RESEARCH ARTICLE

EVALUATING THE ECOLOGICAL SUSTAINABILITY OF A PONDEROSA PINE ECOSYSTEM ON THE KAIBAB PLATEAU IN NORTHERN ARIZONA

Reuben Weisz*, Jack Triepke, and Russ Truman

USDA Forest Service, Southwestern Region,
Albuquerque, New Mexico 87102, USA

*Corresponding author: Tel.: 001-505-842 3217; e-mail: rweisz@fs.fed.us

ABSTRACT

This paper describes a process to evaluate the ecological sustainability of fire-adapted ecosystems, using a case study based on ponderosa pine (*Pinus ponderosa*) forests. We evaluated ecological sustainability by: 1) using reference conditions and models to describe the historical range of natural variability; 2) using recent remote sensing-based mid-scale mapping of existing vegetation to describe current conditions; and 3) retooling the reference condition models to incorporate current natural and anthropogenic processes to project future conditions of ecosystems. Finally, we discuss a process for incorporating consequences of climate change. Using the Vegetation Dynamics Development Tool (VDDT), we constructed state-and-transition models (STM) for cold ponderosa pine bunchgrass systems of northern Arizona. We included historic and contemporary fire frequencies in the respective models, and integrated forest insect and disease events. For the contemporary model, we added anthropogenic transitions based on the types and frequencies of current management activities. We calculated the historic proportion of each vegetation state by averaging model outputs from multiple 1000 yr simulations. We summarized current conditions from remote-sensing based existing vegetation map data, and then used the contemporary model to generate out-year projections as expressions of current management practices. Finally, we generated ecological departure ratings based on disparities between current and historic conditions, and between projected and historic conditions. Our analysis indicated that fire suppression coupled with infrequent management activities contributed to already significant trends in departure from reference conditions. We concluded by recommending additional steps for evaluating the effects of climate change, as well as the effects of alternative management scenarios for addressing climate change issues.

Keywords: climate change, historical range of variability, landscape ecology, landscape models, ponderosa pine, reference conditions, sustainability

Citation: Weisz, R., J. Triepke, and R. Truman. 2009. Evaluating the ecological sustainability of a ponderosa pine ecosystem on the Kaibab Plateau in northern Arizona. Fire Ecology 5(1): 100-114. doi: 10.4996/fireecology.0501100

INTRODUCTION

Ecological sustainability analysis is a two-tiered evaluation of ecosystems (ecosystem diversity) and their associated species (i.e., species diversity). Landscape assessments should use multiple spatial and temporal scales for ecological sustainability analysis (Jensen and Bourgeron 2001). Ecological sections and subsections (Cleland *et al*. 2007) are suitable spatial scales of analysis for comparison of conditions on the planning unit itself (technically, the third scale).

Current diversity of native species is an expression of the ecological health of a system (Moore *et al*. 2004). A guiding principle of ecosystem management (FEMAT 1993) is to replicate reference conditions to enable the persistence of species diversity. Accordingly, current or projected conditions that depart from reference conditions imply that an ecosystem is at risk. A comprehensive evaluation of ecological sustainability includes the consideration of all natural resources including soil, water, and air, but here we focus on vegetation diversity and related ecological processes such as fire.

In this paper, we focus on the temporal analysis conducted on the planning unit. We had three objectives:

1. Describe a methodology that can be used to evaluate the ecological sustainability of fire-adapted ecosystems.
2. Provide a case study of a ponderosa pine bunchgrass forest (*Pinus ponderosa*) ecosystem to demonstrate our methodology.
3. Provide recommendations to support considerations of climate change.

METHODS

Framing the Analysis

We stratified the Kaibab National Forest (NF) and its associated ecological sections (Cleland *et al*. 2007) by potential natural vegetation types (PNVTs) (Smith 2006), a coarse ecosystem framework defined by site potential and historic fire regimes that provides a basic framework for analyzing ecosystem diversity. We documented the analysis process conducted on the 16 PNVTs on the Kaibab NF, using the 218 935 ha ponderosa pine bunchgrass forest PNVT as a case study.

For each PNVT, we conducted quantitative analyses to define key ecosystem characteristics as defined by forest structure, species composition, and ecological processes. We framed the analysis, described reference and current conditions, projected future conditions, and evaluated results. We made quantitative model projections for all forest and woodland PNVTs (see following section, Projecting Future Conditions).

We described historic, current, and future structural conditions according to standard classification schemes based on average tree size (diameter) and canopy cover class (Brohman and Bryant 2005; Table 1). Due to disparities in historic and current condition references and how they were developed, we used crosswalks to normalize across references and to compare between historic and current condition. For instance, the US Forest Service mid-scale mapping that was used to depict current condition (Mellin *et al*. 2008) uses a canopy cover break of 30 % to distinguish open and closed, versus The Nature Conservancy model, which employs a 40 % break.

We portrayed historic, current, and future composition conditions according to a southwestern regional classification of existing vegetation based on dominance types (Triepke *et al*. 2005). Dominance types, defined by the relative abundance and dominance of tree species, are similar to Society of American Foresters or Society for Range Management cover types (Eyre 1980, Shiflet 1994), but are keyable, exhaustive, and mutually exclusive.

Table 1. Crosswalk used to facilitate ecosystem diversity analysis of ponderosa pine bunchgrass forests and the comparison of historic, current, and future conditions.

