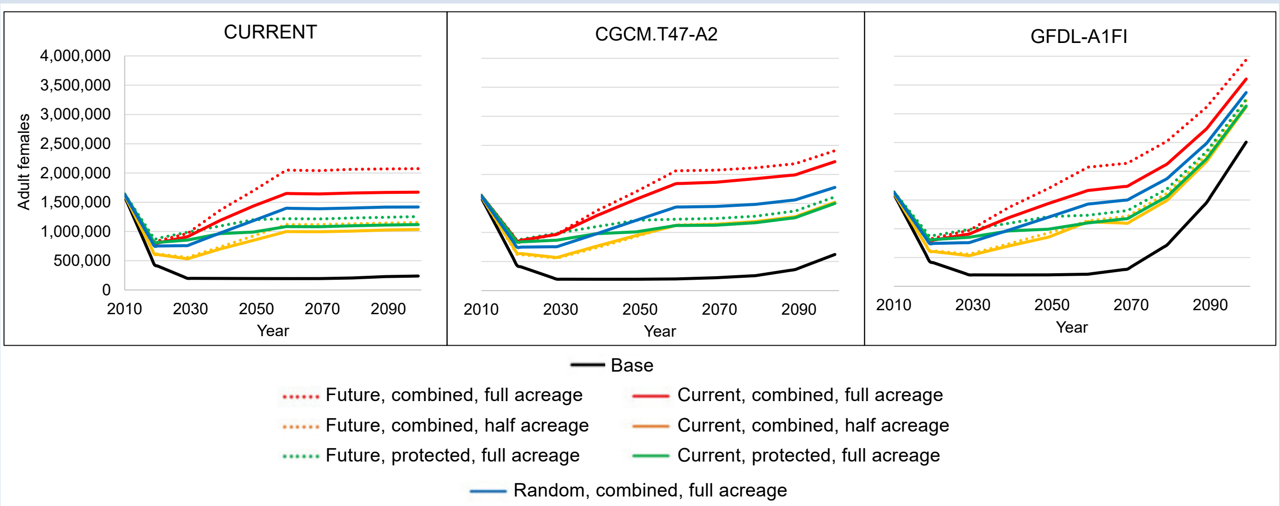

Figure 31.—Projected changes in carrying capacity of adult female cerulean warblers in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

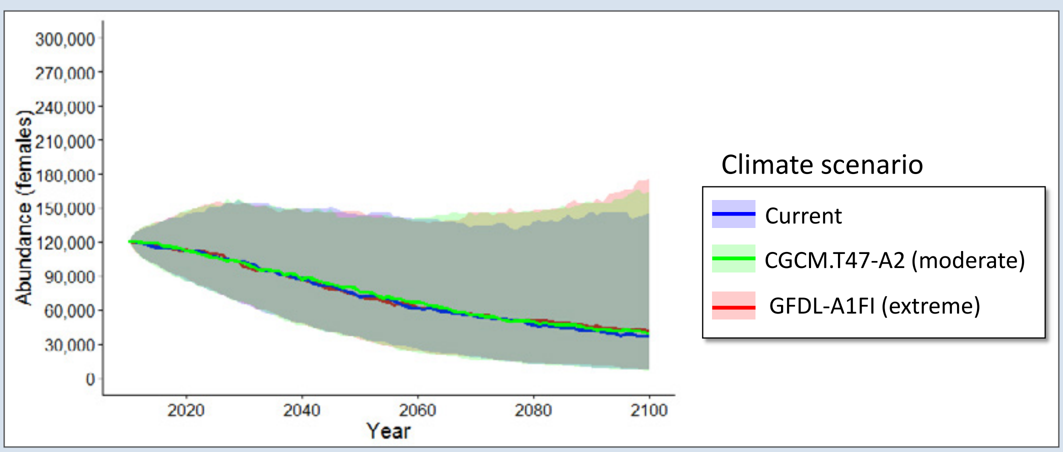

Figure 32.—Projected population growth of cerulean warblers in the Ozark Highlands for a baseline scenario of no forest restoration. Predictions obtained from dynamic-landscape metapopulation models applied to landscapes projected under urbanization and models of a current climate, moderate (CGCM. T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

Box 9 continued.

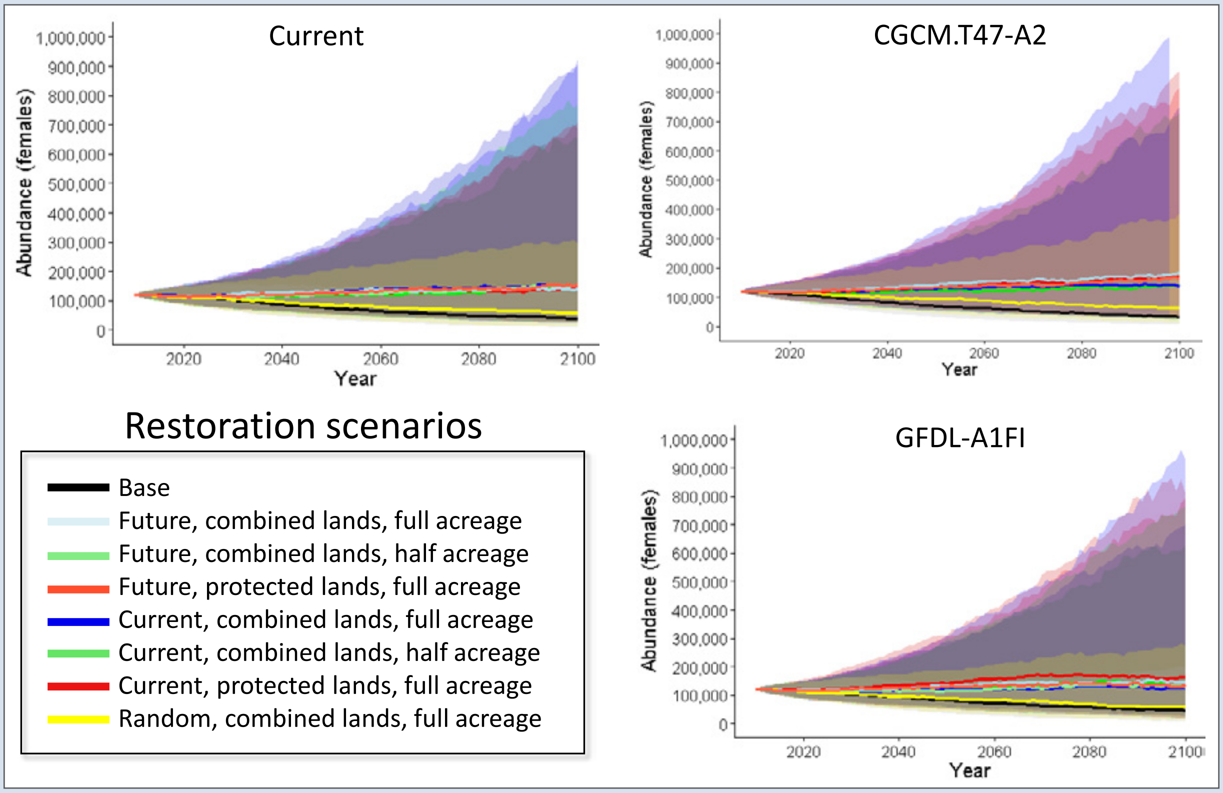

Figure 33.—Projected population growth of cerulean warblers in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM. T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

