# Landscape Development and Mule Deer Habitat in Central Oregon

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## Abstract

This research explored the ecological consequences of rural residential development and different management regimes on a tract of former industrial timberland in central Oregon known as the Bull Springs. Forage quality and habitat suitability models for mule deer (*Odocoileus hemionus*) winter range were joined to the outputs of a spatially explicit vegetation dynamics model under two management scenarios. In one scenario, the tract was managed as a working forest excluding development, and in the other, development was allowed to occur at historical rates. Landscape pattern analysis was used to measure differences between the outcomes of the two scenarios. Our efforts showed that allowing development on the tract could potentially lead to greater isolation, smaller habitat patches, and decreased extensiveness of patches used for foraging across mule deer winter range. Patches providing multiple habitat functions also became more isolated and less numerous in our simulations. Although neither scenario prevented habitat degradation, restricting development on Bull Springs had slightly more favorable simulated outcomes for forage and multifunctional habitat conditions. Management of this tract as a working forest in a region under pressure for more residential development could reduce the negative effects of development on an iconic species in the region. This research provides insight into how the land use change trajectory of a small portion of the landscape can influence the larger ecological conditions of a region undergoing rapid rural residential development.

Keywords: Mule deer; landscape ecology; rural residential development; alternative development scenarios; habitat suitability.

## Introduction

Low-density housing has expanded into rural lands and the wildland-urban interface (WUI) across the United States (Theobald and Romme 2007) and represents an accelerating phenomenon in the West (Brown et al. 2005) capable of altering social and ecological landscapes. The relative permanency of development distinguishes it from extractive land uses such as logging and grazing (Hansen et al. 2005), and its impacts extend beyond the walls of individual homes. Disturbance regimes, biodiversity, and myriad other ecosystem services have all demonstrated sensitivity to the extent and nature of residential development on rural lands (Dale et al. 2005, Hansen et al. 2005, Rindfuss et al. 2004). For example, individual residences create localized disturbance zones for wildlife (Theobald et al. 1997), and developments propagate disturbances, especially fire and invasive species, into adjacent undeveloped lands (Hansen and DeFries 2007). The cumulative effects of individual land use change decisions can lead to substantial ecological impacts (Theobald et al. 2005), and uncoordinated residential development over time can have disproportionate effects on potential wildlife habitat (Spies et al. 2007) and disrupt migration corridors (Hansen and Defries 2007).

Central Oregon has exemplified the western American trend of residential development taking place on forested land previously managed for timber and other non-residential uses. When plans emerged to develop housing on a

13 000-ha private tract of former timberland just west of Bend, the region's largest city, questions arose about how different management actions on the tract would affect the region's natural resources. Although the tract, known as the Bull Springs, was zoned for forest use and a minimum lot size of 97 ha at the time of the proposal, there was a precedent of converting land zoned for forest use to residential use in other parts of the state (Lettman 2002, 2004). Furthermore, the tract overlaps a substantial portion of the observed winter range for mule deer (*Odocoileus hemionus*). One potential outcome of land use, ownership, and management changes in Central Oregon is a shift in the amount, quality, and spatial pattern of habitat for wildlife such as mule deer. Mule deer use higher elevation woodlands in the summer, however, their movement to lower elevation valleys and sagebrush in the winter could bring them into contact with the proposed development. As mule deer require large home ranges (up to 500 ha for solitary bucks) and long dispersal distances (often exceeding 1 km), the proposed development could fragment important habitat patches or disrupt migration corridors. Development in the winter range also has potential to impact mule deer foraging spaces as well as valuable hiding habitats that provide camouflage from predators and thermal cover habitats that protect them from wind and sun (Csuti et al. 1997, Johnson and O'Neil 2001, Oregon Department of Fish and Wildlife (ODFW) 2003). An iconic species in the region and an important source of game (ODFW 2003), mule deer and their winter range requirements have often been at the center of the debate over the possible policy options for guiding land use change on the Bull Springs tract following its ownership change, and so we chose mule deer habitat as the indicator of the ecological outcomes of alternative policies for this study.

To better understand the impacts of shifts in the landscape patterns resulting from changes in land ownership and land use on mule deer winter range habitat, we analyzed differences in the landscape patterns between two management options by examining potential habitat configurations simulated under each option. We present methods that quantify the landscape consequences of differing policy options and, by providing a means to weigh alternative policy scenarios, may be useful to decision makers. Two alternative development scenarios for the Bull Springs tract were considered plausible in the future:

(1) The tract is managed as a working forest and not developed (WF).

(2) The tract becomes developed for residential use at historical rates (DEV).

Development was allowed to occur outside of the Bulls Springs regardless of the scenario.

To understand how different policy options might influence the spatial arrangement of mule deer habitat, we used spatial pattern metrics to quantify change under both scenarios for a mule deer forage quality model and a habitat suitability model. These types of analyses can help identify the effects of working forest management on mule deer habitat in the region, and show the role the Bull Springs tract could play in the larger suite of developments forecast to occur within the region.

## Data and Methods

Vegetation dynamics in the region surrounding the Bull Springs tract and mule deer winter range were simulated under two scenarios for 60 years. The two scenarios were based on the proposed options for the tract arising in the public policy debate in Oregon. The Working Forest (WF) scenario excluded development from the Bull Springs tract and managed it for restoration goals. The Development (DEV) scenario allowed development to occur at historical rates on the tract. We classified the initial (2000) and final (2060) vegetation conditions into forage and habitat suitability categories using wildlife-habitat relationship models (described below). These categorical maps were then analyzed using spatial pattern metrics to quantify the changes in mule deer habitat and compare the ecological outcomes of the scenarios. Analyses were conducted at three spatial extents: the full study area, mule deer winter range, and Bull Springs boundary (fig. 1).

### Study Area

Centered on the Bull Springs tract, the study area is 430 000 ha of the dry, eastern slopes of the Cascade Range and the western edge of Oregon's high desert and ranges in

