Research

Ecography 33: 198–208, 2010
doi: 10.1111/j.1600-0587.2009.05684.x
© 2010 The Author. Journal compilation © 2010 Ecography
Subject Editor: Jens-Christian Svenning. Accepted 7 May 2009

Assessing ecosystem threats from global and regional change: hierarchical modeling of risk to sagebrush ecosystems from climate change, land use and invasive species in Nevada, USA

Bethany A. Bradley

B. A. Bradley (bethanyb@princeton.edu), Woodrow Wilson School, Princeton Univ., Princeton, NJ 08544, USA.

Global change poses significant challenges for ecosystem conservation. At regional scales, climate change may lead to extensive shifts in species distributions and widespread extirpations or extinctions. At landscape scales, land use and invasive species disrupt ecosystem function and reduce species richness. However, a lack of spatially explicit models of risk to ecosystems makes it difficult for science to inform conservation planning and land management. Here, I model risk to sagebrush (*Artemisia* spp.) ecosystems in the state of Nevada, USA from climate change, land use/land cover change, and species invasion. Risk from climate change is based on an ensemble of 10 atmosphere-ocean general circulation model (AOGCM) projections applied to two bioclimatic envelope models (Mahalanobis distance and Maxent). Risk from land use is based on the distribution of roads, agriculture, and powerlines, and on the spatial relationships between land use and probability of cheatgrass *Bromus tectorum* invasion in Nevada. Risk from land cover change is based on probability and extent of pinyon-juniper (*Pinus monophylla*; *Juniperus* spp.) woodland expansion. Climate change is most likely to negatively impact sagebrush ecosystems at the edges of its current range, particularly in southern Nevada, southern Utah, and eastern Washington. Risk from land use and woodland expansion is pervasive throughout Nevada, while cheatgrass invasion is most problematic in the northern part of the state. Cumulatively, these changes pose major challenges for conservation of sagebrush and sagebrush obligate species. This type of comprehensive assessment of ecosystem risk provides managers with spatially explicit tools important for conservation planning.

Applying scientific knowledge to management planning is becoming more important as threats to ecosystems from global change increase (Millenium Ecosystem Assessment 2003). Climate change affects species physiology (Root et al. 2003), poses a threat to richness of native species (Thuiller et al. 2005, Araujo and Rahbek 2006), and is likely to increase the rate of extirpations and extinctions (Thomas et al. 2004). Land use/land cover change also threatens native systems as humans increasingly appropriate natural resources (Vitousek et al. 1997, Foley et al. 2005, Turner et al. 2007). Non-native invasive species are an additional driver of global change, displacing native species and affecting ecosystem processes (Vitousek et al. 1996). Comprehensive assessments of risk from these components of global change are needed for setting successful, long-term conservation and management priorities.

At regional scales, climate change poses a substantial threat to ecosystems. Climate change may lead to large-scale shifts in species distribution, which will require new management strategies for conservation and restoration as the definition of ''native'' species changes (Harris et al. 2006, Millar et al. 2007). Climate change is also likely to create novel climate conditions that currently do not occur, which adds further uncertainty to conservation planning (Williams and Jackson 2007). Regional modeling of relationships between native ecosystems and climate can help to identify the distribution of ecosystems at risk from climate change as well as locations where species are most likely to remain viable within their current range.

Many studies have looked at the impacts of climate change on species distributions (Bakkenes et al. 2002, Iverson and Prasad 2002, Thuiller et al. 2005). Climate variables can be used to model habitat using bioclimatic envelope modeling (BEM) (Guisan and Zimmermann 2000, Guisan and Thuiller 2005), which in turn can be projected onto future climate projections from atmosphere-ocean general circulation models (AOGCMs) (IPCC 2007, Randall et al. 2007) to forecast future habitat. Due to inconsistencies in AOGCM projections (Randall et al. 2007) and differences between BEM methodologies (Elith et al. 2006, Tsoar et al. 2007), Araujo and New (2007) recommend using an ensemble approach to model changes to species distribution. An ensemble involves combining the projections of multiple bioclimatic envelope and/or circulation models to identify consistencies, which are likely more robust than any single projection. Distributional modeling

provides species-specific assessments of regional risk from climate change.

At landscape and local scales, land use and invasive species threaten native ecosystems. Land use/land cover change is often a more immediate concern than climate change, with disturbances such as appropriation of surface water and groundwater, exurban expansion, and energy development increasingly threatening ecosystems (Foley et al. 2005). Previous studies have shown that land use-related disturbances increase the likelihood of plant invasions (Gelbard and Belnap 2003, Bradley and Mustard 2006), and significantly alter bird species abundances (Leu et al. 2008). Understanding the spatially explicit risks from land use is therefore critical for planning effective land management and conservation strategies.

The process of translating scientific results into management planning can be arduous, as managers must learn and incorporate a range of modeling and experimental results into decision-making. There is a great need for scientists to provide planning tools in a framework that is user-friendly. One way to enhance the transfer of knowledge from science to practice is through spatially explicit mapping tools in a GIS environment (Rotenberry et al. 2006, Leu et al. 2008). Managers often have familiarity with GIS and may find this framework easier to apply. Due to the many different ecosystem threats from global change, risk models may relate to regional, landscape, or local scales. Further, different risks may be more or less of a priority depending on management goals and available resources. Modeling multiple risks in a hierarchical framework (Pearson and Dawson 2003, Pearson et al. 2004) provides scientific tools for developing robust short and long-term management strategies.

Here, I present a series of models of global change risk to sagebrush (*Artemisia* spp.) in Nevada, USA. Sagebrush shrublands are common across the intermountain western U.S. They encompass a range of sagebrush species (Table 1) as well as other shrubs and diverse perennial bunchgrasses. Sagebrush ecosystems provide critical habitat for vulnerable species such as greater sagegrouse *Centrocercus urophasianus* and pygmy rabbit *Brachylagus idahoensis*, as well as many other obligate species (Wisdom et al. 2000, Knick et al. 2003, Rowland et al. 2006). Sagebrush ecosystems are also used extensively for grazing by domesticated livestock, especially cattle.

Sagebrush shrublands exist across a strong latitudinal gradient in the western United States, suggesting adaptation to a range of temperature conditions. However, sagebrush shrublands transition to salt desert shrublands (characterized by *Atriplex* spp.) in areas receiving <200 mm of annual precipitation (Houghton et al. 1975) and into pinyon-juniper woodland at higher elevations and precipitation levels, possibly due to greater competition for deep soil water (Leffler and Caldwell 2005). Hence, the distribution of sagebrush habitat is likely to shift if precipitation patterns are altered with climate change.

Table 1. Species included in the regional distribution of sagebrush used to create the bioclimatic envelope model.

Common name	Species
Basin big sagebrush	*Artemisia tridentata* ssp. *tridentata*
Wyoming big sagebrush	*Artemisia tridentata* ssp. *wyomingensis*
Threetip sagebrush	*Artemisia tripartita*
Black sagebrush	*Artemisia nova*
Low sagebrush	*Artemisia arbuscula*
Rigid sagebrush	*Artemisia rigida*
Silver sagebrush	*Artemisia cana*

In addition to climate change, a major threat to sagebrush ecosystems is invasion by non-native species such as cheatgrass, a Eurasian annual that increases regional fire frequency (Whisenant 1990, D'Antonio and Vitousek 1992). Changes in land use and climate may also affect the relationship between cheatgrass and sagebrush. For example, roads and other forms of infrastructure have been shown to increase the probability of cheatgrass establishment (Gelbard and Belnap 2003, Bradley and Mustard 2006), and invasion rates are affected by precipitation and temperature conditions (Chambers et al. 2007).

