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Title: <u>Modeling Sage-Grouse Habitat Using a State-and-Transition Model</u> Abstract approved:

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Habitat for wildlife species that depend on sagebrush ecosystems is of great management concern. Evaluating how management activities and climate change may affect the abundance of moderate and high-quality habitat necessitates the development of models that examine vegetation dynamics, but modeling tools for rangeland systems are limited. I developed state-and-transition models using a combination of scientific literature and data for climate, soils, and wildfire to examine how different types of natural events, management activities, changing climate, and potential future vegetation dynamics may interact and affect the abundance of habitat for the greater sage-grouse (*Centrocercus urophasianus*). Specific periods examined include the era prior to 1850, the current era, and late in the 21st century in southeastern Oregon. A primary purpose of this study was to evaluate the use of climate data to define most event probabilities and, subsequently, the relative mix of ecological states, community phases, and sage-grouse habitat with an eye towards a modeling approach that was objective, repeatable, and transferrable to other locations.

Contrary to expectations, model results of the conditions prior to 1850 indicated fire may not have been the most important disturbance factor influencing sage-grouse habitat abundance, merely the most visible. Other, more subtle disturbances that thinned sagebrush density, such as drought, herbivory, and weather-related mortality, may have been equally or more important in shaping sage-grouse habitat. Sage-grouse breeding habitat may have been slightly more abundant than levels currently recommended by sage-grouse biologists, broodrearing habitat may have been as or more abundant, but wintering habitat may have been less abundant.

Under the current conditions, livestock grazing during severe drought, postfire seeding success, juniper expansion probabilities, and the frequency of vegetation treatments were the most important determinants of sage-grouse habitat abundance. The current vegetation trajectory would lead to considerably less nesting, brood-rearing, and wintering habitat than sage-grouse biologists recommend. Model results suggested reducing or eliminating livestock grazing during severe drought, increasing postfire seeding success, and treating at least 10% of the so-called expansion juniper each year was necessary to maintain higher levels of sage-grouse habitat, although nesting and brood-rearing habitat remained in short supply.

I examined three potential future climates based on long-term climate trends in southeastern Oregon and modeled climate and ecosystem projections for the Pacific Northwest generally. The first scenario produced warmer and drier conditions than present, the second scenario warmer and wetter conditions in winter, and the third scenario warmer and wetter conditions in summer. The implications for sage-grouse habitat abundance were very different between these three scenarios, but all would likely result in the loss or near complete loss of cooler, moister sagebrush communities important for nesting and brood-rearing. Salt desert shrub and warmer, drier sagebrush communities could expand under the first scenario but would have a high risk of displacement by cheatgrass. Juniper woodlands could increase in density and salt desert shrub may expand slightly under the second scenario. The remaining sagebrush communities would remain at high risk of displacement by cheatgrass. Pinyonjuniper woodland could largely displace sagebrush in the third scenario. Sage-grouse habitat quality likely would decline in all three scenarios and the abundance decrease significantly in the second and third scenario. Modeling Sage-Grouse Habitat Using a State-and-Transition Model

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

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Modeling Sage-Grouse Habitat Using a State-and-

Transition Model

Chapter 1 – General Introduction

Prior to the 1980s, the main goal of rangeland management was to reduce sagebrush and increase forage for domestic livestock (Pechanec et al. 1944, Frischknecht and Bleak 1957, Johnson 1958, Johnson and Payne 1968, Johnson 1969, Frischknecht and Baker 1972, Harniss and Murray 1973, Bartolome and Heady 1978, Britton et al. 1981). During the 1980s management emphasis on public lands and some private lands shifted towards restoring healthy, functioning rangeland communities to provide for a variety of social and economic values such as water, recreational opportunities, and wildlife habitat in addition to livestock forage (Bunting et al. 2002, Hemstrom et al. 2002, Bunting et al. 2003). Conditions prior to 1850 (historical) are widely believed to have been significantly different from present conditions (Bunting et al. 2002) and perceived to have better met today's societal desires, often resulting in a call to restore those conditions. However, changes to sagebrush ecosystems over the last 150 years threaten their ability to provide many of these values in the future (Miller and Eddleman 2000, Connelly et al. 2004). Of particular concern are threats to wildlife habitat and especially the habitat of sagebrush obligate species, such as greater sage-grouse (*Centrocercus urophasianus*).

Under the current management paradigm, federal land managers are required to use the "best available science" in the formulation of land management and project plans. Part of this paradigm includes the use of reference conditions by which to judge current conditions, understand how different management strategies and practices may alter vegetation trajectories, and, most recently, consider how changing climate might alter management strategies. The most commonly selected reference conditions are those believed to have been present prior to 1850; however, little data is available on what those conditions might have been. Rangeland managers lack vegetation simulation tools to examine how different management strategies and changing climate may alter vegetation trajectories and the associated social and economic values. Managers desire a single tool that is based on empirical data, repeatable, applicable to different geographic areas, and simple to use. Ideally, this tool should allow managers to evaluate past, present, and possible future conditions and different management scenarios, and work in a data-poor environment.

Several modeling tools exist that potentially could meet the needs of rangeland managers, such as dynamic vegetation models, vegetation simulators, and state-and-transition modeling frameworks. Each type of model has advantages and disadvantages, and varying ability to meet analysis needs in a data-poor environment. Dynamic global vegetation models combine a biogeographical vegetation model with a biogeochemical model to predict vegetation distribution based on climate, soils, and various ecological processes; such as photosynthesis, respiration, and soil hydrology (Haxeltine and Prentice 1996, Bachelet et al. 2001a, Sitch et al. 2003, Gerten et al. 2004, Woodward and Lomas 2004, Morales et al. 2005). These models are often linked to outputs of global or regional circulation models, allowing examination of vegetation-climate feedbacks and predictions of past and future vegetation (Bachelet et al. 2001a, Bachelet et al. 2001b, Woodward and Lomas 2004, Bachelet et al. 2008, Salzmann et al. 2009). Managers need not include preconceived assumptions about what vegetation type occurs where (Neilson 1995, Woodward and Lomas 2004). Although dynamic global vegetation models could be used at the individual species scale, the necessary information for such fine resolution analysis is lacking (Woodward and Lomas 2004). To date, dynamic global vegetation models describe vegetation only in broad categories of limited use to land

managers, such as temperate deciduous forest and C₃ grassland (Neilson 1995, Bachelet et al. 2001a). Additionally, these models rarely include disturbances other than fire and do not include land management activities, such as thinning or postfire seeding (Neilson 1995, Bachelet et al. 2001a, Sitch et al. 2003, Gerten et al. 2004). Lastly, dynamic global vegetation models require a higher level of computer skills and resources than are typically available to land managers, and the primary outputs, such as leaf area index and net primary productivity are not used to develop vegetation management plans. These outputs would need further processing to produce the type of data that managers typically use, such as estimates of livestock forage production.

The Forest Vegetation Simulator is a mechanistic model that predicts individual tree growth and yield (Crookston and Dixon 2005). The basic unit in this type of model is the stand, but the Forest Vegetation Simulator has been used to assess changes in vegetation structure at larger landscapes. The system includes several natural disturbances, as well as silvicultural and fuels management activities. Regional variants address differences in tree growth rates and species compositions. Recently, the Forest Vegetation Simulator has been linked to a biogeochemical model that allows managers to examine interactions between management activities and ecosystem processes, such as photosynthesis, respiration and carbon pools (Dixon 2010). Although most commonly used to predict future stands, the Forest Vegetation Simulator can be used to assess past conditions (Fornwalt et al. 2002). However, the Forest Vegetation Simulator includes only commercially important tree species and no equivalent tool is available for rangeland vegetation. Vegetation simulators require inventory data to initialize a model run, but rangeland vegetation typically is not inventoried due to the expense and the high interannual variability of herbaceous vegetation. In the absence of suitable mechanistic models, probabilistic state-and-transition modeling frameworks such as the Vegetation Dynamics Development Tool (VDDT) (ESSA Technologies Ltd. 2007) appear to provide the next best approach (Barrett 2001, Merzenich and Frid 2005). State-and-transition modeling frameworks operate at a variety of spatial and temporal scales, can incorporate management actions and natural disturbances, and fit directly with current theories on vegetation succession in rangelands ((Westoby et al. 1989, Stringham et al. 2003, Briske et al. 2006) but see also (Bestelmeyer et al. 2004)). These frameworks can operate in data-poor environments, require little training, and can draw on a mix of empirical data and expert opinion-based information on ecosystem function. They allow managers to assess other properties important to land management, such as forage and habitat availability and fire risks (Merzenich and Frid 2005, Hemstrom et al. 2007, Vavra et al. 2007, Wales et al. 2007, Wondzell et al. 2007). Such models do not use a process-based approach to shift plant communities as the water balance changes, but climate variables can form the basis of event probabilities to predict plant community changes and transitions to alternative vegetation states. Users can develop historical reference conditions for the plant community or community group of interest, explore current conditions and evaluate potential future conditions under a variety of scenarios using state-and-transition models.

The LANDFIRE project used a state-and-transition modeling framework to develop models of reference conditions intended to describe historical plant communities and successional dynamics (Rollins and Frame 2006). That project was the first attempt to provide, among other products, complete nationwide maps and consistent models of the potential natural vegetation for use in identifying ecological risks and setting national and regional priorities for vegetation management on federal lands. The LANDFIRE models include only fire whereas other disturbances, such as insect outbreaks and herbivory, are important factors that may

interact with fire and shape wildlife habitat. Time constraints during initial LANDFIRE model development necessitated an approach based largely on expert opinion (LANDFIRE National Implementation 2007a, b, c) so that individual models are not repeatable; a different set of experts would likely result in a different model for the same situation.

This study is an attempt to develop an objective, repeatable, and transferable process for estimating disturbance probabilities for past, present and future conditions and predicting the abundance of different ecosystem services using the VDDT modeling framework. To evaluate the usefulness of this process and state-and-transition models, we tested it within the frame of sage-grouse habitat abundance due to the importance of this species in rangeland management for the federal agencies. The study used information pertinent to south-central Oregon as a basis. The approach taken included:

- Use of the published literature to determine what natural factors in addition to fire may have been important drivers of sagebrush dynamics and how these factors might have facilitated or caused shifts between community phases and states,
- Use of climate variables (temperature, precipitation, and snow) to the extent feasible to estimate probabilities of natural events and land management activities,
- Comparison of the results to information available about historical and current vegetation conditions and fire frequencies in eastern Oregon,
- Determination of the relative importance of both natural and human-related disturbances in sagebrush dynamics, and
- Exploration of different climate change scenarios and what impacts each scenario might have on vegetation, disturbance regimes, and sage-grouse habitat and the implications for land management activities.

Chapter 2: Estimating Historical Sage-Grouse Habitat Abundance Using a State-and-Transition Modeling Framework

Abstract. Habitat for wildlife species that depend on sagebrush ecosystems is of great management concern. Evaluating how management activities and climate change may affect the abundance of moderate and high-guality habitat necessitates the development of comparative reference conditions, but modeling tools for rangeland systems are limited and those that exist are hard for land managers to use. We developed a state-and-transition model using a combination of scientific literature and data for climate (temperature, precipitation, and snow), soils (soil surveys and ecological site descriptions), and fire (occurrence) to evaluate how different disturbances may have interacted and affected the abundance of seasonal habitat for the greater sage-grouse (Centrocercus urophasianus) prior to 1850 in southeastern Oregon. A primary purpose of this study was to evaluate use of climate data to define most disturbance probabilities and, subsequently, the relative mix of community phases and sage-grouse habitat. Contrary to our expectations, model results indicated fire might not have been the most important disturbance factor influencing sagegrouse habitat abundance, merely the most visible. Historically, sage-grouse breeding habitat may have been slightly more abundant (83% of the area) than levels currently recommended by sage-grouse biologists (80% of the area), breeding habitat may have been more abundant (64% of the area vs. recommended 40%), but wintering habitat may have been less abundant (53% of the area vs. recommended 80%). Our study demonstrated the use of climate data to derive probabilities of certain types of disturbance, providing an objective approach to estimating historical sage-grouse habitat abundance.

Keywords: State-and-transition, sagebrush, sage-grouse, reference conditions, climate

2.1 Introduction

Sagebrush (*Artemisia* spp. L.) ecosystems provide many important economic and social values in the Intermountain West, such as livestock forage, water, recreational opportunities, and habitat for a variety of wildlife species. Changes to sagebrush ecosystems over the last 150 years threaten their ability to provide many of these values in the future (Miller and Eddleman 2000, Connelly et al. 2004). Human-related disturbances, invasive species, expansion of conifer woodlands, changes in fire regimes, and changes in climate have all been involved in reducing the area occupied by sagebrush ecosystems by an estimated 14.8 million ha across the western United States (USDI Bureau of Land Management 2004). How interactions among these disturbances, particularly climate, affect sagebrush ecosystems are poorly understood.

Habitat for wildlife species that depend on sagebrush ecosystems is of great environmental concern in many areas of the interior West (Bunting et al. 2002, Knick et al. 2003, Connelly et al. 2004). In particular, habitat for the greater sage-grouse (*Centrocercus urophasianus*), a candidate species for listing under the Endangered Species Act, has been greatly reduced from historical conditions (Connelly et al. 2004). Climate has long been identified as a key factor in vegetation composition and dynamics, yet land management agencies have little understanding of how climate may alter habitat availability, quality, or connectivity through impacts on disturbances and successional dynamics. Dynamic global vegetation models, which examine broad-scale changes in vegetation potential and disturbance regimes (Neilson 1995, Beerling et al. 1997, Bachelet et al. 2001a, Smith et al. 2001, Sitch et al. 2003, Woodward and Lomas 2004, Morales et al. 2005, Gritti et al. 2006), provide a tool for predicting future vegetation changes at broad scales. However, at present, it is difficult to link broad-scale projections made with dynamic global vegetation models to potential local

changes in habitat conditions and disturbances. Local vegetation and disturbance models for forested ecosystems, such as the Forest Vegetation Simulator (Crookston and Dixon 2005) and DecAID (Mellen et al. 2002), can project changes in habitat conditions resulting from altered disturbance regimes and management activities, but there are no such models for sagebrush ecosystems.

Dynamic global vegetation models project the effects of climate change at the scale of biomes (e.g. temperate coniferous forests) and large landscapes (millions of hectares). They allow plant communities to shift as water balance changes in response to climate and weather events, and include some degree of positive and negative feedbacks that may shape future plant communities (Neilson 1995, Sitch et al. 2003, Gerten et al. 2004, Woodward and Lomas 2004, Morales et al. 2005). Dynamic global vegetation models do not usually include human management (Gerten et al. 2004) or disturbances other than fire (Neilson 1995, Sitch et al. 2003), and they lack the necessary parameters to operate below the biome level (Neilson 1995, Bachelet et al. 2001a, Sitch et al. 2003, Woodward and Lomas 2004).

State-and transition modeling frameworks, such as the Vegetation Dynamics Development Tool (VDDT) (ESSA Technologies Ltd. 2007), properly constructed, may allow connection of finer scale landscapes to the influences of climate trends projected by dynamic global vegetation models. These modeling frameworks operate at the finer scale of plant communities or community groups (e.g., warm, dry grand fir) and medium to large landscapes (hundreds to millions of hectares), incorporate management actions and relevant natural disturbances, and fit directly with the current rangeland ecology paradigm ((Westoby et al. 1989, Stringham et al. 2003, Briske et al. 2006), but see also (Bestelmeyer et al. 2004)). Since state-and-transition models are probabilistic instead of mechanistic, they can operate based on a combination of empirical data and expert opinion where empirical data are lacking; a common condition in rangeland management. State-and-transition models do not allow plant communities to shift as changing climate alters the water balance, but climate variables can form the basis of event probabilities to predict plant community changes. Users can develop historical reference conditions for the plant community or community group of interest, explore current conditions, and evaluate potential future conditions under a variety of scenarios using state-and-transition models.

The national LANDFIRE effort developed state-and-transition models and supporting data for historical reference conditions for plant community groups, or biophysical settings, of the United States (Rollins and Frame 2006). Although useful as a starting point, these generalized models include only fire as a disturbance agent whereas other disturbances, such as insect outbreaks and severe browsing or grazing, are important factors that may interact with fire and shape wildlife habitat. Time constraints during initial LANDFIRE model development necessitated an approach based largely on expert opinion (LANDFIRE National Implementation 2007a, b, c).

Our goals in this study were to evaluate the use of climate variables as a basis for event probabilities and evaluate how historical disturbances influenced reference conditions in sagebrush communities with an emphasis on the quantity and quality of greater sage-grouse habitat. Our primary objective was to develop VDDT-based models to examine the effects of these disturbances on plant community dynamics using fire, soils, and climate data; information from the scientific literature on sagebrush-steppe ecosystems; and selected rules used in dynamic global vegetation models. Questions explored included; 1) what disturbance factors may have been important in shaping sage-grouse habitat quantity and quality prior to 1850 (historically) and 2) how might the abundance of sage-grouse habitat historically have differed from the levels recommended by sage-grouse biologists presently. Our selected historical reference period began 500 years prior to 1850, a period commonly known as the Little Ice Age. Although general climate in this reference period was cooler and wetter than present, it had shifted into a winter-dominant precipitation regime with plant communities very similar to present (Miller and Wigand 1994). Prior to this period, climate was warmer than present with evidence of prolonged drought and widespread fire, significant shifts in plant communities, and less dominance of winter precipitation (Miller and Wigand 1994), a time commonly referred to as the Medieval Warm Period.

We used literature, climatic records, and a limited amount of expert opinion to develop probabilities of disturbance occurrence and successional pathways and rates for four sagebrush groupings. Sagebrush groupings were based on ecological site descriptions, which are descriptions of the physiographic and soil features, climate, characteristic and historical plant communities including estimated annual productivity in different types of years, and site suitability for different land uses such as livestock grazing, wildlife, recreation, and so forth for each site type. Using these values, we estimated the amount of each community phase and the resulting quantity of sage-grouse habitat within each sagebrush group and for the landscape as a whole.

2.2 Study Area

We selected the 4-million ha Malheur High Plateau major land resource area (NRCS 2006) in southeastern Oregon (figure 2.1) as the physical basis for model development; this area closely corresponds to the High Desert Ecological Province (Anderson et al. 1998). Much of the area lies between 1190 m and 2105 m elevation, with Steens Mountain reaching 2967 m. The terrain is interspersed with hills, buttes, isolated mountains, and north-south trending fault-block mountains. The area contains no major rivers and little surface water but has numerous springs, shallow lakes, and playas. Perennial streams and small rivers are mostly located on the periphery. Using soil series descriptions (available at

http://soils.usda.gov/technical/classification/osd/index.html), we estimated that 98% of the soils in the sagebrush ecological types of the Malheur High Plateau were Mollisols and Aridisols. In the uplands, soils are primarily loamy to clayey, well-drained and shallow (25 to 50 cm) to moderately deep (50 to 90 cm) on uplands. In the valley and basin bottoms, soils tend to be poorly to well-drained and deep to very deep (>90 cm). The average annual precipitation ranges from 105 mm to 305 mm over most of the area, with Steens Mountain receiving as much as 1450 mm at its upper elevations. Winter and spring are the wettest periods with most precipitation falling in November, December, January and May, while summer is the driest. January is the coolest month, averaging -2°C, and July the warmest, averaging 19°C. Sagebrush-steppe (*Artemisia* spp. L. and cespitose grasses) is the dominant vegetation type with salt desert shrub (*Sarcobatus vermiculatus* (Hook.) Torr.-*Grayia spinosa* (Hook.) Moq.) on saline soils in basins, western juniper (*Juniperus occidentalis* Hook. var. *occidentalis*) in rockier upland sites, and aspen (*Populus tremuloides* Michx.) at the higher elevations.

2.3 Methods

To evaluate potential vegetation responses to various disturbances, we sorted the sagebrush ecological sites for the Malheur High Plateau (<u>http://esis.sc.egov.usda.gov</u>) into four groups based on perennial grass productivity for low, average and high productivity years (table 2.1 and Appendix A). We assumed productivity provided an indicator of potential growth and recovery rates. We searched the literature to identify which disturbance factors were

important determinants of vegetation trajectories, what climatic factors were important to those disturbance factors, and how quickly sagebrush moved through different community phases in the absence of disturbance.

On the basis of the literature review, we selected fire; drought severe enough to kill sagebrush (drought); herbivory from insects, voles (*Microtus* spp.), and pronghorn (*Antilocapra americana*); freezekill; and snow mold as the disturbances to include in the models (see Model Design for more detail). We modeled fire, drought, and insect outbreaks as phenomena that affected the full area occupied by sagebrush (area-wide disturbance), and freezekill, snow mold, vole-related sagebrush mortality, and pronghorn browsing-related sagebrush mortality as phenomena that affected only a portion of the sagebrush-steppe (limited-area disturbance). We modeled fire, pronghorn browsing, freezekill, and snow mold as random events; drought, insects, and voles as cyclical events.

We obtained monthly precipitation and temperature data from 1895-2009 for Oregon Climate Division 7 (available at http://www7.ncdc.noaa.gov/CDO/CDODivisionSelect.jsp) and snow data from 1967-1996 for the Reynolds Creek Experimental Range (Hanson et al. 2001, Marks et al. 2001). Although Reynolds Creek Experimental Range lies outside the Malheur High Plateau, it has a climate that is similar; this data set provided detailed information on snowpack not available for Oregon Climate Division 7. We summarized monthly and seasonal means (temperature) and medians (precipitation) using a temperature-based definition of winter and summer that better matches plant phenology and hydrological cycles than the typical 3-month definitions (Neilson et al. 1992). We estimated the mean and standard deviation for snowpack duration, snow depth, and snowmelt date for the highest elevation station on the experimental range. We created a Monte Carlo Multiplier file to incorporate variability in both the occurrence and impact of fire, drought, insects, voles, and pronghorn browsing. In VDDT, this file type contained a random set of disturbance probability multipliers. For random disturbances, these multipliers were applied to the base disturbance probability, either increasing or decreasing the probability that the disturbance would affect a given spot. For cyclical disturbances, these multipliers affected whether the disturbance type occurs or not within the range of years defining the maximum and minimum interval between outbreaks and the minimum and maximum number of years within each outbreak (ESSA Technologies Ltd. 2007). We included variability in fire and pronghorn impacts by estimating the percentage of years in different severity categories (low, average, high, and extreme), calculating the average number of hectares affected using equation 1:

$$1 = (Lb_{l} + A + Hb_{h} + Eb_{e})x$$
 (1)

Where:

A = percentage of years classified as average

L = percentage of years classified as low

H = percentage of years classified as high

E = percentage of years classified as extreme

 b_y = additional hectares burned relative to 1 hectare in an average year

x = derived multiplier

The derived x was multiplied by the ratio of the additional hectares to create probability modifiers for the different severity categories. All disturbances were focused on the impacts to sagebrush and not to other plant life forms.

We constructed four state-and-transition models using VDDT version 6.0.9 (ESSA

Technologies Ltd. 2007), which enabled us to evaluate vegetation dynamics at fine, medium, and coarse scales . All models used four community phases: (1) grasses and forbs dominate the early seral (ES) community phase with sagebrush seedlings present, (2) sagebrush is subdominant and grasses and forbs remain dominant in the midseral open (MSO) community phase, (3) sagebrush, grass and forbs co-dominate in the late seral open (LSO) community phase, and (4) sagebrush is dominant in the late seral closed (LSC) community phase (Figure 2.2). A community phase consists of a distinctive plant community and its associated dynamic soil property levels that occur over time (Bestelmeyer et al. 2009). We used sagebrush cover as the trigger for a shift from one community phase to the next in the absence of disturbance (Table 2.1) and ignored sprouting shrubs to simplify model development.

All models were initialized with an equal proportion of the community phases. We ran 50 simulations for 500 years each, saved the area in each community phase every 10 years, and estimated the average annual area affected by each disturbance type. To allow ample time for the models to come into dynamic equilibrium, we analyzed model outputs for only the last 250 years of the 500-year simulation runs. We compared the predicted fire rotation in models to estimated fire frequencies published in the literature as a type of model validation, assuming that if the predicted rotations were similar to the published frequency estimates then the models were producing reasonable results. Because community phases in all models were prone to outliers except the ES phase, we based all analyses on medians rather than means.

We estimated average frequencies of the different disturbance types by taking the inverse of the average annual area impacted by a given disturbance type. For disturbances that applied to a limited portion of any sagebrush group, we multiplied the percentage of annualized average area impacted calculated in VDDT by the percentage of area in which the disturbance is believed to occur and used the inverse of the result to estimate disturbance frequencies. Because VDDT allows only one disturbance per time-step, we also totaled the annualized percentages of each disturbance and took the inverse to estimate the approximate frequency of any disturbance in each sagebrush group.