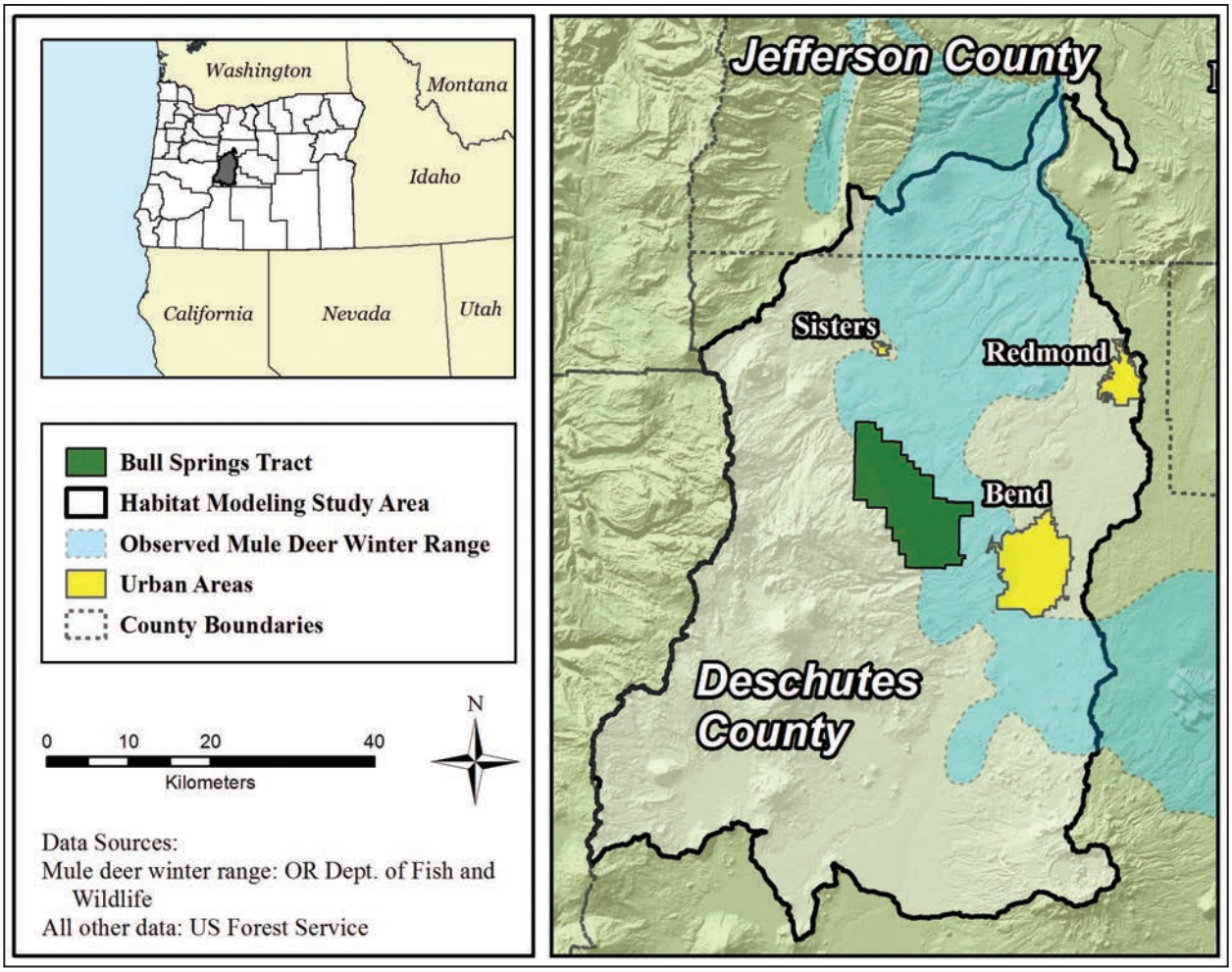


Figure 1—The study area and its ownership context. The solid black line shows the boundary of the study area used in the STM; the dashed line shows the observed extent of the mule deer winter range. The Bull Springs tract is shown in the center of the modeling area.

elevation from 590 m to 3150 m (fig. 1). High elevation forests are dominated by spruces (*Picea* spp.) and firs (*Abies* spp.) whereas lower regions of the mountain slopes consist primarily of ponderosa pine (*Pinus ponderosa* C. Lawson) and lodgepole pine (*Pinus contorta* Douglas ex Louden). East of the foothills, the landscape is dominated by juniper (*Juniperus occidentalis* Hook.) and sagebrush (*Artemisia* spp.) plant communities. With precipitation ranging from 25.4 cm to 239.7 cm annually with large seasonal variation (Thorson et al. 2003) and soils with low water retention capacity, available soil water is often the limiting factor for plant growth (Simpson 2007). The ownership landscape is a mixture of federal and private ownerships. The USDA Forest Service (FS) and the U.S. Bureau of Land Management (BLM) administer much of the land in the study area, with 29.4 percent in private industrial and nonindustrial ownership. The mapped mule deer winter range[1] consists of 133 100 ha (30.7 percent) of the total study area. The Bull Springs tract covers 3.0 percent of the total study area and 9.8 percent of the observed mule deer winter range (fig. 1).

## Vegetation Dynamics

Spatially-explicit state and transition models (STMs) were used to simulate residential development and vegetation

[1] Personal communication with Glenn Ardt, Biologist, Oregon Department of Fish and Wildlife (ODFW).

dynamics. STMs have been applied to various ecological systems in which multiple stable states are possible (Westoby et al. 1989, Hemstrom et al. 2007, Vavra et al. 2007, Petersen et al. 2009). Vegetation was simulated for 60 years (year 2000 to 2060) using a suite of STMs representing individual development stages, or *states*, within potential vegetation types (PVTs) for climax communities. PVTs are vegetation assemblages that take into account physical settings and species communities (Hall 1998). States are defined by a *cover type*, typically the dominant species or vegetation type, and a *structural stage* that describes the vegetation size class, canopy cover, and vertical layering. Transitions, such as wildfire, management activities, and development, defined possible pathways between states. Base transition rates control the speed at which vegetation assemblages change from one state to another given a particular transition, however, rates may be modified using multipliers to increase or decrease base rates to represent, for example, different intensities of the same management type among ownership types. The order, occurrence, and location of transitions are determined stochastically for each annual time step. Other transition-inducing mechanisms modeled were defoliators (e.g., western pine beetle) and parasites (e.g., mistletoe). We used the Tool for Exploratory Landscape Scenario Analyses (Version 3.06) to model landscape dynamics and the Vegetation Dynamics Development Tool (Version 6.0.25) to design, build, and calibrate the STMs (ESSA 2007, ESSA 2008). Models were designed by local ecologists or derived from previous work (Hemstrom et al. 2007) and ongoing planning activities. Fourteen potential vegetation types were modeled, three shrub steppe types and 11 forest types. PVTs modeled were: mountain shrub/meadow, Wyoming big sage/juniper, mountain big sage/juniper, ponderosa pine dry (pumice soils), ponderosa pine dry (hot dry; residual soils), ponderosa pine/lodgepole pine, lodgepole pine dry (pumice soils), lodgepole pine wet, mixed conifer dry (pumice soils), mixed conifer dry (other soils), mixed conifer moist, Shasta red fir (dry), upper montane (cold), and subalpine parklands.

Initial conditions were created by intersecting spatial data sets that represented vegetation stand boundaries, PVT boundaries, ownership/allocation boundaries, and development zones. The vegetation stand boundaries were derived using segmentation in eCognition 5 (Baatz et al. 2004) over a multi-image stack, where image segments represented homogeneous patches of vegetation, such as stands. The multi-image stack was composed of individual Landsat 5 and NAIP bands and image texture. Vegetation cover and structure attributes assigned to vegetation stands came from a vegetation layer developed using a gradient nearest neighbor (GNN) analysis technique where imputation is used to assign plot-level vegetation data to pixels (Ohmann and Gregory 2002; LEMMA 2011). PVT boundaries were determined from plant association maps developed by the USDA FS. Plant associations were grouped to form PVTs and represented the entire geographic extent over which PVTs could occur rather than the mean. Ownership-allocation boundaries were developed by the Oregon Department of Forestry. Residential development was restricted to development zones derived from projections of future building densities based on development rates from the 1970s to 2000 and environmental factors such as slope, elevation, distance to roads, zoning, and distance to other buildings (see Kline 2005 and Kline et al. 2010). For this study, we assumed PVT distribution, ownership-allocation, and development zone boundaries remained static over time. We used the TELSA Voronoi tessellation algorithm (Kurz et al. 2000) to subdivide our landscape into simulation polygons with an average size of 1 ha.

