

AN ABSTRACT OF THE DISSERTATION OF

Casey A. Matney for the degree of Doctor of Philosophy in Rangeland Ecology and Management presented on May 28, 2010.

Title: An Assessment of Plant Community Structure, Herbivory, Soils, and State-and-Transition Theory on a Winterfat (*Krascheninnikovia lanata* [Pursh] A.D.J. Meeuse & Smit) Ecological Site

Abstract approved:

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State-and-transition models (STMs) have been successfully used to describe ecological dynamics in woodlands, shrublands, grasslands, and several other ecosystems. Changes in vegetation and soil are measured to gauge and predict plant community dynamics within ecological states and transitions between alternative ecological states. Ecological states and their boundaries are defined by a range of variability in vegetation and soil parameters. The ecological site classification system used in the United States requires the development of a unique STM model for each defined ecological site. The foundation of the ecological site concept is the estimation and quantification of a historical reference plant community. Historical information, land manager experience, and scientific data are used to quantify the historical reference plant community and gauge changes towards alternative stable states. In regards to winterfat plant communities and the Silty 6-10 PZ ecological site, conceptual STMs have not been fully developed or tested. The objective of this study was to develop, test,

and refine a process-based, data supported, STM for the Silty 6-10 PZ ecological site. The predominant factor of disturbance on this ecological site for the last 100 years has been grazing by livestock. To develop the process-based STM for the ecological site, we conducted a piosphere study centered on a livestock watering point and a five-year grazing exclosure study. With these two studies we were able to quantify the resilience of the ecological site when protected from disturbance (exclosure study) and also to quantify the trends of the ecological site when under a continued grazing disturbance (piosphere study). We used the information gathered from both of these studies to refine the STM. The four strongest factors that indicated differences between ecological states were amount of biological soil crust, soil surface aggregate stability, density of winterfat, and frequency of Indian ricegrass. Data from this study supported the initial proposed STM for the Silty 6-10 PZ ecological site. States defined were a Reference Winterfat State (State 1), Sickle Saltbush State (State 2), and a Creeping Wildrye State (State 3). Each ecological state corresponds to a site characterized by high, moderate, and low ecological functioning, respectively.

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An Assessment of Plant Community Structure, Herbivory, Soils, and State-and-
Transition Theory on a Winterfat (*Krascheninnikovia lanata* [Pursh] A.D.J.
Meeuse & Smit) Ecological Site

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Casey A. Matney

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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.

Casey A. Matney, Author

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CONTRIBUTION OF AUTHORS

Dr. Tamzen Stringham assisted with the data collection, design, and writing of all chapters.

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CHAPTER 1: GENERAL INTRODUCTION

State-and-transition models (STMs) are commonly used to predict and describe changes in ecological attributes and plant communities. Although STMs are often used, data and analyses to construct and support models are often lacking. We collected data in the Catlow Valley of southeastern Oregon to develop, test, and refine a three state STM for the Silty 6-10 PZ ecological site. This research project involved a number of scientific topics, plant species, and relevant historical accounts. Many sources of literature were gathered and reviewed to establish a basic knowledge of all the pertinent information available. The following review focused on the history of the Catlow Valley and the plant species winterfat (*Krascheninnikovia lanata* [Pursh] A.D.J. Meeuse & Smit) and sickle saltbush (*Atriplex falcata* [M.E. Jones] Standl.). Plant species were reviewed and organized according to important ecological attributes and physiological parameters. Also, the history of the Catlow Valley was summarized through the compilation of oral, government, and scientific records as well as relevant societal contextual information of the late 1900's. Other topics reviewed were piospheres/watering points and rangeland management modeling tools.

WATERING POINTS AND PIOSPHERES

Distribution of resources and impacts of disturbances are not spatially homogenous (Coughenour 1991). In arid regions, large herbivores are forced to congregate within a maximum distance from watering points (Thrash and Derry 1999). When large native herbivores use watering points in arid regions the area around the watering point is heavily trampled and vegetation heavily foraged (Glover 1963; Napier Bax and Sheldrick 1963; Savidge 1968; Van Wyk and Fairall 1969; Young 1970; Weir 1971; Spence & Angus 1971; Joubert 1976; Ayeni 1979; Cumming 1981; and Kalikawa 1990). In Africa, the maximum distance native large herbivores were able to travel from water was 10 to 16 km (Vand der Schijff 1957; Western 1975; Goodman 1982; and Collinson 1983). Young (1970) estimated that wild

herbivores seldom use only one watering point. Native large herbivores do not spend much time around watering points after drinking (Young 1970 and Senzota and Mtahko 1990). Most zebra and wildebeest have been found to spend less than three minutes at a watering point (Senzota and Mtahko 1990). In contrast, it has generally been found that livestock spend considerably more time around watering points than wildlife, and spending much of their time chewing cud or resting near watering points (Thrash and Derry 1999). Werger (1977) found that the disparity in amount of time spent at watering points between livestock and native large herbivores led to differences in utilization gradients.

In the Bear's Paw Mountain of Montana, Bailey (2004) found that cows spent approximately 8.5 h a day near water. During this 8.5 hour period, 90% of their time was spent within 100 to 200 m of water. Cows grazed from early evening (1800 h) to about mid-morning (1000 h) the following day, after which they were at or near water until they left to graze at early evening again. When near water, cows were usually lying down and often ruminating. When grazing, cows were typically more than 600 m from water (Bailey 2004).

During summer in eastern Oregon, Ganskopp (2001) found that the average maximum distance that cattle would travel from water was 2.92 km. Ganskopp's study was conducted from June through July during 1999. Actual average maximum distance traveled from water was 2.13 km, but average distance traveled from water was 1.16 km. Movement patterns were more strongly influenced by water than by salt. Cows spent approximately 11 hours grazing and 10 hours resting. Approximate grazing periods during the day were sunrise (0500 h) to mid-morning (1000 h) and from afternoon (1600 h) to 2 hours after sunset (2250 h; Ganskopp 2001). During the late summer in northeastern Oregon, average distance cows were found from streams was about 100 to 150 m (Porath *et al.* 2002).

In northeastern Arizona, Glendening (1944) found that grazing use of forage grasses decreased with distance from water. Forage use of black grama was shown to decrease with distance from water on the Jornada Experimental Range in southern

New Mexico (Campbell 1943). Some arid rangelands are grazed or browsed only in winter when animals can use snow to quench their thirst. Stoddart and Smith (1955) observed that cattle may travel 8-16 km for water if water is scarce, but traveling such distances is not common. Traveling further distances may increase the overall amount of trampling, and time spent traveling may limit the amount of time cattle have available for foraging. They also observed that grazing or browsing on level land is generally greatest nearest the watering point (Stoddart and Smith 1955).

Winter range on the eastern Montana plains showed trends of use by livestock to exhibit a broken-piece regression model (Holscher *et al.* 1953). From 0 to 300 m there was a steep decline in the amount of forage used with a decline from 100% to 37% for blue grama (*Bouteloua gracilis* [Willd. ex Kunth] Lag. ex Griffiths) and 100% to 50% for western wheatgrass (*Pascopyrum smithii* [Rydb.] A. Löve). However, from 300 m to 1600 m the decline in forage use was very gradual with a decline from 37 % to 29% for blue grama and 50% to 25% for western wheatgrass (Holscher *et al.* 1953).

