

Chapter 9

Modeling Landscape Fire and Wildlife Habitat

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9.1 Introduction

Global climate is expected to change rapidly over the next century (Thompson et al. 1998; Houghton et al. 2001; IPCC 2008). This will affect forest ecosystems both directly by altering biophysical conditions (Neilson 1995; Neilson and Drapek 1998; Bachelet et al. 2001) and indirectly through changing disturbance regimes (Baker 1995; McKenzie et al. 1996; Keane et al. 1999; Dale et al. 2001; McKenzie et al. 2004; Westerling et al. 2006). Changes in biophysical conditions could lead to species replacement in communities and latitudinal and altitudinal migrations (Iverson and Prasad 2002; Neilson et al. 2005). Expected increases in the frequency, size, and severity of wildfires (Mearns et al. 1984; Overpeck et al. 1990; Solomon and Leemans 1997; IPCC 2008), and other disturbances such as insect outbreaks, may further amplify changes in vegetation structure, species composition, and diversity (Christensen 1988; McKenzie et al. 2004). These shifts in distributions of plant species may have large impacts on many aspects of ecological diversity and function (Peters and Lovejoy 1992; Miller 2003).

Despite the magnitude of these potential ecosystem changes, relatively little attention has been given to the effects of interactions between climate and natural disturbance regimes on wildlife populations. Wildlife populations are critically dependent on sufficient amount, quality, and spatial distribution of habitat. The environmental conditions that provide habitat for each species in turn are a dynamic product of the prevailing disturbance regime, in interaction with regional climate. In the western United States, fire, as arguably the dominant landscape-scale disturbance process, plays a keystone role in establishing and maintaining habitat conditions for wildlife. Some species evolved in fire-dominated ecosystems and consequently may have been negatively impacted by fire exclusion, whereas by

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contrast others may be sensitive to loss of habitat from fire. If climate change drives more frequent and extensive fires in forest ecosystems (McKenzie et al. 2004; Flannigan et al. 2005; Westerling et al. 2006), species dependent on extensive late-successional forest may decline while species dependent on early-successional habitat may increase. Furthermore, if a warmer climate increases disturbance by insects, changes to habitat could be accentuated by the interactions between fire and insect disturbance regimes.

In this context of changing climate and associated changes in natural disturbance regimes, managers need to decide whether to suppress fires, or manage vegetation, or both. From a management perspective, interactions of climate-induced changes in disturbance regimes with fire suppression and fuels treatment programs are difficult to anticipate based on a simplistic understanding of each factor acting independently. Consequently, it is important to evaluate the interactions among climate change, fire and insect disturbance processes, fire suppression, and fuels treatment more formally. Landscape dynamic simulation models are the most appropriate tool to conduct such evaluations. In this chapter, we illustrate this approach by evaluating the changes in natural disturbance regimes that could result from changes in climate, fire suppression, and vegetation management on the habitat capability of two wildlife species of concern, the American marten (*Martes americana*) and flammulated owl (*Otus flammeolus*) in the northern Rocky Mountains. Recent research on landscape habitat relationships in the study region has found these two species have contrasting ecological relationships. Marten is associated with late-successional closed-canopy mesic forests, whereas flammulated owl is associated with open-canopy, large size-class, dry forest types. These two ecological conditions may be expected to respond differently to changes in fire regime and vegetation management.

9.2 Methods

9.2.1 The Study Landscape

Prospect Creek Basin is a 47,058 ha watershed in the Lolo National Forest of western Montana (Fig. 9.1). We chose this landscape because a regional landscape analysis of biophysical characteristics identified it as highly representative of the surrounding 1,827,400 ha comprising three subsections (Coeur d'Alene Mountains, St. Joe-Bitterroot Mountains, and Clark Fork Valley and Mountains) of the Bitterroot Mountains Ecoregion (Table 9.1). We classified the sample landscape into land cover classes based on the LANDFIRE project (<http://www.landfire.gov>). Specifically, land cover classes represent unique biophysical settings (BpS) or potential vegetation types (PVT). The only significant change we made to this classification scheme was to combine three separate BpS classes corresponding to



Fig. 9.1 Study area orientation map

“riparian” settings into a single “riparian” class. Full documentation of how these BpS classes were derived is available at the LANDFIRE website.

The spatial resolution of the landscape was set at 30 m, consistent with that of the data sources used in the LANDFIRE project. The spatial extent of the landscape was based on the hydrological watershed of Prospect Creek, a tributary of Clark Fork River, but for simulation purposes we included a 2-km wide buffer zone around the basin, bringing the total extent of the simulation landscape to 69,293 ha.

Table 9.1 Comparison of biophysical composition in the Prospect Creek basin study area to that of the surrounding ecosection. The table reports percentages of each landscape in eight biophysical types (<http://www.landfire.gov>)

Type	Percentage of landscape	
	Ecosection	Prospect creek
Mesic-Wet Spruce Fir	23.323	25.654
Mixed-Conifer Ponderosa Pine Douglas-fir	22.779	33.518
Western Hemlock Western Redcedar	20.599	11.269
Mixed Conifer Grand Fir	16.633	17.804
Riparian	5.326	4.649
Mixed Conifer Western Larch	4.382	3.269
Water	1.742	0.443
Subalpine Park	0.897	0.783
Total Area (ha)	1,827,400	47,058

9.2.2 Landscape Simulation Model

We used the Rocky Mountain Landscape Simulator (RMLANDS) (<http://www.umass.edu/landeco/research/rmlands/rmlands.html>) to simulate a variety of disturbance scenarios representing fire and insect disturbance regimes under current and future climate, two fire management strategies, and two vegetation management strategies. RMLANDS is a grid-based, spatially explicit, stochastic landscape model that simulates disturbance and succession processes affecting the structure and dynamics of Rocky Mountain landscapes. RMLANDS simulates two key processes: succession and disturbance. These processes are fully specified by the user (i.e., via model parameterization) and are implemented sequentially within 10-year time steps for a user-specified period of time.

9.2.2.1 Succession

RMLANDS simulates succession using a state-based transition approach in which discrete vegetation states are defined for each cover type. Each cover type has a separate transition model that uniquely defines its successional stages. Succession involves the probabilistic transition from one state to another over time and it occurs at the beginning of each time step in response to gradual growth and development of vegetation. Transition probabilities are typically based on the age of the stand (i.e., the time since the last stand-replacing event), but they can be based on any number of parameters, such as the abiotic setting (e.g., topographic setting) or disturbance history.

Succession is entirely patch-based. Specifically, each cell belongs to a patch, defined as contiguous (touching based on the eight-neighbor rule) cells sharing the same values for each of the attributes used to define succession probabilities. For example, age, time since low-mortality fire, and aspect are all used to define transition probabilities of a particular cover type transition model; contiguous cells with

the same values for these three attributes will be treated as a patch and undergo probabilistic succession transitions together. Successional patches are not static; they change throughout the simulation in response to disturbance events, which can act to break up single patches into several new patches or to coalesce several patches into a single patch by changing the disturbance history at the cell level. This patch-based approach for succession avoids the salt-and-pepper effect that can occur with stochastic cell-based succession.

9.2.2.2 Disturbance Processes

RMLANDS simulates both natural and anthropogenic disturbances. Natural disturbances include wildfire and a variety of insect or pathogen outbreaks (e.g., mountain pine beetle). Each natural disturbance process is implemented separately, but can affect and be affected by other disturbance processes to produce changes in landscape conditions. For example, trees killed by mountain pine beetle can affect the local probability of ignition and spread of wildfires.

