AN ABSTRACT OF THE DISSERTATION OF

Patrick L. Shaver for the degree of Doctor of Philosophy in Rangeland Ecology and Management presented on January 22, 2010.

Title: Quantification of State-and-Transition Model Components Utilizing Long-term Ecological Response Data Following One-seed Juniper Treatment on a Deep Sand Savannah Ecological Site

Abstract Approved: ____

Tamzen K. Stringham William C. Krueger

A proposed state-and-transition model (STM) for the Deep Sand Savannah ecological site in central New Mexico was developed using historical data and expert knowledge. This STM was tested utilizing data from short and long term one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg.) control experiments initiated in 1981 and 1985. Utilizing data from the individual plots within the identified states, the proposed model was refined with definitions of specific processes and indicators associated with each state and transition. At-risk community phases, feedback mechanisms and threshold values were identified and described for soil aggregate stability and vegetative cover variables.

Short term response data were collected in 1984 and long term response data from 1985 through 1989 and in 2003. Soil moisture data indicated the treated plots contained significantly (α=0.05) more available moisture than control plots especially during the drought year of 1989. The treated plots were significantly (α=0.05) different from controls for all the vegetation and soil variables. Vegetation measurements were repeated in 2003 along with additional vegetation attributes and soil aggregate stability.

Eighteen years after treatment, data analysis indicated significantly (α=0.05) different treatment effects in most variables and significant (α=0.05) ranch by treatment interactions for many others. Linear regression showed

expected correlations and several weak ($r^2<0.30$) but significant ($\alpha=0.05$) relationships.

Results suggest soil aggregate stability variables provided the best integrator of long term ecological responses to change in vegetation production, soil moisture, cover, bare ground, litter accumulation and bare patch size. Surface soil stability was a reliable indicator and predictor of state membership and provided indication of value ranges within states for itself and other data elements. The STM includes feedback mechanisms that build resilience into each of the three identified states, at-risk community phases within the Reference and Juniper States and threshold values between the three states. This model will assist managers with identification of potential ecological thresholds and at-risk community phases, thus providing information to plan actions that facilitate the maintenance of ecological and economic sustainability while providing the broadest array of ecosystem services possible within the potential of the ecological site.

Quantification of State-and-Transition Model Components Utilizing Long-term Ecological Response Data Following One-seed Juniper Treatment on a Deep Sand Savannah Ecological Site

By
Patrick L. Shaver

A DISSERTATION

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Doctor of Philosophy

Presented January 22, 2010
Commencement June 2010

Doctor of Philosophy dissertation of Patrick L. Shaver presented on January 22, 2010

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

Patrick L. Shaver, Author

ACKNOWLEDGEMENTS

I wish to express my thanks first to the ranchers involved in this study. They have shown continued interest in this project since 1981 and have endured my interruption of their lives and businesses since that time. Without their continued long time support and interest in the effective and economical management of this ecosystem, this work would not have been possible.

The employees and my supervisors in the USDA-Natural Resources Conservation Service all deserve individual acknowledgement, but space prevents that. However, I must mention and thank the following individuals: Dick, Dan, John, Marcus, Richard, Darrel and Santiago. My friend and co-worker, Javier Ruiz, deserves more thanks than is possible. Javier spent three days every two weeks for five years with me collecting data. I especially owe Lendon Parker thanks for saving the box full of original data and giving it to me in 1992. I appreciate the encouragement from Dr. Ken Spaeth who has constantly encouraged me for the last 20 years to do something with the data. The opportunity to return to school and enter into the PhD program would not have been possible without my previous supervisor, Dr. Larry Butler. Larry provided me the flexibility in my work schedule to pursue the course work and additional data collection needed to complete the degree requirements. Although retired, Larry continues to encourage me in this endeavor. I have been constantly encouraged by George Chavez, who wants to use the STM in the ecological site description he is responsible for. It is on the way George. My current supervisor, Bruce Newton, has provided all the encouragement possible, and has supported my time off work to complete this project.

I owe many thanks to Dr. William Krueger, who found me office space and welcomed me into the department after the USDA-NRCS reorganization in 1995. His continued questioning and challenges led me to enroll in this program. He agreed to serve on my committee and recommended other members of my committee. Dr. Rick Miller has always shown interest and a desire to help and has encouraged me to complete this project.

Dr. Tamzen Stringham, my major professor, has been a source of support and guidance. I have thoroughly enjoyed and learned from Dr. Stringham’s insights and knowledge. I am especially grateful to her for the patience, challenges and encouragement she has provided over the last 8 years. I appreciate most of all that this association has allowed us to work together on several projects and develop both a beneficial professional relationship and a solid friendship. The patience and persistence she showed during the last week of preparation of this manuscript, during a stressful time in her life was above and beyond the call of mentoring. I will always be grateful for that.

Dr. Jeff Herrick has been the foundation of much of the last 8 years on this project. His friendship and mentoring have been highlights in my career. His willingness to help with all aspects of this project has been extremely helpful. All of the time spent with Jeff on the ground both on this project and others have been learning experiences. I cannot repay what Jeff has given me both professionally and personally. Jeff introduced me to Dr. Leigh Murray, who taught me more about statistics in three days than I thought possible. Her help was extremely valuable. Jeff also introduced me to David Toledo who helped with my data collection and entering the data into the computer, which was a tremendous help.

The comic harassment I have received from my sons and their wives has been a source of fun and pleasure. Their encouragement and faith in my persistence has been uplifting and truly appreciated. Finally and most important, I want and need to acknowledge my wonderful wife Sharon. She has suffered with my obsession over this project for more than 25 years. She has been and continues to be my most vocal and stalwart critic and supporter. The time I have spent away from her, both physically and mentally, working on this since 1981 can never be repaid. She has continually believed in and supported my work and has always expected my best. All that I have been able to accomplish in my life is due to her support and the grace of God.

CONTRIBUTION OF AUTHORS

Patrick L. Shaver was the lead scientist involved with experimental design, data collection, data analysis, theoretical modeling, and writing of each chapter in this dissertation. Dr. Tamzen Stringham and Dr. Jeff Herrick were involved with experimental design, and model concept development. Dr. Leigh Murray was involved in statistical advice and Dr. Tamzen Stringham was involved in manuscript editing.

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CHAPTER 1: GENERAL INTRODUCTION

MODELS OF ECOLOGICAL DYNAMICS

Succession-Regression Model

Building on Clements' (1928) concepts of linear succession to a climatic climax, Sampson's (1919) classification ideas and the edaphic polyclimax of Tansley (1935), E. J. Dyksterhuis (1949) proposed the quantitative climax model. This model is based on the succession-regression concept of plant dynamics and provided a quantitative way in the field to determine the status of a plant community. It is based on the concept that competition driven plant succession will return a plant community to its site potential once an outside disturbance has been removed. The status or condition of the functional edaphic unit (later called range site) was based on the proportions of decreaser, increaser and invader plant species in the existing plant community (Dyksterhuis 1958).

The quantitative climax model was integrated into range site descriptions and became the model for directing management decisions and plant community response to disturbance levels. Range sites were defined as a distinctive kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation (USDA 2003). The quantitative climax model worked well for many parts of the country, especially in areas of summer precipitation with plant communities dominated by herbaceous plant species. However, as introduced plants and invasive alien species became more common this model did not explain the resulting dynamics. The quantitative climax model also did not adequately explain much of the plant community dynamics observed in semi-arid and arid ecosystems and in the shrub steppe (Westoby et al. 1989, Laycock 1991, Fuhlendorf et al. 1996, Fuhlendorf and Smeins 1997), especially when natural disturbance regimes were altered (e.g., fire frequency, herbivory levels).

Non-Equilibrium Model

Non-equilibrium theory as applied to rangelands was developed from observations that succession did not always return a plant community to its site potential. These observations, and later research results, recognized that linear succession failed to adequately predict the observed vegetation changes (West et al. 1984, Archer 1989, Laycock 1991).

The idea of alternative steady states as distinct entities for management purposes was first identified by Westoby et al. (1989). Working in South Texas, Archer (1989) discussed the changes in structure and process in a plant community that may accompany a change in disturbance regime. He specifically addressed the cessation of fire as a change in the disturbance regime that results in the establishment of woody vegetation in a system previously dominated by grasses. This change from herbaceous to woody plant domination caused a change in structure and process that could not be reversed by reintroduction of the historic disturbance regime. Significant inputs of energy (machinery, seed, etc.) would be required to return the system to its original structure and function. He also stated that the change is likely to be short lived without continued energy inputs. Friedel (1991) discussed the concept of thresholds between domains from one state to another. She discussed several examples and recognized two main types of thresholds, changes from herbaceous to woody dominated systems and changes from stable to eroding soils.

In 1997 the USDA-Natural Resources Conservation Service officially incorporated concepts of multiple steady states and the use of state-and-transition models (STM) into range sites and enlarged the application into the current ecological site concept (USDA 2003). Ecological sites are the basic unit of inventory, assessment and management on rangeland in the United States. Bestelmeyer et al. (2009) suggested a revision to the range site definition to accommodate the state-and-transition concepts that are now being incorporated into ecological sites and the descriptions of those sites. An ecological site was defined as a class of land based on recurring soil, landform, geological, and climate characteristics that differs from other such classes in the production and composition of plant species under the disturbance regime of reference conditions, associated dynamic soil property levels, and ecosystem services provided. Bestelmeyer et al. (2009) continued by discussing the uniqueness of the ecological site in its responses to management and the processes of degradation and restoration. They further state that ecological sites reoccur on similar soil components, and that within the range of the ecological site it can be observed in one or more ecological states.

Stringham et al. (2003) provided definitions and a framework to develop STM. These proposed models are at the ecological site scale, with each unique ecological site having its own STM. These definitions stress the relationship of the soil and vegetation components to the functioning of the ecological processes within a state. The soil component, having been developed through time integrating the parent material, climate, landscape position and with the interaction of the resulting biota, determines the capability of the ecological site. This interaction between soil and vegetation determines the functional relationship of the ecological processes and the site resiliency and resistance to change in ecological process function. Table 1.1 displays the definitions for states, community phases, community pathways, transitions and thresholds.

Table 1.1. Definitions of the components of a state-and-transition model adapted from Stringham et al. (2003).

State	A recognizable, resilient complex of two components, the soil base and the vegetation structure. The components are connected through integrated ecological processes that interact to produce a resilient equilibrium expressed by a specific suite of vegetative communities.
Community Phase	Different assemblages within a state that represent the natural range of variability within the state. These dynamics may be driven by succession and regression and/or non-equilibrium events.
Community Pathway	Causes of change between community phases within a state.
Transition	Trajectory of change caused by natural or management actions that degrade the integrity of one or more of the state's primary ecological processes beyond the point of self repair.
Threshold	A boundary in space and time between states or along a transitions where one or more of the state's primary ecological processes has been degraded beyond self repair.

The definitions and framework shown in Figure 1.1 provided the necessary structure to begin the process of developing process based STM derived from data for use in understanding the ecological dynamics involved in rangeland management.

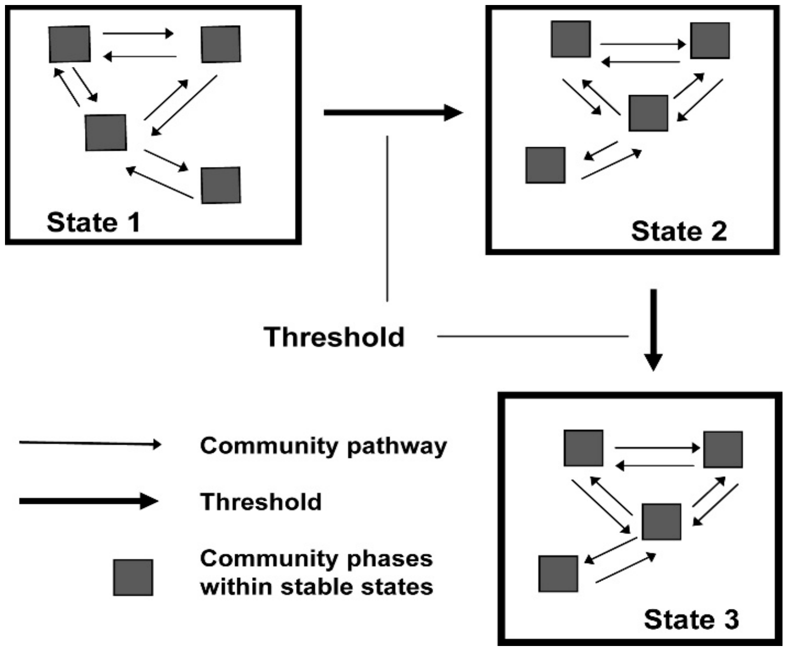

Figure 1.1. Model framework adapted from Stringham et al. (2003) showing community dynamics within a state, transitions and thresholds.

Briske et al. (2008) further refined the model to include more emphasis on resilience within the state and the feedbacks involved in building resilience. Figure 1.2 illustrates this refinement.

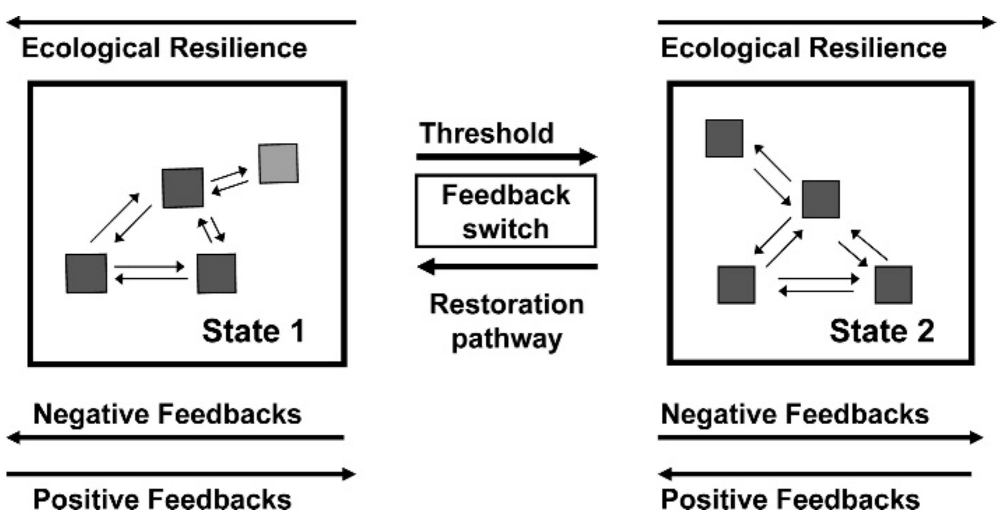

Figure 1.2. Illustration of resilience concepts in state-and-transition models using feedback mechanisms to help explain the movement toward and away from thresholds (adapted from Briske et al. 2008).

Peterson et al. (1998) explained resilience as the amount of change or disruption that is required to transform a system from being maintained by one set of mutually reinforcing processes and structures to a different set of processes and structures. The community pathways between the community phases are feedback mechanisms that maintain the mutually reinforcing processes and structures. Briske et al. (2008) provide additional definitions and suggested inclusion of resilience based concepts into the state-and-transition models (Table 1.2). Bestelemeyer et al. (2009) further refined many of the definitions and added concepts of reference state and reference community phase and their definitions (Table 1.3). Together the definitions and frameworks from Stringham et al. (2003), Briske et al. (2008) and Bestelmeyer et al. (2009) form the basic concepts of state and transition model development in use today.

Table 1.2. Additional definitions and resilience-based concepts for state-and-transition model framework (adapted from Briske et al. 2008).

At-risk community phase	Plant community phase within a state that is most vulnerable to exceeding the resilience limits of the state.
Ecological resilience	Amount of change or disruption that is required to transform a system from being maintained by one set of mutually reinforcing processes and structures to a different set of processes and structures.
Feedback mechanisms	Ecological processes that enhance (negative) or decrease (positive) ecosystem resilience.
Feedback switch	Point at which feedbacks shift from a dominance of negative feedbacks that maintain ecosystem resilience to a dominance of positive feedbacks that decrease ecosystem resilience and contribute to a state change.
Restoration pathways	Re-establishment of prethreshold states following active restoration of negative feedback mechanisms necessary to maintain the resilience of these states.
States	A suite of plant community phases occurring on similar soils that interact with the environment to produce persistent functional and structural attributes associated with a characteristic range of variability.
Thresholds	Conditions sufficient to modify ecosystem structure and function beyond the limits of ecological resilience, resulting in the formation of alternative states.
Triggers	Biotic or abiotic variables or events, acting independently or in combination, that initiate threshold-related processes by contributing to the immediate loss of ecosystem resilience.

Table 1.3. Additional definitions of STM components including reference state and reference community phase (adapted from Bestelemyer et al. 2009).

State	A suite of temporally-related plant communities and associated dynamic soil properties that produce persistent, characteristic structural and functional ecosystem attributes.
Reference state	The state supporting the largest array of potential ecosystem services and from which all other states and phases can be derived; often considered to represent a historical or natural range of variability or the set of conditions most preferred by a society.
Community phases	Distinctive plant communities and associated dynamic soil property levels that can occur over time within a state.
Reference phase	The phase of the reference state exhibiting the structural and functional properties that impart resilience to this state.
At-risk phase	The community phase that is most vulnerable to transition to an alternative state (i.e., least resilient).
Transition	The mechanisms by which one state is transformed into another state.
Trigger	Events processes, and drivers that initiate a transition to an alternative state. Triggers can be indicated by changes in plant community patterns that result in altered feedbacks or increased risk of sudden transition from the at-risk phase.
Threshold	Conditions defined by vegetation/soil characteristics and related processes that distinguish alternative states and that preclude autogenic (unassisted) recovery of the former state.

JUNIPER ECOLOGY

Juniper dominated plant community associations currently cover approximately 40 million hectares in the western United States (West et al. 1975, Fowler et al. 1985, Romme et al. 2009). These plant communities are generally transitional between lower elevation arable lands and higher elevation coniferous forests and provide critical seasonal habitat for wildlife and forage for livestock (Roundy and Vernon 1999).

In parts of the southwest, juniper species including one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg) have encroached into adjacent grasslands or increased in stand density (Woodbury 1947, Jameson 1962, Mueggler 1976, Lanner and Van Devender 1998, Gottfried 1999). Explanations for this encroachment include a decrease in fine fuels from grazing, active fire suppression, reduction of grass competitiveness from overgrazing, spread of seed by livestock and wildlife, and climate change (Johnsen 1962, White 1965, Miller and Wigand 1994, Allen and Breshears 1998, Lanner and Van Devender 1998). Increased juniper cover is generally associated with reduced herbaceous cover and production (Jameson 1962, Arnold 1964, Arnold et al. 1964, Jameson 1967, Jameson 1970, Clary and Jameson 1981, and Pieper 1983) and increased runoff and erosion (Arnold 1964, Jameson 1970, Wilcox et al. 1996a, Wilcox et al. 1996b, Pierson et al. 2007, Petersen and Stringham 2008).

One-seed juniper is common in the desert grassland and piñon-juniper ranges throughout New Mexico and in southeastern and north-central Arizona. This species also occurs in southern Colorado, western Texas and western Oklahoma (Pieper 1977, Allen and Peet 1990). One-seed juniper is a native, long lived, evergreen tree with often shrubby form, 3-12 m tall with several curved limbs arising near the base. One-seed juniper produces small, berry like cones. Mature cones are dark blue to purple or brownish, and succulent, or at least somewhat fleshy with one seed per fruit (Tueller and Clark. 1975).

Mature one-seed junipers have both tap and lateral root systems. The taproots range from 46 cm to more than 3.7 m in length. Lateral roots are widespread, commonly being 2.5 to 3 times as long as the tree is tall. Most lateral roots are in the surface 1 m of the soil, most of those concentrated below the surface 15 cm (Johnsen 1962). The deep root system of mature one-seed junipers is adapted for growth on dry sites (Johnsen 1962). Foxx and Tierney (1987) reported rooting depths ranging from 5-60 m. One-seed juniper has the ability to stop active growth when moisture is limited but can resume growth when moisture availability improves (Herman 1956). Johnsen (1962) reported three growth rings formed in the summer of 1956 corresponding to three rainy periods separated by three dry periods in which the soil dried. This growth pattern represents an important adaptation allowing junipers to survive on harsh, arid and semi-arid sites.

Trees first produce seed at 10 to 30 years of age, although maximum seed production generally does not occur until 50 years of age (Johnsen and Alexander 1974, Schott and Pieper 1986). Trees as short as 46 cm in height can produce seed (Johnsen 1962). One-seed juniper typically produces large seed crops at 2- to 5-year intervals (Johnsen and Alexander 1974). Dispersal of one-seed juniper seeds may occur through water, gravity, or by any of a number of birds and mammals (Johnsen 1962). Animal dispersal may be particularly important, as digestive processes may enhance germination (Balda 1987). Most seed cones occur on the outer edges of trees where they are most visible and accessible to birds (Salomonson 1978). On some sites in New Mexico, as much as 95% of juniper reproduction could be attributed to bird dispersal (Balda 1987). Domestic sheep and cattle may also aid in seed dispersal (Johnsen 1962). According to Johnsen (1962), seed viability is not harmed by long periods of drought. Viable seed in the soil may endure prolonged drought and still germinate when conditions become favorable (Johnsen 1962). Seedling establishment of one-seed juniper is often very poor even when good germination occurs (Schott and Pieper 1987). The growth rate has been characterized as slow with medium vigor. Shade may be important for good establishment and early growth of one-seed juniper (Jameson 1965). Emergence appears to be somewhat greater under trees or shrubs than in interspaces where humidity and temperature fluctuations are more extreme (Johnsen 1962).

PROJECT RATIONALE

The quantitative climax model of plant community dynamics has been used for more than 50 years in the management of rangelands. It has not been effective in providing managers with timely options for restoration of many ecosystems (Laycock 1991, Fuhlendorf et al. 1996, Fuhlendorf and Smeins 1997). State-and-transition models are becoming the decision support tool of choice for developing, adapting and selecting management options, especially when dealing with invasive plant encroachment into grassland ecosystems. Currently, most STMs are based on experience and professional knowledge with little data to support the descriptions of plant community pathways and transitions. The use of models to describe states and predict transitions across thresholds using quantitative indicators has been widely called for (Friedel 1991, Laycock 1991, Herrick et al. 2002, Bestelmeyer et al. 2003, Bestelmeyer et al. 2004, Briske et al. 2005, Briske et al. 2006, and Bestelmeyer et al. 2009).

The increase in extent and density of juniper has been well documented (West et al. 1975, Fowler et al. 1985, Romme et al. 2009) and some effects of juniper have been well studied. Increased juniper cover is generally associated with reduced herbaceous cover and production (Jameson 1962, Arnold 1964, Arnold et al. 1964, Jameson 1967, Jameson 1970, Clary and Jameson 1981, Pieper 1983, and Miller et al. 2000) and increased runoff and erosion (Arnold 1964, Jameson 1970, Wilcox et al. 1996a, Wilcox et al. 1996b, Pierson et al. 2007, Petersen and Stringham 2008). However, there have been relatively few studies exploring the effects of juniper mortality on soil water availability, and most have focused on medium to fine textured soils and included only near surface soil moisture measurements (Gifford 1973, Wilcox et al. 1996a, Wilcox et al. 1996b, Reid et al. 1999, Bates et al. 2000). Newman et al. (1997) and Weltzin and McPherson (1997) quantified water use throughout the soil profiles of untreated stands of one-seed juniper and Emery oak (*Quercus emoryi* Torr.) on fine textured soils and under a winter dominated precipitation pattern.

Long term studies have been limited to historical, pre-Columbian, and Paleo-Indian disturbance regimes and mechanisms (Betancourt 1987, Allen et al. 1998, Mueller et al. 2005, Briggs et al. 2007, Romme et al. 2009), but these do not address the effects of juniper on the processes involved in ecological resilience. Most long term studies of junipers have focused on landscape level plant community composition changes in response to changes in climate or fire regimes (Brown et al. 1997, Swetnam and Betancourt 1998, Floyd et al. 2000, Brown et al. 2001, Rees et al. 2001, Breshears et al. 2005, Bates et al. 2007). One exception is Barnitz et al. (1990) who examined site specific juniper treatment in south central New Mexico over a 31 year time span. However, they looked only at vegetation composition changes and tree density following treatment and did not address ecological processes such as water, energy or nutrient cycles.

The study described here provided an opportunity to use long term ecological response data to test a proposed STM, and to identify resilience feedback mechanisms. Indicators of state properties and of feedback switches were also identified and quantified.

This study was conducted in three main phases on four private ranches in central New Mexico. Data were collected for the first phase of this study in 1984, with supplemental data collected from 1981 – 1986. The objectives of the first phase of the study were to test the effects of juniper mortality on soil water in the top 125 cm of a coarse textured soil. Non-replicated data were also collected on plant production by species as a basis for developing future studies on relationships between changes in soil water and plant composition and production.

Data for the second phase of the study were collected from 1985 – 1989 and again in 2003. The objectives of this phase were to test the hypothesis that vegetation production, cover and soil moisture will increase after juniper treatment on coarse textured soils and that these changes would be sustained over time.

The third phase of this study was to test a proposed process based state-and-transition model for the Deep Sand Savannah ecological site (070CY123NM) to further develop and to quantify the model using data collected within this ecological site over a period of 22 years.

LITERATURE CITED

Allen, C. D. and D. D. Breshears 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings, National Academy of Sciences of the United States of America. 95(25): 14839-14842.

Allen, C. D., J. L. Betancourt and T. W. Swetnam. 1998. Landscape changes in the southwestern United States: techniques, long-term data sets, and trends. *In:* T. D. Sisk [ED.]. Perspectives on the land use history of North America: a context for understanding our changing environment. Reston, VA, USA: US Geological Survey, Biological Science Report USGS/BRD/BSR-1998-0003. pp.71–84.

Allen, R. B. and R. K. Peet. 1990. Gradient analysis of forests of the Sangre de Cristo Range, Colorado. Canadian Journal of Botany. 68: 193-201.

Archer, S. 1989. Have southern Texas savannas been converted to woodland in recent history? American Naturalist 134: 545-561.

Arnold, J. F. 1964. Zonation of understory vegetation around a juniper tree. Journal of Range Management. 47: 41-42.

Arnold, J. F., D. A. Jameson and E. H. Reid. 1964. The pinyon-juniper type of Arizona; effects of grazing, fire and tree control. U. S. Dept. Agr. Prod. Res. Rep. No. 84. Washington, D. C.

Balda, R. P. 1987. Avian impacts on pinyon-juniper woodlands. *In:* Everett, Richard L., compiler. Proceedings--pinyon-juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 525-533. [4993]

Barnitz, J. A. Jr., S. M. Armentrout, V. W. Howard, Jr., R. D. Pieper and G. M. Southward. 1990. Vegetational changes following two-way cabling of pinyon-juniper in south-central New Mexico. New Mexico Agr. Exp. Sta. Bull. 749. 36p.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2000. Understory dynamics in cut and uncut western juniper woodlands. Journal of Range Management 53:119-126.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2007. Long-term vegetation dynamics in a cut western juniper woodland. Western Norrth American Naturalist 67:549-561.

Bestelmeyer, B. T., J. R. Brown, K. M. Havstad, R. Alexander, G. Chavez and J. E. Herrick. 2003. Development and use of state-and-transition models for rangelands. Journal of Range Management 56: 114-126.

Bestelmeyer, B. T., J. E. Herrick, J. R. Brown, D. A. Trujillo and K. M. Havstad. 2004. Land Management in the American Southwest: A State-and-Transition Approach to Ecosystem Complexity. Environmental Management 34(1): 38-51.

Bestelmeyer, B. T., A. J. Tugel, G. L. Peacock Jr., D. G. Robinett, P. L. Shaver, J. R. Brown, J. E. Herrick, H. Sanchez and K. M. Havstad. 2009. State-and-transition models for heterogeneous landscapes: A strategy for development and application. Rangeland Ecology and Management 62(1): 1-15.

Betancourt, J. L. 1987. Paleoecology of pinyon-juniper woodlands: summary. *In:* R. L. Everett [ED]. Proceedings-Pinyon-Juniper Conference; 13 -16 January 1986; Reno, NV, USA. Ogden, UT, USA: USDA, Forest Seervice, Intermountain Research Station, GTR INT-215. pp. 129–139.

Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice, W. H. Romme, J. H. Kastens, M. L. Floyd, J. Belnap, J. J. Anderson, O. B. Myers and C. W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences 102(42): 15144– 5148.

Briggs, J. M., H. Schaafsma and D. Trenkov. 2007. Woody vegetation expansion in a desert grassland: prehistoric human impact? Journal of Arid Environments 69: 458-472.

Briske, D. D., S. D. Fuhlendorf and F. E. Smiens. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management 58(1): 1-10.

Briske, D. D., S. D. Fuhlendorf and F. E. Smiens. 2006. A unified framework for assessment and application of ecological thresholds. Rangeland Ecology and Management 59(3): 225–236.

Briske, D. D., B. T. Bestelmeyer, T. K. Stringham and P. L. Shaver. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management 61(4): 359–367.

Brown, J. H., T. J. Valone and C. G. Curtin. 1997. Reorganization of an arid ecosystem in response to recent climate change. Proceedings of the national Academy of Sciences 94(18): 9729–9733.

Brown, J. H., T. G. Whitham, S. K. Morgan Ernest, and C. A. Gehring. 2001. Complex species interactions and the dynamics of ecological systems: long-term experiments. Science. 293(5530): 643–650.

Clary, W. P., and D. A. Jameson. 1981. Herbage production following tree and shrub removal in the Pinyon-Juniper type in Arizona. Journal of Range Management. 34: 109-113.

Clements, F. E. 1928. Plant succession and indicators. The H.W. Wilson Co., New York City, NY.

Dyksterhuis, E. J. 1949. Condition and management of range land based on quantitative ecology. Journal of Range Management 2(3): 104 – 115.

Dyksterhuis, E. J. 1958. Range conservation as based on sites and condition classes. Journal of Soil and Water Conservation 13: 151-155.

Floyd, M. L., W. H. Romme and D. Hanna. 2000. Fire history and vegetation pattern in Mesa Verde National Park. Ecological Applications 10: 1666–1680.

Fowler, J. M., B. E. Peacock and M. J. Schaber. 1985. Pinyon-juniper woodland type in New Mexico: Asset or liability. New Mex. State Univ. Agr. Exp. Sta. Bull. 718. Las Cruces, NM.

Foxx, T. S. and G. D. Tierney. 1987. Rooting patterns in the pinyon-juniper woodland. *In:* Everett, Richard L., compiler. Proceedings--pinyon-juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: pp. 69-79.

Friedel, M. H. 1991. Range condition assessment and the concept of thresholds: A viewpoint. Journal of Range Management 44(5): 422–426.

Fuhlendorf, S. D., F. E. Smeins and W. E. Grant. 1996. Simulation of a fire-sensitive ecological threshold: A case study of Ashe juniper on the Edwards Plateau of Texas, USA. Ecological Modeling 90: 245-255.