Model State			Existing vegetation classes		
Name	**Description**		**Dominance type**		**Size[1]–cover class[2] combinations**
A, J	Grass seedling sapling COMBINED WITH uncharacteristic grassland	=	Recently burned, all corresponding herb and shrub types[3]		n/a
F	Seedling and sapling >10 % tree cover	=	Ponderosa pine	AND	Seed and sap (all cover classes)
B	Ponderosa pine young forest, <30 % cover	=	Ponderosa pine	AND	Small-open
C	Ponderosa pine mid-age forest, <30 % cover	=	Ponderosa pine	AND	Med-open
D, E	Ponderosa pine mature/old forest with regeneration, <30 % cover	=	Ponderosa pine	AND	Very large-open
Contemporary landscapes only…					
G	Ponderosa pine young forest, >30 % cover	=	Ponderosa pine	AND	Small-closed
H	Ponderosa pine mid-age forest, >30 % cover	=	Ponderosa pine	AND	Med-closed
I	Ponderosa pine mature/old forest with regeneration, >30 % cover	=	Ponderosa pine	AND	Very large-closed

[1] Size classes based on dbh for forest tree species and drc (diameter at root collar) for woodland species: seedling/sapling = < 13 cm; small = 13 cm to 24.9 cm; medium = 25 cm to 50 cm; very large >50 cm.
[2] Overstory cover classes: sparse = < 10 %; open = 10 % to 29.9 %; closed = >29.9 %.
[3] 'Corresponding herb and shrub types' refers to those dominance types expected to occur within the ponderosa pine life zone.

Specific combinations of size, cover, and dominance type classes that are characteristic to each PNVT are expressed in terms of vegetation states identified for each PNVT. Each vegetative state represents an important phase in the ecosystem dynamics of a PNVT. The historic ponderosa pine forest ecosystem has been described (Smith 2006) as having open states (A, B, C, D, and E; Table 1). Frequent surface fires maintained the forest in these reference conditions.

Ecological process reflects the ability of natural and anthropogenic events such as fire, insect infestations, disease, and management activities to alter vegetation composition and structure and, in turn, wildlife habitat and species diversity (Perry and Amaranthus 1997). Along with site potential, the characteristic frequency and severity of fire are differentia of the PNVT classes themselves. Plant communities with the same site potential, say for ponderosa pine and Gambel oak (*Quercus gambelii*), but with differing historic fire regimes, promoted significantly different structure and composition,and would be classified separately—e.g., the ponderosa pine bunchgrass forest PNVT versus the Gambel oak shrubland (USDA Forest Service 1997).

Describing Reference Conditions

Introduction to the modeling software. The Vegetation Dynamic Development Tool (VDDT) (ESSA 2006) has been used by the National LANDFIRE program (Ryan *et al*. 2006) and others such as The Nature Conservancy (TNC) (Smith 2006) to develop state-and-transition models and descriptions of reference conditions. The VDDT software moves cells (representing a unit of area) from one state to another based on a set of transitions. Deterministic transitions are succession changes (aging) in the absence of disturbance. Probabilistic transitions reflect the quantitative assessments of discrete natural and anthropogenic events including fire, insects and diseases, grazing, harvesting, and severe weather events. Each probabilistic transition typically has three characteristics that define its pathway: 1) its return frequency or probability; 2) its severity or impact on vegetation; and 3) the destination state in which the cell will reside after transition.

Descriptions of reference conditions, and the VDDT models used to develop them, are available on the LANDFIRE and The Nature Conservancy websites (TNC 2007, LANDFIRE 2008). The TNC website also contains documentation on their peer review process (Marshal 2006). We retooled these models to project future conditions for the ponderosa pine bunchgrass forest model by replacing historic probabilities and transitions with contemporary transitions and attendant frequencies that reflect current land management. We also added contemporary and possible future vegetative states.

VDDT models for reference conditions. We based reference condition descriptions and models on peer-reviewed journal articles as well as published conference proceedings, reports, theses, dissertations, and book chapters. TNC further limited its survey of scientific literature to studies with a geographical emphasis on Arizona and New Mexico to ensure relevance to southwestern ecosystems (Smith 2006). From their synthesis, TNC created a ten-state model for the ponderosa pine bunchgrass forest in the southwestern United States. The first five states (A to E) depict historic or reference conditions, while the last 5 states represent contemporary conditions that have resulted from anthropogenic influences including fire suppression, timber harvest, and the introduction of exotic species (Table 1).

Assumptions for reference conditions. We base the reference conditions model for ponderosa pine bunchgrass on four key assumptions. First, model developers (Smith 2006) assumed that, because of the lack of empirical information, regeneration occurred gradually over time at a localized level within tree gaps, and not episodically in even-aged stands. As a result, the proportion of seedling and sapling vegetation in reference conditions may be underestimated. Second, we assumed that the surface fire transition type occurred on the average of once every 15.6 yr, based on fire scar data (Swetnam and Baisan 1996, Sneed *et al*. 2002, Fulé *et al*. 2003), indicating that surface fire occurred in the range of 5.4 yr to 36.3 yr. Third, stand-replacing fire was historically rare and minimal in extent, and was not included in the model; tree mortality occurred mostly at the scale of individual trees rather than patches (Moir *et al*. 1997, Fulé *et al*. 2003, Falk 2004). Fourth, deterministic transitions based on plant growth assumed a residence time of 40 yr within each state before natural succession to the next state (Reynolds *et al*. 1992).