Climate plays a significant role in determining the temporal and spatial characteristics of the natural disturbance regime. RMLANDS uses a global parameter, as a proxy for climate, which affects initiation, spread, and mortality of all disturbances within a time step. This parameter can be specified as a constant mean or median with a user-specified level of temporal variability, a trend over time (with a specified variability), or as a user-defined trajectory reflecting the climate conditions during a specific reference period.

Disturbance events are initiated at the cell level, in contrast to succession. In each time step, each cell has a probability of initiation that is a function of its susceptibility to disturbance, and optionally, its spatial or temporal proximity to previous disturbance events or landscape features (e.g., roads). Susceptibility to wildfire, for example, is a function of factors that influence fuel mass and fuel moisture including: cover type, stand condition, time since last fire, time since last insect outbreak, elevation, aspect, and slope. Wildfire susceptibility is also a function of road proximity, which influences the risk of human-caused ignition.

Once initiated, the disturbance spreads to adjacent cells probabilistically. Each cell has a probability of spread that is a function of its susceptibility to disturbance (as above), which is modified by its topographic position relative to a burning cell (i.e., fires can burn more readily upslope), wind direction, and the influence of potential barriers (e.g., roads and streams). The probability of spread is further modified to reflect variable weather conditions associated with the disturbance event. This event modifier affects the final size of the disturbance and is specified as a user-defined size distribution. There is also an optional provision for the spotting of disturbances during spread so that disturbances are not constrained to contiguous spread only. The spotting feature as used in this analysis for both fire and insect disturbances.

Following disturbance spread, each cell is evaluated to determine the magnitude of ecological effect (i.e., severity) of the disturbance. Each cell can exhibit either

high or low mortality of the dominant plants. High mortality occurs when all or nearly all (>75%) of the dominant plant individuals are killed; low mortality is assigned when less than 75% individuals are killed. Cells are aggregated into vegetation patches for purposes of determining mortality response, where patches are defined as spatially contiguous cells having the same cell attributes (e.g., identical disturbance history and age). So-called mixed severity fires produce a heterogeneous mixture of low- and high-mortality cells.

Following mortality determination, each vegetation patch is evaluated for potential immediate transition to a new stand condition (state). Transition pathways and rates of transition between states are defined uniquely for each cover type and are conditional on several attributes at the patch level. These disturbance-induced transitions are different from the successional transitions that occur at the beginning of each time step that represent the gradual growth and development of vegetation over time.

RMLANDS can also simulate a variety of vegetation treatments that result in immediate transition to a new state. These treatments are implemented via management regimes defined by the user. Management regimes are uniquely specified within management zones, or user-defined geographic units (e.g., urban-wildland interface vs. interior). Management zones are further divided into one or more management types based on cover type. Each cover type can be treated separately or it can be combined with other cover types to form aggregate management types. Each management type is then subject to a unique management regime, which consists of one or more treatment types and associated spatial and temporal constraints.

9.2.3 *Wildlife Habitat Capability Model*

We used HABIT@ (<http://www.umass.edu/landeco/pubs/pubs.html>) to quantify the habitat capability of the simulated landscapes for American marten and flammulated owl. HABIT@ is a multi-scale GIS-based system for modeling wildlife habitat capability (Fig. 9.2). We define habitat capability as the ability of the environment to provide the local resources (e.g., food, cover, nest sites) needed for survival and reproduction in sufficient quantity, quality, and distribution to meet the life-history requirements of individuals and local populations. Habitat capability is synonymous with habitat suitability.

HABIT@ models use GIS grids representing environmental variables such as cover type, stand age, canopy density, slope, hydrological regime, roads, and development. Input grids can represent anything pertinent to the species being modeled at any scale, depending only upon the availability of data. Complex derived grids representing specialized environmental variables (such as stream channel constraints, cliffs suitable for nesting, or rainfall patterns) can also be incorporated in HABIT@ models.

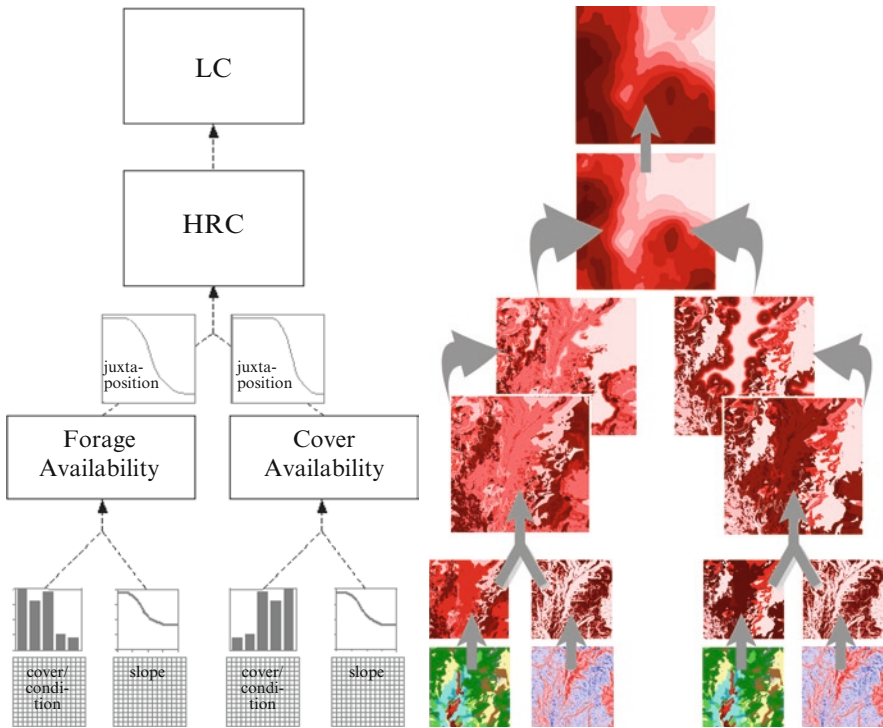


Fig. 9.2 HABIT@ is hierarchically organized into three primary levels. The lowest level comprises one or more local resources, each derived from one or more local resource indices based on GIS data. These are combined and summarized within home range equivalent areas at the second level – Home Range Capability (HRC). Home Range Capability is evaluated over an area much larger than the home range to produce an index of Landscape Capability (LC)

HABIT@ is spatially explicit and models habitat capability at three scales, corresponding to three levels of biological organization:

1. Local Resource Availability (LRA): the availability of resources important to the species life history, such as food, cover, or nesting, at the local, finest scale (a single cell or pixel).
2. Home Range Capability (HRC): the capability of an area corresponding to an individual's home range to support an individual, based on the quantity and quality of local resources, configuration and accessibility of those resources, and condition as determined by intrusion from roads and development.
3. Landscape Capability (LC): the capability of an area to support multiple home ranges; i.e., the ability of an area to support not only a single individual, but a local population.

HABIT@ returns a real number between 0 (no habitat value) and 1 (prime habitat) that represents the relative habitat capability for each cell. At the local resource

level, the cell value indicates LRA, or the resources available at that cell (e.g., value as nesting habitat). At the home range level, the cell value indicates HRC, the resources available in a circular home range of a fixed size centered on that cell. Although the assumption of circular fixed size home ranges is seldom strictly true, it is not an unreasonable generalization when used consistently for comparative purposes. At this level, the HRC value will reflect if there are impediments to movement (e.g., food and nesting resources are across a road from one another). At the LC level, the resulting cell value indicates the value of a home range centered on that cell given that there is habitat in the neighborhood sufficient to support a local population. Specifically, the LC value of a cell reflects the quality of that location within a home range weighted by the sufficiency of the surrounding landscape to support additional proximal home ranges.