We conducted three rounds of model testing to evaluate the sensitivity of the mix of community phases to event probabilities other than those initially developed. We first varied the probability of single disturbance, ranging between 0 and 2 times the initial probability, and compared the differences in disturbance frequencies and the mix of community phases for each model to the initial predictions. We then varied the probability of fire and one other disturbance simultaneously, ranging between 0 and 2 times the initial probabilities, to determine what differences in both disturbance frequencies and the mix of community phases might arise if two factors were different from the original model. We used the same Monte Carlo multiplier file for all runs and followed the same procedures as in the initial model runs.

The first two sets of results indicated a need to test additional modifications to fire, drought, and insects. For fire, we borrowed a concept from fire danger rating that assumed that on 15% of the days the fire danger was low, on 75% of the days it was moderate, on 7% of the days it was high, and on 3% of the days it was extreme. Substituting years for days, we modified the frequency of the different types of fire years (low, average, high, and extreme) in the Monte Carlo multiplier file to explore whether the use of modern fire seasons was an appropriate basis for these multipliers and to explore how sensitive the models were to differing frequencies of fire season severity. Comparing initial model results and sensitivity

testing to the limited literature also suggested that our original models might have overstated the influence of drought and insects. We tested these modifications separately and then in combinations to evaluate the impacts on disturbance frequencies and the mix of community phases relative to our original models.

After finalizing the models based on sensitivity testing, we assigned seasonal habitat types and habitat quality ratings (low, moderate, high, none) to each community phase based on expected amounts of sagebrush cover, horizontal cover, forb abundance and timing of senescence, and expected duration of the habitat in the absence of disturbance as described in the sage-grouse literature (Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005) (Table 2.2). Seasonal sage-grouse habitat types included lekking, pre-laying hens, nesting, early and late brood-rearing, and wintering. We estimated the amount of sage-grouse seasonal habitat in each sagebrush group and the combined groups and compared the landscape amount to the recommendations from sage-grouse biologists (Connelly et al. 2000). Terminology in this study follows that used in the state-and-transition literature (Bestelmeyer et al. 2009). Plant nomenclature follows the U.S. Department of Agriculture online PLANTS database (<u>http://plants.usda.gov</u>).

2.4 Model Design

2.4.1 Sagebrush Groups

In developing the sagebrush groups, we assumed that site productivity was a strong influence on recovery rates. Since fire was the most-studied natural disturbance in sagebrush-steppe and many studies indicated that site productivity is a strong influence on recovery rates following fire (Blaisdell 1953, Harniss and Murray 1973, Young and Evans 1978, West and Hasson 1985, Boltz 1994, Wambolt et al. 2001, West and Yorks 2002, Lesica et al. 2007, Seefeldt et al. 2007, Bollinger and Perryman 2008), we based our indicator of site productivity on fine fuel production. We used grass production of 672 kg ha⁻¹ as the threshold for these divisions since that level of production is considered the minimum needed to support fire spread in bunchgrass fuels under moderate burning conditions (Gruell et al. 1986, Bunting et al. 1987).

The Warm-Moist Sagebrush Group (WM Group) typically resides on xeric, mesic, deep to very deep soils (Table 2.1). Water storage capacity is high and many sites are sub-irrigated. This group occurs mostly in swales, terraces, and near or in riparian areas below 1220 m elevation. It is the least common of the ecological site groups. Based on incomplete soil surveys (electronically available at http://www.or.nrcs.usda.gov/pnw soil/or data.html), this group occupies an estimated 11% of the Malheur High Plateau.

The Cool-Moist Sagebrush Group (CM Group) is found on xeric, frigid, moderately deep to deep soils mostly above 1220 m elevation (Table 2.1). Soils still have a high water storage capacity, but sub-irrigation is rare to nonexistent. This group typically occurs on northerly aspects and ridges, but can occur on higher elevation, cooler south aspects as well, and occupies an estimated 16% of the Malheur High Plateau.

The Warm-Dry Sagebrush group (WD Group) is found on aridic, mesic, moderately deep to shallow soils up to 1400 m elevation (Table 2.1). Water holding capacity is moderate to low and sites tend to become quite dry by mid to late summer. This group occurs mostly on southerly aspects, well-drained soils, and relatively shallow soils in basin bottoms and terraces, and occupies approximately 61% of the province.

The Shallow-Dry Sagebrush Group (SD Group) resides on aridic, mesic to frigid, shallow to very shallow soils at any elevation, from basin bottoms to high, shallow-soiled ridgetops (Table 2.1). Soils typically have low water storage capacity and high evaporation rates from temperature, wind, or both and become quite dry by late spring or early summer. The SD Group covers an estimated 12% of the Malheur High Plateau.

2.4.2 Successional Rates

We modeled deterministic movement through the four community phases as a function of sagebrush establishment and expansion of sagebrush cover across the thresholds we established (Table 2.1). Soil moisture availability in spring and early summer, not germination rates, appear to govern sagebrush establishment (Lomasson 1948, Mueggler 1956, Johnson and Payne 1968, Daubenmire 1975, Harniss and McDonough 1976, Boltz 1994, Meyer 1994, Forman et al. 2007). Since there were no obvious limitations on overall soil moisture availability in the WM Group, we assumed that random factors not specifically accounted for in the state-and-transition model affected sagebrush establishment in that group, such as the specific timing of precipitation events and any heat waves or cold snaps (Table 2.3). We based the probability of sagebrush establishment in the CM Group on the frequency of springs with average temperatures and average to greater than average precipitation. Since the WD and SD groups occupy drier soils, we based sagebrush seedling establishment on the frequency of wetter than average springs in the WD Group and both average spring temperatures and wetter than average conditions in the SD Group to avoid early drying as well as late frost injury.

We used estimated crown area of individual adult sagebrush grown in the wild and in common gardens (Pringle 1960, Tisdale et al. 1965, Johnson 1969, McArthur and Welch 1982,

Wambolt and Sherwood 1999, Anderson and Inouye 2001, Wambolt et al. 2001), including a crown area adjustment factor to account for crown size differences between wild and experimentally grown plants. We applied this adjustment factor to sagebrush species with measures only from a common garden (McArthur and Welch 1982) to estimate potential crown area for individual wild plants. The crown area of basin big sagebrush (*Artemisia tridentata* Nutt. spp. *tridentata*) represented the WM Group, mountain big sagebrush (*A. t.* Nutt. spp. *vaseyana* (Rydb.) Beetle) the CM Group, Wyoming big sagebrush (*A. t.* Nutt. spp. *wyomingensis* Beetle & Young) the WD group, and low sagebrush (*A. arbuscula* Nutt.) the SD Group.

We based the number of years in each community phase on the frequencies of sagebrush seedling establishment estimated above and an exponential establishment pattern beginning with a single plant and doubling the number of plants until the percentage cover of sagebrush crossed the threshold for each community phase (table 2.2) (Johnson 1969, Harniss and Murray 1973, Daubenmire 1975, Winward 1991, Miller and Eddleman 2000, Perryman et al. 2001, Forman et al. 2007, Lesica et al. 2007). For example, beginning with a single mountain big sagebrush plant, we doubled the population every 2 years and assumed it took 6 years for a seedling to reach physical and sexual maturity (McArthur and Welch 1982). At that rate for that subspecies, the CM Group reaches the LSC community phase in about 31 years.

2.4.3 Disturbances

We used the literature to identify potential historical disturbance factors to consider in the models. An extensive review of the sagebrush literature helped us develop estimates of the frequency, intensity and extent of those disturbance factors and which climate factors could serve as a basis for those estimates. Disturbance factors included fire (Knick et al. 2003,

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Connelly et al. 2004, Knick et al. 2005), drought (Ellison and Woolfolk 1937, Pechanec et al. 1937, Allred 1941), freezekill (Hanson et al. 1982, Walser et al. 1990), snow mold (Nelson and Sturges 1986, Sturges and Nelson 1986, Sturges 1986, 1989) and herbivory. Native herbivores impacting sagebrush ecosystems included bison (*Bison bison*) (Mack and Thompson 1982, Daubenmire 1985, Van Vuren 1987, Grayson 2006), elk (*Cervus canadensis*) (Hoffman and Wambolt 1996, Wambolt and Sherwood 1999), mule deer (*Odocoileus heminous*) (McArthur et al. 1988, Hoffman and Wambolt 1996, Clements and Young 1997), pronghorn (Hansen and Clark 1977, MacCracken and Hansen 1981, Howard 1995a, Verts and Carraway 1998), blacktailed jackrabbit (*Lepus californicus*) (Currie and Goodwin 1966, MacCracken and Hansen 1984, Anderson and Shumar 1986), voles (Hubbard and McKeever 1961, Mueggler 1967), and several species of insects (Allred 1941, Gates 1964, Hall 1965, Welch 2005).

We dropped bison, elk, mule deer, and black-tailed jackrabbits from further consideration. Although once present in Oregon, bison populations were believed to be small, isolated, and inbred. Bison apparently began dying out about the same time that the precipitation regime shifted to one of winter dominance (Daubenmire 1985, Van Vuren 1987, Williams 2005, Grayson 2006) with the species extirpated by about 1800 (Van Vuren 1987). There was little evidence of significant elk presence beyond the periphery of the Malheur High Plateau during the reference period (Davies 1961, Aikens and Couture 2007). Mule deer numbers were apparently limited during the reference period (Davies 1961, Clements and Young 1997) and mule deer consume more antelope bitterbrush (*Purshia tridentata* (Pursh) DC.) than sagebrush (Leckenby et al. 1982, Clements and Young 1997). We initially included blacktailed jackrabbits, but model testing indicated minimal to no effect on sagebrush dynamics during jackrabbit population peaks. Further literature review also indicated that black-tailed jackrabbit herbivory likely was not significant even during population peaks as other plants were preferred over sagebrush (Rice and Westoby 1978, Clark and Innis 1982).

Fire. We estimated the initial probability of a fire using a combination of precipitation and temperature to determine when sufficient fuel would be present. Since the WM Group can produce sufficient fuel each year, we assumed an earlier than average start to summer indicated early senescing of grass, resulting in low fuel moisture and a greater likelihood of a fire. Sufficient fuel in the CM Group occurred when spring precipitation recharged the surface soil layers and summers were drier than average. The WD Group required springs that were both wetter and longer than average to create sufficient available fuels. Lastly, the SD Group produced at least some grass to aid fire spread into this group under the same conditions as for the WD Group. Based on the Oregon Climate Division 7 records, sufficient fuel occurred in the WM Group 9.73% of years, in the CM Group 7.08% of years, and in the WD and SD Groups 5.31% of years. We then adjusted all these probabilities downward by 25% to account for having burnable fuels present but no ignition, a common situation in wildland fuels.

We used modern fire occurrence records for the 1980 through 2006 period for Lakeview and Burns Bureau of Land Management (BLM) Districts and Hart Mountain National Wildlife Refuge to estimate the frequency of low, average, high, and extreme years and the ratios in hectares burned in those types of years. We defined average years as those between the 25th and 75th percentiles, low years as those between the 5th and 25th percentiles, and high years as those between the 75th and 95th percentiles. Extreme years were outliers above the 95th percentile. We divided fire into two burn patterns-heterogeneous and homogeneous. We assumed homogeneous burn patterns represented rare events where fires burned across all community phases. Estimated rates of fire spread using BEHAVE Plus (Andrews and Bevins 2005) indicated a wind speed of 24 km hr⁻¹ measured at 6 m above the ground represented the rare event. Hourly wind data from remote automated weather stations currently in operation in the study area (data available at http://raws.dri.edu/index.html) indicated 10-minute average winds of this magnitude and higher occur 11.6% of the time in August, the time when fire danger typically peaks. We multiplied fire probabilities calculated above for the WM, CM, and WD groups by 11.6% to estimate the occurrence of homogeneous burn patterns and subtracted the result from the general fire probability to determine the likelihood of a heterogeneous burn pattern (Table 2.4). For example, the general probability of a fire in the CM Group was 5.31%, the probability of a homogeneous fire 0.62%, and the probability of a heterogeneous fire 4.69%. We assumed that heterogeneous burn patterns occur in low, average, and high years and homogeneous burn patterns occur in high and extreme years.

We modeled homogeneous fire as a stand-replacing event that reset any community phase back to the ES phase and heterogeneous fire as a thinning event within a given community phase. We assumed that depleted understories prohibited the occurrence of heterogeneous fire once a site reaches the LSC community phase in all groups and in the LSO phase as well in the WD Group (Daubenmire 1975, Bradford and Laurenroth 2006, Derner et al. 2008). Although the SD Group generally lacked sufficient available fuel by definition, fires have been observed to burn into and across this group during extreme burning conditions. Therefore, we assumed the SD Group supported only homogeneous fire. **Drought.** Drought-related mortality in sagebrush has been documented only during the 1930s drought (Ellison and Woolfolk 1937, Pechanec et al. 1937, Allred 1941) so the degree of impact beyond the small patch scale remains unclear. Drought frequencies based historical drought reconstructions using tree ring data (Keen 1937, Graumlich 1987, Cook et al. 1999, Cook et al. 2004, Knapp et al. 2004) did not seem appropriate since no relationship between growth declines in trees and mortality in sagebrush has been established. Therefore, we based our estimate of drought frequencies on drought reconstructions that compared past droughts to the 1930s drought (Keen 1937, Graumlich 1987, Cook et al. 1997, Cayan et al. 1998, Cook et al. 1999, Gedalof and Smith 2001, Cook et al. 2004, Knapp et al. 2004, Stahle et al. 2007) resulting in an estimated frequency of 100 to 200 yr. We modeled drought impacts as occurring in a single year (Table 2.4).

Drought did not affect the ES phase owing to lack of sagebrush. We modeled drought as a thinning event in the MSO and LSO phases in all models. Since Knapp et al. (2004) reported that drought impacts appear to be less in the CM Group during the reference period, we modeled drought effects in the LSC community phase the same as in the LSO and MSO phases. In the other three groups, we assumed drought reduced sagebrush cover enough to move the LSC community phase back to the LSO phase.

Insects. Aroga moth, a defoliator, appeared to be the primary insect affecting sagebrush (Gates 1964, Hall 1965, Hsaio 1986), but little work has been conducted on this species. The outbreak dynamics of Pandora moth (*Coloradia pandora* Blake) seemed most similar to what is known about aroga moth (Gates 1964, Hall 1965, McBrien et al. 1983, Hsaio 1986, Speer et al. 2001). Based on Pandora moth dynamics, we assumed the buildup and crash phases for aroga moth outbreaks lasted 2 years each and affected about 6% of the area and population

peaks lasted up to 2 years and affected 38.5% of the area (Gates 1964, Hall 1965, Meyers 1988, Speer et al. 2001). Allowing for no peak population in the model allowed for relatively minor outbreaks as well as major outbreaks. Outbreaks were modeled to occur every 20 to 48 years. Because the phases last multiple years, we used the fourth root of the estimated percentage of area affected to determine the probability that a given location would be affected during the buildup and crash phases and the square root of the estimated percentage area affected to determine the probability during the population peak phase (Table 2.4).

We assumed insect outbreaks affected only the later community phases when sagebrush density was relatively high (Hsaio 1986). The population peak reduced sagebrush cover enough to move the affected community phase back one phase. For example, a population peak in the LSO phase moves the location back to the MSO phase. The buildup and crash phases reduced sagebrush cover without conversion to an earlier community phase.

Pronghorn Browsing. Pronghorn browsing could thin sagebrush during severe winters when animals were concentrated and populations were high. We constructed a very simple model of pronghorn population dynamics (Kindschy et al. 1982, O'Gara and Yoakum 2004, Smyser et al. 2006, Yoakum 2006) to estimate the frequency of low, average and high populations. Using a temperature-based definition of winter (Neilson et al. 1992), an analysis of Oregon Climate Division 7 data indicated that winter typically lasted 2.5 to 3 months. Therefore, we defined a severe winter as one that was 3 to 4 months long, colder than average, and average to wetter than average, resulting in deep persistent snow (Table 2.4). We assumed the animals used 25% of the WD and SD Groups during severe winters (O'Gara 1978). We modeled pronghorn browsing as reducing sagebrush cover within the MSO and LSO community phases without conversion to an earlier phase and removing enough
sagebrush cover to move the LSC community phase back to the LSO phase (Smith 1949, McArthur et al. 1988, Bilbrough and Richards 1993, Hoffman and Wambolt 1996).

Voles. We restricted vole damage to the CM Group where deeper and persistent snowpacks are more frequent compared to the other groups (Mueggler 1967, Frischknecht and Baker 1972, Parmenter et al. 1987) and to community phases with a mix of sagebrush and relatively high grass production (Frischknecht and Baker 1972, Parmenter et al. 1987). We assumed these conditions were most probable in ephemeral draws and swales in northerly aspects, or about 10% of the CM Group. A 4- to 5-year outbreak cycle (Murray 1965, Frischknecht and Baker 1972) meant that over a 100-year period, 23% of the years would have vole populations high enough to thin sagebrush. As with pronghorn, we also assumed that volerelated mortality was likely only when high populations coincided with a severe winter, increasing the likelihood that voles would exhaust their preferred foods before snowmelt. We varied the probability of mortality by community phase, taking into account the relative proportion of grass to sagebrush. The highest probability of vole outbreaks occurred in the LSO phase, the lowest in the LSC phase owing to lack of grass and intermediate probability in the MSO phase owing to the relative sparseness of sagebrush (Table 2.4). We modeled volerelated mortality as a thinning event that reduced sagebrush cover in the MSO and LSC community phases without conversion to an earlier community phase and that moved the LSO phase back to the MSO phase.

Freezekill. Sagebrush normally covered by snow was subject to freezekill under a certain series of events. Below-average snowpacks or a late start to snow accumulation, combined with episodes of very cold temperatures froze soils deeper than usual in the early part of winter but left sagebrush incompletely or uncovered by snow (Hanson et al. 1982, Walser et

al. 1990, Hardy et al. 2001, DeGaetano and Wilks 2002). Soil remained frozen longer than typical with limited ability to thaw from the bottom up (Hardy et al. 2001). A subsequent warm spell in late winter or early spring could then trigger loss of dormancy and the onset of physiological activity in sagebrush (Walser et al. 1990), resulting in physiological drought when plants were unable to extract water from the still-frozen soil (Hanson et al. 1982, Walser et al. 1990, Hardy et al. 2001).

Episodes of very cold temperatures occurred when arctic air masses drop into eastern Oregon (Taylor and Hatton 1999) but were not possible to detect in the monthly temperature data (Nelson and Tiernan 1983). We assumed the probability of freezekill was highest in a dry winter with a warmer than average January or February, but also more likely to occur on north aspects and higher elevations in the CM Group. We estimated 25% of group was susceptible to freezekill (table 2.4). We modeled freezekill as reducing sagebrush cover within the MSO community phase without conversion to an earlier community phase, and moving both the LSO and LSC phases back to the MSO phase (Hanson et al. 1982, Nelson and Tiernan 1983).

Snow Mold. Deep, persistent snowpack can trigger snow mold (Sturges 1986, 1989). The fungus was inactive in years where snow depth was less than 400 mm (Sturges and Nelson 1986). Using data from the Reynolds Creek Experimental Range (Hanson et al. 2001), we identified winters where snowpack duration, snow depth, and snowmelt date were greater than one standard deviation from the mean at the highest elevation station and assumed that snow mold occurs only on 25% of the CM Group (Table 2.4). The effects of snow mold were modeled the same as for freezekill.

2.4.4 Greater Sage-Grouse Habitat Suitability

Table 2.2 depicts the sage-grouse habitat suitability rating for each sagebrush group and community phase. We evaluated seasonal habitat suitability, based on community structure, for the different community phases in each group. Seasonal habitat types for sage-grouse included lekking, pre-laying hen, nesting, early and late brood-rearing, and wintering. Open areas with little or no sagebrush cover surrounded by big sagebrush were lekking habitat (Connelly et al. 2000, Braun et al. 2005). Pre-laying hens derived many of the nutrients they need for high-guality egg production from areas rich in forbs (Barnett and Crawford 1994, Crawford and Gregg 2001, Gregg 2006); we assumed forb abundance decreased as sagebrush cover increased in the WM, CM, and WD Groups (Ersch 2009). Nesting habitat consisted of individual large sagebrush plants 40 to 80 cm tall within an area of 15 to 25% shrub cover, 15+% grass cover, and 10% forb cover with herbaceous plants at least 18 cm tall (Call and Maser 1985, Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005, Goodrich 2005, Gregg and Crawford 2009). Early brood-rearing habitat was very similar to nesting habitat, although canopy cover was slightly lower at 10 to 25% and this habitat required an abundance of insects (Call and Maser 1985, Braun et al. 2005, Goodrich 2005). Late brood-rearing habitat also had similar characteristics but was more moist with forbs that remained succulent later into the summer or early fall; sagebrush and an abundance of insects were not essential parts of the habitat (Call and Maser 1985, Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005). Wintering habitat consisted of sagebrush that extended 25 to 30+ cm above the snow with 10-30% canopy cover (Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005).

2.5 Results

Disturbance Frequencies. All models were sensitive to the frequency of fire and insect outbreaks and the frequency of the different types of fire years, ranging from low to extreme. Altering the frequency of the different types of fire years had a large impact on fire rotation and the mix of community phases, particularly in the abundance of the ES phase, in all four groups. The natural fire rotation lengthened 2.7 times in the WM and CM Groups and 3.5 times in the WD and SD Groups, well outside that reported in the literature (Table 2.5) (Burkhardt and Tisdale 1976, Whisenant 1990, Miller and Rose 1999, Miller et al. 2001, Knick et al. 2005, Baker 2006, Heyerdahl et al. 2006, Mensing et al. 2006, Miller and Heyerdahl 2008).

The WM and CM groups were not sensitive to drought within the range tested, but the WD and SD Groups were somewhat sensitive. The CM Group was moderately sensitive to the frequency of vole outbreaks and sensitive to the frequency of freezekill and snow mold within the range tested. Both the WD and SD Groups were sensitive to the frequency of pronghorn browsing.

When we varied the probability of fire in conjunction with another disturbance type and the effects on the abundance of community phases were similar to when each disturbance was varied separately, the changes in abundance generally were additive in nature. However, if the effects of the other disturbance were opposite to that of fire, the combined effects somewhat canceled each other, trending towards less change. Other types of interactions also appeared, although the interactions were not always large. For example, varying the probability of drought and insects simultaneously also affected the frequency of vole

outbreaks in the CM Group. Varying the probability of drought or insect outbreaks also affected fire rotation in the WD and SD Groups.

Contrary to our expectations, fire had less influence than all other disturbance types, excepting drought (Table 2.6). Estimated fire rotations were 24, 33, 83, and 196 years for the WM, CM, WD, and SD Groups, respectively. Other disturbance types took fewer years to affect an area equivalent to the entire landscape in our simulations, whereas drought took several hundred years to do the same.

In each model, some sort of disturbance occurred rather frequently across the landscape as a whole. General disturbance intervals ranged from a minimum interval of 4 years in that part of the CM Group subject to all disturbances to a maximum of 26 years in that part of the SD Group subject only to fire, insects, and drought. Portions of the landscape subjected only to the fire, insects, and drought had a longer interval between any disturbances than the portions also subjected to voles, snow mold, and freezekill in the CM Group and also subjected to pronghorn browsing in the WD and SD groups.

Mix of Community Phases. The LSC community phase was the most common phase in all Groups (Figure 2.3), although the dominance of this community phase differed by model, likely owing to differences in recovery rates. The least common community phase was the MSO phase in the WM, WD, and SD Groups and the ES phase in the CM Group. All groups were subject to outlier values in at least one community phase, with sudden decreases or increases in abundance followed by a rapid return to values closer to the median.

In the WM Group increasing the probability of fire, either singly or in combination with drought and insects, decreased the abundance of the LSC community phase and increased the MSO and LSO phases. In the CM Group, altering the probability of fire, insects, voles, and freezekill affected the abundance of the MSO and LSC community phases more than the LSO phase; increasing the probability of drought or insects above that in the initial model had no effect on the abundance of any community phase. The abundance of the LSC community phase least affected. In the cases of fire plus snow mold and fire plus insects, the effects on the MSO and LSO community phases were similar.

Varying the probability of fire in the WD Group had the most impact on the abundance of the LSO community phase and least on the LSC phase, whereas varying the probability of insect outbreak had the most impact on the LSC phase and the least on the LSO phase. Varying drought had the least effect in all community phases. As with the WM Group, increasing the probability of drought or insects up to double the initial probability had no effect. Varying fire and insects simultaneously had the greatest impact on the abundance of the LSC community phase; varying fire and drought together had the greatest impact in the LSO phase; and varying fire in conjunction with pronghorn browsing had the greatest impact on the MSO phase.