Expansion of native pinyon-juniper woodland also threatens sagebrush ecosystems (Tausch et al. 1981, Miller and Rose 1999, Weisberg et al. 2007, Bradley and Fleishman 2008). Both climate and land-use change have been suggested as drivers of woodland expansion (Burkhardt and Tisdale 1976). As a result, expansion into sagebrush seems likely to continue with ongoing land-use and climate change.

In this paper, I first use an ensemble modeling approach to estimate likely change to climatic habitat of sagebrush based on the projections of two BEMs using 10 AOGCMs. Second, I use spatial modeling to develop state-wide landscape-scale risk assessments associated with land use and invasion of cheatgrass *Bromus tectorum*. Finally, I integrate previous experimental, observational, and geospatial studies to model risk of and expansion of pinyon-juniper (*Pinus monophylla*, *Juniperus* spp.) woodland into sagebrush ecosystems. All risk layers are available individually and can be used by land managers to customize maps of threatened and stable sagebrush ecosystems depending on management goals.

Materials and methods

Study area and species

Risk to sagebrush species is assessed for the state of Nevada, U.S. Nevada is characterized by basin and range topography, or north-south trending parallel mountain ranges which can reach over 3000 m in elevation. The southern part of the state is Mojave Desert, while the central and northern parts are Great Basin Desert. Sagebrush species are prominent in the Great Basin Desert. Most land in the state is publicly owned and managed by the Bureau of Land Management, U.S. Forest Service, or the Dept of Defense.

Sagebrush distributions were derived from Sagestitch, which is a compilation of the distribution of all sagebrush species derived from EPA/GAP land cover maps for the western U.S. (Comer et al. 2002). All species of basin sagebrush (Table 1) were included because the species frequently co-occur and because species-level differentiation

based on remote sensing may be inaccurate. Mountain big sagebrush *Artemisia tridentata* ssp. *vaseyana*, which occurs in high elevation montane ecosystems, was excluded from the analysis. I resampled the Sagestitch data to 0.04166 dd (~4 km) using a majority filter. This resolution was selected to correspond to the PRISM interpolated climate data set (Daly et al. 2002).

Invasive species and land use

Cheatgrass distributions were based on a Nevada state-wide map (Peterson 2006) that identifies cheatgrass based on the difference between early spring and early summer greenness. Cheatgrass has an earlier growing season than sagebrush and other native perennial shrubs and grasses. As a result, cheatgrass is more likely to be green in the early spring, but brown in the early summer, whereas native species are more likely to be brown in the early spring and green in the early summer (Peterson 2005). However, in drier systems of the Mojave Desert (southern Nevada), early season growth of natives is more common than in the Great Basin Desert (northern Nevada), and natives are more likely to be confused with cheatgrass. Therefore, Clark County (the southernmost county in Nevada) was excluded from analysis.

The state-wide cheatgrass map was initially produced in units of percent cover at 30 m resolution (Peterson 2006). These continuous values were reclassified to presence/absence based on 2325 validation points collected throughout northern Nevada between 2004 and 2006 (Bradley and Mustard 2005). In the resulting presence/absence map, 73% of observed points dominated by cheatgrass were correctly identified. The map had a 15% false positive rate, for an overall accuracy of 79%.

Geographical relationships between cheatgrass presence and distance to a given type of land use were developed for the state using the presence/absence map (Peterson 2006) and based on the methods described by Bradley and Mustard (2006). Probability of presence is defined as the number of pixels (map units) with cheatgrass present divided by the total number of pixels. Probability of presence was calculated for discrete ranks of distance to a given land use. For example, probability of cheatgrass presence was calculated for pixels within 30 m of any road, between 30 and 60 m of any road, and at discrete intervals out to a total distance of 2000 m from any road. The total probability of cheatgrass presence throughout the state was subtracted from the probability of cheatgrass presence at each discrete interval to determine the relative probability of cheatgrass presence at each discrete interval away from land use. Relative probabilities >0 indicated elevated invasion risk associated with land use.

The probability of cheatgrass presence was calculated relative to roads, agriculture, power lines, and elevation (Table 2). Invasion risk associated with distance from cheatgrass dominated lands was taken directly from Bradley and Mustard (2006). In that study, cheatgrass maps from 1973 and 2001 encompassing a portion of northwest Nevada were compared to calculate relative probability of cheatgrass presence in 2001 with distance to cheatgrass presence in 1973. This relationship, derived from a subset of the state, was used because state-wide maps of cheatgrass distribution in 1973 were not available.

Table 2. Summary of data sources.

Name	Description	Source
Agriculture	Agricultural fields in Nevada	USDA Aerial Photography Field Office
Roads	Paved and unpaved roads in Nevada	2000 census
Power lines	High voltage power lines	2000 census
Elevation	Elevation (m)	National Elevation Dataset (NED 1999)
Cheatgrass	Cheatgrass distribution in Nevada	Landsat-derived state map (Peterson 2006)
Sagebrush	Sagebrush distribution in the western U.S.	Sagestitch (Comer et al. 2002)
Pinyon-juniper	Pinyon-juniper distribution in Nevada	Southwest Re-GAP (USGS 2004)

Risk maps based on relative probability of invasion were created at a 30-m spatial resolution for the state. For land-use relationships (distance to roads, power lines, agriculture, and cheatgrass dominated lands), only positive relative probability values were retained. For elevation, both positive and negative relative probability values were retained. Risk of cheatgrass invasion for each pixel within the state was calculated by summing the individual relative probability layers. Any pixel with a relative probability value >0 is at some risk of cheatgrass invasion.

Native woodland expansion

Pinyon-juniper distribution in Nevada was based on remote sensing maps produced by the southwest ReGAP project at 30-m resolution (USGS 2004). All pixels identified as "Great Basin Pinyon-Juniper Woodland" by the land cover classification were included in the analyses.

Pinyon-juniper woodland is expanding into lower elevation sagebrush ecosystems (Tausch et al. 1981, Miller and Rose 1999, Weisberg et al. 2007, Bradley and Fleishman 2008). The relationship between distance to pinyon-juniper and probability of woodland expansion is unknown. Wisdom et al. (2003) assume some risk to all sagebrush from pinyon-juniper expansion, with elevated risk under higher precipitation regimes and adjacent to woodland. However, expansion over the short-term (decades) is limited by seed dispersal, primarily by birds and rodents (VanderWall and Balda 1981, VanderWall 1997, Chambers 2001). Dispersal by rodents occurs within 100 m of woodland (VanderWall 1997), while dispersal by birds averages 1–5 km from woodland (J. C. Chambers pers. comm.). I assumed elevated probability of loss of sagebrush within 5 km of woodland, with risk from pinyon-juniper expansion decreasing with distance from woodland. Pinyon-juniper expansion at the shrubland/woodland interface between 1986 and 2005 was observed on up to 20% of pixels in central Nevada (Bradley and Fleishman 2008). This value was used as the maximum probability of risk associated with woodland expansion.