Fuhlendorf, S. D. and F. E. Smeins. 1997. Long-term vegetation dynamics mediated by herbivores, weather and fire in a *Juniperus-Quercus* savanna. Journal of Vegetation Science 8: 819-828.

Gifford, G. F. 1973. Runoff and sediment yields from runoff plots on chained pinyon-juniper plots in Utah. Journal of Range Management. 26: 440-443.

Gottfried, G. J. 1999. Pinyon-juniper woodlands in the southwestern United States. *In:* Ffolliott, Peter F.; Ortega-Rubio, Alfredo, eds. Ecology and management of forests, woodlands, and shrublands in the dryland regions of the United States and Mexico: perspectives for the 21st century. Co-edition No. 1. Tucson, AZ: The University of Arizona; La Paz, Mexico: Centro de Investigaciones Biologicas del Noroeste, SC; Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: pp. 53-67.

Herman, F. R. 1956. Growth and phenological observations of Arizona junipers. Ecology. 37: 193-195.

Herrick, J. E., J.R. Brown, A. J. Tugel, P. L. Shaver and K. M. Havstad. 2002. Application of soil quality to monitoring and management. Agronomy Journal 94: 3-11.

Jameson, D. A. 1962. Effects of burning on a galleta-black grama range invaded by juniper. Ecology. 43(4): 760-763.

Jameson, D. A. 1965. Arrangement and growth of pinyon and one-seed juniper trees. Plateau. 37: 121-127.

Jameson, D. A. 1967. The relationship of tree overstory and herbaceous understory vegetation. Journal of Range Management. 20: 247-249.

Jameson, D. A. 1970. Juniper root competition reduces basal area of blue grama. Journal of Range Management. 23: 217-218.

Johnsen, T. N., Jr. 1962. One-seeded juniper invasion of northern Arizona grasslands. Ecological Monographs. 32(3): 187-207.

Johnsen, T. N., Jr. and R. A. Alexander. 1974. Juniperus L. juniper. *In:* Schopmeyer, C. S., tech. coord. Seeds of woody plants in the United States. Agric. Handb. 450. Washington, DC: U.S. Department of Agriculture, Forest Service: pp. 460-469.

Lanner, R. M. and T. R. Van Devender. 1998. The recent history of pinyon pines in the American Southwest. *In:* Richardson, David M., ed. Ecology and biogeography of Pinus. Cambridge, United Kingdom: The Press Syndicate of the University of Cambridge: pp. 171-182.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North American rangelands: A viewpoint. Journal of Range Management 44(5): 427–433.

Miller, R. F. and P. E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. Bioscience. 44(7): 465-474.

Miller, R. F., T. J. Svejcar, and J. A. Rose. 2000. Impacts of western juniper on plant community composition and structure. Journal of Range Management 53: 574-585.

Mueggler, W. F. 1976. Ecological role of fire in western woodland and range ecosystems. *In:* Use of prescribed burning in western woodland and range ecosystems: Proceedings of the symposium; 1976 March 18-19; Logan, UT. Logan, UT: Utah State University, Utah Agricultural Experiment Station: pp. 1-9.

Mueller, R. C., C. M. Scudder, M. E. Porter, R. T. Trotter III, C. A. Gehring and T. G. Whitham. 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. Journal of Ecology 93(6): 1085–1093.

Newman, B. D., A. R. Campbell and B. P. Wilcox. 1997. Tracer based studies of soil water movement in semi-arid forests of New Mexico. Journal of Hydrology. 196: 251-270.

Petersen, S. L. and T. K. Stringham. 2008. Infiltration, runoff, and sediment yield in response to western juniper encroachment in southeast Oregon. Rangeland Ecology and Management. 61: 74–81.

Peterson, G., C. R. Allen and C. S. Holling. 1998. Ecological resilience, biodiversity, and scale. Ecosystems 1: 6–18.

Pieper, R. D. 1977. The southwestern pinyon-juniper ecosystem. *In:* Aldon, Earl F.; Loring, Thomas J., technical coordinators. Ecology, uses, and management of pinyon-juniper woodlands: Proceedings of the workshop; 1977 March 24-25; Albuquerque, NM. Gen. Tech. Rep. RM-39. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: pp. 1-6. [17251]

Pieper, R. D. 1983. Overstory-understory relationships: Pinyon-juniper and juniper woodlands. *In:* Bartlett, E. T. and D. R. Betters (Eds.) Overstory-understory relationships in western forests. West. Reg. Res. Pub. No. 1. Colo. State Univ. Agr. Exp. Sta. Ft. Collins, CO.

Pierson, F. B., J. D. Bates, T. J. Svejcar, and S. P. Hardegree. 2007. Runoff and erosion after cutting western juniper. Rangeland Ecology and Management 60:285-292.

Rees, M., R. Condit, M. Crawley, S. Pacala and D. Tilman. 2001. Long-term studies of vegetation dynamics. Science. 293(5530): pp. 650–655.

Reid, K. D., B. P. Wilcox, D. D. Breshears and L. MacDonald. 1999. Runoff and erosion in a Piñon-Juniper woodland: Influences of vegetation patches. Soil Science Society of America Journal. 63: 1869–1879.

Romme, W. H., C. D. Allen, J. D. Bailey, W. L. Baker, B. T. Bestelmeyer, P. M Brown, K. S. Eisenhart, M. L. Floyd, D. W. Huffman, B. F. Jacobs, R. F. Miller, E. H. Muldavin, T. W. Swetnam, R. J. Tausch and P. J. Weisberg. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Roundy, B. A. and J. L. Vernon. 1999. Watershed values and conditions associated with pinon-juniper communities. *In* Proceedings: Ecology and Management of Pinon-Juniper Communities Within the Interior West. USDA Forest Service. Proceedings RMRS-P-9. pp. 173-187.

Salomonson, M. G. 1978. Adaptations for animal dispersal of one-seed juniper seeds. Oecologia. 32: 333-339.

Sampson, A. W. 1919. Plant succession in relation to range management. U.S. D.A. bulletin 791.

Schott, M. R. and R. D. Pieper. 1986. Succession in pinyon-juniper vegetation in New Mexico. Rangelands. 8(3): 126-128.

Schott, M. R. and R. D. Pieper. 1987. Succession in tree pits following cabling in pinyon-juniper communities. The Southwestern Naturalist. 32(3): 399-402.

Stringham, T. K., W. C. Krueger and P. L. Shaver. 2003. State-and-transition modeling: an ecological process approach. Journal of Range Management 56: 106-113.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American southwest. Journal of Climate. 11: 3128–3147.

Tansley, A. G. 1935. The use and abuse of vegetational concepts and terms. Ecology 16:284-307.

Tueller, P. T. and J. E. Clark. 1975. Autecology of pinyon-juniper species of the Great Basin and Colorado Plateau. *In:* The pinyon-juniper ecosystem: a symposium: Proceedings; 1975 May; Logan, UT. Logan, UT: Utah State University, College of Natural Resources, Utah Agricultural Experiment Station: pp. 27-40.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2003. National range and pasture handbook, revision 1. Washington, D.C.

Weltzin, J. F. and G. R. McPherson. 1997. Spatial and temporal soil moisture resource partitioning by trees and grasses in a temperate savanna, Arizona, USA. Oecologia 112: 156-164.

West, N. E., K. H. Rea and R. J. Tausch. 1975. Basic synecological relationships in pinyon-juniper woodlands. *In:* Gifford, G. F. and F. E. Busby (Eds.). The pinyon-juniper ecosystem. Utah State Univ. Agr. Exp. Sta. Logan, UT.

West, N. E., F. D. Provenza, P. S. Johnson and M. K. Owens.1984. Vegetation change after 13 years of Livestock grazing exclusion on sagebrush semidesert in west central Utah. Journal of Range Management 37(3): 262-264.

Westoby, M., B. Walker and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42(4): 266–274.

White, L. D. 1965. The effects of a wildfire on a desert grassland community. [thesis]. Tucson, AZ: University of Arizona. 107p.

Wilcox, B. P., C. D. Allen, B. D. Newman, K. D. Reid, D. Brandes, J. Pitlick and D. W. Davenport. 1996a. Runoff and erosion on the Pajarito Plateau: observations from the field. pp. 433-439. *In:* New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Regions, 1996. New Mexico Geological Society. Albuquerque, N.M.

Wilcox, B. P., J. Pitlick, C. D. Allen,and D. W. Davenport. 1996b. Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. pp. 61-77. *In:* M. G. Anderson and S. M. Brooks (eds.) Advances in Hillslope Processes, Vol. 1. John Wiley and Sons, New York.

Woodbury, A. M. 1947. Distribution of pigmy conifers in Utah and northeastern Arizona. Ecology. 28(2): 113-126.

CHAPTER 2: SHORT TERM RESPONSE OF SOIL MOISTURE AND VEGETATION TO ELIMINATION OF ONE-SEED JUNIPER CANOPY

ABSTRACT

The effects of the removal of one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg) on soil moisture and vegetation composition and production were studied. One-seed juniper was killed on site by the application of the herbicide tebuthiuron in the fall of 1981. Soil moisture was measured at four depths during the growing season of 1984. A repeated measures analysis of variance showed that the soil moisture in the treated plots was significantly higher throughout the season. The available soil moisture in the treated profile was consistently above the 1.5 MPa moisture content and for most of the season had twice as much as the control plots. These differences were associated with reductions in woody vegetation and increases in herbaceous production from 1981-1986 in treated plots, resulting in higher total production in the treated plots for 1981-1986. The potential options for management aimed at livestock production, wildlife habitat and watershed values are much greater with increased available soil moisture and the resultant change in vegetation composition and production.

INTRODUCTION

Juniper dominated plant community associations currently cover approximately 40 million hectares in the western United States (West et al. 1975, Fowler et al. 1985, Romme et al. 2009). In parts of the southwest, juniper species including one-seed juniper (*Juniperus monosperma* Engelm.) Sarg) have encroached into adjacent grasslands, or increased in stand density (Woodbury 1947, Jameson 1962, Mueggler 1976, Lanner and Van Devender 1998, Gottfried 1999). Explanations for this encroachment include fire suppression, a decrease in fine fuels from grazing, reduction of grass competitiveness by overgrazing, spread of seed by livestock and wildlife, and climate change (Johnsen 1962, White 1965, Miller and Wigand 1994, Allen and Breshears 1998, Lanner and Van Devender 1998).

Increased juniper cover is generally associated with reduced herbaceous cover and production (Jameson 1962, Arnold 1964, Arnold et al. 1964, Jameson 1967, Jameson 1970, Clary and Jameson 1981, and Pieper 1983) and increased runoff and erosion (Arnold 1964, Jameson 1970, Wilcox, et al. 1996a, Wilcox et al. 1996b, Pierson et al. 2007, Peterson and Stringham 2008). It has been argued that these changes should lead to reductions in soil water availability (Arnold 1964, Jameson 1970). If true, juniper mortality should result in increased soil water availability. There have been relatively few studies exploring the effects of juniper mortality on soil water availability, and most have focused on medium to fine textured soils and included only near surface soil moisture measurements (Gifford 1973, Wilcox et al. 1996a, Wilcox et al. 1996b, Breshears et al. 1997a, Reid et al. 1999, Bates et al. 2000). Exceptions include Newman et al. (1997) and Weltzin and McPherson (1997). These studies looked at water use throughout the soil profiles on mature untreated stands of one-seed juniper and Emery oak *Quercus emoryi* Torr. Breshears et al. (1997a) showed that plant water potential of one-seed juniper correlates with variations in soil moisture at shallow depths, indicting direct competition for soil available moisture with grasses.

One-seed junipers have both tap and lateral root systems. The taproots range from 46 cm to more than 3.7 m in length. Lateral roots are widespread, commonly being 2.5 to 3 times as long as the tree is tall. Most lateral roots are in the surface 1 m of the soil, and most are concentrated below the surface 15 cm, expanding to fully occupy the interspaces between trees (Johnsen 1962). Foxx and Tierney (1987) reported rooting depths ranging from 5-60 m. One-seed juniper has the ability to stop active growth when moisture is limited but can resume growth when moisture availability improves (Herman 1956). This growth pattern may represent an important adaptation allowing junipers to survive on harsh, arid sites.

STUDY OBJECTIVES

The objective of this study was to test the hypothesis that juniper mortality would increase soil water in the top 125 cm of a coarse textured soil in central New Mexico, USA. Non-replicated data were also collected on plant production by species as a basis for developing future studies on relationships between changes in soil water and plant composition and production.

METHODS

Study Area

The study was completed on a private ranch located approximately 3 km southwest of the Salinas National Monument, Gran Quivira Unit in central New Mexico at an elevation of 1920 m. The area has a flat to rolling dune topography. Soils in the area are similar and formed from aeolian sand deposits derived from mixed sources. The dominant characteristic is the coarse surface texture. The soil moisture regime at this location is aridic-ustic and the soil is classified as mixed, mesic, Ustic Torripsamments. The soil map unit component on the study area is Mespun fine sand (USDA 1988). This classification was confirmed with a soil pit and several auger holes at the study site. The soil is deep and excessively drained, the surface layer is a brown fine sand about 28 cm thick. The underlying material is strong brown fine sand in excess of 155 cm deep (USDA 1988). This map unit component is correlated to the Deep Sand Savannah Ecological Site in MLRA G070C, Central New Mexico Highlands, site number 070CY123NM (USDA 2004).

The climate of the study area is characterized as semi-arid continental. The average annual precipitation ranges from 33-40 cm depending on the period of record. The thirty year average (1951-1980) immediately preceding the study was 36.2 cm. Precipitation during the study period, measured at the Gran Quivira Unit of the Salinas National Monument was above average. Precipitation during the first three years of the study (1981-1983) was 6.8, 3.1 and 5.4 cm above average. In 1984, 1985 and 1986 it was 8.88, 13.08 and 23.77 cm above average. Seventy five percent of the precipitation falls from April to October, primarily in the form of high intensity thunderstorms. Temperatures are characterized by distinct seasonal changes and large annual and diurnal fluctuations. The average annual air temperature is about 10^{o}C, with extremes of -34^{o}C in the winter to 40^{o}C in the summer. The average frost free season is 130–160 days. The last killing frost occurs in early May and the first killing frost is in early October (Western Regional Climate Center, 2009).

At the time of the study the vegetation in the study area was dominated by one-seed juniper greater than 4 meters tall and canopy cover in excess of 25%. Initial annual plant production transects indicated a total annual production of less than 400 kg/ha. The herbaceous component of the total was less than 270 kg/ha.

Procedures

An area of approximately four hectares was fenced and sixteen 38.1 m X 38.1 m plots were established. There was a 15.2 m buffer between each plot. Three different chemical application rates and a control plot were randomly assigned to plots and replicated four times. In the fall of 1981, tebuthiuron pellets were applied by hand in a crisscross pattern across the entire plot area. The active ingredient application rates were 1.1, 1.6 and 2.2 kg/ha. At the same time ten 0.89 m^2 (9.6 ft^2) frames were established in treated and control plots to use for double sampling herbaceous vegetation production in response to the chemical treatment (Wilm et al. 1944, Cook and Bonham 1977, USDA 2003). Two of the ten plots were expanded to 40.5 m^2 plots to estimate woody plant production (USDA 2003). Figure 2.1 shows the layout of the plots, application rates and location of the double sampling frames.

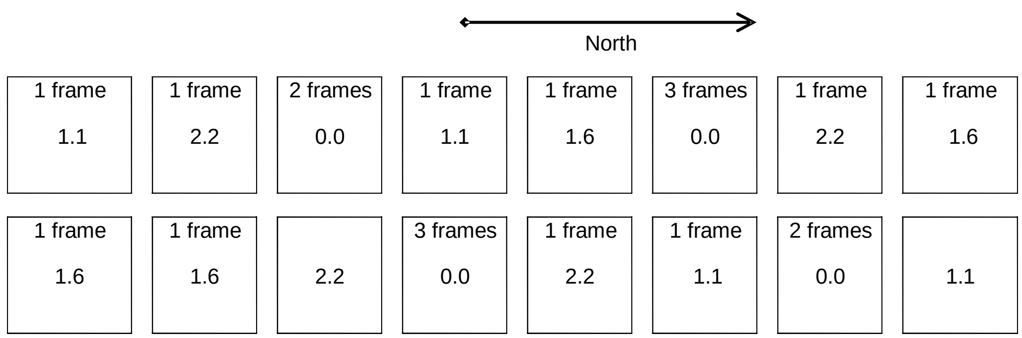

Figure 2.1. Plot design, double sampling frames and chemical application rates (kg/ha ai. Tebuthiuron). Each plot is 38.1 m X 38.1 m with a 15.2 m buffer strip between plots in all directions.

Double sampling was completed annually, after the first frost from 1981 to 1986. Annual production of each species in the 0.89 m^2 frame was estimated for each of the ten frames. After all ten frames were estimated, two frames were then clipped by species, and the clipped production was separated into a paper bag and weighed. Samples were oven dried and weighed again to calculate dry weight. Weight estimates were adjusted based on the clipped weights. Individual species weights were converted to kilograms per hectare and a total production value was summed.

It was noted that in the fall of 1982 most of the sand sagebrush (*Artemisia filifolia* Torr.) appeared to be dead and all the one-seed juniper showed effects of the chemical application. By the fall of 1983 all one-seed juniper and sand sagebrush on the treated plots appeared to be dead at all application rates. Therefore, all treatment plots were combined and the study compared treated and control plots.

The monitoring of the vegetation response continued through the fall of 1986. Due to time constraints and objectives of the study, the vegetation data collection part of this project was not replicated. Therefore, it is inappropriate to conduct any statistical analysis of the vegetation response to the treatment. Soil moisture measurements were begun in the spring of 1984. Soil moisture samples were collected thirteen times from 17 April 1984 to 27 November 1984 at approximately two week intervals. Locations within both the treated and control 38.1 m X 38.1 m plots were randomly selected for soil moisture collection. Four depths were used for sampling. The first depth was 25-30 cm below the surface. The next three depths were 43-51 cm, 91-99 cm and 122-130 cm below the soil surface. These depths represented the bottom of the soil surface horizon, the top, middle, and bottom respectively of the moisture control section for the soil. Soil moisture samples were collected using a soil auger. Samples were collected into a soil can, weighed, oven dried at 105^{o}C for 24 hours and weighed again (Gardner 1986). Wet weight minus dry weight divided by dry weight produces the percent of gravimetric moisture. This number was then converted to volumetric moisture by multiplying the gravimetric moisture content by the bulk density of the soil at each of the measured depths. Bulk density is the mass of a unit volume of soil and was determined at each of the depths in each plot by using a cylindrical core sampler (Gardner 1986). The known volume of soil was then oven dried and weighed to determine the bulk density.

Statistical Analysis

A repeated measures analysis of variance was performed on the soil moisture data using Statistica 7.1 software (StatSoft, Inc. 2005). The dependent variables were the data taken at each sample date and the categorical predictors were treatment and depth.

RESULTS

One-Seed Juniper Mortality

All of the one-seed juniper and sand sagebrush plants in the study area were killed at all tebuthiuron application rates. After two years of observations, no further work was done on the juniper mortality portion of this study.

Vegetation Response

The nomenclature used in this study is from *The PLANTS database* (USDA 2010). The response of the herbaceous vegetation to the decrease in one-seed juniper was readily apparent. The actual values for the total annual production of herbaceous species on the treated and control plots each year, 1981 - 1986, are shown in Table 2.1.

Table 2.1. Production and composition of vegetation on treated and control plots from 1981 – 1986.

			Schizachyrium scoparium	Andropogon hallii	Sporobolus cryptandrus	Sporobolus contractus	Muhlenbergia pungens	Bouteloua gracilis	Aristida purpurea	Achnatherum hymenoides	Other annual grasses	Other annual forbs	Other perennial forbs	Herb. Production
1981	Treated	kg/ha	53	26	28	31	34	27	0	0	12	67	20	298
		%	18	9	9	11	11	9	0	0	4	23	7	
	Control	kg/ha	92	23	25	19	23	18	15	0	0	65	13	294
		%	31	8	8	6	8	6	5	0	0	22	5	
1982	Treated	kg/ha	50	22	17	17	38	23	0	0	4	27	19	218
		%	23	10	8	8	17	11	0	0	2	12	9	
	Control	kg/ha	87	22	18	12	22	16	12	0	0	56	12	259
		%	34	9	7	5	9	6	5	0	0	22	5	
1983	Treated	kg/ha	207	12	25	67	44	22	0	0	226	59	34	697
		%	30	2	4	10	6	3	0	0	32	9	5	
	Control	kg/ha	92	17	23	20	22	12	17	0	0	56	13	273
		%	34	6	9	7	8	5	6	0	0	20	5	
1984	Treated	kg/ha	280	17	17	202	39	45	6	6	146	56	31	844
		%	33	2	2	24	5	5	1	1	17	7	4	
	Control	kg/ha	63	17	22	18	22	28	13	0	0	56	13	253
		%	25	7	9	7	9	11	5	0	0	22	5	
1985	Treated	kg/ha	286	314	45	134	67	34	28	22	78	95	39	1143
		%	25	27	4	12	6	3	2	2	7	8	3	
	Control	kg/ha	90	56	34	28	28	22	6	6	0	67	22	364
		%	25	15	9	8	8	6	3	2	0	18	6	
1986	Treated	kg/ha	476	230	9	205	103	22	9	9	16	142	183	1404
		%	34	16	1	15	7	2	1	1	1	10	13	
	Control	kg/ha	157	87	8	22	22	28	13	7	25	55	13	438
		%	36	20	2	5	5	6	3	2	6	13	3	

Total annual production of the treated plots increased more than 1000 kg/ha from 428 to 1438 kg/ha in the six years of the study. During the same six years the total annual production of the control plots also increased from 425 to 566 kg/ha. Herbaceous production also showed a large difference between the treated and control plots. The herbaceous species included all the grasses and forbs on the site. The annual herbaceous production on the treated plots far exceeded that of the control plots. The treated plot herbaceous annual production increased from 298 to 1404 kg/ha while the control plot herbaceous annual production changed from 294 to 438 kg/ha. Figure 2.2 illustrates the changes in total annual production on the treated and control plots.

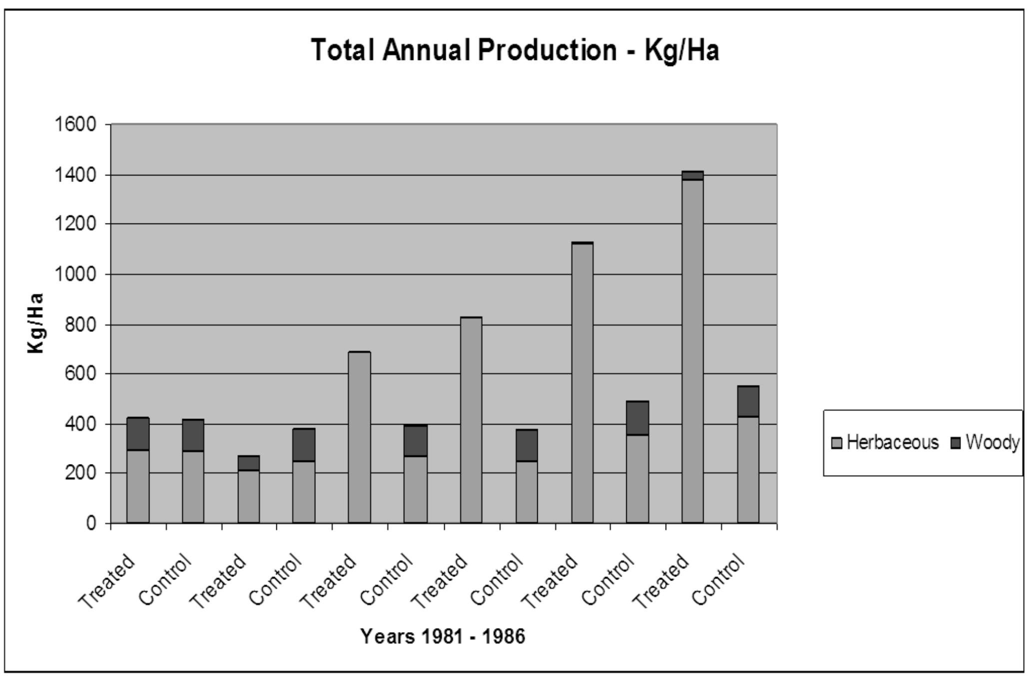

Figure 2.2. Total annual production for 1981–1986. Bars denote total annual production with stacks indicating herbaceous and woody production.

Species Composition

Herbaceous species composition by weight is shown in Table 2.1. The largest change in both species production and percent composition by weight was in the one-seed juniper and sand sagebrush. Both of these species were killed by the application of the herbicide and these data are not included in Table 2.1. It should be noted however, that by the end of the study, sand sagebrush seedlings were establishing in the treated plots as reflected in the difference in the total annual production and total annual herbaceous production for the treated sites (Figure 2.2) in 1985 and 1986. Most of the herbaceous species on the site increased in production following treatment. The most responsive were little bluestem (*Schizachyrium scoparium* (Michx.) Nash) and sand bluestem (*Andropogon hallii* Hack.) which were 3 and 2.6 times greater, respectively.

Soil Moisture

Soil moisture increased significantly ($F(1,8)=2766.6$, $p<0.0000$) for the season at all depths (Figure 2.3). When the three depths were compared separately, there was a significant ($\alpha=0.05$) treatment by depth interaction ($F(3,8)=104.66$, $p=0.0000$). Figure 2.4 shows the treatment by depth interaction.

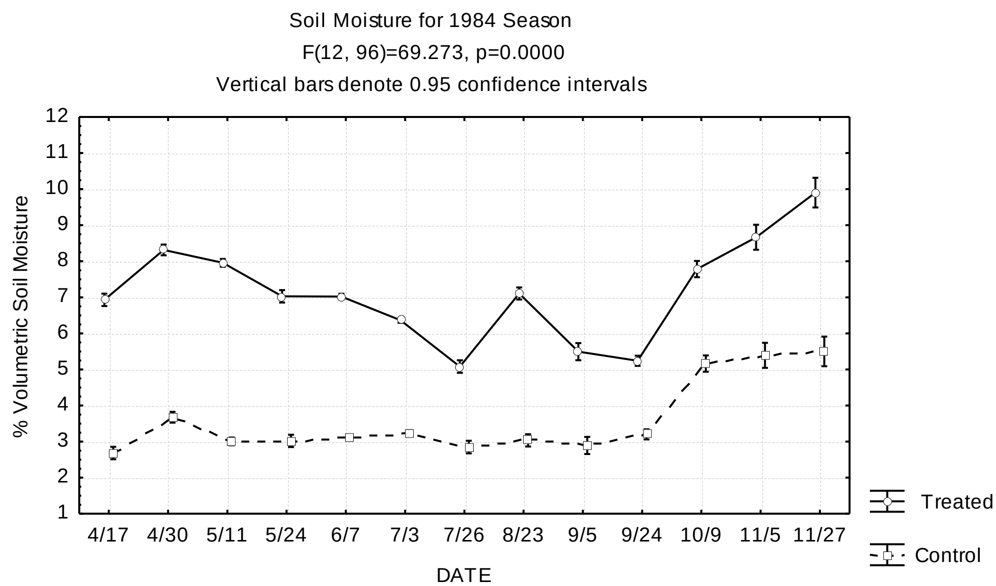

Figure 2.3. Differences in the mean volumetric soil moisture throughout the 1984 growing season.

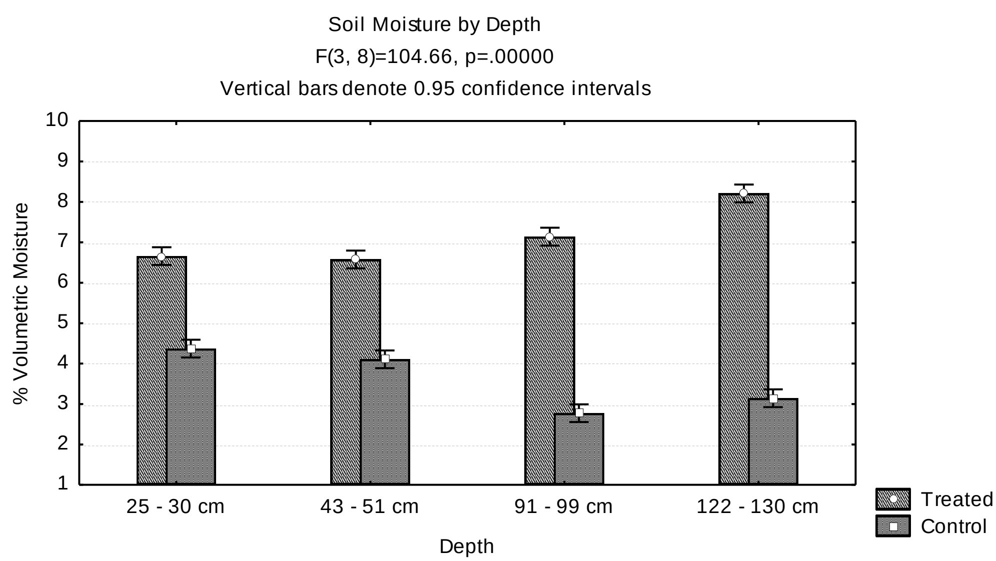

Figure 2.4. Soil moisture differences by soil profile depth from 17 April, 1984 to 27 Nov., 1984.

At every depth measured, the treated plots had significantly more moisture than the control plots. At the bottom of the soil surface horizon (25-30 cm) the volumetric soil moisture for the treated plots was 6.7%

while the control plots had 4.4% volumetric soil moisture. The next depth, which was the top of the control section for the soil, at 43-51 cm below the surface showed the moisture in the treated plots at 6.6% and in the control plots at 4.1%. The differences in soil moisture between the treated and control plots however diverged even more at the next two depths. At 91-99 cm, which is the middle of the control section, the soil moisture in the treated plots increased to 7.1% while the control plot moisture decreased to 2.8%. The difference was even greater at the bottom of the control section, 122-130 cm below the soil surface. At this depth, the treated plot soil moisture increased to 8.2% while the control plots were at 3.1%. Table 2.2 shows the average percent soil moisture by depth along with the standard deviation and standard error.

Table 2.2. Mean, Standard Deviation and Standard Error for soil moisture at each depth during the 1984 season.

	Mean	Standard Deviation	Standard Error
25-30cm Treated	6.66	1.92	0.38
25-30cm Control	4.37	2.06	0.40
43-51cm Treated	6.58	1.57	0.31
43-51cm Control	4.11	1.95	0.04
91-99cm Treated	7.14	1.56	0.31
91-99cm Control	2.77	0.52	0.10
122-130cm Treated	8.21	1.80	0.35
122-130cm Control	3.14	0.30	0.06

The differences in soil moisture also extended throughout the season. Volumetric soil moisture throughout the soil profile was constantly greater in the treated plots than the control (Figure 2.5).