We found that the responses of the SD Group were very similar to that of the WD Group with a few exceptions. The difference between the greatest amount and least amount of the LSO and MSO community phases was narrower in the SD Group than in the WD Group when we varied the probability of fire. Varying fire in conjunction with insects resulted in a more similar magnitude of change in abundance between the LSC and MSO community phases in the SD Group than in the WD Group.

Sage-Grouse Seasonal Habitat. Our simulations suggested that the historical Malheur High Plateau landscape provided lekking habitat on 21% of the area, nesting habitat on 56% of the area, and pre-laying hen habitat on 83% of the area (Figure 2.4). Assuming these three categories of seasonal habitat meet the definition of breeding habitat discussed in Connelly et al. (2004), then the modeled study area provided breeding habitat on 86% of the Malheur High Plateau in Oregon. However, only about 22% of the area was simulated as high quality breeding habitat. Most of the lekking habitat was moderate quality due to the much greater abundance of the WD Group, even though the proportion of this group providing lekking habitat was less than the proportion of the SD Group. Most of the nesting habitat was high quality and pre-laying hen habitat was about equally divided between moderate and high quality habitat.

Our simulations indicated that the historical landscape provided early brood-rearing habitat over 58% of the model area and late brood-rearing habitat over 24% of the modeled area. Collectively, the historical Malheur High Plateau landscape may have provided brood-rearing habitat over 64% of the simulated area (Figure 2.4). Most brood-rearing habitat of both types was of moderate quality. Wintering habitat was simulated as occurring on 53% of the historical landscape, which much of this habitat high quality. Although Table 2.4 includes the SD Group as wintering habitat, we did not include it in this estimate as this group serves as wintering habitat only in low snow years.

2.6 Discussion

This project demonstrates that state-and-transition models may be useful for examining potential historical ecosystem dynamics and habitat availability in sagebrush-steppe communities. It also demonstrates how climate data can be used to develop objective disturbance probabilities and examine how climate can influence ecosystem dynamics at finer scales than currently possible in dynamic global vegetation models. Our study also provides objective criteria, which could be used to evaluate expert opinion and the logical arguments that underpin such opinion.

Disturbance Frequencies. Although consensus among fire managers and sage-grouse biologists has been that fire was the most important disturbance factor in sagebrush dynamics, model results suggested that fire may simply have been the most easily observed disturbance under historical conditions. Instead, the less visible and subtler disturbance types (particularly insects) may have played a much greater role in sagebrush dynamics prior to 1850 either through their effect on a much larger area than fire or through a more frequent occurrence rate. The models may understate the role insects, particularly aroga moth, and drought may have played in past sagebrush dynamics. We modeled both disturbance types as thinning agents rather than agents that cause a shift into another community phase. The literature is limited, but there is some evidence that aroga moth and drought could be standreplacing disturbances under certain conditions (Ellison and Woolfolk 1937, Pechanec et al. 1937, Allred 1941, Gates 1964, Hall 1965, Hsaio 1986); however, these conditions remain poorly defined. Long-term drought can affect the abundance of sagebrush (Mehringer 1987, Miller and Wigand 1994, Mensing et al. 2006, Miriti et al. 2007); what is much less clear is how severe or how prolonged the drought must be (McDowell et al. 2008). Because aroga moth outbreaks are not monitored, there are no data on outbreak frequencies, outbreak size

and size variation, and impacts at the landscape scale to inform the modeling effort. The modeled effects when drought and insects are treated as thinning agents suggest the potential for even greater influence on sagebrush dynamics if these disturbances could be treated in a similar fashion as fire—as a thinning agent under one set of conditions and a stand-replacing agent under another set.

The models were very sensitive to the frequency distribution of the different types of fire year and frequency of insect outbreaks and less sensitive to the limited-area disturbances and drought. We had expected limited sensitivity to drought because of the long return interval modeled, but included it anyway to provide a foundation for examining climate change in future work. The CM Group model in particular demonstrated that disturbances interact. For example, the frequency of insect outbreaks altered the frequency of vole outbreaks apparently by altering the abundance of the late seral open community phase—the phase in which a vole outbreak is most likely to occur. A change in the frequency of one disturbance tended to alter the frequency of another disturbance in the two dry sagebrush groups than in the two moist sagebrush groups. The reason for that outcome is unclear and may be a modeling artifact.

We expected that the probabilities of other disturbances would also differ by community phase, setting up even more interactions, but lack the evidence needed to determine how much they should differ. For example, we defined the late seral closed community phase as one that is beginning to lose bunchgrass understory, indicating that the probability of a fire should be less in all groups than in the earlier community phases with a greater abundance of grass. However, we do not have the information required to define how much grass biomass may differ between the late seral closed and late seral open community phases nor whether fire spread differs significantly between the two phases. One effect of a thinning disturbance that shifts the community phase, such as an insect outbreak, should be to make more site resources available to herbaceous species, leading to an increase in grass fuel loading and continuity and increased ability to support fire spread. If we were able to include an increase in the probability of fire following such a thinning disturbance, additional interactions might more apparent and the relative importance of fire might change.

We speculate that modern burned-hectare totals per fire season in our study area may not be much different from those prior to 1850. Use of fire by Native Americans was well documented in the Great Basin (Gruell 1985, Robbins 1999, Griffen 2002, Stewart 2002), although the number of hectares affected was not. Most fires observed by Euro-Americans in the mid-1800s were at higher elevations (Robbins 1999, Griffen 2002), but Stewart (2002) documented the frequent use of fire by Native Americans at lower elevations. Tree-ring studies of fire extent in pre-1850 forests indicate that regional fire years (years where fire is widespread throughout a large area, the equivalent of extreme fire years today) occurred at about the same frequency prior to 1850 as in modern fire records (Swetnam and Betancourt 1998, Hessl et al. 2004, Heyerdahl et al. 2008). One possible difference between the time prior to 1850 and since 1980 is average fire size, as compared to total hectares burned per year. Before 1850, a year where a great many hectares burned may have consisted of a large number of small to medium-sized fires. Since 1980, such years consisted of a few very large fires, believed to be largely due to changes in fuel structure resulting from a combination of past grazing, aggressive fire suppression, and a variety of other human-caused changes (Connelly et al. 2004, Knick et al. 2005, Heyerdahl et al. 2006). The resulting landscape patterns and sage-grouse habitat quality and availability would have been very different prior

to 1850 than since 1980 even if the frequencies of the different types of fire season were similar.

Mix of Community Phases. Comparing modeled results to actual historical vegetation dynamics prior to 1850 was not possible. Ideally, we would compare the modeled results against written descriptions of the area, photos, and research data or results from other ecological studies in similar environments. Only a few sources exist for the Malheur High Plateau. Human population densities in the Great Basin have always been low (Kelly 1997, Griffen 2002), limiting the availability of ethnographic descriptions and accounts from explorers and settlers. What accounts do exist suggest that grass was relatively abundant and sagebrush cover was relatively low to moderate (Rich 1950, Davies 1961, Clark and Tiller 1966, Menefee and Tiller 1977a, b). However, the information is not sufficiently detailed to describe relative community phase abundance, other than to vaguely suggest that the LSC community phase may not have been widespread in the lower elevations of the study area.

We also evaluated whether the soil orders present might provide some indirect support to the modeled projections of historical conditions. Mollisols develop under grasslands and shrubsteppes, and strong mollic epipedons indicate where grasses have been co-dominant to dominant for a prolonged period (Buol et al. 2003, Heyerdahl et al. 2006, Miller and Heyerdahl 2008). Aridisols develop where conditions are very dry and potential evapotranspiration demand typically exceeds precipitation by a significant amount (Buol et al. 2003), but they provide little information about the plant life forms that dominate, owing to the wide range of temperature regimes in these soils. Mollisols dominate in the WM and CM Groups, Aridisols in the WD Group, and Mollisols and Aridisols occur in nearly equal proportions in the SD Group. The only conclusion this information supports is that grasses have been dominant to codominant for a prolonged period throughout the study area and that the WD and SD Groups tend to occur on very dry sites.

Although the LSC community phase dominates within the WM and CM models, the abundance of this phase tends to change somewhat rapidly every 10 years with relatively common outliers in abundance at the lower end. This volatility and the outliers at the lower end but not the upper end combined with the relatively rapid transitional rate between community phases and short fire rotations suggests relatively frequent cycling between the LSC community phase and earlier community phases in both groups, likely promoting the overall co-dominance of grass over time and the development of Mollisols.

We also compared the estimated fire rotation against tree-ring based estimates and expert opinion estimates of fire frequency. We assumed that if the fire frequencies in the literature and fire rotations from the models were relatively close, the model results were sufficiently accurate that the calculated mix of community phases was a reasonable representation of the reference period. Tree-ring studies at the sagebrush-conifer ecotone indicate an average fire return interval of 10 to 35 years (Burkhardt and Tisdale 1976, Miller and Rose 1999, Miller et al. 2001, Heyerdahl et al. 2006, Miller and Heyerdahl 2008). Expert opinion for fire return intervals range from 10 to 25 years on more productive sites, 30 to 80 on less productive sites, and over 100 years on very dry, low-productivity sites (Miller and Rose 1999, Knick et al. 2005, Miller and Heyerdahl 2008) (see also Table 2.5). The modeled fire rotations all fell within these general categories, although the fire rotation in the CM Group was at the longer end of the expected range. Thus, the indirect evidence suggests the mix of community phases is reasonable. **Sage-Grouse Seasonal Habitat.** Seasonal habitat descriptions indicate that sage-grouse generally lek in the SD Group; nest, begin brood-rearing, and winter in the WD Group; and complete brood-rearing and summer in the CM Group (Call and Maser 1985, Connelly et al. 2000, Braun et al. 2005, Goodrich 2005). Some nesting and wintering occurs at the lower elevations of the CM Group and some lekking occurs in the early community phases of both the CM and WD Group, although more frequently in the WD Group. Sage-grouse use of the WM Group remains speculative on our part as most of this sagebrush-steppe type was lost to agriculture before sage-grouse studies began. Goodrich (2005) suggested this group could provide some nesting and wintering habitat where shrubs are not too tall, and brood-rearing where succulent forbs are readily available.

Our predicted quantity of sage-grouse seasonal habitats appears to be similar to that recommended by sage-grouse biologists, with the exception of wintering habitat (Connelly et al. 2000, Connelly et al. 2004). Our models indicated slightly more breeding habitat was present historically than the recommended 80% of the landscape, but only about 70% of the recommended amount of wintering habitat (Connelly et al. 2000). We estimated that 1.5 times the recommended amount of brood-rearing habitat may have been present in the past (Connelly et al. 2000), albeit heavily skewed toward early brood-rearing habitat. If the recommended amount were evenly divided between early and late brood-rearing, then our models provided three times the early brood-rearing habitat produced by our models is at least somewhat misleading. Sage-grouse chicks tend to use the edges of the SD Group (Aldridge 2000, 2005, Goodrich 2005) but we excluded the SD Group from our calculations since we could not be very certain of the actual proportion used. The proportion of the SD Group used depends on the size and configuration of these patches relative to nearby cover.

Our models do not include spatial patch characteristics, such as edge-to-interior ratio, that could further refine this estimate.

Our model results also suggest that the availability of wintering habitat and possibly late brood-rearing habitat might have been population bottlenecks; limiting adult recruitment, survival, and total population. Although sage-grouse will winter in the SD Group in many locations, the majority of wintering populations in Oregon have been observed in more productive sites dominated by big sagebrush (Connelly et al. 2004), placing them in either the WD or CM Groups. Much of the modeled seasonal habitat fell into the moderate category, which may have also limited total population. At least one community phase in each model is prone to outlier values, and means tend to be skewed to the right or left of the median. Outliers tended to quickly return towards the median, but indicate that historical variability in sage-grouse habitat may have been higher than either managers or sage-grouse biologists would prefer and may be a driver in bird population swings.

Modeled recovery rates in the WM and CM Groups imply that frequent disturbance of some type likely maintained forb production needed for brood-rearing and kept sagebrush cover at levels that supported nesting in the pre-1850 landscape (Connelly et al. 2000, Connelly et al. 2004, Aldridge 2005, Moynahan et al. 2007). Even though the LSC community phase, in which grass production begins to decline, dominated in these two groups, the frequency of thinning disturbances (drought, insects, heterogeneous fire, etc.) might have kept this phase in a relatively open condition and generated frequent cycling between the LSO and LSC phases. The use of non-spatial state-and-transition models means we cannot evaluate spatially dependent ecological processes and habitat requirements that depend on spatial attributes. For example, contagion of disturbances creates spatial mosaics that are important characteristics of sage-grouse habitat (Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005). Spatially explicit state-and-transition models require much more expertise to calibrate and run, require a much higher number of Monte Carlo simulations to account for patch-scale variation in disturbance patterns, necessitate a greater number of detailed assumptions about disturbance contagion, and generally are computationally demanding. We expect spatially explicit models would be more practical at fine to middle scales (e.g., thousands to hundreds of thousands of hectares), but non-spatial models can provide useful information, make fewer assumptions about disturbance behavior, and are relatively easy for managers to use.

Ignoring sprouting shrubs meant we could have missed significant effects these species may have on successional rates, either increasing the rate by providing safe sites for sagebrush seedlings or retarding the rate through competition for moisture and nutrients, and altering disturbance probabilities. Rabbitbrush species (*Chrysothamnus viscidiflorus* (Hook.) Nutt. and *Ericameria nauseosa* (Pall. ex Pursh) G.L. Nesom & Baird) are the most widespread sprouting shrub species in the study area, and recently were identified as a potentially important component of brood-rearing habitat (Ersch 2009). Root systems of sprouting shrubs are usually fully developed at the time of disturbance allowing rapid recovery when these species are top-killed, and allowing abundant seed production with relatively high seedling establishment (Young and Evans 1974, Tirmenstein 1999a, b). However, rabbitbrush species are early to midseral species with minimal presence in later community phases (Blaisdell 1953, Harniss and Murray 1973, Young and Evans 1974, Tirmenstein 1999a, b, Mata-González et al. 2007).

2.7 Conclusions

Our modeled historical fire rotations were within the range reported in the literature and suggested by expert opinion. Fire may not be the most important disturbance factor shaping landscape patterns and habitat available, just the most visible and easily studied factor. The frequency of the different types of fire seasons is an important, but possibly overlooked factor in how fire shapes habitat availability. Disturbances in general interact, not a new finding, but these interactions affect not only the specific disturbances examined, but also other disturbances by altering the abundance of different community phases. The interaction may be stronger when the probability of an unrelated disturbance varies by community phase. Relatively subtle disturbances, such as drought, herbivory, and weather-related mortality, may have been more important in historical sagebrush-steppe communities than previously recognized, even in the drier sagebrush types.

Sage-grouse breeding habitat availability may have been somewhat greater under historical conditions than that recommended by sage-grouse biologists, but late brood-rearing and wintering habitat may have been less. If so, these shortage categories along with a predominance of less than optimal habitat may indicate population bottlenecks that could have limited sage-grouse population potential both in the past and in the present.

Constructing the models revealed how little is known about the so-called natural disturbance types and regimes that occur in sagebrush ecosystems. We drew on big sagebrush literature from across the range of the species without certain knowledge that all the disturbance types included in the models actually occur within the Malheur High Plateau area. There is almost no literature concerning disturbance factors in low sagebrush. We were surprised by what appears to be the greater importance of insect outbreaks relative to fire, but that modeled outcome may be due to our use of a forest-based surrogate to inform the modeling effort instead of a real effect of sagebrush defoliators. Given the sensitivity displayed to the probability of insect outbreak effects and the lack of data for outbreak size, we may have over-estimated the probable size of most outbreaks, thereby artificially increasing the relative importance of insects. Table 2.1. Characteristics of each sagebrush group including modal potential natural plant community, grass production, and sagebrush cover by community phase. The top portion of the table describes modal site characteristics. The bottom portion of the table identifies sagebrush cover thresholds for the different community phases in each sagebrush group.

	Warm-Moist			Shallow-Dry
	Group	Cool-Moist Group	Warm-Dry Group	Group
Modal plant association	Artemisia tridentata Nutt. ssp. tridentata/ Leymus cinereus (Scribn. & Merr.) A. Löve	Artemisia tridentata Nutt. ssp. vaseyana (Rydb.) Beetle/ <i>Festuca</i> idahoensis Elmer	Artemisia tridentata Nutt. ssp. wyomingensis Beetle & Young/ Pseudoroegneria spicata (Pursh) A. Löve-Achnatherum thurberianum (Piper) Barkworth	Artemisia arbuscula Nutt./ Poa secunda J. Presl
Years producing at least 672 kg ha ⁻¹	All	High and Average	High	None
Dominant soil moisture regime	Xeric	Xeric	Aridic	Aridic
Dominant soil temperature regime	Mesic	Frigid	Mesic	Mesic to frigid
General soil depth	Deep to very deep	Moderately deep to deep	Shallow to moderately deep	Very shallow to shallow
		Sagebrush cover	by community phase	
Early seral	<1%	<1%	<1%	<0.25%
Midseral open	1-10%	1-10%	1-8%	0.25-1%
Late seral open	10-25%	10-30%	8-20%	1-5%
Late seral closed	>25%	>30%	>20%	>5%

Note: Characteristics are based on ecological site information for the Malheur High Plateau (available at <u>http://esis.sc.egov.usda.gov</u>), Winward 1991, and Miller and Eddleman 2000.

				Early		
		Pre-laying	N 1 11	brood-	Late brood-	
	Leks	hens	Nesting	rearing	rearing	Wintering
Warm-Mois	st Group					
ES	Low	Moderate	N/A	Low	Moderate	N/A
MSO	N/A	Moderate	Low	Moderate	High	N/A
LSO	N/A	Low	High	High	High	High
LSC	N/A	N/A	Moderate	Low	Low	Moderate
Cool-Moist	Group					
ES	Low	High	N/A	Low	Moderate	N/A
MSO	N/A	High	Low	Low	High	N/A
LSO	N/A	Moderate	High	High	High	High
LSC	N/A	N/A	Moderate	Moderate	Moderate	Moderate
Warm-Dry Group						
ES	Moderate	High	N/A	Low	Low	N/A
MSO	Low	High	N/A	High	Low	Low
LSO	N/A	Moderate	High	High	Low	High
LSC	N/A	Moderate	Moderate	Moderate	Low	High
Shallow-Dr	y Group					
ES	High	High	N/A	High ^a	Moderate	High ^b
MSO	Moderate	High	N/A	High ^a	Moderate	High ^b
LSO	Low	High	N/A	High ^a	Low	High ^b
LSC	N/A	High	N/A	High ^a	Low	High ^b

Table 2.2. Habitat suitability (low, moderate, high) for greater sage-grouse by model and community phase

^a High along edges, dropping to low in interior

^b High until or unless buried by snow

Notes: Descriptions of each habitat element are based on Call and Maser 1985; Connelly et al. 2000, 2004; Crawford and Gregg 2001; Goodrich 2005; Braun et al. 2005; and Gregg 2006. Quality rating is based on how closely a given community phase in each sagebrush group matches the definition of that habitat and its expected duration (see text under "Greater Sage-Grouse Habitat Suitability").

NA = not applicable, ES = early seral, MSO = midseral open, LSO = late seral open, LSC = late seral closed.

	Establishment		Years in community phase			
Sagebrush group	Probability	Frequency	Early Seral	Midseral open	Late seral open	Late seral closed
				years		
Warm- Moist	0.75	1.5	0-14	15-20	21-24	25+
Cool-Moist	0.62	2	0-18	19-25	26-30	31+
Warm-Dry	0.18	6	0-48	49-66	67-78	78+
Shallow- Dry	0.15	7	0-55	56-76	77-90	91+

Table 2.3. Establishment and successional rates in the absence of disturbance

Note: Establishment parameters based on probabilities of certain climatological conditions in Oregon Climate Division 7. Years in each community phase based on establishment frequency and estimates of canopy closure derived from the literature (see text under Successional Rates).

	Warm-Moist Group	Cool-Moist Group	Warm-Dry Group	Shallow-Dry Group
Homogeneous fire	0.0085	0.0062	0.0046	0.0046
Heterogeneous fire	0.0645	0.04969	0.0352	
Drought	0.375	0.375	0.375	0.375
Insect buildup and crash	0.2943	0.2943	0.2943	0.2943
Insect peak	0.4388	0.4388	0.4388	0.4388
Pronghorn			0.0199	0.0199
Voles		0.0360 – MSO 0.0785 – LSO 0.0100 - LSC		
Freezekill		0.0088		
Snow mold		0.0177		

Table 2.4. Probabilities of disturbance within each model for each applicable disturbance factor

MSO = midseral open, LSO = late seral open, LSC = late seral closed.

Table 2.5. Comparison of fire rotation between the final models, the altered fire variant, and literature. In the final model, the frequency of different types of fire years was based on modern fire records. In the altered fire variant, the frequency of different types of fire years was based on the fire danger paradigm.

Sagebrush		Altered Fire			
Group	Final Model	Variant		Literature	
		Years			
Warm-Moist	24	64	10-20	Knick et al. 2005	
Cool-Moist	33	88	16-33 3-28 3-32 10-25 2-84 70-200 <25	Burkhardt and Tisdale 1976 Miller and Rose 1999 Miller et al. 2001 Knick et al. 2005 Heyerdahl et al. 2006 Baker 2006 Miller and Heyerdahl 2008	
Warm-Dry	83	250	50->100 up to 100 100-240 >80	Knick et al. 2005 Mensing et al. 2006 Baker 2006 Miller and Heyerdahl 2008	
Shallow-Dry	196	667	148+ >100 325-450	Miller and Rose 1999 Knick et al. 2005 Baker 2006	
Unspecified			60-110	Whisenant 1990	

Warm-Moist Group Cool-N		Cool-Moist Group	Warm-Dry Group	Shallow-Dry Group
	Insects Snow mold		Pronghorn	Pronghorn
	Fire	Voles	Insects	Insects
	Drought	Insects	Fire	Fire
		Freezekill	Drought	Drought
		Fire		
		Drought		

Table 2.6. Relative importance of disturbance types in each sagebrush model based on the estimated disturbance rotation period



Figure 2.1. Location of the study area in Oregon. The High Desert Province Ecological Province and the Malheur High Plateau Major Land Resource Area occupy approximately the same area, with the exception of the area to the east of Steens Mountain. The Malheur High Plateau Major Land Resource Area includes some area to the east of Steens Mountain while the High Desert Ecological Province does not. The area to the east of Steens Mountain lies within the rain shadow of the mountain and has a different climate. The town of Burns is the largest community within the study area.



Figure 2.2. Model structure. Arrows pointing to the right indicate deterministic transitions resulting from succession. Arrows pointing to the left indicate probabilistic transitions to an earlier community phase. Circles indicate probabilistic transitions that remain in the same community phase. ES = early seral, MSO = midseral open, LSO = late seral open, LSC = late seral closed.



Figure 2.3. Mix of community phases. The late seral closed (LSC) phase is the most common in all models, although more dominant in the Warm-Moist (WM) and Cool-Moist (CM) Groups. The midseral open (MSO) phase is the least common in the WM, Warm-Dry (WD), and Shallow-Dry (SD) Groups while the early seral (ES) phase is the least common in the CM Group.



Figure 2.4. Amount of moderate and high quality seasonal sage-grouse habitat. Landscape amounts are based on the proportions of each group as determined from soil surveys in the Malheur High Plateau major land resource area. Not all groups provide all types of seasonal habitat.

Chapter 3: Modeling Current Conditions for Sage-Grouse Habitat in Southeastern Oregon Using a State-and– Transition Modeling Framework

Abstract. Sagebrush (Artemisia spp. L.) ecosystems and habitat for greater sage-grouse (Centrocercus urophasianus) in the Intermountain West are considered imperiled due to a combination of natural and human-related disturbances. We developed two state-andtransition models using a combination of scientific literature and data for climate (temperature, precipitation, and snow), soils (soil surveys and ecological site descriptions), fire occurrence records, to examine how fire, insect outbreak, drought, snow mold, voles, freezekill, livestock grazing, juniper expansion, cheatgrass, and vegetation treatments might influence the abundance and quality of seasonal habitat for in the Malheur High Plateau major land resource area in southeastern Oregon. Both models included multiple ecological states and up to four community phases in each state. In drier sagebrush plant associations, typically characterized by Wyoming big sagebrush (Artemisia tridentata Nutt. spp. wyomingensis Beetle & Young) communities, overgrazing by domestic livestock during severe drought and postfire seeding success were drivers of the abundance of the different states and community phases. In moister sagebrush associations, typically characterized by mountain big sagebrush (A. tridentata Nutt. spp. vaseyana (Rydb.) Beetle) communities, juniper expansion and vegetation treatments were drivers. Fires that produced mosaic burn patterns were also a strong influence on fire rotations in moister sagebrush associations. In both models, changes in fuel continuity and loading resulted in fire rotations similar to those reported in the literature with an average fire rotation of 12 years in drier sagebrush associations and an average fire rotation of 81 years in moister sagebrush associations.