Climate habitat

I calculated regional risk that sagebrush ecosystems will become non-viable in response to climate change using BEMs based on sagebrush distribution in the western U.S. from Sagestitch (Comer et al. 2002). Presence of sagebrush was compared spatially to 39 separate interpolated climate variables: monthly and annual average precipitation, temperature minimum, and temperature maximum from 1971 to 2000 (Daly et al. 2002).

Each of the 39 individual climate variables taken from the PRISM dataset (Daly et al. 2002) were ranked by how well they constrain sagebrush distribution relative to all of the western U.S. Constraint was calculated as the median distance from the sample mean for the total (western U.S.) population divided by the median distance from the sample mean for the sample (sagebrush) population (Bradley 2009). Constraint is similar to "specialization" defined by Hirzel (2002), except I use median distance rather than standard deviation of distance due to the large range of climate conditions in the western U.S. Higher constraint ratios indicate that the sample population encompasses a range of values increasingly smaller than the total population. Hence, variables with higher constraint ratios are considered better predictors of sagebrush distribution.

I ranked the 39 predictor variables based on their constraint ratios. Correlation coefficients for locations of sagebrush presence were then calculated, and predictor variables were selected based first on highest constraint ratio and second on low correlation with previously selected predictors. An R^2 threshold of <0.60 was used to reduce correlation between predictor variables. Goodness of fit was measured using a receiver operating characteristic (ROC) curve, which compares the fraction of total western U.S. pixels captured by the model to the fraction of sagebrush occurrences captured. A higher fraction of sagebrush relative to total pixels indicates a better fit.

The initial set of potential predictors was then reduced incrementally and change in the model fit based on the ROC curve was recalculated. The number of predictor variables was reduced until the point at which the model fit suffered. Limiting the number of variables helps to both identify the most important predictors of sagebrush presence and reduces overfitting of the model.

Once climate predictor variables were identified, I used two bioclimatic envelope methodologies to model sagebrush climatic habitat: Mahalanobis distance (MD) (Farber and Kadmon 2003, Tsoar et al. 2007), and Maximum entropy (Maxent) (Phillips et al. 2006). Both of these techniques perform well relative to other commonly used bioclimatic envelope models (Elith et al. 2006, Tsoar et al. 2007, Phillips and Dudik 2008). MD produces an ellipsoidal "envelope" around presence points in n-dimensional space (where n = the number of predictor variables). The shape of the ellipsoid is proportional to the covariance of the distribution of each climate predictor. Hence, if the range of values is small for a given predictor, the width of the envelope is correspondingly narrow. This approach reduces overprediction common in some other bioclimatic models (Tsoar et al. 2007). Smaller Mahalanobis distances are more likely to be within the bioclimatic envelope, and thus are more likely to be climatically suitable for sagebrush. MD was calculated using Matlab (2007).

Maxent models climatic habitat by calculating simple functions relative to each predictor variable based on species presence and absence (Phillips et al. 2006). The Maxent model was constructed using Maxent ver. 3.2.1. Larger Maxent values are more likely to be climatically suitable for the modeled species. To create a current model of sagebrush climatic suitability in the western U.S., I identified the thresholds of MD and Maxent that included 95% of the current mapped distribution of sagebrush shrubland.

I estimated the climatically suitable areas for sagebrush in 2100 using the two envelope models and an ensemble of 10 AOGCM projections based on IPCC scenario SRESA1B. This scenario is the "middle of the road" projection with CO_2 levels reaching 720 ppm by 2100 (Nakicenovic and Swart 2000). I calculated change in average monthly climate by subtracting mean 1970–2000 conditions from mean 2090–2100 conditions. Climate change by 2100 was added to current climate conditions (Daly et al. 2002) to create down-scaled ($\sim$4-km) future projections of mean monthly and annual precipitation and temperature. I calculated the future spatial distribution of climatic habitat by projecting sagebrush's bioclimatic envelope onto each AOGCM climate prediction. The bioclimatic envelope was based on the thresholds of Mahalanobis distance or Maxent values that encompassed 95% of the current sagebrush distribution. I summed the climatic habitat maps projected by each AOGCM to produce a final ensemble product (Araujo and New 2007) with values for each pixel ranging from 0 to 20, where 0 indicated that no AOGCM in either envelope model projected suitability in 2100 and 20 indicated that all AOGCMs in both envelope models projected suitability in 2100.

AOGCMs used in this study were from the following modeling groups: Canadian Centre for Climate Modelling and Analysis (CGCM3.1), Centre National de Recherches Meteorologiques (CM3), Geophysical Fluid Dynamics Laboratory (GFDL2.1), Goddard Inst. for Space Studies, Hadley Centre for Climate Prediction (CM3), Inst. for Numerical Mathematics (CM3), Inst. Pierre Simon Laplace (CM4), Model for Interdisciplinary Research on Climate (MIROC3.2), Max Planck Inst. for Meterology (echam5), National Center for Atmospheric Research (CCSM3).

Hierarchical risk modeling

The individual layers of risk of loss of sagebrush associated with invasive species, land use, expansion of pinyon-juniper, and climate change are stored separately within a GIS to enable users to assign their own criteria for assessing risk. For example, some users might rank cheatgrass invasion as a more immediate threat than climate change and choose to increase its relative weighting in a risk assessment. Landscape scale risk from land use has a spatial resolution of 30 m, while regional risk from climate change has a spatial resolution of 4 km. The hierarchical modeling framework, and examples of how sites could be evaluated based on short and long-term risk, is shown in Fig. 1. Using this framework, I present an example assessment in which I identify lands in Nevada with low short and long-term risk, which

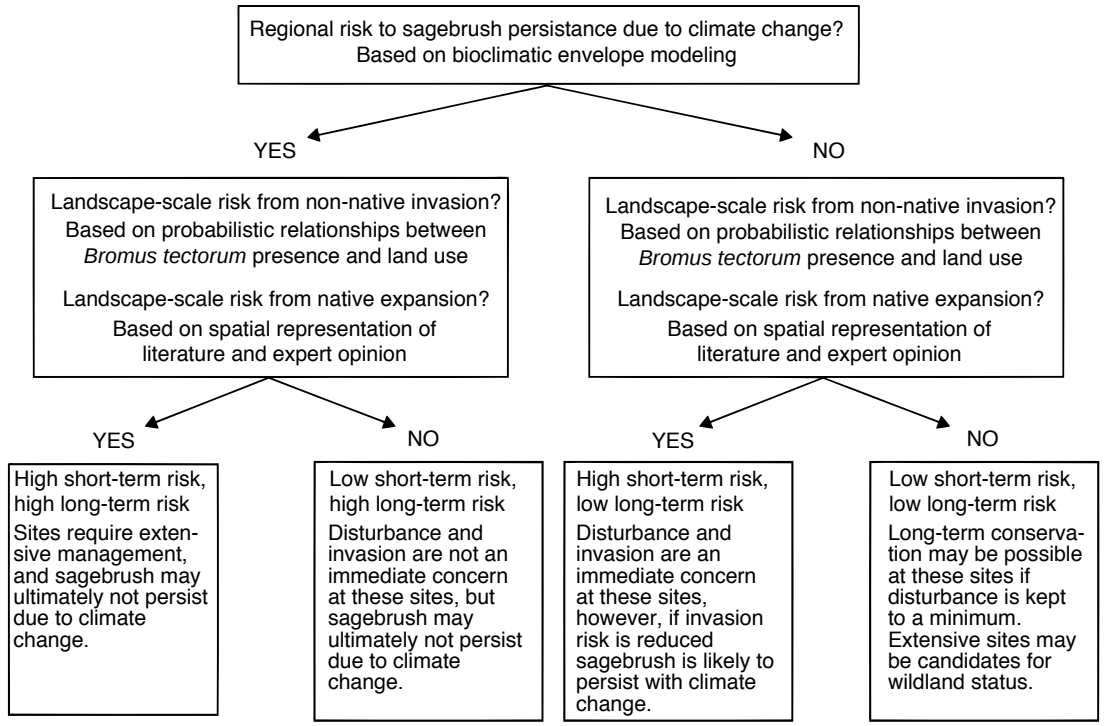

Figure 1. Analytical and management framework for assessing short and long-term risk to sagebrush (*Artemisia* spp.) ecosystems.