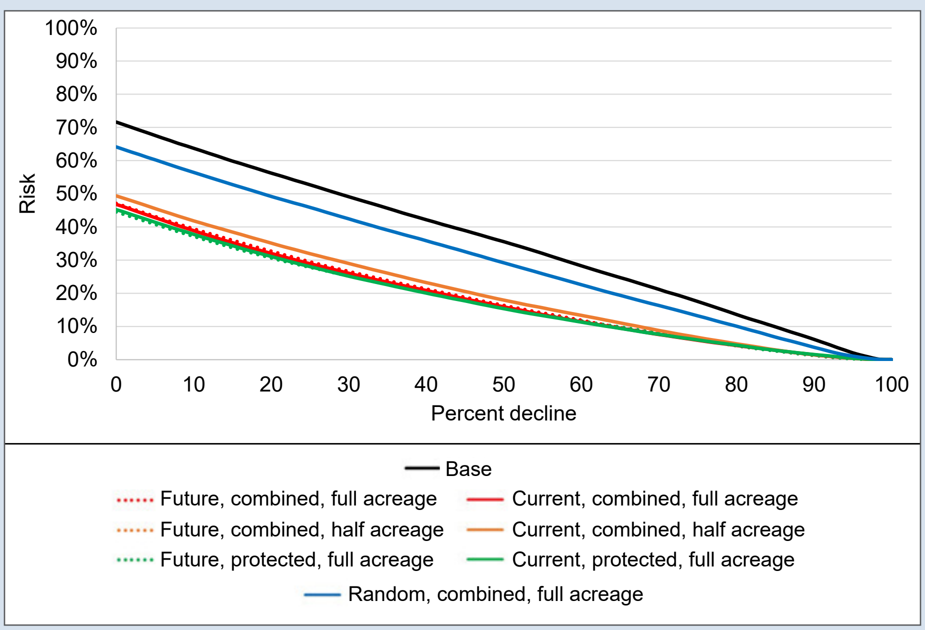

Figure 34.—Projected risk of the Ozark Highlands population of cerulean warblers declining by various levels under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Image by the authors.

BOX 10. *MYOTIS* SPP.

Habitat Model

Photo by U.S. Fish and Wildlife Service, Flickr Creative Commons, https://www.flickr.com/photos/.

Our *Myotis* spp. model reflected habitats for Indiana and northern long-eared bats. Both of these species breed in mature forests and hibernate in caves during winter. Primary non-winter habitat components are roosting sites, foraging areas, and water (Kurta 2002, Larson et al. 2003).

To develop the *Myotis* HSI model, we combined components from models for northern long-eared bats and Indiana bats. The first component identified pole and saw seral stages of various forest cover type as suitable habitat. Roost sites are predominantly in large dead trees, under exfoliating bark, or inside hollows or crevices. (Foster and Kurta 1999). Therefore, we used the quadratic function by Rittenhouse et al. (2007) that identified roost tree availability for Indiana bats. This function estimates snag density by tree age class. We also varied suitability with distance to nearest water source, where sites <1 km from water were most suitable (Rittenhouse et al. 2007). We adapted a variable from Larson et al. (2003) that reduces suitability by half for sites >120 m from a forest edge, given northern long-eared bats forage in mature deciduous forest with small gaps. Finally, recent work has shown the importance of the surrounding landscape for occurrence of northern long-eared bats. We varied suitability with the percent of forest within 10 km (Starbuck et al. 2014). We also reduced suitability for sites with increased small-stem densities (Starbuck et al. 2014).

Population Model

Carrying capacity and initial abundances of *Myotis* spp. were estimated from HSI model assuming density at K = 0.1 pairs/ha when HSI = 1.0 (Womack, K. 2015, Personal communication. Resource staff scientist, Missouri Department of Conservation, 3500 East Gans Road, Columbia, MO 65201). Because these species breed in colonies, we assumed a maximum home range size of 1,500 ha and removed any habitat that K supported <60 breeding females over the area (Womack, pers. comm.). We assumed initial abundances occurred at 25 percent of K.

We used demographic rates from a stage structured model of Indiana bats with three stages (Thogmartin et al. 2013). We set survival rates for adults, juveniles, and pups (fall and winter survival only) at 0.9, 0.57, and 0.898, respectively. We assumed fertility for adults is 0.42 fem/fem/year and for juveniles, 0.279 fem/fem/year (Thogmartin et al. 2013).

Because of the limited dispersal in these species, we did not model dispersal between populations. We modeled density dependence using a Ricker contest model where survival decreased with increasing abundance of bats (Erickson et al. 2015).

Results

Habitat increased for *Myotis* in the baseline simulations such that K for bats grew from its initial estimate of 597,608 females by >70,000 under all climate models. However, we projected 20 percent declines in the simulated population under the three future climates by 2100.

Restoration scenarios did little to affect habitat for *Myotis* and only slightly lowered risk. Except for scenarios under the CGCM.T47-A2 model, restoration did not alter K. For this climate model restoration prioritized by the current landscape did result in K above the baseline scenario. Our models did not project any response by *Myotis* population to climate change or restoration. In all scenarios we estimated a 34-36 percent risk of a 25 percent decline through 2100. Restoring half of the acreage on private and protected lands based on the current prioritization presented the lowest risk of decline (34 percent).

Box 10 continued.