HABIT@ models are static; HABIT@ does not model population dynamics nor population viability. The results are relative measures of habitat capability, and do not necessarily correspond to animal density or fitness. HABIT@ is not an individual-based model, in that it does not explicitly model animal movement (although movement is accounted for implicitly in the assumption of home range size). HABIT@ models are, of course, limited by the availability, scale, and accuracy of available data, and the applicability of these data to the species being modeled. As with all habitat modeling, the greatest limitation is usually lack of knowledge of the habitat requirements of the species being modeled, and HABIT@ models are only as good as the biological information used to build them.

9.2.4 The Simulation Experiment

We established a full 2×2×2 factorial design to evaluate the effect on habitat capability of three factors (climate, fire management, and vegetation management) and their interactions (Table 9.2).

Table 9.2 Factors in the modeling experiment

Factor	Levels	Description
Climate	Historical (HC)	Frequency, size, and severity of fire and insect disturbances calibrated to historical (1600–1900) regimes.
	Future (FC)	Frequency and probability of spread of fire and insect disturbances = 1.1 × HC
Fire management	No suppression (NOSUP)	Frequency and size of fires calibrated to historical range of variability.
	Suppression (SUP)	Frequency of fires same as NOSUP; size of fires as in Table 9.3.
Vegetation management	No treatment (NOTRT)	No active vegetation management.
	Treatment (TRT)	WUI fuel reduction treatments with 3,000 ha per decade target; non-WUI post-disturbance salvage treatments up to 2,000 ha per decade.

9.2.4.1 Climate Factor

The two levels for the climate factor represented a contrast between natural disturbance regimes that have occurred under historical climate conditions and those that might be expected under future climate conditions.

Historical climate (HC). To represent natural disturbance regimes that have occurred under historic climate, we set the climate parameter in RMLands for the two dominant disturbance processes as follows:

- *Wildfire*—based on the historical record as represented by the mean Palmer Drought Severity Index (PDSI) (source: National Climatic Data Center), averaged over five sample locations in the vicinity of the study landscape for each 10-year interval for the period 1600–1900 (this sequence was repeated to create the 1,000 year time series needed for the simulation). The climate parameter affected the frequency and spread (i.e., size) of fires, which in combination affected the total area burned (Westerling and Swetnam 2003).
- *Pine beetle*—based on the historical record as represented by the *cumulative threshold* PDSI, averaged over five sample locations in the vicinity of the study landscape for each 10-year interval for the period 1600–1900, as above. The cumulative threshold PDSI is based on the maximum cumulative consecutive years of drought within each 10-year interval, but timesteps with an index < 1 are set to 0, preventing pine beetle disturbances from occurring. This results in periodic or episodic outbreaks (or epidemics) against a background of endemic levels of disturbance. This parameterization was based on local expert opinion and was consistent with published knowledge on beetle-climate interactions (e.g., Rogers 1996).

Future climate (FC). To represent natural disturbance regimes that might occur under future climate conditions, we set the climate parameter as above except increased the mean climate value by 10% (from 1 to 1.1). As implemented, the frequency and probability of spread (at the cell level) of disturbances both increased by 10%. This did not necessarily increase total area disturbed by 10%, however, because the climate parameter is only one of several variables affecting the disturbance processes. This factor level represents the case in which the frequency and severity of climate conditions conducive to burning and bark beetle outbreaks are increased by 10%. Although it is only one of many possible alternative future climate scenarios, we considered it to be within expectations of GCM predictions.

9.2.4.2 Fire Management

Two fire management factor levels represented a contrast between a “no suppression” policy and an “aggressive fire suppression” policy.

No suppression (NOSUP). To represent no suppression fire management policy, the frequency, size and severity of wildfires are based solely on the estimated historic range of variability (<http://www.landfire.gov>; Table 9.3).

Table 9.3 Percentage of fires in each size class under suppression and no suppression scenarios. Percentages in each size category are estimates from expert opinion

Size (ha)	Percentage of fires	
	No suppress	Suppress
1	76.25	92.25
10	12.00	4.00
100	6.00	2.00
1,000	3.00	1.00
10,000	1.50	0.50
100,000	0.75	0.25

Suppression (SUP). There are many possible ways to emulate the effect of fire suppression on fire frequency, size, and severity. For this factor level, we assumed that fire suppression per se does *not* change the frequency of ignitions or the severity of fires—although indirectly it will likely increase both over the long term if the vegetation becomes more flammable with age. Instead, we assumed that fire suppression directly affects the probability of fire spread, and thus directly influences the distribution of fire sizes. To emulate this effect, we modified the size distribution of fires in the spread parameters for wildfire (Table 9.2).

The settings here are designed to emulate a fire suppression policy that is reasonably but not perfectly effective in preventing the spread of fires. Consequently, while the distribution has many more small fires, very large fires (including the maximum fire size) still occur under the suppression scenario, but with a reduced probability.

9.2.4.3 Vegetation Management

Two vegetation management factor levels represented a contrast between “no treatment” and “aggressive vegetation treatment” management strategies.

No treatment (NOTRT). We simulated no active vegetation management to represent a “do nothing” management strategy.

Vegetation treatment (TRT). To represent an aggressive vegetation management policy, we attempted to emulate the current National Forest management focus on: (1) fuels reduction in the wildland-urban interface and (2) salvage of timber following large-scale disturbance events in the non-wildland-urban interface. Other management objectives, such as ecosystem restoration and timber stand improvement, were addressed only indirectly as by-products of the vegetation treatments aimed at fuels reduction and postfire salvage. Treatments were excluded from private lands, unsuitable timberland (as designated), riparian zones, and roadless areas. All other lands were considered eligible for treatments. The parameterization of vegetation treatments in RMLANDS can be quite complex. Rather than try to

describe the detailed parameterization, a summary of the important distinctions is given below:

1. Wildland-urban interface (WUI) zone:

Objective. WUI treatments were designed primarily to reduce fuels, thus reduce the risk of high-severity fire and improve the likelihood of effective fire suppression.

Treatment intensity. The goal was to treat all eligible lands within the WUI (approximately 15,000 ha). Recognizing that even under the best circumstances, it is highly unlikely—and may not be desirable—to treat 100% of the eligible land, we instead assumed a target of 3,000 ha of land treated per decade on a 40-year treatment interval, with a goal of treating 12,000 ha every 40 years. However, numerous factors conspire against meeting this target in the simulation. For example, although closed-canopy forest conditions were targeted for treatment, previous occurrence of wildfire can leave considerably less closed-canopy forest to treat, and in some cases, less than the target. If we constrain treatments to sufficiently large contiguous areas of eligible lands containing suitable forest conditions for logistical and economic reasons, there will be locations and times when patches of eligible forest are simply too small or too scattered for efficient treatment. Thus, the targeted “maximum treatment area per timestep” was not necessarily met, but rather was a flexible target that varied depending on the vegetation conditions and due to other constraints.

Treatment regime. Two silvicultural treatments were simulated: (1) “restoration” treatments, which involve the combination of individual tree removal (i.e., basal area reduction) and prescribed underburning (i.e., low mortality fire); and (2) “individual tree selection”, which involves individual tree removal without underburning. Treatments were aggregated in units of 4–200 ha.