## Development and Land Management

To model development, we determined the rates of change within five development density classes based on initial and ending development patterns determined by Kline et al. (2010), and applied these rates to our models as the annual probabilities for development. In other words, for each development density class, we defined development transitions and a probability for a development event to occur; the probabilities were determined from work by Kline et al. (2010). Development was assumed to occur linearly so that a patch must be developed at the lowest density class before being developed to a higher density and must pass through all density classes, in ascending order, before reaching the

highest density class possible given the initial development zone. Exact locations of development were assigned stochastically within the development zones. Development density classes were defined, going from lowest to highest density, as more than 194 ha per structure (NotDev), 97 to 194 ha per structure (D2), 32 to 97 ha per structure (D3), 4 to 32 ha per structure (D4) and less than 4 ha per structure (D5). Most of the landscape began in the lowest development class (NotDev). When housing densities exceeded one house per 97 ha, active forest management ceased, as sustainable forest management has been shown to decrease with parcelization (Germain et al. 2007). Development analyses are presented for the whole study area below, but model outcomes were examined at all three spatial extents.

Modeled management activities varied by ownership (e.g., private industrial), federal land allocations (e.g., wilderness), and vegetation types and were applied to the landscape based on ownership class and development density. Federal lands were managed primarily for restoration and to reduce fuel loads, with lower rates of treatments than private lands. Management activities modeled for federal lands included pre-commercial thinning, commercial thinning, prescribed fire, and other harvest types; the frequencies at which management occurred varied by ownership, vegetation type, cover type, and structure. Overall, private land without residential development was assumed to be managed using similar methods to federal lands, but at higher intensities. For example, salvage activities were assumed to be 50 times more likely to occur on private ownership types than on USDA Forest Service land. Under the working forest scenario, the Bull Springs tract was managed for restoration of open ponderosa pine stands typical of the region under historical conditions prior to Euro-American settlement (Youngblood et al. 2004).

## Quantifying Changes in Mule Deer Habitat

To represent the landscape in terms of mule deer habitat, the states in the initial and final vegetation maps were classified into habitat categories using a wildlife-habitat relationship (WHR) model (Johnson and O'Neil 2001). WHR models map the habitat of a particular wildlife species to the landscape based on vegetation type and structure, and have been compiled for many species in Oregon (Johnson and O'Neil 2001). With these models, the spatial structure of the landscape is linked to ecologically relevant life history traits such as home range size and dispersal distances, allowing species-specific responses to changes in landscape structure over time to be inferred (Johnson and O'Neil 2001).

The wildlife-habitat model for mule deer for the central Oregon region was developed in consultation between the USDA FS and wildlife biologists from ODFW. Vegetation type and structure were classified in three dimensions: forage quality (poor/none, low, moderate, high), thermal cover (yes, no), and hiding cover (yes, no). In addition, these dimensions were combined into a single rating model called the habitat suitability index (HSI). Classifications drew mainly on natural history information and sources reviewed above, but some modifications were made based on the expert knowledge of area biologists. We limited our pattern analysis of habitat area and patches to the observed mule deer winter range. All outputs generated by the vegetation dynamics model were converted to raster data sets with a 30-m cell size to coincide with the nominal grain size of the GNN vegetation data and analyzed in FRAGSTATS 3.3 (McGarigal et al. 2002). Habitat patches were defined in FRAGSTATS as adjacent cells sharing a cell boundary.

## Forage Quality

Mule deer forage quality is related to a combination of forest structure (corresponding to structural stages in the STMs) and dominant tree species (cover types in the STMs). High-quality forage patches were typically open, with grass/forb, closed shrub, and seedling/sapling conditions in areas that supported most conifer tree species or were older stands of very large trees with multistory, open canopies. The exceptions were mesic stands of high elevation mixed conifer species, ponderosa pine, white fir (*Abies concolor* (Gord. and Glend.) Lindl. ex Hildebr.), and lodgepole pine, which were considered to be of low-quality due to snow accumulation in the winter. Moderate-quality forage consisted of younger stands of mostly conifer species and open canopies, with the exceptions listed above. Development densities less than one structure per 97 ha were considered moderate-quality forage provided the cover type was a tree

**Table 1—Vegetation conditions defining each forage quality rating level and the habitat suitability index scores obtained when these levels are combined with hiding and thermal cover classifications (see text for definitions of hiding and thermal cover conditions). Certain combinations were not possible in these models due to the structural requirements for habitat to be called hiding or thermal cover, and are indicated with '--' in the table**

| | | Habitat suitability index scores when combined with hiding and thermal cover | | | |
|---|---|---|---|---|---|
| **Forage quality** | **Description of forage quality ratings** | **Hiding and thermal cover** | **Hiding cover** | **Thermal cover** | **Neither hiding nor thermal cover** |
| High | *Structure:* open, grass/forb, closed shrub, seedling/sapling or older very large trees with multistory, open canopies<br>*Composition:* all species except mesic, high elevation stands of mixed conifer species (ponderosa pine, white fir, and lodgepole pine), where snow accumulates in winter | -- | 6 | -- | 3 |
| Moderate | *Structure:* younger and open canopies<br>*Composition:* mostly conifer species | -- | 5 | -- | 2 |
| Low | *Structure:* closed canopy stands<br>*Composition:* any species when canopy is closed, and any occurrence of mesic, high elevation stands of mixed conifer species (e.g., ponderosa pine, white fir, and lodgepole pine) | 7 | -- | 4 | -- |
| Poor/none | *Structure and composition*: sagebrush, juniper, grassland or development densities higher than 97 ha per structure | 6 | 3 | 3 | 0 |

species and the site was not mesic and high in elevation. All closed-canopy stands were considered to be low-quality forage. Forage quality in states with development higher than one structure per 97 ha or in sagebrush, juniper, or grassland trajectories were classified as poor/none, following the concept of disturbance zones (Theobald et al. 1997). These classifications are summarized in table 1.

## Habitat Suitability Index

To capture change in modeled states that provide multiple habitat benefits or functions, we defined a habitat suitability index. The index combined forage quality habitat classes with thermal and hiding cover classes to provide a means to consider all habitat types together. Hiding cover was based on vegetation structure. Seedling/sapling stands, denser stands of poles, and all stands with a multilayered canopy and trees greater than 25 cm in diameter provided hiding cover. Closed shrub conditions in aspen types and dry ponderosa types were also considered hiding cover. All other modeled states, including all development states, provided no hiding cover. Canopy closure was the primary determinant of thermal cover, as it provides both shade in the summer and reduced wind exposure in the winter. All areas with coniferous tree species larger than 25 cm in diameter and canopy closure exceeding 40 percent provided adequate thermal cover. Numerical equivalents were given to each level of habitat classification: forage quality received scores of 0 through 3 corresponding to each level defined above from poor/none to high, while hiding and thermal cover were each given a score of 0 when absent and 3 when present. These numerical equivalents were then summed to generate the HSI (table 1). This formulation of habitat suitability distinguished between vegetation states that