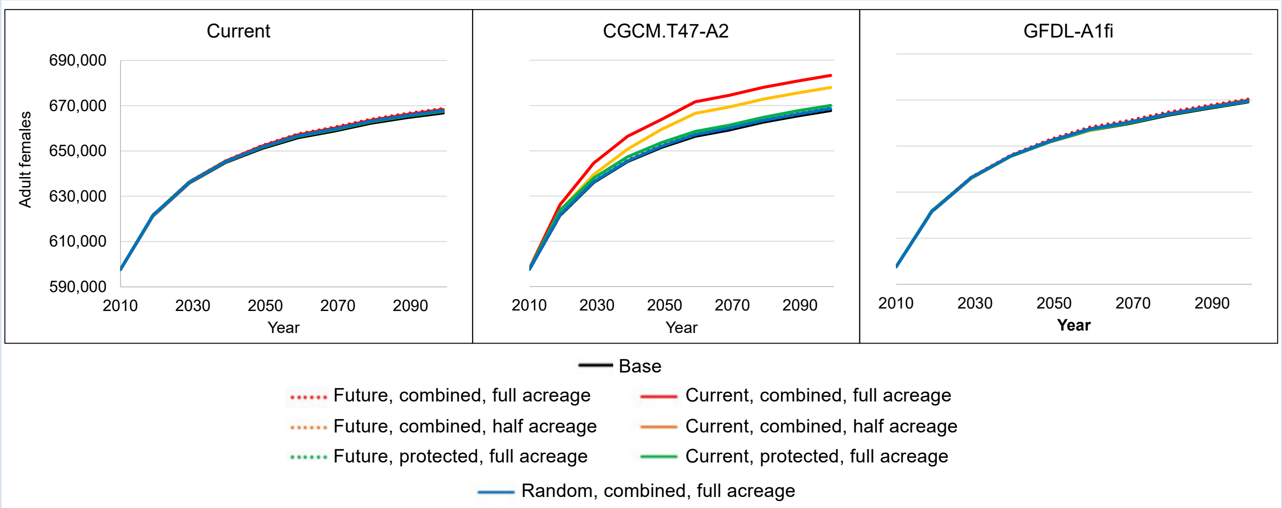

Figure 35.—Projected changes in carrying capacity of adult females for a *Myotis* bat in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

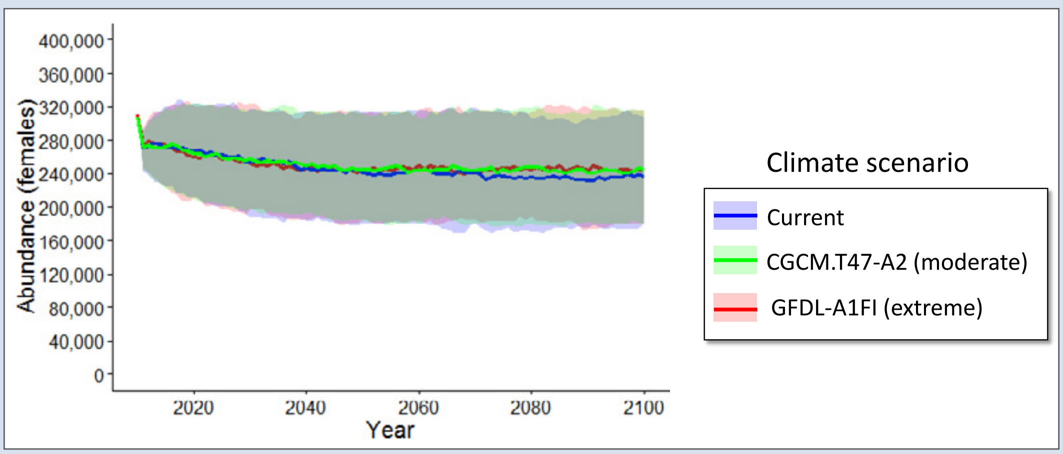

Figure 36.—Projected population growth of a *Myotis* bat in the Ozark Highlands for a baseline scenario of no forest restoration. Predictions obtained from dynamic-landscape metapopulation models applied to landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

Box 10 continued.

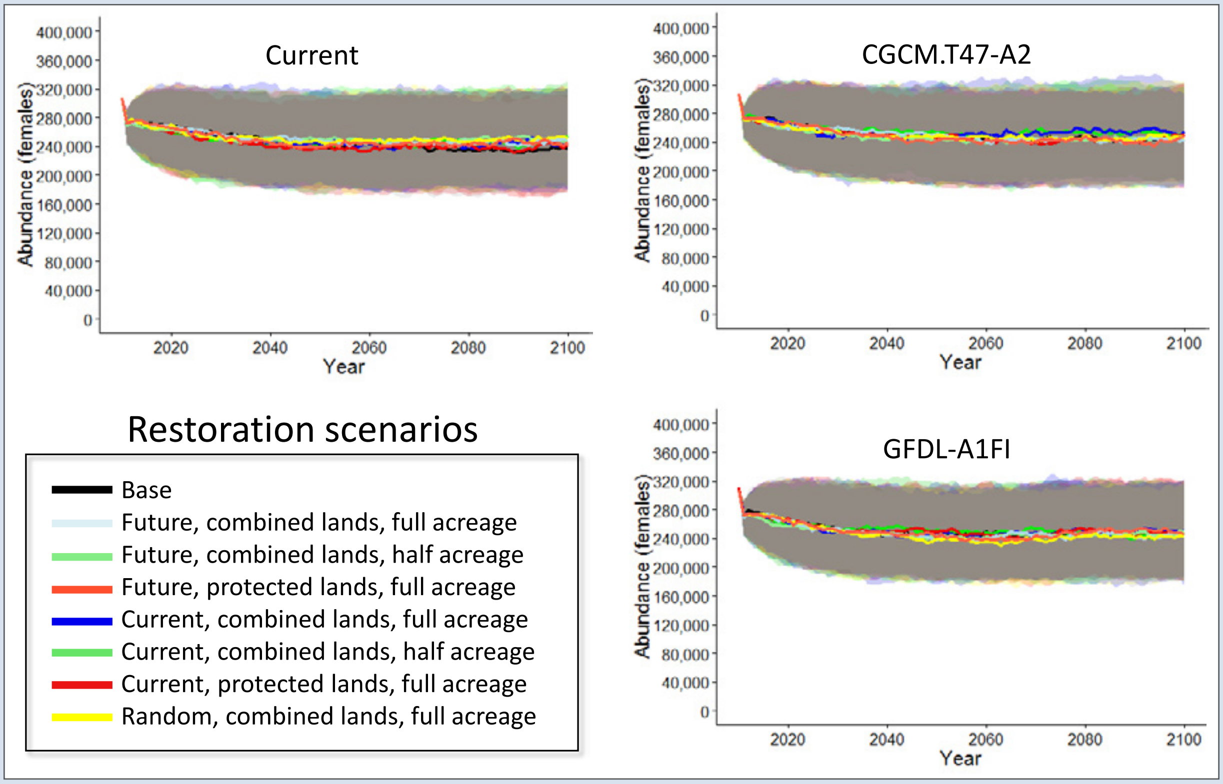

Figure 37.—Projected population growth of a *Myotis* bat in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM. T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