may be candidates for long-term conservation of sagebrush. Low risk areas of sagebrush shrubland were defined as pixels with $<5\%$ increased risk from proximity to land use, cheatgrass invasion, or pinyon-juniper expansion, and sustained climatic suitability projected by more than half of the combined BEM and AOGCM projections.

Results

On the basis of remote sensing analysis, sagebrush ecosystems were most prevalent in the northern, central, and eastern portions of Nevada (Fig. 2), with 95% of the sagebrush identified by Sagestitch (Comer et al. 2002) occurring in areas with annual precipitation >19 cm, but <52 cm. Average annual minimum temperatures for 95% of sagebrush shrubland range from -5.5°C to 3.9°C, while average maximum temperatures range from 10.8°C to 18.5°C.

Cheatgrass invasion is most extensive in the northwest corner of the state. The relationships between cheatgrass invasion and land use provide a framework for developing state-wide risk maps of increased probability of invasion of cheatgrass and other invasive species. Statewide, probability of cheatgrass invasion was elevated proximal to agriculture, power lines, and roads. Probability of invasion increased by 24% on pixels directly adjacent to agriculture, and was elevated to a distance of 4 km from agriculture (Fig. 3A). Probability of cheatgrass invasion was 10% higher on lands directly adjacent to high voltage power lines, which remove wide swaths of vegetation when they are emplaced, and was elevated to a distance of several kilometers from power lines (Fig. 3B). Probability of invasion of cheatgrass was 5% higher on lands directly adjacent to roads and was elevated to a distance of 500 m (Fig. 3C).

A previous study showed that distance to areas currently dominated by cheatgrass also influences probability of invasion. Lands directly adjacent to cheatgrass in 1973 had a 26% higher probability of cheatgrass presence 30 yr later than lands that were not adjacent to cheatgrass (Bradley and Mustard 2006). Although the relationship between past and current presence of cheatgrass was not derived from the full extent of the geographic distribution of cheatgrass, I assumed that the relationship holds throughout the state.

Across Nevada, probability of cheatgrass presence also was strongly related to topography. Elevations between 1300 and 1500 m a.s.l. were up to 30% more likely to be cheatgrass dominated. Elevations below 1300 and between 1500 and 1700 were 10% more likely to contain cheatgrass. Elevations above 1700 m were up to 15% less likely to contain cheatgrass (Fig. 4).

Risk of sagebrush loss associated with land use and cheatgrass invasion was highest in portions of the state already dominated by cheatgrass and near lands used for agriculture (Fig. 5). However, roads and powerlines also caused elevated risk of sagebrush loss and were present throughout the state.

Expansion of pinyon-juniper woodland was a risk to sagebrush at low elevations adjacent to current woodland. Expansion risk was prevalent throughout the state because pinyon-juniper woodland exists in almost all montane areas (Fig. 6). The highest expansion probability value of 0.20 was assigned to pixels within 1000 m of pinyon-juniper woodland. Pixels 1000–2000 m from woodland were assigned probabilities of 0.10, and pixels 2000–5000 m from woodland were assigned a probability of 0.05.

The best model for regional sagebrush presence used four climate predictor variables: June precipitation, June maximum temperature, August precipitation, and annual

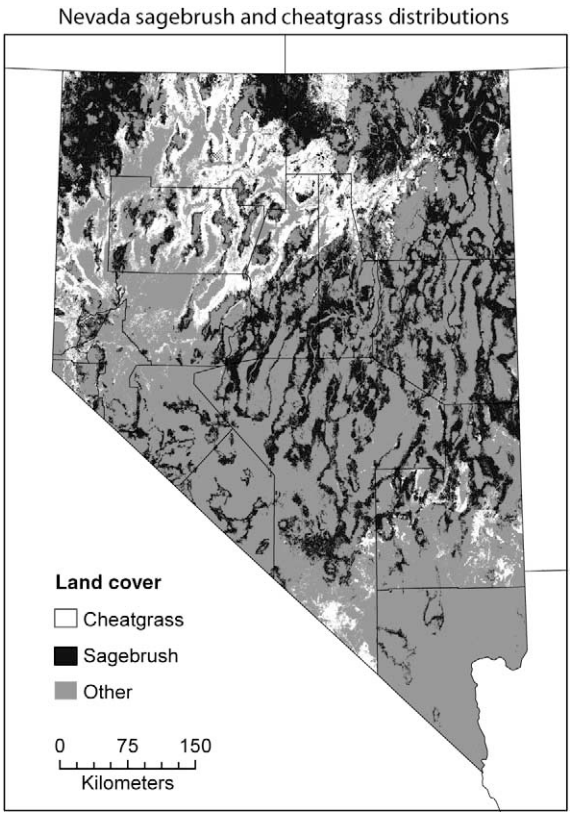

Figure 2. Native sagebrush (*Artemisia* spp.) is prevalent throughout the state of Nevada. Invasive cheatgrass *Bromus tectorum* poses a significant threat in the northwest portion of the state.

precipitation (Fig. 7). Although two of the climate predictors were summer precipitation variables, June and August precipitation were poorly correlated ($R^2 = 0.37$), making them reasonably independent. None of the four predictor variables had a correlation coefficient > 0.42 (Table 3).

The current distribution of sagebrush, and models of both current and future climatic suitability are shown in Fig. 8. Current projections of suitable climate based on the two BEM methodologies were quite similar, with differences only at the edge of sagebrush's range. Projections of future climatic suitability for sagebrush were also similar for the two envelope models (not pictured). An ensemble of the two BEMs $\times$ 10 AOGCM projections is shown in Fig. 8C. Regionally, there was considerable spatial heterogeneity in future climatic suitability for sagebrush. Areas with the lowest risk from climate change occurred in Wyoming, eastern Idaho, central Oregon, and northern Nevada. The majority of sagebrush in the state of Nevada, particularly locations further south, was at high risk from climate change.

As an example assessment of sagebrush ecosystems at low risk, pixels with elevated risk of sagebrush loss due to land use, cheatgrass invasion, and pinyon-juniper expansion of $< 5\%$ that maintained climatic suitability in at least half of the combined BEMs and AOGCMs are shown in Fig. 9. Lands meeting all of these criteria encompassed over 300 000 ha, but represented $< 3\%$ of all sagebrush shrubland in the state of Nevada. Sagebrush ecosystems at low risk occurred primarily in the northern and eastern parts of the state.