2. Non-wildland-urban interface (non-WUI) zone:

Objective. Non-WUI treatments were designed primarily to salvage timber following major wildfires and insect outbreaks.

Treatment Intensity. Given the spatial constraints outlined above, approximately 17,000 ha of land were eligible for treatments in the non-WUI zone. The goal was to salvage up to a *maximum* of 2,000 ha in any decade experiencing extensive wildfires or insect outbreaks.

Treatment regime. Silvicultural treatments included a combination of “clearcut” and “individual tree selection”. Both treatments were single-entry treatments without follow-up. Treatments were aggregated in units of 4–40 ha for clearcut and 10–200 ha for individual tree selection.

9.2.4.4 Landscape Capability Analysis

Each of the eight treatment combinations in the $2 \times 2 \times 2$ factorial was simulated using RMLANDS. Simulations were run for 1,000 years, at 10 year time steps.

Each treatment combination was simulated once for 1,000 years, with 10-year time steps. The simulation landscape was initialized with the current vegetation type and seral stage obtained from the Landfire Program (<http://www.landfire.gov>) and temporal dynamics stabilized within 200 years. Output grids of the vegetation cover type and condition plus a group of grids associated with wildfires and pine beetles were saved at each time step for each simulation. We described the composition and configuration of the cover-condition grids at each timestep using FRAGSTATS (McGarigal et al. 2002). Metrics that have been shown to be important predictors of habitat capability for two target species were computed (see below).

We described habitat capability using RMLands output and recently developed HC models for American marten (Wasserman et al. 2008) and flammulated owl (Cushman et al. unpublished data). The habitat suitability model for American marten in northern Idaho suggests that marten select landscapes with high average canopy closure and low fragmentation (Wasserman et al. 2008). Within these unfragmented landscapes, marten select foraging habitat at a fine scale within middle-elevation, late-successional, mesic forests. In northern Idaho, optimum American marten habitat therefore consists of landscapes with low road density, low density of patches and high contrast edges, with high canopy closure and large areas of middle-elevation, late-successional, mesic forest.

As implemented in HABIT@, the model estimates LRA as the product of three Local Resource Indices (LRIs): one for cover, one for adverse edge effects, and one for road intensity. LRI_{cover} is based on the cover-condition grid at the focal cell (Wasserman et al. 2008). LRI_{edge} takes into account the distance to adverse edges, as defined in Wasserman et al. (2008). LRI_{edge} increases with distance to an adverse edge according to a logistic function (Wasserman et al. 2008). LRI_{roads} is based on the distance-weighted road density within a 2,000-m radius circular window. The home range capability for marten is simply the mean LRA across a 630 m radius circular home range (125 ha), inversely scaled by distance. Landscape Capability (LC) is based on the number of 630 m radius home ranges that can be placed on the landscape by tiling non-overlapping home ranges starting with the cells with the highest HRC. Home ranges are then dropped by using the HRC as the probability of retaining each home range (e.g., a home range centered on a cell with a HRC of 0.85 will have a 15% chance of being dropped). Landscape Capability is the total number of home ranges in the landscape for that time step.

The habitat suitability model for flammulated owl includes canopy closure, elevation, edge density, patch richness density, correlation length of warm-dry forest types, landscape percentage area in grass cover types and riparian cover types (Cushman et al. unpublished data). Optimal flammulated owl habitat, as predicted by this model, consists of middle-elevation landscapes, with extensive warm-dry forest, relatively large amounts of grass and intermediate amounts of riparian cover types, intermediate canopy closure, and relatively high landscape heterogeneity, as indicated by the variables patch richness density and edge density.

The flammulated owl model is based on seven environmental variables that predict LRA: canopy cover, elevation, edge density, patch richness density, correlation length of warm-dry forest types, landscape composition by grass cover types, and landscape composition by riparian cover types. LRA is based on implementing

the logistic-regression equation for the flammulated owl habitat model in Cushman et al. (submitted). The HRC for flammulated owl is simply the mean LRA across a 630 m radius circular home range (125 ha), inversely scaled by distance. LC is based on the number of 630 m radius home ranges that can be placed on the landscape by tiling non-overlapping home ranges starting with the cells with the highest HRC. Home ranges are then dropped by using the HRC as the probability of retaining each home range (e.g., a home range centered on a cell with a HRC of 0.85 will have a 15% chance of being dropped). Landscape Capability is the total number of home ranges in the landscape for that time step.

We applied the American marten and flammulated owl HABIT@ models to the simulation output for each timestep under each scenario. Figure 9.3 shows a snapshot of a LRI and the HRC for American marten under one of the eight

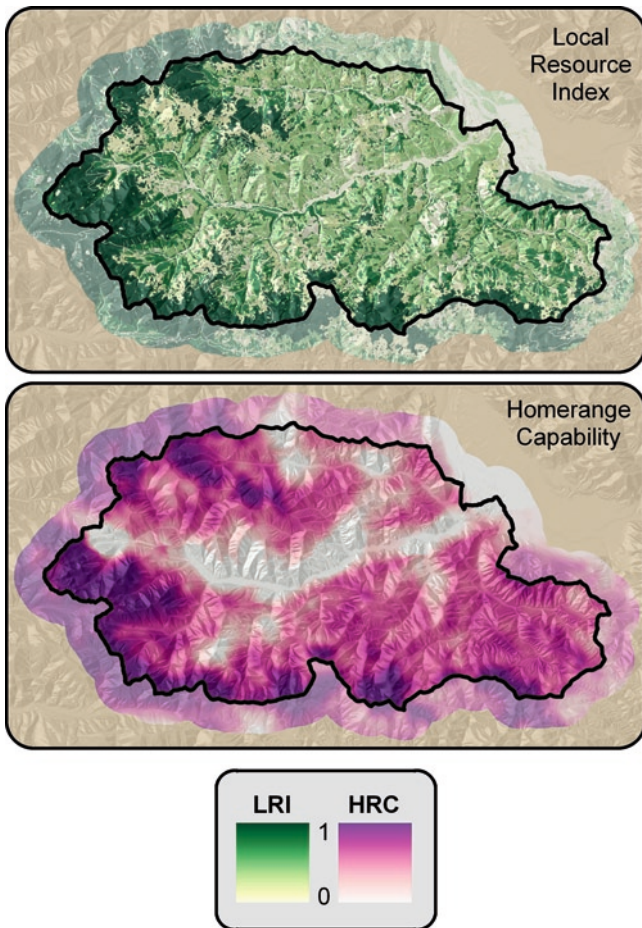


Fig. 9.3 Local resource index for cover-condition (LRI) (*top* figure) and Home Range Capability (HRC) (*bottom* figure) derived from the marten HABIT@ model for the 200 year timestep under the historical climate–no suppression–no vegetation treatment scenario

scenarios after 200 years of the simulation, which is when system dynamics stabilize. For each of the eight scenarios and each species, we calculated the total expected number of home ranges in the landscape (LC) based on HRC for each species at each time step. We summarized the differences among the eight scenarios by computing the mean, median, inter-quartile range, and standard error of LC across 10-year time steps. We tested for significant differences in LC in relation to climate, suppression, treatment, and their interactions using factorial analysis of variance.