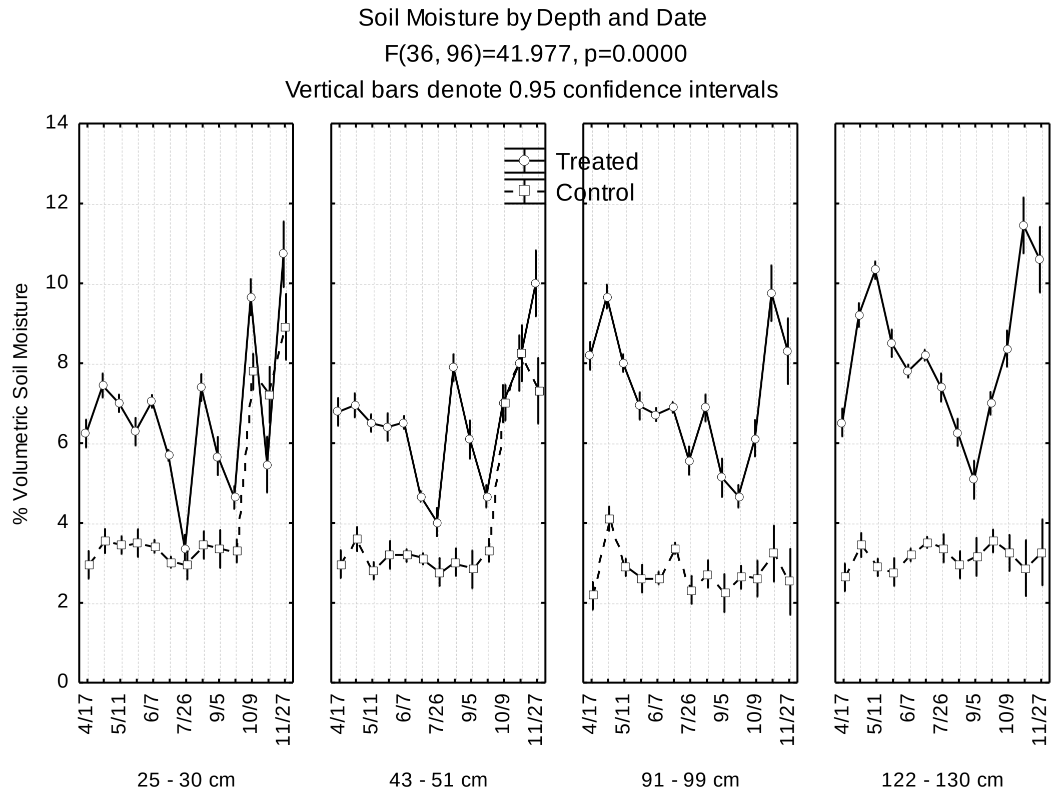

Figure 2.5. Average volumetric soil moisture content by depth through the 1984 season.

On 26 July 1984, the treated plots showed the minimum soil moisture content of 5.1%. The maximum soil moisture content, 9.9% occurred at the last measurement on 27 November. Both treated and control plots increased in soil moisture between 24 September and 9 October. The lowest soil moisture in the control plots was 2.7% which occurred on 17 April while the highest was 5.5% on 27 November.

DISCUSSION

The total annual production for this site in average growing condition years (1981 – 1983) as reported in the ecological site description is 1155 kg/ha of which 950 kg/ha should be herbaceous. The total annual production in favorable years (1984-1986) is reported in the site description as 1980 kg/ha of which about 1640 kg/ha should be herbaceous vegetation (USDA 2004). Two species, little bluestem (*Schizachyrium scoparium (Michx.) Nash*) and sand bluestem, (*Andropogon hallii Hack.*) are shown in the ecological site description to be the dominant plants making up 20% - 30% of the total composition by weight or 230 - 350 kg/ha. Although not tested statistically, these two species showed an increase from 79 kg/ha and 27% of the composition to 706 kg/ha and 50% of the composition of the herbaceous species in the treated plots. This was a nine fold increase in production and a doubling in percent composition by weight. In the control plots these two species ranged from a low of 80 kg/ha and 32% of the herbaceous production to a high of 244 kg/ha and 56% of the herbaceous production. This was a threefold increase in production and a 1.8 increase in percent composition by weight. These increases in measured herbaceous annual production following treatment are consistent with research studies conducted in one-seed juniper systems showing high levels of interannual variability associated with weather (Jameson 1970, Pieper 1983).

The increase in available soil moisture suggests that soil moisture for plant growth may be the major reason for the vegetation differences measured. This agrees with Johnsen (1962) and Breshears et al. (1997a) who both found direct competition for soil moisture between one-seed juniper and herbaceous vegetation, and with Jameson (1970) who showed increased herbaceous basal area after removal of one-seed juniper root activity. The soil moisture analysis showed significantly (α=0.05) more available soil moisture in the treated plots than in the control plots. The difference of more than 2% soil moisture by volume in the profile throughout the season is highly important for plant growth. This difference is reflected throughout the soil profile at all four depths measured and throughout the season from mid April to late November. The two lower depths (middle and bottom of the moisture control section) showed a much greater difference in soil moisture than the two upper depths (soil surface and top of the moisture control section). This suggests that one-seed juniper and sand sagebrush were extracting moisture from all depths and that the herbaceous vegetation was primarily extracting moisture from the upper depths. It also suggests that even though the herbaceous vegetation produces more annual growth than the woody vegetation, the one-seed juniper and sand sagebrush extracted more soil moisture than the herbaceous plants. This agrees with the finding of Breshears et al. (1997a), however, they offer an alternative explanation that there was greater soil surface evaporation in the one-seed juniper dominated community (Breshears et al. 1997b). Regardless of the mechanism, the available soil moisture for plant growth was higher in the treated plots.

It is also notable that the observed increase in both total and herbaceous production in the treated plots occurred in conjunction with the increase in soil moisture. Using the permanent wilting point of 1.5 MPa, the volumetric water percentage was 2.9%. Although one-seed juniper can extract soil moisture at as low as -2.4 MPa (Johnsen 1962), 1.5 MPa is a good reference for comparing the available water in the soil profile. The volumetric moisture content in the control plots was close to the 1.5 MPa throughout most of the season. It was consistently higher only after the killing frost at the end of September. The available soil moisture in the treated profile was consistently above the 1.5 MPa moisture content and for most of the season had twice as much moisture as the control plots. This difference in available soil moisture occurred while apparent total production was much higher in the treated plots than the control plots.

Due to the large increase in soil moisture at the lower two depths, there is a suggestion of the possibility of increased ground water recharge. Some of this increase in the treated plots, especially in the earliest part of the season and at the end of the season, could be due to the decrease in transpiration of the herbaceous species allowing deep penetration of the soil moisture. In the control plots transpiration continues as long as the one-seed juniper remained active. Johnsen (1962) showed that while air temperature did not matter, one-seed juniper roots actively grew any time of the year when soil temperature was above 10 °C and there as available soil moisture. The actual precipitation during the study period was above average and the 1984 precipitation was 8.88 cm above average. Studies needed to determine increased groundwater recharge should be long term and should include additional hydrologic variables. Many studies including Wilcox et al. (1996a) suggest that there is little or no additional ground water recharge from woody plant removal in systems of less than 50 cm precipitation. It should be pointed out however, that their study was conducted on sites with much finer textured soils.

SUMMARY and CONCLUSIONS

While there were several years of vegetation data collected, there was no replication within the yearly data collection. However, the apparent differences observed in total and herbaceous annual production, ground cover differences and species composition changes occurred in a relatively short time frame and were obvious to the casual observer.

The results of this study clearly showed a significant increase in available soil water in the upper 125 cm, at least in the short term following one-seed juniper mortality. This increase occurred when no apparent change in runoff was observed and in spite of a dramatic increase in total annual production on the treated sites. These results suggest the need for additional work to relate the differences observed and measured here to long term changes in soil erosion, plant cover and production and groundwater recharge. Wind erosion on this ecological site was evident to the point that much of the area was active sand dunes with one-seed juniper.

Livestock production values and management options were greatly enhanced by the change in vegetation composition and production. Wildlife habitat values for most species of concern were also enhanced. Both food and cover values changed radically and different habitat elements were developed as a result of the changes in structure and composition of the vegetation. With more available water in the soil profile and the possibility of increased ground water recharge, the water cycle and resulting watershed values were enhanced by the treatment.

LITERATURE CITED

Allen, C. D. and D. D. Breshears 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings, National Academy of Sciences of the United States of America. 95(25): 14839-14842.

Arnold, J. F. 1964. Zonation of understory vegetation around a juniper tree. Journal of Range Management. 47: 41-42.

Arnold, J. F., D. A. Jameson and E. H. Reid. 1964. The pinyon-juniper type of Arizona; effects of grazing, fire and tree control. U. S. Dept. Agr. Prod. Res. Rep. No. 84. Washington, D. C.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2000. Understory dynamics in cut and uncut western juniper woodlands. Journal of Range Management 53:119-126.

Breshears, D. D., O. B. Myers, S. R. Johnson, C. W. Meyer and S. N. Martens. 1997a. Differential use of spatially heterogeneous soil moisture by two semiarid woody species: *Pinus edulis* and *Juniperius monosperma*. Journal of Ecology 85(3): 289–299.

Breshears, D. D., P. M. Rich, F. J. Barnes and K. Campbell. 1997b. Overstory-imposed heterogeneity in solar radiation and soil moisture in a semiarid woodland. Ecological Applications 7(4): 1201–1215.

Clary, W. P. and D. A. Jameson. 1981. Herbage production following tree and shrub removal in the Pinyon-Juniper type in Arizona. Journal of Range Management. 34: 109-113.

Cook, C. W. and C. D. Bonham. 1977. Techniques for vegetation measurements and analysis for a pre-and post-mining inventory. Colo. State Univ. Range Sci. Dept. Sci. Ser. No. 28. Ft. Collins, CO.

Fowler, J. M., B. E. Peacock and M. J. Schaber. 1985. Pinyon-juniper woodland type in New Mexico: Asset or liability. New Mex. State Univ. Agr. Exp. Sta. Bull. 718. Las Cruces, NM.

Foxx, T. S. and G. D. Tierney. 1987. Rooting patterns in the pinyon-juniper woodland. In: Everett, Richard L., compiler. Proceedings--pinyon-juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: pp. 69-79.

Gardner, W. H. 1986. Water content. Klute, A.(ed.). Methods of soil analysis. Part I. pp. 493-544. Madison, WI, ASA and SSSA. Agron. Monogr.

Gifford, G. F. 1973. Runoff and sediment yields from runoff plots on chained pinyon-juniper plots in Utah. Journal of Range Management. 26: 440-443.

Gottfried, G. J. 1999. Pinyon-juniper woodlands in the southwestern United States. In: Ffolliott, Peter F.; Ortega-Rubio, Alfredo, eds. Ecology and management of forests, woodlands, and shrublands in the dryland regions of the United States and Mexico: perspectives for the 21st century. Co-edition No. 1. Tucson, AZ: The University of Arizona; La Paz, Mexico: Centro de Investigaciones Biologicas del Noroeste, SC; Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: pp. 53-67.

Herman, F. R. 1956. Growth and phenological observations of Arizona junipers. Ecology. 37: 193-195.

Jameson, D. A. 1962. Effects of burning on a galleta-black grama range invaded by juniper. Ecology. 43(4): 760-763.

Jameson, D. A. 1967. The relationship of tree overstory and herbaceous understory vegetation. Journal of Range Management. 20: 247-249.

Jameson, D. A. 1970. Juniper root competition reduces basal area of blue grama. Journal of Range Management. 23: 217-218.

Johnsen, T. N., Jr. 1962. One-seeded juniper invasion of northern Arizona grasslands. Ecological Monographs. 32(3): 187-207.

Lanner, R. M. and T. R. Van Devender. 1998. The recent history of pinyon pines in the American Southwest. In: Richardson, David M., ed. Ecology and biogeography of Pinus. Cambridge, United Kingdom: The Press Syndicate of the University of Cambridge: pp. 171-182.

Miller, R. F. and P. E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. Bioscience. 44(7): 465-474.

Mueggler, W. F. 1976. Ecological role of fire in western woodland and range ecosystems. In: Use of prescribed burning in western woodland and range ecosystems: Proceedings of the symposium; 1976 March 18-19; Logan, UT. Logan, UT: Utah State University, Utah Agricultural Experiment Station: pp. 1-9.

Newman, B. D., A. R. Campbell and B. P. Wilcox. 1997. Tracer based studies of soil water movement in semi-arid forests of New Mexico. Journal of Hydrology. 196: 251-270.

Pieper, R. D. 1983. Overstory-understory relationships: Pinyon-juniper and juniper woodlands. In: Bartlett, E. T. and D. R. Betters (Eds.) Overstory-understory relationships in western forests. West. Reg. Res. Pub. No. 1. Colo. State Univ. Agr. Exp. Sta. Ft. Collins, CO.

Pierson, F. B., J. D. Bates, T. J. Svejcar, and S. P. Hardegree. 2007. Runoff and erosion after cutting western juniper. Rangeland Ecology and Management 60: 285-292.

Reid, K. D., B. P. Wilcox, D. D. Breshears and L. MacDonald. 1999. Runoff and erosion in a Piñon-Juniper woodland: Influences of vegetation patches. Soil Science Society of America Journal. 63: 1869–1879.

Romme, W. H., C. D. Allen, J. D. Bailey, W. L. Baker, B. T. Bestelmeyer, P. M Brown, K. S. Eisenhart, M. L. Floyd, D. W. Huffman, B. F. Jacobs, R. F. Miller, E. H. Muldavin, T. W. Swetnam, R. J. Tausch and P. J. Weisberg. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

StatSoft, Inc. (2005). STATISTICA (data analysis software system), version 7.1. www.statsoft.com.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2003. National range and pasture handbook, revision 1. Washington, D.C.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2004. Ecological site descriptions. Field office technical guide, section II. Albuquerque, NM.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2010. *The* PLANTS *Database* (http://plants.usda.gov, 2 January 2010). National Plant Data Center, Baton Rouge, LA 70874-4490 USA.

U.S. Department of Agriculture. Soil Conservation Service. 1988. Soil survey Socorro area, New Mexico. Washington, D.C.

Weltzin, J. F. and G. R. McPherson. 1997. Spatial and temporal soil moisture resource partitioning by trees and grasses in a temperate savanna, Arizona, USA. Oecologia 112: 156-164.

West, N. E., K. H. Rea and R. J. Tausch. 1975. Basic synecological relationships in pinyon-juniper woodlands. In: Gifford, G. F. and F. E. Busby (Eds.). The pinyon-juniper ecosystem. Utah State Univ. Agr. Exp. Sta. Logan, UT.

Western Regional Climate Center. 2009. New Mexico Climate Summaries, Gran Quivira National Monument, NM. http://www.wrcc.dri.edu/summary/climsmnm.html.

White, L. D. 1965. The effects of a wildfire on a desert grassland community. [thesis]. Tucson, AZ: University of Arizona. 107p.

Wilcox, B. P., C. D. Allen, B. D. Newman, K. D. Reid, D. Brandes, J. Pitlick and D. W. Davenport. 1996a. Runoff and erosion on the Pajarito Plateau: observations from the field. pp. 433-439. In: New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Regions, 1996. New Mexico Geological Society. Albuquerque, N.M.

Wilcox, B. P., J. Pitlick, C. D. Allen and D. W. Davenport. 1996b. Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. pp. 61-77. In: M.G. Anderson and S.M. Brooks (eds.) Advances in Hillslope Processes, Vol. 1. John Wiley and Sons, New York.

Wilm, H. G., D. F. Costello and G. E. Klipple. 1944. Estimating forage yield by the double-sampling method. Journal of the American Society of Agronomy. 36: 194-203.

Woodbury, A. M. 1947. Distribution of pigmy conifers in Utah and northeastern Arizona. Ecology. 28(2): 113-126.

CHAPTER 3: LONG TERM RESPONSE OF SOIL MOISTURE AND VEGETATION TO THE REDUCTION OF ONE-SEED JUNIPER CANOPY COVER

ABSTRACT

The response of soil moisture and several vegetation and soil attributes were measured following the application of herbicide on one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg). Treatment was applied in 1985 and soil moisture, annual herbaceous production and vegetative cover were measured for five years. In 2003 these measurements were repeated along with additional vegetation attributes and soil aggregate stability. A repeated measures analysis of variance was used to test the effects of the original treatment including the repeated measurements in 2003. This analysis indicated a significant treatment effect on most variables. It also indicated that the effects of the 1985 treatment were still significant in 2003. The proportion of time that moisture was available in the soil profile was much greater in the treated plots. Herbaceous production and various ground cover measurements were also significantly greater in the treated plots. These results may explain the significant differences measured in soil aggregate stability. The increase in soil aggregate stability in the treated sites suggests that the vegetation changes resulting from the treatment improved the organic matter inputs into the system and the resulting nutrient cycle. Improved soil aggregate stability has been associated with improved water infiltration, reduced evaporation and erosion, and a resultant feedback for improved herbaceous production and soil aggregate stability. These changes have important implications for the development of state-and-transition models and for management systems aimed at maintaining or improving the ecological and economic sustainability of rangelands.

INTRODUCTION

Juniper dominated plant community associations currently cover approximately 40 million hectares in the western United States (West et al. 1975, Fowler et al. 1985, Romme et al. 2009). In parts of the southwest, juniper species including one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg) have encroached into adjacent grasslands or increased in stand density since European settlement (Woodbury 1947, Jameson 1962, Mueggler 1976, Lanner and Van Devender 1998, Gottfried 1999). Explanations for this encroachment include a decrease in fine fuels from grazing, active fire suppression, reduction of grass competitiveness from overgrazing, spread of seed by livestock and wildlife, and climate change (Johnsen 1962, White 1965, Miller and Wigand 1994, Allen and Breshears 1998, Lanner and Van Devender 1998). Increased juniper cover is generally associated with reduced herbaceous cover and production (Jameson 1962, Arnold 1964, Arnold et al. 1964, Jameson 1967, Jameson 1970, Clary and Jameson 1981, and Pieper 1983), and increased runoff and soil erosion (Arnold 1964, Jameson 1970, Wilcox et al. 1996a, Wilcox et al. 1996b, Petersen and Stringham 2008). It has been argued that these changes should lead to reductions in soil water availability (Arnold 1964, Jameson 1970, Breshears, et al. 1997). If true, juniper mortality should result in increased herbaceous production and cover, increased soil water availability, and decreases in associated soil erosion. There have been relatively few studies exploring the effects of juniper mortality on soil water availability, and most have focused on medium to fine textured soils and included only near surface soil moisture measurements (Gifford 1973, Wilcox et al. 1996a, Wilcox et al. 1996b, Reid et al. 1999). Exceptions include Newman et al. (1997) and Weltzin and McPherson (1997) which quantified water use throughout the soil profile of untreated stands of one-seed juniper and Emery oak (*Quercus emoryi* Torr.), respectively.

These studies were conducted on fine textured soils and with a winter dominated precipitation pattern. There have been few studies on large spatial or temporal scales assessing the effects of juniper treatment on vegetation and soil changes on coarse textured soils with a summer, monsoon dominated precipitation pattern.

The first objective of this study was to test the hypothesis that vegetation production, cover and soil moisture would increase after juniper treatment on coarse textured soils. The second objective was to test the hypothesis that these changes would be sustained over time.

METHODS

Study Area

The study was conducted on four private ranches located in central New Mexico. The four ranches are all within a 46 km diameter area. The area has a flat to rolling dune topography with elevations ranging from 1750 m to 2000 m. The soils in the area are very similar and formed from aeolian sand deposits derived from mixed sources. The dominant characteristic of these deep soils is the coarse texture. The soil moisture regime is aridic-ustic and the soils at all locations are classified as mixed, mesic, Ustic Torripsamments (USDA 1970, USDA 1988). This classification was confirmed with a soil pit and several auger holes at each of the study sites. The soil is deep and excessively drained and the surface layer is a brown fine sand about 28 cm thick. The underlying material is strong brown fine sand in excess of 155 cm deep (USDA 1970, USDA 1988). The soil map unit components are all correlated to the Deep Sand Savannah Ecological Site in MLRA G070C, Central New Mexico Highlands, site number 070CY123NM (USDA 2004).

The Gran Quivira Unit of the Salinas National Monument is located on one of the ranches in the study. Summarizing Spanish journals and records, Horgan (1954) described the mission of Gran Quivira and the area to the east of the mission as golden-grassed plains where no natural barriers divide them from the country of hunters (Comanche). The territorial land survey (McLeullough, 1882) describes several sections west and north of Gran Quivira as sandy with good grass cover and scattered piñon and juniper. The same survey describes several sections north and east of Gran Quivira as having good grass cover with no timber on the sandy soils. Adolph Bandelier (1884) described the area as seen from the mission ruin looking southeast as treeless but very grassy. He also noted the grass was three feet high in places.

The climate of the study area is characterized as semi-arid continental. The average annual precipitation at the Gran Quivira National Monument ranges from 33-40 cm depending on the period of record. The thirty year average (1951-1980) immediately preceding the study was 36.2 cm (Western Regional Climate Center, 2009). Precipitation during the first four years of the study (1985-1988) was above average with 49.3 cm, 60.0 cm, 52.9 cm and 45.1 cm, respectively. In 1989 the precipitation was below average at 25.0 cm. The period 1989 – 2003 shows the variability common to the area with 2001 – 2003 being very dry (Figure 3.1). Seventy five percent of the precipitation falls from April to October, primarily in the form of high intensity thunderstorms. Temperatures are characterized by distinct seasonal changes and large annual and diurnal fluctuations. The average annual air temperature is about 10^{o}C, with extremes of -34^{o}C in the winter to 40^{o}C in the summer. The average frost free season is 130–160 days. The last killing frost occurs in early May and the first killing frost is in early October (Figure 3.2).

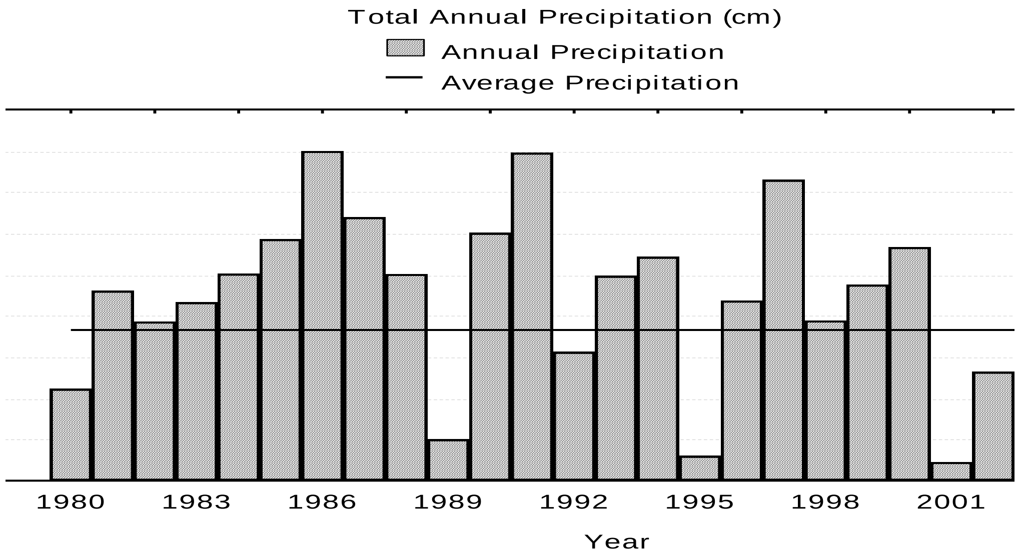

Figure 3.1. Actual and average precipitation for the study site from 1980–2007.

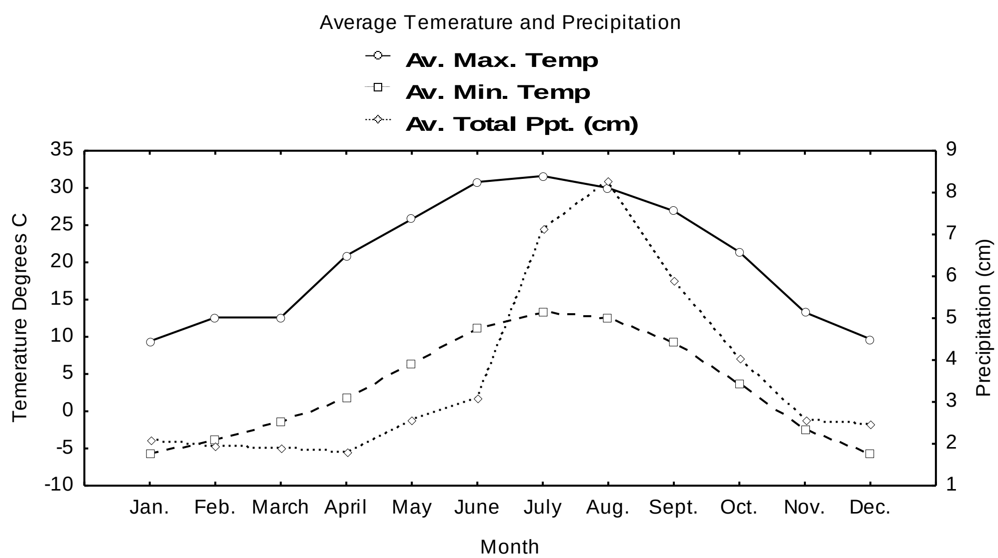

Figure 3.2. Average annual high and low temperature and average total precipitation.

Vegetation measurements prior to the treatment in 1985, indicated the vegetation in the area was dominated by one-seed juniper greater than 4 m tall with canopy cover in excess of 25%. Total annual production of all species was less than 150 kg/ha (Shaver 2010).

Procedures

Long Term Effects Study

The herbicide tebuthiuron was aerially applied to approximately 1300 ha on four ranches in the fall of 1985. Application rates ranged from 0.66 kg/ha to 0.99 kg/ha. The objective of this study was to determine the effects of juniper mortality, not tebuthiuron effectiveness, thus all application rates were grouped for analysis. The design of the study was a complete randomized block for ranches 1 and 2 and as generalized blocks – unbalanced for ranches 3 and 4. Plot locations were selected and established at the time of herbicide application. Ranches 3 and 4 were treated with more than one treatment rate, and as a result, data were collected on two randomly located treatment plots. One location in the control area at each of the four ranches was also randomly selected for data collection. Ten permanent transect locations were established (Figure 3.3). These transects were oriented magnetic north and data collection started at the south end. The soil moisture locations were selected randomly within 20 feet of the beginning marker for the vegetation transects.

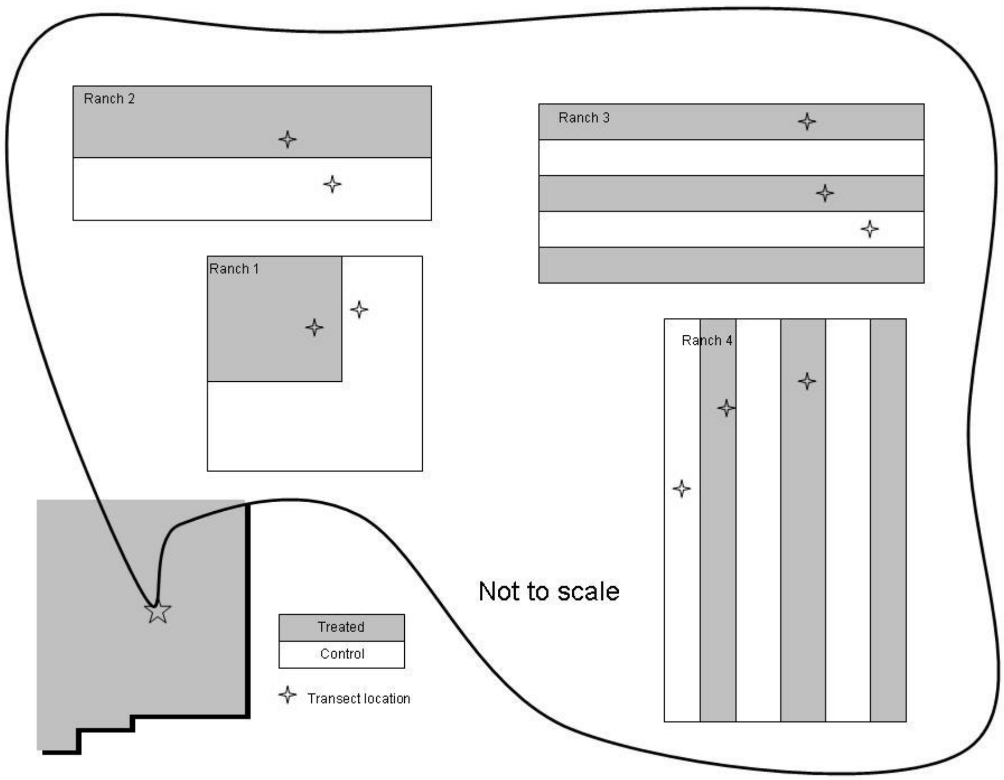

Figure 3.3. Plot layout for long term effects study showing the relative location of ranches and treated and control plots.

Vegetation Measurements. Vegetation production and composition by weight were measured in the autumn of each year 1985 – 1989, after killing frost, using double sampling (Wilm et al. 1944, Cook and Bonham 1977, USDA 2003). Ten 0.89 m^2 (9.6 ft^2) frames located along a permanent 100 pace transect line at each of the 10 locations were sampled. Two locations along each transect had grazing exclusion cages to allow for reconstruction of grazed production. These cages were moved each year after sampling. Two of the ten plots on each line were expanded to 40.5 m^2 (0.01 ac.) to estimate woody plant production using weight units and weight size relationships tables developed by USDA-NRCS (USDA 2003).

Point cover data were collected at the same time as the double sampling. A 100 pace transect was sampled along the production transect. At each pace bare ground, herbaceous foliar cover, litter and live juniper canopy cover at a point on a frequency frame were recorded (Allison 1982). Percent bare ground, herbaceous foliar cover and litter together totaled 100%.

Soil Measurements. Soil moisture was measured using a Troxler model 503, neutron probe (Gardner 1986) at four depth intervals within the soil profile. The first interval was centered at 17.8 cm below the surface. The resulting sphere of measurement represented the bottom of the soil surface horizon. The next three depths were centered at the top, middle and bottom of the moisture control section as determined for each soil at each of the locations. Data were correlated to volumetric soil moisture. Soil moisture samples collected with a soil auger at each depth were weighed, oven dried at 105^{o}C for 24 hours and weighed again (Gardner 1986). Bulk density was determined one time at each of the depths in each plot using a cylindrical core sampler (Gardner 1986). Volumetric soil moisture was determined by multiplying the gravimetric value by the bulk density of the soil at each of the measured depths. This procedure was followed until enough moisture samples were taken to yield a correlation to volumetric soil moisture of at least an r^2=0.80. After that time, the neutron probe readings alone were used to determine the soil moisture content.

Expansion Study

In the fall of 2003, at each of the original locations, vegetation production and point cover were repeated using the same methodology as 1985-1989. In addition to these data, ten new transects each on the treated and untreated locations at each ranch were established. Ranch 3 and 4 had an additional three transects each in the treated areas (Table 3.1).