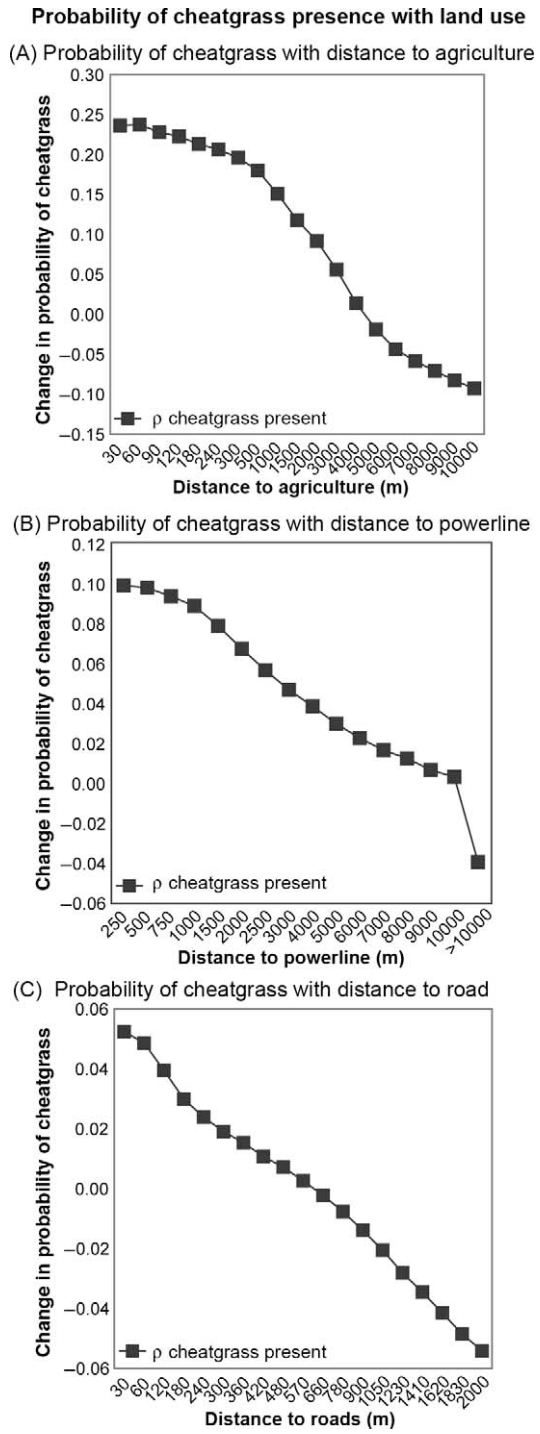

Figure 3. State-wide relationships between cheatgrass *Bromus tectorum* presence and land use. (A) Lands within 4 km of agriculture are more likely to contain cheatgrass. (B) Lands within 7 km of power lines are more likely to contain cheatgrass. (C) Lands within 500 m of roads are more likely to contain cheatgrass.

Discussion

Comprehensive assessments of ecosystems threats associated with local, regional, and global change are important for conservation prioritization. For state-level planning, spatially explicit risk models of land use/land cover change, invasive species, and climate change provide important

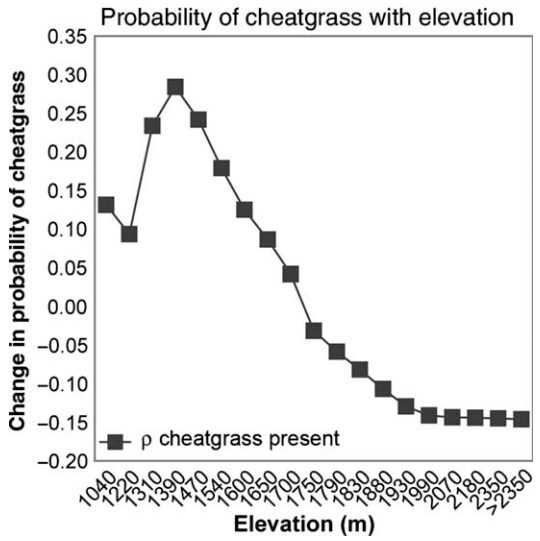

Figure 4. Cheatgrass *Bromus tectorum* presence is most likely between 1300 and 1500 m a.s.l. and is more likely at all elevations below 1700 m a.s.l.

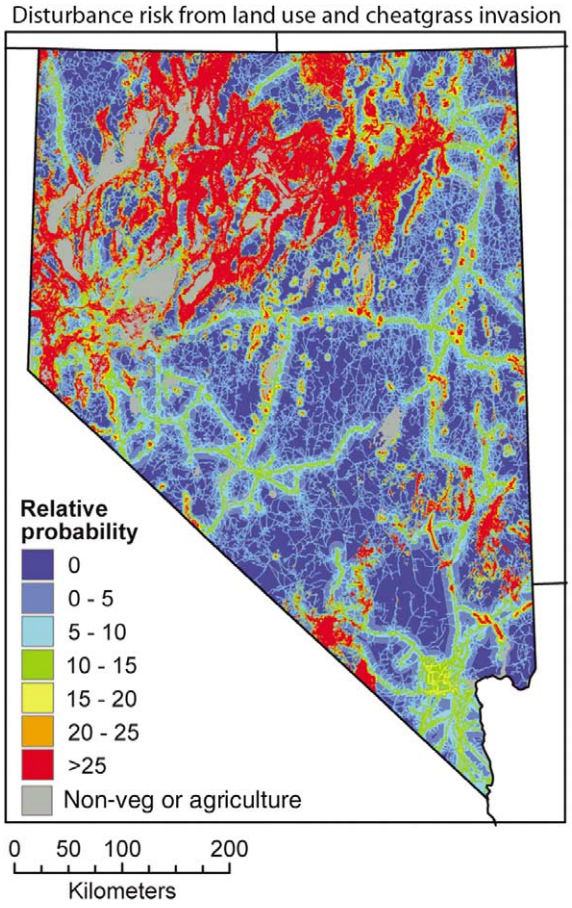

Figure 5. State-wide probability of change in cover of sagebrush (*Artemisia* spp.) associated with proximity to roads, power lines, and agriculture and proximity to cheatgrass *Bromus tectorum* dominated lands.

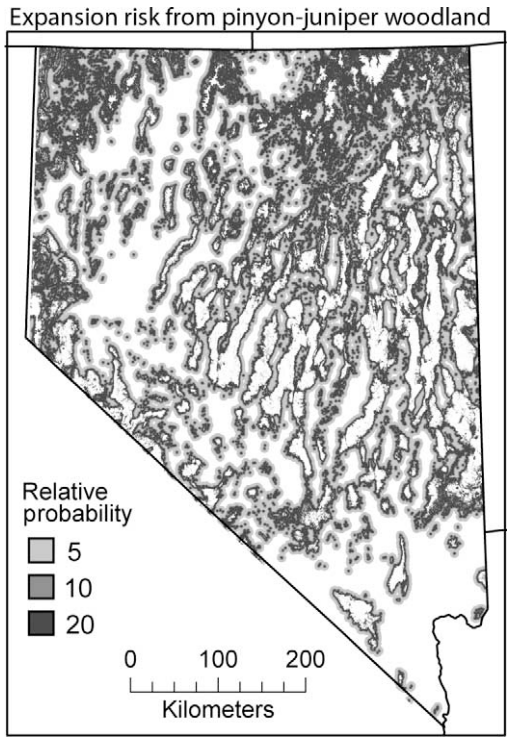

Figure 6. Locations within 5 km of the lower-elevation extents of pinyon-juniper (*Pinus* spp.; *Juniperus* spp.) are considered to have the highest risk of woodland expansion.

decision-making tools. Risk models may be based on empirical relationships, experimental or observational evidence, or expert opinion, but are most useful for management if provided in a GIS framework. The scope of threats to sagebrush ecosystems demonstrates the importance of including multiple risk assessments into management planning.