**Table 2—The landscape metrics used, a description of what they measure, and their units**

| Level | Metric | Definition | Ecological interpretation | Units |
|---|---|---|---|---|
| Class | Mean patch area | Average area in hectares of all patches in the same class | Home range size needed by an individual for mating, breeding, and foraging | Hectares |
| | Mean patch radius of gyration | Average distance between each cell in a patch and the patch centroid, averaged over all patches | Connectivity and corridor characteristics of the landscape | Meters |
| | Mean Euclidean nearest neighbor | Average distance from the edge distance of a patch to the edge of the nearest neighbor of the same type for all patches in that class | Overall isolation of patches in each class | Meters |
| | Number of patches | Total number of patches in each class | Fragmentation or consolidation of a class | Count |
| Landscape | Shannon's diversity index | richness and evenness of the distribution of patch types | Sensitive measure of rare patch types | None |
| | Largest patch index | percentage of the landscape occupied by the single largest patch | Homogeneity and the dominance of patches within the | Percent |
| | Contagion | The likelihood that adjacent cells will be of the same class type | General aggregation of the patches in the landscape and connectedness | Percent |

provided just one habitat function (single-function classes) from those that provided 2 or 3 habitat functions (multi-functional classes). A rating of 2, for example, corresponded to only moderate-quality forage being present, whereas a rating of 7 indicated that both hiding and thermal cover were present in combination with low forage quality. HSI values of 2 or 3 only reflected forage conditions (except for very large multistory stands of open-canopy white fir, which provided only hiding cover), while HSI values of 4, 5, 6, and 7 represented multifunctional classes. Values of 0 signified no habitat provision and values of 8 and 9 were not possible, as all states that are moderate- or high-quality forage are inherently poor thermal and hiding cover; similarly, HSI = 1 does not occur because low-quality forage conditions provide hiding or thermal cover.

## Landscape Pattern Analysis

Landscape pattern analysis can quantify the effects of human activities on the spatial arrangement of the landscape, and many metrics have been developed for measuring these arrangements (McGarigal et al. 2002, Turner 2005, Turner et al. 2001). Various studies in the field of landscape ecology have used landscape pattern metrics to study the relationship between ownership, land use change, and ecological processes (McComb et al. 2007, Stanfield et al. 2002, Swenson and Franklin 2000). Landscape structure can be measured within habitat types (class level) or across all habitat types (landscape level), and depends on the spatial extent considered and the grain size of the smallest unit of area. Metrics were calculated at the class and landscape levels, where habitat quality and HSI values represented classes, using FRAGSTATS 3.3 and were chosen to minimize redundancy while capturing the broadest set of landscape characteristics possible. Within the observed mule deer winter range, patches were defined as adjacent pixels of the same forage and HSI classes. At the class level, we calculated mean patch size, total class area, mean Euclidean nearest neighbor, mean radius of gyration and the number of patches (table 2). At the landscape level, we calculated the Shannon's Diversity Index (SHDI), contagion, and LPI (table 2). These metrics were calculated for each habitat model component for the initial and final conditions under

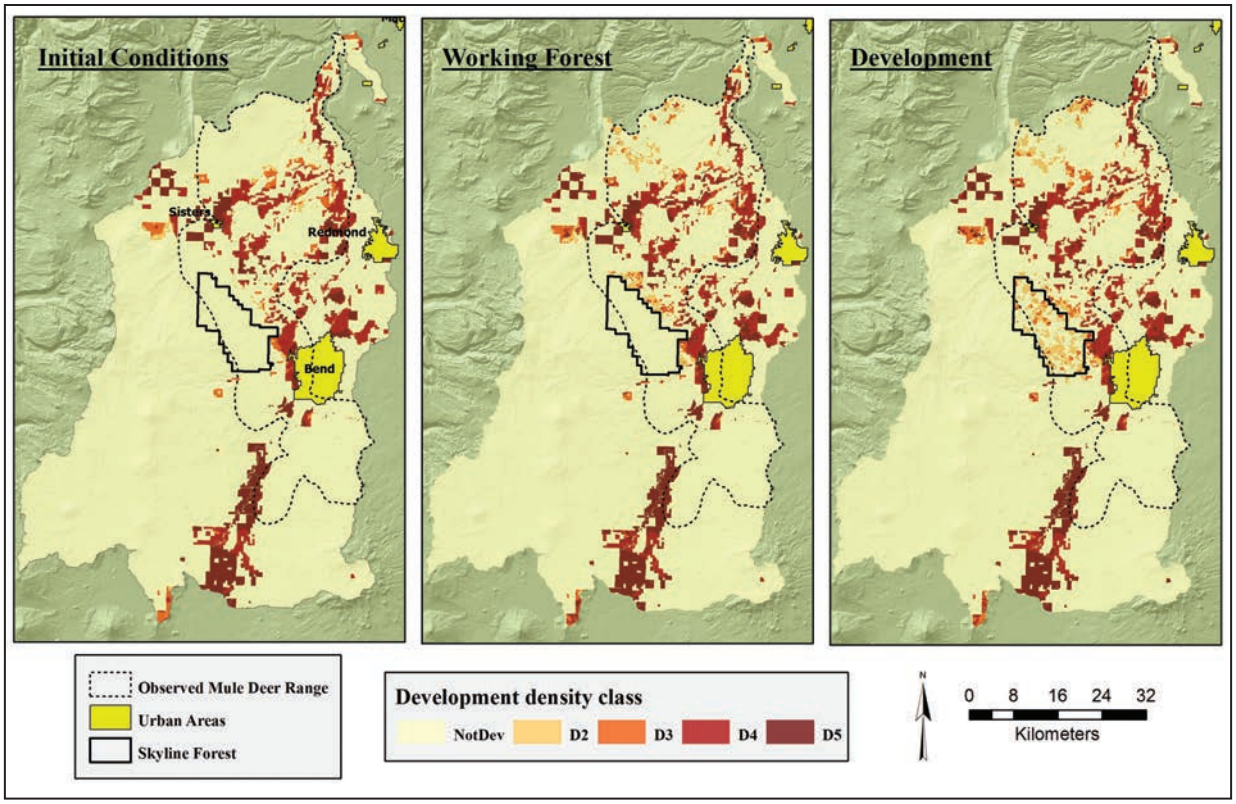


Figure 2—Spatial distribution of development classes for initial conditions and working forest (WF) and development (DEV) scenarios at 60 years. NotDev = greater than 194 ha/structure; D2 = 97 to 194 ha/structure; D3 = 32 to 97 ha/structure; D4 = 4 to 32 ha/structure; D5 = less than 4 ha/structure.

each scenario within the mule deer winter range. Percentage changes and absolute changes from present to future were calculated. In addition, change in developed land and vegetation states were calculated to provide context for interpreting the results of the pattern analysis. Our approach did not include a weight for neighborhood or proximity to high quality habitat. For example, moderate quality patches located beside high quality patches did not receive a higher habitat value than those far from high quality patches.