9.3 Results

For marten, simulated fire suppression significantly increased Landscape Capability (LC), while simulated future climate significantly decreased LC (Table 9.4), and appears to increase its variability (Fig. 9.4). Similarly, simulated fire suppression significantly increased predicted habitat capability and simulated future climate decreased habitat capability for the flammulated owl (Table 9.4). The dominant effect is increased variability in the home range capability of the study area in the simulated future climate regime (Fig. 9.4). For both species, it appears that LC in the future climate regime will be lower than in the current climate regime regardless of the management scenario implemented. Vegetation fuels treatments did not appear to have a significant effect on LC for either species (Table 9.4)

To clarify the impact of climate change, fire suppression, and vegetation treatment on LC, we present box plots for each of these main effects, across the levels of the others. First, for both marten and flammulated owl, simulated increases in rates of disturbance under future climate nominally reduced the mean and increased the variability of LC in this study area (Fig. 9.5). Similarly, for both marten and flammulated owl, the higher disturbance rates associated with no suppression of wildfire nominally increased mean and variability of LC (Fig. 9.6, Table 9.4). In contrast, simulated vegetation treatments had virtually no effect on the mean or variability of LC for either species (Fig. 9.7, Table 9.4).

Table 9.4 Results of factorial analysis of variance for American marten and flammulated owl Landscape Capability in relation to climate, fire suppression and fuels treatment. For both species, capability is significantly higher under current than future climate and significantly higher under fire suppression than no suppression. There were no significant effects of vegetation treatment or interactions

Term (treatment)	Marten landscape capability		Owl landscape capability	
	Estimate	Prob> t	Estimate	Prob> t
Intercept	232.985360	<0.0001	36.576241	<0.0001
CLIMATE	-8.581242	0.0002	-1.017071	0.0373
BURN	-9.758519	<0.0001	-1.371524	0.0056
TREAT	1.801136	0.4156	-0.176619	0.7131
Timestep	-0.133488	0.0026	-0.002687	0.0051
CLIMATE*BURN	1.546739	0.7261	-0.002687	0.3836
CLIMATE*TREAT	2.559228	0.5625	-0.838995	0.7497
BURN*TREAT	0.911939	0.8363	-0.306442	0.6611

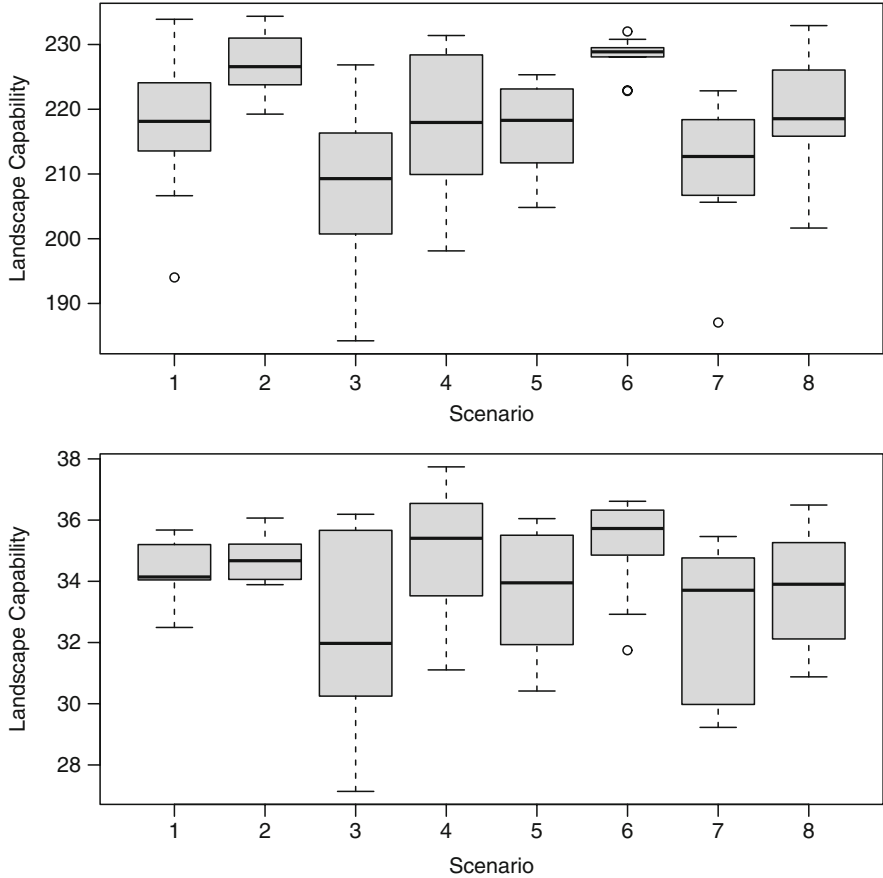


Fig. 9.4 Boxplots for the eight scenarios. (*top*) marten; (*bottom*) flammulated owl. The y-axis is the Habitat@ Landscape Capability index based on home range habitat capability in the Prospect Creek study area. The eight scenarios are distributed along the x-axis. Scenarios are numbered as in Table 9.1: 1: historical climate, no suppression, no vegetation treatment; 2: historical climate, suppression, no vegetation treatment; 3: future climate, no suppression, no vegetation treatment; 4: future climate, suppression, no vegetation treatment; 5: historical climate, no suppression, vegetation treatment; 6: historical climate, suppression, vegetation treatment; 7: future climate, no suppression, vegetation treatment; 8: future climate, suppression, vegetation treatment

9.4 Discussion

To evaluate the relative expected impacts of climate change, fire suppression, and fuels treatment on wildlife habitat in the northern Rocky Mountains, we used the most recent and robust empirical models of species-habitat relationships for American marten and flammulated owl available. This provides the strongest available understanding of the factors that predict the occurrence of these two species as functions of multi-scale habitat conditions. We coupled multi-scale empirical models

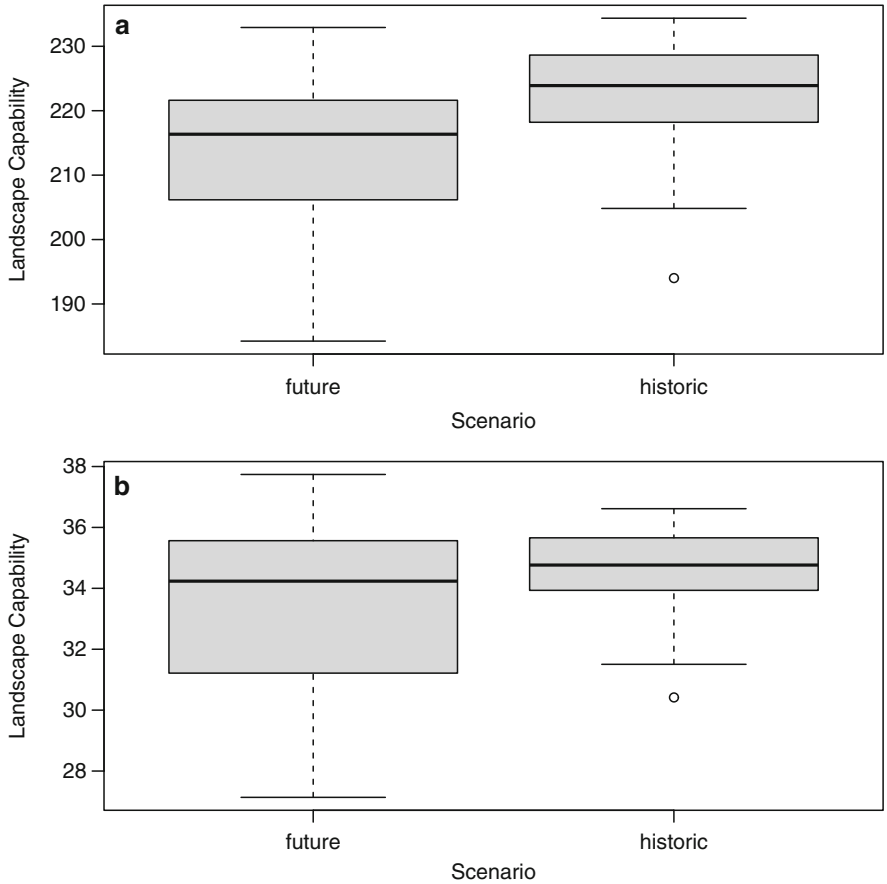


Fig. 9.5 Boxplots for climate change effects. **(a)** marten; **(b)** flammulated owl. The y-axis is the HABIT@ Landscape Capability index based on home range habitat capability in the study area. The x-axis is the climate scenarios (future vs. historic) pooled across the other factors