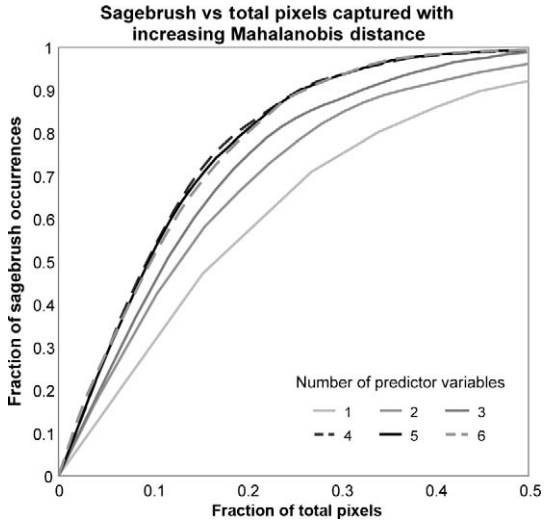

Figure 7. Receiver operating characteristic (ROC) curves show the fraction of western U.S. pixels vs the fraction of sagebrush pixels captured with increasing Mahalanobis distance (not pictured). The fit improves with the addition of the first 4 climate predictor variables, but declines thereafter. Hence, the best model is based on the top 4 climate predictor variables.

Table 3. Correlation matrix of climate predictor variables for areas with sagebrush present.

	June precip	June tmax	August precip	Annual precip
June precip	1			
June tmax	−0.29	1		
August precip	0.37	−0.01	1	
Annual precip	0.42	−0.41	0.22	1

At the landscape scale, a major risk to sagebrush ecosystems is invasion by non-native plants. The most widespread non-native invader in Nevada is cheatgrass; however, other invasive plants are also a concern. Several types of land use are important drivers of cheatgrass invasion, and likely increase probability of invasion by other non-natives as well. Hence, land use poses a direct risk to sagebrush ecosystems (e.g. by removal of shrubs to establish roads and power lines) (Leu et al. 2008), as well as an indirect and long-term risk associated with disturbance-facilitated invasion by non-native species (Gelbard and Belnap 2003, Bradley and Mustard 2006) (Fig. 3). Sagebrush ecosystems far from human land use and existing populations of cheatgrass have the lowest probability of cheatgrass invasion (Fig. 5).

Another observed threat to sagebrush ecosystems is the expansion of pinyon-juniper woodland (Tausch et al. 1981, Miller and Rose 1995, Miller et al. 2000, Weisberg et al. 2007, Bradley and Fleishman 2008). Expansion of trees typically occurs at the lower elevation extents of pinyon-juniper. The cause of expansion is unclear, but changes in land use and climate have both been implicated (Burkhardt and Tisdale 1976, Miller and Wigand 1994, Miller and Rose 1995). If pinyon-juniper expansion is indeed linked to climate, possibly due to higher precipitation during the last century, then the geographic distribution of woodland may continue to increase with climate change in the future. Sagebrush ecosystems adjacent to pinyon-juniper woodland are the most likely to be vulnerable to woodland expansion (Fig. 6).

Climate change poses a substantial long-term risk to sagebrush ecosystems. Climate conditions may render given locations less suitable for sagebrush than for other species, creating potential shifts in ecosystem distributions. Summer precipitation and temperature are the best predictors of sagebrush regional distribution, suggesting that changing summer conditions may have the most impact on long-term viability. Climate change risk to sagebrush due to changing

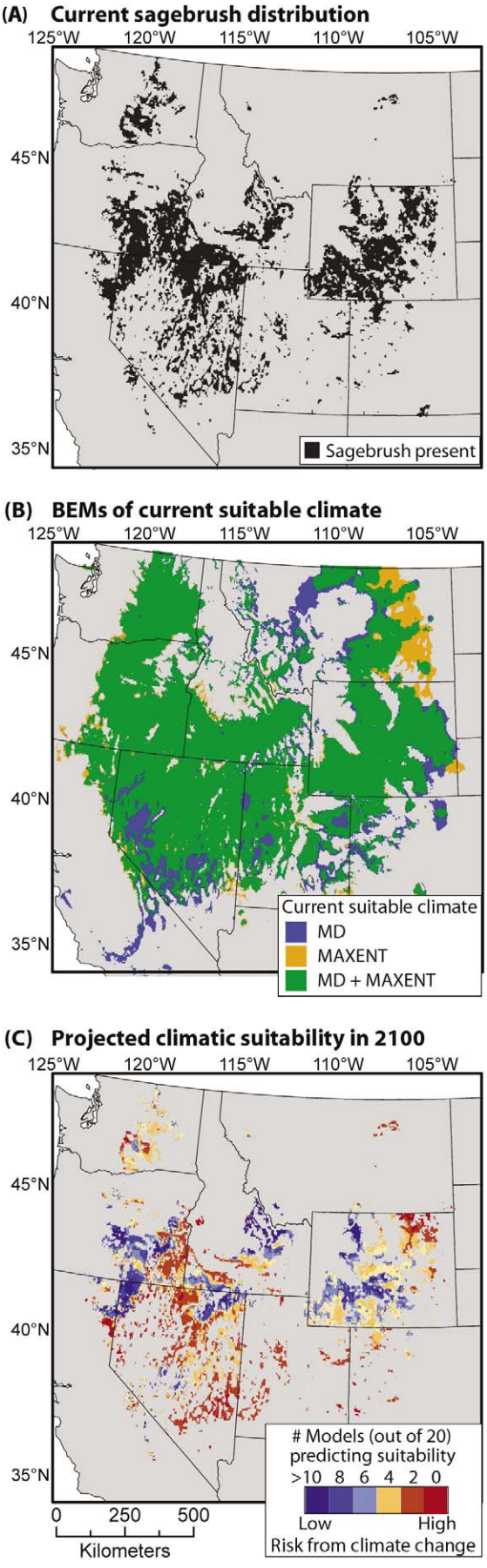

Figure 8. Current climatic suitability for sagebrush (*Artemisia* spp.) in the western United States and future climate change risk to sagebrush. (A) Current sagebrush distribution based on sagestitch (Comer et al. 2002) scaled to 4 km resolution is shown in black. (B) Suitable climate for sagebrush from bioclimatic envelope models based on Mahalanobis distance (MD) and maximum entropy (MAXENT). (C) Within the current sagebrush distribution, combined number of envelope models and AOGCMs (out of 20) that project maintained climatic suitability by 2100. Warmer colors are projected to be climatically suitable under fewer climate scenarios and are at greater risk from climate change. Cooler colors remain climatically suitable in multiple climate scenarios and are at lesser risk from climate change.