Modeled reference conditions. Simulations with the reference condition model, based on the above assumptions, indicated that approximately 99 % of the area occurred in State E (old forest with an open canopy cover), 0.8 % in State D (mature forest with open canopy cover), and 0.1 % of the forest in State C (mid-age forest with open canopy cover).

Describing Current Conditions

We mapped PNVTs using Terrestrial Ecosystem Survey (TES) data for the Kaibab NF (Brewer *et al.* 1991). The TES is a terrestrial ecological unit inventory that formulates map units based on similarities in climate, soils, landform, and potential vegetation at the map scale of 1:24 000 (Winthers *et al*. 2005). Among the map unit attributes are classes that indicate historic events (zootic, fire), making TES map data the best available resource for PNVT mapping. Interpretation is necessary to accurately crosswalk TES and PNVT classes. Beyond Forest Service lands and the extent of TES information, southwest ReGAP map data (Prior-Magee *et al*. 2007) were used to supplement PNVT mapping across ecological sections.

In 2004, the Forest Service's Southwestern Region initiated mid-scale mapping of existing vegetation at 1:100 000 across all national forests and national grasslands (Mellin *et al*. 2008). This mapping included the three principal existing vegetation map components previously mentioned—dominance type, size class, canopy cover. With the description of model states (Table 1), these map data allowed for the quantitative analysis of current conditions within each PNVT.

We intersected PNVT mapping in a GIS with the existing dominance type, size class, and canopy cover layers from mid-scale vegetation mapping products to produce tabular summaries of current conditions within each PNVT class. These summaries were in turn synthesized to give hectares and percent of each vegetation state within each PNVT. We eventually compared these percents to historic and projected conditions for the ecosystem.

Along with each condition reported (historic, current, or projected), we calculated ecosystem condition class values using the same equation employed by LANDFIRE to compute Fire Regime Condition Class (FRCC) (Hann *et al*. 2005). But unlike FRCC, which provides percentages for each departure class (1, 2, or 3), our own ecosystem condition class (ECC) provides one overall departure rating for a given analysis area. The ECC is computed for each comparison, either current vs. reference condition or projected vs. reference condition (Table 2), based on the departure of all states in total from their reference conditions. In each calculation, the sum of the lesser of percent values for each state, either reference or current,

Table 2. Calculation of the departure index and ecosystem condition class based on the disparity between reference and current conditions of ponderosa pine bunchgrass forests on the Kaibab National Forest.

Ponderosa pine-Bunchgrass model state		**Historic**	**Current**	**Calculation**
Name	Description	Mean (%)	Mean (%)	Lesser of reference and current condition (%)
A, J	Grass seedling sapling COMBINED WITH uncharacteristic grassland	0.0	5.5	0.0
F	Seedling/sapling, >10 % tree cover	0.0	1.7	0.0
B	Ponderosa pine young forest, <3 0% cover	0.0	3.7	0.0
C	Ponderosa pine mid-age forest, <30 % cover	0.1	14.7	0.1
D	Ponderosa pine, <160 years forest with regeneration, <30 % cover	0.8	4.7	0.8
E	Ponderosa pine, 160 years plus with regeneration, <30 % cover	99.1	1.2	1.2
G	Ponderosa pine young forest, >30 % cover	–	8.6	0.0
H	Ponderosa pine mid-age forest, >30 % cover	–	51.1	0.0
I	Ponderosa pine mature and old forest with regeneration, >30 % cover	–	9.0	0.0
			Sum ≥	2.0
			Departure index = 100 % - Sum =	98.0
			Ecosystem condition class (0–33 = 1; 34–65 = 2; >66 = 3) =	"3"

is subtracted from 100 to provide one overall departure index on a scale of 1 % to 100 %, higher values representing more departed conditions. From there, three classes make up the ECC rating system:

- ECC 1 (within reference condition) represents departure index values ≤33;
- ECC 2 (moderately departed) represents departure index values >33 and ≤66; and
- ECC 3 (severely departed) represents departure index values >66.

We employed ECC values as an objective means of quantifying departure for an entire PNVT, in part because our VDDT models were not spatial. The ECC calculations for the ponderosa pine bunchgrass forests indicated an overall departure index of 97.95 % and a severely departed ecosystem condition class—ECC 3 (Table 2).

Recently developed FRCC map data for LANDFIRE map zones in Arizona (LANDFIRE 2008) corroborate our findings, as do regional studies of these systems (e.g., Fulé *et al*. 2003, Swetnam and Baisan 1996). Current trends in the incidence of insect and disease outbreaks, trends in management activities, along with decreased frequency and increased severity of fire (Kaibab National Forest, unpublished report), collectively explain the departure of the Kaibab's ponderosa pine ecosystem.

Projecting Future Conditions

Retooling reference condition models. Typically, reference conditions models are based on a survey of the literature supplemented by empirical data as well as expert opinion (LANDFIRE 2008). Often these models are applicable to a large map zone or, in the case of the TNC reference used in this paper, to a large region like Arizona and New Mexico. To retool these models to project conditions under existing or proposed management schemes, managers can modify reference condition models to: 1) include new states or modified states that reflect vegetation classes that did not exist under reference conditions; 2) incorporate current and projected natural and anthropogenic processes; and 3) incorporate current and projected transition probabilities. We illustrated by example how the Kaibab NF retooled the TNC ponderosa pine bunchgrass model for this purpose with the assumption of no climate change.