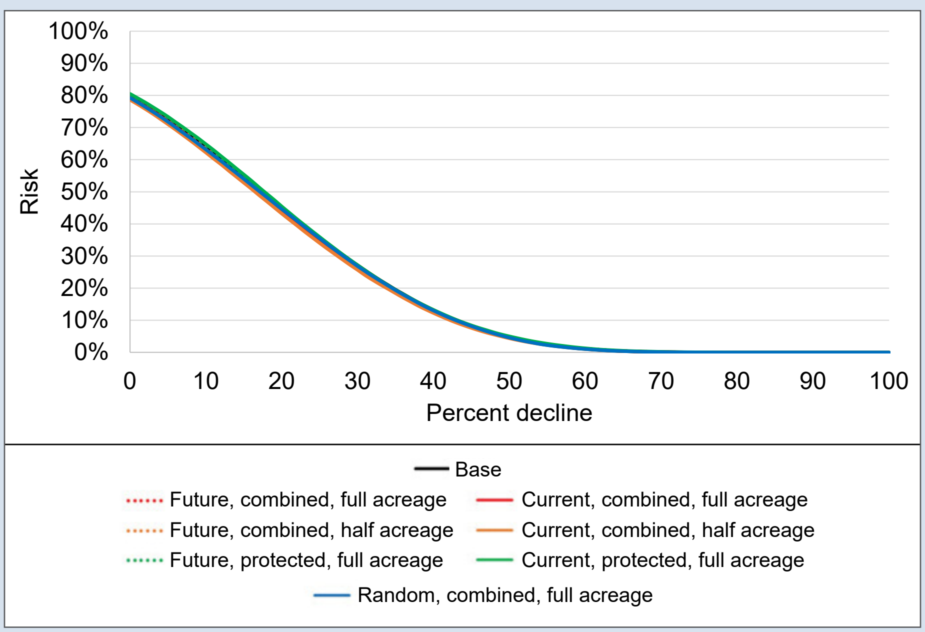

Figure 38.—Projected risk of the Ozark Highlands population of a *Myotis* bat declining by various levels under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Image by the authors.

BOX 11. BROWN-HEADED NUTHATCH

Habitat Model

Photo by leshoward, Flickr Creative Commons, https://www.flickr.com/photos/.

Brown-headed nuthatch are endemic to pine forests of the Southeastern United States and are rarely seen far from pine-dominated areas (Slater et al. 2013). The habitat of this species is defined by two habitat elements: mature pines for foraging and cavities for nesting (Dornak et al. 2004, Wilson and Watts 1999).

We used the brown-headed nuthatch model, developed by Tirpak et al. (2009). In the model, suitability of forests varied by landform, landcover, and seral stage, with later stages of mixed and evergreen forests being most suitable. Because LANDIS does not model snags, we relied on projected stand ages as a proxy for the original snag density variable that increased suitability with snag density (Rittenhouse et al. 2007, Tirpak et al. 2009). The model also considered small-stem density to account for the preference of the brown-headed nuthatch for open understories. Finally, suitability indices decline with increasing hardwood basal area as birds are less abundant in habitats with a greater hardwood component (Slater et al. 2013, Wilson et al. 1995, Wilson and Watts 1999).

Population Model

Carrying capacity and initial abundances of brown-headed nuthatch were estimated from HSI model assuming density at K = 0.5 pairs/ha when HSI = 1.0 (Wilson et al. 1995). We assumed a maximum territory size of 3.2 ha when filtering habitat not supporting territories (Slater et al. 2013). Because the brown-headed nuthatch is likely habitat limited, we set initial abundance 75 percent of K.

We assumed an adult survival rate of 0.67 based on the average of female rates in two Florida populations (Cox and Slater 2007). No estimates of juvenile survival exist, so we used convention estimates of 0.3 (Bonnot et al. 2011). We set maternity at 1.3 fem/fem/year (Slater et al. 2013) and did not model parasitism in this population (Cox and Slater 2007).

The model assumed that 90 percent of juveniles and 10 percent of adults dispersed annually. Dispersal distance was modeled as a negative exponential function with a mean distance of 1.27 km, based on models for another short-distance dispersing resident, the golden-cheeked warbler (Riedy et al. 2018). We accounted for density dependence using a modified ceiling model.

Results

Under baseline scenarios with no conservation, we projected reductions in brown-headed nuthatch habitat over the next century. As an open-pine specialist, K for this species decreased from succession and dominance of hardwoods. Carrying capacities under the two climate change scenarios finished higher than the current climate, probably due to the effects of climate that opened forest structure and favored pine species. These differences in habitat availability were apparent in their base population projections. Brown-headed nuthatch declined 19 percent through 2100 under the current climate model, whereas the population was stable and grew 16 percent under the CGCM.T47-A2 and GFDL-A1FI models, respectively.

In general, restoration moderately offset habitat loss. For most scenarios, we projected final Ks to be 5-45 percent above baseline scenarios. Under the CGCM.T47-A2 model, the two scenarios that restored habitat on combined private and protected lands, prioritized by future landscapes, resulted in lower K compared to the baseline scenario.

Box 11 continued.

Despite scenarios providing habitat, we projected major differences between brown-headed nuthatch responses for restoration efforts alone and when restoration scenarios were simulated with translocation. Without translocation, all restoration scenarios, regardless of climate model, negatively affected population growth. These scenarios reduced the population 15 percent below baseline projections when averaged across climate models. This species is currently limited to the Southern portion of the Ozark Highlands in the Boston and White River Hills subsections.

Most open-pine restoration occurred outside of its current distribution in the Current River Hills and Black River Ozark Border subsections. Therefore, the limited dispersal of brown-headed nuthatch prevented it from utilizing the newly restored habitat; and the restoration where they were distributed targeted oak woodlands, which removed habitat and caused population declines. This realization was the motivation for the team's decision to combine translocation efforts with restoration scenarios, giving the species a chance to colonize newly created habitat. When paired with translocation, all restoration scenarios reversed population declines. The new populations of birds grew substantially and drove abundances 30-102 percent greater than initial estimates of 19,428 brown-headed nuthatch females. Likely limited most by habitat, this population was relatively stable. With only a 7 percent chance of a 25 percent decline, most restoration scenarios reduced this risk 2-4 points. The random restoration scenario presented the lowest risk for this species (3 percent).