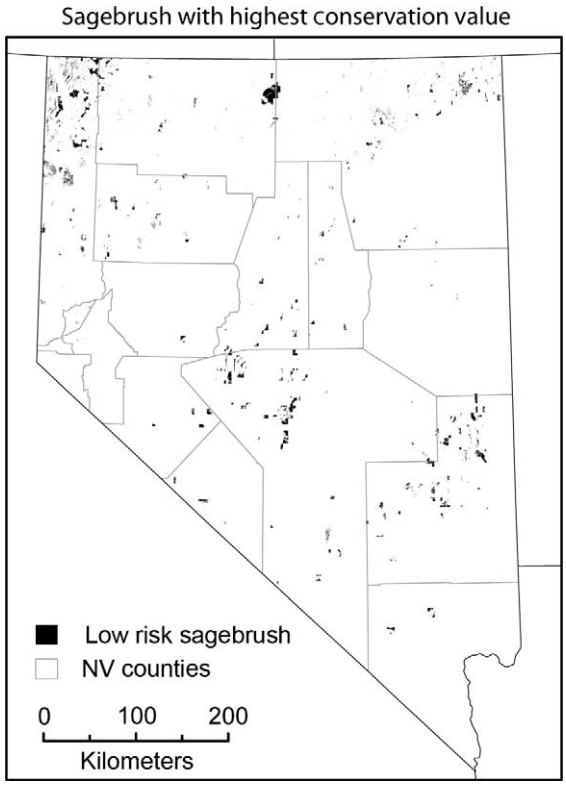

Figure 9. Sagebrush (*Artemisia* spp.) ecosystems in Nevada with the lowest risk of loss from land use, cheatgrass *Bromus tectorum* invasion, and climate change.

summer conditions is most pronounced in southern Nevada, where decreased precipitation and/or rising temperatures make current habitat climatically unsuitable in the future (Fig. 8). Under warmer and drier conditions, salt desert shrublands or non-native invasive species may replace some sagebrush ecosystems (Bradley 2009). This threat is most imminent at the low elevation edges of sagebrush distribution. Monitoring to assess ecosystem changes in these areas, and testing of management options to increase sagebrush viability should be a priority.

The projection of climate change risk for sagebrush presented here uses an ensemble of 2 BEMs and 10 AOGCMs to predict shifts in sagebrush distribution individually, and then combines the results to identify areas that remain climatically suitable in multiple (low risk) and few (high risk) scenarios (Araujo and New 2007). Differences between climate models increase the uncertainty associated with any single future climate scenario, particularly in the western United States where complex topography and the El Niño cycle further complicate projections (Randall et al. 2007). An ensemble approach is less susceptible to inconsistencies associated with any single model alone, and may produce more robust and reliable projections of distribution shifts (Araujo and New 2007). Land areas with more overlap between model projections (those with more consistent projections of high or low risk from climate change) have higher certainty than areas where models conflict in projections of risk.

It should be noted, however, that the quality of the model depends to a large degree on the quality of the distribution dataset. Thuiller et al. (2004) show that restricted distribution data may result in a much more conservative estimate of climatic habitat, particularly if data towards the edges of the environmental range are missing. Although the regional distribution data encompass sagebrush shrubland in the U.S., distributions in Canada are unavailable and have not been included. Further, sagebrush distributions in large parts of Montana are underrepresented in the remote sensing maps (USGS 2004). As a result, it is possible that the upper edge of the cold/wet tolerance of sagebrush is poorly constrained. Although the model provides a best estimate of future climatic suitability (Fig. 8), further analysis on a site-by-site basis is needed along with estimates of sagebrush sensitivity to a range of changes in summer precipitation and temperature to inform an adaptive management framework.

Land-use and climate change are important components of assessing threats to ecosystems (Millenium Ecosystem Assessment 2003) and assigning conservation value. However, additional spatial components should be considered in land management planning. For example, long-term goals such as landscape connectivity for wildlife may inform management decisions. Other spatial components such as land ownership and available patch size will affect the relative feasibility of conservation in some locations. These factors can be considered independently or collectively with risks from land use, invasive species and climate change.

Risk of sagebrush loss within Nevada can be assessed using a hierarchical framework (Pearson and Dawson 2003, Pearson et al. 2004) that considers both risk from climate change and risk from land use (Fig. 1). Depending on the planning goals and focus, layers can be included or weighted differently. For example, short-term planning may emphasize risk from land use, while long-term planning might equally weight risk from land use and risk from climate change. Only a small portion of the state was assessed to contain low risk sagebrush (Fig. 9), with 97% of the sagebrush in Nevada at risk from one or more of the factors modeled here. This suggests that long-term state and regional management strategies, such as the Healthy Lands Initiative in the western United States (HLI 2008), will be critically important for sustaining sagebrush ecosystems.

Assessing the spatial impacts of land use and climate change on native ecosystems is an important component of long-term land management planning. Providing risk analysis tools in a GIS framework supports land use planning and conservation prioritization. Climate change, land use and invasive species are major threats to ecosystems that are likely to become even more important in the future (Millenium Ecosystem Assessment 2003). Conservation planning strategies that encompass these risks have the highest potential for reducing the impacts of global change on native ecosystems.

Acknowledgements – This work was supported by the Wilburforce Foundation through a grant for Nevada Conservation Area Design. I thank Erica Fleishman, Nathan Gregory, Dave Marvin, and two anonymous reviewers for their constructive comments on earlier versions of the manuscript. I gratefully acknowledge the modeling groups for providing their data for analysis, the Program for Climate Model Diagnosis and Intercomparison (PCMDI) for collecting and archiving the model output, and the JSC/CLIVAR Working Group on Coupled Modelling (WGCM) for organizing

the model data analysis activity. The IPCC Data Archive is supported by the Office of Science, U.S. Dept of Energy.

References

Araujo, M. B. and Rahbek, C. 2006. How does climate change affect biodiversity? – Science 313: 1396–1397.

Araujo, M. B. and New, M. 2007. Ensemble forecasting of species distributions. – Trends Ecol. Evol. 22: 42–47.

Bakkenes, M. et al. 2002. Assessing effects of forecasted climate change on the diversity and distribution of European higher plants for 2050. – Global Change Biol. 8: 390–407.

Bradley, B. A. 2009. Regional analysis of impacts of climate change on cheatgrass invasion shows potential risk and opportunity. – Global Change Biol. 15: 196–208.

Bradley, B. A. and Mustard, J. F. 2005. Identifying land cover variability distinct from land cover change: cheatgrass in the Great Basin. – Remote Sens. Environ. 94: 204–213.

Bradley, B. A. and Mustard, J. F. 2006. Characterizing the landscape dynamics of an invasive plant and risk of invasion using remote sensing. – Ecol. Appl. 16: 1132–1147.

Bradley, B. A. and Fleishman, E. 2008. Relationships between expanding pinyon-juniper cover and topography in the central Great Basin, Nevada. – J. Biogeogr. 35: 951–964.

Burkhardt, J. W. and Tisdale, E. W. 1976. Causes of juniper invasion in southwestern Idaho. – Ecology 57: 472–484.

Chambers, J. C. 2001. *Pinus monophylla* establishment in an expanding *Pinus-Juniperus* woodland: environmental conditions, facilitation and interacting factors. – J. Veg. Sci. 12: 27–40.

Chambers, J. C. et al. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? – Ecol. Monogr. 77: 117–145.

Comer, P. et al. 2002. Current distribution of sagebrush and associated vegetation in the western United States (excluding NM and AZ). – Interagency Sagebrush Working Group.

Daly, C. et al. 2002. A knowledge-based approach to the statistical mapping of climate. – Clim. Res. 22: 99–113.