New or modified states. Typically, models for current and projected conditions contain as many or more states than reference condition models. In the case of the ponderosa pine bunchgrass PNVT, TNC (Smith 2006) suggests ten states to describe current conditions versus the five states representing reference conditions.

Quantifying current transitions. To retool reference condition models to reflect contemporary processes, four steps are followed: 1) identify the contemporary transitions; 2) replace reference transitions with contemporary ones; 3) model future conditions; and 4) interpret the results. Each contemporary transition is identified in terms of its type, transition class (groups of transition types), frequency, and effects. We used four transition classes in our current model: wildland fire, management activities, insect and disease, and deterministic transitions. Transition types within each transition class may have unique frequencies and effects unto themselves. The management activities transition class contains, for example, mechanical thinning, prescribed burning, etc.

Wildland fire transitions. We used LANDFIRE definitions of fire severity based on how much overstory canopy mortality would occur during a wildland fire: nonlethal (or low severity), <25 % mortality; mixed severity fire, 25 % to 75 % mortality; and stand replacement fire, >75% mortality (Hann *et al*. 2005). We generated fire frequencies for each of the transition classes using local fire history data on the planning unit for the period 1960 through 2005.

Spatial data was available for approximately 50 wildland fires >40 ha in size for the period 1960 to 2005. Fire mortality mapping was available for three incidents including the Pumpkin (1996), Warm (2006) and Muddersback (2005) fires. For other fires that occurred after 2000, fire officials provided estimates of the percentage of non-lethal, mixed severity, and stand replacement fire that occurred. We estimated fire mortality for the remaining fires using orthophotos in GIS, estimating fire extent and mortality based on patterns of top-kill and regeneration. For example, 98 % of the 1974 Moquitch Fire burned within the ponderosa pine bunchgrass PNVT. When the fire perimeter is compared to the 1996 orthophoto, one sees large areas of pine seedlings, areas with very open pine, and areas with closed pine stands, indicating different levels of mortality. Fires previous to 1986 are not as well documented, often captured only in fire perimeter maps and qualitative descriptions.

We summarized these results as average annual probabilities per hectare of each fire type: nonlethal fire (0.0038), mixed severity fire (0.0025), and stand replacing fire (0.0027). The effects of a fire on a cell within the model depend on pre-fire canopy cover and the severity of the fire (i.e., the fire mortality class; Table 3).

Table 3. Canopy cover and fire mortality proportion table.

Beginning canopy cover class	Fire severity class	Ending percentage by canopy cover classes
10 % to 30 % (open)	Non-lethal	9 % → sparse (0 % to 10 %)
		91 % → open (10 % to 30 %)
	Mixed severity	55 % → sparse (0 % to 10 %)
		45 % → open (10 % to 30 %)
	Stand replacement	100 % → sparse (0 % to 10 %)
30 % to 60 % (closed)	Non-lethal	16 % → open (10 % to 30 %)
		84 % → closed (30 % to 60 %)
	Mixed severity	2 % → sparse (0 % to 10 %)
		79 % → open (10 % to 30 %)
		19 % → closed (30 % to 60 %)
	Stand replacement	87 % → sparse (0 % to 10 %)
		13 % → open (10 % to 30 %)

Management activity transitions. We quantified management activities using the Forest Activity Tracking System (FACTS) database (M. Pitts, Forest Service, unpublished data). We queried all activities recorded on the planning unit from 1985 through 2006, and then eliminated activities that did not affect broad-scale vegetation composition and structure from further analysis (such as wildlife inventories and mine reclamation). We summarized the remaining 8747 management activities into standardized transition classes such as prescribed burning, fuels treatment, and harvest thinning. As with the wildland fire transitions, we calculated average annual probability per hectare values for each PNVT. Less than 2 % of the ponderosa pine bunchgrass forest PNVT was affected by these activities in a typical year during the sampled time period.

Insect and disease transitions. We quantified insect and disease transitions using data from 1918 through 2006 for the Kaibab NF and Grand Canyon National Park (Ann Lynch, Forest Service, unpublished report). For the ponderosa pine bunchgrass forest PNVT, we only used non-endemic mountain pine beetle and ips beetle outbreaks from 1950 through 2006 for which the severity, extent, and period could be determined. Severity is based on the degree of canopy cover loss resulting from an outbreak and the potential transition from one state to another. With a low severity event, for example, the affected forest stays in the same state. With a moderate severity outbreak, the closed state moves to an open state, and an open state moves to more open and younger states. A high severity outbreak transitions to the grass, seedling, and sapling state. For example: in the 1970s, we experienced high severity on 4047 ha and low severity on 30 351 ha; in 1983, we experienced a moderate occurrence on 12 141 ha; in 2002, we recorded 29 137 ha of low severity infestations; and in 2004, 24 281 ha of moderate severity infestations occurred. As with fire, we computed average annual probability per hectare values for each PNVT—0.0870 for the ponderosa pine bunchgrass forest PNVT.

Deterministic transitions. We included several types of deterministic transitions in the model. As in the reference condition model, we assumed the 40 yr residence times for natural succession. However, in the absence of surface fire, our residence times between open and closed states were reduced to 25 yr based on Covington and Moore (1994), Swetnam and Baisan (1996), Allen *et al*. (2002), and Fulé *et al*. (2003).