Fusco *et al.* (1995) found that on a rangeland in good condition, black grama (*Bouteloua eriopoda* [Torr.] Torr.), mesa dropseed (*Sporobolus flexuosus* [Thurb. ex Vasey] Rydb.), threeawn (*Aristida sp.*), and total perennial grass standing crop increased as distance from water increased within 1 700 m from water. On fair condition rangeland, black grama and threeawn showed no association with distance from water. Broom snakeweed (*Gutierrezia sarothrae* [Pursh] Britton & Rusby) decreased with distance from water on rangeland in good condition. The majority of poisonous plants decreased with distance from watering points on both good and fair condition rangeland. Relationships between distance from water and response variables were tested using linear regression models. Poisonous plants and annual forbs dominated the zone from 0 to 500 m from water. Grazing effects on perennial grasses were greatly reduced on areas more than 1 000 m from water. On one site forage was scarce under 1 700 m from water, while only being scarce under 500 m at another (Fusco *et al.* 1995). Martin and Ward (1970) suggested that rotating access to

watering points may be an effective means for increasing perennial grass production, as opposed to fencing.

Nash *et al.* (2003) discovered that microtopography within 450 m of watering points in the Chihuahuan desert grasslands was severely degraded relative to distances of 1 050 m from water. Loss of microtopography around watering points exacerbated erosion processes and desertification. Perennial plant vegetative cover increased with distance from water. Nash *et al.* (2003) suggested that the reduction in height and number of micromounds as well as the lack of plants in conjunction with increased soil compaction resulted in reduced infiltration. They also suggested that the depth of standing water could be shallower following rains. They hypothesized that this resulted in increased evaporation losses and reduced water storage when compared with soils further away from watering points. Levels of soil moisture and nutrients were higher in microcatchments (Nash *et al.* 2003). Average size of bare soil patches decreased with increasing distance from water (Nash *et al.* 2003). DeSoyza *et al.* (1997) found that soil stability and long-lived vegetative cover increased with distance from water. Nash *et al.* (1999) found that long-lived vegetative cover increased as square distance increased from water. Following drought, plants that were farther than 450 m from livestock watering points had higher survivorship and greater recovery of total vegetative cover (Whitford *et al.* 1999).

Smuts (1972) found that rangeland became noticeably grazed, out to about 10 km when watering points were established in a new area. Smuts (1972) also found that pioneer annual plants became more dominant over the whole area while perennial species declined. Effects from natural and artificial watering points did not appear to be different (Thrash 1998b). When areas were changed from seasonal use to year-round use due to the establishment of perennial watering points, herbaceous basal cover declined out to 10 km (Smuts 1972; Delany and Happold 1979; Riney 1982; and Thrash *et al.* 1991).

Georgiadis and McNaughton (1990) observed that nutritional quality of forage was higher at 50 m from watering points than at 800 m from watering points. It was

hypothesized that the difference between 50 m and 800 m was caused by a species composition shift to annual plants and the soil having a higher nutrient content near the watering point (Georgiadis and McNaughton 1990). Weedy annual plants near the watering point are not likely to be used by herbivores to an appreciable extent (Thrash and Derry 1999).

Woody plants that are vulnerable to browsing or trampling suffer increased mortality near watering points (Thrash and Derry 1999). Browsing pressure is highest near watering points (Thrash and Derry 1999) and browse resilient shrubs replace other shrubs over time (Coetzee 1983; Andrew and Lange 1986b; Du Toit 1988; Bastin *et al.* 1993; and Ross 1995). Reduction in herbaceous species leads to reduced competition among plants for soil moisture and also reduces the intensity of fires near watering points, which often increases woody plant density (Van Vegten 1983; Tolsma *et al.* 1987; and Friedel and Blackmore 1988). Perkins and Thomas (1993a) found that shrub canopy cover decreased within 400 m of the watering point, but from 400 m to 1 500 m the shrub canopy cover increased. Beyond 1 500 m the shrub canopy cover decreased again.

Herbaceous plants decline in areas immediately adjacent to newly constructed watering points (Thrash and Derry 1999). Most of the herbaceous species found to decline are grasses (Spence and Angus 1971; Edroma 1981; Riney 1982; McNaughton 1984; Friedel 1988; Friedel and Blackmore 1988; Smit and Rethman 1989; Van Rooyen *et al.* 1990; Perkins and Thomas 1993a; Cowley and Rogers 1995; and Navie and Rogers 1997). Among the herbaceous species that survive, perennial species decrease while annual species increase (Weir 1971; Kelly and Walker 1976; Walker 1980; Riney 1982; Cowley and Rogers 1985; Tolsma *et al.* 1987; Stroleny and Mentis 1989; Georgiadis and McNaughton 1990; and Bastin *et al.* 1993). Weeds increase near watering points (Tolsma *et al.* 1987; Andrew 1988; Stroleny and Mentis 1989; McClaran and Anable 1992). Plant species diversity declined near watering points in Botswana, Africa (Tolsma *et al.* 1987); however a study in Australia found that seed bank species diversity increased near watering points (Navie *et al.* 1996). Overall

there was not a consistent relationship between distance from water and herbaceous species diversity (Thrash *et al.* 1993b).

Senzota and Mtahko (1990) found that the area immediately adjacent to watering points was more negatively affected by trampling than by grazing. It was hypothesized that the sacrifice area occurring around watering points was the direct effect of trampling by large herbivores and not grazing (Thrash and Derry 1999). When there was only a few watering points, competition for drinking space increased (Young 1970 and Zambatis 1980). As competition increased, animals spent more time around watering points and trampling increased (Young 1970 and Zambatis 1980). Herbivore trampling gradients can be indexed by the number of hoofprints around watering points. Studies of hoofprints show that herbivore trampling decreased as distance from water increased (Lange 1969 and Andrew and Lange 1986a). Soil compaction and soil capping (development of a hard and compacted physical crust at the soil surface) tend to occur around watering points (Lason 1979; Walker 1980; Riney 1982; Van Wijngaarden 1985; and Andrew and Lange 1986a), though a compaction layer did not seem to occur in sandy soils (Perkins and Thomas 1993a). Soil capping was caused by the combined effect of trampling moist soil and the removal of protective plant cover along with co-occurring splash erosion (Rauzi 1963 and Van den Berg *et al.* 1976). As grazing pressure increased, soil infiltration capacity decreased (Rhoades *et al.* 1964 and Van den Berg *et al.* 1976). Soil infiltration capacity increased with increasing distance from watering points (Kelley and Walker 1976; Dunn 1977; Riney 1982; Van Wijngaarden 1985; Andrew and Lange 1986a; and Thrash 1997). Soil compaction and infiltration capacity were important to consider on heavier textured soils (Walker 1980) but were less important on very sandy soils (Perkins and Thomas 1993a).

Researchers have found higher rates of soil erosion around watering points (Young 1970; Lason 1979; Edroma 1981; and Perkins and Thomas 1993a). At Kruger National Park in South Africa, one researcher estimated that 2.55 million tons of soil was lost over many years of grazing due to wind erosion around a single watering

point (Venter 1990). When wind erosion occurs, the finer particles of soil are lost first (O'Connor 1985), and sometimes the texture of the soil is found to change along a gradient radiating outward from the watering point (Thrash 1997). Accretion of dung and urine causes high concentrations of nutrients to develop near the watering point (Weir 1971; Georgiadis 1987; Tolsma *et al.* 1987; and Perkins and Thomas 1993a) and are only likely to occur when the watering point has been very heavily used or used for an extended period of time (Georgiadis 1987).