Table 3.1. Summary of the number of plots in the original 1985 – 1989 study and in the expansion study of 2003

	Original Plots		Expansion Plots	
Ranch	Treated	Control	Treated	Control
1	1	1	10	10
2	1	1	10	10
3	2	1	13	10
4	2	1	13	10

Three of the 2003 transects were established at each of the original treatment and control locations. The remainder of the new transects were randomly selected throughout the extent of the treated and control areas on each ranch. The purposes of this new data and additional transect locations were to increase the sample numbers and precision of the analysis and to test the representativeness of the original transect locations. The following data were collected at each of the new transect locations in 2003.

Vegetation and ground cover. A transect line 50 m long was established with a point reading at 0.5 m intervals starting at 0.5 m. All foliar cover was counted regardless of height and recorded as dead or alive. Species were recorded only once at each point. A basal hit of a plant species automatically warranted a canopy hit of that species. Woody litter (>5mm), herbaceous litter, embedded litter, duff, lichen, moss, bedrock, rock and bare soil were also recorded (Herrick et al. 2005).

Vegetation Gaps. Gaps in both plant canopy and plant bases >20 cm were recorded along the point transect. The line was read from the zero end of the tape, reading left to right. Woody litter (not attached to plant base) that was >5 mm diameter, 15 cm long and in contact with the soil or fine litter, was considered a stop in basal gap. Woody litter that was above the soil surface was considered a stop in canopy gap. Gap sizes were analyzed by the following groupings: 25 – 50 cm, 51 – 100 cm, 101 – 200 cm, and >200 cm (Herrick et al. 2005).

Soil stability. Nine surface samples and nine subsurface samples, 6 – 8 mm in diameter, were collected along the point transect at a randomly predetermined distance perpendicular to the left side of the line. Samples were taken at 5 m intervals for a total of nine samples per line. Sample soil peds were placed in the sample boxes and allowed to dry if wet. Deionized water was used in the test and each sample was placed in the water and observed for five minutes. After five minutes, each sample was sieved in and out of the water five times. The soil peds were then rated on a scale of 1 – 6 (from least to most stable) based on the percent of ped remaining before and after sieving (Herrick et al. 2001).

Statistical analysis. Statistica 7.1 software (StatSoft, Inc. 2005) was used for all analyses. A repeated measures analysis of variance (ANOVA) was performed on the long term effects study vegetation data. Dependent variables were the data elements taken on each of the sample dates and the categorical predictors were treatment and ranch (block). Due to the unbalanced nature of the design, the effective hypothesis approach (Hocking et al. 1980, Hocking 1985) was employed. Based on Levene's test for normality (Levene 1960) all data except soil moisture were evenly distributed. Therefore, Friedman ANOVA (Friedman 1937), a nonparametric alternative to one-way repeated measures ANOVA, was used to compare the percent of time soil moisture was above 1.5 MPa in all or in any part of the moisture control section. Simple linear regression analysis was conducted to determine the relationship between the data elements.

The expansion study data were analyzed using a factorial ANOVA with the various values obtained from the data elements being the dependent variables and the ranch (block) and treatment being the categorical predictors. Simple linear regression analysis was conducted to determine the relationships between the data elements. Significance level was alpha = 0.05.

RESULTS

Long Term Effect Study

Vegetation Production. The repeated measures ANOVA for the 1985 – 1989 data showed a significant treatment by years interaction ($F(4,8)=7.0051$, $p=0.01$). The mode of action of the herbicide tebuthiuron was likely a key factor influencing this interaction through the delayed response in annual production. It was not until the end of the first growing season after application (1986) before any herbicide effects could be observed on the juniper trees. It was the end of the second season (1987) or the middle of the third growing season (1988) before the juniper trees exhibited full effects of the herbicide. A Dunnett's pair wise comparison (Dunnett 1980, Winer et al. 1991, Westfall et al. 1999) was used to more fully assess the differences shown in the repeated measures ANOVA and the effects of treatment by year interaction. Dunnett's pair wise comparison determined the difference between a control mean and the treatment means. The test showed significant differences between annual herbaceous production in the control plots and treated plots through time. Using the 1985 control plots as the starting pair, 1987, 1988 and 1989 treated plots were significantly different from the 1985 control plots ($p=0.0048$, $p=0.0003$ and $p=0.0008$, respectively).

When the 2003 annual herbaceous production data were added to the analysis, the interaction between treatment and years was still significant ($F(5,10)=5.7229$, $p=0.0095$). Dunnett's pair wise comparison showed the 2003 treated plots were significantly different from the 1985 control plots ($p=0.047$). This test also showed no difference in the 1985 and 2003 control plots (p=0.85). Figure 3.4 shows the effect of time on the response of the herbaceous production to the treatment of the juniper.

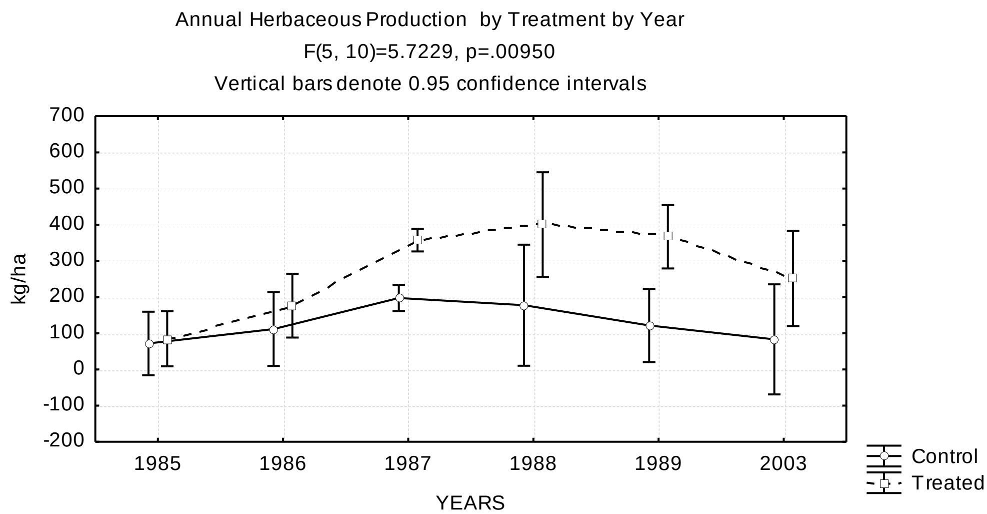

Figure 3.4. Annual herbaceous production by year by treatment, 1985-1989 and 2003.

Table 3.2 lists the p-values for the Dunnett's pair wise comparison for annual herbaceous production.

Table 3.2. p-values for Dunnett's pair wise comparison using 1985 control plots as comparison to illustrate effects of treatment and year interaction.

	Control					Treated					
	'86	'87	'88	'89	'03	'85	'86	'87	'88	'89	'03
1985-1989	0.79	0.33	0.42	0.74	N/A	0.88	0.42	0.005	0.0003	0.0008	N/A
1985-1989 & 2003	0.69	0.13	0.22	0.62	0.85	0.85	0.30	0.002	0.0001	0.0004	0.02

Bare Ground. Year by treatment interaction was significant (F(4,8)=3.5954,p=0.049). This interaction clearly shows the effects of time on the amount of bare ground present due to juniper treatment. When the 2003 data were added to the analysis, the years by treatment interaction was again significant (F(5,10)=3.4029,p=0.047). This interaction shows the effects of time on the amount of bare ground present due to juniper treatment. The decrease in percent bare ground occurred somewhat sooner than the increase in annual herbaceous production, but otherwise the pattern of the results match that of the annual herbaceous production (Figure 3.5). Dunnett's pair wise comparisons showed significant differences in 1986 (p=0.0046), 1987 (p=0.0006), 1988 (p=0.0011), 1989 (p=0.0016) and 2003 (p=0.0004).

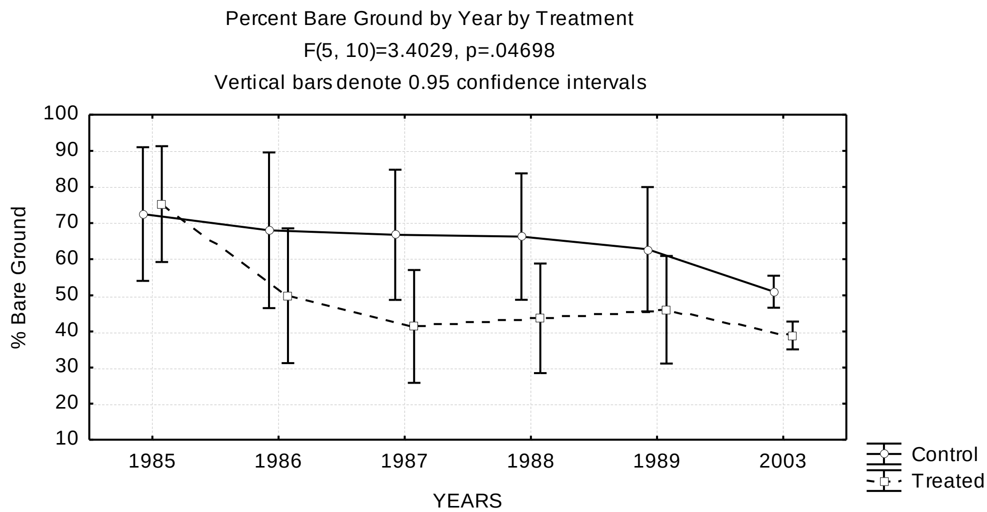

Figure 3.5. Percent bare ground by year by treatment for 1985–1989 & 2003.

Litter Cover. The amount of litter cover was significantly different (F(1,2)=41.143 p=0.023) between the treated and control plots. The treated plots averaged 36.9% ± 1.16 litter cover and the control plots averaged 25.5% ± 1.34 litter cover. When the 2003 data were added to the analysis, the difference between treated and control plots was still significant (F(1,2)=52.261, p=0.019) with the treated plots having 39.06% ± 0.97 and the control plots having 28.33% ± 1.12 average litter cover.

Juniper Canopy. The analysis of the 1985 – 1989 data showed significant treatment by year interaction (F(4,8)=23.992, p=00016) demonstrating the effects of time on the herbicide activity. When the 2003 data were added to the analysis, that interaction was again significant (F(5,10)=5.2309, p=0.013) . It is interesting to note that the decrease in juniper canopy cover over time matched very closely with the increase in annual herbaceous production, and the decrease in percent of bare ground. Dunnett's pair wise comparison showed significant treatment effects for 1988 and 1989 (p=0.024 and p=0.023 respectively), but no difference for 2003 (p=0.966) (Figure 3.6), indicating that overall the juniper canopy cover had recovered from the treatment. The recovery in juniper canopy cover is likely due to the establishment of one-seed juniper seedlings since herbicide treatment in 1985.

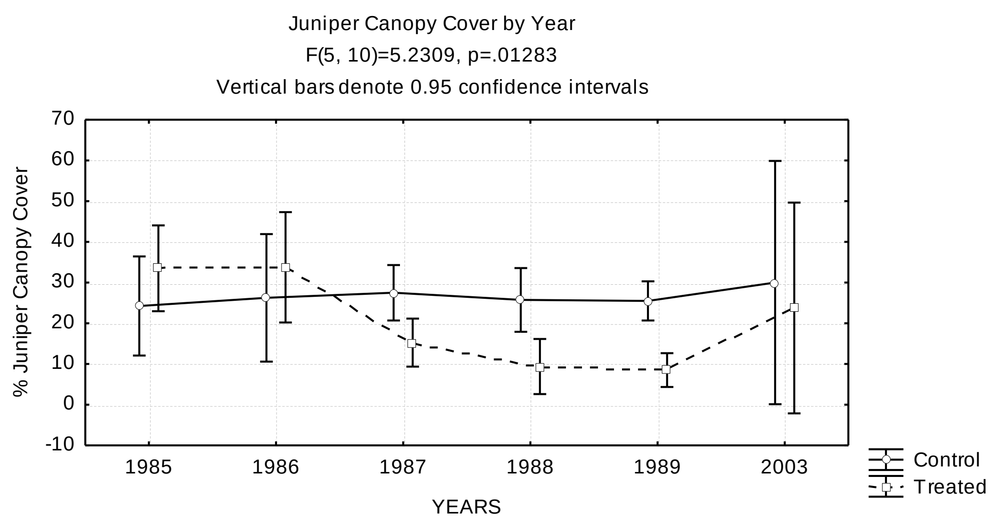

Figure 3.6. One-seed juniper canopy cover for treated and control plots by year for 1985–1989 & 2003.

The analysis with the 2003 data also showed a significant ranch by treatment interaction (F(3,2)=22.406, p=0.043) (Figure 3.7). Dunnett's pair wise comparison showed no effect on juniper canopy in the treated plots on ranches 1, 2, and 4 but on ranch 3 the differences were significant (p=0.013).

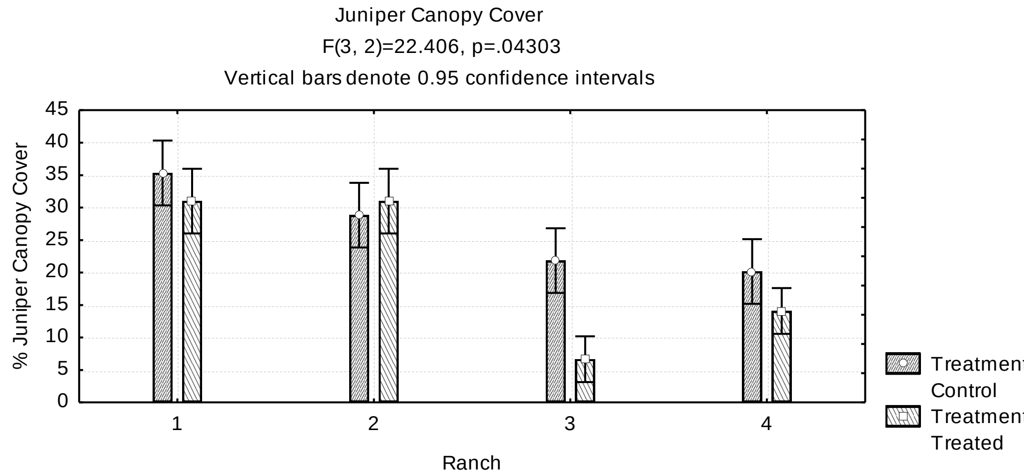

Figure 3.7. Average one-seed juniper canopy cover by ranch with 1985 – 1989 and 2003 data, showing ranch by treatment interaction.

These differences indicate the effects of the juniper treatment may be ending and juniper canopy was increasing both in absolute terms and relative to the control on ranches 1, 2 and 4.

Soil Moisture. There were significant differences between the treated and control plots in the amount of time soil moisture was above 1.5 MPa in all parts of the control section (p=0.0016). Significant differences also appeared between the treated and control plots in the amount of time soil moisture was above 1.5 MPa in any part of the control section (p=0.0003) (Table 3.3).

Table 3.3. Proportion of time soil moisture was above 1.5 MPa in the all or part of the moisture control section of the soil profile.

	Any Part of Moisture Control Section			All Parts of Moisture Control Section		
	Mean	Std Dev	p	Mean	Std Dev	p
Treated	0.99	0.03	0.003	0.91	0.29	0.0016
Control	0.79	0.23		0.60	0.13	

The treatment effect was especially noticeable during the dry year of 1989 when only 25 cm or 69% of normal precipitation was available (Figure 3.8).

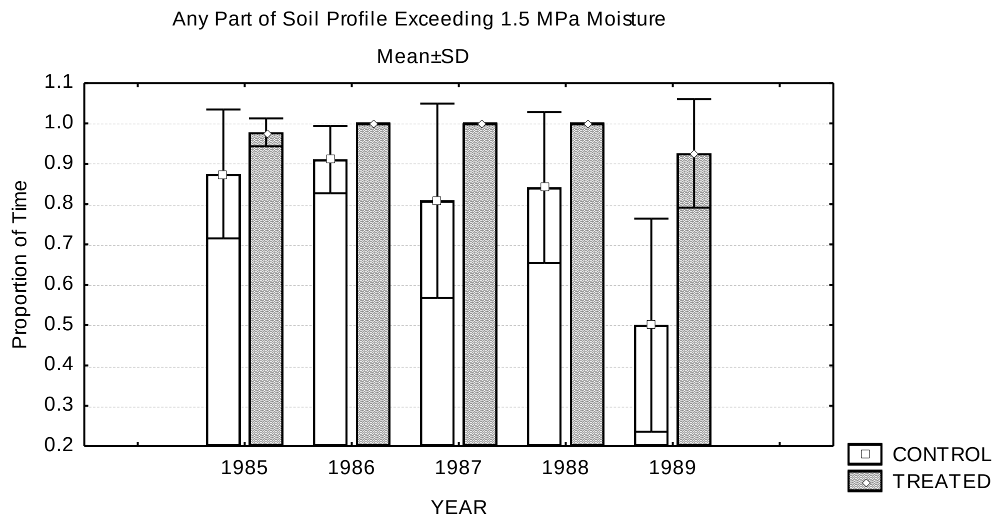

Figure 3.8. Proportion of time soil moisture in the profile exceeds 1.5 MPa in any part. Note: Treated plots during 1986 – 1988 had some part of the profile above 1.5 MPa 100% of the time.

Relationships. Table 3.4 summarizes the results of simple linear regression between the data elements for the 1985 – 1989 data.

Table 3.4. Results of simple linear regression of the data collected in the long term study, 1985 - 1989.

	1985 - 1989		
Data Elements	r	r^2	p
Annual Herbaceous Production : Bare Ground	-0.88	0.78	0.0007
Annual Herbaceous Production : Litter Cover	0.89	0.79	0.0006
Annual Herbaceous Production : Juniper Canopy	-0.93	0.86	0.0001
Bare Ground ; Litter Cover	-0.95	0.90	0.00002
Bare Ground : Juniper Canopy	0.94	0.89	0.0001
Litter Cover : Juniper Canopy	-0.76	0.59	0.0099

When 2003 data were included in the regression the negative correlation between annual herbaceous production and juniper cover remained strong (r^2=0.8535, p=0.00002, y=34.5111-0.0596*x). The negative correlation between annual herbaceous production and bare ground weakened (r^2=0.5884, p=0.0036) as did the positive correlation between annual herbaceous production and litter cover (r^2=0.5138, p=0.0087), however, both remained highly significant.

Expansion Study

Bare ground and total ground cover showed significant treatment differences with no ranch interactions. Table 3.5 summarizes those effects.

Table 3.5. Treatment differences for per cent bare ground and total ground cover.

	Bare Ground			Total Ground Cover		
	Average	Std. Dev	p-value	Average	Std. Dev	p-value
Control	38.53	1.85	0.0032	52.3	1.9	0.0065
Treated	30.76	1.74		59.61	1.79	

Juniper Foliar Cover. Analysis showed a significant interaction between ranch and treatment (F(3,78)=4.5539, p=0.00542). The differences in the ranches affected the overall effects of the treatment. Post hoc analysis using Fisher's LSD (Winer et al. 1991) to test differences in the means for the interaction showed no significant differences in the means for ranches 1 and 2 (p=0.1061 and p=0.4288, respectively) and significant differences between the means for ranches 3 and 4 (p=0.0001 and p=0.0000, respectively) (Figure 3.9).

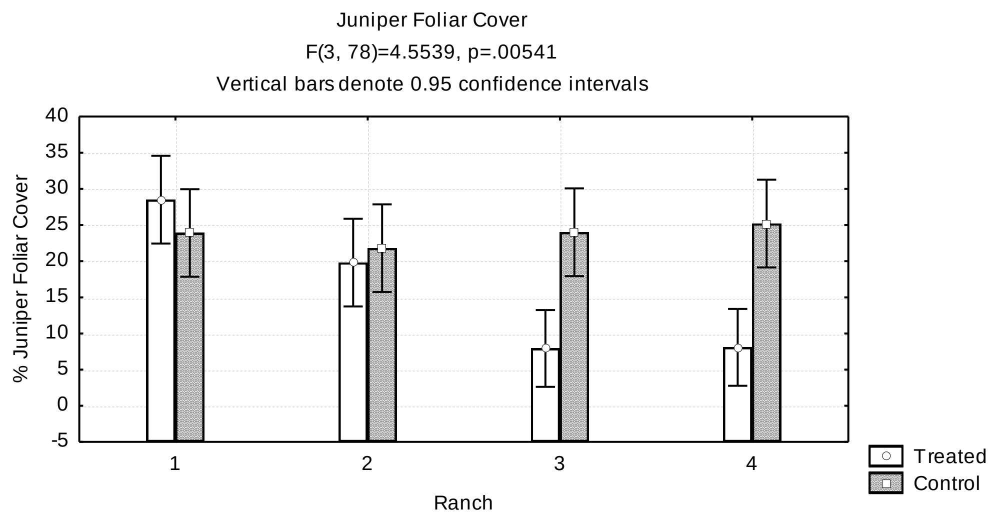

Figure 3.9. Percent juniper foliar cover by ranch by treatment.

Soil Stability. The average stability rating for all surfaces, both protected and unprotected, showed significant (F(3,78)=3.8953, p=0.0119) ranch by treatment interaction. Post hoc analysis using Fisher's LSD showed that all ranches had significant difference in treatment means (p<0.0001 for all pairs) (Figure 3.10).

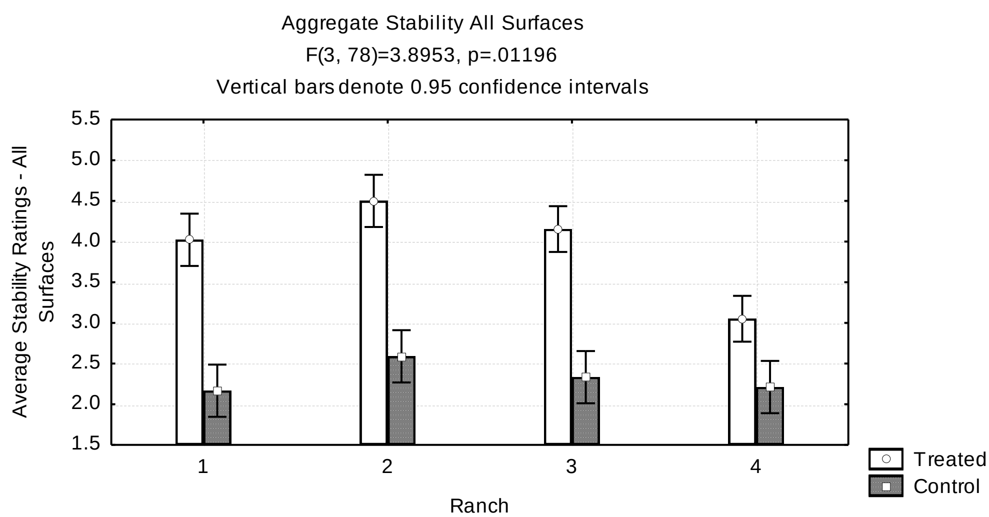

Figure 3.10. Average surface soil stability by ranch showing treatment interaction effects.

The average stability ratings for the protected subsurface samples showed a significant treatment difference ($F(1,77)=26.737$, $p<0.0001$). The treated protected subsurface samples had an average stability rating of 2.93 ± 0.13 and the control plots averaged 1.95 ± 0.14.

The average stability ratings for unprotected surface samples showed a significant ($F(3,77)=5.2134$, $p=0.0025$) ranch by treatment interaction. Post hoc analysis using Fisher's LSD showed significant differences for ranches 1, 2 and 3 as $p<0.001$ and for ranch 4 as $p=0.05$ (Figure 3.11).

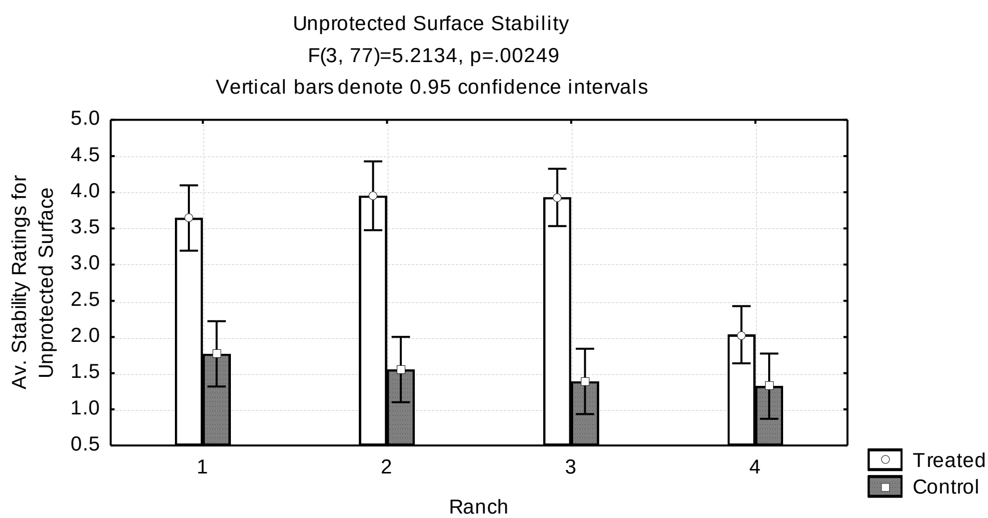

Figure 3.11. Average treated and control unprotected surface soil stability ratings by ranch.

The average stability ratings for the unprotected subsurface samples also showed a significant ($F(3,76)=3.0579$, $p=0.0333$) ranch by treatment interaction. Figure 3.12 shows the differences in the average unprotected subsurface soil stability ratings by ranch and treatment. Post hoc analysis using Fisher's LSD showed that ranches 1 and 3 had significant differences in treatment means ($p=0.0009$ and $p<0.0001$ respectively).

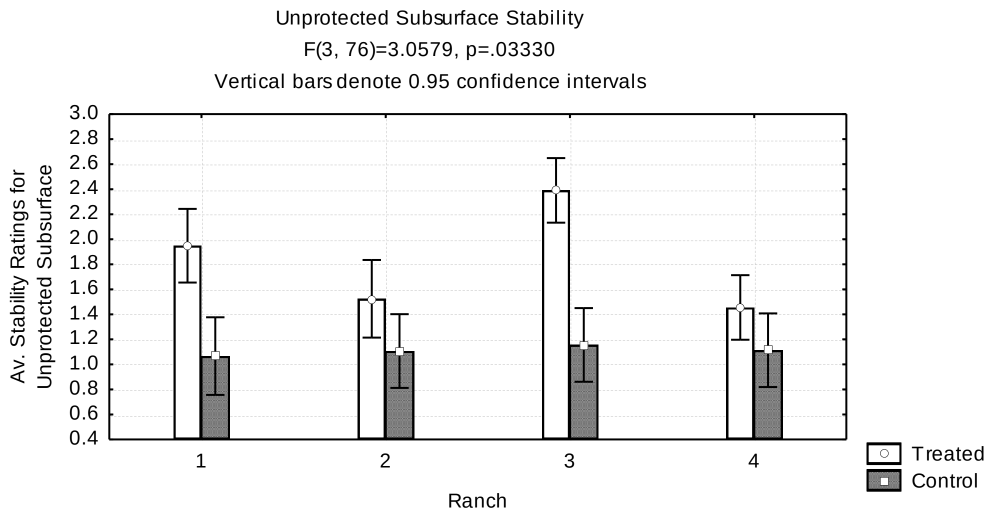

Figure 3.12. Average unprotected subsurface soil stability ratings by ranch and treatment.

Relationships. Simple linear regression was used to determine the relationship between the data elements from the expanded study. Table 3.6 summarizes selected results of those regressions that showed relationships with significant p-values (α=0.05).

Table 3.6. Selected results of simple linear regression of the data collected in the expansion study showing significant p-values (α=0.05).

Data Element	r	r^2	p
Foliar Cover : Bare Ground	-0.8125	0.6602	0.0001
Total Ground Cover : Bare Ground	-0.8798	0.7741	0.0001
Total Litter : Total Ground Cover	0.885	0.7832	0.0001
Annual Herbaceous Production : Litter	0.6927	0.4799	0.0264
Annual Herbaceous Production : Juniper Foliar Cover	-0.6704	0.4494	0.0338
Litter : Juniper Foliar Cover	0.4396	0.1933	0.00002
Basal Cover : All Surface Stability	0.4312	0.1859	0.00003
Canopy Gaps 25–50 cm : Protected Surface Stability	0.5128	0.263	0.0000005
Basal Gaps 25–50 cm : Protected Surface Stability	0.4734	0.2241	0.000005
Canopy Gaps >200 cm : All Surface Stability	-0.4610	0.2125	0.000008
Canopy Gaps >200 cm : Protected Surface Stability	-0.4824	0.2327	0.000005
All Surface Stability : Juniper Area	-0.4690	0.2200	0.000005

Based on the significance of the simple linear regression, multiple regression was used to look at the relationship between aggregate stability of all surfaces and selected other data elements. When aggregate stability of all surfaces was used as the dependent variable and aggregate stability of all subsurfaces, canopy gaps >200 cm, basal gaps >200 cm, basal cover, juniper foliar cover, foliar cover and bare ground were used as the independent variables, there was a significant relationship ($F(7,77)=17.71$, $p<0.0000$, $R=0.79$, $R^2=0.62$). This indicates that aggregate stability of all surfaces is a good integrator of the other data elements.

DISCUSSION

Soil aggregate stability is a function of the interactions of the soil microbes, mineral and organic inputs, the existing plant community and disturbance history (Kemper and Koch 1966, Tisdall and Oades 1982, Goldberg et al. 1988, Topp et al. 1996, Bird et al. 2007). The identified significant non-linear relationships may indicate the presence of thresholds of change (Friedel, 1991) and changes in feedback mechanisms affecting resilience of steady states (Briske et al. 2008). Bestelmeyer et al. (2006) showed the relationship between soil aggregate stability and bare patch size. Their work supports the concept of a discontinuous relationship between vegetation attributes, bare patch size and soil degradation that might be used to define thresholds. Their results suggest that soil aggregate stability data elements provided the best integrator of long term ecological responses to change in vegetation production and cover, bare ground, litter accumulation and bare patch size. They also suggest that soil aggregate stability may provide insight into the ecological dynamics of the ecological site.

The differences in the disturbance history between plots in the study area (herbicide application and reduction of juniper on treated sites) have affected the amount and kind of organic matter inputs. Organic matter inputs occur from both above ground and below ground sources. The increase in annual herbaceous production in the treated plots would indicate a greater organic matter input from all sources. Total litter cover for the expanded study plots totaled 48.4%. However, in central New Mexico arid and semiarid sites, Vanderbilt et al. (2008) suggest that unless litter incorporation into the surface occurs, biological decomposition of litter is less likely to occur than losses due to abiotic processes, primarily oxidation, physical weathering and loss due to wind. Biological crusts have the potential to directly stabilize soil surfaces and to add organic matter to the soil (Fletcher and Martin 1948). Only 0.58% of the ground was covered by moss, lichen crust or well developed cyanobacteria. The underground sources including root exudates, mycorrhizal fungi, glomalin, and root turnover are likely the largest contributor to the increased organic matter inputs. Gill and Jackson (2000) show that root turnover can be as high as 70% for blue grama

(*Bouteloua gracilis (Willd. ex Kunth) Lag. ex Griffiths*), 45 – 60% for little bluestem (*Schizachyrium scoparium (Michx.) Nash*) and 35 – 50% for sand bluestem (*Andropogon hallii Hack.*). These three grass species make up more than 80% of the total plant basal hits on the line point transects. Gill and Jackson (2000) also state that given the differences in root turnover between grasses and shrubs, it is expected that a lower proportion of root biomass is turned over annually in shrub invaded ecosystems than in grasslands. This further supports the conclusion of an increase in organic matter inputs in the treated sites due to the increase in annual herbaceous production.