We included an "uncharacteristic grassland" state (State J) that results from stand replacement fire in the contemporary model. We initially assumed permanent deforestation following high severity fire in ponderosa pine, based on work by Savage and Mast (2005). A more detailed review and analysis of this research, along with vegetation map data, indicated that while uncharacteristic grasslands and shrublands result from stand replacement fire, these consequences are localized. On the whole, the Savage and Mast (2005) study seems to indicate that tree regeneration is protracted by decades and, in some cases, by a century or longer. On average, tree canopy cover does increase by approximately 0.3 % per year in the years following fire, with a range of 0.0 % to 0.6 % among the study sites. The average increase in tree diameter at breast height (dbh) per year was calculated at 0.254 cm. At these rates, it would take approximately 39 years to achieve 10 % canopy cover (the threshold for a tree-dominated class and state), and approximately 41 years to achieve 12.7 cm at dbh. These values of tree cover and diameter recovery translate to a probability of succession and tree encroachment from State J of approximately 9 % of the residual area per year.

As a result of our assessment of State J, we retooled the ponderosa pine bunchgrass forest model to provide succession from State J. As it turned out, a subsequent sensitivity analysis using regeneration scenarios ranging from 0 % to 9 %, and short-term projections, produced no significant difference in overall ecosystem condition class. Also, current management policy on the Kaibab NF promotes reforestation of all deforested units by planting if necessary.

Model runs. VDDT is a non-spatial model and results of runs are based on a summary of up to 50 000 sample units or cells. We used 1000 sample units in our runs since our earlier work, and work conducted by TNC and LANDFIRE, indicated that this number produced reasonable and consistent projections. If we increased the number of cells beyond 1000, results of the analysis are not significantly changed, but running time is increased significantly.

In the next step of the remodeling process, we initialized the starting hectares in each state based on current conditions indicated by mid-scale vegetation mapping products. We ran multiple simulations to estimate the long-term effects of continuing current management under the existing land management plan. We ran 10 simulations with each simulation projecting conditions annually for 1000 years; these runs were based on data and assumptions described earlier and summarized in Table 4. We compared the average annual results of these simulations with current conditions and reference conditions.

Table 4. Summary of transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their projected effects in the model. This table assumes no projected change in climate. The last section of this paper describes how to modify the models to incorporate the effects of projected climate change.

Transition type	Transition frequency or length	Sources	Assumptions
Fuel build-up	After 25 years of uninterrupted growth	The cessation of surface fires around 1880 and the resulting accumulation of fuels (Covington and Moore 1994, Swetnam and Baisan 1996, Allen *et al.* 2002, Fulé *et al.* 2003)	It would take approximately 25 years since the last fire or management activity mimicking fire to move from an open canopy state (<30 % canopy cover) to a higher canopy state (>30 %).
Wildland fire	Frequencies for nonlethal, mixed severity and stand replacing fire are based on empirical data in sources column.	Fire history data on the planning unit for the period 1960 through 2005	Projected fire frequencies will be the same as those experienced the last half century.
Management activities	Frequencies for management activities (fires and thinning) are based on empirical data in sources column.	Forest Activities Tracking System (FACTS) database of management activities recorded on the planning unit from 1985 through 2006 (M. Pitts, Forest Service, unpublished data)	Projected frequencies of prescribed burning, other fuels treatment, and harvest thinning will be the same as those experienced the last half century.
Insect and disease	Frequencies for insects and diseases incidence are based on empirical data in sources column.	Insect and disease transitions were quantified using data from 1918-2006 (A. Lynch, Forest Service, unpublished report).	Projected frequencies of insect and disease incidence will be the same as those experienced since 1950.
Regeneration from seed	Varies between 0 and 9 % per hectare per year after stand replacing fire	Savage and Mast (2005)	Projected frequencies of seedling recruitment are the same as those observed in the recent literature and in the planning area.
Plant growth	40 years between states	Transitions among model states are taken from silvicultural data summarized by Reynolds *et al.* (1992).	We assume that transitions between states (for example, from seedling and sapling to young forest, from young to mid-age forest) take 40 yr.

RESULTS

The results shown in Table 5 help to answer the question, "How do current and projected conditions compare to reference conditions?" Again, we derived reference conditions from VDDT models that TNC developed to quantify the historical proportion of major vegetation states. For current conditions, we summarized existing vegetation mapping according to the same vegetation state concepts used for the reference condition model. We generated out-year projections from the retooled VDDT model for 20, 40, 50, 100, 250, 500, and 1000 years. The bulk of ponderosa pine bunchgrass forest on the historical landscape occurred in older, open states (D and E) characterized by a plurality of larger diameter trees. Current condition is shown as severely departed; the overall departure exceeds 96 %.

Table 5. Comparison of the ponderosa pine PNVT successional states to reference, current and projected conditions for the North Kaibab (62 811 ha), Tusayan (42 444 ha), and Williams (118 662 ha) ranger districts, and the Kaibab National Forest (223 916 ha).