The simulated amount of sage-grouse nesting, brood-rearing and wintering habitat was considerably less than that recommended by sage-grouse biologists. Habitat suitability was also lower due to the influence of cheatgrass and juniper. Model results in the drier sagebrush associations indicated postfire seeding success alone was not sufficient to limit the area of cheatgrass domination. The incidence of overgrazing by domestic livestock during severe drought was also a key influence on sage-grouse habitat abundance and quality. In the moister sagebrush, results indicated that at least 10% of the juniper expansion area should be treated annually to keep juniper in check and maintain sage-grouse habitat. Regardless, juniper seedlings and saplings would remain abundant. Keywords: State-and-transition, sage-grouse, sagebrush, juniper, cheatgrass, livestock grazing, vegetation treatments

3.1 Introduction

Since the mid-19th century, domestic livestock grazing, introduction of non-native invasive plants (e.g., cheatgrass [*Bromus tectorum* L.]), changes in wildfire occurrence, conversion of sagebrush-steppe to western juniper (*Juniperus occidentalis* var. *occidentalis* Hook.) woodlands (Miller and Wigand 1994), and a history of treatments to eradicate or modify sagebrush (*Artemisia* spp. L.) communities have produced broad-scale alterations of sagebrush ecosystems throughout the western United States (Bunting et al. 2002, Hemstrom et al. 2002, Connelly et al. 2004). Prior to the mid-1980s, vegetation management in the Intermountain West emphasized the reduction of sagebrush to increase livestock forage production (Pechanec et al. 1944, Frischknecht and Bleak 1957, Cooper and Hyder 1958, Johnson 1958, 1969, Harniss and Murray 1973, Bartolome and Heady 1978, Britton et al. 1981). Since the mid-1980s, the focus has shifted to sagebrush restoration on public lands and many private lands to address declining populations of several sagebrush-obligate

species, most notably greater sage-grouse (*Centrocercus urophasianus*). A number of studies have examined the effects of natural and human-related disturbances in sagebrush dynamics a single factor at a time. A few studies have examined the interactions between natural and human-related disturbances (Bunting et al. 2002, Hemstrom et al. 2002) but these studies relied mostly on expert opinion as a basis for the disturbance probabilities and were very broad in scope.

Land managers charged with developing long-term management plans to address these challenges need a tool to evaluate alternative strategies and the potential impact of such strategies on plant communities and various ecosystem services at the landscape scale. Key questions they must contend with and that this study examines include: 1) what influence might domestic livestock grazing have on sage-grouse habitat over the long-term as it interacts with climate and other disturbances, 2) how important is active vegetation management in retaining sage-grouse habitat, and 3) how might the combination of domestic livestock grazing and vegetation management affect sage-grouse habitat availability with respect to the amount recommended by sage-grouse biologists. We investigated the how the interaction of natural disturbances and current management activities might affect the quantity and quality of sage-grouse habitat in southeastern Oregon to answer the questions posed above. Better understanding of these interactions can guide or inform the development and evaluation of alternative approaches to land management. In particular, increased understanding can highlight where changes in the type or intensity of management activities may be warranted to protect or restore sage-grouse habitat.

We selected the 4 million ha Malheur High Plateau major land resource area (NRCS 2006) in southeastern Oregon to construct state-and-transition models and examine the combined

effects of natural disturbances and typical management activities on sage-grouse habitat quantity and quality. We divided the sagebrush plant communities of the Malheur High Plateau into four ecological units based on estimates of site productivity in ecological site descriptions. Ecological site descriptions consist of the physiographic and soil features, climate, characteristic and historical plant communities including estimated annual productivity in different types of years, and site suitability for different land uses such as livestock grazing, wildlife, recreation, and so forth for each site type. We used the Vegetation Dynamics Development Tool (VDDT) (ESSA Technologies Ltd. 2007) to construct state-and-transition models for the two ecological units where most management activities take place. To create state-and-transition models that were more objective and less dependent on expert opinion, we based as many probabilities of events as feasible on measured climatic factors.

3.2 Study Area

We selected the Malheur High Plateau (figure 3.1) because it is considered a stronghold for sage-grouse (Connelly et al. 2004) and much of the land is managed by the Bureau of Land Management and U.S. Fish and Wildlife Service with a mandate to protect, enhance, and restore habitat for sagebrush obligate species. Much of the study area lies between 1190 m and 2105 m elevation, with Steens Mountain reaching 2967 m. The area consists of interspersed hills, buttes, isolated mountains and north-south trending fault-block mountains. The Malheur High Plateau contains no major rivers and little surface water but has numerous springs, shallow lakes and playas. Perennial streams and small rivers are mostly located on the periphery of the province. Using soil survey data

(http://soils.usda.gov/technical/classification/osd/index.html), we calculated that 98 percent of the soils in the sagebrush ecological types of the Malheur High Plateau were Mollisols and Aridisols. Soils are primarily loamy to clayey, well-drained and shallow to moderately deep on uplands, and poorly to well-drained and very deep in basins. Soil temperature regimes range from mesic in the lower elevations, frigid in the mid-elevations, and cryic generally above 2000 m elevation. Soil moisture regimes range from xeric at mid and upper elevations to aridic at mid and lower elevations. The average annual precipitation ranges from 105 mm to 305 mm over most of the area, with Steens Mountain receiving as much as 1450 mm at its upper elevations. Winter and spring are the wettest periods with most precipitation falling in November, December, January and May, while summer is the driest. January is the coolest month, averaging -2°C, and July the warmest, averaging 19°C. Sagebrush-steppe is the dominant vegetation type with salt desert shrub (*Sarcobatus vermiculatus* (Hook.) Torr.-*Grayia spinosa* (Hook.) Moq.) on saline soils in basins, western juniper expanding out of rockier areas, and aspen (*Populus tremuloides* Michx.) at the highest elevations.

3.3 Methods

We sorted sagebrush ecological sites for the Malheur High Plateau

(http://esis.sc.egov.usda.gov) into four groups based on perennial grass productivity for low, average and high productivity years, assuming productivity provided a measure of potential growth and recovery rates. In this study, we constructed VDDT models only for the two groups where most active management occurs. Each model consisted of several states, defined as a suite of temporally related plant communities and their associated dynamic soil properties (Bestelmeyer et al. 2009), with one or more community phases in each state. A community phase consists of a distinctive plant community and its associated dynamic soil property levels that occur over time within a given state (Bestelmeyer et al. 2009).

We searched the literature to identify which disturbance factors were important determinants of vegetation trajectories and sage-grouse habitat, selecting fire, insect outbreaks, drought severe enough to kill shrubs (shrub-drought), freezekill, snow mold, vole outbreaks, domestic livestock grazing, postfire seeding, juniper expansion, and juniper vegetation treatments (prescribed burning, cutting, and cutting and burning) (see also Chapter 2). We also used our literature search to determine what climatic factors were important to the likelihood of fire (Britton et al. 1981, Brown 1982, Bunting et al. 1987, Whisenant 1990, Peters and Bunting 1994), shrub-drought (Keen 1937, Graumlich 1987, Cook et al. 1997, Cayan et al. 1998, Cook et al. 1999, Gedalof and Smith 2001, Cook et al. 2004, Knapp et al. 2004, Stahle et al. 2007), freezekill (Hanson et al. 1982, Walser et al. 1990, Hardy et al. 2001, DeGaetano and Wilks 2002), snow mold (Sturges 1986, 1989), vole outbreaks (Mueggler 1967, Frischknecht and Baker 1972, Parmenter et al. 1987), overgrazing and drought-grazing (Craddock and Forsling 1938, Houston 1961, Brotherson and Brotherson 1981, Angell 1997, Adler et al. 2005), postfire seeding success (Klomp and Hull 1972, Hull 1974, Cox and Anderson 2004, Thompson et al. 2006), and the initiation of juniper expansion (Miller and Wigand 1994, Romme et al. 2009).

We obtained monthly precipitation and temperature data from 1895-2009 for Oregon Climate Division 7 (available at http://www7.ncdc.noaa.gov/CDO/CDODivisionSelect.jsp) and snow data from 1967-1996 for the Reynolds Creek Experimental Range (Hanson et al. 2001, Marks et al. 2001). Although Reynolds Creek Experimental Range lies outside the Malheur High Plateau, it has a similar climate and provided more detailed information on snowpack than that available for Oregon Climate Division 7. We summarized monthly and seasonal means (temperature) and medians (precipitation) using a temperature-based definition of winter and summer that better matches plant phenology and hydrological cycles than the typical 3-month definitions (Neilson et al. 1992). We estimated mean and standard deviation for snowpack duration, snow depth, and snowmelt date for the highest elevation station on the experimental range. We applied Neilson et al.'s (1992) biome rules to each water-year for Oregon Climate Division 7, defining a low production year as one that would produce a shrubland and a severe drought year as one that would produce a desert. We also used climatic data to estimate the probabilities for fire, freezekill, snow mold, severe winters for voles, overgrazing, drought-grazing, postfire seeding success, and the initiation of juniper expansion. We developed the probabilities for insect outbreaks using Pandora moth (*Coloradia pandora* Blake) dynamics as this forest defoliator's dynamics seemed most similar to what is known about the sagebrush defoliator, aroga moth (*Aroga websteri* Clark) (Gates 1964, Hall 1965, McBrien et al. 1983, Hsaio 1986, Speer et al. 2001). We assigned sagegrouse seasonal habitat quality ratings based on descriptors in the sage-grouse literature (Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005).

We constructed two state-and-transition models using VDDT and created a Monte Carlo Multiplier file to incorporate variability in both the occurrence and impact of fire, shrubdrought, insects, voles, and postfire seeding. In VDDT, the Monte Carlo Multiplier file type contained a random set of disturbance probability multipliers. For random disturbances, these multipliers were applied to the base disturbance probability, either increasing or decreasing the probability that the disturbance would affect a given spot. For cyclical disturbances, these multipliers affected whether the disturbance type occurs or not within the range of years defining the maximum and minimum interval between outbreaks and the minimum and maximum number of years within each outbreak (ESSA Technologies Ltd. 2007). We treated fire, freezekill, snow mold, overgrazing, drought-grazing, postfire seeding, juniper expansion, and vegetation treatments as random events and shrub-drought, insect outbreaks, and voles as cyclical events. We ran 50 simulations for 500 years each, saved the area in each community phase every 10 years, and estimated the average annual area affected by each disturbance type. To allow ample time for the models to come into dynamic equilibrium, we analyzed model outputs for only the last 250 years of the 500-year simulation runs. We compared the average annual percentage of area affected by fire in each state-and-transition model to estimated current fire frequencies published in the literature as a form of model validation.

Because the results data were not normally distributed with equal variances, we tested for differences in the abundance of community phases and states using the Kruskal-Wallis one-way ANOVA on ranks in SigmaPlot 11.0 (Systat Software 2008). When significant differences were found, we tested for differences between model variants using the Tukey test with a significance level of <0.05. We examined the importance of: 1) the initial proportions of the community phases used at the beginning of each run, 2) the effect of adding overgrazing, drought-grazing, postfire seeding, juniper expansion, and vegetation treatments to the natural disturbances of fire, shrub-drought, insect outbreaks, freezekill, snow mold, and vole outbreaks, and 3) model sensitivity to variations in the probabilities for mosaic fire, livestock grazing, postfire seeding, juniper expansion and vegetation treatments.

To test the importance of the initial proportions of community phases, we compared model results where the landscape initially had an equal proportion of community phases to results where all area was initially assigned to the earliest community phase of the reference state. We tested the importance of each added disturbance by first running each model with only the natural disturbances, then by adding each new disturbance type singly (e.g., overgrazing or postfire seeding) and in combinations (e.g., overgrazing and postfire seeding). We compared the resulting abundance of each community phase and state to the full model. We also varied the probability of livestock grazing, postfire seeding, juniper expansion and

vegetation treatments between 0 and 2 times the initial probability and compared the differences. The final series of tests consisted of varying the probability of mosaic fire between 0 and 2 times the initial probability to examine our assumption concerning the effect of livestock grazing on fire probabilities. We used the same Monte Carlo sequences for all runs and followed the same procedures as in the initial model runs.

We assigned quality ratings for sage-grouse seasonal habitat to each community phase in each state-and-transition model (table 3.1). Seasonal habitat types included lekking, prelaying hens, nesting, early and late brood-rearing, and wintering. The rating for each community phase was based on expected amounts of sagebrush cover, horizontal cover, and forb abundance. We estimated the amount of sage-grouse seasonal habitat in each sagebrush group and the combined groups and compared the landscape amount to the recommendations from sage-grouse biologists (Connelly et al. 2000). Terminology in this study follows that used in the state-and-transition literature (Bestelmeyer et al. 2009). Plant nomenclature follows the U.S. Department of Agriculture online PLANTS database (http://plants.usda.gov).

3.4 Model Design

The two groups of interest for this study were the Cool-Moist Sagebrush Group (CM Group), which produces at least 672 kg ha⁻¹ yr⁻¹ of perennial grass in high and average productivity years, and the Warm-Dry Sagebrush Group (WD Group), which produces at least 672 kg ha⁻¹ yr⁻¹ of perennial grass in high productivity years only. Although both groups include more than one subspecies of sagebrush the most common subspecies in the CM Group was mountain big sagebrush (*A. t.* Nutt. ssp. *vaseyana* (Rydb. Beetle), and Wyoming big sagebrush (*A. t.* Nutt. ssp. *wyomingensis* Beetle & Young) in the WD Group. Sage-grouse use
both groups at various times of the year for breeding, brood-rearing, and wintering. We constructed a state-and-transition model for each group.

Both state-and-transition models included fire, insect outbreaks, shrub-drought, overgrazing, drought-grazing, and postfire seeding. The CM Group also included freezekill, snow mold, voles, juniper expansion, and juniper vegetation treatments (prescribed burning, cutting, and cutting and burning). The reference state in both models included four community phases: 1) Early Seral (ES) dominated by grasses and forbs, 2) Midseral Open (MSO) dominated by grasses and forbs, but with sagebrush now present, 3) Late Seral Open (LSO) with sagebrush, grasses and forbs co-dominant, and 4) Late Seral Closed (LSC) with sagebrush dominant and grasses and forbs in decline. We used sagebrush cover to define the breakpoints between each community phase, with different breakpoints in each model.

WD Group. Table 3.2 and figure 3.2 depict the basic structure of our model for the WD Group. We used a phase approach similar to that used by Miller et al. (2005) for western juniper to structure the state-and-transition model into four states – Minimum Cheatgrass, Phase I Cheatgrass, Phase II Cheatgrass, and Phase III Cheatgrass (figure 3.2). The Minimum Cheatgrass state was the reference state. Cheatgrass was present in the Minimum Cheatgrass state but shrubs and perennial herbs were the dominant influence on ecological processes. Cheatgrass and perennial grasses co-dominated ecological processes in the herbaceous layer of the Phase I Cheatgrass state. Cheatgrass state. Finally, in the Phase III Cheatgrass state, cheatgrass dominated ecological processes.

We also defined four general community phases for the Minimum Cheatgrass, Phase I Cheatgrass, and Phase II Cheatgrass states. Grasses and forbs dominated in the early seral with sagebrush cover less than 1% in the (ES) community phase. Sagebrush was subdominant, with 1-8% cover, and grasses and forbs remained dominant in the midseral open (MSO) community phase. Sagebrush at 8-20% cover, grass, and forbs were codominant in the late seral open (LSO) community phase. Sagebrush at greater than 20% cover was dominant in the late seral closed (LSC) community phase. All four community phases were present in the Minimum Cheatgrass and Phase I Cheatgrass states; the Phase II Cheatgrass state lacked the ES community phase. Phase III Cheatgrass consisted of a single community phase, which we designated the same as the state name.

We assumed cheatgrass was present throughout the WD Group (Hubert 1955, Tisdale et al. 1965, Hosten and West 1994, Kindschy 1994, Tausch et al. 1994b, Bradford and Laurenroth 2006). Fire (Young and Evans 1978, Whisenant 1990, D'Antonio and Vitousek 1992, Billings 1994, Pellant 1996, Bunting et al. 2002), overgrazing (Griffiths 1902, Cottam and Evans 1945, Pechanec and Stewart 1949, Mueggler 1950, Reid et al. 2008) and drought-grazing (Klemmedson and Smith 1964, Tisdale et al. 1965, Kindschy 1994, Tausch et al. 1994b, Bradford and Laurenroth 2006) promote cheatgrass dominance.

CM Group. Table 3.3 and figure 3.3 display the basic structure of our model for the CM Group. Vegetation management included prescribed burning, cutting juniper and burning, cutting juniper, and postfire seeding. We structured the CM Group model into the No Juniper, Juniper Expansion, and Phase III Cheatgrass states (figure 3.3). The No Juniper state was the reference state, where sagebrush and perennial grasses controlled site resources. This state contained four community phases with the same designations as the WD Group but with

different sagebrush cover. Grasses and forbs dominated the ES community phase with sagebrush cover of less than 1%. Sagebrush at 1-10% cover was subdominant to grasses and forbs in the MSO community phase. Sagebrush cover was 10-30% in the LSO community phase with grasses and forbs generally co-dominant. Sagebrush cover in the LSC community phase exceeded 30%.

The Juniper Expansion state included Phase I Juniper (J1), Phase II Juniper (J2), Phase III Juniper (J3) and old growth juniper (OG) community phases and juniper controlled site resources. The J1, J2, and J3 community phase descriptions followed those in Miller et al. (2005). The OG phase in this model referred to old juniper woodland that would develop from juniper expansion into the CM Group, not the old growth juniper woodland that was the initial source of juniper expansion. Stands with 75% or more of trees exhibiting one or more morphological characteristics associated with old trees comprised the OG community phase (Waichler et al. 2001). Since the duration of each phase depends largely on site characteristics and site potential, we used the intermediate level age data from Johnson and Miller (2006) to model movement from the J1 community phase to the OG phase in the absence of disturbance. The Phase III Cheatgrass state, which consisted of a single community phase the same as in the WD Group, occurred in the warmer, drier portion of the CM Group where cheatgrass could assume control of site resources, representing the boundary with the WD Group.

Wildfire. In each state-and-transition model, we estimated the initial probability of a fire using a combination of precipitation and temperature to define when sufficient fuel would be present, reduced that probability by 25% to account for limited ignitions. We divided fire into homogeneous burn patterns (stand-replacing fire) and heterogeneous burn patterns (mosaic

fire) with different probabilities for each type. In the reference states of both state-andtransition models and in the Phase I Cheatgrass state of the WD Group, we reduced the probability of mosaic fire still further to account for reduced fuel loading and fuel continuity from livestock grazing. In the WD Group, we increased fire probability in the Phase II and Phase III Cheatgrass states to reflect the continuous fuels cheatgrass provided (Britton et al. 1981, Whisenant 1990). We assumed that sufficient cheatgrass fuel was present in all years except in very low production years and reduced the resulting probability by 25% to account for the presence of sufficient fuel but absence of ignition. In the CM Group, we assumed only stand-replacing fire was possible in the Juniper Expansion state and fire probability in the Phase III Cheatgrass state is the same as in the WD Group.

In the WD Group, we assumed stand-replacing fire in the Minimum Cheatgrass state did not result in cheatgrass dominance until a location reached age 100 in the LSC community phase (Cline et al. 1977, Young and Evans 1978, Hosten and West 1994, Chambers et al. 2007, Davies et al. 2007, Davies et al. 2008). A mosaic fire in the ES and MSO community phases had equal chances of remaining in the Phase I Cheatgrass state or transitioning into the Phase III Cheatgrass state due to interannual variability in the production of both cheatgrass and perennial bunchgrasses. Stand-replacing fire in all community phases of the Phase I and Phase II Cheatgrass states resulted in a transition to the Phase III Cheatgrass state, after which fire maintained that state (Knapp 1996, Bradford and Laurenroth 2006, Reid et al. 2008). If a site in the Phase III Cheatgrass state escaped fire for at least 20 years, sagebrush reestablished and transitioned back to the Phase II Cheatgrass state and MSO community phase (Young and Evans 1973, Peters and Bunting 1994, Mata-González et al. 2007).

In the CM Group, mosaic fire in the No Juniper state reduced sagebrush cover, but did not result in a shift to a different community phase. In both the No Juniper and Juniper Expansion states, stand-replacing fire shifted the site into the ES phase of the No Juniper state with one exception. In the J3 and OG phases of the Juniper Expansion state, we included a 2% probability that stand-replacing fire would transition the site into the Phase III Cheatgrass state, based on the proportion of the CM Group where cheatgrass could assume site dominance.

Overgrazing and Drought-Grazing. In both state-and-transition models, livestock grazing included both cattle (Greer 1994, 2000) and feral horses since the impacts from these two species were difficult to separate (Hansen and Reid 1975, Olsen and Hansen 1977, McInnis and Vavra 1987). We assumed the predominant grazing system used was deferred-rotation in a 4-pasture setup, that the number of assigned animal unit months was based on moderate utilization of the grass production in an average year, and that animal numbers were not adjusted based on actual grass production. We modeled two levels of excessive grazing in both models. Overgrazing occurred in a low production year, equivalent to a single-year or mild drought. We assumed utilization levels were high enough to cause a low level of mortality in grasses and forbs, but that recovery was relatively quick (Julander 1945, Van Poollen and Lacey 1979, Holechek et al. 2004). Drought-grazing occurred in a very low production year, equivalent to a multi-year or severe drought (see Methods) with the assumption that all effects occurred in the latter stages of the drought. We assumed the combination of severe drought and very high utilization resulted in significant damage to grasses and forbs with moderate to high mortality in these life forms, fragmentation of some grass bunches, and slow recovery rates (Craddock and Forsling 1938, Julander 1945, Van Poollen and Lacey 1979, Butler and Briske 1988, Angell 1997, Holechek et al. 2004, Loeser et

al. 2007). We reduced probabilities of both overgrazing and drought-grazing by 25% since one pasture would not be grazed at a time it would have the most impact on plant vigor under our assumed grazing system (Holechek et al. 2004).

In both state-and-transition models, overgrazing increased sagebrush canopy cover by reducing competition for water and nutrients, accelerating the rate of aging in a given community phase. Drought-grazing in the WD Group increased both sagebrush cover and cheatgrass abundance (Julander 1945, Pechanec and Stewart 1949, Paulsen and Ares 1961, Billings 1994, Tausch et al. 1994a, Loeser et al. 2007), causing a transition to another state with the same community phase and preserving the relative age of the phase in the transition. Drought-grazing in the CM Group accelerated aging at twice the rate of overgrazing alone, facilitating very rapid movement into the community phases where juniper expansion becomes a possibility (Romme et al. 2009).

Postfire Seeding. In both state-and-transition models, seeding desirable species occurred after a wildfire to reduce or avoid dominance by invasive species (Robichard et al. 2000, Eiswerth et al. 2009), but only a narrow window of opportunity exists to keep cheatgrass from attaining or retaining site dominance (Getz and Baker 2007, Keeley and McGinnis 2007). We assumed that appropriate planting techniques and seed mixes were used (Holechek et al. 2004), and used fall and May-June precipitation as a primary indicator of seeding success. We assumed that for every three hectares that had successful establishment in high precipitation years, one hectare successfully established in adequate years and one-tenth hectare successfully establishment during low precipitation years. We included postfire seeding only in the Phase III Cheatgrass state where success resulted in a transition to the ES community phase of the Minimum Cheatgrass state or the No Juniper state.

Shrub-Drought and Insect Outbreaks. Shrub-drought and insect outbreaks occurred in both state-and-transition models using the same assumptions and parameters. We based the probability of shrub-drought on estimates of the frequency of droughts comparable to the 1930s drought (Cook et al. 2004, Stahle et al. 2007). Shrub-drought affected all vegetation states, except Phase III Cheatgrass, in the WD Group and affected the No Juniper state in the CM Group. In the MSO and LSO community phases, shrub-drought reset the relative age, while in the LSC community phase it shifted the community phase back to LSO. We assumed insect outbreaks occurred only in the LSO and LSC community phases of all states (Hsaio 1986) and resulted in a shift to an earlier community phase.