to spatial simulation of eight scenarios in a factorial framework enabling the evaluation of the relative effects of climate change, fuels treatment, and fire suppression on the habitat capability for each of these species. These scenarios reflect reasonable relative effects of management and potential climate change. In terms of management effects, we simulated both potential effects of fire suppression and of fuels reduction treatments. Our simulation assumed that fire suppression does *not* change the frequency of ignitions or the severity of fires, but does directly affect the probability of fire spread, and thus directly influences the distribution of fire sizes. Our fire suppression scenarios reflect aggressive suppression that may over-represent actual management effectiveness. Thus, we view our fire suppression scenarios as illustrating the largest reasonable effect possible due to suppression. Similarly, the fuels

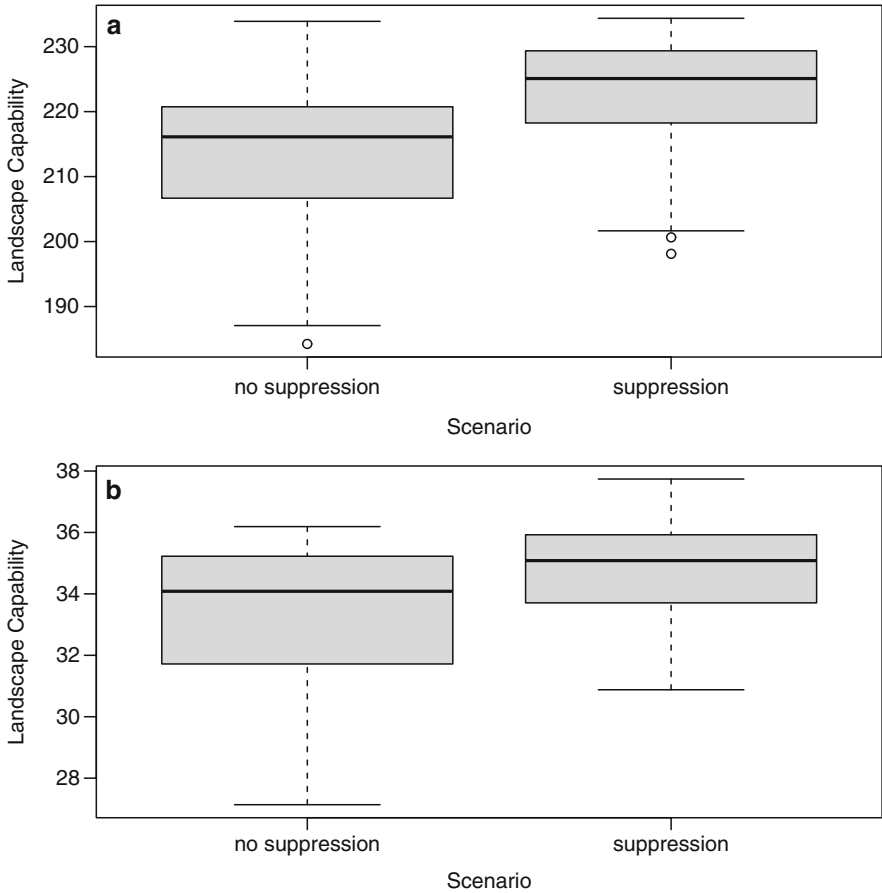


Fig. 9.6 Boxplots for fire suppression effects: (a) marten, (b) flammulated owl. The y-axis is the HABIT@ Landscape Capability index based on home range habitat capability in the study area. The x-axis is the fire suppression scenarios (nosuppress vs suppress) pooled across the other factors

treatment scenarios reflect aggressive vegetation management. We emulated the current National Forest management focus on fuels reduction and salvage of timber with a goal of treating all eligible lands within the WUI on roughly a 40-year treatment interval. This highly aggressive simulated fuels treatment program is probably more ambitious than would be possible to implement given legal, logistic, and financial limitations facing forests, and thus we view our fuels treatment scenarios as the largest reasonable effect possible due to fuels treatment.

We simulated future climate effects on disturbance regimes as a 10% increase in the frequency and severity of climate conditions that are conducive to burning and bark beetle outbreaks. Climate change is expected to have complex effects on fire

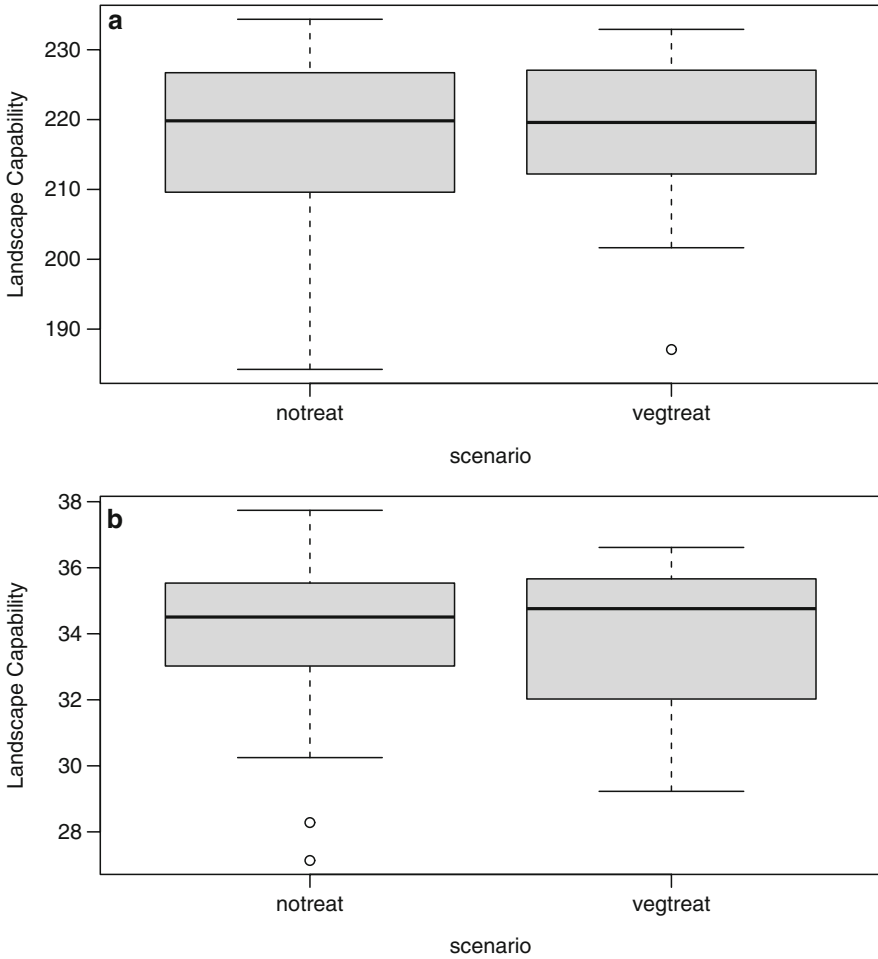


Fig. 9.7 Boxplots for fuels treatment effects: (a) marten; (b) flammulated owl. The y-axis is the HABIT@ Landscape Capability index based on home range habitat capability in the study area. The x-axis is the vegetation treatment scenarios (notreat vs. vegtreat) pooled across the other factors