	Vegetation state											
	A	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**Departure**	**ECC**
	Reference condition											
	0	**0**	**0.1**	**0.8**	**99.1**	**0**	**0**	**0**	**0**	**0**		
	Current condition											
N Kaibab	7.13	3.69	20.56	8.23	2.15	0.48	3.75	35.39	15.94	2.68	96.95	3
Tusayan	4.89	7.66	21.65	6.77	1.29	0.22	7.68	43.75	6.09	0.00	97.81	3
Williams	3.1	2.23	8.44	1.72	0.5	2.99	11.79	63.16	6.06	0.00	98.6	3
Kaibab NF	4.65	3.74	14.65	4.65	1.15	1.69	8.59	51.08	9	0.8	97.95	3
	Projected trends – Kaibab National Forest											
20 yr	14.31	6.28	5.86	2.68	2.15	1.15	2.72	32.28	32.45	0.12	96.95	3
40 yr	21.95	7.07	3.03	1.46	2.01	1.39	0.75	16.25	46.08	0.01	97.09	3
50 yr	21.32	10.32	1.87	1.24	1.95	1.48	0.78	13.93	47.08	0.03	97.15	3
100 yr	20.18	19.34	1.94	1.42	1.74	2.39	0.85	11.42	40.7	0.02	97.36	3
250 yr	19.71	19.21	2.89	0.9	1.56	2.31	1.05	15.73	36.63	0.01	97.54	3
500 yr	20.04	18.48	2.79	1.15	1.81	2.17	1.17	15.52	36.86	0.01	97.29	3
1000 yr	19.99	17.74	2.91	1.01	1.38	2.36	1.13	15.87	37.57	0.04	97.72	3

Under current management practices, little change in future conditions is expected according to the projections we provide. The level of departure from one out-year projection to the next remains nearly static, at or just above 97 %, with the proportion in each state moving slightly farther away from reference conditions and then stabilizing after 100 years. The only notable exceptions to this pattern occur within particular vegetation states where the proportion in State B (open, 13 cm to 24.9 cm diameter) nearly triples from 6.3 % to 17.7 %, while the acres in State H (closed, 25 cm to 50 cm) are halved, suggesting that some amount of area may be passed to a younger, more open condition as a result of fire.

In total, our findings indicate that current conditions and projected conditions in the ponderosa pine bunchgrass PNVT are approximately 97 % departed from reference conditions, as represented by an ecosystem condition class of 3 (severe departure). Given our analysis methods, trends of further departure would not be detected, pending changes in the pattern of stand-replacing fire into the future (i.e., unforeseen change in fire frequency).

DISCUSSION

Analysis Process

As indicated earlier, TNC and the LANDFIRE program and others have made a significant investment in the development of reference condition descriptions and models. We retooled these reference condition models, which in turn enabled us to project future conditions. Current and future conditions can be compared with reference conditions to answer two questions: 1) is there a current departure from reference conditions; and 2) will conditions remain the same or trend towards or away from reference conditions? Trends away from reference condition may indicate an ecosystem at risk. If so, the model can be further retooled to help evaluate the effectiveness of potential management strategies.

In an ongoing effort to make modeling more reliable, we are refining forest and woodland VDDT models by calibrating residence times with Forest Inventory and Analysis (FIA) data (Miles *et al*. 2001) and the Forest Vegetation Simulator (FVS) framework (Dixon 2002, Havis and Crookston 2008).

Evaluating Our Results

Several assumptions were necessary in the face of uncertainties concerning the historic condition that we modeled (Table 4). For example, although available empirical evidence points to the uneven-aged nature of historic ponderosa pine stands (Swetnam 1996, Fulé *et al*. 2003, Moore *et al*. 2004, Savage and Mast 2005), it is reasonable to assume that, although uncharacteristic, even-aged patches did occasionally exist. A related assumption was made for stand scale, where the resolution of plant communities was constrained indirectly by available map data, with map features averaging 6 ha (Mellin *et al*. 2008) and the smallest map features still averaging about 0.1 ha (i.e., the size of a LANDSAT pixel). Here again, the reader can assume that tree regeneration was present in the reference conditions, and is present in the current conditions, despite what may be implied by the size class of the state. This is particularly the case in more open-canopied stands where tree regeneration would be favored. The model's only sensitivity in this regard is in the written descriptions for the oldest open-canopied states that tend to be more uneven-aged (Smith 2006) and have a greater proportion of seedlings and saplings (Smith 2006).

Our analysis indicates that the ponderosa pine bunchgrass ecosystem in the Kaibab NF is severely departed from reference conditions, and that this trend will continue into the future under the existing land management plan. Fire suppression coupled with infrequent forest management activities contributes to an already significant departure from reference conditions. Thus, the continued current implementation of the existing land management plan creates a risk to the ecological sustainability of this ecosystem.

Others such as Arno and Fiedler (2005) have explored deteriorated forest conditions in western North America and reached similar conclusions. Our contribution is to illustrate how reference condition models developed by LANDFIRE and others can be retooled to assess the ecological sustainability of these ecosystems within a more empirical, systematic framework for developing strategic land management plans. Once these models are created, they also can be utilized to explore alternative management scenarios for revising strategic land management plans to restore functionality to fire-adapted ecosystems.

Addressing Climate Change

Future extensions of our methodology include projecting the effects of climate change on ecological sustainability, and providing spatial simulations (Miller 2007). We also advocate evaluating adaptive and mitigation strategies as outlined by Millar *et al*. (2007). Carbon accounting for mitigation strategies can currently be facilitated by using the carbon extension of the Forest Vegetation Simulator (Havis and Crookston 2008) and do not require climate change projections. But adaptation strategies do necessitate predictions about future vegetation patterns and, at this time, we are considering the advantages and disadvantages of alternative approaches to modeling climate change, and we are evaluating the benefits of doing such modeling.