Many studies discuss the reallocation of soil resources (e.g. soil moisture and nutrients) and soil aggregate stability as grassland conditions deteriorate, and describe the process of desertification (Davenport et al. 1996, Weltzin and McPherson 1997, Breshears and Barnes 1999, Reid et al. 1999, Bird et al. 2002, McIntyre and Tongway 2005, Bestelmeyer et al. 2006, Bird et al. 2007). These studies showed that as grass cover and reproduction declined and erosion of interspaces occurred, soil moisture, nutrients and organic matter decreased in the interspaces and increased under remaining shrubs. This study showed that the reverse is also true and that as grass cover increased, herbaceous production increased, bare ground decreased and soil aggregate stability improved.

The years 1985 – 1989 were above average rainfall years and the soil profiles in the treated plots were above 1.5 MPa over 90% of the time while the control plots had periods of drying. In 1989, which was a dry year with only 25 cm or 69% of normal precipitation, the difference in soil moisture between the control and treated plots was most pronounced. When precipitation was limited the control plots were dry a much higher percent of the time than the treated plots. It is believed that this increase in available soil moisture contributed to the treatment results. As shown by the analysis of the 1985 – 1989 data, the treated plots were significantly (α=0.05) different from the control plots for all the vegetation and soil cover data elements. Annual herbaceous production was negatively correlated to juniper canopy cover (r^2=0.86) and bare ground and juniper canopy cover were positively correlated (r^2=0.89). Litter was positively correlated to annual herbaceous production (r^2=0.79) and negatively correlated to bare ground (r^2=0.78). These correlations clearly indicate that as juniper canopy cover decreased, herbaceous production increased. These cover relationships and the increase in annual herbaceous production should lead to a corresponding increase in organic matter being added to the soil.

When the 2003 data were added to the analysis, the results showed that the treatment effect was still significant 18 years following treatment. The negative correlation between annual herbaceous production and juniper canopy remained strong (r^2=0.85). The negative correlation between annual herbaceous production and bare ground weakened (r^2=0.59), as did the positive correlation between annual herbaceous production and litter cover (r^2=0.51). The analysis of the extended study data also supported this conclusion. Eighteen years after treatment there was a significant (α=0.05) treatment effect for bare ground, and total ground cover and protected subsurface soil stability. Juniper foliar cover, aggregate stability of all surfaces, unprotected surfaces and unprotected subsurface all showed significant (α=0.05) ranch by treatment interactions. Although still significant, the changes between the 1985 – 1989 data and the 2003 and expanded study data along with the recovery of juniper canopy cover indicate the effect of treatment may be reaching an end. Analysis of the juniper canopy cover during 1985 – 1989 showed no treatment by ranch interaction, but when the 2003 data were added that interaction was significant (α=0.05). This indicates increased juniper canopy between 1989 and 2003 on most ranches in the study.

The linear regression models for the expanded study data showed several expected correlations. Foliar cover and total ground cover were both negatively correlated to bare ground (r^2=0.66 and, r^2=0.77, respectfully). There were also several weak (r^2<0.30) but statistically significant (α=0.05) relationships. Multiple regression using the aggregate stability of all surfaces as the dependent variable showed a significant (α=0.05) strong relationship (R^2=0.62) with selected other data elements.

SUMMARY and CONCLUSIONS

The differences in surface soil stability have major implications for use and management (Herrick et al. 2002). While the treatment of juniper with herbicide is very expensive, the resulting conditions may allow for continued economic use of the land. Results indicate that with no follow up treatment the effects of herbicide treatment may have reached their peak and are now declining. Wright (1990) states that non-sprouting junipers less than 4 ft. tall are susceptible to a ground fire of herbaceous fuels. Prescribed burning as a secondary treatment is relatively inexpensive and very effective in reducing the juniper seedlings and saplings. The use of fire to maintain the herbaceous production and resulting organic matter inputs would provide management with a tool that would facilitate the continued economical and sustainable use of the resources. Anecdotally, a visit to the study sites in 1990 indicated that a prescribed fire would carry with the fine fuels present and would reduce the number of juniper trees that had germinated and established since treatment. Based on the annual herbaceous production data collected in 2003 (<300 kg/ha), there currently is not enough fine fuel to carry a prescribed fire. It appeared that the herbicide treatment was necessary to restore soil aggregate stability and the function of the associated ecological processes, but that some follow up treatment such as prescribed fire, would be necessary to maintain those ecological functions.

LITERATURE CITED

Allen, C. D., and D. D. Breshears 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings, National Academy of Sciences of the United States of America. 95(25): 14839-14842.

Allison, C. D. 1982. A simple range trend monitoring procedure. New Mexico State University Cooperative Extension Service. 400 B-2. Las Cruces, NM.

Arnold, J. F. 1964. Zonation of understory vegetation around a juniper tree. Journal of Range Management. 47: 41-42.

Arnold, J. F., D. A. Jameson and E. H. Reid. 1964. The pinyon-juniper type of Arizona; effects of grazing, fire and tree control. U. S. Dept. Agr. Prod. Res. Rep. No. 84. Washington, D. C.

Bandelier, A. 1884. The southwest journals of Adolph F. Bandelier, 1883 – 1884. Carroll L. Riley and Charles H. Lange Eds. University of New Mexico Press, Albuquerque, NM.

Bestelmeyer, B. T., J. P. Ward, J. E. Herrick, and A. J. Tugel. 2006. Fragmentation effects on soil aggregate stability in a patchy arid grassland. Rangeland Ecology and Management 59(4): 406–415.

Bird, S. B., J. E. Herrick, M. M. Wander, and S. F. Wright. 2002. Spatial heterogeneity of aggregate stability and soil carbon in semi-arid rangeland. Environmental Pollution 116: 445–455.

Bird, S. B., J. E. Herrick, M. M. Wander and L. Murry. 2007. Multi-scale variability in soil aggregate stability: Implications for understanding and predicting semi-arid grassland degradation. Geodermia 140: 106–118.

Breshears, D. D., O. B. Myers, S. R. Johnson, C. W. Meyer and S. N. Martens. 1997. Differential use of spatially heterogeneous soil moisture by two semiarid woody species: *Pinus edulis* and *Juniperius monosperma*. Journal of Ecology 85(3): 289–299.

Breshears, D. D., and F. S. Barnes. 1999. Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landscapes within the grassland/forest continuum: a unified conceptual model. Landscape Ecology. 14(5): 465-478.

Briske, D. D., B. T. Bestelmeyer, T. K. Stringham, and P. L. Shaver. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management 61(4): 359–367.

Clary, W. P., and D. A. Jameson. 1981. Herbage production following tree and shrub removal in the Pinyon-Juniper type in Arizona. Journal of Range Management. 34: 109-113.

Cook, C. W. and C. D. Bonham. 1977. Techniques for vegetation measurements and analysis for a pre-and post-mining inventory. Colo. State Univ. Range Sci. Dept. Sci. Ser. No. 28. Ft. Collins, CO.

Davenport, D. W., B. P. Wilcox, and D. D. Breshears. 1996. Soil morphology of canopy and intercanopy sites in piñon-juniper woodland. Soil Science Society of America Journal 60: 1881–1887.

Dunnett, C. W. 1980. Pairwise Multiple Comparisons in the Homogeneous Variance, Unequal Sample Size Case. Journal of the American Statistical Association. 75(372): 789-795.

Fletcher, J. E. and W. P. Martin. 1948. Some effects of algae and molds in the raincrust of desert soils. Ecology 29: 95–100.

Fowler, J. M., B. E. Peacock and M. J. Schaber. 1985. Pinyon-juniper woodland type in New Mexico: Asset or liability. New Mex. State Univ. Agr. Exp. Sta. Bull. 718. Las Cruces, NM

Friedel, M. H. 1991. Range condition assessment and the concept of thresholds: A viewpoint. Journal of Range Management 44(5): 422–426.

Friedman, M. 1937. The use of ranks to avoid the assumption of normality implicit in the analysis of variance. Journal of the American Statistical Association 32 (200): 675–701.

Gardner, W. H. 1986. Water content. Klute, A.(ed.). Methods of soil analysis. Part I. pp. 493-544. Madison, WI, ASA and SSSA. Agron. Monogr.

Gifford, G. F. 1973. Runoff and sediment yields from runoff plots on chained pinyon-juniper plots in Utah. Journal of Range Management. 26: 440-443.

Gill, R. A., and R. B. Jackson. 2000. Global patterns of root turnover for terrestrial ecosystems. New Phytol. 147: 13–31.

Goldberg, S., D. L. Suarez, and R. A. Glaubig. 1988. Factors affecting clay dispersion and aggregate stability of arid-zone soils. Soil Science 146: 317-325.

Gottfried, G. J. 1999. Pinyon-juniper woodlands in the southwestern United States. In: Ffolliott, Peter F.; Ortega-Rubio, Alfredo, eds. Ecology and management of forests, woodlands, and shrublands in the dryland regions of the United States and Mexico: perspectives for the 21st century. Co-edition No. 1. Tucson, AZ: The University of Arizona; La Paz, Mexico: Centro de Investigaciones Biologicas del Noroeste, SC; Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: pp. 53-67.

Herrick, J. E., W. G. Whitford, A. G. de Soyza, J. W. Van Zee, K. M. Havstad, C. A. Seybold and M. Walton. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. Catena 44(1): 27–35.

Herrick, J. E., J. R. Brown, A. J. Tugel, P. L. Shaver, and K. M. Havstad. 2002. Application of soil quality to monitoring and management. Agronomy Journal 94: 3-11.

Herrick, J. E., J. W. Van Zee, K. M. Havstad, L. M. Burkett and W. G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. UDSA-ARS Jornada Experimental Range, Las Cruces, NM. Distributed by University of Arizona Press.

Hocking, R. R. 1985. The analysis of linear models. Monterey, CA: Brooks/Cole.

Hocking R. R., F.M. Speed, and A.T. Coleman. 1980. Hypotheses to be tested with unbalanced data. Communications in Statistics - Theory and Methods, 9(2): 117–129.

Horgan, P. 1954. Great River. Reinhart Co. New York, NY.

Jameson, D. A. 1962. Effects of burning on a galleta-black grama range invaded by juniper. Ecology. 43(4): 760-763.

Jameson, D. A. 1967. The relationship of tree overstory and herbaceous understory vegetation. Journal of Range Management. 20: 247-249.

Jameson, D. A. 1970. Juniper root competition reduces basal area of blue grama. Journal of Range Management. 23: 217-218.

Johnsen, T. N., Jr. 1962. One-seeded juniper invasion of northern Arizona grasslands. Ecological Monographs. 32(3): 187-207.

Kemper, W. D. and E. J. Koch. 1966. Aggregate stability of soils from western United States and Canada. USDA-ARS Tech. Bull., vol.1355. U.S. Govt Print. Office, Washington D.C.

Lanner, R. M. and T. R. Van Devender. 1998. The recent history of pinyon pines in the American Southwest. In: Richardson, David M., ed. Ecology and biogeography of Pinus. Cambridge, United Kingdom: The Press Syndicate of the University of Cambridge: pp. 171-182.

Levene, H. 1960. Robust tests for equality of variances.*In* Ingram Olkin, Harold Hotelling et al. Contributions to Probability and Statistics: Essays in Honor of Harold Hotelling. Stanford University Press. pp. 278-292.

McIntyre, S. and D. Tongway. 2005. Grassland structure in native pastures: links to soil surface condition. Ecological Management and Restoration 6 (1): 43–50.

McLeullough, G. T. 1882. Field notes of the survey of the sub-division of T.1N., R.8E., of the principle base meridian in New Mexico. Territorial Survey. (survey commenced September 7, 1882 and concluded September 13, 1882).

Miller, R. F., and P. E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. Bioscience. 44(7): 465-474.

Mueggler, W. F. 1976. Ecological role of fire in western woodland and range ecosystems. *In*: Use of prescribed burning in western woodland and range ecosystems: Proceedings of the symposium; 1976 March 18-19; Logan, UT. Logan, UT: Utah State University, Utah Agricultural Experiment Station: pp. 1-9.

Newman, B. D., A. R. Campbell, and B. P. Wilcox. 1997. Tracer based studies of soil water movement in semi-arid forests of New Mexico. Journal of Hydrology. 196: 251-270.

Petersen, S. L. and T. K. Stringham. 2008. Infiltration, runoff, and sediment yield in response to western juniper encroachment in southeast Oregon. Rangeland Ecology and Management. 61: 74–81.

Pieper, R. D. 1983. Overstory-understory relationships: Pinyon-juniper and juniper woodlands. *In:* Bartlett, E. T. and D. R. Betters (Eds.) Overstory-understory relationships in western forests. West. Reg. Res. Pub. No. 1. Colo. State Univ. Agr. Exp. Sta. Ft. Collins, CO.

Reid, K. D., B. P. Wilcox, D. D. Breshears, and L. MacDonald. 1999. Runoff and erosion in a Piñon-Juniper woodland: Influences of vegetation patches. Soil Science Society of America Journal. 63: 1869–1879.

Romme, W. H., C. D. Allen, J. D. Bailey, W. L. Baker, B. T. Bestelmeyer, P. M Brown, K. S. Eisenhart, M. L. Floyd, D. W. Huffman, B. F. Jacobs, R. F. Miller, E. H. Muldavin, T. W. Swetnam, R. J. Tausch and P. J. Weisberg. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Shaver, P. L. 2010. Short term response of soil moisture and vegetation to elimination of one-seed juniper canopy cover. *IN* Shaver, Patrick L. Quantification of State-and-Transition Model components utilizing long-term ecological response data following one-seed juniper treatment on a Deep Sand Savannah ecological site. PhD Dissertation, Oregon State University, Corvallis, OR.

StatSoft, Inc. (2005). STATISTICA (data analysis software system), version 7.1. www.statsoft.com.

Tisdall, J. M., J. M. Oades. 1982. Organic matter and water stable aggregates in soils. Journal of Soil Science. 33: 141-163.

Topp, G. C., W. D. Reynolds, F. J. Cook, J. M. Kirby and M. R. Carter. 1996. Physical attributes of soil quality. IN: Gregorich, E. G.., M.R. Carter, (Eds), Soil quality for crop production and ecosystem health. Elsevier, Amsterdam, pp. 21- 58.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2004. Ecological site descriptions. Field office technical guide, section II. Albuquerque, NM.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2003. National range and pasture handbook, revision 1. Washington, D.C.

U.S. Department of Agriculture. Soil Conservation Service. 1970. Soil survey Torrence area, New Mexico. Washington, D.C.

U.S. Department of Agriculture. Soil Conservation Service. 1988. Soil survey Socorro area, New Mexico. Washington, D.C.

Vanderbilt, K. L., C.S. White, O. Hopkins and J.A. Craig. 2008. Aboveground decomposition in arid environments: Results of a long-term study in central New Mexico. Journal of Arid Environments. 72(5): 696-709.

Weltzin, J. F. and G. R. McPherson. 1997. Spatial and temporal soil moisture resource partitioning by trees and grasses in a temperate savanna, Arizona, USA. Oecologia 112: 156-164.

West, N. E., K. H. Rea, and R. J. Tausch. 1975. Basic synecological relationships in pinyon-juniper woodlands. *In:* Gifford, G. F. and F. E. Busby (Eds.). The pinyon-juniper ecosystem. Utah State Univ. Agr. Exp. Sta. Logan, UT.

Western Regional Climate Center. 2009. New Mexico Climate Summaries, Gran Quivira National Monument, NM. http://www.wrcc.dri.edu/summary/climsmnm.html.

Westfall, P. H., R. D. Tobias, D. Rom, R. D. Wolfinger, and Y. Hochberg. 1999. Multiple Comparisons and Multiple Tests Using the SAS System, SAS Institute.

White, L. D. 1965. The effects of a wildfire on a desert grassland community. [thesis]. Tucson, AZ: University of Arizona. 107p.

Wilcox, B. P., C. D. Allen, B. D. Newman, K. D. Reid, D. Brandes, J. Pitlick, and D. W. Davenport. 1996a. Runoff and erosion on the Pajarito Plateau: observations from the field. pp. 433-439. *In:* New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Regions, 1996. New Mexico Geological Society. Albuquerque, N.M.

Wilcox, B. P., J. Pitlick, C. D. Allen, and D. W. Davenport. 1996b. Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. pp. 61-77. *In:* M.G. Anderson and S.M. Brooks (eds.) Advances in Hillslope Processes, Vol. 1. John Wiley and Sons, New York.

Wilm, H. G., D. F. Costello, and G. E. Klipple. 1944. Estimating forage yield by the double-sampling method. Journal of the American Society of Agronomy. 36: 194-203.

Winer, B. J., D. R. Brown, K. M Michels. (1991). Statistical principals in experimental design. (3rd ed.). New York: McGraw-Hill.

Woodbury, A. M. 1947. Distribution of pigmy conifers in Utah and northeastern Arizona. Ecology. 28(2): 113-126.

Wright, H. A. 1990. Role of fire in the management of Southwestern ecosystems. pp. 1–5 *In:* J. S. Krammes, technical coordinator. Effects of Fire Management of Southwestern Natural Resources. General Technical Report RM-191. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, CO. 293p.

CHAPTER 4: PROCESS BASED STATE AND TRANSITION MODEL FOR THE DEEP SAND SAVANNAH ECOLOGICAL SITE

ABSTRACT

A proposed three-state state-and-transition model (STM) for the Deep Sand Savannah ecological site was developed using historical data, experience, expert and scientific knowledge. The model was tested using k-means clustering, a non-hierarchical multivariate exploratory technique based on 22 years of soil and vegetation data from a manipulative restoration experiment. Three significantly (α=0.05) different clusters were produced, which supports the identification of three states in the proposed model. Surface soil stability data, were excluded from the k-means clustering analysis in order to utilize the data to validate the results. Mean stability ratings for State 1, the Reference State; State 2, the Juniper State; and State 3, the Eroded State; were 4.31±0.65, 2.78±0.55 and 2.05±0.38 respectfully. The relatively wide mean separation and low variances support the identification of three states. A qualitative analysis suggested that mean and range values for soil stability and the other soil and vegetation variables were generally consistent with the proposed model. Using the data from the individual plots within the three identified states, the proposed model was refined with the definition of specific processes and indicators associated with each state and transition. The STM includes feedback mechanisms that build resilience into each of the three identified states, at-risk community phases within the Reference and Juniper States and threshold values between the three states. This model will assist managers with identification of potential ecological thresholds and at-risk community phases, thus providing information to plan actions that facilitate the maintenance of ecological and economic sustainability while providing the broadest array of ecosystem services possible within the potential of the ecological site.

INTRODUCTION

The USDA Natural Resources Conservation Service uses a hierarchical classification system to organize landscapes into interpretive units for the purposes of inventory, assessment, and development of management plans. This system is made up of the following components: Land Resource Regions (LRR), Major Land Resource Areas (MLRA) and Land Resources Units (LRU) and finally Ecological Sites (USDA 2006). Ecological sites had their inception as functional edaphic units initially within the USDA Soil Conservation Service as Range Sites (Dyksterhuis 1949). Ecological sites are the basic unit of inventory, assessment and management on rangeland in the U.S. An ecological site is defined as a distinctive kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation (USDA 2003). The quantitative climax model (Dyksterhuis 1949) was integrated into the range site descriptions and became the model for directing management decisions and plant community response to disturbance levels. This model works well in many places, however as non-native species increase this model does not explain the resulting ecosystem dynamics. The quantitative climax model also does not adequately explain much of the plant community dynamics observed in the semi-arid and arid parts of the world and the shrub steppe (Westoby et al. 1989 and Laycock 1991).

The idea of alternative steady states as distinct entities for management purposes was first identified by Westoby et al. (1989). Working in south Texas, Archer (1989) discussed the changes in structure and function in a plant community following the cessation of fire resulting in the establishment of woody vegetation in a system previously dominated by grasses. The change from herbaceous to woody plant domination caused a change in structure and mineral cycling and hydrologic function that could not be reversed by reintroduction of the historic disturbance regime (the quantitative climax model). Significant inputs of energy (machinery, seed, etc) would be required to return the system to its original structure and function. Archer (1989) also indicated that rehabilitation would be short lived without continued energy inputs. Friedel (1991) discussed the concept of thresholds between domains from one state to another. She recognized two main types of thresholds: 1) change from herbaceous to woody dominated systems and 2) change from stable to eroding soils. The use of models to describe states and predict transitions across thresholds using quantitative indicators has been widely called for (Friedel 1991, Laycock 1991, Herrick et al. 2002, Bestelmeyer et al. 2003, Bestelmeyer et al. 2004, Briske et al. 2005, Briske et al. 2006, and Bestelmeyer et al. 2009).

Stringham et al. (2003) provided definitions and a framework to develop state-and-transition models (STM). These proposed concepts function at the ecological site scale, with each unique ecological site having its own STM. The definitions stress the relationship of the soil and vegetation components of the state in determining the functional capacity of ecological processes integral to ecological site integrity. The soil component, having been developed through time integrates parent material, climate, and landscape position with the interaction of the resulting biota, thus determining the capability of the ecological site. The interaction between soil and vegetation determines the functional status and the site resiliency and resistance to change in ecological process function. These definitions and framework provided the necessary structure to begin the development of process based STM derived from data for use in understanding the ecological dynamics involved in rangeland management.

Briske et al. (2008) further refined the model to include more emphasis on resilience within the state and the feedbacks involved in building resilience. Peterson et al. (1998) explained resilience as the amount of change or disruption that is required to transform a system from being maintained by one set of mutually reinforcing processes and structures to a different set of processes and structures. The community pathways between the community phases are feedback mechanisms that maintain the mutually reinforcing process and structure. Briske et al. (2008) provide additional definitions and suggested inclusion of resilience based concepts into the STM. Together with the definitions and frameworks from Stringham et al. (2003), they form the basics of state and transition concepts in use today.

There were three objectives to this study:

1. Develop a process based state-and-transition model for the Deep Sand Savannah ecological site (070CY123NM) using historical data, experience, expert and scientific knowledge.
2. Test the model using data collected during a long term effects study (Shaver 2010b) on this ecological site over a period of 22 years.
3. Use the data together with understanding of the biophysical processes on the ecological site to refine the model for use as a decision making tool.

METHODS

Study Area

The study was conducted on four private ranches located in south central Torrance and northeastern Socorro counties, New Mexico. The ranches are within an area defined by a 46 km diameter circle around the Gran Quivira Unit of the Salinas National Monument.

Elevation of the study area ranges from 1750 m to 2000 m with a flat to rolling dune topography. The soils in the area are all very similar and formed from aeolian sand deposits derived from mixed sources. The dominant characteristic of these deep soils is the coarse texture. The soil moisture regime at these locations is aridic-ustic and the soils at all locations are classified as mixed, mesic, Ustic Torripsamments (USDA 1970, USDA 1988). This classification was confirmed with a soil pit and several auger holes at each of the study sites. The soil is deep and excessively drained and the surface layer is a brown fine sand about 28 cm thick. The underlying material is strong brown fine sand in excess of 155 cm deep (USDA 1970, USDA 1988). The soil map unit components are all correlated to the Deep Sand Savannah ecological site in MLRA G070C, Central New Mexico Highlands, site number 070CY123NM (USDA 2004) (Figure 4.1).

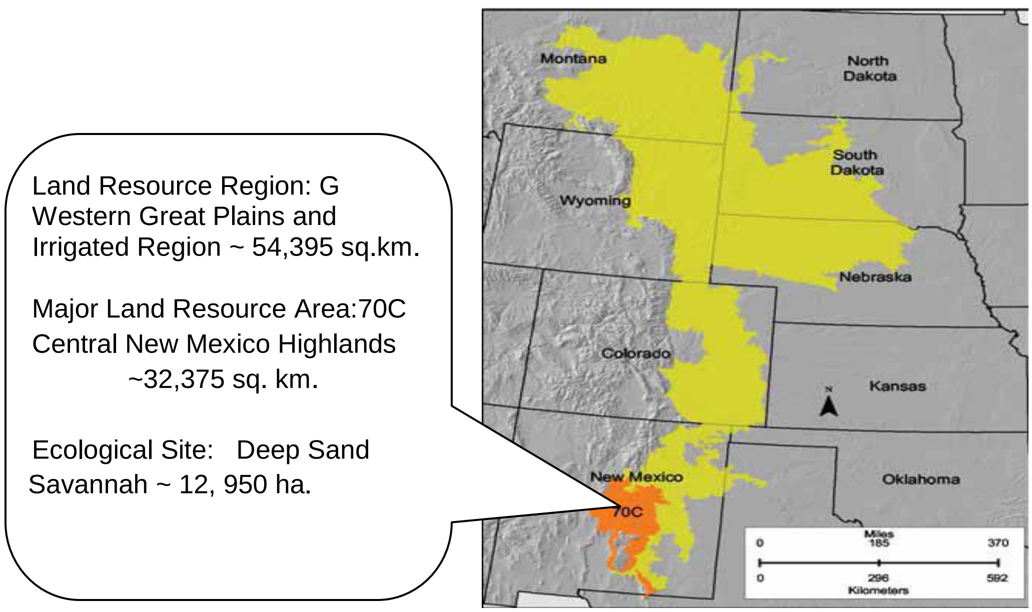

Figure 4.1. Map of Great Plains area of the U.S. showing location and size of Land Resource Region - G; Major Land Resource Area - 70C; and Ecological Site – Deep Sand Savannah.

The climate of the study area is characterized as semi-arid continental. The average annual precipitation at the Gran Quivira

National Monument ranges from 33-40 cm depending on the period of record. The thirty year average (1951-1980) immediately preceding the study was 36.2 cm. The actual precipitation during the study period (1985-1989) was above average with 46.5 cm. The most recent thirty year record (1971 – 2000) showed an average of 43.9 cm. Seventy five percent of the precipitation falls from April to October, primarily in the form of high intensity thunderstorms. Temperatures are characterized by distinct seasonal changes and large annual and diurnal fluctuations. The average annual air temperature is about 10^{o}C, with extremes of -34^{o}C in the winter to 40^{o}C in the summer. The average frost free season is 130–160 days. The last killing frost occurs in early May and the first killing frost in early October (Western Regional Climate Center 2009).

Information Used to Develop the Proposed State-and-Transition Model

A state-and-transition model was developed using archeological and historical data, professional experience, expert and scientific knowledge. A brief summary of the information used is included here.

In the early 1600's Gran Quivira (Jumano Pueblo) may have had as many as 1000 inhabitants. These people traded with the Pueblo peoples in the Rio Grande valley to the west, the Comanche in the east and the Apache in the south. Vivian (1961) quotes Spanish documents in which Nicolas de Aguilar, a Spanish "Encomendero", in 1663 states:

> "It has never been possible to keep livestock in said Pueblo because there is not water, for what there is comes only from wells which are a quarter of a league (~850 – 900 m) from the place, forty or fifty estados (~70 - 85 m) in depth. And therefore it costs a great deal to get water and it makes a lot of work for the Indians in obtaining it, and the wells are exhausted and there is an insufficient water supply for the people, for their lack of water is so great they are accustomed to save their urine to water the land and to build walls."

Excavations of the ruin of Gran Quivira show that over half of the animal bones found were blacktailed jackrabbit *(Lepus californicus)* and pronghorn antelope *(Antilocapra americana)*. Cottontail rabbits *(Sylvilagus auduboni)*, domestic and wild sheep *(Ovis spp)*, mule deer *(Odocoileus hemionus)*, domestic horse *(Equus caballus)* and bison *(Bison bison)* were also present in much smaller amounts. Trace amounts of bones were found from various birds, cougar *(Felis concolor)*, Gray fox *(Urocyon cinereoargenteus)* and black-tailed prairie dog *(Cynomys ludovicianus)* (Vivian 1961). The indication is that native herbivory prior to European influence, and even after Spanish reconquest and colonization in 1692, was mainly by lagomorphs and pronghorn antelope.

This area has historically been described as grassland with few junipers dotting the landscape. One-seed juniper (*Juniperus monosperma* (Engelm.) Sarg) was confined to ridge tops and the foot slopes of adjoining mountains (Bandelier 1884, Horgan 1954, McLeullough 1982). While not specific to the peoples of the Jumano Pueblo, Stewart (2002) states that the Apache, Navajo and Pueblo inhabitants of this area used fire as a management tool for hunting, to draw game into the area, for clearing crop fields and to increase the yield of grass seeds used for grain. Other authors support this contention with fire frequencies for the area ranging from 4 – 20 years (Allen 1989, Baisan and Swetnam 1997, and Frost 1998). This fire regime would be frequent enough to create and maintain a grassland aspect and herbaceous dominated plant community. Wright (1990) indicated that fires every 10 – 30 years kept juniper on shallow, rocky, rough places. He also indicated that non-sprouting junipers less than four feet tall are readily killed by ground fires of herbaceous fuel. Dwyer and Pieper (1967) show that one-seed juniper less than four feet tall were killed with a ground fire.

Given this description, it is very unlikely that large numbers of domesticated or wild animals were grazing in this region until the Anglo expansion into New Mexico occurred in the mid 1800's. It is estimated that there were several hundred thousand head of sheep in New Mexico from 1788 onward to about 1870 (Denevan 1967). Domestic livestock grazing increased rapidly following Anglo expansion. From 1870 to 1890 sheep numbers increased to around 5 million state wide (Denevan 1967). Cattle numbers were approximately 137,000 in 1880 and reached 1.3 million by 1889. Numbers of sheep and cattle increased from the 1880's to the end of World War I and have been decreasing since 1920 (Schickedanz 1980). In 1906 there were approximately 1 million cattle and 6 million sheep in New Mexico. By 1979, cattle had increased to 1.5 million, while sheep had decreased to 600,000. This increase in livestock numbers undoubtedly impacted the natural disturbance regime of frequent fires and low levels of herbivory. Many studies have linked juniper expansion to increased livestock and decreased fire frequency (Jameson 1962, Johnsen 1962, Arnold 1964, Arnold et al. 1964, White 1965, Jameson 1967, Jameson 1970, Clary and Jameson 1981, and Pieper 1983, Miller and Wigand 1994, Allen and Breshears 1998, Lanner and Van Devender 1998).