and insect disturbance processes at landscape scales (Cushman et al. 2007), making simple parameterization of a single climate-change scenario difficult. Identifying the most likely potential future effect of expected climate change is a major research task requiring the integration of the latest downscaled climate models with sophisticated fire and insect disturbance models (Cushman et al. 2007). This challenging task will likely require the integration of multiple modeling efforts and take a number of years to produce reliable predictions (Cushman et al. 2007). Instead, we chose to implement a relatively simple but nonetheless plausible potential

future. Our scenario of a 10% increase in the frequency and severity of conditions favoring wildfire and bark beetles is generally consistent with recent research suggesting that recent climate changes have increased the area burned in wildfire (Westerling et al. 2006; Littell et al. 2009) and impacted by bark beetles (Logan et al. 2003; Berg et al. 2006). Future climate warming is expected to further increase the area impacted by fire and bark beetles in the Western United States (McKenzie et al. 2004; Running 2006; Westerling et al. 2006). Our choice of a 10% increase in these parameters is admittedly arbitrary but is probably conservative given observed and expected changes in relation to changing climate. Thus, in contrast with the management options, the effects of climate change in our scenarios probably reflect the minimum expected change due to climate forcing. Therefore, the scenarios simulated are preliminary and hypothetical, in that we did not exhaustively evaluate a range of potential climate change effects or explicitly simulate any particular predictions of future climate from current GCMs. While this is not a definitive exploration of potential climate change effects, we believe it is reasonable and that the results suggest plausible landscape and habitat changes in the future.

The first major conclusion from these simulations is that the expected changes in disturbance regimes due to climate change will likely have a larger effect on landscape habitat capability of these two species than active fire and vegetation management, and that even very aggressive management involving suppression and fuels treatment will not substantially alter climate-driven effects on habitat for these species in the northern Rocky Mountains. This conclusion follows from the simulation results, which show that increased fire and insect disturbance under the future climate decreases the median and increases the variability of Landscape Capability for both species, and this holds true even when aggressive fuels treatment and fire suppression are simulated. Our simulation is preliminary, but the results are striking because our simulated climate change scenario was intentionally conservative and our suppression and fuels treatment scenarios were intentionally somewhat more aggressive than realistic.

Simulations illustrate that altered disturbance regimes may have substantial effects on wildlife habitat capability due to changes in the structure and composition of vegetation communities. In this simulation both American marten and flammulated owl appear to be negatively impacted by these changes even though they have very different relationships to vegetation. *A priori*, one could reasonably expect that increased fire and insect activity would adversely affect American marten, which is a closed-canopy, late-successional, interior forest species. In contrast, one would expect that increased fire and insect disturbance could benefit flammulated owl, given its preference for intermediate levels of canopy closure and high levels of landscape heterogeneity. However, the LC for both species was reduced slightly under the simulated future climate regime and variability over time increased substantially. The increase in variability may have important implications for population viability, as it is possible that both populations could face severe bottlenecks even if LC only infrequently drops to relatively low levels under these more variable future conditions. These bottlenecks could increase the

chance of local extinction due to demographic stochasticity in temporarily reduced populations.

Another implication of these results regards the relative impacts of fire suppression and fuels treatment on habitat capability of these species. In the case of marten, one might expect that fuels treatments and fire suppression would have positive effects by increasing late-successional closed-canopy conditions at middle and high elevations. This was true for suppression, which nominally increased marten habitat capability under both historical and future climate. Fire suppression, however, did not decrease variability in habitat capability over time; indeed, suppression slightly increased variability in habitat capability under the future climate scenario. Fuels treatments did not appear to have any discernible effect on the expected median habitat capability for American marten under either historical or future climate, probably because even extremely aggressive fuels treatments did not substantially alter the area and severity of fire and insect disturbance. In the case of the flammulated owl, one might expect that fire suppression and fuels treatments would have opposite effects on habitat capability, with fire suppression reducing the suitability of low elevation warm-dry forest types for this species by encouraging transition to closed-canopy conditions and homogenization of the landscape mosaic, and increasing habitat suitability by decreasing canopy closure and increasing heterogeneity among patches in the landscape mosaic. These opposing effects were not observed in our simulation results, however. Fire suppression appeared to have a very minor effect on flammulated owl habitat capability, nominally increasing median habitat capability.

Also contrary to expectation, fuels treatment did not have any discernible effect on habitat capability for flammulated owl. In the simulation model, the amount of disturbance is what affects the cover condition most dramatically. The treatment targets had relatively little effect on disturbance regimes, and thus treatments did not markedly affect habitat capability. The area affected by natural disturbance regimes is impressive when compared to what is feasible with management.

9.5 Conclusion

Climate change and natural resources management activities will likely interact in complex ways to affect forest ecosystems and wildlife habitat. Importantly, altered disturbance regimes, fire suppression, and fuels treatments may result in unexpected outcomes. Simulation modeling is a powerful tool available to investigate these interactions and evaluate possible outcomes. In this analysis we explored potential interactions of climate change, fire suppression, and fuels treatment on the habitat capability of two wildlife species with highly contrasting habitat relationships in the northern Rocky Mountains. We expected climate change to negatively affect habitat quality for American marten, and we expected that fire suppression and fuels treatments would benefit this species by increasing the proportion of the

landscape in late-successional closed-canopy conditions. Likewise, we expected that climate change would benefit flammulated owl by increasing the amount of low- and mixed-severity disturbance in warm-dry forest types, thereby increasing average canopy openness and landscape heterogeneity. We also expected that fuels treatments would benefit the owl, while fire suppression would lead to reductions in habitat capability. The simulations, however, indicated that many of these expectations were not likely outcomes.

While this exploration is preliminary, we believe it makes three main points. First, climate change and management may interact in complex and nonintuitive ways that defy expectations based on simplistic understanding of each process acting separately. Formal evaluations of the interaction of these multiple processes under realistic scenarios is essential to guide adaptation and mitigation strategies for addressing the effects of climate change. Second, climate-driven changes to disturbance regimes may have larger effects than fire suppression or vegetation treatment. Thus, management tools available to mitigate climate change may have limited power to substantially alter the trajectory of landscape change under altered future disturbance regimes (but see Chap. 10). Third, fuels treatments in particular appear to be of very limited utility in altering habitat capability of these species under current or future climate. This is because they have limited ability to alter overall disturbance regimes, even when they appear to be on aggressive schedules.

We remind the reader that this analysis is preliminary, with simplistic assumptions and representations of how natural disturbance regimes might change with changes in climate and fire management strategies. These are reasonable scenarios that can suggest the kind and approximate magnitude of effects on LC, but they are not comprehensive. Therefore, further research is needed to explore how a range of climate change scenarios and concomitant effects on disturbance regimes interact with a range of management scenarios to affect landscape dynamics and wildlife habitat capability.