## Results

The two scenarios generated different spatial patterns of development (fig. 2) and affected differences in vegetation structure and cover combinations after 60 years of simulation. Overall, the DEV scenario showed greater amounts of land conversion from lower to higher densities of development than the WF scenario. Urban densities increased in both scenarios (1.0 percent in WF and 0.7 percent in DEV), and undeveloped land area decreased by 1.2 percent and 2.5 percent in the WF and DEV scenarios respectively. In the observed mule deer winter range 5.8 percent of the land moved from undeveloped into the lowest developed density (D2, 97 to 142 ha per structure) in DEV as compared to only 3.0 percent in WF (fig. 3). In contrast, 42 percent of the Bull Springs tract converted from undeveloped to developed land (D2) under the DEV scenario, with 3670 ha of undeveloped land converted to D2 and 1890 ha converted to development densities of 32 to 97 ha per structure (D3) at the end of 60 years. On the Bull Springs tract alone, nearly twice as much land entered the lowest density class in the DEV scenario as compared to the WF scenario.

### Forage Quality

Mapping the initial and future forage quality conditions illustrated differences in the locations and nature of land use conversions between scenarios (fig. 4A) and the resulting distributions of forage quality levels (fig. 4B).

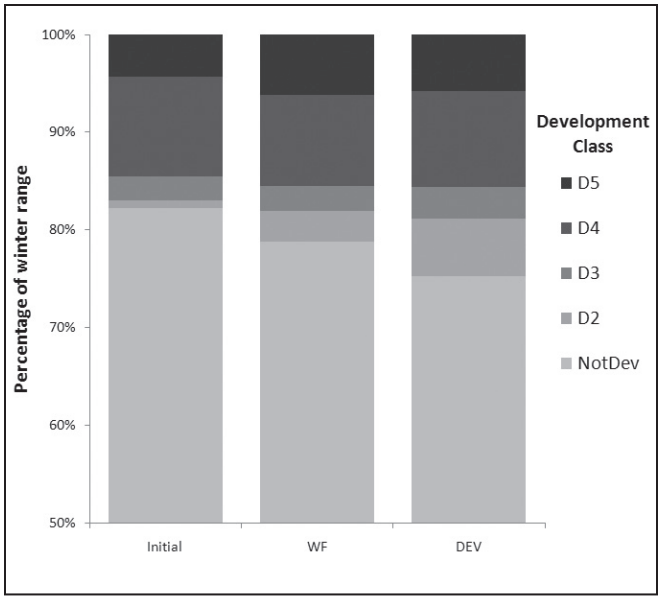


Figure 3—Proportions of development in five classes within mule deer winter range for the development (DEV) scenario and the working forest (WF) scenario. Winter range comprises 9800 ha of the Bull Springs tract and 133 100 ha of the entire study area. NotDev = greater than 194 ha/structure; D2 = 97 to 194 ha/structure; D3 = 32 to 97 ha/structure; D4 = 4 to 32 ha/structure; D5 = less than 4 ha/structure. As NotDev comprises most of the winter range, the vertical axis begins at 50 percent to better illustrate the differences in the other development classes.

Landscape level metrics did not change dramatically from initial conditions for either scenario or show substantial differences between scenarios (table 3). Overall, low-quality forage experienced the greatest negative change from the initial conditions for both scenarios in all landscape metrics (fig. 5). The largest total increases in patch area, abundance, and extensiveness occurred in high- and moderate-quality classes, with slightly larger increases in total area, largest patch index, patch area, and extensiveness in the WF scenario. Nonforage patches were less numerous, less extensive, and closer together under the WF scenario compared to DEV; under the DEV scenario nonforage patches became smaller and constituted a larger percentage of the landscape. Moderate-quality forage patches became larger and more extensive under WF compared to DEV, but they also were more isolated, according to nearest neighbor distances. High-quality forage patches became smaller on average in both scenarios. The decrease in mean high-quality patch area was less pronounced in the WF scenario, but fragmentation through the creation of more patches was more pronounced. High-quality patches also became less isolated and more extensive, with somewhat greater gains under the WF scenario compared to DEV. The largest differences between the scenarios were an increase in nearest neighbor distances of low-quality forage patches and a reduction in the patch abundance under DEV compared to a moderate increase in patch number under WF.

## Habitat Suitability Index

Similar to outcomes for forage quality, the spatial mapping of the habitat suitability index showed subtle differences between the initial conditions and the two scenarios in terms of the location and nature of habitat quality changes (fig. 6A), as well as the overall distribution of habitat suitability levels in the two scenarios (fig. 6B). Landscape-level metrics were fairly similar between scenarios (table 3), but at the class level, differences were more apparent (fig. 7). For single-function habitat types (HSI = 2 or 3), the mean patch size increased in both scenarios although the increase was slightly smaller in DEV. Other metrics showed little difference between the two scenarios. In contrast, multifunctional habitat types (HSI = 4–7) displayed substantial differences between scenarios. For all multifunctional habitat types, mean radius of gyration was slightly lower under DEV than under WF. The patch isolation (mean

**Table 3—Landscape level metrics calculated for the forage and habitat suitability index landscapes**

| | Forage | | | HSI | | |
|---|---|---|---|---|---|---|
| **Metric** | **Initial** | **Development** | **Working Forest** | **Initial** | **Development** | **Working Forest** |
| Largest patch index | 19.60 | 19.68 | 19.69 | 19.61 | 19.68 | 19.69 |
| Contagion | 53.22 | 61.95 | 63.33 | 52.22 | 63.33 | 61.95 |
| Shannon's Diversity Index | 1.57 | 1.32 | 1.27 | 1.57 | 1.32 | 1.27 |