D'Antonio, C. M. and Vitousek, P. M. 1992. Biological invasions by exotic grasses, the grass fire cycle, and global change. – Annu. Rev. Ecol. Syst. 23: 63–87.

Elith, J. et al. 2006. Novel methods improve prediction of species, distributions from occurrence data. – Ecography 29: 129–151.

Farber, O. and Kadmon, R. 2003. Assessment of alternative approaches for bioclimatic modeling with special emphasis on the Mahalanobis distance. – Ecol. Model. 160: 115–130.

Foley, J. A. et al. 2005. Global consequences of land use. – Science 309: 570–574.

Gelbard, J. L. and Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. – Conserv. Biol. 17: 420–432.

Guisan, A. and Zimmermann, N. E. 2000. Predictive habitat distribution models in ecology. – Ecol. Model. 135: 147–186.

Guisan, A. and Thuiller, W. 2005. Predicting species distribution: offering more than simple habitat models. – Ecol. Lett. 8: 993–1009.

Harris, J. A. et al. 2006. Ecological restoration and global climate change. – Restor. Ecol. 14: 170–176.

Hirzel, A. H. et al. 2002. Ecological-niche factor analysis: how to compute habitat-suitability maps without absence data? – Ecology 83: 2027–2036.

HLI 2008. Healthy Lands Initiative. – U.S. Dept of the Interior, <www.doi.gov/initiatives/healthylands.html>.

Houghton, J. G. et al. 1975. Nevada's weather and climate. – Nevada Bureau of Mines and Geology.

IPCC 2007. Climate change 2007: the physical science basis. – Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge Univ. Press.

Iverson, L. R. and Prasad, A. M. 2002. Potential redistribution of tree species habitat under five climate change scenarios in the eastern US. – For. Ecol. Manage. 155: 205–222.

Knick, S. T. et al. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. – Condor 105: 611–634.

Leffler, A. J. and Caldwell, M. M. 2005. Shifts in depth of water extraction and photosynthetic capacity inferred from stable isotope proxies across an ecotone of *Juniperus osteosperma* (Utah juniper) and *Artemisia tridentata* (big sagebrush). – J. Ecol. 93: 783–793.

Leu, M. et al. 2008. The human footprint in the west: a large-scale analysis of anthropogenic impacts. – Ecol. Appl. 18: 1119–1139.

Matlab 2007. – The Mathworks, Natick, MA, USA.

Millar, C. I. et al. 2007. Climate change and forests of the future: managing in the face of uncertainty. – Ecol. Appl. 17: 2145–2151.

Millenium Ecosystem Assessment 2003. Ecosystems and human wellbeing: a framework for assessment. – Island Press.

Miller, R. F. and Wigand, P. E. 1994. Holocene changes in semiarid pinyon-juniper woodlands. – Bioscience 44: 465–474.

Miller, R. F. and Rose, J. A. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. – Great Basin Nat. 55: 37–45.

Miller, R. F. and Rose, J. A. 1999. Fire history and western juniper encroachment in sagebrush steppe. – J. Range Manage. 52: 550–559.

Miller, R. F. et al. 2000. Impacts of western juniper on plant community composition and structure. – J. Range Manage. 53: 574–585.

Nakicenovic, N. and Swart, R. 2000. Special report on emissions scenarios. – Cambridge Univ. Press.

NED 1999. USGS 30 meter resolution national elevation dataset (NED). – U.S. Geological Survey.

Pearson, R. G. and Dawson, T. P. 2003. Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? – Global Ecol. Biogeogr. 12: 361–371.

Pearson, R. G. et al. 2004. Modelling species distributions in Britain: a hierarchical integration of climate and land-cover data. – Ecography 27: 285–298.

Peterson, E. B. 2005. Estimating cover of an invasive grass (*Bromus tectorum*) using tobit regression and phenology derived from two dates of Landsat ETM plus data. – Int. J. Remote Sens. 26: 2491–2507.

Peterson, E. B. 2006. A map of invasive annual grasses in Nevada derived from multitemporal Landsat 5 TM imagery. – Report for the USDI Bureau of Land Management, Nevada State Office, Reno, Nevada Natural Heritage Program.

Phillips, S. J. and Dudik, M. 2008. Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. – Ecography 31: 161–175.

Phillips, S. J. et al. 2006. Maximum entropy modeling of species geographic distributions. – Ecol. Model. 190: 231–259.

Randall, D. A. et al. 2007. Climate models and their evaluation. – In: Solomon, S. et al. (eds), Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourther Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press, p. 74.

Root, T. L. et al. 2003. Fingerprints of global warming on wild animals and plants. – Nature 421: 57–60.

Rotenberry, J. T. et al. 2006. Gis-based niche modeling for mapping species' habitat. – Ecology 87: 1458–1464.

Rowland, M. M. et al. 2006. Greater sage-grouse as an umbrella species for sagebrush-associated vertebrates. – Biol. Conserv. 129: 323–335.
Tausch, R. J. et al. 1981. Tree age and dominance patterns in Great-Basin pinyon-juniper woodlands. – J. Range Manage. 34: 259–264.
Thomas, C. D. et al. 2004. Extinction risk from climate change. – Nature 427: 145–148.
Thuiller, W. et al. 2004. Effects of restricting environmental range of data to project current and future species distributions. – Ecography 27: 165–172.
Thuiller, W. et al. 2005. Climate change threats to plant diversity in Europe. – Proc. Nat. Acad. Sci. USA 102: 8245–8250.
Tsoar, A. et al. 2007. A comparative evaluation of presence-only methods for modelling species distribution. – Divers. Distrib. 13: 397–405.
Turner, B. L. et al. 2007. The emergence of land change science for global environmental change and sustainability. – Proc. Nat. Acad. Sci. USA 104: 20666–20671.
USGS 2004. National gap analysis program. Provisional digital land cover map for the southwestern United States', ver. 1.0. – RS/GIS Laboratory, College of Natural Resources, Utah State Univ.
VanderWall, S. B. 1997. Dispersal of singleleaf pinon pine (*Pinus monophylla*) by seed-caching rodents. – J. Mammal. 78: 181–191.
VanderWall, S. B. and Balda, R. P. 1981. Ecology and evolution of food-storage behavior in conifer-seed-catching Corvids. – J. Comp. Ethol. 56: 217–242.
Vitousek, P. M. et al. 1996. Biological invasions as global environmental change. – Am. Sci. 84: 468–478.
Vitousek, P. M. et al. 1997. Human domination of Earth's ecosystems. – Science 277: 494–499.
Weisberg, P. J. et al. 2007. Spatial patterns of pinyon-juniper woodland expansion in central Nevada. – Rangeland Ecol. Manage. 60: 115–124.
Whisenant, S. G. 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implications. – In: McArthur, E. D. et al. (eds), Symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. USDA Forest Service General Technical Report INT-276, pp. 4–10.
Williams, J. W. and Jackson, S. T. 2007. Novel climates, no-analog communities, and ecological surprises. – Front. Ecol. Environ. 5: 475–482.
Wisdom, M. J. et al. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia basin: broad-scale trends and management implications. – USDA Forest Service General Technical Report PNW-GTR-485.
Wisdom, M. J. et al. 2003. A prototype regional assessment of habitats for species of conservation concern in the Great Basin Ecoregion and State of Nevada. – USDA Forest Service, Pacific Northwest Research Station.