The assumptions in our projection models can be modified in the following ways to incorporate the emerging evidence from climate research (M.A. Hemstrom and J. Merzenich, Forest Service, personal communication):

1. Types of states: Climate change may result in the addition or removal of states within a PNVT as new vegetation composition and structural patterns are introduced with changing site potential and processes (such as the introduction of exotic species).
2. Types of transitions: Climate change may result in the addition or removal of transitions within a PNVT, with novel patterns of vegetation composition, structure, and process.
3. Rates of transitions: The rates of transitions between model states for existing transitions, for example, stand-replacing fire, may change within the PNVT and planning area.
4. New (adventive) PNVTs: Adventive PNVTs may need to be modeled, depending on the climate scenario.
5. Transitions between PNVTs: In addition to transitions within PNVT models, transitions between PNVTs may be necessary to reflect the movement of area between PNVT classes as climate changes.
6. New management activities: New management activities may be necessary to respond to adaptive and mitigation strategies (Millar *et al*. 2007) and modification to the rates of existing transitions.
7. Projected climate variability: Changes in the annual variation of phenomena such as wet years, dry years, insect and disease incidence, etc., may be explicitly modeled within existing VDDT software.
8. Addressing multiple climate scenarios: Current assumptions (Table 4) can be modified to reflect each climate change scenario that needs to be considered by management; for example, in scenario 1, the planning area may be getting warmer and drier, and in scenario 2, the planning area may be getting warmer and wetter.

There are several limitations associated with using VDDT models in the presence of climate change. A solid foundation of data and evidence may not be available to modify our existing assumptions to reflect new and altered environments, adventive PNVTs, and adventive cover and structural classes and transitions within existing PNVTs. Nevertheless, there are increasingly compelling reasons to consider using these models, however speculative, to develop and evaluate adaptation and mitigation strategies for future landscapes in the presence of climate change.

ACKNOWLEDGEMENTS

We acknowledge the contributions of our many Forest Service colleagues who made this work possible, and especially Wayne Robbie who envisioned an advanced process for ecological sustainability analysis in the southwest. We also thank Ed Smith and our collaborators in The Nature Conservancy, and our partners in the LANDFIRE program.

LITERATURE CITED

Allen, C., M. Savage, D.A. Falk, K.F. Suckling, T.W. Swetnam, T. Schulke, P.B. Stacey, P. Morgan, M. Hoffman, and J.T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12: 1418-1433.

Arno, S., and C. Fiedler. 2005. Mimicking nature's fire: restoring fire-prone forests in the west. Island Press, Washington, D.C., USA.

Brewer D.G., R.K. Jorgensen, L.P. Munk, W.A. Robbie, and J.L. Travis. 1991. Terrestrial ecosystem survey of the Kaibab National Forest. USDA Forest Service, Southwestern Region technical report.

Brohman, R.J., and L.D. Bryant. 2005. Existing vegetation classification and mapping technical guide version 1.0. USDA Forest Service General Technical Report WO-GTR-67.

Cleland, D.T., J.A. Freeouf, J.E. Keys, G.J. Nowacki, C.A. Carpenter, and W.H. McNabb. 2007. Ecological subregions: sections and subsections of the coterminous United States. USDA Forest Service, General Technical Report WO-76.

Covington, W.W., and M.M. Moore. 1994. Postsettlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. Journal of Sustainable Forestry 2: 153-181.

Dixon, G.E. 2002. Essential FVS: a user's guide to the Forest Vegetation Simulator, reference manual. USDA Forest Service, Forest Management Service Center. Fort Collins, Colorado, USA.

ESSA [ESSA Technologies Ltd.]. 2006. Vegetation dynamics development tool user guide, Version 5.1. ESSA Technologies Ltd., Vancouver, British Columbia, Canada. <http://essa.com/downloads/vddt/download.htm>. Accessed 31 December 2008.

Eyre, F.H., editor. 1980. Forest cover types in the United States and Canada. Society of American Foresters. Washington, D.C., USA.

Falk, D.A. 2004. Scaling rules for fire regimes. Dissertation, University of Arizona, Tucson, USA.

FEMAT. 1993. Forest ecosystem management: an ecological, economic, and social assessment. Report of the Forest Ecosystem Management Assessment Team. US Government Printing Office, Washington, D.C., USA.

Fulé, P.Z., T.A. Heinlein, W.W. Covington, and M.M. Moore. 2003. Assessing fire regimes on Grand Canyon landscapes with fire scar and fire record data. International Journal of Wildland Fire 12:129-145.

Hann, W., A. Shlisky, D. Havelina, K. Schon, S. Barrett, T. DeMeo, K. Pohl, J. Menakis, D. Hamilton, J. Jones, M. Levesque. 2005. Interagency fire regime condition class guidebook. Version 1.2. National Interagency Fuels Technology Team. <http://www.landfire.gov/documents_related_tech.php>. Accessed 23 December 2008.

Havis, R.N., and N.L. Crookston, compilers. 2008. Third forest vegetation simulator conference. USDA Forest Service proceedings RMRS-P-54.

Jensen, M.E., and P.S. Bourgeron, editors. 2001. A guidebook for integrated ecological assessments. Springer-Verlag, New York, New York, USA.

LANDFIRE. 2008. Homepage of the LANDFIRE Project, USDA Forest Service, US Department of the Interior. <http://www.landfire.gov/index.php>. Accessed 23 December 2008.