Juniper Expansion. Juniper expansion occurred only in the CM Group. We assumed that all locations were equally exposed to juniper seed sources (Chambers et al. 1999, Miller et al. 2005). Initiation of the J1 community phase occurred when a year of adequate spring moisture for germination was followed by a year with adequate winter and spring moisture (Miller and Wigand 1994, Romme et al. 2009). Most junipers establish under sagebrush (Burkhardt and Tisdale 1976, Eddleman et al. 1994, Miller and Rose 1995, Chambers et al. 1999, Zophy 2006), but we found no studies that established a minimum threshold of shrub cover needed. We assumed the J1 phase begins in the LSO and LSC community phases of the NJ state. Once expansion began, establishment was relatively continual (Burkhardt and Tisdale 1976, Chambers et al. 1999, Soulé et al. 2004), so we used empirically measured juniper tree age to determine the shift into subsequent community phases (Johnson and Miller 2006). **Vegetation Treatments.** Prescribed burning, cutting, and cutting plus burning occurred only in the CM Group. Cutting treatments reduced large juniper tree populations in the Juniper Expansion state while prescribed burning affected all tree sizes. We assumed a 10% overall chance of any combination of cutting and burning with an equal chance for any given treatment. Based on conversations with BLM fuels managers, we included prescribed burning in the J1 community phase, all vegetation treatments in the J2 community phase, and the combination of cutting and burning in the J3 and OG phases. We treated prescribed burning as a stand replacing fire, shifting the site to the ES community phase in the No Juniper state. Cutting and prescribed burning had the same outcome as prescribed burning alone. Cutting shifted the site back to the J1 phase.

Freezekill, Snow Mold, and Voles. Freezekill, snow mold, and vole damage occurred only in the CM Group, No Juniper state, and were restricted to those areas where deeper snowpacks are characteristic. Freezekill and snow mold reduced sagebrush cover within the MSO community phase without conversion to an earlier community phase, and moved both the LSO and LSC phases back to the MSO phase (Hanson et al. 1982, Nelson and Tiernan 1983). Vole damage reduced sagebrush cover in the MSO and LSC community phases without conversion to an earlier community phase and moved the LSO phase back to the MSO phase.

3.5 Results

In both state-and-transition models, the initial proportion among community phases produced no significant difference on the median abundance of community phases, states, and disturbance frequencies. All subsequent results were based on an equal proportion of community phases at the beginning of each model run. The different disturbance variables had varying effects on the two sagebrush groups modeled.

WD Group. The WD Group was very sensitive to the probabilities of drought-grazing and postfire seeding success but only somewhat sensitive to the probabilities of overgrazing and mosaic fire. Drought-grazing in the absence of postfire seeding resulted in cheatgrassdominated states over the entire landscape occupied by this group. Without drought-grazing, cheatgrass-dominated states occupied only a minor portion of the WD Group. The absence of overgrazing from the full model produced cheatgrass-dominated states over slightly more than one third of the area where the WD Group occurs. Doubling the probability of postfire seeding success still resulted in just under half of the WD Group in cheatgrass-dominated states. Less than 10% of the WD Group supported the LSO and LSC community phases except when drought-grazing was absent or less than half the modeled probability. Overgrazing in the absence of both drought-grazing and postfire seeding resulted in nearly two-thirds of the WD Group area in cheatgrass-dominated states. The reference state occupied only about a third of the WD Group with drought-grazing present and nearly all of the WD Group landscape with drought-grazing absent. Varying the probability of mosaic fire produced only minor changes in the abundance of community phases, states, and frequency of other disturbance, and had no effect on fire rotation length.

CM Group. The CM Group was very sensitive to the probabilities of juniper expansion and vegetation treatments, including postfire seeding success, and somewhat sensitive to the probabilities of overgrazing, drought-grazing, and mosaic fire. The JE state occupied slightly over half of the landscape where the CM Group occurred when juniper expansion was included and vegetation treatments excluded from the model, with most of this area in the J3,

OG, and cheatgrass community phases. Adding drought-grazing and overgrazing, either singly or in combination, to juniper expansion reduced the abundance of the J3, OG and cheatgrass phases relative to when both were absent. The combination of both vegetation treatments at our assumed rates and juniper expansion reduced the extent of the JE state to less than half of the CM Group and the J3, OG, and cheatgrass phases were absent. Including vegetation treatments and excluding juniper treatments resulted in the removal of juniper from the CM Group within 150 years.

In most model variants, early community phases dominated the No Juniper state while a large proportion of the older sagebrush community phases had juniper (J1 community phase). Generally, the abundance of juniper in the CM Group increased as the probability of juniper expansion increased or the probability of vegetation treatments decreased. Doubling the probability of vegetation treatments increased the presence of the reference state somewhat and reduced the presence of the juniper state nearly half. Doubling the probability of juniper expansion reduced the presence of the reference state and increased the presence of the juniper state only slightly. Varying the probability of mosaic fire had only minor effects on the abundance of community phases, states, and some disturbance frequencies but a large effect on fire rotation. The absence of mosaic fire resulted in a fire rotation of 182 years, while doubling the probability resulted in a fire rotation of 50 years.

Sage-Grouse Habitat. We summarized results for sage-grouse habitat using only community phases rated as moderate and high quality (Table 3.1). In the full model, 72% of the WD Group provided sage-grouse habitat, primarily for lekking and pre-laying hens, while 94% of the CM Group provided habitat, primarily for pre-laying hens, late brood-rearing, and wintering (Figure 3.4). Within the study area, the WD Group occurred on about 79% of the

area while the CM Group occurred on about 21% of the area. Taking into account these proportions, 62% of the combined groups provided breeding habitat, 30% provided brood-rearing habitat, and 22% provided wintering habitat over the long-term and under current management conditions (Figure 3.4). Sage-grouse biologists recommended that at least 80% of the landscape function as breeding habitat and wintering habitat and at least 40% function as brood-rearing habitat (Connelly et al. 2000). Much of the habitat provided was moderate quality and about half would be rated as functioning at-risk. Even within the breeding habitat, projected conditions were skewed towards lekking and pre-laying hen habitat.

Examining the abundance of the different types of seasonal habitat in the model variants highlighted four different combinations with similar results in both the WD Group (Table 3.4) and CM Group (Table 3.5). In both models, we included a variant with just the historical disturbances for comparative purposes, although these results differ somewhat from those in Chapter 2. In the WD Group, the differences are due to the absence of pronghorn browsing and the inclusion of cheatgrass community phases and states. In the CM Group, they are due to the presence of juniper and cheatgrass community phases and states and possibly the reduced probability of mosaic fire.

The absence of postfire seeding or the combined absence of postfire seeding and overgrazing from the full model resulted in no sage-grouse habitat in the WD Group (Table 3.4). Adding overgrazing alone to the historical set of disturbances resulted in little or no difference in sage-grouse habitat abundance compared to just the historical disturbances. Similarly, the absence of overgrazing from the full model also resulted in little or no difference in habitat abundance compared to the full model also resulted in little or no difference in habitat abundance compared to the full model. Lastly, the absence of drought-grazing from the full model.

amounts of seasonal sage-grouse habitat. Little or no lekking habitat occurred in the absence of postfire seeding. The absence of drought grazing resulted in considerably more nesting, early brood-rearing, and wintering habitat, and considerably less lekking habitat with mixed results in the abundance of pre-laying hen habitat, depending on whether postfire seeding was included or not.

In the CM Group, adding juniper expansion to the historical disturbances, with or without overgrazing and drought-grazing, or removing vegetation treatments from the full model resulted in little sage-grouse habitat of any type. Adding vegetation treatments at our assumed level to the historical disturbances, with or without overgrazing and drought-grazing, or removing juniper expansion from the full model resulted in similar amounts of seasonal habitat as the historical disturbances alone, and produced the greatest amount of nesting, brood-rearing, and wintering habitat. Adding both juniper expansion and vegetation treatments at our assumed level to the historical disturbances or removing either overgrazing or drought-grazing from the full model predicted similar abundances of sage-grouse habitat as the full model. These combinations produced high levels of late brood-rearing habitat and moderate levels of breeding, early brood-rearing and wintering habitat. Lastly, adding overgrazing and drought-grazing to the historical disturbances, either singly or in combination resulted in similar abundances of the different types of seasonal sage-grouse habitat. These combinations also produced high levels of late brood-rearing habitat, moderate to high levels of nesting, early brood-rearing, and wintering habitat, but relatively low levels of pre-laying hen habitat. In all model variants that included juniper expansion, a large proportion of the nesting, early brood-rearing, and wintering habitat would include juniper seedlings and saplings.

3.6 Discussion

Current trends in the Malheur High Plateau appear to be consistent with the predicted dynamic equilibrium conditions in both state-and-transition models, particularly in the CM Group. The resulting dynamic equilibrium in both the WD and CM models does not fully represent actual current conditions as observations of the Malheur High Plateau indicate both groups are still within a transitional period that began around 1850. Both our models took 100-150 years to reach a dynamic equilibrium and included the presence of all community phases as part of the initial conditions. The lack of sensitivity to the initial proportion of community phases is useful when the initial distribution is not known and the simulation length is long enough to allow a dynamic equilibrium to develop.

<u>3.6.1 Warm-Dry Sagebrush Group</u>

Drought-grazing vs. Overgrazing. Our model simulations indicate that within the WD Group even a limited amount of drought-grazing (once every 23 years), coupled with fire, drove the system into greater cheatgrass domination and greater loss of sage-grouse habitat, whereas occasional overgrazing (once every 12 to 13 years) outside of severe drought periods may not be as detrimental as commonly perceived. Other studies reported increased shrub cover and annual species and reduced perennial grass cover, numbers, or productivity under very intensive grazing (Griffiths 1902, Pickford 1932, Shinn 1977, Van Poollen and Lacey 1979, Brotherson and Brotherson 1981), particularly when coupled with drought (Craddock and Forsling 1938, Julander 1945, Loeser et al. 2007). That portion of the Malheur High Plateau in northern Nevada (not included in our study area), along with the Snake River Plain and parts of the Owyhee High Plateau and Humbolt Area (Pellant 1996, Connelly et al. 2004) appear to have experienced this outcome. While intensive grazing similar to drought-grazing has clearly been implicated in the transition to cheatgrass-dominated states (Pickford 1932,

Brotherson and Brotherson 1981, Hosten and West 1994, Adler et al. 2005, Díaz et al. 2007, Loeser et al. 2007) the exact mechanism is unknown and most studies couple grazing and fire when discussing this transition.

Several studies reported spatial differences in grazing intensity that affect plant community composition (Van Poollen and Lacey 1979, Weber et al. 1998, Holechek et al. 2004, Adler et al. 2005, Briske et al. 2008). We found no studies that examined the effects of temporal differences in grazing intensity under modern stocking levels; however, our model results appear to be consistent with these studies on the spatial differences. Essentially in our models, damage during low production years, when utilization would exceed moderate levels, recovered during high production years, when utilization would be the equivalent of light grazing (Van Poollen and Lacey 1979, Anderson and Holte 1981, Milchunas and Laurenroth 1993, Anderson and Inouye 2001, Chambers et al. 2007). The literature is clear that grazing intensity matters more than grazing system (Holechek et al. 2004, Briske et al. 2008).

Postfire Seeding. Our model results also indicated that postfire seeding was critical for limiting cheatgrass domination in the WD Group, but postfire seeding alone was not sufficient to restore sage-grouse habitat. Instead, our model indicated that restoration of sage-grouse habitat also required reduction (less than once every 40 years) or elimination of all drought-grazing in addition to relatively high postfire seeding success. Evans and Young (1978) reported high seeding failure rates where grazing utilization was very high shortly after seeding. In the absence of active restoration efforts cheatgrass can retain site dominance where the native perennial grasses have been lost from the site and seedbank (Young and Evans 1973, Billings 1994, Bollinger and Perryman 2008). Frequent fire, along with the loss of vesicular arbuscular mycorrhizae and high nitrogen availability that often follows a fire and

grazing pressure on remaining perennial grasses, favor continued dominance of cheatgrass (Robertson and Pearse 1945, Pyke 1986, 1987, McLendon and Redente 1994, Knapp 1996). Although we could not directly incorporate the other factors besides fire, our results appeared to be consistent with these findings.

Both Eiswerth et al. (2009) and Boyd and Davies (2010) suggested that postfire seeding success was higher immediately after a fire that resulted in a transition from the Phase I or Phase II Cheatgrass states to the Phase III Cheatgrass state, and lower following subsequent fires in the Phase III Cheatgrass state. Cheatgrass alters soil physical and chemical properties such that the longer cheatgrass occupies a site, the lower the probability of postfire seeding success (Norton et al. 2004). Yet we lacked the information needed to estimate how the probability of postfire seeding success should change over time. Had we been able to adjust postfire seeding success rate based on the number of past fires in the Phase III Cheatgrass state, our modeled outcomes may have been different.

Fire Rotation. Our model simulations produced an estimated fire rotation of approximately 12 years for the WD Group, with approximately 64% of the landscape area cheatgrass dominated; similar to other published reports of fire frequencies in dry sagebrush areas dominated by cheatgrass (Whisenant 1990, Knapp 1996, Pellant 1996, Knick et al. 2003, Connelly et al. 2004, Bradford and Laurenroth 2006, Reid et al. 2008). No clear relationship has been established to directly compare fire rotation to fire frequency based on point data, but a common assumption is that estimates of fire rotation are generally longer than fire frequencies for the same landscape. Our modeled fire rotation for the WD Group was more consistent with the current conditions on the Snake River Plain, an area lower in elevation and somewhat drier than the study area. We may have overestimated the probability of stand-

replacing fire in the Phase II and Phase III Cheatgrass states or the probability of transition from the Phase I Cheatgrass state to the Phase II or Phase III Cheatgrass state.

Simulated vs. Observed Conditions. No inventory of current conditions on the Malheur High Plateau was available; however, observations of the study area indicated much less area dominated by cheatgrass even though cheatgrass has been present for over 100 years. One possible explanation for the differences between the observed conditions in the Malheur High Plateau and simulated conditions is land-use history. Southeastern Oregon was one of the last areas settled in Oregon or in the surrounding areas in western Idaho and northern Nevada due to the lack of surface water (Shinn 1977). Fewer areas were farmed in the late 1800s and early 1900s on the Malheur High Plateau than the surrounding areas. Soil profiles were disrupted over a much smaller area and less opportunity provided for cheatgrass to become entrenched over a wide area when farms were abandoned in the 1920s and 1930s (Piemeisel 1938, Stewart and Hull 1949). Although Griffiths (1902) reported severe degradation due to continual high utilization within 20 years of the onset of livestock grazing on Steens Mountain, cheatgrass was not specifically mentioned as present. The generally cooler and moister conditions on the Malheur High Plateau may have left the area less degraded and better able to recover once grazing pressure eased than in surrounding areas.

3.6.2 Cool-Moist Sagebrush Group

Juniper Expansion. In our model, unchecked juniper expansion led to a landscape dominated by the Juniper Expansion state and greatly reduced sage-grouse habitat. There are no known natural factors that would limit or halt juniper expansion in the CM Group under the current climate (Burkhardt and Tisdale 1969, 1976, Miller and Rose 1995, Knapp and Soulé 1998, Chambers et al. 1999, Soulé et al. 2004). We assumed that all locations in the

LSO and LSC community phases were equally vulnerable to the initiation of juniper expansion, however, vulnerability actually depends on proximity to seed-producing trees (Chambers et al. 1999). In field studies, the observed juniper spread pattern was in the form of bands expanding from a locus (Barney and Frischknecht 1974, Soulé et al. 2003, Miller et al. 2005).

Vegetation Treatments. Our model indicated that annually treating a relatively small portion of the Juniper Expansion state retained more sage-grouse habitat, although the J1 community phase was still abundant. However, our model did not deal with the spatially explicit aspects of vegetation treatments on the abundance of the Juniper Expansion state. Both Chambers et al. (1999) and Bates et al. (2005) reported the return to tree dominance was relatively rapid in small scale treatments, treatments that left the juniper seedbank more-or-less intact, and treatments that left either seedlings or surviving mature trees. Conversely, juniper recolonization was slow in very large treatments where no surviving seeds or nearby seed sources remained (Barney and Frischknecht 1974, Goodrich and Barber 1999, Bates et al. 2005).

In our model, when the probability of juniper expansion was less than the probability of vegetation treatments the abundance of the Juniper Expansion state declined and the reverse occurred when the probability of vegetation treatments was less that the probability of juniper expansion. We assumed a combined probability of treating 10% of the Juniper Expansion state annually, regardless of how much area it occupied, which resulted in juniper expansion and vegetation treatments occurring at about the same rate. However, there is likely an upper limit to the extent of area that can be treated each year that could result in a vegetation treatment rate that is lower than the juniper expansion rate, allowing for continued

spread. Indeed, the observed rate of juniper expansion suggests that our modeled probability of vegetation treatments may be too high (Gedney et al. 1999, Azuma et al. 2005).

At present, most herbicide use is legally enjoined on lands managed by the Bureau of Land Management in Oregon, resulting in just one effective vegetation treatment type (prescribed burning) in the J1 community phase in most of the study area. We modeled the probability of this treatment, prescribed burning, at 3.3% per year, consistent with limitations on prescribed burning due to a variety of factors, such as maintaining larger sagebrush to serve as sagegrouse habitat. The resulting abundance of the J1 community phases was consistent with the estimated current extent of this phase in the study area (Gedney et al. 1999, Azuma et al. 2005). The addition of another treatment option in the J1 phase, such as a chemical treatment specific to juniper and that preserved some presence of sagebrush, would likely decrease the abundance of the Juniper Expansion state and J1 phase and increase the abundance of the No Juniper state and its later community phases, thereby providing more high quality sage-grouse habitat.

Overgrazing and Drought-Grazing. Overgrazing and drought-grazing had little direct effect in our simulations of the CM Group. They increased the abundance of the later community phases in the reference state, which was only apparent by excluding juniper expansion in the model. By accelerating the movement into the later sagebrush community phases, it appears overgrazing and drought-grazing indirectly provided greater opportunities for juniper expansion to initiate, consistent with several studies and reviews that implicate past grazing practices in rapid juniper expansion (Burkhardt and Tisdale 1976, Miller and Wigand 1994, Chambers et al. 1999, Soulé et al. 2003, Romme et al. 2009).

Fire Rotation. Estimates of current fire frequencies for mountain big sagebrush communities are typically more than 100 years, based on fire scar studies at the sagebrushconifer ecotone (Miller and Rose 1999, Miller et al. 2001, Heyerdahl et al. 2006, Miller and Heyerdahl 2008). These studies showed a sudden decrease in fire frequency that corresponded to the onset of continual high intensity livestock grazing. Baker (2006) estimated a fire rotation of 70-200 years based on the growth rates of sagebrush and an assumption that late seral community phases dominated, although this estimate does not account for the effects of livestock grazing on fire occurrence and spread. Our modeled estimate of 81 years for the average fire rotation in the CM Group is less than most estimates based on fire scar studies, although near the lower end of Baker's (2006) estimated range. When we reduced the probability of mosaic fire by 50% as part of the sensitivity analysis, the fire rotation was 112 years, more consistent with the frequencies reported in the literature, with little effect on the abundance of the different community phases. Our model did not explicitly include fire suppression, but further reductions in the probability of mosaic fire would be consistent with the current distribution of fire sizes under the practice of full suppression of all unplanned fires. Further reductions in the probability of mosaic fire might be warranted to reflect a combination of the effects of fire suppression and a greater effect from livestock grazing than we estimated.

Simulated vs. Observed Conditions. Although we do not know the actual extent, observations indicate the J2 and J3 community phases may be more widespread than our model predicts. If so, that would be a indication that our modeled vegetation treatment rate of 10% is too high. The highest probability of treatment occurred in the J2 community phase in our model, where all types of treatment methods were included. This model structure keeps the J3 and OG community phases, and therefore the Phase III Cheatgrass state, out of

the simulated landscape until such treatments annually affect 5% or less of the Juniper Expansion state. Johnson and Miller (2008) concluded that cutting treatments on a frequency of less than 50 years would maintain juniper savannah in southwestern Idaho. The juniper expansion OG community phase is still absent from the study area but that lack is largely due to insufficient time for development (Johnson and Miller 2006).

3.6.3 Sage-grouse Habitat

Both the WD and CM Group models produced less sage-grouse habitat and lower quality habitat than estimated historically (see Chapter 2) consistent with documented declines (Miller and Eddleman 2000, Knick et al. 2003, Connelly et al. 2004). The amount provided was also less than the levels recommended by sage-grouse biologists (Connelly et al. 2000). The greatest reduction in sage-grouse habitat was in the WD Group, considered especially important for wintering habitat (Call and Maser 1985, Braun et al. 2005, Bruce 2008). We believe the degree of decline is not yet as great as the WD Group model indicates since cheatgrass-dominated states are not very widespread on the Malheur High Plateau in Oregon. Even in the reference state, the ES community phase was the most abundant phase, providing only lekking and pre-laying hen habitat. The projected lack of nesting, early broodrearing, and wintering habitat is especially problematic.

3.7 Conclusions and Management Implications

This study indicated how managers could use state-and-transition models, such as VDDT, in conjunction with the current state of knowledge on sagebrush ecosystem dynamics and sagegrouse habitat needs to examine how management activities and natural disturbance regimes might interact to shape sagebrush communities and sage-grouse habitat. It also demonstrated how the use of climate variables to develop probabilities of disturbance and management success could be incorporated into such models, potentially setting the stage for examining how changing climate and management activities might interact.

Our results suggest that controlling drought-grazing, increasing postfire seeding success, and treating large enough areas to control juniper are important factors in continuing to provide sufficient sage-grouse habitat quality and quantity at recommended levels. Drought-grazing in the WD Group directly facilitates rapid movement into cheatgrass-dominated states, while in the CM Group it indirectly facilitates the initiation of the Juniper Expansion state by accelerating the shift into community phases that are more vulnerable to juniper seedling establishment. In both sagebrush groups, our results suggest reductions in the occurrence of drought-grazing would enhance retention of sage-grouse habitat.

In the absence of vegetation treatments, sage-grouse habitat all but disappears, lost to cheatgrass and to juniper where seed sources are nearby. What habitat would remain might be highly fragmented, potentially triggering loss of sage-grouse from the Malheur High Plateau due to inbreeding depression and lack of genetic variability (Connelly et al. 2004, Frankham 2005). The sensitivity analysis suggests a need to increase the actual level of vegetation treatments to further slow the expansion of juniper and restore cheatgrass-dominated areas to sagebrush steppe. Even at twice the modeled success rate of postfire seeding, cheatgrass-dominated states still comprise a large proportion of the simulated landscape. The dominance of the ES and J1 community phases in the CM Group point to a need to develop vegetation treatments that remove all sizes of juniper and still preserve some level of large sagebrush.

Our results support other studies indicating passive management will not restore sagebrush ecosystems altered by cheatgrass and juniper expansion. Restoration treatments can reverse the effects of drought-grazing and juniper expansion when conducted at sufficient intensity and success rates, although the actual mix of community phases and quantity of different seasonal habitats likely will differ from historical conditions. However, vegetation treatments are unlikely to eliminate either cheatgrass or juniper as long as seed sources remain. Cheatgrass dominated areas will continue to persist in the WD Group and juniper will continue to establish in the CM Group unless general conditions become unfavorable for either species. Table 3.1. Habitat suitability (low, moderate, high) for greater sage-grouse by model, state, and community phase. Descriptions of each habitat element are based on Call and Maser 1985; Connelly et al. 2000, 2004;Crawford and Gregg 2001; Goodrich 2005; Braun et al. 2005; and Gregg 2006. Quality rating is based on how closely a given community phase in each sagebrush group matches the definition of that habitat and its expected duration.