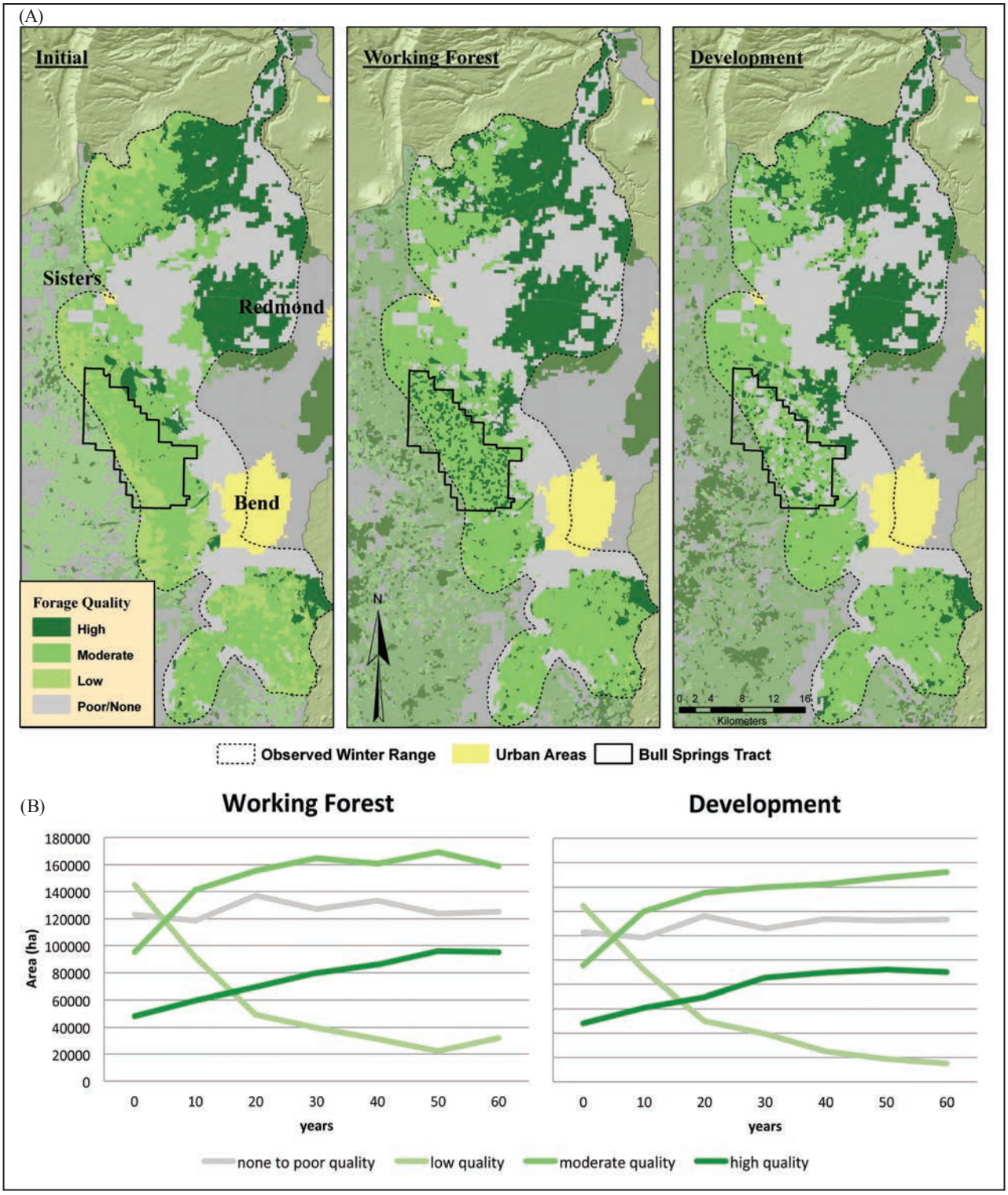


Figure 4—(A) Initial forage quality conditions (left) as well as the simulated forage quality conditions for the working forest scenario (middle) and the development scenario (right). The conditions within the mule deer winter range are highlighted and the location of the Bull Springs tract is shown in each map, as well as the major urban areas in the landscape. (B) Trajectories of the total amount of land in each forage quality class over the simulation.

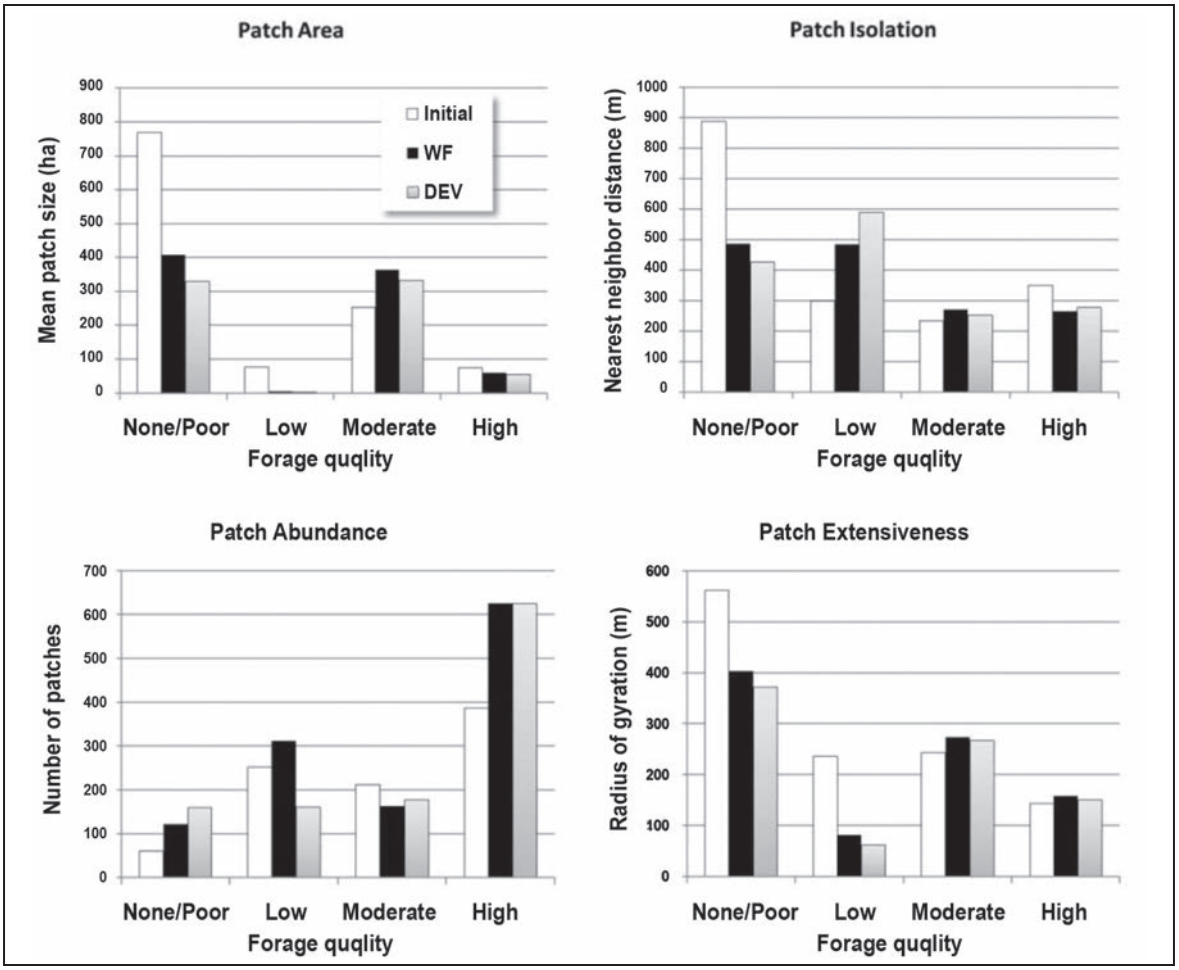


Figure 5—Differences in patch metrics between the initial conditions and modeled scenarios indicate changes in the spatial patterns of mule deer forage quality classes. The calculated values of the landscape metrics are given for the mean patch area (top left), the number of patches (bottom left), the mean Euclidean nearest neighbor (top right), and the mean radius of gyration (bottom right).

nearest neighbor distances) under DEV nearly doubled for HSI values of 4, 5 and 7. These habitat types provide low- or moderate-quality forage, in combination with either hiding or thermal cover. Our results suggest that vegetation conditions supplying more than one habitat function on the landscape would become more isolated, less extensive, and smaller in the future, with these effects amplified under DEV in most cases.

Interestingly, the outcomes for HSI values of 6 differed from those of the other multifunctional classes. Total class area remained within 10 percent of initial conditions, but both scenarios resulted in very large increases in the number of patches and little change in isolation and extensiveness. Coupled with the overall decrease in patch size, this indicates greater fragmentation of this class compared to the other multifunctional classes. There were also marked differences between scenarios in terms of nearest neighbor distance, with WF resulting in increased isolation compared to decreased isolation under DEV for this value.