The historical information, the Deep Sand Savannah ecological site description, 070CY123NM (UDSA 2004), and existing data (Shaver 2010a, 2010b) indicate these physical and environmental conditions have led to ecological dynamics that have produced an ecological site characterized by tall and mid warm season grasses. Warm and cool season mid and short grasses were the sub-dominant plant functional groups on this ecological site. Observation indicates the forb component was variable depending on timing and amount of precipitation and with the season. The woody plant component was both spatially and temporally variable depending on time since the last fire, but was always a minor component of the plant community (Allen 1989, Baisan and Swetnam 1997, Frost 1998, Stewart 2002). The production of a continuous fine fuel load and resulting fires were important negative feedbacks that limited the abundance of the woody components and maintained an herbaceous dominance on the site. One-seed juniper trees less than four feet tall were readily killed by ground fire fueled by herbaceous fuels (Dwyer and Pieper 1967, Wright 1990). The resilience of the ecological site was maintained by the continued input of organic matter primarily from root turnover of the herbaceous species (Gill and Jackson 2000) and herbaceous litter. Development and maintenance of stable soil aggregates is a function of the interactions of soil microbes and organic inputs, and the continuation of herbaceous litter production and root turnover. The resulting soil aggregate stability is integral to the negative feedback mechanisms responsible for ecological resilience. High soil aggregate stability provides optimal rates of infiltration, water holding capacity, aeration and mineral cycling, which maintains herbaceous production. Herbaceous production provides for a uniform distribution of soil nutrients and water throughout the soil profile (Schlesinger et al. 1990). This uniform distribution maintains uniform organic matter inputs (Kemper and Koch 1966, Tisdall and Oades 1982, Goldberg et al. 1988, Topp et al. 1996, Bird et al. 2007) and strengthens the resilience of the site to the periodic fires that were necessary to control the establishment and increase of one-seed juniper.

As European settlement progressed, domestic livestock numbers increased rapidly and the resulting grazing pressure decreased fine fuel for fires (Savage and Swetnam 1990, Swetnam et al. In Press). One-seed juniper increased in density and cover with an increase in the time since the last fire, also causing a decrease in herbaceous production (Jameson 1962, Johnsen 1962, Arnold 1964, Arnold et al. 1964, White 1965, Jameson 1967, Jameson 1970, Clary and Jameson 1981, and Pieper 1983, Miller and Wigand 1994, Allen and Breshears 1998, Lanner and Van Devender 1998). Decreased herbaceous production caused a decline in organic matter inputs, resulting in lowered soil aggregate stability (Tisdall and Oades 1982). As herbaceous production and cover were reduced and bare ground and erosion increased, fine fuels for fire became inadequate and resilience was weakened. Soil moisture, nutrients and organic matter decreased in the interspaces and increased under the trees and shrubs (Padien and Lajtha 1992, Davenport et al. 1996, Weltzin and McPherson 1997, Breshears and Barnes 1999, Reid et al. 1999, Bird et al. 2002, McIntyre and Tongway 2005, Bestelmeyer et al. 2006 and Bird et al. 2007). The decreased herbaceous production and organic matter in the interspaces likely decreased soil aggregate stability, infiltration, water holding capacity and mineral cycling. As the redistribution of resources continued the strength of the feedback mechanisms began to shift and site resilience decreased.

Plant interspaces continued to lose resources resulting in a lower proportion of root biomass available for annual turnover (Gill and Jackson 2000) further concentrating resources under the one-seed juniper. Gill and Jackson (2000) discussed the difference in root turnover from grasses and shrubs and state that the turnover from woody plants is much less proportionately than that of grasses. When juniper increases, the resulting reallocation of underground resources is well documented (Padien and Lajtha 1992, Davenport et al. 1996, Weltzin and McPherson 1997, Breshears and Barnes 1999, Reid et al. 1999, Bird et al. 2002, McIntyre and Tongway 2005, Bestelmeyer et al. 2006 and Bird et al. 2007). This change develops abiotic feedback mechanisms controlled by wind and water erosion, leading to desertification that builds very strong resilience feedback mechanisms and resistance to change (Schlesinger et al. 1996, Whisenant 1999, Bestelmeyer et al. 2006).

Testing the Model

The proposed model was tested using data collected from treated and control plots from the long term and expanded study (Shaver 2010a). A significant response of available soil moisture and herbaceous vegetation production to juniper treatment was found in a short term response study on this ecological site (α=0.05) (Shaver 2010a). These results led to the design of a long term study to quantify available soil moisture and vegetation responses from 1985 – 1989 and again in 2003. The long term study analyzed the effects on selected vegetation and soil variables to a 1985 herbicide application on one-seed juniper. The long term study experimental design was a complete randomized block on two of the ranches and generalized blocks - unbalanced on the other two ranches (Shaver 2010b). Eighteen years after treatment there were significantly (α=0.05) different treatment effects for parameters measured, which included bare ground, total litter cover, total ground cover and protected subsurface soil stability. Juniper foliar cover, aggregate stability of all surfaces, of unprotected surfaces and of unprotected subsurface all showed significant (α=0.05) ranch by treatment interactions. There were significant (α=0.05) relationships (R^2=0.62) between aggregate stability of all surfaces and many other data elements tested. These results led to the development of a hypothesis of ecological dynamics involved in a proposed process based state-and-transition model. Figure 4.2 shows the relative layout and locations of the plots in the long term study.

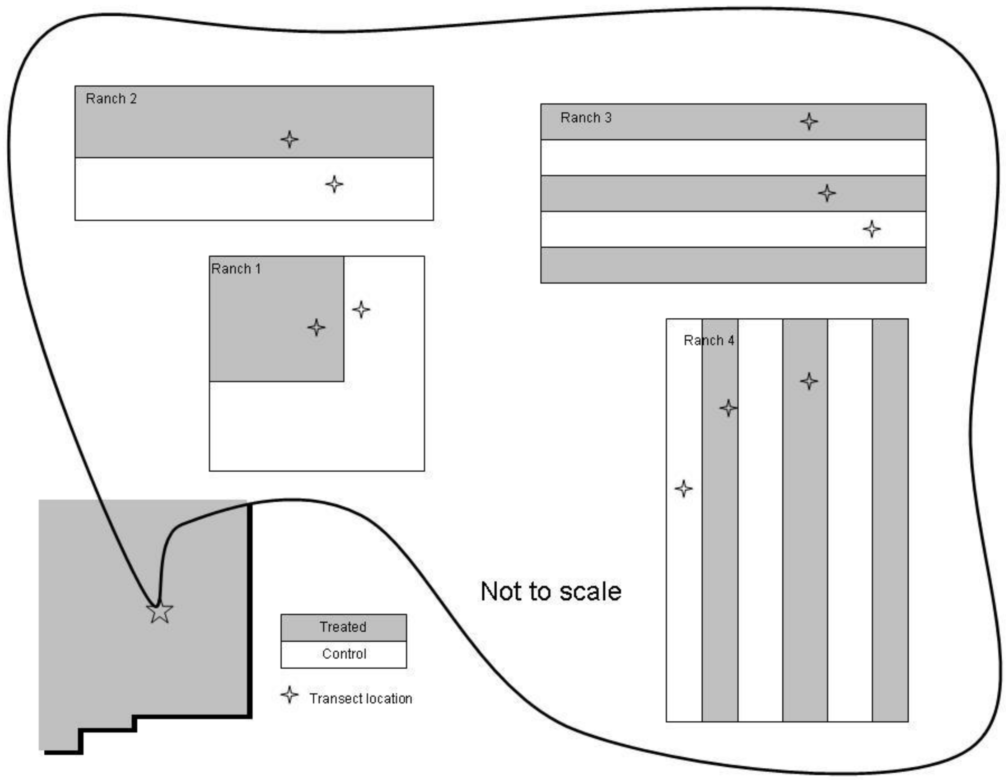

Figure 4.2. Plot layout for long term effects study showing the relative location of ranches and treated and control plots.

Refining the Model

The model was refined using professional experience and expert knowledge gained over the last 22 years of working on this ecological site. Scientific knowledge gained from the long term and expanded studies (Shaver 2010b) and the scientific and historical literature was also used.

Soil and Vegetation Measurements

The moisture, vegetation and soil variable responses to the short term response study conducted in 1984, and the long term response study conducted from 1985 – 1989, provided the foundation for this expanded study. The long term study contained 10 plots on four ranches (Shaver 2010a, Shaver 2010b). In the fall of 2003, a total of eighty-six 50 m transects were established as part of the long term effects study. Three of these were placed at each of the ten original plot locations with the remaining 56 transects randomly located within treatments on the four ranches in the study (Shaver 2010b). Table 4.1 summarizes the number of plots in each treatment on each ranch for both the long term study and the expanded study.

Table 4.1. Summary of number of plots in original study set up in 1985 and in the expansion study set up in 2003.

	Original Plots		Expanded Study	
Ranch	Treated	Control	Treated	Control
1	1	1	10	10
2	1	1	10	10
3	2	1	13	10
4	2	1	13	10

The objectives of both the long term and expanded studies were to test the hypothesis that vegetation production, cover and soil moisture will increase after juniper treatment on coarse textured soils and that these changes would be sustained over time. The data from these studies as well as the expanded plot data were used in this study to test the proposed STM and develop state and community phase characteristics. The following data elements collected on all 86 transects in 2003 were used in this analysis: herbaceous foliar cover (%), bare ground (%), basal cover (%), canopy gaps >200 cm (%), basal gaps >200 cm (%), juniper foliar cover (%), surface soil stability (rating 1-6) and subsurface soil stability (rating 1-6). These data elements were selected due to their significance to the long term response portion of the larger study (Shaver 2010b), their use in the development of the proposed STM and their importance in resilience feedback mechanisms (Schlesinger et al. 1999, Whisenant 1999, Bestelmeyer et al. 2006, Petersen et al. 2009).

Vegetation, ground cover and vegetation gaps. A 50 m transect was established and a point reading was taken at 0.5 m intervals starting at 0.5 m. Canopy was counted regardless of height and whether dead or alive. Species were recorded only once in the canopy. A basal hit of a plant species automatically warranted a canopy hit of that species. Woody litter (>5mm), herbaceous litter, embedded litter, duff, lichen, moss, bedrock, rock and bare soil were also recorded. Intercanopy vegetation gaps >20 cm in both canopy and plant bases were measured (Herrick et al. 2005).

Soil stability. Nine surface samples and nine subsurface samples 0.64 cm in diameter, were collected at set distances perpendicular to the left side of the point transect line. Samples were taken at 5 m intervals for a total of nine samples per line. An in-field soil stability test developed by Herrick et al. (2001) was used. Soil peds were rated on a scale of 1 – 6 (from least to most stable) based on the percent of ped remaining before and after sieving.

Statistical analysis. Statistica 7.1 software (StatSoft, Inc. 2005) was used for all analyses. The proposed STM was tested using all the transect level data except surface soil stability data, collected in 2003. The multivariate exploratory technique k-means clustering was used to group each of the 86 transects into meaningful classes. This nonhierarchical method, examines the means of each cluster on each dimension to assess how distinct from one another the clusters were. To test the proposed model, clustering was performed with 2, 3, 4, and 5 clusters and average Euclidean distances between clusters were calculated. The number of clusters with the highest average distance between clusters was used as the optimum number (StatSoft, Inc. 2007). This optimum number of clusters was compared to the proposed state-and-transition model and considered to represent the number of states in the STM. This model structure was then used for further development of state and community phase characteristics.

In order to test the number and distinctness of the cluster defined states (StatSoft, Inc. 2007), surface soil stability data were tested separately. Surface soil stability was used for this test due to the relationship to the other data elements (Shaver 2010b) and to changes in organic matter inputs associated with resilience building feedback mechanisms. A one-way analysis of variance (ANOVA) was used to test the differences between clusters using surface soil stability as the dependent variable.

The values for the other data elements were relativized to the same scale for data analysis. Herbaceous foliar cover, basal cover and subsurface soil stability data were also inverted so that the direction of interpretation was the same (McCune and Grace 2002).

RESULTS

Developing the State-and-Transition Model

A three state model based on historical information, experience, expert and scientific knowledge was developed. State characteristics, at-risk community phases, thresholds, resilience feedback mechanisms and transitions were proposed. Figure 4.3 shows the proposed state-and-transition model.

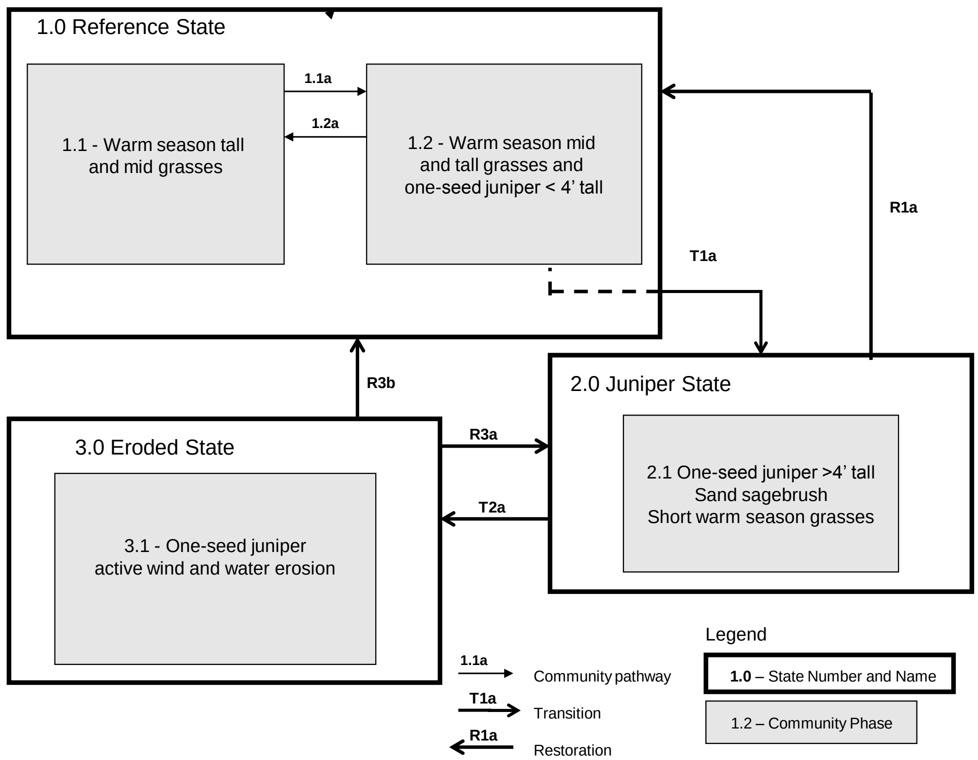

Figure 4.3. Proposed state and transition model for the Deep Sand Savannah Ecological Site.

Testing the Model

The k-means clustering procedure was used with two, three, four and five clusters. The average Euclidean distances between clusters showed the optimum number of clusters as three. Distances for each of the clustering analyses are shown in Table 4. 2.

Table 4.2. Average Euclidean distances between clusters when two, three, four and five clusters were used.

	2 Clusters	3 Clusters	4 Clusters	5 Clusters
Distance	0.140124	0.172404	0.160947	0.158287

The resulting three clusters or states in the state-and-transition diagram determined the transect membership for the state. Individual plot membership in the three states showed that all transects in state 1 had been treated with herbicide in 1985. Table 4.3 provides a summary of the number of transects in each state and how many of those transects were treated in 1985.

Table 4.3. Number of total, treated and control transects in states.

	Total	Treated	Control
State 1	31	31	0
State 2	35	13	22
State 3	20	2	18

The results of the ANOVA test of the surface soil stability data showed that the states were significantly ($\alpha=0.05$) different from one another ($F(2,83)=113.39$, $p=0.0000$) (Figure 4.4).

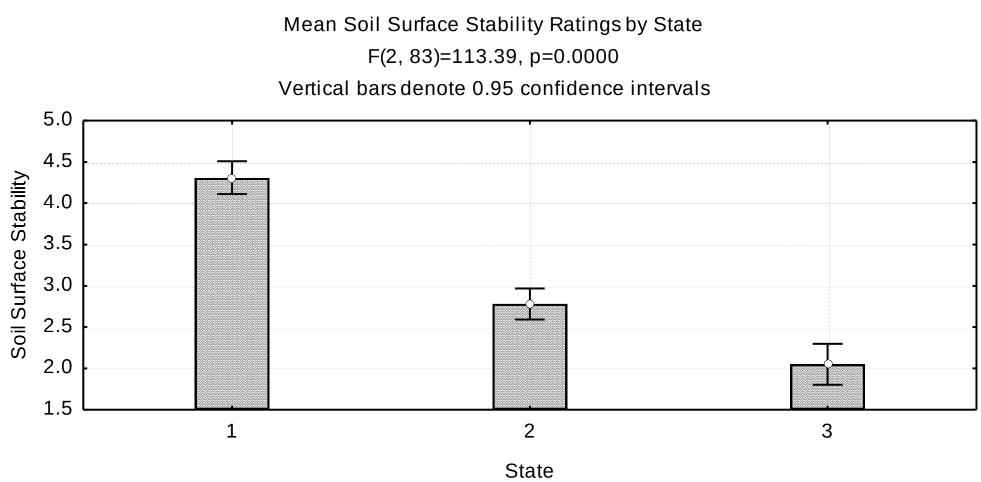

Figure 4.4. Soil surface stability differences between states.

Table 4.4 shows the means, standard deviations, standard errors and ranges within a 95% confidence interval for each data element in each state. Differences between the states as well as the variability of those data elements within the states were evident.

Table 4.4. Means, standard deviations and standard errors and range for a 95% confidence interval for each data element in each state.

Data Element	State	Mean	Standard Deviation	Standard Error	Confidence Interval -95%	Confidence Interval +95%
Surface Soil Stability	1	4.3	0.6	0.1	4.1	4.5
	2	2.8	0.6	0.1	2.6	3.0
	3	2.0	0.4	0.1	1.9	2.3
Subsurface Soil Stability	1	2.7	0.7	0.1	2.4	2.9
	2	1.7	0.3	0.1	1.6	1.8
	3	1.4	0.3	0.1	1.3	1.6
Canopy Gaps > 200cm (%)	1	7.2	7.9	4.2	4.3	10.1
	2	14.8	12.6	10.4	10.4	19.1
	3	35.8	14.7	3.3	28.9	42.7
Basal Gaps > 200 cm (%)	1	13.0	11.7	2.1	8.7	17.3
	2	22.8	17.7	3.0	16.8	29.0
	3	45.4	25.6	5.7	33.5	57.4
Basal Cover (%)	1	9.0	4.6	0.8	7.3	10.7
	2	6.9	5.5	0.9	5.0	8.8
	3	2.7	2.0	0.5	1.7	3.7
Juniper Foliar Cover (%)	1	12.1	11.4	2.0	7.8	16.3
	2	22.2	12.8	2.2	17.8	26.6
	3	23.2	9.9	2.2	18.5	27.8
Herb. Foliar Cover (%)	1	49.5	10.3	1.8	45.7	53.3
	2	50.0	13.5	2.3	45.4	54.7
	3	35.7	10.5	2.4	30.8	40.6
Bare Ground (%)	1	29.6	8.6	1.5	26.5	32.8
	2	32.8	13.1	2.2	28.3	37.3
	3	44.8	10.5	2.3	39.4	49.7

Refining the Model

This variability within transects clustered into a state may provide evidence of potential at-risk community phases. Viewing the data in two- and three- dimensional scatter graphs provided the opportunity to observe the variability within states and the patterns and exceptions in the distribution of the data values. The lowest value for surface soil stability in state 1 was 2.7 and the next lowest was 3.4. The highest value in state 3 was 2.8. The percent of canopy gaps >200 cm in state 1 was below 15%, with two exceptions and was above 24% in state 3 with three exceptions. The percent of canopy gaps >200 cm was always greater than 10% in state 3 (Figure 4.5).

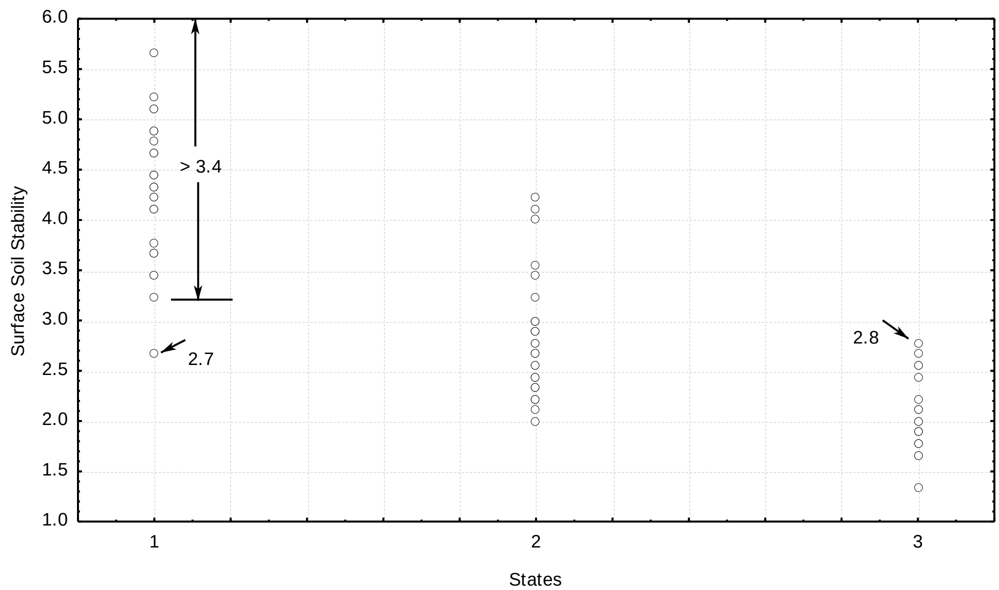

Figure 4.5. 2-dimensional scatter graph for surface soil stability ratings and the three states identified using the k-means joining method. Also note the distribution of the surface soil stability rating within the states, especially the low values in state 1 and the high values in state 3.

When viewed in a 3-D scatter graph with surface soil stability the pattern of transect distribution by state within surface soil stability space can easily be seen. Transects in state 2 clearly show a unique distribution and relationship in three dimensional space from the transects in state 1 and state 3. Exceptions to the patterns in State 1 and State 2 can also be seen (Figure 4.6).

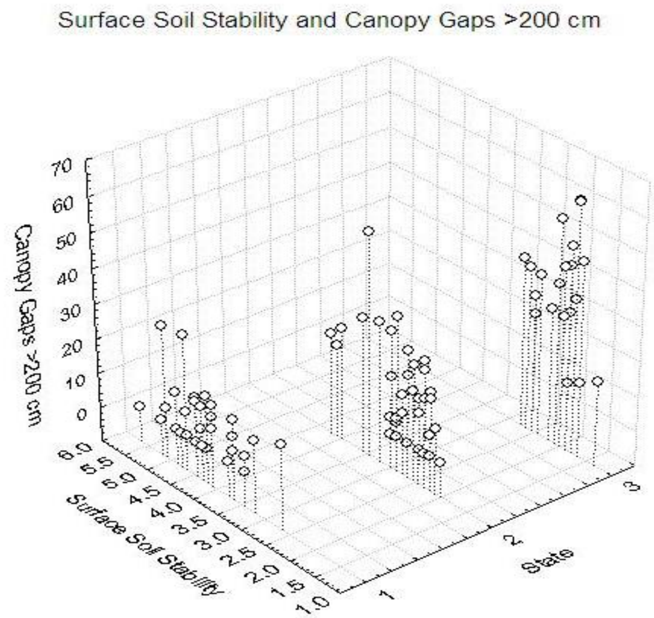

Figure 4.6. 3-dimensional scatter graph by state with surface soil stability and % canopy gaps >200 cm. This graph shows the unique distribution of large canopy gaps and surface soil stability values by state.

State 1 values for the percent of basal gaps >200 cm, showed that with one exception less than 30% were covered by basal gaps >200 cm, while in state 3 only five transects had less than 30% covered by basal gaps >200 cm. In three dimensional space, the distribution of percent basal gaps >200 cm in the three states is distinct as is the identification of exceptions to the pattern (Figure 4.7).

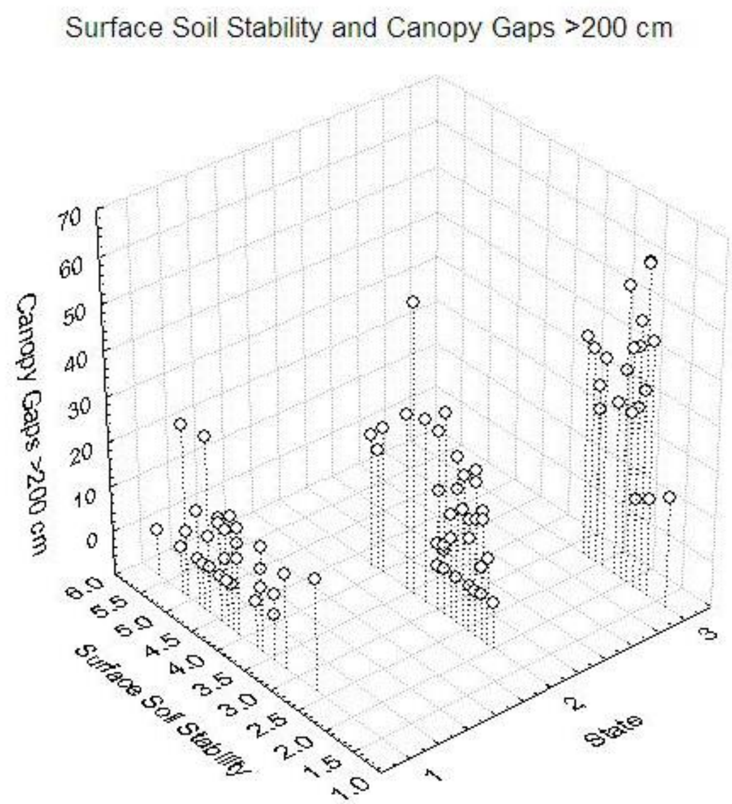

Figure 4.7. 3-dimensional scatter graph by state with surface soil stability and % basal gaps >200 cm. This graph shows the unique distribution and relationship of large basal gaps by state in surface soil stability space.

There was a great deal of overlap in percent basal cover for all three states, and the variability in the mean values. However, the basal cover in state 3 never exceeded 7% and the distribution of basal cover by state in surface soil stability space showed a distinct pattern of low surface soil stability ratings and low basal cover in state 3 and higher values for both variables as they moved from state 3 to state 2 to state 1 respectively (Figure 4.8). The exceptions to the pattern in state 2 were readily identified for further examination.

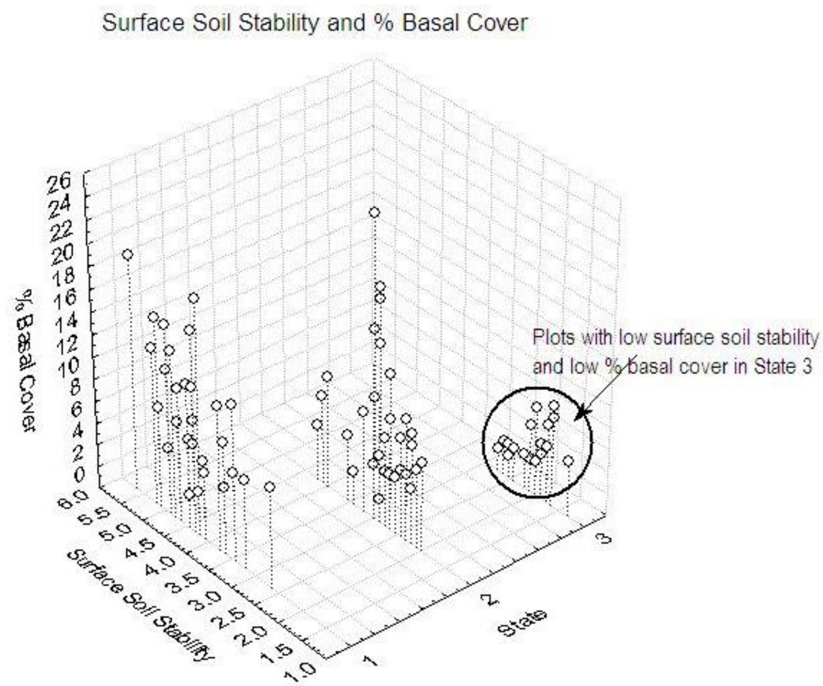

Figure 4.8. 3-dimensional scatter graph by state with surface soil stability and % basal cover. This graph clearly shows the unique distribution of basal cover by state in surface soil stability space.

Subsurface soil stability values showed similar trends with the other data elements. State 1 values were above 1.8, with one exception and state 3 values were below 1.8 with two exceptions, one of which had a rating of 1.9. Subsurface soil stability ratings for state 3 never exceeded 2.3.

When viewed in three dimensional space the pattern and relationship of basal and canopy gaps as grouped into states 1, 2 and 3, is clear (Figure 4.9). Exception to the patterns can also be readily identified for further evaluation.

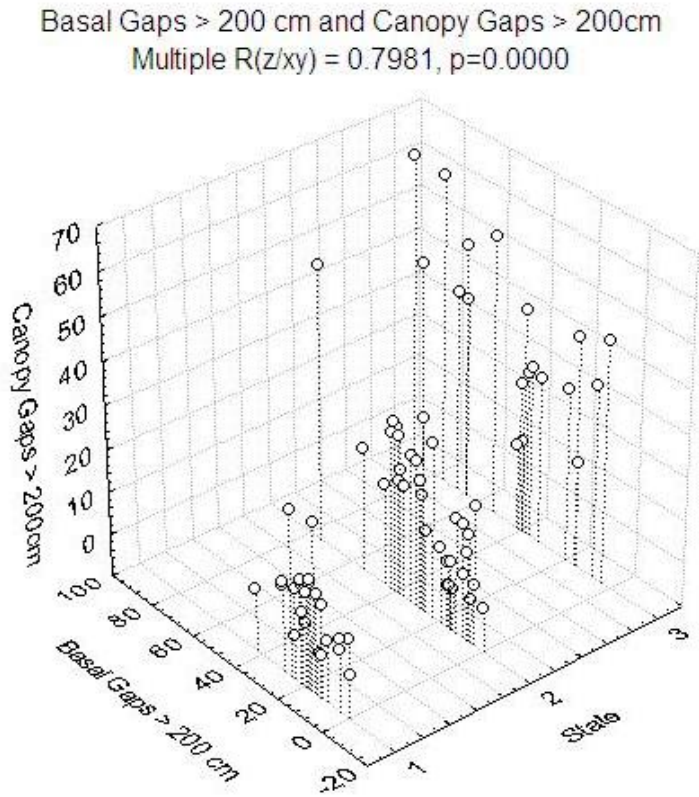

Figure 4.9. 3-dimensional scatter graph by state with basal gaps >200 cm and canopy gaps >200 cm. This graph clearly shows the unique relationship, distribution and importance of basal and canopy gaps by state.

When viewed in three dimensional space the pattern and relationship of surface soil stability and subsurface soil stability as grouped into states 1, 2 and 3, is evident (Figure 4.10). Exceptions to the patterns can also be readily identified for further evaluation.