In 1947, Valentine surmised that even “well-watered” range is not equally used near and away from water. Utilization of forage was found to decrease perceptibly and consistently as distance from water increased. The relationship between forage use and distance from watering point varied according to many factors including season of use, class and age of livestock, grazing system, temporary availability of outlying water, snow, or succulent forage, presence of salt/mineral supplement, presence of roads, trails, and other cultural features that could influence cattle movement, resistance of forage species to grazing/browsing use and trampling, and topography (Valentine 1947 and Thrash and Derry 1999).

In 1943-1944, Valentine found a significant utilization gradient on a 985 ha pasture of good black grama grassland in New Mexico that was used by cattle on a year-long basis. A graphical display of the data (Fig. 1.1), assuming 100% use at 0 km from water, demonstrates a forage use gradient very similar to the gradient described by Holscher and Woolfolk (1953). Valentine's gradient would also fit a broken piece regression model with 2 gradients (Fig. 1.1). The first gradient being steep from 0 km to approximately 0.4 km while the second gradient was much more gradual from 0.4 km to approximately 3.3 km. A natural log model also fits the data well. Valentine (1947) suggested that a sacrifice area (area of highly degraded soils and vegetation) should not extend more than 0.4 to 0.8 km from the watering point though he clarified that the creation of a sacrifice area is unavoidable. Valentine showed that three vegetative communities were found along the grazing gradient out to 3.22 km from water; a mostly broom snakeweed community found from 0 to 0.6

km, a grass community from 0.6 to 2.8 km, and grasses only from 2.8 km to 3.22 km (Valentine 1947).

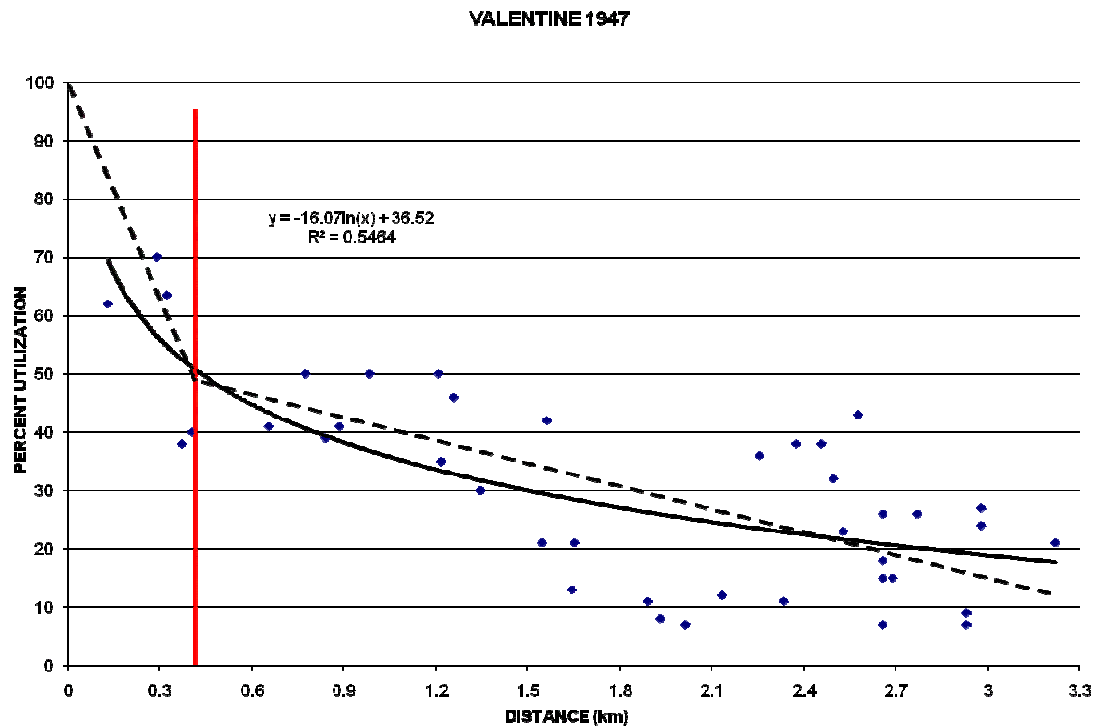


Figure 1.1. Examples of regression model fits for percent utilization data in relation to distance from water, collected by Valentine (1947). Broken piece linear regression model is displayed by dashed line and natural log model displayed by the solid black line. The vertical line indicates the outer perimeter of the sacrifice zone.

Valentine's early work and the research of later scientists led to the development of the term "piosphere", which is used to describe the sum response of soil and vegetation changes that occur around watering points (Valentine 1947). "Pios" is the Greek word for drink and "sphere" describes the weakening impact of the disturbance equally distant from the resource patch center (Lange 1969). Lange further states that "a piosphere is an ecological system of interactions between a watering point, its surrounding vegetation and the grazing animal." Forman and Godron (1981) defined a piosphere as "the result of the impact of a disturbance (*e.g.*, livestock grazing) at an environmental resource patch such as water, shade, mineral licks, etc." Andrew (1988) suggested that an animal's disturbance impact is highest

at the center of a resource patch and attenuates radially with increased distance from the patch center. The term piosphere was coined by Lange (1969) while researching the effects of sheep grazing at water points. Researchers have suggested that piospheres can be good indicators of whether or not desertification and land degradation are occurring (Glantz 1977; Dregne 1983; Sinclair and Fryxell 1985; and Pickup *et al.* 1994).

Since Valentine's research, a large number of gradient models have been developed to describe changes in ecosystem processes in relation to distance from watering points (Thrash and Derry 1999). In general, a gradient of utilization pressure develops which is greatest near the watering point and decreases as a function of distance from it (Thrash and Derry 1999). Often, a zone of very heavy use develops directly adjacent to the watering point which is referred to as the sacrifice area (Graetz and Ludwig 1978 and Thrash and Derry 1999). Areas outside the normal traveling distance of herbivores from the watering point receive the least amount of utilization (Van Wyk and Fairall 1969 and Graetz and Ludwig 1978). Forage availability has been found to exponentially increase as a function of distance from the watering point (Perkins and Thomas 1993a). The following factors have all been found to be directly proportional to distance from watering points: range condition (Foran 1980 and Barker *et al.* 1989), grass production (Edroma 1989), biomass (Andrew and Lange 1986b), understory cover (Bastin *et al.* 1993; Cowley and Rogers 1995; and Pickup and Bastin 1997), standing crop (Thrash 1998a), and basal cover (Thrash 1998a).

Van der Schijff (1959) used a five zone model to describe arid rangelands in South Africa. The first zone was termed the sacrifice area in scientific literature (Graetz and Ludwig 1978). Van der Schijff (1959) found the sacrifice area to extend out to 91 m from water and was characterized by dust and little to no vegetation. This sacrifice area has also been documented by Delany and Happold (1979) and Senzota and Mtahko (1990). Additionally in Botswana, Perkins (1991) found the sacrifice area to extend out to approximately 200 m from water. Zone two extended out 1.6 km from the watering point and was characterized by short heavily grazed vegetation and

trampled soil. The third zone extended from 1.6 km to 5 km and was grazed short with few signs of trampling. The fourth zone extended from 5 km to 8 km with light and selective use. The fifth zone extended beyond 8 km where there was much standing dead vegetation and few signs of use by herbivores.