Marshall, R. 2006. Southwest forest assessment project peer review process. The Nature Conservancy in Arizona. <http://azconservation.org/dl/TNCAZ_SWFAP_HRV_PeerReviewProcess.pdf>. Accessed 24 December 2008.

Mast, J.N., P.Z. Fulé, M.M. Moore, W.W. Covington, and A.E.M. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications 9: 228-239.

Mellin, T., F.J. Triepke, and P. Joria. 2008. Mapping existing vegetation at the mid-scale level in the Forest Service Southwestern Region. Proceedings of the twelfth biennial USDA Forest Service remote sensing applications conference. Salt Lake City, Utah, 15-19 April 2008, CD-ROM. <http://svinetfc4.fs.fed.us/RS2008/t_mellin/index.htm>. Accessed 6 January 2009.

Miles, P.D., G.J. Brand, C.L. Alerich, L.F. Bednar, S.W. Woudenberg, J.F. Glover, and E.N. Ezzell. 2001. The forest inventory and analysis database: database description and users manual Version 1.0. USDA Forest Service General Technical Report NC-GTR-218.

Millar, C.I., N.L. Stephenson, and S.L. Stephens. 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications 17: 2145-2151.

Miller, C. 2007. Simulation of the consequences of different fire regimes to support wildland fire use decisions. Fire Ecology 3(2): 83-102.

Moir, W.H., and J.H. Dieterich. 1988. Old growth ponderosa pine from succession on pine-bunchgrass habitat types in Arizona and New Mexico. Natural Areas Journal 8: 17-24.

Moir, W.H., B. Geils, M.A. Benoit, and D. Scurlock. 1997. Ecology of southwestern ponderosa pine forests. Pages 3-27 in W.M. Block and D.M. Finch, editors. Songbird ecology in southwestern ponderosa pine forests: a literature review. USDA Forest Service General Technical Report RM-GTR-292.

Moore, M.M., D.W. Huffman, P.Z. Fulé, W.W. Covington, and J.E. Crouse. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. Forest Science 50: 162-176.

Perry, D.A., and M.P. Amaranthus. 1997. Disturbance, recovery, and stability. Pages 31-56 in: K.A. Kohm and J.F. Franklin, editors. Creating a forestry for the 21st century: the science of ecosystem management. Island Press, Washington, D.C., USA.

Prior-Magee, J.S., K.G. Boykin, D.F. Bradford, W.G. Kepner, J.H. Lowry, D.L. Schrupp, K.A. Thomas, and B.C. Thompson, editors. 2007. Southwest regional gap analysis project final report. USGS Gap Analysis Program, Moscow, Iaho, USA. <http://fws-nmcfwru.nmsu.edu/swregap/report/SWReGAP%20Final%20Report.pdf>. Accessed 24 December 2008.

Reynolds, R.X., R.T. Graham, M.H. Reiser, R.L. Bassett, P.L. Kennedy, D.A. Boyce, Jr., G. Goodwin, R. Smith, and E.L. Fisher. 1992. Management recommendations for the northern goshawk in the southwestern United States. USDA Forest Service General Technical Report RM-GTR-217.

Ryan, K.C., K.M. Lee, M.G. Rollins, Z. Zhu, J. Smith, and D. Johnson. 2006. Landfire: national vegetation and fuel mapping for fire management planning. Forest Ecology and Management 234: S220.

Savage, M., and J.N. Mast. 2005. How resilient are southwestern ponderosa pine forests after crown fires? Canadian Journal of Forest Research 35: 967-977.

Shiflet, T.N. 1994. Rangeland cover types. Society for Range Management, Denver, Colorado, USA.

Smith, E.B. 2006. Historical range of variation and state and transition modeling of historical and current landscape conditions for ponderosa pine forest and woodland of the southwestern US. The Nature Conservancy in Arizona. <http://azconservation.org/dl/TNCAZ_SWFAP_HRV_Ponderosa_Pine.pdf>. Accessed 31 October 2008.

Swetnam, T.W., and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 in: C.D. Allen, editor. Fire effects in southwestern forests: proceedings, 2nd La Mesa fire symposium, 29-31 March 1994, Los Alamos, New Mexico. USDA Forest Service General Technical Report RM-GTR-286.

The Nature Conservancy. 2007. Homepage for historical range of variation for potential natural vegetation types of the southwest. The Nature Conservancy in Arizona. <http://azconservation.org/downloads/data/historical_range_of_variation_for_potential_natural_vegetation_types/>. Accessed 24 December 2008.

Triepke, F.J., W.A. Robbie, and T.C. Mellin. 2005. Dominance type classification: existing vegetation classification for the Southwestern Region. USDA Forest Service Forestry Report FR-R3-16-1, Southwestern Region, Albuquerque, New Mexico, USA.

USDA Forest Service. 1997. Plant associations of Arizona and New Mexico Volume I: forests. Third edition. <http://www.nm.nrcs.usda.gov/technical/tech-notes/range/range99-publication-forests.pdf>. Accessed 27 December 2008.

Winthers, E., D. Fallon, J. Haglund, T. DeMeo, G. Nowacki, D. Tart, M. Ferwerda, G. Robertson, A. Gallegos, A. Rorick, D. Cleland, and W. Robbie. 2005. Terrestrial ecological unit inventory technical guide: landscape and land unit scales. USDA Forest Service General Technical Report WO-GTR-68. <http://www.fs.fed.us/emc/rig/includes/TEUI_guide.pdf>. Accessed 6 January 2009.