	Sage-grouse Seasonal Habitat Type								
			Pre-		Early	Late			
	Community		laying		Brood-	Brood-			
State	phase	Lekking	Hens	Nesting	rearing	rearing	Wintering		
Warm, Dry Sagebrush Group									
	ES	Mod	High		Low	Low			
Minimum	MSO	Low	High		High	Low	Low		
Cheatgrass	LSO		Mod	High	High	Low	High		
	LSC		Mod	Mod	Mod	Low	High		
	ES	Mod	High		Low	Low			
Phase I	MSO	Low	High		High	Low	Low		
Cheatgrass	LSO		Mod	High	High	Low	High		
	LSC		Mod	Mod	Mod	Low	High		
Dhace II	MSO		Low		Low				
Cheatgrass	LSO			Low	Low		Mod		
<u>-</u>	LSC						Mod		
Phase III Cheatgrass	PIII	Low							
Cool, Moist Sagebrush Group									
	ES	Low	High		Low	Mod			
No Juniner	MSO		Mod	Low	Low	High			
No Julipei	LSO		Mod	High	High	High	High		
	LSC			Mod	Mod	Mod	Mod		
Phase I Juniper	PI		Low	Low	Low	Low	Mod		
Phase II Juniper	PII			Low	Low	Low	Low		
Phase III Juniper	PIII								
Juniper Old Growth	OG								
Phase III Cheatgrass	CHT	Low							

		Fire		Ins	ects	<u> </u>			
State	Community Phase	Mosaic	Stand- replacing	Shrub Drought	Buildup/ Crash	Peak	Over grazin	Drought- Grazing	Postfire Seeding
	ES	0.018	0.005				0.112	0.059	
мс	MSO	0.018	0.005	0.375			0.112	0.059	
ne	LSO		0.005	0.375	0.294	0.439	0.112	0.059	
	LSC		0.005	0.375	0.294	0.439	0.112	0.059	
	ES	0.018	0.005				0.112	0.059	
C1	MSO	0.018	0.005	0.375			0.112	0.059	
CI	LSO		0.005	0.375	0.294	0.439	0.112	0.059	
	LSC		0.005	0.375	0.294	0.439	0.112	0.059	
	MSO		0.23	0.375			0.112	0.059	
C2	LSO		0.23	0.375	0.294	0.439	0.112	0.059	
	LSC		0.23	0.375	0.294	0.439			
C3	C3		0.23						0.75

Table 3.2. Probabilities of different disturbances in the Warm-Dry Sagebrush Group by community phase.

Abbreviations: MC=Minimum cheatgrass, C1=Phase I cheatgrass, C2=Phase II cheatgrass, C3=Phase III cheatgrass, ES=early seral, MSO=midseral open, LSO=late seral open, LSC=late seral closed.

Dicturbanco	No Juniper				Juniper Expansion				Phase
Туре	ES	MSO	LSO	LSC	J1	J2	J3	OG	Cheat
Fire									
Mosaic	0.023	0.023	0.023						
Stand- replacing	0.006	0.006	0.006	0.006	0.006	0.006	0.006	0.006	0.23
Shrub Drought		0.375	0.375	0.375					
Insects									
Buildup/ Crash			0.294	0.294					
Peak			0.439	0.439					
Freezekill		0.009	0.009	0.009					
Snow mold		0.018	0.018	0.018					
Voles		0.036	0.078	0.01					
Overgrazing	0.112	0.112	0.112						
Drought- grazing	0.059	0.059	0.059						
Juniper Expansion			0.158	0.158					
Vegetation Treatments					0.033	0.1	0.034	0.034	0.75

Table 3.3. Probabilities of different disturbances in the Cool-Moist Sagebrush Group by community phase

Abbreviations: ES=early seral, MSO=midseral open, LSO=late seral open, J1=Phase I juniper, J2=Phase II juniper, J3=Phase III juniper, OG=juniper old growth

Table 3.4. Abundance of sage-grouse seasonal habitat for each model variant of the Warm-Dry Sagebrush Group. We created variants by adding current disturbances to the historical model structure or removing them from the full model structure to examine what role different types of management may play in sage-grouse habitat abundance. The Warm-Dry Sagebrush Group does not provide late brood-rearing habitat.

	Seasonal Habitat Type						
		Pre-laying		Early Brood-			
Model Variant	Lekking	Hen	Nesting	rearing	Wintering		
		F	Percent of Area	a			
Full Model	57.1	63.8	2.4	6.7	3.1		
Full minus Overgrazing	58.8	63.7	1.5	4.9	2.0		
Historical Only	4.6	30.0	23.6	25.4	23.6		
Historical plus Grazing	3.6	28.8	23.7	25.2	23.7		
Historical plus Seeding	22.0	98.8	68.9	76.8	68.9		
Full minus Drought- grazing	20.0	98.5	71.3	78.6	71.3		
Historical plus Drought- grazing	0.0	0.0	0.0	0.0	0.0		
Full minus Seeding	0.0	0.0	0.0	0.0	0.0		

Table 3.5. Abundance of sage-grouse seasonal habitat for each model variant of the Cool-Moist Sagebrush Group. We created variants by adding current disturbances to the historical model structure or removing them from the full model structure to examine what role different types of management may play in sage-grouse habitat abundance. The Cool-Moist Sagebrush Group does not provide lekking habitat.

	Seasonal Habitat Type						
	Pre-		Early	Late			
Model Variant	laying	Nostina	Brood-	Brood- Bearing	Wintering		
	<u>nen</u>		Percent of Ar		wintering		
Full Model	55.5	45.9	45.9	94.4	45.9		
Historical plus Juniper and Veg. Treatments	60.6	41.4	41.4	94.9	41.4		
Full minus Overgrazing	57.9	43.9	43.9	94.7	43.9		
Full minus Drought- grazing	57.8	43.9	43.9	94.6	43.9		
Historical Only	37.5	76.3	76.3	100.0	76.3		
Historical plus Veg. Treatments	37.3	76.4	76.4	100.0	76.4		
Historical plus Overgrazing and Veg. Treatments	34.6	78.4	78.4	100.0	78.4		
Historical plus Drought- grazing and Veg. Treatments	35.6	78.5	78.5	100.0	78.5		
Full minus Juniper	32.8	80.1	80.1	100.0	80.1		
Historical plus Juniper	13.2	12.8	12.8	24.4	12.8		
Historical plus Juniper and Overgrazing	12.1	12.9	12.9	23.5	12.9		
Historical plus Juniper and Drought-grazing	12.0	12.9	12.9	23.3	12.9		
Full-Veg. Treatments	11.1	13.1	13.1	22.6	13.1		
Historical plus Overgrazing	30.2	66.3	66.3	85.2	66.3		
Historical plus Drought- grazing	30.4	66.5	66.5	85.5	66.5		
Historical plus Overgrazing and Drought-grazing	28.5	67.9	67.9	85.4	67.8		



Figure 3.1. Location of the study area in Oregon. The study area lies in southeastern Oregon between the Owyhee River basin to the east, the Malheur and John Day River basins to the north, Deschutes and Klamath River basins to the west and the Nevada border to the south.



Figure 3.2. Simplified diagram of the successional pathways for the Warm-Dry Sagebrush Group. Solid lines with arrows indicate deterministic pathways in the absence of disturbance; dashed lines with areas indicate probabilistic pathways due to disturbance; and circles indicate disturbances that reset the relative age within a community phase. Community phase codes are ES = Early Seral, MSO = Midseral Open, LSO = Late Seral Open, LSC = Late Seral Closed. Broken lines indicate which community phases belong in which state. Minimum Cheatgrass is the reference state.



Figure 3.3. Simplified diagram of the successional pathways for the Cool-Moist Sagebrush Group. Solid lines with arrows indicate deterministic pathways in the absence of disturbance; dashed lines with areas indicate probabilistic pathways due to disturbance; and circles indicate disturbances that reset the relative age within a community phase. Community phase codes are ES = Early Seral, MSO = Midseral Open, LSO = Late Seral Open, LSC = Late Seral Closed, J1 = Phase I Juniper, J2 = Phase II Juniper, J3 = Phase III Juniper, OG = Juniper Old Growth, CHT = cheatgrass. Broken lines indicate which community phases belong in which state. No Juniper is the reference state.



Figure 3.4. Predicted abundance of seasonal sage-grouse habitat for the two sagebrush groups separately and in combination. The Cool-Moist Sagebrush Group provided abundant late brood-rearing and wintering habitat but consists of only 21% of the combined area. The combined area provided little nesting, brood-rearing, and wintering habitat.

Chapter 4: Potential Impacts of Climate Change on Sagebrush Steppe in Southeastern Oregon

Abstract. Climate change poses a threat to sagebrush (*Artemisia* spp. L.) ecosystems and the wildlife species that depend on them. Land managers need a method for downscaling climate projections to a scale useful in land management planning and need to consider multiple possible futures to address the uncertainties associated with any climate projection. We downscaled regional climate projections for the Pacific Northwest by applying the estimated degree of change (the 'delta' method) to a subregional climate zone in southeastern Oregon and we projected the long-term trends for this same zone to develop three different scenarios for future climate. Using the available literature on general and specific plant responses to changes in temperature, precipitation, and atmospheric carbon dioxide concentrations, we speculated on how general vegetation classes, disturbance regimes, and greater sage-grouse (*Centrocercus urophasianus*) habitat might change under each scenario. Under all three scenarios, we expect the quantity and quality of sage-grouse habitat would decline significantly and the cooler, moister sagebrush communities important for nesting, late brood-rearing, and some wintering habitat would likely disappear. Under two scenarios, the sagebrush plant communities and sage-grouse habitat would remain very vulnerable to cheatgrass (Bromus tectorum L.). Under two scenarios, juniper (Juniperus spp. L.) woodland would likely displace a large proportion of sagebrush and sage-grouse habitat. The remaining sage-grouse habitat would be limited and possibly fragmented. We conclude with a discussion of possible adaptation strategies.

Key words: climate change, sagebrush, juniper, sage-grouse, Oregon

4.1 Introduction

Changing climate is expected to result in a re-ordering of many ecosystems with the potential for novel climates and novel plant communities (Williams et al. 2007). The sagebrush region is considered one of the most threatened in North America from a variety of non-climate factors, such as altered disturbance regimes and invasive species (Knick et al. 2003, Welch 2005). Changing climate poses yet another threat to sagebrush ecosystems and the wildlife species that depend on sagebrush (Connelly et al. 2004). Federal land managers are required to consider potential changes to plant communities and ecosystem services that could arise from climate change when developing or revising land management plans.

There are several problems with applying the current projections for climate change at local scales. Available regional climate models operate at spatial scales that are too coarse to be used directly in land management planning, typically encompassing several different subregional climates (Mote 2003, Mote and Salathé 2009). Regional projections currently cannot incorporate finer scale climate features that may be important factors in subregional climate trajectories (Henry 1928, Tang and Reiter 1984, Adams and Comrie 1997, Sun et al. 2007, Salathé et al. 2008). Dynamic global vegetation models depend on the outputs from global and regional climate models, which lack the necessary detail to allow simulation of vegetation below very broad cover types, such as short grass or needle-leaved evergreen forest (Neilson 1995, Sitch et al. 2003, Gerten et al. 2004, Woodward and Lomas 2004). While there is high certainty that temperatures will continue to rise, there is less certainty over how much and there is considerable uncertainty over changes in precipitation (IPCC 2007, Mote and Salathé 2009). Federal land management planning for both climate and

vegetation and need to consider multiple possible futures to address the uncertainties surrounding climate change.

We selected the 4 million ha Malheur High Plateau major land resource Area (NRCS 2006) in southeastern Oregon to examine a downscaling approach for climate and vegetation. We were particularly interested in trying to answer two questions for this area: 1) how might the potential distribution of general plant communities and disturbance regimes change in the future and 2) how might these changes in plant communities and disturbance regimes affect habitat for the greater sage-grouse (*Centrocercus urophasianus*) (sage-grouse). We divided the sagebrush plant communities in Malheur High Plateau into four ecological groups based on current site productivity, as measured by perennial grass production (data available through the <u>Natural Resources Conservation Service</u>). This approach also resulted in a division approximately based on current soil moisture and temperature regimes. We then downscaled ensemble regional climate predictions for the Pacific Northwest created by Mote and Salathé (2009) and observed trends in a subregional climate zone to predict possible changes in precipitation and temperature regimes. Using the literature, we estimated how these changes might influence disturbance regimes and plant species composition, and how these vegetation changes might affect habitat for sage-grouse.

4.2 Study Area

The Malheur High Plateau (figure 4.1) includes the full range of habitat types used by sagegrouse and falls within a single climate division, Oregon Climate Division 7, as established by the National Oceanic and Atmospheric Administration (NOAA). Much of our study area lies between 1190 m and 2105 m elevation, with Steens Mountain reaching 2967 m. The area consists of interspersed hills, buttes, isolated mountains and north-south trending fault-block mountains. The Malheur High Plateau contains no major rivers and little surface water but has numerous springs, shallow lakes and playas. Perennial streams and small rivers are mostly located on the periphery of the province. Using data from soil surveys (http://soils.usda.gov/technical/classification/osd/index.html), we calculated that 98 percent of the soils in the sagebrush ecological types of the Malheur High Plateau were Mollisols and Aridisols. Soils are primarily loamy to clayey, well-drained and shallow to moderately deep on uplands, and poorly to well-drained and very deep in basins. Soil temperature regimes range from mesic in the lower elevations, frigid in the mid-elevations, and cryic generally above 2000 m elevation. Soil moisture regimes range from aridic at lower and mid-elevations to xeric at mid and upper elevations. Sagebrush-steppe (Artemisia spp. L.-cespitose grasses) is the dominant vegetation type, occurring across nearly all soil moisture and temperature regimes. Salt desert shrub (Sarcobatus vermiculatus (Hook.) Torr.-Grayia spinosa (Hook.) Mog.) is found mostly on saline soils in basins under the mesic temperature regime and aridic moisture regime. Aspen (Populus tremuloides Michx.) occurs at the highest elevations under cyric temperature regime and xeric moisture regime. Western juniper (Juniperus occidentalis Hook var. occidentalis) is currently expanding out of rockier areas and displacing sagebrushsteppe where the soil temperature regime is mesic to frigid and the moisture regime is xeric (Miller et al. 2005). Cheatgrass (Bromus tectorum L.), a non-native annual grass, is also displacing sagebrush-steppe where the soil temperature regime is mesic (Chambers et al. 2007). The average annual precipitation ranges from 105 to 305 mm over most of the area with a winter-spring dominant regime. The average annual temperature is 8°C. Winter is the coldest and wettest period, summer the warmest and driest.

4.3 Methods

We compared long-term temperature, precipitation and snow water equivalent trends in Oregon Climate Division 7 to the general trends documented for the Pacific Northwest as a whole to determine what differences might exist between the subregional and regional climate. We created simple linear regressions in Sigma Plot 11 (Systat Software 2008) for annual, seasonal, and monthly precipitation and temperature (data available at http://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp) and for April 1 snow water equivalent at the two Snotel sites within the study area (data available at http://www.wcc.nrcs.usda.gov/snow/). Data for Oregon Climate Division 7 included the period 1895 through 2009; data for the two Snotel stations included the period 1939 through 2010. We compared these results to documented changes in the Pacific Northwest as described by Mote (2003) and the Climate Impacts Group (2004) to determine if enough difference existed between the subregional and regional climate to use the long-term trends in Oregon Climate Division 7 as a basis for one scenario.

We then developed three climate change scenarios to evaluate possible changes to vegetation and sage-grouse habitat. We based the first two climate scenarios on the most recent ensemble projections for the Pacific Northwest and which used the A1B emissions scenario (Mote and Salathé 2009); the A1B emissions scenario assumes little change in the current emissions trends for greenhouse gases. To downscale these results, we applied them to the data for Oregon Climate Division 7, using the same base period (1970-1999) and the same three months in each season defined by Mote and Salathé (2009). Known as the 'delta' approach, this technique preserves monthly and seasonal patterns in the existing climatology. To provide a range of possible futures, one scenario used the 75th percentile projection for temperature in combination with the 25th percentile projection for precipitation while the
second used the 25th percentile projection for temperature combined with the 75th percentile projection for precipitation. While not the traditional approach to developing climate scenarios, we felt this approach was adequate for understanding the relative differences between different possible futures particularly when projecting so far into the future. We based the third scenario on a projection of the observed long-term trend in precipitation and temperature for Oregon Climate Division 7, assuming no change in these trends. All scenario projections were for near the end of the 21st century.

To evaluate the potential impacts of changing climate and disturbance regimes on the plant communities used by sage-grouse, we estimated the actual evapotranspiration to potential evapotranspiration ratio using the method developed by Thornthwaite and Mather (1955) and estimated the likely biome using the rule-based system of Neilson et al. (1992). We interpreted these broad biome categories into cover types reflective of local conditions. Using ecological site descriptions (available at http://esis.sc.egov.usda.gov) and soil survey information for the Malheur High Plateau, we sorted sagebrush ecological sites into four groups based on grass productivity, assuming that grass productivity was the single best predictor of recovery potential from disturbance (see also Chapter 2 for more complete descriptions). Ecological site descriptions consist of the physiographic and soil features, climate, characteristic and historical plant communities including estimated annual productivity in different types of years, and site suitability for different land uses such as livestock grazing, wildlife, recreation, and so forth for a given type of site.

We dropped one group, the Warm-Moist Sagebrush Group, as most of this group had been converted to agriculture. The remaining groups we considered were the Cool-Moist Sagebrush Group (CM Group) typically found on sites with xeric soil moisture regime and frigid soil temperature regime, the Warm-Dry Sagebrush Group (WD Group typically found on sites with an aridic soil moisture regime and mesic soil temperature regime, and the Shallow-Dry Sagebrush Group (SD Group) typically found on sites with shallow to very shallow soil depth and an aridic soil moisture regime. Sage-grouse use all three groups at various times of the year for breeding, brood-rearing, and wintering. Along with the sagebrush groups, other general vegetation types currently found in the Malheur High Plateau that we considered were salt desert shrub, juniper woodland, and aspen.

Lastly, we reviewed the literature on plant-climate and plant-atmospheric CO₂ interactions to understand how feedbacks and interactions may modify potential responses based solely on changes in temperature and precipitation regimes. We assumed that broader scale projections in climatic extremes, such as an increased frequency of high temperature events (Christensen et al. 2007, Caprio et al. 2009); increased areal extent, duration, and severity of drought (Cook et al. 2004, Wang et al. In Press); continued changes in snow pack (Knowles et al. 2006, Christensen et al. 2007, Brown and Mote 2008); and changes in the frequency, size, and intensity of rain events (Sun et al. 2007), also applied to the Malheur High Plateau. We then speculated on how the different climate scenarios might alter the presence or abundance of general cover types, such as salt desert shrub and juniper woodland, natural disturbance regimes and management activities, and sage-grouse habitat quantity and distribution. Plant nomenclature follows the U.S. Department of Agriculture PLANTS database (http://plants.usda.gov).

4.4 Results and Discussion

4.4.1 Observed Climate Trends

Over the last approximately 100 years in both the Pacific Northwest and Oregon Climate Division 7, average annual temperature increased, with slightly less increase in Oregon Climate Division 7 (table 4.1). Average winter and summer temperature increased at both scales, although the winter temperature increase was much greater for the Pacific Northwest than for Oregon Climate Division 7. Average spring temperature increased in the Pacific Northwest and decreased in Oregon Climate Division 7. Precipitation changes were very different between the Pacific Northwest and Oregon Climate Division 7. In the Pacific Northwest, average annual precipitation increased by over 30% and increased in all seasons, whereas in Oregon Climate Division 7, average annual precipitation increased only slightly and decreased in fall and winter. In the Pacific Northwest, April 1 snow water equivalent decreased at nearly all elevations, with the largest decrease occurring below 1800 m elevation (Mote 2003, Climate Impacts Group 2004), while in Oregon Climate Division 7, April 1 snow water equivalent decreased below 2100 m elevation and increased above 2300 m elevation. The decrease in the Malheur High Plateau below 2100 m was consistent with the Pacific Northwest trend, although the degree of decrease was less. Whether the observed increase in the Malheur High Plateau above 2300 m elevation was consistent with the observed changes in the Pacific Northwest is less certain as there are few stations in the Pacific Northwest at or above 2100 m and a large number of stations at lower elevations. These differences supported the creation of the climate scenario based on long-term trends in Oregon Climate Division 7.

The observed trends in Oregon Climate Division 7 were consistent with studies that note the potential for contrary trends in rain shadow landscapes and those that lack geographic features that promote orographic uplift (e.g., Sun et al. 2007). The observed increase in summer precipitation may be a result of enhanced monsoonal circulation over the study area brought about by several changes in winter and summer conditions across western North America (Houghton 1979, Tang and Reiter 1984, Adams and Comrie 1997, Hawkins et al. 2002, Grantz et al. 2007). Reduced snow cover in winter apparently has affected the positions and strength of surface high pressure systems and reduced the opportunity for precipitation in winter from the passage of troughs and weak cold fronts (Henry 1928, Regonda et al. 2005, Knowles et al. 2006).

4.4.2 Future Climate Scenarios

Scenario 1. Scenario 1 (S1), based on the 25th percentile precipitation and 75th percentile temperature expected for the Pacific Northwest, would result in much warmer and drier conditions than present (table 4.2). Winter, as defined by temperature, would shrink to only two months and become much warmer than present. Summer would expand to approximately five months. The precipitation regime would become weakly spring-summer dominant. Winter would remain the coldest season and summer the warmest, but spring would become the wettest season and fall the driest. Soil moisture stress would be greater in all seasons than present. The soil moisture regime would likely become aridic throughout much of the Malheur High Plateau and xeric at the highest elevations. A thermic soil temperature regime could develop in the lower elevations and the mesic soil temperature regime could shift upward. The cryic and frigid temperature regimes would likely disappear. Deep soil water recharge would be less than present in most years and saturated soils less likely. The dominant cover type likely would be a grassland of short grasses (Neilson et al.

1992) possibly similar to the desert grasslands of the Southwest. However, if the thermic soil temperature regime developed as seems possible, salt desert shrub cover types could become widespread.

Scenario 2. Scenario 2 (S2), based on the 25th percentile change in temperature and 75th percentile change in precipitation for the Pacific Northwest, would result in a warmer and wetter climate than present with a winter-spring dominant precipitation regime (table 4.2). Season length would be approximately the same as present, although all seasons would be warmer. Winter would remain the coldest and wettest season and summer the warmest and driest. Soil moisture stress would increase from mid-spring through fall, although to a lesser degree than in S1. The aridic soil moisture regime likely would expand upwards, shifting the xeric soil moisture regime upward as well, although the shift would not be as great as under S1. A thermic soil temperature regime might develop in the warmest locations and the mesic temperature regime might expand upward, also shifting the frigid temperature regime upward. The cryic soil temperature regime likely would disappear. Deep soil water recharge would be similar to or greater than present. The expected increase in spring precipitation would result in a somewhat higher probability of saturated soils than at present. The expected dominant biome would remain shrub-steppe (Neilson et al. 1992) similar to the present, but cover types would likely shift as the soil moisture and temperature regimes shifted. Juniper woodland potential would remain where the soil temperature regime remained mesic to frigid.

Scenario 3. Scenario 3 (S3), based on current temperature and precipitation trends for Oregon Climate Division 7, would result in a warmer and wetter climate than present, but with a spring-summer dominant precipitation regime (table 4.2). The length of the seasons would

remain unchanged from present, with winter, spring and summer slightly warmer than present and fall slightly cooler. Winter would remain the coldest season and summer the driest, but as in S1, spring would become the wettest season and fall the driest. Soil moisture stress would decrease in spring and summer and increase in fall. The aridic soil moisture regime likely would contract to the driest locations and the xeric moisture regime would expand downwards. An ustic moisture regime might develop in the higher elevations. The mesic temperature regime would likely expand upward, shifting the frigid temperature regime upward as well. The cryic soil temperature regime would contract to the highest elevations. Deep soil water recharge would be less than present and saturated soils would be more likely in spring than present. The expected dominant biome would be a grassland dominated by short grasses (Neilson et al. 1992) likely similar to the shortgrass prairie of the northern Great Plains. However, the potential for juniper woodland would continue where the soil temperature regime remained mesic to frigid.

Entire plant communities would not shift as climate changes; new plant communities would likely develop as individual species shift (Williams et al. 2007). However, our use of the sagebrush groups provides a useful simplification that aided in discussing the implications for sage-grouse habitat, similar to the use of plant functional groups in the ecology literature. All sagebrush groups include more than one big sagebrush subspecies (Appendix A), but one subspecies tends to dominate in a given group (table 4.3). Each sagebrush group tends to occur in a particular combination of soil moisture and temperature regimes, and a characteristic soil depth that indicates potential water holding capacity.

4.4.3 Expected Changes in Vegetation

Each of these scenarios carries different implications for the mix of vegetation types and the species compositions within those types and their associated disturbance regimes. Variability in climate is more important than changes in means for plant species occurrence and productivity (Zavaleta et al. 2003). Since we were unable to examine variability directly, we used combinations of soil temperature regime and soil moisture regime and assigned different vegetation types to each combination as a device to evaluate the differences between the three scenarios (table 4.4). Some combinations could support more than one type, depending on other factors, such as soil water holding capacity and disturbance regimes.