On cattle ranches in sub-Saharan Africa, Perkins and Thomas (1993a) developed a five zone piosphere model. Perkins and Thomas divided the sacrifice zone, or first zone, into a subset of two areas: 0 m to 50 m, and 50 m to 400 m. In the first zone soil nutrients were enriched to the point of toxicity, soils were heavily trampled, and there was almost no vegetation. The second zone consisted of heavily grazed vegetation and trampled soil. The third zone extended from 200 m to 800 m from the watering point, was grazed short, and had fewer signs of trampling. The fourth zone extended from 800 m to 2 km from the watering point and consisted mostly of shrubby vegetation. The fifth zone extended beyond 2 km from the watering point and consisted of vegetation that showed few signs of herbivory. Thrash and Derry (1999) believe that the difference between Van der Schijff's piosphere model and Perkins and Thomas' piosphere model is that domestic cattle did not graze as far away from the watering point in Perkins and Thomas' study as did the native herbivores in Van der Schijff's study. It is important to note that both Van der Schijff's research and Perkins and Thomas' research were qualitative studies and not based on quantitative data, though some quantitative studies have found similar results as to both Perkins and Thomas and Van der Schijff's. Studies of domestic (Lange 1969; Andrew & Lange 1986a, Hart *et al.* 1991; Pinchak *et al.* 1991; Hart *et al.* 1993) and wild herbivores (Young 1970; Child *et al.* 1971; Weir 1971; Senzota & Mtahko 1990; Thrash *et al.* 1993a, 1995; and Hunter 1996) have shown that negative impacts from herbivores decreased with increasing distance from watering points.

Heshmatti (1997) characterized three zones around watering points: disturbed, sensitive and outer zones for close, moderate, and distant areas, respectively. Zones were distinguished by types and amounts of vegetative cover, as well as changes in some soil properties. Heshmatti hypothesized that high cryptogam cover and

moderately hard soil crust may be useful indicators for increased soil infiltration and nutrient cycle status in outer zones. Destroyed cryptogamic cover and brittle, non-coherent sub-crust were considered indicators of degraded/disturbed zones. Changes in cryptogam cover, slaking performance, infiltration capacity, rate of nutrient cycling and soil stability were useful indicators for detecting the three zones around watering points. Organic carbon did not appear as a useful indicator for detecting zones around watering points.

Graetz and Ludwig (1978) suggested that the typical piospheric response is usually depicted as a sigmoidal response function and is accompanied by four diagnostic characteristics for which a persistent trend indicates degradation. These diagnostics are: 1) the development of a permanent grazing gradient in the area directly surrounding the resource patch, where vegetation cover is permanently reduced or nonexistent during wet periods; 2) a decrease of the wet-period response toward the dry period; 3) a decrease of the dry-period response toward plant extinction; and 4) an increase in the distance of either the inflection point or asymptote away from the resource patch. Alternatively, Graetz and Ludwig (1978) found that a decrease in the slope of the line between the asymptote and the inflection points over time is an indicator of degradation. An opposite trend to each of these responses is indicative of recovery at a resource patch, such as a water point.

Pickup *et al.* (1994) observed that there were three more types of piosphere gradient responses in addition to the sigmoidal response, which they termed inverse, composite and complex. An inverse piosphere consists of a decrease in cover from the water point, except in the vicinity of the water point. This pattern suggests the presence of unpalatable species immediately surrounding the water source. A composite gradient is a combination of normal and inverse gradients meaning that cover decreases with distance from water and then begins to increase. Pickup *et al.* (1994) suggested that a composite gradient results in either an increase in woody plants or build-up of unpalatable species. If the gradient persists for a short time after a rain event it is probably indicative of an annual flush. Finally, a complex piosphere is

when the mean response of the index is not sufficiently robust to detect changes but the index's variance does detect the disturbance. Complex gradients appear in areas where reduced growth of vegetation in runoff and eroded areas is offset by increased growth in run-on and sediment sinks. The topographic mosaic of source and sinks average out the mean response of the index and a weak gradient is detected. However, the variance of the vegetation index is more robust where topographic effects are present. Perkins and Thomas (1993a) suggest that formation of a permanent gradient is an indication that changes in soil quality as well as quantity have occurred.

By measuring changes that occur as a function of distance from the watering point, instead of just changes over time, piosphere analysis can help scientists separate grazing impacts from climatic effects (Pickup *et al.* 1994). However, Hosten (1995) did not detect a piosphere response in the sagebrush steppe of southern Idaho. Hosten suggested that the response could have been lessened due to variable topography. It could have also been muted perhaps due to average to wet conditions during the period of study.

Mphinyane (2001) found in Botswana that two of three studied piospheres had leaf volume and dry mass decreasing with increasing distance from water; however, this was an area dominated by large shrubs that sometimes exceeded 2 m in height. For some of the piospheres, plant density was found to increase with increasing distance from water, while at another piosphere the relationship was reversed. The immediate areas around watering points were dominated by shrubby plants and annual plant species. Plant species diversity was found to increase as distance from water increased. Perennial grass species were found to be nonexistent near watering points due to trampling and heavy grazing pressure. Amounts of dung, urine, and soil trampling were higher near water where animals would concentrate. High levels of nutrients were also found near the watering points. Some species of grass showed a limited relationship to distance from water due to their poor preference by livestock. Within 500 m of the watering point, plants had elevated nutrient contents which were correlated with increased nutrient content in the soil due to accumulations of dung and

urine. Trends in vegetation around watering points were highly variable among different watering points (Mphinyane 2001).

Overall, general regression models and logistic models are the most common techniques used to model vegetation and soil responses around watering points (Valentine 1947; Graetz and Ludwig 1978; Kotliar and Weins 1990; Laca and Demment 1991; Ellis *et al.* 1993; Thrash *et al.* 1993b; Buckland *et al.* 1997; Thrash 1998a, 1998c; and Verlinden *et al.* 1998). Washington-Allen *et al.* (2004) found remote sensing to be a useful tool for modeling piosphere responses around watering points. Their premise was that piosphere analysis measures the effects of a disturbance's radial attenuation on the condition of vegetation and soil resources.

In all, the above information provides overwhelming evidence of the widespread and known effects that occur in vegetation and soil adjacent to watering points. These effects offer the ability to study grazing and trampling gradients as a function of distance from watering points.

SICKLE SALTBUUSH

Sickle saltbush is a common shrub in the Catlow Valley of Oregon (Fig. 1.2). The scientific name for sickle saltbush is *Atriplex falcata* (M.E. Jones) Standl. Sickle saltbush was previously listed as a variant under Nuttall's saltbush (*Atriplex nuttallii* var. *falcata* (Jones)). It was also listed as Gardner's saltbush (*Atriplex gardneri* (Moq.) D. Dietr. var. *falcata* (Jones) S.L. Welsh). There is an absence of information directly regarding sickle saltbush in the scientific literature. Due to similarities and synonymy within the genus, information for *sickle saltbush* has been generalized from Nuttall's and Gardner's saltbush. Common names for sickle saltbush are saltsage, saltbush, Nuttall's saltbush, and moundscale. Sickle saltbush belongs to the plant family *Chenopodiaceae*.



Figure 1.2. A sickle saltbush (*Atriplex falcata* [M.E. Jones] Standl.) seedling.

Saltbush is a dominant shrub in salt desert shrub communities of Wyoming. Associated species include Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis* Beetle & Young), black greasewood (*Sarcobatus vermiculatus* [Hook.] Torr.), winterfat (*Krascheninnikovia lanata* [Pursh] A.D.J. Meeuse & Smit.), galleta grass (*Pleuraphis jamesii* Torr.), alkali sacaton (*Sporobolus airoides* [Torr.] Torr.), Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth), shadscale (*Atriplex confertifolia* [Torr. & Frém.] S. Watson), and bottlebrush squirreltail (*Elymus elymoides* [Raf.] Swezey; Aldon 1978 and Krysl *et al.* 1984). Saltbush can be a climax species in greasewood-saltbush communities, but is not considered an indicator of early secondary succession on many desert rangelands (McKell and Goodin 1975). Saltbush-dominated communities are characterized as having few saltbush plants or clusters of plants separated by bare soil (Driese and Reiners 1997). A study of 128 sickle saltbush plants collected from across the Catlow Valley suggested the maximum age of sickle saltbush is approximately 50 years, with an

average age of 35 years (Matney 2010a). Saltbush is characterized as a subshrub ranging from 0.1 to 0.5 m tall (Hitchcock and Cronquist 1973). The basal stem of saltbush is woody and has herbaceous stems that arise from the woody base.