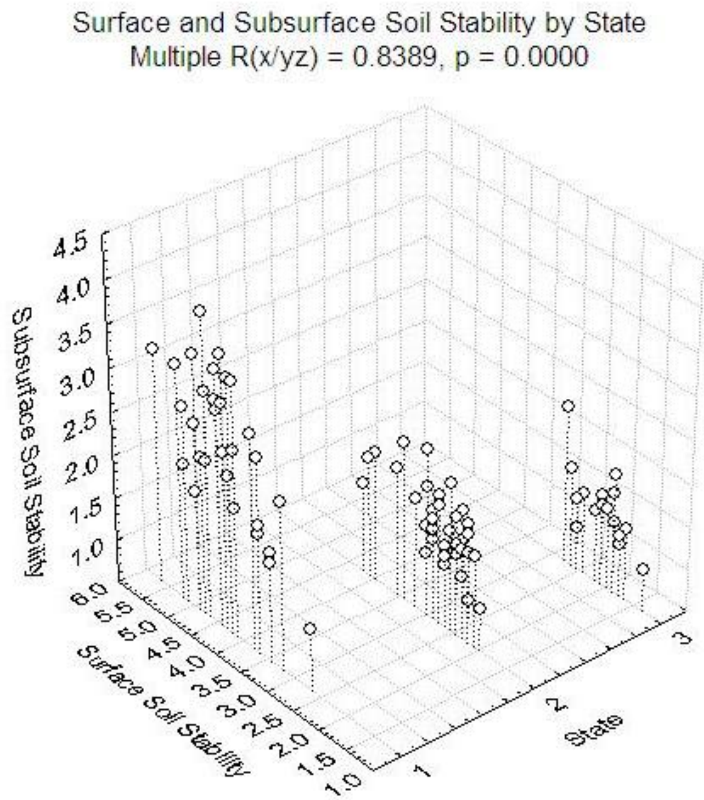

Figure 4.10. 3-dimensional scatter graph by state with surface soil stability and subsurface soil stability. This graph clearly shows the unique relationship and distribution of surface soil stability and subsurface soil stability by state.

DISCUSSION

Developing the Model

The development of STMs using available historical and professional knowledge must be preceded by a good understanding of the concepts and parameters of the specific ecological site. The range of variability that is expressed throughout the spatial extent of the ecological site must be observed over time to understand the temporal and spatial variability of the ecological site. However, without data to support the conceptual understanding of the ecological dynamics, interpretation is limited and use of the model as a decision making tool is restricted.

Testing the Model

The k-means clustering method produced three states that were significantly different (α=0.05) from one another. When used to validate the clustering of multiple variables into ecological states, surface soil stability showed significant (α=0.05) differences between states. These differences support the use of surface soil stability as an indicator and predictor of state membership. The data presented supports the hypothesis of a three state ecological process based state-and-transition model.

Using the 95% confidence interval values for the variables the three states can be described in terms of those data elements. Table 4.5 summarizes the range of values for each data element used to describe three states in the STM for this ecological site.

Table 4.5. Summary of data values within 95% confidence interval of the mean value for variables in state descriptions.

	Reference State 1.0	Juniper State 2.0	Eroded State 3.0
Surface Soil Stability	>4.0	2.6 – 3.0	< 2.4
Subsurface Soil Stability	>2.4	1.6 – 1.8	< 1.6
Canopy Gaps > 200cm	<10%	10% - 20%	> 28%
Basal Gaps > 200cm	< 17%	17% - 29%	> 33%
Basal Cover	>7%	5% - 9%	< 4%
Juniper Foliar Cover	<17%	18% - 27%	>20%
Herb. Foliar Cover	>45%	>45%	<41%
Bare Ground	<33%	28% - 37%	>39%

Refining the Model

These ranges in data element values provide evidence for community phase identification in the STM. They also provide indication of the feedbacks involved in changing resilience from one state to another. The indication of different community phases within states due to the variability of the within state data led to further examination of the distribution patterns in the 2- and 3- dimensional scatter plots and individual data for each transect identified in each state. Because of the importance of the soil stability and vegetation gap distribution to ecological function, these data elements were used to sort transects in each state. There were six transects grouped into the Reference State whose surface soil stability values were <4.0 and subsurface soil stability values were <2.4. All of these transects had at least one of the vegetation gap data elements whose value was outside the 95% C.I. for the Reference State. There were two additional transects where either surface or subsurface stability was less than the 95% C.I. yet contained values for both vegetation gap elements that were greater than the 95% C.I. for those data elements. In the Juniper State there were five transects with values for both soil stability elements greater than the 95% C.I. These five transects also had at least one other data element that was outside the 95% C.I. on the Reference State side. All five of these transects had been treated with herbicide in 1985.

The values for the soil stability variables and the vegetation gap variables for these five transects and the eight transects identified in the Reference State were grouped and the mean and 95% C.I. were computed. The same computation was made for the remaining transects in the Reference State and Juniper State. When those values were compared, there was very little overlap and the preponderance of evidence indicates that the fourteen transects are a community phase in the Reference State. This procedure was continued with the other transects in the Juniper and Eroded States producing an additional community phase (Table 4.6). When comparing these ranges in values to the previous state ranges (Table 4.5), community phases 1.2 and 2.2 appear to be at-risk community phases approaching a threshold from one state to another. The resulting state-and-transition diagram is shown in Figure 4.11.

Table 4.6. Summary of data values within 95% confidence interval of the mean value for community phases within states.

State	Reference State		Juniper State		Eroded State
Community Phase	1.1	1.2	2.1	2.2	3.1
Surface Soil Stability	>4.3	3.4 – 4.1	2.5 – 2.8	2.4 – 2.8	<2.1
Subsurface Soil Stability	>2.7	1.8 – 2.1	1.5 – 1.8	1.2 – 1.8	<1.5
Canopy Gaps > 200cm	<8%	12 – 27%	7 – 13%	18 – 33%	>29%
Basal Gaps > 200cm	<15%	15 – 36%	12 – 25%	29 – 55%	>30%
Basal Cover	>7%	5 – 9%	5 – 10%	<4%	<4%
Juniper Foliar Cover	<8%	11 – 24%	18 – 28%	16 – 32%	>29%
Herb. Foliar Cover	>46%	40 – 54%	47 – 57%	32 – 46%	<30%
Bare Ground	<32%	24 – 42%	27 – 37%	33 – 47%	>39%

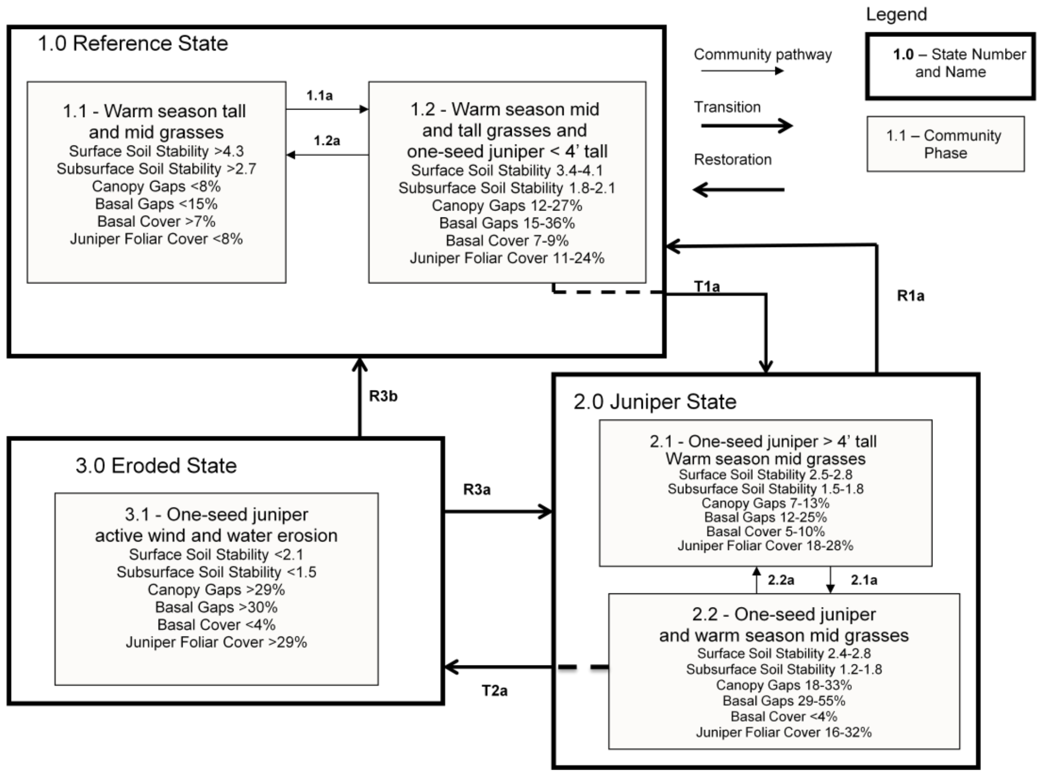

Figure 4.11. State-and-transition model diagram using data elements to identify state boundaries and at-risk community phases.

The community pathways designated 1.1a and 1.2a represent the feedback mechanisms that maintain the resilience of the Reference State. As time since the last fire increased, the one-seed juniper in community phase 1.2 increased in size and number and the negative feedback mechanisms associated with site resilience weakened. Positive feedbacks associated with degradation increased, making this the at-risk community phase in the Reference State. The average surface stability ratings in community phase 1.2 was 3.7, well within the range for the Reference State, but the average subsurface stability rating was 1.9, at the lowest end of the Reference State range. Canopy gaps >200 cm and basal gaps >200 cm were 20% and 25% respectively, both within the range for the Reference State. Juniper foliar cover for these nine transects averaged 21%, outside the range of the Reference State and within the range of the Juniper State indicating these transects are at risk of crossing an ecological threshold.

As the one-seed juniper increased in size and density, soil and water resources began to concentrate under and around the juniper plants, reducing herbaceous production. This reduction in herbaceous production increased gap size, reduced fine fuel for fires and reduced organic matter inputs for soil aggregate stability. This agrees with Archer (1989) who suggested that changes to natural disturbance regimes might cause increases in woody plants. Bestelmeyer et al. (2006) showed that as the size of bare patches increased, aggregate stability decreased. Shaver (2010b) showed the results of herbicide treatment on these treated plots, although long lasting, were beginning to decline. This suggests that without the reintroduction of fire, the feedback mechanisms of increased organic matter inputs were not able to limit the increase or encroachment of one-seed juniper onto the site. The resilience of the Reference State was weakened and the processes of infiltration, nutrient cycle, aggregate stability and annual production were nearing a threshold into the Juniper State. Threshold values from the Reference State to the Juniper State for surface soil stability were between 3.4 and 2.8 and for subsurface soil stability at 1.8. Once the threshold was crossed, along the transition (T1a) the positive feedbacks for change become negative feedbacks strengthening the resilience of the Juniper State.

There were ten transects that appeared to constitute an at-risk community phase approaching a threshold from the Juniper State to the Eroded State. The average surface stability rating for these transects was 2.7 and the average subsurface stability rating was 1.5. The canopy gaps >200 cm were 26% and basal gaps >200 cm were 42%. While the soil stability values were within the Juniper State ranges, the vegetation gap values were well within the Eroded State ranges. It is unclear from the data how the feedback mechanisms maintain the resilience of the Juniper

State. Experience indicates that management actions intended to maintain or increase the production and cover of the short grasses are needed to control wind erosion and to ensure a level of organic matter inputs necessary to maintain soil aggregate stability. As gap size continued to increase due to nutrient pooling around one-seed juniper, soil aggregate stability decreased. Lower soil aggregate stability decreased infiltration which further decreased production and organic matter inputs. Larger gap sizes allowed for higher erosion rates from both wind and water. Indications based on the increase in vegetation gaps and bare ground coupled with decreases in both basal and herbaceous foliar cover (Shaver 2010b) support the crossing of an abiotic threshold where the physical processes of wind and water erosion are the feedback mechanisms that build the resilience in the Eroded State. Threshold values from the Juniper State to the Eroded State were canopy gaps >200cm and basal gaps >200cm both covering more 30% of the transect line with <4% basal cover of plants and 40% or more bare ground. The indication of crossing an abiotic threshold is supported by several authors who discussed the sequence of crossing biotic thresholds and as deterioration continues crossing abiotic thresholds (Westoby et al. 1989, Whisenant 1999, Petersen et al. 2009).

SUMMARY and CONCLUSIONS

Expert knowledge and k-means clustering were used to develop and test a state-and-transition model to organize our understanding of the dynamics of this ecological site. A model with three distinct states was developed with ranges in values for each of the data elements measured. Examining the state membership data showed patterns and inconsistencies in the state membership. When further examined with considerations of the ecological processes the model was refined. This refinement suggested composition and attributes of community phases at risk of losing resilience and transitioning across thresholds to an alternative state.

On this ecological site, the data suggest that the herbicide treatment was necessary to restore the ecological function of the site and enable the feedback mechanisms to develop resilience within the restored state. The data also suggest that although the feedback mechanisms have persisted for 18 years, without the re-establishment of one or more properties or processes that limit the re-establishment of junipers, (such as frequent ground fires), site resilience is not strong enough to maintain the system in the Reference State.

Surface soil stability proved to be a reliable indicator and predictor of state membership and provided indication of value ranges within states for itself and when combined with the other data elements, for those elements as well. The clear relationship between soil aggregate stability and gap size distribution is critical to understanding the feedback mechanisms responsible for site resilience. These relationships suggest that soil aggregate stability and gap size distribution can be incorporated into management systems and monitoring activities to ensure ecological function and vegetation structure in a state that provides the optimum in ecosystem services. This can provide decision makers the opportunity to manage the rangeland in a productive and sustainable manner. The understanding of resilience and the identification of feedback mechanisms and the change in dominance from negative to positive feedbacks is a powerful tool allowing managers to make decisions to maintain the desirable plant community function and structure before a threshold is crossed.

The identification and quantification of the components of a STM expands the ecological foundation of the STM concepts by linking them to process based feedback mechanisms and ecological resilience. The inclusion of resilience and feedbacks into the STM promotes adaptive management by focusing on indicators of the feedbacks responsible for resilience before thresholds are crossed. This provides managers with information to restore degraded sites, to ensure that feedback mechanisms are in place to maintain restoration and to make management decisions based on maintaining state integrity through ecological resilience. Process based STMs derived from data analysis and evaluation can be a useful tool in organizing data into state-and-transition models. However, professional experience and knowledge must be employed to interpret and understand relationships between the data, the ecological processes and management options.

LITERATURE CITED

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Allen, C. D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. Dissertation. University of California, Berkeley, California, USA.

Allen, C. D., and D. D. Breshears 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings, National Academy of Sciences of the United States of America. 95(25): 14839-14842.

Archer, S. 1989. Have southern Texas savannas been converted to woodland in recent history? American Naturalist 134: 545-561.

Arnold, J. F. 1964. Zonation of understory vegetation around a juniper tree. Journal of Range Management. 47: 41-42.

Arnold, J. F., D. A. Jameson and E. H. Reid. 1964. The pinyon-juniper type of Arizona; effects of grazing, fire and tree control. U. S. Dept. Agr. Prod. Res. Rep. No. 84. Washington, D. C.

Baisan, C. H., and T. W. Swetnam. 1997. Interactions of fire regimes and land use history in the central Rio Grande Valley. U.S. Forest Research Paper RM-RP-330.

Bandelier, A. 1884. The southwest journals of Adolph F. Bandelier, 1883 – 1884. Carroll L. Riley and Charles H. Lange Eds. University of New Mexico Press, Albuquerque, NM.

Bestelmeyer, B. T., J. R. Brown, K. M. Havstad, R. Alexander, G. Chavez and J. E. Herrick. 2003. Development and use of state-and-transition models for rangelands. Journal of Range Management 56: 114-126.

Bestelmeyer, B. T., J. E. Herrick, J. R. Brown, D. A. Trujillo and K. M. Havstad. 2004. Land Management in the American Southwest: A State-and-Transition Approach to Ecosystem Complexity. Environmental Management 34(1): 38-51.

Bestelmeyer, B. T., J. P. Ward, J. E. Herrick and A. J. Tugel. 2006. Fragmentation effects on soil aggregate stability in a patchy arid grassland. Rangeland Ecology and Management 59(4): 406–415.

Bestelmeyer, B. T., A. J. Tugel, G. L. Peacock Jr., D. G. Robinett, P. L. Shaver, J. R. Brown, J. E. Herrick, H. Sanchez and K. M. Havstad. 2009. State-and-transition models for heterogeneous landscapes: A strategy for development and application. Rangeland Ecology and Management 62(1): 1-15.

Bird, S. B., J. E. Herrick, M. M. Wander and S. F. Wright. 2002. Spatial heterogeneity of aggregate stability and soil carbon in semi-arid rangeland. Environmental Pollution 116: 445-455.

Bird, S. B., J. E. Herrick, M. M. Wander and L. Murry. 2007. Multi-scale variability in soil aggregate stability: Implications for understanding and predicting semi-arid grassland degradation. Geodermia 140: 106–118.

Breshears, D. D. and F. S. Barnes. 1999. Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landscapes within the grassland/forest continuum: a unified conceptual model. Landscape Ecology. 14(5): 465-478.

Briske, D. D., S. D. Fuhlendorf and F. E. Smiens. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management 58(1): 1–10.

Briske, D. D., S. D. Fuhlendorf, and F. E. Smiens. 2006. A unified framework for assessment and application of ecological thresholds. Rangeland Ecology and Management 59(3): 225–236.

Briske, D. D., B. T. Bestelmeyer, T. K. Stringham and P. L. Shaver. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management 61(4): 359–367.

Clary, W. P., and D. A. Jameson. 1981. Herbage production following tree and shrub removal in the Pinyon-Juniper type in Arizona. Journal of Range Management. 34: 109-113.

Davenport, D. W., B. P. Wilcox and D. D. Breshears. 1996. Soil morphology of canopy and intercanopy sites in piñon-juniper woodland. Soil Science Society of America Journal 60: 188–1887.

Denevan, W. M. 1967. Livestock numbers in nineteenth-century New Mexico, and the problem of gullying in the southwest. Annals of the Association of American Geographers 57(4): 691-703.

Dwyer, D. D. and R. D. Pieper. 1967. Fire effects on blue grama-pinyon-juniper rangeland in New Mexico. Journal of Range Management. 20(6): 359–362.

Dyksterhuis, E. J. 1949. Condition and management of range land based on quantitative ecology. Journal of Range Management 2(3): 104-115.

Friedel, M. H. 1991. Range condition assessment and the concept of thresholds: A viewpoint. Journal of Range Management 44(5): 422–426.

Frost, C. C. 1998. Presettlement fire frequency regimes of the United States: A first approximation. *In:* Teresa L. Pruden and Leornard A. Brennan (eds.). Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings, No. 20 Tall Timbers Research Station, Tallahassee, FL: pp 70-81.

Gill, R. A. and R. B. Jackson. 2000. Global patterns of root turnover for terrestrial ecosystems. New Phytol. 147: 13–31.

Goldberg, S., D. L. Suarez and R. A. Glaubig. 1988. Factors affecting clay dispersion and aggregate stability of arid-zone soils. Soil Science 146: 317-325.

Herrick, J. E., W. G. Whitford, A. G. de Soyza, J. W. Van Zee, K. M. Havstad, C. A. Seybold and M. Walton. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. Catena 44(1): 27–35.

Herrick, J. E., J. R. Brown, A. J. Tugel, P. L. Shaver and K. M. Havstad. 2002. Application of soil quality to monitoring and management. Agronomy Journal 94: 3-11.

Herrick, J. E., J. W. Van Zee, K. M. Havstad, L. M. Burkett and W. G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. UDSA-ARS Jornada Experimental Range, Las Cruces, NM. Distributed by University of Arizona Press.

Horgan, P. 1954. Great River. Reinhart Co. New York, NY.

Jameson, D. A. 1962. Effects of burning on a galleta-black grama range invaded by juniper. Ecology. 43(4): 760-763.

Jameson, D. A. 1967. The relationship of tree overstory and herbaceous understory vegetation. Journal of Range Management. 20: 247-249.

Jameson, D. A. 1970. Juniper root competition reduces basal area of blue grama. Journal of Range Management. 23: 217-218.

Johnsen, T. N., Jr. 1962. One-seeded juniper invasion of northern Arizona grasslands. Ecological Monographs. 32(3): 187-207.

Kemper, W. D. and E. J. Koch. 1966. Aggregate stability of soils from western United States and Canada. USDA-ARS Tech. Bull., vol.1355. U.S. Govt Print. Office, Washington D.C.

Lanner, R. M. and T. R. Van Devender. 1998. The recent history of pinyon pines in the American Southwest. *In:* Richardson, David M., ed. Ecology and biogeography of Pinus. Cambridge, United Kingdom: The Press Syndicate of the University of Cambridge: pp. 171-182.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North American rangelands: A viewpoint. Journal of Range Management 44(5): 427–433.

McCune, B. and J. A Grace. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach OR.

McIntyre, S. and D. Tongway. 2005. Grassland structure in native pastures: links to soil surface condition. Ecological Management and Restoration 6 (1): 43–50.

McLeullough, G. T. 1882. Field notes of the survey of the sub-division of T.1N., R.8E., of the principle base meridian in New Mexico. Territorial Survey. (survey commenced September 7, 1882 and concluded September 13, 1882).

Miller, R. F., and P. E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. Bioscience. 44(7): 465-474.

Padien, D. J. and K. Lajtha. 1992. Plant spatial pattern and nutrient distribution in pinyon-juniper woodlands along an elevational gradient in northern New Mexico. International Journal of Plant Science. 153(3): 425-433.

Petersen, S. L., T. K. Stringham and B. A. Roundy. 2009. A process-based application of state-and-transition models: A case study of Western Juniper (Juniperus occidentalis) encroachment. Rangeland Ecology and Management. 62: 186-192.

Peterson, G., C. R. Allen and C. S. Holling. 1998. Ecological resilience, biodiversity, and scale. Ecosystems 1: 6-18.

Pieper, R. D. 1983. Overstory-understory relationships: Pinyon-juniper and juniper woodlands. *In:* Bartlett, E. T. and D. R. Betters (Eds.) Overstory-understory relationships in western forests. West. Reg. Res. Pub. No. 1. Colo. State Univ. Agr. Exp. Sta. Ft. Collins, CO.

Reid, K. D., B. P. Wilcox, D. D. Breshears and L. MacDonald. 1999. Runoff and erosion in a Piñon-Juniper woodland: Influences of vegetation patches. Soil Science Society of America Journal. 63: 1869–1879

Savage, M., and T. W. Swetnam. 1990. Early 19th-century fire decline following sheep pasturing in a Navajo Ponderosa pine forest. Ecology 71:2374-2378.

Schickedanz, J. G. 1980. History of grazing in the Southwest. *In:* McDaniel, K.C. and C. Allison [EDS]. Grazing management systems for southwest rangelands. A symposium. Range Improvement Task Force. New Mex. State Univ., Las Cruces, NM.

Schlesinger, W. H., J. F.. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virgina and W. G. Whitford. 1990. Biological feedbacks in global desertication. Science 247: 1043-1048.

Shaver, P. L. 2010a. Short term response of soil moisture and vegetation to elimination of one-seed juniper canopy cover. *In:* Shaver, Patrick L. Quantification of State-and-Transition Model components utilizing long-term ecological response data following one-seed juniper treatment on a Deep Sand Savannah ecological site. PhD Dissertation, Oregon State University, Corvallis, OR.

Shaver, P. L. 2010b. Long term response of soil moisture and vegetation to the reduction of one-seed juniper canopy cover. *In:* Shaver, Patrick L. Quantification of State-and-Transition Model components utilizing long-term ecological response data following one-seed juniper treatment on a Deep Sand Savannah ecological site. PhD Dissertation, Oregon State University, Corvallis, OR.

StatSoft, Inc. (2007). Electronic Statistics Textbook. Tulsa, OK: StatSoft. WEB: http://www.statsoft.com/textbook/stathome.html.

StatSoft, Inc. (2005). STATISTICA (data analysis software system), version 7.1. www.statsoft.com.

Stewart, O. C. 2002. The effects of burning of grasslands and forests by aborigines the world over. The mountain west. *In:* Henery T. Lewis and M.Kat Anderson [EDS.]. Forgotten Fires. Norman, OK. University of Oklahoma Press. pp. 217-224.

Stringham, T. K., W. C. Krueger and P. L. Shaver. 2003. State-and-transition modeling: an ecological process approach. Journal of Range Management 56: 106-113.

Swetnam, T. W., C. Baisan, and J. M. Kaib, editors. In Press. Forest fire histories of La Frontera: fire-scar reconstructions of fire regimes in the United States/Mexico borderlands. Univeristy of New Mexico.

Tisdall, J. M. and J. M. Oades. 1982. Organic matter and water stable aggregates in soils. Journal of Soil Science. 33: 141-163.

Topp, G. C., W. D. Reynolds, F. J. Cook, J. M. Kirby and M. R. Carter. 1996. Physical attributes of soil quality. *In:* Gregorich, E. G.., M.R. Carter, (Eds), Soil quality for crop production and ecosystem health. Elsevier, Amsterdam, pp. 21-58.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2003. National range and pasture handbook, revision 1. Washington, D.C.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2004. Ecological site descriptions. Field office technical guide, section II. Albuquerque, NM.

U.S. Department of Agriculture, Natural Resources Conservation Service. 2006. Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin. United States Department of Agriculture Handbook 296. Washington D.C.

U.S. Department of Agriculture. Soil Conservation Service. 1970. Soil survey Torrence area, New Mexico. Washington, D.C.

U.S. Department of Agriculture. Soil Conservation Service. 1988. Soil survey Socorro area, New Mexico. Washington, D.C.

Vivian, G. 1961. Excavations in a 17th century Jumano pueblo, Gran Quivira, New Mexico. Archeological Research Series No. 8. National Park Service. USDI, Washington, DC.

Weltzin, J. F. and G. R. McPherson. 1997. Spatial and temporal soil moisture resource partitioning by trees and grasses in a temperate savanna, Arizona, USA. Oecologia 112: 156-164.

Western Regional Climate Center. 2009. New Mexico Climate Summaries, Gran Quivira National Monument, NM. http://www.wrcc.dri.edu/summary/climsmnm.html.

Westoby, M., B. Walker and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42(4): 266-274.

Whisenant, S. G. 1999. Repairing Damaged Wildlands: biological conservation, restoration, and sustainability. Cambridge, United Kingdom: Cambridge University Press. 312p.

White, L. D. 1965. The effects of a wildfire on a desert grassland community. [thesis]. Tucson, AZ: University of Arizona. 107p.

Wright, H. A. 1990. Role of fire in the management of Southwestern ecosystems. P. 1–5 in J. S. Krammes, technical coordinator. Effects of Fire Management of Southwestern Natural Resources. General Technical Report RM-191. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, CO. 293p.

CHAPTER 5: GENERAL CONCLUSION

The quantitative climax model of plant community dynamics has been used for more than 50 years in the management of rangelands. It has not been effective in providing managers with timely options for restoration of many ecosystems. State-and-transition models (STM) are becoming the decision support tool of choice for management options, especially when dealing with invasive plant encroachment into grassland ecosystems. Currently, most STM are based on experience and professional knowledge with little data to support the descriptions of plant community pathways and transitions. In an effort to quantify the components of a processed based STM, the effects of the manipulation of one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg) on selected soil and vegetation variables were studied over the course of 22 years.

Short term response in available soil moisture and vegetation production and composition were collected from small plots treated with the herbicide tebuthiuron in 1981. Soil moisture data were collected at four depths during the growing season of 1984 and vegetation production and composition data were collected from 1981 – 1986. Analysis showed that the soil moisture in the treated plots was significantly ($\alpha=0.05$) higher throughout the season. The available soil moisture in the treated profile was consistently above the 1.5 MPa and for most of the season contained twice as much available water as the control plots. These differences were associated with reductions in woody vegetation and increases in herbaceous production. The vegetation measurements contained no replicates, therefore no statistical analysis was done. There were large differences, however, in the measured production and composition between the treated and control plots.

Data for long term responses were collected from plots in field size application of the herbicide. Treatment was applied in 1985 and soil moisture, annual herbaceous production and vegetative cover were measured from 1985 – 1989. In 2003, these vegetation measurements were repeated along with additional vegetation attributes and soil aggregate stability. Soil moisture data indicated that the treated plots consistently contained significantly (α=0.05) more available moisture than the control plots. This was especially evident during the drought year of 1989. The treated plots were significantly (α=0.05) different from the control plots for all the vegetation and soil data elements. Annual herbaceous production was negatively correlated to juniper canopy cover (r^2=0.86) and bare ground and juniper canopy cover were positively correlated (r^2=0.89). Litter was positively correlated to annual herbaceous production (r^2=0.79) and bare ground was negatively correlated (r^2=0.78). These correlations clearly indicate that as juniper canopy cover decreased, herbaceous production increased. As annual herbaceous production increased, there was a corresponding increase in organic matter being added to the soil. When the 2003 data were added to the analysis, the results showed that the treatment effect was still significant 18 years following treatment. The negative correlation between annual herbaceous production and juniper canopy remained strong (r^2=0.85). The negative correlation with annual herbaceous production and bare ground weakened (r^2=0.59) as did the positive correlation between annual herbaceous production and litter cover (r^2=0.51).

Analysis of the expanded study data also showed that the treatment effects lasted for the 18 years of the study. Eighteen years after treatment, data analysis indicated significantly (α=0.05) different treatment effects for bare ground, total litter cover, total ground cover and protected subsurface soil stability. Juniper foliar cover, aggregate stability of all surfaces, of unprotected surfaces and of unprotected subsurface, all showed significant (α=0.05) ranch by treatment interactions. The linear regression models for the expanded study data showed several expected correlations. Foliar cover and total ground cover were both negatively correlated to bare ground (r^2=0.66, r^2=0.77 respectfully).

There were also several weak (r^2<0.30) but statistically significant (α=0.05) relationships. When surface soil stability was the dependent variable and seven other variables were used as the independent variables the resulting relationship (R^2=0.62) was highly significant (p<0.0000). These results suggest that the soil aggregate stability data elements provided the best integrator of long term ecological responses to change in vegetation production, cover, bare ground, litter accumulation and bare patch size. They also suggest that soil aggregate stability may provide insight into the ecological dynamics of this ecological site.

The increase in soil aggregate stability in the treated sites suggests that the vegetation changes resulting from the treatment improved the organic matter inputs into the system and the resulting nutrient cycle. Improved soil aggregate stability has been associated with improved water infiltration, reduced evaporation and erosion, and a resultant feedback for improved herbaceous production and soil aggregate stability. The 2003 data also suggest that the effects of the herbicide treatment are nearing the end of their lifespan. These changes have important implications for the development of STM and for management aimed at maintaining or improving the ecological and economic sustainability of rangelands.

A proposed STM was verified and validated using the data from the short term, long term and expanded studies. Surface soil stability proved to be a reliable indicator and predictor of state membership. Surface soil stability also provided indication of value ranges within states for itself and when combined with the other data elements, for those elements as well. The understanding of the clear relationship between soil aggregate stability and gap size distribution is critical to identifying feedback mechanisms. These relationships also suggest that these variables can be used to develop an ecological process based STM supported by field data. This relationship was used to identify and quantify a matrix of state characteristics and threshold values between states in the STM (Table 5.1).