## Discussion

Our spatial models demonstrated that rural residential development and forest management have the potential to alter the Bull Springs tract and the surrounding landscape regardless of the development fate of the Bull Springs

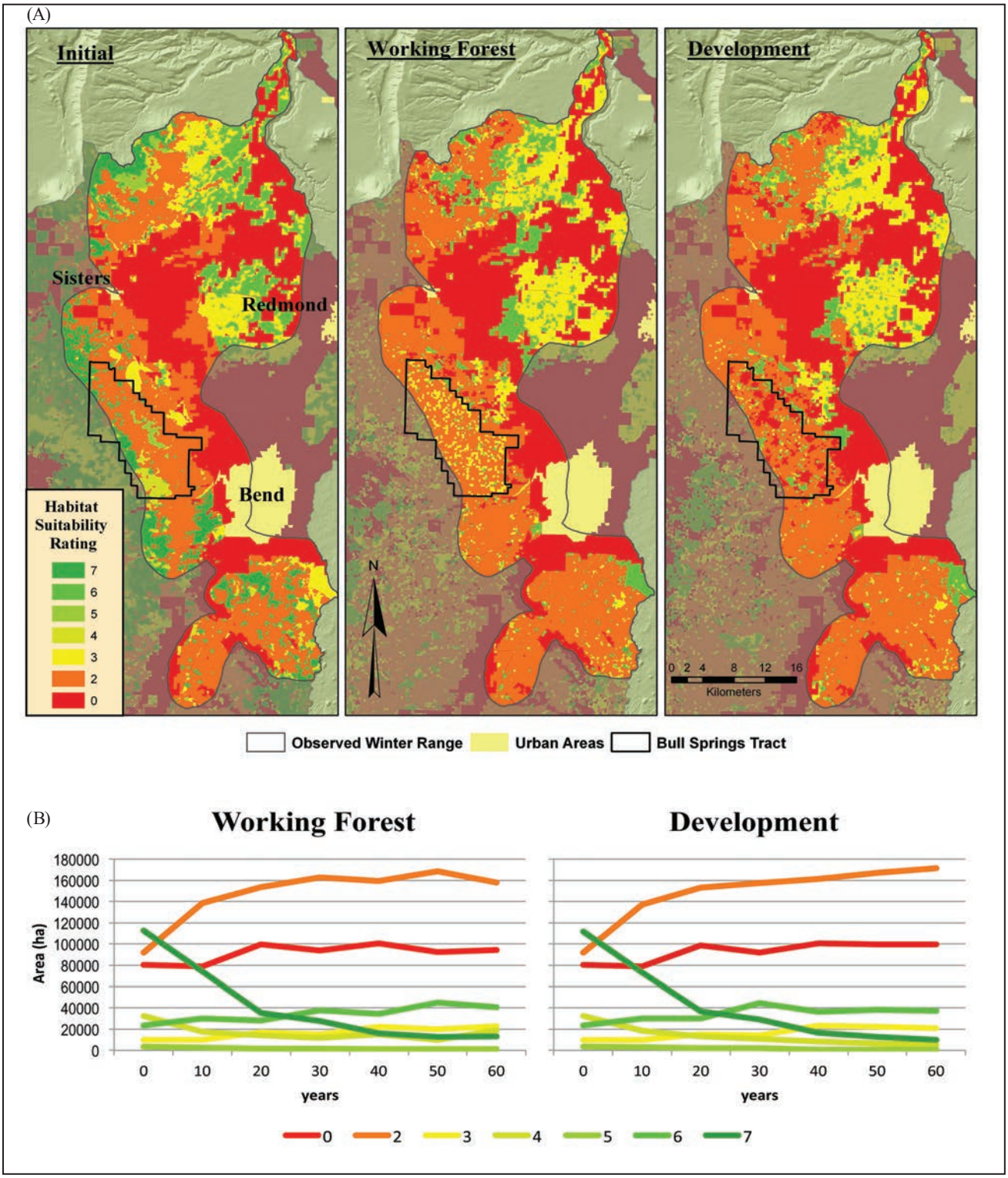


Figure 6—Maps of the habitat suitability conditions initially (left) and after 60 years under the working forest (WF, center) and development (DEV) scenarios (right). The conditions within the mule deer winter range are highlighted and the location of the Bull Springs tract is shown in each map, as well as the major urban areas in the landscape.

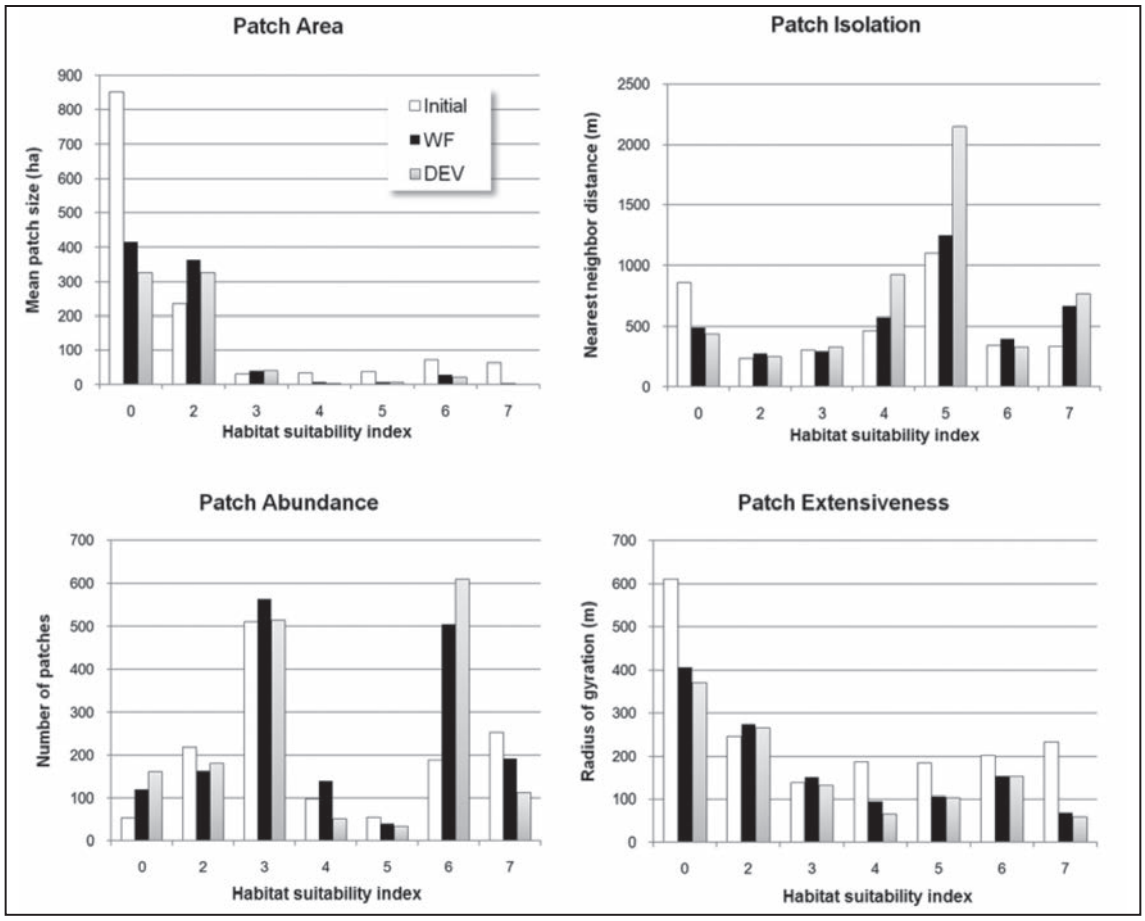


Figure 7—Class-level measures of the spatial pattern of the mule deer habitat suitability index. HSI = 0 has no habitat functions, HSI = 2 or 3 represent single-function habitat classes, HSI > 3 represent multifunction habitat classes. The calculated values of the landscape metrics are given for the mean patch area (top left), the number of patches (bottom left), the mean Euclidean nearest neighbor (top right), and the mean radius of gyration (bottom right).