Saltbush generally loses its leaves during cold winter months (winter deciduous). Saltbush comprises 55% of total canopy cover when dominant in a stand (Ibrahim *et al.* 1972). The average height and width of this species is 3 and 5 dm, respectively. Biomass of saltbush in pure stands averages $100 \text{ g}\cdot\text{m}^{-2}$ (Goodman 1973). Saltbush grown on furrowed land produces more biomass and is more robust than non-furrowed land (Wein and West 1971). Individual plants occupy 100 dm^{-3} (Stutz and Sanderson 1998). Saltbush density in Wyoming ranged from 0.5-2.4 plants $\cdot\text{m}^{-2}$ (Driese and Reiners 1997).

Saltbush initiates growth in spring. The flowering period extends from May through July and is variable according to seasonal and yearly water variability (Wein and West 1971; Goodman 1973; and Blauer *et al.* 1976). Foliage remains succulent even during hot, dry summer months. The leaves of saltbush are alternate, linear-spatulate to narrowly obovate (Hitchcock and Cronquist 1973). Leaves are green-grey and scurfy (Stubbenieck *et al.* 2003). Saltbush has an extensive and branched root system. Roots of a 60 cm tall plant have been found to reach 140 cm deep and spread laterally 200 cm (Nord *et al.* 1971, Wein and West 1971). Saltbush is dioecious or rarely monoecious (Stubbenieck *et al.* 2003 and McArthur *et al.* 1978). The inflorescence and staminate flowers are yellow in glomerate, terminal, leafy-bracteate spikes. Pistillate flowers are in axillary and terminal spikes (Stubbenieck *et al.* 2003). It can reproduce by seed or vegetatively by layering and sprouting from the root (Nord *et al.* 1969 and Foiles 1974). Flowers are wind pollinated. Plants can be grown from cuttings.

Sickle saltbush is generally found on alkaline soils from southeast Washington south to California, east to Montana, Utah, and Nevada (Hitchcock and Cronquist 1973). Saltbush is usually found on clay soils with a pH between 7.8 and 8.6 that are typically low in available phosphorus, nitrogen, and potassium (Frischknecht and

Ferguson 1984). However, saltbush can be found on sodic, silty, or sandy soils (McArthur *et al.* 1978; Carlson 1984; Fisser and Joyce 1984; and Frischknecht and Ferguson 1984). Saltbush may be found on dry alluvial flats, eroded slopes, badlands, and on saline and alkaline clay soils (Wein and West 1971 and Miller *et al.* 1982). Saltbush sites are usually harsh and arid, with widely fluctuating temperatures and high winds (Nord *et al.* 1969 and Ansley and Abernethy 1984). It is considered to be a warm season, or C4, plant. Saltbush has a wide range of salt tolerance depending on ecotype. Salt-tolerant ecotypes respond positively to increases in soil salt content. Saltbush plants growing in pure stands are not as salt tolerant as those growing in stands of mixed vegetation (Driese and Reiners 1979).

Saltbush seeds require stratification, scarification, after-ripening, or washing to overcome dormancy (Ansley and Abernathy 1984). Saltbush plants grown in furrows have higher seed weights compared to saltbush plants at non-furrowed sites (Wein and West 1971). Saltbush seeds mature in the fall, but seeds may remain on the plant for 2 years. Seeds are wind dispersed and long-lived (Nord *et al.* 1969 and Foiles 1974). Recommended seeding rates are 32 pure live seeds·m⁻² (Schuman *et al.* 1994).

Saltbush is top-killed by fire, and it burns very slowly when ignited (Nord *et al.* 1969 and 1971). It is fire resistant because it contains high concentrations of minerals that increase char formation but has low concentrations of flammable compounds (Philpot 1968 and 1970). Saltbush communities may require 2-3 years to recover from fire. Saltbush vigorously resprouts from roots after burning and plants can recolonize areas from seeds (Nord *et al.* 1969 and 1971). Planting saltbush for fuel breaks has been suggested in California due to its tendency to not carry fire (Nord and Countryman 1972).

Saltbush is an important early season forage because it develops leaf area earlier than most other associated plants (Goodman 1973). Crude protein averages 7.2% throughout the growing season and can be as high as 10% in late fall (Hamilton and Gilbert 1972 and Davis and Welch 1984). Saltbush provides forage for livestock, antelope, mule deer, and rabbits (Nord *et al.* 1969; Floyd *et al.* 1988; and Banner

1992). Winter grazing of up to 35% of total biomass allows for maintenance of saltbush stands (Fisser and Joyce 1984). New plants are 1.4 times more numerous on stands where livestock were excluded compared to areas where livestock had been grazing (Philpot 1970).

Saltbush stabilizes soils with its extensive, highly branched root system. Saltbush is used to reclaim mined land (Powell *et al.* 1990 and Schuman *et al.* 1994) on soils that have high clay content, are saline, are sodic, and have low permeability (Smith *et al.* 1986 and Schuman *et al.* 1994). Transplanted seedlings had 100% survival in Wyoming (Luke and Monsen 1984). Gypsum applied to the soil surface improves productivity and canopy cover of saltbush communities on sodic bentonite mine spoils (Schuman *et al.* 1994, Smith *et al.* 1986). Saltbush responds favorably to contour furrowing (Wein and West 1971; Branson *et al.* 1976; and Powell *et al.* 1990). Saltbush vegetative and reproductive growth is positively affected by increased moisture provided by furrows (Branson *et al.* 1966 and Wein and West 1971).

Dried saltbush fruits were sometimes ground to make pinole flour by Native Americans (Stubbendiek *et al.* 2003).

WINTERFAT

The scientific name and authority for winterfat is *Krascheninnikovia lanata* (Pursh) A.D.J. Meeuse & Smit. (Fig. 1.3). Former scientific names for winterfat are *Ceratoides lanata* (Pursh) J.T. Powell and *Eurotia lanata* (Pursh) Moq. Common names for winterfat include: white sage, winter sage, lambstail, and hierba lanosa (Stubbendiek *et al.* 2003). Winterfat belongs to the plant family Chenopodiaceae and is a cool season plant (Stubbendiek *et al.* 2003). It has a potential longevity of 120 years or more (Clarke and Tisdale 1945) and is considered a climax species in the northern Great Basin (Coupland 1950 and Whisenant and Wagstaff 1991).



Figure 1.3. Photo of winterfat (*Krascheninnikovia lanata* [Pursh] A.D.J. Meeuse & Smit.) shrubs in the Catlow Valley, OR. Photo taken during May of 2007.