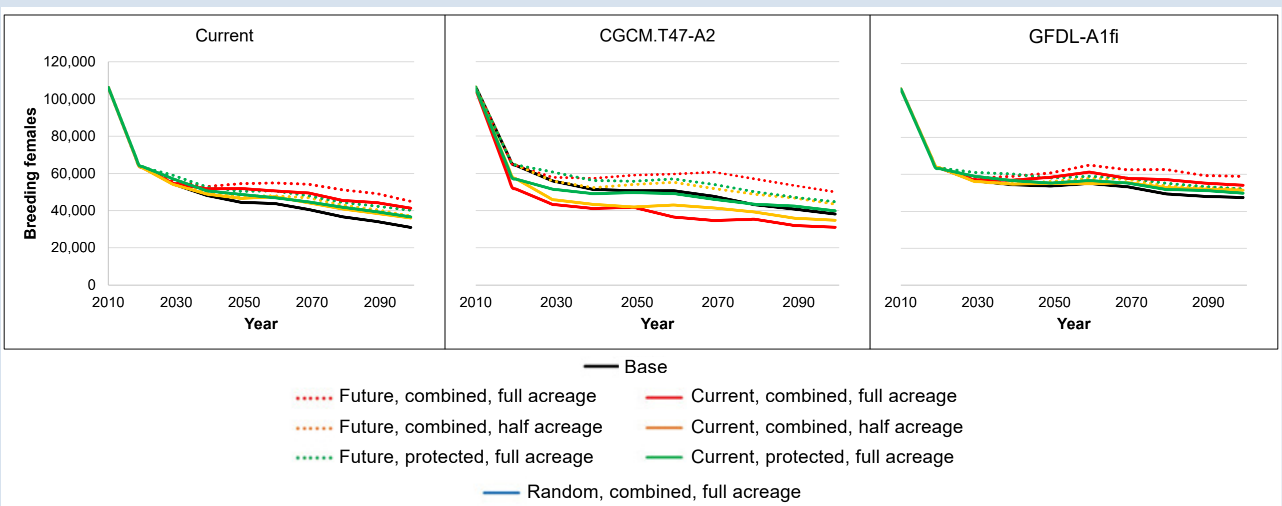

Figure 39.—Projected changes in carrying capacity of adult female brown-headed nuthatch in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

Box 11 continued.

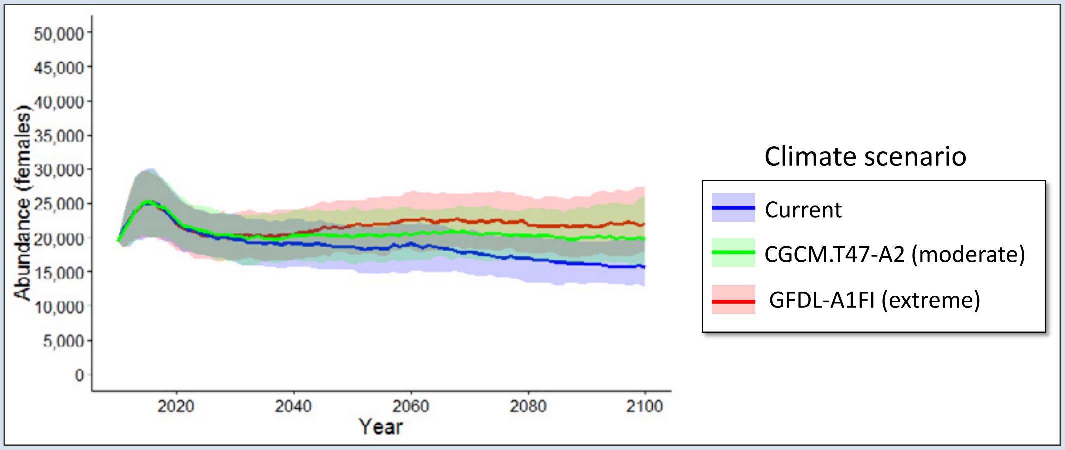

Figure 40.—Projected population growth of brown-headed nuthatch in the Ozark Highlands for a baseline scenario of no forest restoration. Predictions obtained from dynamic-landscape metapopulation models applied to landscapes projected under urbanization and models of a current climate, moderate (CGCM. T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

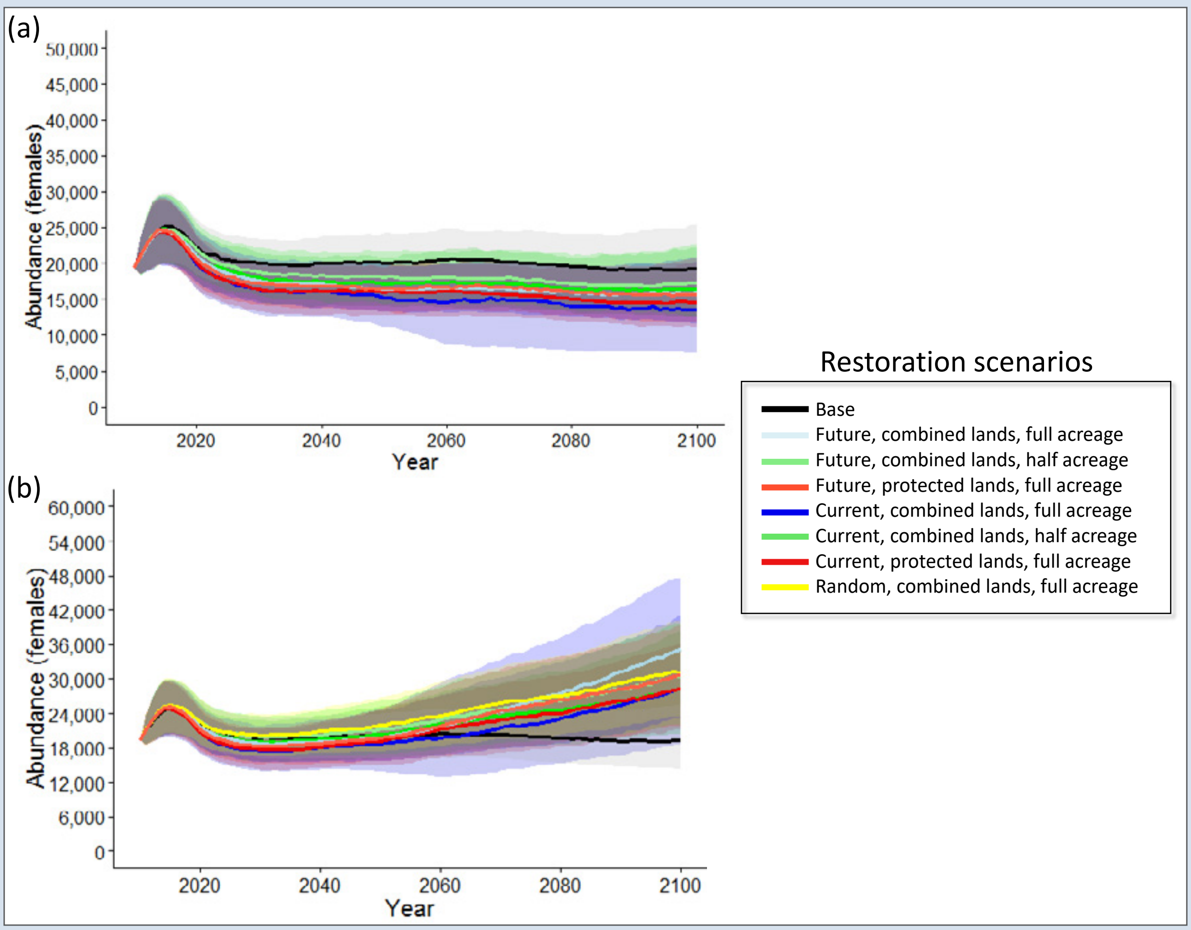

Figure 41.—Contrasting population growth of brown-headed nuthatch in the Ozark Highlands under conservation scenarios that (a) focus only on restoration and (b) combine translocation efforts with restoration from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate conservation on landscapes projected under urbanization and moderate (CGCM.T47-A2) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

Box 11 continued.