Table 5.1. Summary of data values within 95% confidence interval of the mean value for variables in state descriptions.

	Reference State 1.0	Juniper State 2.0	Eroded State 3.0
Surface Soil Stability	>4.0	2.6 – 3.0	< 2.4
Subsurface Soil Stability	>2.4	1.6 – 1.8	< 1.6
Canopy Gaps > 200cm	<10%	10% - 20%	> 28%
Basal Gaps > 200cm	< 17%	17% - 29%	> 33%
Basal Cover	>7%	5% - 9%	< 4%
Juniper Foliar Cover	<17%	18% - 27%	>20%
Herb. Foliar Cover	>45%	>45%	<41%
Bare Ground	<33%	28% - 37%	>39%

At-risk community phases were also identified and quantified with value ranges for the indicators (Table 5.2).

Table 5.2. Summary of data values within 95% confidence interval of the mean value for community phases within states.

State	Reference State		Juniper State		Eroded State
Community Phase	1.1	1.2	2.1	2.2	3.1
Surface Soil Stability	>4.3	3.4 – 4.1	2.5 – 2.8	2.4 – 2.8	<2.1
Subsurface Soil Stability	>2.7	1.8 – 2.1	1.5 – 1.8	1.2 – 1.8	<1.5
Canopy Gaps > 200cm	<8%	12 – 27%	7 – 13%	18 – 33%	>29%
Basal Gaps > 200cm	<15%	15 – 36%	12 – 25%	29 – 55%	>30%
Basal Cover	>7%	5 – 9%	5 – 10%	<4%	<4%
Juniper Foliar Cover	<8%	11 – 24%	18 – 28%	16 – 32%	>29%
Herb. Foliar Cover	>46%	40 – 54%	47 – 57%	32 – 46%	<30%
Bare Ground	<32%	24 – 42%	27 – 37%	33 – 47%	>39%

These ranges in values provide evidence for threshold identification in the STM. They also provide indication of the feedbacks involved in changing resilience from one state to another. State resilience is weakened as one-seed juniper increases, reducing available soil moisture and annual herbaceous production. As herbaceous production is decreased, gap size increased and organic matter input decreased. Reduced organic matter inputs weaken soil aggregate stability making the soil more susceptible to erosion from wind and water. Once the threshold is crossed into the new state, these same feedback mechanisms act to create resilience in the new state. The data suggests that the herbicide treatment was necessary to restore the ecological function of the site and enable the feedback mechanisms to develop resilience within the restored state. The data also suggest that without reestablishing the disturbance of a ground fire as part of the feedback mechanism to limit the increase in one-seed juniper, the resilience was not strong enough to resist the reinvasion of one-seed juniper into the system. The resulting STM diagram is shown in Figure 5.1.

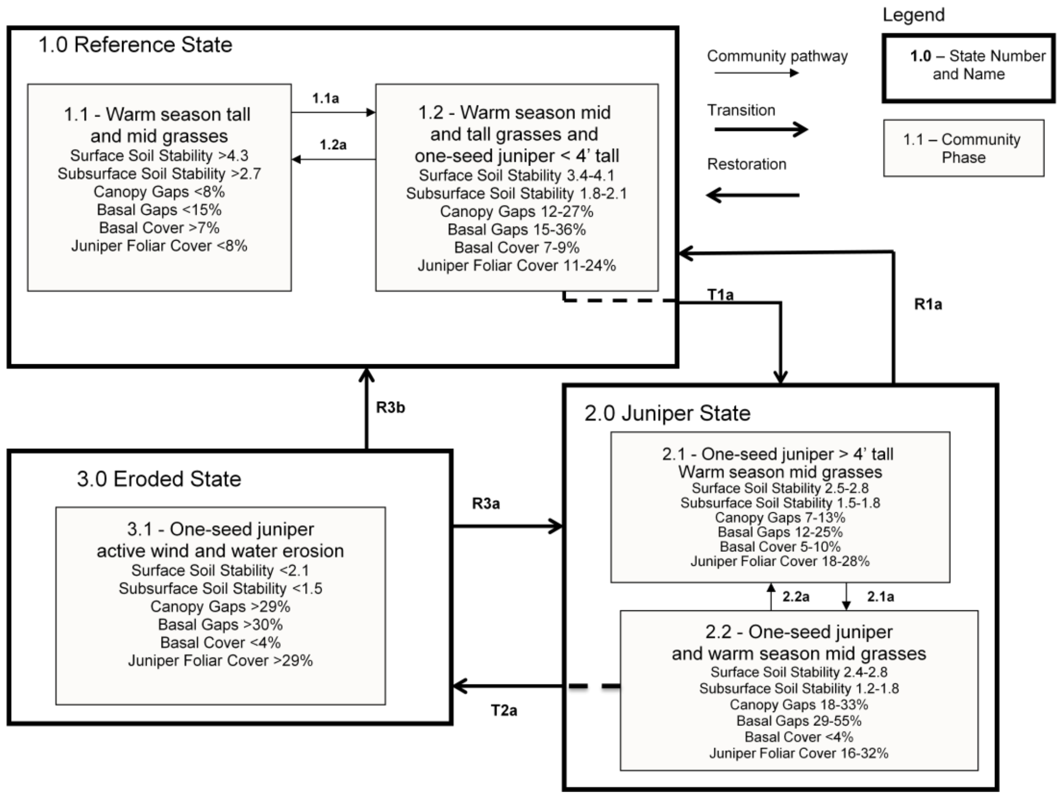

Figure 5.1. State-and-transition model diagram using data elements to identify state boundaries and at-risk community phases.

The identification and quantification of the components of a STM expands the ecological foundation of the STM concepts by linking them to process based feedback mechanisms and ecological resilience. The inclusion of resilience and feedbacks into the STM promotes adaptive management by focusing on the indicators of feedback mechanisms responsible for resilience before thresholds are crossed. This provides managers with information needed to restore degraded sites, to ensure that feedback mechanisms are in place to maintain restoration, and to make management decisions based on maintaining state integrity through ecological resilience.

BIBLIOGRAPHY

Allen, C. D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. Dissertation. University of California, Berkeley, California, USA.

Allen, C. D., and D. D. Breshears 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings, National Academy of Sciences of the United States of America. 95(25): 14839-14842.

Allen, C. D., J. L. Betancourt and T. W. Swetnam. 1998. Landscape changes in the southwestern United States: techniques, long-term data sets, and trends. *In:* T. D. Sisk [ED.]. Perspectives on the land use history of North America: a context for understanding our changing environment. Reston, VA, USA: US Geological Survey, Biological Science Report USGS/BRD/BSR-1998-0003. pp. 71-84.

Allen, R. B. and R. K. Peet. 1990. Gradient analysis of forests of the Sangre de Cristo Range, Colorado. Canadian Journal of Botany. 68: 193-201.

Allison, C. D. 1982. A simple range trend monitoring procedure. New Mexico State University Cooperative Extension Service. 400 B-2. Las Cruces, NM.

Archer, S. 1989. Have southern Texas savannas been converted to woodland in recent history? American Naturalist 134: 545-561.

Arnold, J. F. 1964. Zonation of understory vegetation around a juniper tree. Journal of Range Management. 47: 41-42.

Arnold, J. F., D. A. Jameson and E. H. Reid. 1964. The pinyon-juniper type of Arizona; effects of grazing, fire and tree control. U. S. Dept. Agr. Prod. Res. Rep. No. 84. Washington, D. C.

Baisan, C. H., and T. W. Swetnam. 1997. Interactions of fire regimes and land use history in the central Rio Grande Valley. U.S. Forest Research Paper RM-RP-330.

Balda, R. P. 1987. Avian impacts on pinyon-juniper woodlands. *In:* Everett, Richard L., compiler. Proceedings--pinyon-juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 525-533. [4993]

Bandelier, A. 1884. The southwest journals of Adolph F. Bandelier, 1883 – 1884. Carroll L. Riley and Charles H. Lange Eds. University of New Mexico Press, Albuquerque, NM.

Barnitz, J. A. Jr., S. M. Armentrout, V. W. Howard, Jr., R. D. Pieper, and G. M. Southward. 1990. Vegetational changes following two-way cabling of pinyon-juniper in south-central New Mexico. New Mexico Agr. Exp. Sta. Bull. 749. 36p.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2000. Understory dynamics in cut and uncut western juniper woodlands. Journal of Range Management 53:119-126.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2007. Long-term vegetation dynamics in a cut western juniper woodland. Western Norrth American Naturalist 67:549-561.

Bestelmeyer, B. T., J. R. Brown, K. M. Havstad, R. Alexander, G. Chavez, and J.E. Herrick. 2003. Development and use of state-and-transition models for rangelands. Journal of Range Management 56: 114-126.

Bestelmeyer, B. T., J. E. Herrick, J. R. Brown, D. A. Trujillo and K. M. Havstad. 2004. Land Management in the American Southwest: A State-and-Transition Approach to Ecosystem Complexity. Environmental Management 34(1): 38-51.

Bestelmeyer, B. T., J. P. Ward, J. E. Herrick and A. J. Tugel. 2006. Fragmentation effects on soil aggregate stability in a patchy arid grassland. Rangeland Ecology and Management 59(4): 406-415.

Bestelmeyer, B. T., A. J. Tugel, G. L. Peacock Jr., D. G. Robinett, P. L. Shaver, J. R. Brown, J. E. Herrick, H. Sanchez and K. M. Havstad. 2009. State-and-transition models for heterogeneous landscapes: A strategy for development and application. Rangeland Ecology and Management 62(1): 1-15.

Betancourt, J. L. 1987. Paleoecology of pinyon-juniper woodlands: summary. *In*: R. L. Everett [ED]. Proceedings-Pinyon-Juniper Conference; 13 -16 January 1986; Reno, NV, USA. Ogden, UT, USA: USDA, Forest Seervice, Intermountain Research Station, GTR INT-215. pp. 129-139.

Bird, S. B., J. E. Herrick, M. M. Wander and S. F. Wright. 2002. Spatial heterogeneity of aggregate stability and soil carbon in semi-arid rangeland. Environmental Pollution 116: 445-455.

Bird, S. B., J. E. Herrick, M. M. Wander and L. Murry. 2007. Multi-scale variability in soil aggregate stability: Implications for understanding and predicting semi-arid grassland degradation. Geodermia 140: 106-118.

Breshears, D. D., O. B. Myers, S. R. Johnson, C. W. Meyer and S. N. Martens. 1997a. Differential use of spatially heterogeneous soil moisture by two semiarid woody species: *Pinus edulis* and *Juniperius monosperma*. Journal of Ecology 85(3): 289–299.

Breshears, D. D., P. M. Rich, F. J. Barnes and K. Campbell. 1997b. Overstory-imposed heterogeneity in solar radiation and soil moisture in a semiarid woodland. Ecological Applications 7(4): 1201-1215.

Breshears, D. D. and F. S. Barnes. 1999. Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landscapes within the grassland/forest continuum: a unified conceptual model. Landscape Ecology. 14(5): 465-478.

Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice, W. H. Romme, J. H. Kastens, M. L. Floyd, J. Belnap, J. J. Anderson, O. B. Myers and C. W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences 102(42): 15144-15148.

Briggs, J. M., H. Schaafsma and D. Trenkov. 2007. Woody vegetation expansion in a desert grassland: prehistoric human impact? Journal of Arid Environments 69: 458-472.

Briske, D. D., S. D. Fuhlendorf and F. E. Smiens. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management 58(1): 1-10.

Briske, D. D., S. D. Fuhlendorf and F. E. Smiens. 2006. A unified framework for assessment and application of ecological thresholds. Rangeland Ecology and Management 59(3): 225-236.

Briske, D. D., B. T. Bestelmeyer, T. K. Stringham and P. L. Shaver. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management 61(4): 359-367.

Brown, J. H., T. J. Valone and C. G. Curtin. 1997. Reorganization of an arid ecosystem in response to recent climate change. Proceedings of the national Academy of Sciences 94(18): 9729-9733.

Brown, J. H., T. G. Whitham, S. K. Morgan Ernest and C. A. Gehring. 2001. Complex species interactions and the dynamics of ecological systems: long-term experiments. Science. 293(5530): 643-650.

Clary, W. P. and D. A. Jameson. 1981. Herbage production following tree and shrub removal in the Pinyon-Juniper type in Arizona. Journal of Range Management. 34: 109-113.

Clements, F. E. 1928. Plant succession and indicators. The H.W. Wilson Co., New York City, NY.

Cook, C. W. and C. D. Bonham. 1977. Techniques for vegetation measurements and analysis for a pre-and post-mining inventory. Colo. State Univ. Range Sci. Dept. Sci. Ser. No. 28. Ft. Collins, CO.

Davenport, D. W., B. P. Wilcox and D. D. Breshears. 1996. Soil morphology of canopy and intercanopy sites in piñon-juniper woodland. Soil Science Society of America Journal 60: 1881-1887.

Denevan, W. M. 1967. Livestock numbers in nineteenth-centry New Mexico, and the problem of gullying in the southwest. Annals of the Association of American Geographers 57(4): 691-703.

Dunnett, C. W. 1980. Pairwise Multiple Comparisons in the Homogeneous Variance, Unequal Sample Size Case. Journal of the American Statistical Association. 75(372): 789-795.

Dwyer, D. D. and R. D. Pieper. 1967. Fire effects on blue grama-pinyon-juniper rangeland in New Mexico. Journal of Range Management. 20(6): 359-362.

Dyksterhuis, E. J. 1949. Condition and management of range land based on quantitative ecology. Journal of Range Management 2(3): 104-115.

Dyksterhuis, E. J. 1958. Range conservation as based on sites and condition classes. Journal of Soil and Water Conservation 13: 151-155.

Fletcher, J. E. and W.P. Martin. 1948. Some effects of algae and molds in the raincrust of desert soils. Ecology 29: 95-100.

Floyd, M. L., W. H. Romme and D. Hanna. 2000. Fire history and vegetation pattern in Mesa Verde National Park. Ecological Applications 10: 1666-1680.

Fowler, J. M., B. E. Peacock and M. J. Schaber. 1985. Pinyon-juniper woodland type in New Mexico: Asset or liability. New Mex. State Univ. Agr. Exp. Sta. Bull. 718. Las Cruces, NM.

Foxx, T. S. and G. D. Tierney. 1987. Rooting patterns in the pinyon-juniper woodland. *In:* Everett, Richard L., compiler. Proceedings--pinyon-juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: pp. 69-79.

Friedel, M. H. 1991. Range condition assessment and the concept of thresholds: A viewpoint. Journal of Range Management 44(5): 422-426

Friedman, M. 1937. The use of ranks to avoid the assumption of normality implicit in the analysis of variance. Journal of the American Statistical Association 32 (200): 675–701.

Frost, C. C. 1998. Presettlement fire frequency regimes of the United States: A first approximation. *In:* Teresa L. Pruden and Leornard A. Brennan (eds.). Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings, No. 20 Tall Timbers Research Station, Tallahassee, FL: pp. 70 -81.

Fuhlendorf, S. D., F. E. Smeins and W. E. Grant. 1996. Simulation of a fire-sensitive ecological threshold: A case study of Ashe juniper on the Edwards Plateau of Texas, USA. Ecological Modeling 90: 245-255.

Fuhlendorf, S. D. and F. E. Smeins. 1997. Long-term vegetation dynamics mediated by herbivores, weather and fire in a *Juniperus-Quercus* savanna. Journal of Vegetation Science 8: 819-828.

Gardner, W. H. 1986. Water content. Klute, A.(ed.). Methods of soil analysis. Part I. pp. 493-544. Madison, WI, ASA and SSSA. Agron. Monogr.

Gifford, G. F. 1973. Runoff and sediment yields from runoff plots on chained pinyon-juniper plots in Utah. Journal of Range Management. 26: 440-443.

Gill, R. A. and R. B. Jackson. 2000. Global patterns of root turnover for terrestrial ecosystems. New Phytol. 147: 13 – 31.

Goldberg, S., D. L. Suarez, and R. A. Glaubig. 1988. Factors affecting clay dispersion and aggregate stability of arid-zone soils. Soil Science 146, 317-325.

Gottfried, G. J. 1999. Pinyon-juniper woodlands in the southwestern United States. *In:* Ffolliott, Peter F.; Ortega-Rubio, Alfredo, eds. Ecology and management of forests, woodlands, and shrublands in the dryland regions of the United States and Mexico: perspectives for the 21st century. Co-edition No. 1. Tucson, AZ: The University of Arizona; La Paz, Mexico: Centro de Investigaciones Biologicas del Noroeste, SC; Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: pp. 53-67.

Herman, F. R. 1956. Growth and phenological observations of Arizona junipers. Ecology. 37: 193-195.

Herrick, J. E., W. G. Whitford, A. G. de Soyza, J. W. Van Zee, K. M. Havstad, C. A. Seybold and M. Walton. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. Catena 44(1) 27-35.

Herrick, J. E., J. R. Brown, A. J. Tugel, P. L. Shaver and K. M. Havstad. 2002. Application of soil quality to monitoring and management. Agronomy Journal 94: 3-11.

Herrick, J. E., J. W. Van Zee, K. M. Havstad, L. M. Burkett and W. G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. UDSA-ARS Jornada Experimental Range, Las Cruces, NM. Distributed by University of Arizona Press.

Hocking, R. R. 1985. The analysis of linear models. Monterey, CA: Brooks/Cole.

Hocking R. R., F.M. Speed and A.T. Coleman. 1980. Hypotheses to be tested with unbalanced data. Communications in Statistics - Theory and Methods, 9(2): 117-129.

Horgan, P. 1954. Great River. Reinhart Co. New York, NY.

Jameson, D. A. 1962. Effects of burning on a galleta-black grama range invaded by juniper. Ecology. 43(4): 760-763.

Jameson, D. A. 1965. Arrangement and growth of pinyon and one-seed juniper trees. Plateau. 37: 121-127.

Jameson, D. A. 1967. The relationship of tree overstory and herbaceous understory vegetation. Journal of Range Management. 20: 247-249.

Jameson, D. A. 1970. Juniper root competition reduces basal area of blue grama. Journal of Range Management. 23: 217-218.

Johnsen, T. N., Jr. 1962. One-seeded juniper invasion of northern Arizona grasslands. Ecological Monographs. 32(3): 187-207.

Johnsen, T. N., Jr. and R. A. Alexander. 1974. Juniperus L. juniper. *In:* Schopmeyer, C. S., tech. coord. Seeds of woody plants in the United States. Agric. Handb. 450. Washington, DC: U.S. Department of Agriculture, Forest Service: pp. 460-469.

Kemper, W. D. and E. J. Koch. 1966. Aggregate stability of soils from western United States and Canada. USDA-ARS Tech. Bull., vol.1355. U.S. Govt Print. Office, Washington D.C.

Lanner, R. M, and T. R. Van Devender. 1998. The recent history of pinyon pines in the American Southwest. *In:* Richardson, David M., ed. Ecology and biogeography of Pinus. Cambridge, United Kingdom: The Press Syndicate of the University of Cambridge: pp. 171-182.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North American rangelands: A viewpoint. Journal of Range Management 44(5): 427-433.

Levene, H. 1960. Robust tests for equality of variances.*In* Ingram Olkin, Harold Hotelling et al. Contributions to Probability and Statistics: Essays in Honor of Harold Hotelling. Stanford University Press: pp. 278-292.

McCune, B. and J. A Grace. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach OR.

McIntyre, S. and D. Tongway, 2005. Grassland structure in native pastures: links to soil surface condition. Ecological Management and Restoration 6 (1): 43-50.

McLeullough, G. T. 1882. Field notes of the survey of the sub-division of T.1N., R.8E., of the principle base meridian in New Mexico. Territorial Survey. (survey commenced September 7, 1882 and concluded September 13, 1882).

Miller, R. F., and P. E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. Bioscience. 44(7): 465-474.

Miller, R. F., T. J. Svejcar, and J. A. Rose. 2000. Impacts of western juniper on plant community composition and structure. Journal of Range Management 53:574-585.

Mueggler, W. F. 1976. Ecological role of fire in western woodland and range ecosystems. *In:* Use of prescribed burning in western woodland and range ecosystems: Proceedings of the symposium; 1976 March 18-19; Logan, UT. Logan, UT: Utah State University, Utah Agricultural Experiment Station: pp. 1-9.

Mueller, R. C., C. M. Scudder, M. E. Porter, R. T. Trotter III, C. A. Gehring and T. G. Whitham. 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. Journal of Ecology 93(6): 1085-1093.

Newman, B. D., A. R. Campbell and B. P. Wilcox. 1997. Tracer based studies of soil water movement in semi-arid forests of New Mexico. Journal of Hydrology. 196: 251-270.

Padien, D. J. and K. Lajtha. 1992. Plant spatial pattern and nutrient distribution in pinyon-juniper woodlands along an elevational gradient in northern New Mexico. International Journal of Plant Sciences. 153(3), Part 1: pp. 425-433.

Petersen, S. L. and T. K. Stringham. 2008. Infiltration, runoff, and sediment yield in response to western juniper encroachment in southeast Oregon. Rangeland Ecology and Management. 61: 74-81.

Petersen, S. L., T. K. Stringham and B. A. Roundy. 2009. A process-based application of state-and-transition models: A case study of Western Juniper (Juniperus occidentalis) encroachment. Rangeland Ecology and Management. 62: 186-192.

Peterson, G., C. R. Allen and C. S. Holling. 1998. Ecological resilience, biodiversity, and scale. Ecosystems 1: 6-18.

Pieper, R. D. 1977. The southwestern pinyon-juniper ecosystem. *In:* Aldon, Earl F.; Loring, Thomas J., technical coordinators. Ecology, uses, and management of pinyon-juniper woodlands: Proceedings of the workshop; 1977 March 24-25; Albuquerque, NM. Gen. Tech. Rep. RM-39. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: pp. 1-6. [17251]

Pieper, R. D. 1983. Overstory-understory relationships: Pinyon-juniper and juniper woodlands. *In:* Bartlett, E. T. and D. R. Betters (Eds.) Overstory-understory relationships in western forests. West. Reg. Res. Pub. No. 1. Colo. State Univ. Agr. Exp. Sta. Ft. Collins, CO.

Pierson, F. B., J. D. Bates, T. J. Svejcar, and S. P. Hardegree. 2007. Runoff and erosion after cutting western juniper. Rangeland Ecology and Management 60:285-292.

Rees, M., R. Condit, M. Crawley, S. Pacala and D. Tilman. 2001. Long-term studies of vegetation dynamics. Science. 293(5530): 650-655.

Reid, K. D., B. P. Wilcox, D. D. Breshears and L. MacDonald. 1999. Runoff and erosion in a Piñon-Juniper woodland: Influences of vegetation patches. Soil Science Society of America Journal. 63: 1869–1879.

Romme, W. H., C. D. Allen, J. D. Bailey, W. L. Baker, B. T. Bestelmeyer, P. M Brown, K. S. Eisenhart, M. L. Floyd, D. W. Huffman, B. F. Jacobs, R. F. Miller, E. H. Muldavin, T. W. Swetnam, R. J. Tausch and P. J. Weisberg. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203-222.

Roundy, B. A. and J. L. Vernon. 1999. Watershed values and conditions associated with pinon-juniper communities. *In:* Proceedings: Ecology and Management of Pinon-Juniper Communities Within the Interior West. USDA Forest Service. Proceedings RMRS-P-9. pp. 173-187.

Salomonson, M. G. 1978. Adaptations for animal dispersal of one-seed juniper seeds. Oecologia. 32: 333-339.

Sampson, A. W. 1919. Plant succession in relation to range management. USDA bulletin 791.

Savage, M., and T. W. Swetnam. 1990. Early 19th-century fire decline following sheep pasturing in a Navajo Ponderosa pine forest. Ecology 71:2374-2378.

Schickedanz, J. G. 1980. History of grazing in the Southwest. *In:* McDaniel, K.C. and C. Allison [EDS]. Grazing management systems for southwest rangelands. A symposium. Range Improvement Task Force. New Mex. State Univ., Las Cruces, NM.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virgina and W. G. Whitford. 1990. Biological feedbacks in global desertication. Science 247: 1043-1048.

Schott, M. R. and R. D. Pieper. 1986. Succession in pinyon-juniper vegetation in New Mexico. Rangelands. 8(3): 126-128.

Schott, M. R. and R. D. Pieper. 1987. Succession in tree pits following cabling in pinyon-juniper communities. The Southwestern Naturalist. 32(3): 399-402.

Shaver, P. L. 2010a. Short term response of soil moisture and vegetation to elimination of one-seed juniper canopy cover. *In:* Shaver, Patrick L. Quantification of State-and-Transition Model components utilizing long-term ecological response data following one-seed juniper treatment on a Deep Sand Savannah ecological site. PhD Dissertation, Oregon State University, Corvallis, OR.

Shaver, P. L. 2010b. Long term response of soil moisture and vegetation to the reduction of one-seed juniper canopy cover. *In:* Shaver, Patrick L. Quantification of State-and-Transition Model components utilizing long-term ecological response data following one-seed juniper treatment on a Deep Sand Savannah ecological site. PhD Dissertation, Oregon State University, Corvallis, OR.

StatSoft, Inc. (2005). STATISTICA (data analysis software system), version 7.1. www.statsoft.com.

StatSoft, Inc. (2007). Electronic Statistics Textbook. Tulsa, OK: StatSoft. WEB: http://www.statsoft.com/textbook/stathome.html.

Stewart, O. C. 2002. The effects of burning of grasslands and forests by aborigines the world over. The mountain west. *In:* Henery T. Lewis and M.Kat Anderson [EDS.]. Forgotten Fires. Norman, OK. University of Oklahoma Press. pp. 217-224.

Stringham, T. K., W. C. Krueger and P. L. Shaver. 2003. State-and-transition modeling: an ecological process approach. Journal of Range Management 56: 106-113.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American southwest. Journal of Climate. 11: 3128-3147.

Swetnam, T. W., C. Baisan, and J. M. Kaib, editors. In Press. Forest fire histories of La Frontera: fire-scar reconstructions of fire regimes in the United States/Mexico borderlands. Univeristy of New Mexico.

Tansley, A. G. 1935. The use and abuse of vegetational concepts and terms. Ecology 16: 284-307.

Tisdall, J. M. and J. M. Oades. 1982. Organic matter and water stable aggregates in soils. Journal of Soil Science. 33: 141-163.

Topp, G. C., W. D. Reynolds, F. J. Cook, J. M. Kirby and M. R. Carter. 1996. Physical attributes of soil quality. *In:* Gregorich, E. G.., M.R. Carter, (Eds), Soil quality for crop production and ecosystem health. Elsevier, Amsterdam, pp. 21-58.

Tueller, P. T. and J. E. Clark. 1975. Autecology of pinyon-juniper species of the Great Basin and Colorado Plateau. *In:* The pinyon-juniper ecosystem: a symposium: Proceedings; 1975 May; Logan, UT. Logan, UT: Utah State University, College of Natural Resources, Utah Agricultural Experiment Station: pp. 27-40.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2003. National range and pasture handbook, revision 1. Washington, D.C.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2004. Ecological site descriptions. Field office technical guide, section II. Albuquerque, NM.

U.S. Department of Agriculture, Natural Resources Conservation Service. 2006. Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin. United States Department of Agriculture Handbook 296. Washington D.C.

U.S. Department of Agriculture. Natural Resources Conservation Service. 2010. *The* PLANTS *Database* (http://plants.usda.gov, 2 January 2010). National Plant Data Center, Baton Rouge, LA 70874-4490 USA.

U.S. Department of Agriculture. Soil Conservation Service. 1970. Soil survey Torrence area, New Mexico. Washington, D.C.

U.S. Department of Agriculture. Soil Conservation Service. 1988. Soil survey Socorro area, New Mexico. Washington, D.C.

Vanderbilt, K. L., C.S. White, O. Hopkins and J.A. Craig. 2008. Aboveground decomposition in arid environments: Results of a long-term study in central New Mexico. Journal of Arid Environments. 72(5): 696-709.

Vivian, G. 1961. Excavations in a 17th century Jumano pueblo, Gran Quivira, New Mexico. Archeological Research Series No. 8. National Park Service. USDI, Washington, DC.

Weltzin, J. F. and G. R. McPherson. 1997. Spatial and temporal soil moisture resource partitioning by trees and grasses in a temperate savanna, Arizona, USA. Oecologia 112: 156-164.

West, N. E., K. H. Rea and R. J. Tausch. 1975. Basic synecological relationships in pinyon-juniper woodlands. *In:* Gifford, G. F. and F. E. Busby (Eds.). The pinyon-juniper ecosystem. Utah State Univ. Agr. Exp. Sta. Logan, UT.

West, N. E., F. D. Provenza, P. S. Johnson and M. K. Owens.1984. Vegetation change after 13 years of Livestock grazing exclusion on sagebrush semidesert in west central Utah. Journal of Range Management 37(3): 262-264.

Western Regional Climate Center. 2009. New Mexico Climate Summaries, Gran Quivira National Monument, NM. http://www.wrcc.dri.edu/summary/climsmnm.html.

Westfall, P. H., R. D. Tobias, D. Rom, R. D. Wolfinger and Y. Hochberg. 1999. Multiple Comparisons and Multiple Tests Using the SAS System, SAS Institute.

Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42(4): 266-274.

Whisenant, S. G. 1999. Repairing Damaged Wildlands: biological conservation, restoration, and sustainability. Cambridge, United Kingdom: Cambridge University Press. 312p.

White, L. D. 1965. The effects of a wildfire on a desert grassland community. [thesis]. Tucson, AZ: University of Arizona. 107p.

Wilcox, B. P., C. D. Allen, B. D. Newman, K. D. Reid, D. Brandes, J. Pitlick and D. W. Davenport. 1996a. Runoff and erosion on the Pajarito Plateau: observations from the field. p. 433-439. *In:* New Mexico Geological Society Guidebook, 47th Field Conference, Jemez Mountains Regions, 1996. New Mexico Geological Society. Albuquerque, N.M.

Wilcox, B. P., J. Pitlick, C. D. Allen and D. W. Davenport. 1996b. Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. p. 61-77. *In:* M.G. Anderson and S.M. Brooks (eds.) Advances in Hillslope Processes, Vol. 1. John Wiley and Sons, New York.

Wilm, H. G., D. F. Costello and G. E. Klipple. 1944. Estimating forage yield by the double-sampling method. Journal of the American Society of Agronomy. 36: 194-203.

Winer, B. J., Brown and D. R. Michels, K. M. (1991). Statistical principals in experimental design. (3rd ed.). New York: McGraw-Hill.

Woodbury, A. M. 1947. Distribution of pigmy conifers in Utah and northeastern Arizona. Ecology. 28(2): 113-126.

Wright, H. A. 1990. Role of fire in the management of Southwestern ecosystems. pp. 1–5 in J. S. Krammes, technical coordinator. Effects of Fire Management of Southwestern Natural Resources. General Technical Report RM-191. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, CO. 293p.