tract. Results from both modeled scenarios suggest that landscape-wide changes projected to occur in the next several decades have potential to affect mule deer habitat, largely because development is expected to continue, but the differences may be subtle at the landscape scale. However, due to the Bull Springs' landscape position, large proportion of mule deer winter range, and large size relative to other private single-ownership tracts, preventing development on Bull Springs may offset landscape-level habitat degradation that could result from nearby rural development. Moreover, the modeled interaction of management and changes in forest structure and composition due to forest maturation suggests improved forage and multifunctional habitat conditions for mule deer over the next several decades if the land is managed as a working forest. This expectation was exemplified by increased patch abundance and area of high-quality forage patches in the working forest scenario when compared to the development scenario. Though more total area of high quality forage was added under the working forest scenario, both scenarios created the same number of high-quality forage habitat patches and resulted in decreased isolation of these patches relative to the initial conditions. This relationship suggests that mule deer may find more high-quality forage patches in the future and have to travel shorter distances between patches once they find the high-quality forage patches. The overall shift from multifunctional to single-function HSI ratings in both scenarios signals a possible stratification of the landscape

and could lead to higher energetic costs if mule deer must travel farther distances between patches of different habitat types to obtain all needed resources. In contrast, if the single-function patches are closely interspersed and all habitat requirements are met within short travel distances, energetic costs may decline.

Mule deer is an iconic species in eastern Oregon, and maintenance of the herd for aesthetic and hunting purposes is a stated goal; our results suggest habitat supply and arrangement will be impacted by Bull Springs tract management decisions. Based on the limits of the observed winter range, there is evidence from the simulation that lower rates of development on the tract could enhance mule deer habitat conditions in the future and offset or ameliorate the impacts of development elsewhere in the region. Promotion of mule deer persistence in their winter range will likely require attention to the location of residential development outside the Bull Springs tract. Attention must also be paid to the potential for isolating high-quality forage and multifunctional habitat conditions and converting these to less suitable types because restricting development within the Bull Springs may increase development pressures elsewhere in the landscape.

Conversion of forested land to rural residences is only one process affecting vegetation dynamics and landscape change. Decision makers are often expected to respond to landscape-scale processes, but may not have access to landscape-scale information to ensure the broader policy governing land use change balances wide-ranging land use conversion with gains made through more localized conservation actions. Landscape simulation modeling provides one approach for investigating the numerous effects future policy choices may have in a region by defining the ecological significance of a particular piece of land in a given landscape as well as the limits of conservation on a single tract when considering the larger suite of changes taking place elsewhere in the landscape. By coupling spatial vegetation dynamics modeling that incorporates both human- and disturbance-driven modifications of the landscape with landscape pattern analysis, we can inform the broader debates on planning for land use changes in the future. Moreover, landscape simulation modeling results such as these can be extended to provide supporting data to a decision support system designed to prioritize scenario outcomes. Another extension of the work presented here that could benefit decision support would be to relate the landscape metrics for forage quality and HSI classes to mule deer carrying capacity. Incorporating the role that proximity to various habitat types plays in carrying capacity on the landscape would be another powerful means to understand the complex interactions of human activity and wildlife requirements. Landscape models and analyses such as the ones presented here provide a means for increasing knowledge of dynamic landscapes that are important for many management objectives.

## Limitations

A major limitation of this study is that the allocation of development and simulation of final vegetation states were done using a single simulation, rather than multiple simulations, which limits the conclusions that can be drawn from the predictions. While we performed limited uncertainty and sensitivity analyses with the nonspatial and spatial vegetation dynamics models, we are limited in our ability to distinguish between the impacts of development and artifacts arising from the spatial simulation method; our results must be tempered by this uncertainty. In addition, the pattern metrics used in this study represent a set of assumptions about what makes “good” mule deer habitat, both in defining patches and in deciding what spatial aspects are more important than others. We also used a limited set of independent and complementary spatial pattern metrics for simplicity, but these are dependent on the scale and grain of measurement (Li and Wu 2004) and can often be highly correlated with cumulative changes in habitat area (Fahrig 2003). While these metrics were chosen to relate to life history requirements of mule deer, they are just one set of measurements of landscape structure in the region, and could not capture all aspects of the spatial structure of habitat, such as inter-class spatial relationships. Other limitations arise from not including other forms of human-wildlife interactions such as subsidized food supplies, suppression of predators, and climate change in the models.

With regards to the input data, the base GNN vegetation data layer was generated from a statistical model. When combined with the spatial vegetation dynamics model there is the potential for nontrivial classification errors (Ohmann and Gregory 2002). As such, spatial results from these products should be considered at the regional level, not the site level. These tools could not predict landowner behavior and changes in management approaches or ownership boundaries, and are limited to our historical understanding of and assumptions about these phenomena. Human and ecological processes and interactions were simplified in our modeling, but while the results must be viewed as only a small subset of many possible outcomes, they provide a point of departure for understanding the different outcomes that could be experienced under alternative policy and management regimes.

## Conclusion

Interactions between land use change, shifts in management priorities, and natural disturbance processes can drive landscape dynamics and resulting patterns. The habitat of mule deer and other wildlife are determined by these dynamics and patterns. Using wildlife-habitat relationships, spatial analysis, and spatial vegetation dynamics modeling, it is possible to provide concise measurements of landscape change and facilitate ecological interpretations of differences between alternative land use policy futures. Our analysis of a large private tract in central Oregon and surrounding landscape showed that alternative management regimes might affect mule deer habitat, but more information is needed to understand the effect on carrying capacity. Notably, working forest management on the Bull Springs could result in somewhat better mule deer habitat outcomes, particularly with respect to higher quality forage and multifunctional habitat types. That said, simulated habitat was degraded to some extent under both scenarios, but marginal gains due to conservation over dispersed rural development could arise through working forest management, according to our simulation outcomes. Managing this tract as working forest could reduce the negative ecological effects of residential development and other land use changes within the region. Future management decision-making should pay continued attention to where residential development is expected to occur outside the Bull Springs tract and consider the isolating effect development might have on important habitat types. Managing the use and conditions of a single portion of the landscape can only do so much when that portion is embedded in a larger context of change, but this research provides critical insight into how land use policies on a specific tract can influence the ecological trajectory of change in the broader landscape.

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