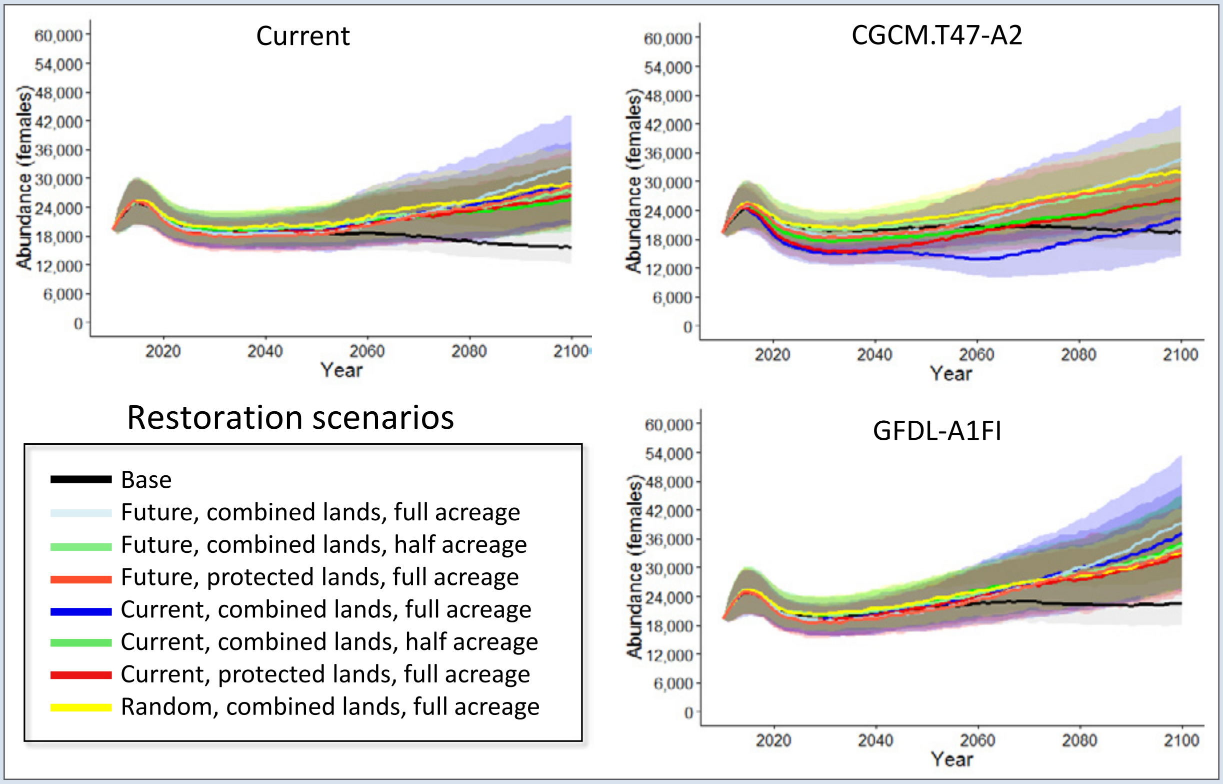

Figure 42.—Projected population growth of brown-headed nuthatch in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate translocation efforts with restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Shaded regions indicate 85 percent credible intervals. Image by the authors.

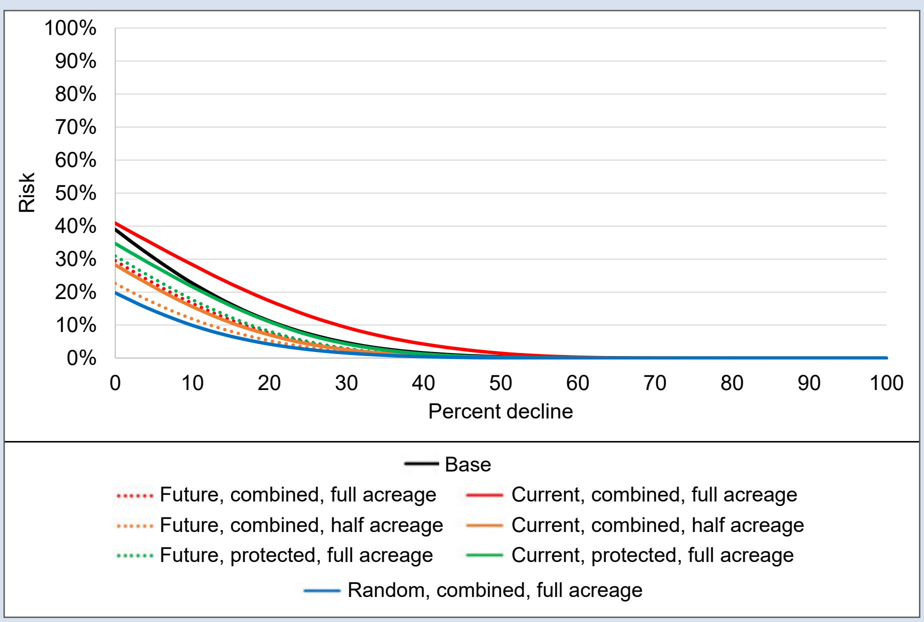

Figure 43.—Projected risk of the Ozark Highlands population of brown-headed nuthatch declining by various levels with translocation and under combinations of restoration and climate change scenarios from 2010–2100. Predictions are based on dynamic-landscape metapopulation models that integrate restoration on landscapes projected under urbanization and models of a current climate, moderate (CGCM.T47-A2) climate change, and extreme (GFDL.A1FI) climate change. Image by the authors.

BOX 12. EASTERN COLLARED LIZARD

Habitat Model

Given the lack of data on habitat for eastern collared lizards, we considered the general use of glades as habitat. However, because colonization of glades by this species in a metapopulation, context depends on their connectedness by woodlands (forests with open canopies and understory vegetation) (Brisson et al. 2003). We considered glades as habitat only if they were connected to ≥ 1 other glades by woodland forests. We evaluated the effects of restoration scenarios by summing the acreage of habitat provided under each scenario.

Photo by Peter Paplanus, Flickr Creative Commons, https://www.flickr.com/photos/.