Winterfat is considered a subshrub with short and tall varieties (Stevens *et al.* 1977 and Welsh *et al.* 1987). Winterfat's taller varieties are found in areas of higher precipitation and are associated with Ponderosa pine (*Pinus ponderosa* C. Lawson) and juniper-pinyon plant communities. The shorter variety, *Krascheninnikovia lanata* [Pursh] J.T. Howell var. *lanata*, has an average height of approximately 38 cm (Stevens *et al.* 1977). Overall size of the plant varies due to ecotypic variation (Stevens *et al.* 1977). The variety common in Oregon seems to be the shorter variety, due to its size. Early accounts by settlers in Oregon estimate winterfat plants at approximately 61 cm in height (Shirk 1956 and French 1964), but by today's accounts the shrub appears to be more of the size indicated by Stevens *et al.* (1977).

The range of winterfat extends from the western United States to northern Mexico and to southern Canada (Coupland 1950; Nielsen 1968; Woodmansee and

Potter 1971; Davis 1979; Stubbendieck *et al.* 2003; and USDA 2007). Winterfat may inhabit elevations at sea level (Stevens *et al.* 1977) up to 3,048 m (Woodmansee and Potter 1971 and Stevens *et al.* 1977). In the Great Basin, winterfat inhabits clay, sand, and rocky loam textured soils (Booth 2006). In Utah it grows in gravelly and sandy loams (Duda 2003), and in silty clays (USFWS 2007). In New Mexico, winterfat can be found in sandy loams and clay loams that can be cherty, stony, gravelly or rocky (Woodmansee and Potter 1971). Gates *et al.* (1956) suggested winterfat distribution is not determined by bulk soil nutrients. Winterfat prefers soils that are slightly alkaline to neutral pH, but can be found on a variety of soil types and textures (Stevens *et al.* 1977). In many instances, winterfat is associated with dry alluvial soils having up to moderate salinity (Coupland 1950; Nielsen 1968; and Woodmansee and Potter 1971). Winterfat can tolerate a maximum salinity with an electrical conductivity of 27.1 dS·m⁻² (Workman and West 1967 and McFarland *et al.* 1990). In Texas, winterfat tolerated salty and/or alkaline soils (13-46 % exchangeable sodium; McFarland *et al.* 1990). Additionally, winterfat has adapted to sites with low available iron (Moore *et al.* 1972).

Winterfat is found most frequently on foothills, plains, and valleys (Woodmansee and Potter 1971) where annual precipitation ranges from 10-102 cm (Stevens *et al.* 1977). In southern Utah, Clary and Holmgren (1982) were able to predict the amount of winterfat above ground production (*i.e.* twigs, leaves, flowers, and fruits) and below ground production based on precipitation data. In the arid valleys of the northern Intermountain region where winterfat is found, most of the precipitation comes as snow during the winter months (Eckert 1954).

Winterfat leaf arrangement is alternate or in fascicles. Leaves are simple with the blades appearing linear to narrowly lanceolate. The apices of leaves are obtuse and leaf margins are entire and revolute. Dense red or white stellate and simple hairs cover leaves. There is a prominent midrib running the length of the leaves, and leaves occur as sessile to short-petioled. Stem twigs are stout and gray to reddish-brown.

Stems are covered with dense stellate and simple hairs. The main trunk's bark is exfoliating and gray-brown (Stubbendieck *et al.* 2003).

Winterfat has an extensive fibrous root system that permeates an area 1.45 m below the surface with a taproot that can extend up to 7.6 m below the soil surface (Stevens *et al.* 1977). Caldwell and Camp (1974) found that the biomass of the winterfat root system is 3 times that of the above ground biomass. It is estimated that winterfat's belowground biomass is often 11 times that of the aboveground biomass, which enables it to survive in low precipitation environments (Caldwell and Camp 1974). Winterfat's high root to shoot ratio, along with its ability to regenerate its fibrous root system to continually seek out water, enables winterfat to be photosynthetically active in all but the driest months of the year (Caldwell and Camp 1974).

Inflorescences of winterfat arise in clusters and are spiciform. Staminate flowers are terminal and axillary in clusters of 6-8. Staminate flowers have 4 obovate calyx lobes (1.5-2 mm long and 0.7-1.2 mm wide). The staminate flowers are white with a green stripe, wooly on the outside and glabrous inside. There are 4 stamens per staminate flower. Pistillate flowers are axillary, solitary or in clusters of 2-4 and apetalous. Pistillate flowers are subtended by 2 bracts, and the bracteoles are ovate, green, and pubescent (Stubbendieck *et al.* 2003). Pistillate flowers contain 2 styles (Booth 2006). Wind is the primary mode of fertilization, and fertilization may occur between plants or within the same plant on monoecious plants (Stevens *et al.* 1977).

Winterfat fruits measure 5 to 8 mm in length and 3 to 5 mm in width depending on ecotype (Stevens *et al.* 1977). The fruits of winterfat are utricles enclosed in 2 bracts. The bracts are lanceolate (5-6 mm long), being 2-horned at the top. Four dense tufts of long white hair cover the fruit. Inside the fruit the embryo is attached to the cotyledons and the radicle at the center. The embryo is enclosed with food storage tissue within the seed coat. Seeds are enclosed by the pericarp, which forms the utricle (Booth 1988 and Stubbendieck *et al.* 2003). Winterfat shrubs produce between 25,000 and 125,000 fruits per pound depending on area of collection and the

degree of seed fill (Stevens *et al.* 1977). The percentage of fruits filled with seeds has a wide range of variation. For instance, among five different New Mexico seed collection sites during one year, the percentage of fruits with seeds ranged from 67% to 95% (Springfield 1974).

Winterfat reproduces from seeds (Booth 1988) and can be monoecious or dioecious (Riedl *et al.* 1964). Winterfat plants typically flower from May to August, fruit ripening occurs from September to November, and diaspores release from the mother plant during fall and winter (Budd and Campbell 1959; Stevens *et al.* 1977; and Romo *et al.* 1995). However, timing of phenological phases follows annual variances in environmental conditions. In south central Idaho, Windle (1960) found that winterfat initiated its annual above ground growth in March and completed its reproduction by October. In northern Utah, West and Gasto (1978) observed that winterfat had a protracted period of growth beginning in April and ending in September. Some plants may flower twice in a growing season if they receive heavy precipitation in August or September (West and Gasto 1978). Winter precipitation resulting in relatively wet springs seems to be the main driver of growth in winterfat (West and Gasto 1978).

Seeds are likely to fully ripen following a hard frost in the fall (USDA 2002). In Montana, the average fully ripe date was approximately the last week of October (USDA 2005). The larger the plant the more seed it produces (Freeman and Emlen 1995). In years of lower precipitation, particularly in the winter, plants have truncated reproductive phases with low or no seed set (West and Gasto 1978). Seeds are shed in the fall or winter and dispersed by wind or by hitchhiking on animals (USDA 2005). Winterfat bracts have been found to aid in wind dispersal (Hilton 1941 and Stevens *et al.* 1977). Our field observations suggest that in the late summer and fall, the reproductive stems of winterfat become dry and somewhat brittle. The ends of annual reproductive stems will break away from the shrub and fall to the ground, near the plant, with seeds still attached to the stem. The primary mechanisms for reproductive stem breakage appear to be disturbance by herbivores and wind.

Winterfat seeds are able to germinate at temperatures as low as 0° C (Dettori *et al.* 1984). Bai *et al.* (1998) suggested that seed imbibition temperatures are related to ecotypic variation across winterfat's natural range. Booth and McDonald (1994) found that seeds imbibed at 25° C had faster rates of hydration than seeds imbibed at 4° C, but the warmer imbibed seeds had reduced seedling vigor. Booth *et al.* (1999) recommended that seeds be imbibed at 0-5° C. Booth found that imbibition at this temperature did not harm seed viability or vigor. It has been theorized that cold imbibition helps retain stored food for seedling growth (Booth 1992). Booth (1992) recommended that seeding be done in late fall, winter or early spring to allow for cold imbibition. Hou and Romo (1997) also recommend seeding in late fall to winter because seedlings are able to survive freezing temperatures and earlier seeding allows maximum seedling growth during the spring growing season. Hilton (1941) found that light was not critical to seed germination but winterfat seeds germinated best when imbibition moisture contained 0.5-1% salt.