The linkage of rigorous, empirical, multi-scale habitat models with spatial simulation of alternative disturbance regimes is a powerful framework to evaluate the potential effects of a range of alternative scenarios on the habitat and populations of wildlife species of interest. For this critical task to be addressed rigorously requires that three major conditions are satisfied. First, reliable spatially explicit models for species of interest must exist. Second, meaningful alternative disturbance scenarios involving climate forcing and realistic alternative management responses must be simulated in a realistic manner using dynamic simulation models such as RMLANDS. Third, the habitat models must be applied to simulation results over time to estimate impacts on habitat under each scenario, requiring that the habitat models and simulation outputs match in terms of attributes and scales of ecological variables. Given sufficient care these conditions can be satisfied for many species and many systems. With this approach, we have the tools in hand to anticipate the potential effects of changing and interacting disturbance regimes on habitat for multiple species.

References

- Bachelet, D., J.M. Lenihan, C. Daly, R.P. Neilson, D.S. Ojima, and W.J. Parton. 2001. *MC1: A dynamic vegetation model for estimating the distribution of vegetation and associated ecosystem fluxes of carbon, nutrients, and water*. General Technical Report PNW-GTR-508. Portland: U.S. Forest Service.
- Baker, W.L. 1995. Longterm response of disturbance landscapes to human intervention and global change. *Landscape Ecology* 10: 143–159.
- Berg, E.E., J.D. Henry, C.L. Fastie, A.D. DeVolder, and S.M. Matsuoka. 2006. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: Relationship to summer temperatures and regional differences in disturbance regimes. *Forest Ecology and Management* 227: 219–232.
- Christensen, N.L. 1988. Succession and natural disturbance: Paradigms, problems, and preservation of natural ecosystems. In *Ecosystem management for parks and wilderness*, eds. J.K. Agee, and D.R. Johnson, 62–81. Seattle: University of Washington Press.
- Cushman, S.A., D. McKenzie, D.L. Peterson, J. Littell, and K. McKelvey. 2007. *Research agenda for integrated landscape modelling*. General Technical Report RMRS-GTR-194. Fort Collins: U.S. Forest Service.
- Cushman, S.A., B. Hahn, and A. Cilimburg. (unpublished data). Multiple scale habitat selection by flammulated owl in the northern Rocky Mountains. Manuscript in review. *Ecological Applications*.
- Dale, V.H., L.A. Joyce, S. McNulty, R.P. Neilson, M.P. Ayres, M.D. Flannigan, P.J. Hanson, L.C. Irland, A.E. Lugo, C.J. Peterson, D. Simberloff, F.J. Swanson, B.J. Stocks, and B.M. Wotton. 2001. Climate change and forest disturbances. *Bioscience* 51: 723–734.
- Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner, and B.J. Stocks. 2005. Future area burned in Canada. *Climatic Change* 72: 1–16.
- Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, and D. Xiaosu, eds. 2001. *Climate Change 2001: The Scientific Basis: Contributions of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge: Cambridge University Press pp 881.
- Iverson, L.R., A.M. Prasad. 2002. Potential redistribution of tree species habitat under five climate change scenarios in the eastern US. *Forest Ecology and Management* 155: 205–222.
- Keane, R.E., P. Morgan, and J.D. White. 1999. Temporal patterns of ecosystem processes on simulated landscapes in Glacier National Park, Montana, USA. *Landscape Ecology* 14: 311–329.
- Littell, J.S., D. McKenzie, D.L. Peterson, and A.L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications* 19: 1003–1021.
- Logan, J.A., J. Regniere, and J.A. Powell. 2003. Assessing the impact of global warming on forest pest dynamics. *Frontiers in Ecology and the Environment* 1: 130–137.
- McGarigal, K., S.A. Cushman, M.C. Neel, and E. Ene. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. <http://www.umass.edu/landeco/research/fragstats/fragstats.html>. Accessed 5 Mar 2010.
- McKenzie, D., D.L. Peterson, and E. Alvarado. 1996. *Predicting the effect of fire on large-scale vegetation patterns in North America*. Research Paper PNW-RP-489. Portland: U.S. Forest Service.
- McKenzie, D., Z. Gedalof, D.L. Peterson, and P. Mote. 2004. Climatic change, wildfire, and conservation. *Conservation Biology* 18: 890–902.
- Mearns, L.O., R.W. Katz, and S.H. Schneider. 1984. Extreme high-temperature events: Changes in their probabilities with changes in mean temperature. *Journal of Climate and Applied Meteorology* 23: 1601–1613.
- Miller, C. 2003. Simulation of effects of climatic change on fire regimes. In *Fire and climatic change in temperate ecosystems of the Western Americas*, eds. T.T. Veblen, W.L. Baker, G. Montenegro, and T.W. Swetnam, 69–94. New York: Springer.

- Neilson, R.P. 1995. A model for predicting continental-scale vegetation distribution and water balance. *Ecological Applications* 5: 362–385.
- Neilson, R.P., and R.J. Drapek. 1998. Potentially complex biosphere responses to transient global warming. *Global Change Biology* 4: 505–521.
- Neilson, R.P., L.F. Pitelka, A.M. Solomon, R. Nathan, G.F. Midgley, J.M. Fragoso, H. Lischke, and K. Thompson. 2005. Forecasting regional to global plant migration in response to climate change. *BioScience* 55: 749–759.
- Overpeck, J.T., D. Rind, and R. Goldberg. 1990. Climate-induced changes in forest disturbance and vegetation. *Nature* 343: 51–53.
- Peter, S., and K. Vranes. Intergovernmental Panel on Climate Change (IPCC). In *Encyclopedia of Earth*, Eds. Cutler J. Cleveland, (Washington, D.C.: Environmental Information Coalition, National Council for Science and the Environment). [First published in the *Encyclopedia of Earth* May 6, 2008; Last revised Date May 6, 2008; Retrieved November 8, 2010 [http://www.eoearth.org/article/Intergovernmental_Panel_on_Climate_Change_\(IPCC\)](http://www.eoearth.org/article/Intergovernmental_Panel_on_Climate_Change_(IPCC))]
- Peters, R.L., and T.L. Lovejoy. 1992. *Global warming and biological diversity*. New Haven and London: Yale University Press.
- Rogers, P. 1996. *Disturbance ecology and forest management: A review of the literature*. General Technical Report INT-GTR-336. Ogden: U.S. Forest Service.
- Running, S.W. 2006. Climate change: Is global warming causing more, larger wildfires? *Science* 313: 927–928.
- Solomon, A.M., and R. Leemans. 1997. Boreal forest carbon stocks and wood supply: Past, present, and future responses to changing climate, agriculture, and species availability. *Agricultural and Forest Meteorology* 84: 137–151.
- Thompson, R.S., S.W. Hostetler, P.J. Bartlein, and K.H. Anderson. 1998. A strategy for assessing potential future changes in climate, hydrology, and vegetation in the western United States. U.S. Geological Survey Circular 1153.
- Westerling, A.L., and T.W. Swetnam. 2003. Interannual to decadal drought and wildfire in the western United States. *Eos Interactions* 84: 545–560.
- Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam 2006. Warming and earlier spring increases western U.S. forest wildfire activity. *Science* 313: 940–943. doi:10.1126/science.1128834. Online supplement.
- Wasserman, T.N. 2008. *Multi-scale habitat relationships and landscape genetics of Martes Americana in Northern Idaho*. M.S. Thesis. Western Washington University.