Results

All restoration scenarios increased habitat for eastern collared lizards. The number of acres of lizard habitat was proportional to the total acreage restored across scenarios. Scenarios targeting full acreage objectives on private and protected lands provided the most habitat. Random restoration scenarios also provided substantial habitat.

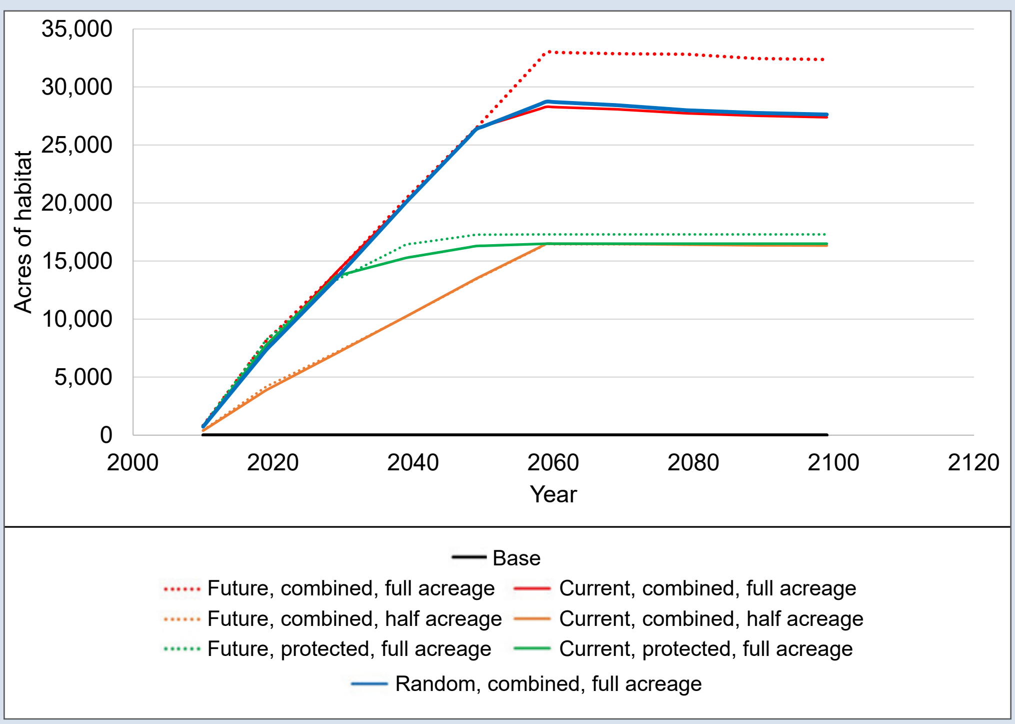

Figure 44.—Projected changes in amount of habitat for eastern collared lizard in the Ozark Highlands under combinations of restoration and climate change scenarios from 2010–2100. Image by the authors.

CONCLUSIONS

Landscape Conservation Design Is Effective

We have confirmed that long-term planning for viable populations can be achieved under global change. Because many impacts of climate change are still decades in the future, a new level of proactive management is required (Lawler 2010). Management will also need to be large in scale to adequately address widespread threats. Ultimately, our team identified a landscape conservation design that best reduced the average risk to focal populations across the Ozark Highlands throughout a century of climate change and urbanization. The framework used was objective, transparent, and repeatable. The process addressed numerous sources of uncertainty, from error in demographic rates to uncertainty among emissions scenarios. The results of the framework were also actionable, including spatial products that provided detailed guidance on specific conservation activities that managers can take to the ground in specific patches. Such a framework can greatly help address the uncertainties and complexities that are inherent in the process of long-term, large-scale conservation planning.

The effectiveness of this approach stems from the integration of SDM and DLMP and how they complement each other. Although DLMPs provide information on the effects of climate change on populations or the responses of populations to conservation actions, this information alone does not make the decision (Beissinger and McCullough 2002). Rather, decisions are still affected by complexity of competing objectives and uncertainties from model error and future climates (Iverson et al. 2016, Naujokaitis-Lewis et al. 2013). Structured decision making facilitates decisions by explicitly incorporating complexity and uncertainty into group decisions. Nevertheless, the degree to which they are good decisions is dictated by how well the effects of climate change and conservation activities on populations are represented in the process (Marcot et al. 2012). Integrating the two methods resolved these issues and provided partners with a structured way to compare the effectiveness of alternative conservation scenarios on multiple species under various future possibilities while being confident that important ecological and population processes were captured in each outcome. Estimating risk is fundamental to population modeling, but it is also fundamental to decision making as it more intuitively conveys the severity and uncertainty of potential impacts to planners and policy-makers (Millspaugh et al. 2009). Thus, a key advantage of this integration lies in the ability of DLMPs to estimate risk and the ability of decision makers to structure their decisions around it.

Structured Decision Making Is Key

Managers must continually account for multiple objectives (e.g., biological benefits, cost, public interest) in conservation planning (Lambeck and Hobbs 2002, Noon et al. 2009). Such considerations create potential for conflicts. Rather than avoid these conflicts, decision theory provides methods to address competing objectives, such as multicriteria decision and tradeoff analyses (Schwenk et al. 2012, Thompson et al. 2013). In this way, the team viewed the needs of multiple species as competing objectives and adapted a multicriteria decision analysis to transparently resolve tradeoffs. The largest source of climate-related uncertainty for the future is not model error but, rather, the path of emissions scenarios taken by society (IPCC 2013). We addressed this uncertainty by pooling risk across possible climate change scenarios, assuming equal probabilities for each future, which may or may not be valid (Kujala et al. 2013). Other summaries of risk, such as maxi-min or mini-max regret, are preferred because the optimal choice does not depend on knowing probabilities of future outcomes (Polasky et al. 2011, Thompson et al. 2013).