General Responses. Some potential plant species and cover type responses would apply across all scenarios. High variability in precipitation amount and timing, higher temperatures, and changed drought regimes would likely reduce overall herbaceous plant cover and density (Kremer et al. 1996, Kremer and Running 1996, Breshears et al. 2005, Pereira et al. 2006, Miriti et al. 2007, McDowell et al. 2008). Increased frequency of heat waves and changes in the drought regime would favor ruderal species and drought-tolerant species (White et al. 2001, Morecroft and Paterson 2006), many of which may be non-native invasive species; and favor shallower rooted C₃ perennial species, C₄ perennials and annuals, and cacti (Ehleringer et al. 1991, Comstock and Ehleringer 1992, Cook and Irwin 1992, Epstein et al. 1997). Increased winter temperatures should result in fewer losses of cheatgrass seedlings from frost heaving (Bates et al. 2006) but also reduced flowering and seed production (Finnerty and Klingman 1962).

The potential response of most forbs to changing climate is unknown, other than with respect to vague generalities at the plant functional group level (Ehleringer et al. 1991, Zavaleta et al. 2003, Reynolds et al. 2004, Menzel and Sparks 2006, Morecroft and Paterson 2006), which may not apply to all species in that group (Zavaleta et al. 2003). Response of forbs has potentially major implications to nutrition and survival of pre-laying hens and chicks (Klebenow and Gray 1968, Barnett and Crawford 1994, Drut et al. 1994, Connelly et al. 2004, Gregg 2006, Gregg et al. 2006, Ersch 2009, Gregg and Crawford 2009). In addition, the semi-arid areas of the western United States support a large number of non-native and invasive plants, with some non-natives important food sources for pre-laying hens and chicks (Klebenow and Gray 1968, Barnett and Crawford 1994, Drut et al. 1994, Gregg 2006) while others degrade sage-grouse habitat (Connelly et al. 2004). As climate changes, new invasive species may establish, the impacts or distributions of existing species may change, and species currently considered non-invasive may become so (Hellmann et al. 2008), resulting in impacts to sage-grouse habitat quantity and quality. Under S1 conditions, reduced water availability in late spring could reduce flowering and seed production in perennial species while the increased water availability under S2 and S3 could have the opposite effect (Sauer and Uresk 1976, Morecroft and Paterson 2006). The increased summer moisture under S3 conditions could favor facultative rhizomatous species such as bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), and rhizomatous grasses such as thickspike wheatgrass (*Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould) (Miller et al. 1986, Ogle 2002a, Ogle 2002b). The drier conditions under S1 would likely result in an increase in the amount of bare soil while the amount of bare soil may fluctuate more than present under S2 and S3, providing sites that cheatgrass could exploit (White et al. 2001, Morecroft and Paterson 2006).

Increasing winter temperatures and reduced snow depth may lead to higher sagebrush productivity at the highest elevations than currently present (Perfors et al. 2003). Big sagebrush plants may become shorter in stature as April precipitation increases (Cook and Irwin 1992). Sagebrush densities may fluctuate during a transition period as older plants die and genotypes adapted to milder winter conditions migrate in (McDonough and Harniss 1974, Meyer et al. 1990, Meyer and Monsen 1992). Drier soils, warmer temperatures, and inadequate chilling of seeds could lead to range contractions and reduced densities of several species of tall shrubs, low sagebrush and western juniper (*Juniperus occidentalis* Hook. var. *occidentalis*), and promote expansion of subshrubs (Young et al. 1988, Baskin and Baskin 1989, Donovan and Ehleringer 1994, Chambers et al. 1999, Zlatnik 1999, Steinberg 2002). Utah juniper (*Juniperus osteosperma* (Torr.) Little), which is generally more tolerant of drier conditions than western juniper (Thompson et al. 1999), or Utah-western juniper hybrids (Vasek 1966, Dealy 1990, Terry In Press) could replace western juniper on some sites.

Sagebrush-Steppe. The area occupied by sagebrush would decrease under all three scenarios, with the greatest decline under S3 conditions and least under S1 conditions (figure 4.2). Because the SD Group typically occupies shallow soils with limited water holding capacity (table 4.3), changes in the area occupied by this sagebrush group tend to occur independent of the other two sagebrush groups. Changes in the WD Group and the CM Group tend to be closely linked in all three scenarios, as the boundary between these two groups is less distinct. Expansion and shifts in the area occupied by one group most often accompanies contraction and shifts in the other group, assuming no other cover types are involved. The SD Group is intermingled with both the WD Group. Changes in the location and area occupied by the sagebrush cover types would be driven by changes in the location and area occupied by the different soil moisture and temperature regime combinations (table 4.4), although other factors, such as site characteristics and changes in disturbance regimes, could further alter the potential outcomes.

We expect the area occupied by the SD Group will decline in the lower elevations under both S1 and S3, but for different reasons. Under S1, salt desert shrub could displace most or all of the SD Group in the lower elevations (Miller and Wigand 1994, Thompson et al. 1998, Shafer et al. 2001), although some shifting onto sites that became too dry for the WD Group may occur. The SD Group occupying ridgetops with shallow soil will likely persist. Under S3, juniper woodland or savanna may displace the SD Group above the basin floors, although some sagebrush may persist as an understory species in any savanna that develops. Under S2 conditions, the SD Group may contract from sites that would be more favorable to salt desert shrub, but could also expand somewhat onto sites currently occupied by the WD Group, resulting in little net change in terms of area. The drought regime may be a large determinant of the degree and type of change in the area occupied by the SD Group under S2 conditions.

The WD Group would likely expand and shift upward in elevation, displacing the CM Group, which likely would disappear under S1 conditions (Thompson et al. 1998, Shafer et al. 2001). The WD Group would also expand and shift upward under Scenario 2, although the expansion would not be as great and the CM Group may persist at the higher elevations where it was not displaced by juniper woodland. The WD Group likely would contract under S3 conditions, and juniper woodland would likely displace the CM Group (Bachelet et al. 2001b, Bachelet et al. 2008, Lenihan et al. 2008).

Salt Desert Shrub. Salt desert shrub communities typically develop on saline soils. Many species associated with salt desert shrub cover types, such as winterfat (*Krascheninnikovia lanata* (Pursch) A. Meeuse & Smit), budsage (*Picrothamnus desertorum* Nutt.), and shadscale

(Atriplex confertifolia (Torr. & Frém.) S. Watson), grow on nonsaline soils but usually cannot compete well against other species under a mesic soil temperature regime (Billings 1949, Comstock and Ehleringer 1992). If a thermic soil temperature regime developed in the lower elevations, it could support taxa typically found further south in the Great Basin and the Mojave Desert (Thompson et al. 1999, Shafer et al. 2001). A thermic regime could provide the salt desert species with a competitive advantage on nonsaline soils, and many of these species are present in the study area. Changes in the drought regime, reduced deep soil water recharge, and vegetation-atmosphere feedbacks could facilitate this shift as temperature increased and many plant species currently occupying these sites became more sensitive to smaller changes in soil moisture availability (Cook et al. 2004, Loik et al. 2004, Breshears et al. 2005, Diffenbaugh 2005a, b, Michels et al. 2007, Stahle et al. 2007, Groisman and Knight 2008, Adams et al. 2009). The S1 conditions would support the widest expansion of the salt desert shrub types, potentially occupying much of the area now occupied by the SD Group in the lower elevations and possibly some of the area now occupied by the WD Group. Scenario 2 conditions would support more limited expansion, most likely from existing locations into some of the area now occupied by the SD Group (Comstock and Ehleringer 1992, Miller and Wigand 1994). Scenario 3 conditions have the lowest temperature increase along with wetter conditions, so likely would not provide for expansion of the salt desert shrub types unless the increased moisture resulted in mobilization of deep salts and salinization of soils (Buol et al. 2003).

Conifers. Western juniper typically occurs in the same combinations of soil moisture and temperature regimes as the CM Group (table 4.4). Utah juniper, currently not found in the study area, is adapted to spring-summer dominant precipitation regimes while western juniper is found in winter-spring dominant precipitation regimes (Vasek 1966, Dealy 1990, Comstock

and Ehleringer 1992, Chambers et al. 1999, Miller et al. 2005). Infrequent fire favors the presence and spread of juniper (Burkhardt and Tisdale 1976, Miller and Rose 1995, 1999, Miller et al. 2005). Western juniper woodlands would likely disappear under S1 conditions, shift upward in elevation and contract under S2, and expand under S3 conditions. However, if Utah juniper or Utah-western hybrids migrated into the Malheur High Plateau the outcomes may differ somewhat. Since Utah juniper tolerates drier conditions than western juniper (Thompson et al. 1999, Lyford et al. 2003), the juniper woodland cover type could expand into the mesic soil temperature regime (table 4.4), reducing the potential area that the WD Group might occupy under all three scenarios. Any juniper woodland would be restricted to the highest elevations under S1 conditions. Under S2 conditions, Utah juniper woodlands in the mid-elevations could develop below western juniper woodlands in the high elevations with both juniper woodland types displacing the CM Group and partly displacing the WD Group, most likely on the more productive sites of this group. Scenario 3 conditions would result in a climate similar to that on the Colorado Plateau (Comstock and Ehleringer 1992), potentially allowing pinyon, likely Pinus edulis Engelm., and Gambel oak (Quercus gambelii Nutt.) to expand into the Malheur High Plateau (Shafer et al. 2001). This potential pinyon-juniper woodland would displace most or all of the CM Group and potentially much of the WD Group (Bachelet et al. 2001b, Bachelet et al. 2008, Lenihan et al. 2008). The movement of Utah juniper or Utah-western juniper hybrids into the Malheur High Plateau seems probable given the current proximity of both to the study area (Vasek 1966, Dealy 1990). Natural movement of pinyon or Gambel oak into the area seems less certain given the distance involved. If an ustic soil moisture regime developed under Scenario 3, then the higher elevations could also support expansion of the existing aspen forest and the development of a conifer forest (Bachelet et al. 2008), if seed sources were available.

Cheatgrass. Cheatgrass as a cover type usually occurs in the same soil moisture and temperature regimes as the WD Group (table 4.4), although the species is present across the landscape (Knapp 1996, Pellant 1996). The presence or absence of cheatgrass as a cover type depends on the fire regime that would develop under the changed conditions, with frequent fire favoring cheatgrass expansion (D'Antonio and Vitousek 1992, Pellant 1996). June rainfall amounts also influence cheatgrass abundance and competiveness with maximum abundance at 20 mm and decreasing abundance at greater and smaller precipitation amounts (Bradley 2009). Predicted average June precipitation under both S1 and S2 would be optimum or near optimum for cheatgrass, indicating widespread vulnerability of sagebrush to displacement by cheatgrass if the fire return interval were frequent enough (D'Antonio and Vitousek 1992). Under S3 conditions, the estimated June precipitation and change in precipitation regime would disfavor cheatgrass, resulting in contraction and reduced vulnerability in what sagebrush remains (Bradley 2009).

Additional Factors. Other factors, such as the degree of deep soil water recharge, increased atmospheric CO₂ concentrations, and changes in the disturbance regimes, could alter the cover types in ways that are difficult to predict, increasing the uncertainty over the potential outcomes under all three scenarios. Deep soil water recharge primarily affects woody species and limited recharge could reduce sagebrush and juniper density or restrict them to higher elevations than the soil moisture and temperature regimes would suggest (Cook and Irwin 1992, Flanagan et al. 1992, Donovan and Ehleringer 1994, Thompson et al. 1998, Shafer et al. 2001, Gillespie and Loik 2004). Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) and low sagebrush (*Artemisia arbuscula* Nutt.) can grow on sites too dry for juniper (Comstock and Ehleringer 1992, Miller and Wigand 1994), suggesting less deep water recharge is needed to maintain relatively high sagebrush

density and cover. Scenarios 1 and 3 would result in less deep soil water recharge than present, suggesting that future sagebrush communities would be more open than present, although the increased summer precipitation in S3 could partly mitigate that outcome (Flanagan et al. 1992, Leffler et al. 2004). Under all three scenarios, both sagebrush and juniper would likely adjust density, stature, and crown size in order to deal with the increased demands for maintenance respiration brought about by the higher temperatures (Kremer et al. 1996), with S1 conditions producing the greatest changes and S3 the least.

Atmospheric carbon dioxide (CO₂) concentrations would further alter potential responses, but predicting those alterations remains difficult. Both juniper and cheatgrass have exhibited responses consistent with increased water use efficiency arising from increased CO₂ concentrations, allowing them to persist on or expand into drier sites than considered typical (Hunter 1991, Miller and Wigand 1994, Knapp 1996, Knapp et al. 2001b, a, Meyer et al. 2001, Beckstead and Augspurger 2004, Ziska et al. 2005, Blank et al. 2006). Sagebrush and some grass species have also responded with increased WUE, but to a lesser degree than juniper and cheatgrass (Smith et al. 1987, Mayeux et al. 1994, Knapp and Soulé 1996, Shaw et al. 2000, Lucash et al. 2005). However, environmental stresses often reduce plant responses to increased atmospheric CO₂ and most species acclimatize to increased CO₂ when some other limitation, most commonly N availability, is reached (Ainsworth and Long 2005, Wang et al. 2006, Ziska and Bunce 2006). Responses differ between life forms and plant functional groups, in wet verses dry years, on dry verses mesic sites, and between ecotypes of a given species (Mayeux et al. 1994, Wand et al. 1999, Knapp et al. 2001a, Ainsworth and Long 2005, Lucash et al. 2005).

4.4.4 Changes in Disturbance Regimes

Changes in disturbance regimes, management activities, and the feedbacks between disturbance agents, vegetation, and climate could also alter the approximate locations and area occupied by each of the cover types. Reductions in snowpack depth and duration under all three scenarios would greatly reduce or eliminate the probability of any disturbance factor tied to snowpack, such as freezekill, snow mold, and vole impacts (Mueggler 1967, Frischknecht and Baker 1972, Hanson et al. 1982, Nelson and Sturges 1986, Sturges and Nelson 1986, Parmenter et al. 1987, Sturges 1989, Walser et al. 1990, Hardy et al. 2001).

Fire. The general fire frequency over the Malheur High Plateau as a whole would likely decrease under all three scenarios where cheatgrass was not abundant, but for different reasons. In semi-arid environments, conditions in prior years control fire occurrence by affecting the abundance and continuity of fine fuels (Swetnam and Betancourt 1998, Brown et al. 2004, Dube 2009, Littell et al. 2009). Under S1 conditions, reduced fire frequencies would be due to the expected increase in bare ground and reductions in grass fuel abundance and continuity while under S2 and S3 conditions, the expansion of juniper woodland would reduce fuel continuity and grass fuel abundance over much of the Malheur High Plateau. The combined effects of increased temperature, which would tend to reduce herbaceous production, and increased precipitation, which would tend to increase herbaceous production, under S2 and S3 conditions make any predictions on how fire frequency might change in sagebrush difficult and very uncertain. If S3 conditions resulted in widespread occurrence of rhizomatous grasses and increased fuel continuity, then fire frequency could increase. Cheatgrass would likely remain a problem fuel type under S1 and S2 conditions, increasing fire frequencies, but not under the expected precipitation regime of S3 (Ziska et al. 2005, Blank et al. 2006, Bradley 2009). Under all three scenarios, the expected fuel arrangement

would tend to produce homogeneous burn patterns more often than heterogeneous burn patterns. Under S1 and S2, the relative proportion of low, average, high, and extreme years and the range in annual hectares burned would be less certain, depending on how widespread cheatgrass would become. Under S3 conditions, the relative proportion of low and average fire years would likely increase and the proportion of high and extreme years decrease, but the range in annual hectares burned is less certain, depending on how prevalent and continuous grass understories or pinyon-juniper savannas would be (Eddleman et al. 1994, Miller and Eddleman 2000).

Drought. Drought severe enough to kill woody plants would occur more frequently, affect a greater area, last longer, and likely become stand-replacing during at least some events under all scenarios (Cook et al. 2004, IPCC 2007, Michels et al. 2007, Stahle et al. 2007, Groisman and Knight 2008). As median annual precipitation declines, plants become increasingly sensitive to small changes in precipitation and temperature (Zavaleta et al. 2003, Austin et al. 2004, Loik et al. 2004), and vegetation-atmosphere feedbacks can increase regional temperatures beyond that expected by CO₂ forcing alone (Diffenbaugh 2005b). Higher temperatures shorten the time to drought-induced mortality and can increase the extent of area affected (Breshears et al. 2005, Adams et al. 2009). Drought-related mortality would be most likely, occur more frequently, and affect the widest area under S1. Some increase in drought-related mortality is less certain. For example, if drought resulted in greater proportional reductions in warm season precipitation, the potential impacts of drought likely would be greater under S3 and if the reductions were proportionally greater in cool season precipitation, the greater impacts would likely occur under S2.

Insects. Projections of potential changes in the frequency, duration, and effects of insect outbreaks are very difficult due to the high level of uncertainty over climate-induced changes in insect species compositions and population dynamics. Increasing CO₂ concentrations will alter leaf C:N ratios, requiring that insect larvae consume more leaf material in order to obtain sufficient N to support development, slowing insect development and increasing mortality in some insect species (Johnson and Lincoln 1991, Cannon 1998, Brooks and Whittaker 1999, Harrington et al. 2001, Goverde and Erhardt 2003, Agrell et al. 2005). Increased leaf consumption from the current insect herbivores, the emergence of new significant insect herbivores (Cannon 1998, Bale et al. 2002), or more frequent outbreaks (Harrington et al. 2001, Bale et al. 2005) may increase sagebrush mortality during insect outbreaks.

Postfire Seeding Success. A common management practice after a wildfire is to seed desirable grasses, forbs, and shrubs to reduce erosion and the risk that invasive plant species, such as cheatgrass, would assume dominance of the site (Robichard et al. 2000, Brooks 2005, Epanchin-Niell et al. 2009). Postfire seeding success is low under the current conditions (James and Svejcar 2010), with many failures attributed to inadequate moisture (Evans and Young 1978, Ratzlaff and Anderson 1995, Abbott and Roundy 2003, Jessup and Anderson 2007, Wirth and Pyke 2009). Increased climate variability under all three scenarios would likely increase variability in postfire seeding success rates (Ratzlaff and Anderson 1995, Abbott and Roundy 2003, Chambers et al. 2007, Jessup and Anderson 2007, James and Svejcar 2010). However, the much drier conditions under S1 would lead to a much lower probability of postfire seeding success using current seed mixes and postfire survival of bunchgrasses may also be higher than present due to wetter falls and winters (Ratzlaff and

Anderson 1995, Brooks 2005, James and Svejcar 2010). Overall moister conditions under S3 could also increase the probability of postfire seeding success, although the drier falls may reduce the survival of some seeded species (Chambers et al. 2007, Jessup and Anderson 2007).

Risk of Overgrazing. Domestic livestock grazing also that affects vegetation, disturbance regimes, and sage-grouse habitat (Connelly et al. 2004, Ersch 2009). While light to moderate grazing can improve or maintain productivity and may confer some resistance to invasive grasses following a wildfire (Holechek et al. 2004, Bates et al. 2009, Davies et al. 2009), high utilization, particularly during a drought, increases grass mortality and facilitates the development of increased density and cover of woody plants (Julander 1945, Holechek et al. 2004, Briske et al. 2008). Under all three scenarios, warmer conditions and greater variability in precipitation would increase the probability of over-grazing if animal numbers were not adjusted to reflect the changed forage availability. Under S1 conditions, high air temperatures and less deep soil water recharge would reduce the recovery potential for perennial grasses (Angell et al. 1990, Anderson and Toft 1993, Volaire et al. 1998, Ebdon and Kopp 2004, Reynolds et al. 2004, Wang and Huang 2004). Higher precipitation would likely increase the recovery potential under S2 and S3 conditions (Harris and Wilson 1970, Harris 1977, Martens et al. 1994, Nasri and Doescher 1995, Goodwin et al. 1996, Bair et al. 2006). If rhizomatous grasses became more common across the Malheur High Plateau under S3 conditions, the effects of overgrazing may be less severe (Holechek et al. 2004, Díaz et al. 2007).

4.4.5 Expected Changes in Sage-grouse Habitat

Changing climate and disturbance regimes, and increasing frequency of extreme weather events should result in significant changes to plant communities and plant species compositions (Botkin and Sobel 1975, Morecroft and Paterson 2006, Pereira et al. 2006, Dube 2009, Krawchuk et al. 2009) affecting the location, quantity, and quality of greater sagegrouse habitat. Preferred sage-grouse habitat consists of several structural and species composition elements to provide breeding, brood-rearing, and wintering habitat (Connelly et al. 2004). In general, sage-grouse need heterogeneous landscapes with a variety of patch types at a variety of sizes (Aldridge 2005, Aldridge and Boyce 2007). We expect that sagegrouse habitat abundance and quality would decline under all climate scenarios.

Some effects will likely be common to all three alternatives. Increasing frequency of extended droughts would likely reduce the abundance of needed forbs, reducing nesting success and chick survival (Barnett and Crawford 1994, Gregg 2006, Gregg et al. 2006, Aldridge et al. 2008, Ersch 2009, Gregg and Crawford 2009). The expected loss or near loss of the CM Group would severely reduce the availability of high quality late brood-rearing habitat and some nesting and wintering habitat (Klebenow and Gray 1968, Call and Maser 1985, Drut et al. 1994, Coggins 1998, Connelly et al. 2000, Connelly et al. 2004, Braun et al. 2005, Gregg 2006, Huwer et al. 2008). Reduced sagebrush height and cover may lessen habitat quality for nesting (Connelly et al. 2000, Aldridge 2005, Braun et al. 2005, Aldridge and Boyce 2007, Hagen et al. 2007, Ersch 2009).

Scenario 1. Potential habitat quantity may be highest under this scenario relative to S2 and S3, but habitat quality may be lowest (Figure 4.2). Reductions in the abundance of the SD Group would reduce the availability of high quality lekking, pre-laying hen, and early brood-

rearing habitat (Call and Maser 1985, Barnett and Crawford 1994, Coggins 1998, Connelly et al. 2000, Connelly et al. 2004, Dunbar et al. 2005, Gregg et al. 2006). The WD Group can provide all aspects of sage-grouse habitat, but conditions are usually too dry to provide high quality late brood-rearing habitat. Under S1 conditions, sage-grouse habitat could become fragmented, limited to the higher elevations such as the Steens Mountain-Pueblo Mountains complex and Hart Mountain, particularly if cheatgrass becomes widely dominant. Other higher peaks, such as Wagontire Mountain, may not provide sufficient habitat for viable populations. Further, the birds may be unwilling to use the steeper slopes at these higher elevations (Call and Maser 1985, Cadwell et al. 1994). Grass cover for nest sites may become marginal as the drier conditions limit grass height and possibly alter grass composition to shorter species, such as squirreltail (*Elymus elymoides* (Raf.) Swezey) and Sandberg bluegrass (*Poa secunda* J. Presl).

Scenario 2. The moister conditions under S2 likely would produce higher quality habitat than S1 conditions, with somewhat better forb production for pre-laying hens and early broods and better grass cover for nesting sites (Angell et al. 1990, Reynolds et al. 2004, Aldridge 2005). Habitat connectivity may be much better, with less fragmentation if cheatgrass does not become widely dominant. However, the quantity of available habitat may be much less (Figure 4.2). The effects of higher temperatures on plant productivity, density, and cover may result in much of the habitat functioning as an attractive sink, unable to provide sufficient quality habitat to assure sustainable populations, instead of remaining a source habitat (Aldridge 2005, Aldridge and Boyce 2007, Aldridge et al. 2008).

Scenario 3. Nearly all sage-grouse habitat could be lost under S3 conditions as increased productivity shifts much the Malheur High Plateau into pinyon-juniper woodland (Figure 4.2)

(Cook and Irwin 1992, Bachelet et al. 2001b, Shafer et al. 2001, Bachelet et al. 2008, Lenihan et al. 2008). Much of the remaining habitat would likely be of intermediate quality relative to S1 and S3. Where sagebrush remains, the shift in precipitation regime would increase forb and grass abundance (Ehleringer et al. 1991), enhancing pre-laying hen and brood-rearing habitat, but decrease sagebrush abundance and size (Ehleringer et al. 1991, Cook and Irwin 1992, Neilson et al. 1992, Poore et al. 2009), degrading nesting and wintering habitat. Lekking habitat may become limited to recent burns as productivity increases replace the SD Group with grassland. Expected increases in grass density and height may reduce chick survival as movement became difficult and predators have greater cover (Aldridge 2005). Patches of sage-grouse habitat may well be too small, too disconnected, and of too low quality to provide for viable populations.