Winterfat has genotypic variation in seed germination and seedling traits (Workman and West 1967; Springfield 1968a; Moyer and Lang 1976; and Meyer and Monsen 1993). Springfield (1972) observed that the optimum temperature for winterfat germination was between 10-27° C. Springfield (1973a) and Hou and Romo (1998a) found that germination rates and average seedling size increased with increases in seed weight. Springfield (1968a) found that warmer temperatures resulted in increased moisture stress and decreased germination. Booth (1987) suggested that winterfat seedlings take advantage of available moisture by tolerating freezing conditions as they progress from imbibed seeds to germinants to nonwoody seedlings. Seeds from populations on warm dry sites have shorter chill requirements than those collected from populations on colder, wetter sites (Meyer and Monsen 1993). Germination can be improved with a treatment of 600 to 1 000 mM ethanol solution for 4 to 8 hours (Hou and Romo 1998b).

Viability depends on the environmental conditions during seed formation and physiological characteristics of the seed (Woodmansee and Potter 1971). Stressors on

the mother plant, including wind sufficient to cause the stomata to close and interrupt photosynthesis, resulted in offspring with decreased vigor (Booth 1990). Lack of Ca^{++} , K^{+} , and Na^{+} can effect seed water adsorption (Booth 1989), seed freezing tolerance (Booth 1987), and seedling vigor (Booth 1990). Others have found that winterfat germination and seedling survival is dependent on the age of the seed (Booth and Schuman 1983), the size of the seed (Springfield 1973a), how the seed is stored after harvest (Springfield 1968b), and imbibition temperature (Booth and McDonald 1994). Booth (1990) found that lack of transpiration in the mother plant after seed set can reduce offspring vigor, and he hypothesized that the lack of transpiration reduced the amount of nutrients reaching seeds.

Weight of winterfat seeds range from 330,000-680,000 seeds·kg⁻¹ (Springfield 1973a and Abouguendia 1995). Larger and heavier seeds generally have higher germination rates (Springfield 1973a; Booth and Schuman 1983; and Hou and Romo 1998a). Higher concentrations of glucose, raffinose, and sucrose in larger seeds provided by the mother plant correlated with better cold tolerance, and larger seeds were also more efficient converters of carbohydrates (Wang *et al.* 2006b). When the mother plant commits fewer resources and creates smaller seeds, those seeds are more sensitive to external conditions and less likely to produce seedlings (Springfield 1973b).

Seed bracts shelter seeds, decrease premature germination (Stevens *et al.* 1977) and store nutrients for the seed (Booth and McDonald 1994). Bracts have also been found to aid seeds in soil adhesion and increase radicle growth, compared to threshed seeds, and they may also influence geotropic response by the radicle and improve radicle growth (Booth and Schuman 1983). Booth and Griffith (1984) also observed that seeds may be damaged in the threshing process. Current recommendations for seed planting suggest using non-threshed seeds (Booth 1988).

As the age of seeds increases the rate of germination decreases (Hilton 1941). Hilton (1941) found that seeds stored for 2 to 3 years showed low germination rates and seeds stored for 4 to 5 years were unviable, but Hilton did not specify the storage

procedure he used. Booth *et al.* (1999) observed that mitochondria and organelle deterioration may be the cause for decreased germination rates as seeds age. Younger seeds were found to have better imbibition temperature tolerance (Booth and Schuman 1983). Springfield (1968b) observed that using cold storage may prolong seed viability.

Booth and Schuman (1983) found that broadcasting onto a rough seedbed (seed depth ≤ 6 mm) is the most successful method of seeding. Springfield (1971) found that a seeding depth ≤ 1.6 mm had the highest success. Romo (2004) observed that a seeding rate of 20 seeds·m⁻² during late autumn allowed seeds time to after-ripen. This is in agreement with Springfield's (1973b) findings that it takes 10-25 weeks for winterfat seeds to after-ripen when stored at 2° C, 7° C, and room temperature. On winterfat endemic sites, seedbeds that have been tilled or treated with glyphosate will increase the amount of available resources for winterfat seedlings (Romo 2004). Romo also found that light to moderate amounts of litter on seedbeds will moderate temperature and moisture extremes. Hou and Romo (1997) observed that litter may shelter seeds from freezing temperatures. Litter may also increase humidity which has been found to have a positive effect on seeding success (Hou *et al.* 1999). Aerial broadcasting of winterfat fruits after light chaining of the surface was effective and late fall or winter seeding was most successful in Utah (Herbel 1986). Degraded rangeland can be improved by seeding winterfat although seedling establishment is not consistent.

Where research has been conducted, the canopy of cover of winterfat has been found to range from 7% to 19% in areas ungrazed by livestock and less than 1% where it has been grazed (West 1985; Carbal and West 1986; and Romo *et al.* 1995). Biomass of winterfat ranged from 30-60 kg·ha⁻¹ on moderately grazed sites in Wyoming (Krysl *et al.* 1984). Ungrazed winterfat in southwest Saskatchewan produced 160 kg·ha⁻¹ (Oldemeyer *et al.* 1971), while grazed winterfat in southeast Alberta produced 25 kg·ha⁻¹ (Romo *et al.* 1995). Use of nitrogen or nitrogen in combination with phosphorus has been found to increase winterfat biomass production

(Howard *et al.* 1977). Density of winterfat communities can range from 1.1 to 11.4 plants·m⁻² (West 1985 and Romo *et al.* 1995).

Bonham and Mack (1990a) observed that winterfat root growth is better when grown in pure stands. Competition with annual plants can reduce the percent of seedling establishment (West 1985). Competition from cheatgrass (*Bromus tectorum* L.) decreases seed production of winterfat plants (Freeman and Emlen 1995). Seedlings and young plants of winterfat have been found to be suppressed when grown in association with cheatgrass, Greene's rabbitbrush (*Chrysothamnus Greenei* (A. Gray) Greene), purple threeawn (*Aristida purpurea* Nutt.), and bud sagebrush (*Picrothamnus desertorum* Nutt.; Freeman and Emlen 1995). Transpiration and leaf conductance in winterfat is reduced when grown in association with cool season grasses (Bonham and Mack 1990b). Winterfat has a 4 to 7 times increased rate of transpiration relative to shadscale saltbush (*Atriplex confertifolia* (Torr. & Frém.) S. Watson) when under similar levels of moisture stress (Moore *et al.* 1972). Sandberg bluegrass (*Poa secunda* J. Presl) may or may not interfere with winterfat seedling establishment (Rosentreter and Jorgensen 1986 and Monsen and Pellant 1989). When grazing is a continuing disturbance, halogeton (*Halogeton glomeratus* (M. Bieb.) C.A. Mey.; Eckert 1954) and broom snakeweed (*Gutierrezia sarothrae* (Pursh) Britton & Rusby; Francis 1986) have been found to invade and dominate winterfat sites. Winterfat has been found to interact with vesicular-arbuscular-mycorrhiza, and on disturbed sites where mycorrhiza are not present, winterfat may not do as well (Miller 1979).