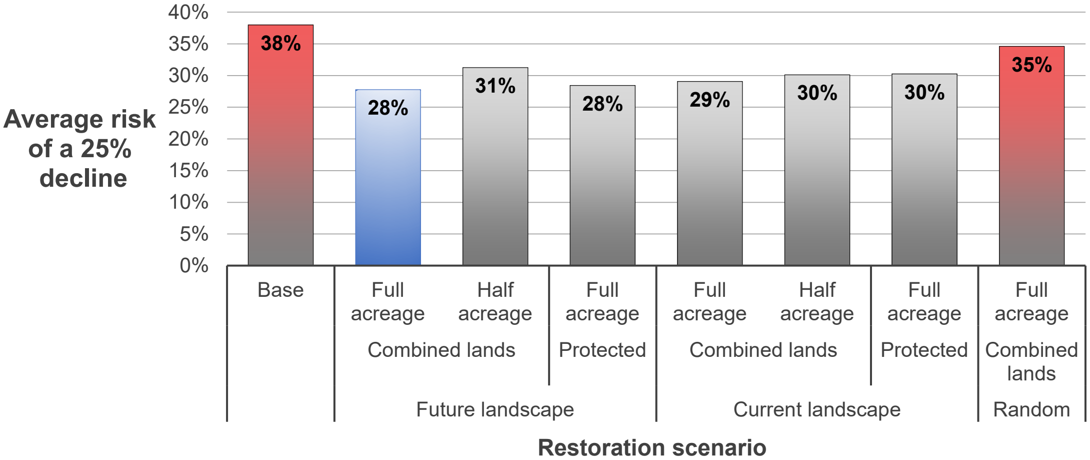

Figure 45.—Average risk of a 25 percent decline for all focal species in the Ozark Highlands under alternative habitat restoration scenarios, proposed by the planning team. Risk is pooled across projections under three future climate models that range from continuation of the current climate, moderate climate change, and extreme climate change. The alternative that restored full acreage targets on both private and protected lands base, prioritized by future landscape conditions, resulted in the lowest risk across species (blue bar). In contrast, the base and random scenarios (red bars) resulted in the highest risk across species. Image by the authors.

Implications for Strategic Habitat Conservation

We suggest that habitat restoration, implemented strategically through a landscape conservation design process, can help conserve multiple species in the face of climate and landscape change. Scenarios with no restoration or randomly placed restoration presented the most risk to species (Fig. 45). Instead, strategic scenarios best reduced risk for the most threatened species (prairie warbler, cerulean warbler, and brown-headed nuthatch) while avoiding added risk to other species. We expected restoration scenarios that largely affected woodland and glade communities to reduce habitat for wood thrush, but its population was unaffected by these changes, with the impacts of fragmentation instead driving declines. Similar patterns were projected for other species. Therefore, restoration scenarios balanced competing habitat needs. The loss and alteration of these communities over the last century were responsible for the concern and risk surrounding prairie warblers. Results also indicated how climate change could alter forest structure later in the century, essentially opening forests and creating woodland habitat. While prairie warblers did respond to these changes, the ability of restoration to provide habitat earlier prevented larger declines in their population and reversed declines of cerulean warblers. Given that adaptation of forest communities to climate change is a current focus in management, the notion that current efforts could restore important habitats for these species, while at the same time adapting landscape for climate change, is an encouraging finding.

More generally, this project illustrated how targeting ecosystem-based management within a decision-support framework reflects a blend of climate adaptation approaches. Increasing ecosystem resilience is a preferred approach to climate adaptation, focusing on the capacities to adapt to changing conditions, maintain key processes, and transform to new states if the previous states become untenable (Polasky et al. 2011). Building resilience can require targeting threats affecting populations, which is an important aspect of DLMPs (Harwood 2000). For example, addressing the habitat loss responsible for recent prairie warbler declines also increased the resiliency of this population to climate change. While the tradeoff with a resiliency approach is that it can be too general to develop specific plans (Polasky et al. 2011), the alternatives used in our example simulated specific plans that can be implemented.

This framework closely resembles scenario planning used in climate adaptation. In scenario planning, contingency plans are developed for a range of future impacts (Amer et al. 2013, Polasky et al. 2011). Scenario planning and our approach both assume that potential impacts are foreseen and that plans will be effective at addressing them. However, our team originally did not expect the positive impact of climate change on prairie warbler habitat. Nor did it foresee the negative impact of habitat restoration without translocation on brown-headed nuthatch. But outputs from the population models provided important insights into what could happen and the potential limitations of our approaches. These instances informed partners of important processes that affected species, allowing them to better develop multifaceted plans to conserve populations under multiple threats (Bonnot et al. 2013).

Important Considerations for the Decision-Support Framework

Although we demonstrated the capability of a decision-support framework to guide conservation decisions, its effectiveness at providing robust decisions across applications will depend greatly on the rigor of the science and the capacities of the decision teams. Optimal decisions require an accurate representation of species responses to conservation from many threats (Beissinger and Westphal 1998). However, one of the weaknesses in our approach was that habitat models for some species did not thoroughly capture the habitat and demographic effects of climate change and restoration activities. For example, although recent work suggests likely changes in surface water distribution from climate change (written communication from Jacob LaFontaine, USGS), these data were not available to us at the time. Consequently, our projections of *Myotis* spp. habitat remained unchanged across scenarios. Including such data would have more accurately projected future changes in *Myotis* spp. habitat. Furthermore, our DLMP models did not incorporate recent evidence of the threat that climate warming poses for songbird reproduction (Bonnot et al. 2018, Cox et al. 2013). Significant population declines caused by processes unrelated to habitat such as temperature on nest survival might decrease the effectiveness of habitat conservation. Therefore, designing quality plans will require decisions that are based on the most thorough and accurate information available.

The utility of this approach will also be determined largely by the ability of the decision teams to correctly frame their problems, agree on their objectives, and design alternatives. Decision environments can range from long-term visions for agency-wide programs to time-sensitive and dynamic scenarios that allow day to day responses to changing conditions (Thompson et al. 2013). Having a clear idea of the planning context for a decision and being able to match objectives and alternatives to that situation are important (Drechsler and Burgman 2004). For our example, considering millions of hectares of habitat restoration to target population viability objectives over the next century made sense from the long-term planning perspective of the GCPO and CHJV. However, team members that are restricted to planning management over smaller areas and shorter time frames with smaller budgets saw less utility to this approach in their daily decisions. Additionally, this project constrained planning to the biological objectives despite a full awareness that stakeholder and partner interest, cost, and other factors will need to be weighed in any decision (Keeney 2007, Powledge 2012). Finally, the team's structuring of alternatives resulted in very similar restoration plans. As a result, population responses were at times similar and the overall decision among risks was close. To avoid these potential difficulties, it will be crucial that decision makers balance creativity and experience to entertain a wide range of alternatives (Gregory et al. 2012). The range of alternatives is limited only by the ability to model their consequences in terms of the objectives. Recruiting diverse interdisciplinary teams that have experience with the system of interest and expertise in different fields can help greatly in such cases (Polasky et al. 2011).

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Planning for sustainable landscapes is hampered by uncertainty in how species will respond to conservation actions amidst impacts from landscape and climate change. Planning decisions, including tradeoffs among competing species objectives, are complex. We developed a decision-support framework that integrates dynamic-landscape metapopulation models (DLMPs) and structured decision making (SDM) to help guide landscape conservation design. With this framework, we demonstrated that planning for viable populations across broad scales can be achieved under global change. Furthermore, the integration of DLMPs with SDM enabled decisions to be more objective and transparent, and thus, more defensible.

KEY WORDS: structured decision making, climate change, urbanization, population model, dynamic, risk, habitat, restoration

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