4.5 Conclusions and Management Implications

Using scenarios of alternative futures allows exploration of the implications to land management strategies and objectives arising from uncertainty in outcomes (Peterson et al. 2003, Millar et al. 2007, Galatowitsch et al. 2009, Glick and Stein 2010). The three scenarios warmed by differing amounts and included both decreases and increases in precipitation and shifts in seasonality. All three scenarios indicate a large contraction in sage-grouse habitat should be expected. At present, the Malheur High Plateau is considered a sage-grouse stronghold with relatively intact habitat (Connelly et al. 2004). Under S1, sage-grouse habitat shifts to the higher elevations, under S2 it contracts to the lower and mid-elevations, and under S3 it largely disappears. Sage-grouse habitat recovery following stand-replacing disturbances such as fire, severe drought and, possibly, severe insect outbreaks, would be slower than present under the drier conditions of S1 and S2 but potentially quicker under the different precipitation regime of S3. Scenarios 1 and 2 carry a high risk of shifting to a cheatgrass-dominated state following such a disturbance (D'Antonio and Vitousek 1992, Bradley 2009, Epanchin-Niell et al. 2009) while S3 apparently does not (Bradley 2009).

Many conservation biologists have developed recommendations for adapting to changing climate (Heller and Zavaleta 2008). Several approaches should be pursued simultaneously to cope with the uncertainties over which future scenario is more likely to occur, with a goal of developing policies and actions that are robust to those uncertainties (Peterson et al. 2003, Millar et al. 2007, Heller and Zavaleta 2008). Increasing resistance to and resilience towards climate change are short-term strategies intended to retain as much of the existing sagegrouse habitat as possible and restore degraded habitat where possible (Millar et al. 2007). Resistance and resilience strategies that could work under all three scenarios are to reduce non-climatic stressors, such as overgrazing, manage juniper encroachment, and increase post-fire seeding success (see Chapter 3). In reducing juniper, traditional management approaches may need to be applied in different seasons and contexts than present, but new approaches that successfully remove juniper while retaining more sagebrush, particularly in the earliest stages of juniper encroachment, are needed as well (Miller et al. 2005, Millar et al. 2007). Improvements in postfire seeding success may require new or different seeding technologies, different seed mixes, and use of wider seed zones for appropriate germplasma for the changing conditions (Galatowitsch et al. 2009, Mawdsley et al. 2009, James and Svejcar 2010). Seed mixes should include species thought to be more resilient to the anticipated changes, which may mean use of non-native species and species from nearby analog climates (Galatowitsch et al. 2009, Mawdsley et al. 2009). Additionally, better assessment methods are needed to determine when post-fire seeding is actually necessary (Ratzlaff and Anderson 1995, Brooks 2005, James and Svejcar 2010).

Longer-term strategies should assist the system in responding to climate change, with different approaches needed under each scenario. The most common recommendation is to facilitate or assist the migration of desirable plant species into the area (Millar et al. 2007, Heller and Zavaleta 2008) although some conservation biologists do not believe we know enough to assist migration without causing significant, unwanted side effects (Ricciardi and Simberloff 2009). In the Malheur High Plateau, facilitating in-migration of Utah juniper, Utah-western juniper hybrids, pinyon, or Gambel oak likely would not be desired for sage-grouse habitat management. However, facilitating the migration of genotypes of sagebrush, grasses, and forbs better adapted to future conditions may well be desirable given that populations of a species may be adapted to a narrower range of conditions than the species as a whole (O'Neill et al. 2008). Longer-term strategies should focus on managing ecological processes and not specific structures or conditions (Millar et al. 2007, Mawdsley et al. 2009) although doing so can be difficult when the goal is conservation of a particular species. Managers should expect surprises and threshold behavior in plant and ecosystem response and not steady or particularly predictable change.

Table 4.1. Observed climate trends over the 20th C in the Pacific Northwest (Mote 2003, Climate Impacts Group 2004) and Oregon Climate Division 7. Pacific Northwest data cover variable periods, depending on data source, while Oregon Climate Division 7 data cover 1895-2009 and Malheur High Plateau Snotel data cover 1939-2010.

	Pacific Northwest	Oregon Climate Division 7	
Average Annual Temperature	+0.8°C	+0.6°C	
Average Annual Precipitation	+37.6%	+1.5%	
Average Seasonal Temperature			
Winter	+2.4°C	+0.7°C	
Spring	+0.3°C	-0.6°C	
Summer	+0.6°C	+0.6°C	
Fall	No change	+0.2°C	
Average Seasonal Precipitation			
Winter	+38%	-24%	
Spring	+51%	+28%	
Summer	+71%	+95%	
Fall	+19%	-29%	
Average April 1 Snowpack	Decreased >30% below 1800 m	Decreased 17% below 2100 m ^a Increased 12% above 2300 m ^b	

^a Silvies Snotel station ^b Fish Creek Snotel station

	Historical		Future			
Factors	1895- 2009 ¹	1970- 1999 ²	Scenario 1	Scenario 2	Scenario 3	
Average Annual Temperature	7.95°C	8.11°C	12.36°C	10.91°C	8.99°C	
Average Annual Precipitation	282.95 mm	272.03 mm	262.58 mm	300.89 mm	311.46 mm	
Precipitation Regime	Winter- Spring	Winter- Spring	Weakly Spring- Summer	Winter-Spring	Spring- Summer	
AE/PE	0.49	0.49	0.37	0.45	0.51	
Average Annual Soil Moisture Deficit	291.01 mm	282.31 mm	445.33 mm	373.00 mm	286.30 mm	
Biome ³	Shrub- steppe	Shrub- steppe	Salt Desert Shrub/Grassland	Shrub- steppe/ Woodland	Woodland/ Grassland	

Table 4.2. Historical values and climate change scenarios for the Malheur High Plateau based on projections for the Pacific Northwest generally and observed long-term trends in Oregon Climate Division 7.

¹ Long-term observed trend in Oregon Climate Division 7 and basis for Scenario 3

 $^{\rm 2}$ Basis for PNW projections and Scenarios 1 and S2

³ Based on Neilson et al. 1992 and local information

	Warm-Moist	Cool-Moist		Shallow-
Characteristic	(WM)	(CM)	Warm-Dry (WD)	Dry (SD)
Modal plant association	<i>Artemisia</i> <i>tridentata</i> Nutt. ssp. <i>tridentata/</i> <i>Leymus</i> <i>cinereus</i> (Scribn. & Merr.) A. Löve	<i>Artemisia tridentata</i> Nutt. ssp. <i>vaseyana</i> (Rydb.) Beetle/ <i>Festuca</i> <i>idahoensis</i> Elmer	Artemisia tridentata Nutt. ssp. wyomingensis Beetle & Young/Pseudoroegne ria spicata (Pursh) A. Löve-Achnatherum thurberianum (Piper) Barkworth	<i>Artemisia arbuscula</i> Nutt./ <i>Poa secunda</i> J. Presl
Dominant soil moisture regime Dominant soil temperature	Xeric Mesic	Xeric Frigid	Aridic Mesic	Aridic Mesic to
eaime		5		frigia
General soil Jepth	Deep to very deep	Moderately deep to deep	Shallow to moderately deep	Very shallow to shallow
Years producing at least 672 kg ha ⁻¹ of perennial grass	All years	High and average productivity years	High productivity years	No years

Table 4.3. Characteristics of each sagebrush group including modal potential natural plant community, soil moisture and temperature regimes, general soil depth, and grass productivity.

Table 4.4. Matrix of common combinations of soil temperature regimes and soil moisture regimes associated with the dominant species or plant functional types mapped by the Natural Resources Conservation Service (NRCS) in the western United States. Soil temperature regimes are arranged by warmest to coolest; soil moisture regimes by driest to wettest that occurs under the current climate or may occur under future climate.

		Soil Temperature Regimes			
		Thermic	Mesic	Frigid	Cryic
Soil Moisture Regimes	Aridic	Salt Desert Shrub	Warm-Dry Sagebrush Shallow-Dry Sagebrush Cheatgrass	Shallow-Dry Sagebrush	
	Xeric	Desert Grassland	Warm-Moist Sagebrush Juniper Woodland Cheatgrass	Cool-Moist Sagebrush Juniper Woodland	Cool-Moist Sagebrush Aspen
	Ustic		Conifer forest	Conifer forest Aspen	Conifer forest Aspen



Figure 4.1. Location of the study area in Oregon. The High Desert Province Ecological Province and the Malheur High Plateau Major Land Resource Area occupy approximately the same area, with the exception of the area to the east of Steens Mountain. The Malheur High Plateau Major Land Resource Area includes some area to the east of Steens Mountain while the High Desert Ecological Province does not. The area to the east of Steens Mountain lies within the rain shadow of the mountain and has a different climate. The town of Burns is the largest community within the study area.



Figure 4.2. Hypothetical arrangement of major cover types of the Malheur High Plateau under the current climate and three potential future climates. Average moisture increases as elevation increases while average temperature declines. Scenario 1 climate is warmer and drier than present. Scenario 2 climate is warmer and wetter in winter than present. Scenario 3 climate is warmer and wetter in summer than present. Breakpoints between the major cover types are approximate.

Chapter 5: General Conclusions

Prior to 1850, herbivory by insects and small ungulates may have been more important to the quantity and quality of sage-grouse habitat on the Malheur High Plateau than previously recognized. Fire was a key factor as well, but perhaps not as significant as commonly perceived. Past conditions appeared to have provided as much breeding habitat and brood-rearing habitat as sage-grouse biologists recommended, although the brood-rearing habitat was heavily skewed towards early brood-rearing, but only about three quarters as much wintering habitat (Connelly et al. 2004). Modeling results suggested that either sage-grouse do not need as much wintering habitat as recommended, or that lower levels of wintering habitat constrained sage-grouse populations.

Under current conditions, overgrazing during severe drought, postfire seeding success, juniper expansion, and the annual treatment rate of newly developing juniper woodlands along with the presence of cheatgrass are now the primary drivers of sage-grouse habitat in southeastern Oregon. These factors have reduced the apparent influence of the past factors. Although the current conditions of the Malheur High Plateau have not yet degraded as far as the models project, model results suggest the current trajectory will not provide sufficient nesting, brood-rearing, and wintering habitat over the long-term. The models also indicated that passive management would not reverse this trajectory and could accelerate the rate of degradation.

In the future, warmer temperatures and shifts in the precipitation regime would likely result in significant reductions in sage-grouse habitat availability and quality. Whether management activities could prevent the changes projected is very uncertain, but would be necessary to

slow the rate of conversion to other vegetation types. Under two of the scenarios analyzed, cheatgrass would remain problematic, but could decline under the third scenario. Aggressive control of juniper and cheatgrass likely would be needed, regardless. Improvements in postfire seeding success rates are needed to ensure desirable grass and forb species are present and undesirable species are controlled. New juniper treatment methods are needed that retain more sagebrush while also reducing the extent of juniper seedlings and saplings.

While I used this study to evaluate sage-grouse habitat, the same models could be used to assess other elements of management interest, such as expected forage for livestock or feral horses, habitat for other sagebrush obligate species such as pygmy rabbit, fire risks, and potential biomass production. Some elements included in the models may not be necessary for evaluating other elements. For example, I included the potential damage from vole outbreaks, even though only a small percentage of the Cool-Moist Sagebrush Group was affected, since the typical sites where voles cause damage are also very productive. In keeping sagebrush cover limited, I speculated that these sites would also provide some of the highest quality late brood-rearing habitat, rich in forbs that remained succulent later into the summer than nearby locations where sagebrush cover was denser or forbs cured earlier. If vole damage were not a significant consideration, the user can disable that factor. However, these models could be missing other disturbance factors important for a different wildlife species or other ecosystem services.

Using climate variables as much as possible to construct state-and-transition models of past and present conditions provided an objective approach for developing reference conditions and assessing the trajectory of the current vegetation in southeastern Oregon. Understanding what aspects of a given vegetation type could be important drivers of the specific focus of analysis, such as understanding the different types of seasonal habitat used by sage-grouse, is a critical first step in designing a state-and-transition model. Reviewing the literature greatly aided in determining which factors might have been important in creating the different types of seasonal habitat as well as identifying whether climate variables could be used as a basis for transition probabilities and which climate variables.

Additional Research Needs. The literature review also revealed several areas of needed research to strengthen these and similar models of sagebrush ecosystems. Little is known about many of the natural disturbance agents that influence sagebrush dynamics. The current literature suggests that scientific attention is turned to natural disturbances other than fire only when a very large event occurs. For example, only a small handful of studies have concerned the biology of sagebrush defoliators, such as aroga moth or chrysomelid beetles (Pringle 1960, Hsaio 1986, Haws et al. 1990). The timing of these studies suggests scientific interest developed after a multi-state outbreak (Gates 1964, Hall 1965) but this interest was not sustained. In forest ecosystems, annual insect and disease surveys conducted by the U.S. Forest Service have provided a wealth of information concerning the start, end, size, duration, and effects of many insect outbreaks. Although researchers still do not understand the drivers behind defoliator outbreaks (Meyers 1988, Cooke and Roland 2000, Maron et al. 2001, Cooke and Lorenzetti 2006) knowing the characteristics of outbreaks allows one to develop probabilities, modifiers, and appropriate transitions. Further, little is known about the actual impact of insects on sagebrush and sage-grouse habitat at the landscape scale. Since management activities interact with natural disturbances, better identification of which disturbances have a significant effect on sagebrush ecosystem dynamics, the conditions associated with the occurrence of those disturbances, and the type and variation in the

impacts could become more critical as changing climate alters disturbance types and regimes, and legal requirements to preserve or sustain sagebrush-dependent and associated species come into play.

Similarly, little is known about the characteristics of droughts severe enough to kill sagebrush. The only documented accounts of drought-caused mortality are from the 1930s drought and none provide sufficient information to understand the degree or extent of mortality (Ellison and Woolfolk 1937, Pechanec et al. 1937, Allred 1941). However, the beginning and end of drought is difficult to define (Byun and Wilhite 1999), the term has numerous definitions, depending on whether the focus is on crops, water supply or some other factor (Thurow and Taylor 1999), and most systems of calculating drought intensity include variables that are hard to measure with accuracy (Alley 1984, Byun and Wilhite 1999). As a result, there is little agreement on when the 1930s drought began and ended and how intense it was. Further, drought may kill plants through carbon starvation or through hydraulic failure, in which drought duration and intensity play different roles (Hanson and Weltzin 2000, Breshears et al. 2005, McDowell et al. 2008). Seasonal photosynthetic characteristics of sagebrush (DePuit and Caldwell 1973) suggest that sagebrush mortality from drought may be due to combinations of drought duration and intensity. Greater understanding of the role drought may play in rangeland dynamics is needed to better evaluate how forecasts of increasing drought frequency, intensity, and duration may affect the plant communities and ecosystem services of southeastern Oregon.

Lastly, I based inclusion of voles, freezekill, and snow mold on accounts of sagebrush mortality in other locations in the western United States without certainty they occurred in the Malheur High Plateau. All information on snow mold is from a few studies in Wyoming, Utah, and Colorado (Nelson and Sturges 1986, Sturges and Nelson 1986, Sturges 1989, 1993). Freezekill is based on a single documented occurrence, but one that was widespread enough to attract scientific attention (Hanson et al. 1982, Nelson and Tiernan 1983). While not specifically documented in Oregon, I assumed it occurred here as well. The inclusion of voles is also based on a small number of papers that documented vole-related mortality in other states (Hubbard and McKeever 1961, Mueggler 1967, Parmenter et al. 1987, Sturges 1993). As a result, I may have included factors that are not applicable in the Malheur High Plateau and missed others that were important.

What about the needs for the future under a changing climate? This study suggests many potential research questions. Under what conditions might western juniper retreat given the apparent benefit the species has received from increasing atmospheric carbon dioxide concentrations (Knapp et al. 2001a, Soulé et al. 2003, 2004)? Could Utah juniper displace western juniper as conditions warm? Sage-grouse were apparently able to survive and persist during the multi-decadal droughts of the Medieval Warm Period, possibly by shifting their habitat to the high basins on Steens Mountain and in the nearby Blue Mountains and eastern Cascade Mountains. Assuming that were true, how likely is it that these same refugia might be available in the future or that new refugia develop? Will the rate of change in habitat availability and quality exceed the capability of sagebrush and sagebrush-associated species to adapt?

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APPENDICES

Appendix A: Sagebrush Groups and Included Plant Associations

Warm-Moist Sagebrush Group

Ecological Site Number	Ecological Site Name	Plant Association
023XY104OR	Loamy Bottom	basin big sagebrush/basin wildrye
023XY202OR	Swale 10-14	basin big sagebrush/basin wildrye- bluebunch wheatgrass
023XY310OR	North Slopes 12-16	mountain big sagebrush-antelope bitterbrush/Idaho fescue-bluebunch wheatgrass-basin wildrye
023XY316OR	Droughty Loam 11-13	basin big sagebrush/Idaho fescue-Thurber's needlegrass-bluebunch wheatgrass
023XY404OR 023XY618OR	Deep North 12-18 Dry floodplain 8-10	mountain big sagebrush-antelope bitterbrush/Idaho fescue-needlegrass basin big sagebrush/basin wildrye
023XY516OR	Stony Loam 10-12	mountain big sagebrush-antelope bitterbrush/Idaho fescue-Thurber's needlegrass

Cool-Moist Sagebrush Group

Ecological Site Number	Ecological Site Name	Plant Association
023XY019OR	Silt Loam Terrace 10-12	basin big sagebrush/bluebunch wheatgrass- basin wildrye
023XY210OR	Pumice 10-12	mountain big sagebrush-antelope bitterbrush/Idaho fescue-western needlegrass-Ross' sedge
023XY213OR	Sandy Loam 10-12	basin big sagebrush/needle-and-thread- Thurber's needlegrass-basin wildrye
023XY219OR	Shallow Ashy 10-12	Wyoming big sagebrush/bluebunch wheatgrass-Idaho fescue
023XY301OR	Droughty South Slopes 11- 13	basin big sagebrush-Wyoming big sagebrush/bluebunch wheatgrass-Thurber's needlegrass
023XY302OR	South Slopes 12-16	antelope bitterbrush-basin big sagebrush/bluebunch wheatgrass
023XY303OR	Sandy Slopes 10-12	antelope bitterbrush-basin big sagebrush/needle-and-thread-Indian ricegrass

Ecological Site Number	Ecological Site Name	Plant Association
023XY308OR	North Slopes 10-12	Wyoming big sagebrush-basin big sagebrush/Idaho fescue-bluebunch wheatgrass
023XY314OR	Gravelly North Slopes 12- 16	threetip sagebrush/Idaho fescue-Sandberg bluegrass-Cusick's bluegrass
023XY318OR	Loamy 12-16	basin big sagebrush-mountain big sagebrush/Idaho fescue-Thurber's needlegrass-bluebunch wheatgrass
023XY406OR	Swale 12-16	mountain big sagebrush-snowberry/basin wildrye-Cusick's bluegrass
023XY412OR	Gravelly Ridge 16+	low sagebrush/rough fescue-Idaho fescue
023XY501OR	Shallow Loam 16-25	fescue
023XY509OR	Subalpine Slopes 16-35	mountain big sagebrush-mountain snowberry/Idaho fescue
023XY512OR	Dry Lakebed 10-12	mountain big sagebrush/Idaho fescue- Thurber's needlegrass
023XY515OR	Droughty Ashy 9-12	mountain big sagebrush/Idaho fescue- bluebunch wheatgrass-Thurber's needlegrass
		Wyoming big sagebrush-basin big sagebrush/bluebunch wheatgrass-Thurber's
023XY604OR	Arid plains 8-11	needlegrass
023X16090R	Droughty bottom 8-10	basin big sagebrusn/basin wildrye
023XY613OR	Pumice north slopes 10-12	bluebunch wheatgrass

Warm-Dry Sagebrush Group

Ecological Site Number	Ecological Site Name	Plant Association
023XY212OR	Loamy 10-12	Wyoming big sagebrush/Thurber's needlegrass-bluebunch wheatgrass- bottlebrush squirreltail
023XY216OR	Claypan 12-16	stiff sagebrush/Idaho fescue-bluebunch wheatgrass-Thurber's needlegrass
023XY220OR	Clayey 10-12	Wyoming big sagebrush/bluebunch wheatgrass-Sandberg bluegrass
023XY300OR	South Slopes 8-12	Antelope bitterbrush-Wyoming big sagebrush/bluebunch wheatgrass-Thurber's needlegrass
023XY312OR	Shallow North 12-16	low sagebrush/Idaho fescue-bluebunch wheatgrass

Ecological		
Site Number	Ecological Site Name	Plant Association
000277000	Doon Loomy 12 16	mountain big sagebrush-antelope bitterbrush/Idaho fescue-Thurber's
023713210R	Deep Loaniy 12-16	neeulegiass
023XY400OR	Loamy 16-20	fescue
023XY502OR	Loamy 25-35	mountain big sagebrush/Idaho fescue-sedge
023XY507OR	Claypan 16-25	low sagebrush/Idaho fescue-onespike oatgrass
023XY514OR	Pumice 8-10	mountain big sagebrush/Idaho fescue- Thurber's needlegrass
023XY606OR	Shrubby pumice plains 8-11	mountain big sagebrush-antelope bitterbrush/needle-and-thread-western needlegrass
023XY607OR	Pumice plains 8-11	mountain big sagebrush-rubber rabbitbrush/needle-and-thread-western needlegrass
023XY608OR	Droughty pumice plains 8- 11	basin big sagebrush-rubber rabbitbrush/needle-and-thread-western needlegrass
023XY612OR	Pumice south slopes 10-14	mountain big sagebrush-antelope bitterbrush/Idaho fescue-bluebunch wheatgrass-Sandberg's bluegrass

Shallow-Dry Sagebrush Group

Ecological		
Site Number	Ecological Site Name	Plant Association
023XY211OR	Pumice Claypan 10-12	stiff sagebrush/Idaho fescue-needlegrass
		low sagebrush/bluebunch wheatgrass-
023XY214OR	Claypan 10-12	Sandberg bluegrass
	Shallow Gravelly Loam 10-	low sagebrush/Thurber's needlegrass-
023XY215OR	12	bluebunch wheatgrass
023XY218OR	Thin Surface Claypan 10-16	low sagebrush/Sandberg bluegrass
		Wyoming big sagebrush/Idaho fescue-
023XY221OR	Gravelly Terrace 10-12	western needlegrass-Thurber's needlegrass
023XY324OR	Shallow Swale 10-14	low sagebrush/Sandberg bluegrass-LETR5
		low sagebrush/Idaho fescue-bluebunch
023XY410OR	Gravelly Ridge 12-16	wheatgrass
		mountain big sagebrush/western
023XY508OR	Pumice Flat 10-12	needlegrass-Ross' sedge-squirreltail
		low sagebrush/bluebunch wheatgrass-
023XY600OR	Shallow south slopes 8-12	Thurber's needlegrass
		basin big sagebrush/needle-and-thread-
023XY619OR	Dry sandy loam 8-10	Indian ricegrass

Ecological Site Number	Ecological Site Name	Plant Association
023XY635OR	Loamy 8-10	Wyoming big sagebrush-basin big sagebrush/Thurber's needlegrass-Indian ricegrass-bluebunch wheatgrass
023XY636OR	Shallow loam 8-10	Wyoming big sagebrush-basin big sagebrush/Thurber's needlegrass-Indian ricegrass-bluebunch wheatgrass
023XY638OR	Droughty south 8-10	Wyoming big sagebrush-basin big sagebrush/bluebunch wheatgrass-Thurber's needlegrass
023XY649OR	Droughty sandy slopes 10- 12	basin big sagebrush/needle-and-thread- bluebunch wheatgrass
023XY650OR	Dry ponded clay 6-10	basin big sagebrush/beardless wildrye- squirreltail
023XY666OR	Stipa fescue plains 8-10	mountain big sagebrush/needle-and-thread- Idaho fescue
023XY670OR	Stipa fescue basin 8-11	basin big sagebrush/needle-and-thread- Idaho fescue

Appendix B: List of Abbreviations

BLM	Bureau of Land Management
CM Group	Cool-Moist Sagebrush Group
ES	Early Seral
J1	Phase I Juniper
J2	Phase II Juniper
33	Phase III Juniper
LSO	Late Seral Open
LSC	Late Seral Closed
MSO	Midseral Open
OG	Old Growth
S1	Scenario 1
S2	Scenario 2
S3	Scenario 3
SD Group	Shallow-Dry Sagebrush Group
VDDT	Vegetation Dynamics Development Tool
WD Group	Warm-Dry Sagebrush Group
WM Group	Warm-Moist Sagebrush Group