Winterfat has been characterized as tolerant to grazing (Stevens *et al.* 1977 and Blaisdell and Holmgren 1984). It can resprout from the crown (Romo *et al.* 1997), and has been found to reproduce naturally on lightly grazed range (Woodmansee and Potter 1971). In Canada, Smoliak (1965) did not find a difference in winterfat canopy cover of a lightly grazed site relative to a protected site. In a Utah study, West (1979) found little difference between survival rates of grazed versus ungrazed plots and attributed his findings to the low intensity of the grazing. Winterfat cover

significantly increased with light to moderate winter grazing in western Utah from 1933 to 1989 (Yorks *et al.* 1992). Winter use of 40-50% utilization by wildlife stimulated forage production of winterfat (Clark 1986). Defoliation of winterfat during the month of May yielded similar production the following year as control plants; June, July, or August defoliation however required a year or more for recovery (Romo *et al.* 1995). Anderson and Shumar (1986) found that significant herbivory by a decadal cyclic peak in black-tailed jackrabbit numbers did not affect above ground production of winterfat the following year. With overgrazing during the growing season winterfat stands have steadily declined and even disappeared in areas where they were once extensive (Stevens *et al.* 1977 and Blaisdell and Holmgren 1984). In Wyoming and Canada, winterfat has been found to decrease with grazing (Smoliak 1965; Krysl *et al.* 1984; and Dormaar *et al.* 1994). Rasmussen and Brotherson (1986) found that late winter grazing of winterfat reduced canopy cover by 2.3% to 13.9%. Cook and Child (1971) found that the extent of canopy cover of winterfat recovered but total biomass of shrubs remained low after 7 years of rest from grazing. Density, frequency, canopy cover and basal cover of winterfat have been found to be higher in areas protected from grazing than on grazed areas in the northern mixed prairie of Saskatchewan (Romo *et al.* 1995). Winterfat survives better when ungrazed or grazed during the winter instead of in the growing season (West 1979 and Chambers and Norton 1993). Grazing by cattle and rabbits, along with deficient moisture, were found to be factors responsible for low vigor and little regrowth of winterfat (Eckert 1954). During population peaks black-tailed jackrabbits can cause considerable damage to rangeland vegetation (McAdoo *et al.* 2002), and they may kill winterfat seedlings (Ogle *et al.* 2001). Winterfat is able to fully recover from heavy browsing that occurs due to cyclical peaks in black-tailed jackrabbit numbers (Anderson and Shumar 1986). Smith and Beale (1980) found that July or August browsing of winterfat caused decreased plant biomass the following year relative to undisturbed plants or plants browsed earlier in the summer. Late winter or early spring grazing negatively affects winterfat (Blaisdell and Holmgren 1984 and Whisenant and

Wagstaff 1991). Eckert (1954) found that summer clipping at all intensities prevented seed production, while winter clipping allowed some viable seed to be produced. Eckert (1954) hypothesized that grazing of winterfat by rabbits and cattle during the summer limited seed production. Studies suggest that winter grazing has the least amount of impact on winterfat (Cook and Stoddart 1963 and Stevens *et al.* 1977), while spring causes the most impact (Cook and Stoddart 1963). Under stressful conditions of winter grazing or drought, winterfat may forego seed formation in any given year (Eckert 1954). There have been various suggestions for what level of browse-use winterfat can endure. Cook and Stoddart (1963) recommended that winter or spring grazing of winterfat should not exceed 30% of the total standing biomass. Blaisdell and Holmgren (1984) suggested that browsing of winterfat be limited to 60% of its annual production. Hodgkinson (1975) suggested that 80% browse-use of winterfat during fall through winter was acceptable as long as it was not grazed at any other time. Additionally, Romo and colleagues (1997) observed that winterfat also resprouts well following spring or fall burning.

In addition to grazing by cattle, winterfat is browsed by rabbits and antelope among other wildlife species (Stevens *et al.* 1977 and Ogle *et al.* 2001). Winterfat and perennial grasses average 80% of jackrabbit diets, with shrubs being grazed particularly in fall and winter (Johnson and Anderson 1984). Season of use is important for wildlife as it is for cattle. Shrubs are the primary winter feed of black-tailed jackrabbits (McAdoo *et al.* 2002). Winterfat is a staple food for black-tailed jackrabbits (Johnson and Anderson 1984 and Anderson and Shumar 1986). Winterfat is also a major forage item for Rocky Mountain bighorn sheep on winter ranges near Yellowstone National Park (Keating *et al.* 1985). It contributed 6% of the diet, by relative density in feces, of Nuttall's cottontail in southern Idaho (Johnson and Hansen 1979). Winterfat is probably eaten by desert tortoise (McArthur *et al.* 1994). Townsend's ground squirrels browse winterfat (Yensen and Quinney 1992). Other animals that browse winterfat include mule deer, white-tailed deer, Rocky Mountain elk, desert bighorn sheep, pronghorn, and Dall sheep (Neilson 1968; Stevens *et al.*

1977; and Smith and Beale 1980). Winterfat seeds are eaten by rodents including the chisel-toothed kangaroo rat and Great Basin pocket mouse (Johnson and Jorgensen 1981). Pronghorn and rabbits browse stems, leaves and seed stalks of winterfat year round, especially during periods of active growth (Stevens *et al.* 1977). Winterfat is used for cover by rodents (Wasser 1982).

Forage value of winterfat has been rated good for sheep, pronghorn, elk and mule deer but is considered only fair for cattle (Stubbendieck *et al.* 2003). Others have rated winterfat palatability as excellent to good for sheep and deer, good to fair for horses and good to fair for cattle and elk (Dittberner and Olson 1983 and USDA 2001). Winterfat is of great importance to the western ranges (Dayton 1931 and Statler 1967), and is considered an important forage plant for livestock and wildlife in salt-desert shrub rangeland and subalkaline flats, especially as winter forage (Blaisdell and Holmgren 1984 and Mozingo 1987). Winterfat's importance in winter is high because nutrient quality of other vegetation, such as grasses, is low during this period (Hodgkinson 1975). Winterfat palatability is above average during all seasons but greatest during periods of active growth (Stevens *et al.* 1977), but overall palatability varies by year (McKell 1989).

Average nutrient content of winterfat in winter is 43.5% in-vitro digestibility, 10% crude protein, 0.11% phosphorus, and 16.8 mg·kg⁻¹ carotene (Welch 1989), but varies by season. Percent crude protein in the spring and summer were found to be 21 and 12.2, respectively (Welch 1989). Crude protein is highest in the spring (Stevens *et al.* 1977). Winterfat is a good source of digestible protein, vitamin A, and calcium (Cook *et al.* 1954). Smoliak and Bezeau (1967) found that winterfat had 23.13% crude protein, 3.96% digestible protein, and 26.3% cellulose during the emerging leaf period. During the flowering or heading stage of growth, winterfat had 18.5% crude protein, 4.40% digestible protein, and cellulose was 26.6%. During the seed ripening stage crude protein was 12.07%, while digestible protein was 4.32%, and cellulose was 32.6%. In Laramie, Wyoming crude protein of winterfat was 17.08% in mid-August and 8.08% in mid-October; crude fiber in mid-October was 36.72% and

29.62% in mid-September (Riedl *et al.* 1964). In Utah, Cook and Harris (1968) reported that during the period of Oct 1-Mar 15 winterfat had 9.1% total protein, 4.8% digestible protein, 13.3% ash, 11.7% lignin, 27.7% cellulose, and a gross energy of 1808 kcal·lb⁻¹. In general, winterfat loses crude protein and gains crude fiber as it matures, but the overall nutrient content of winterfat is comparable to that of alfalfa hay (Riedl *et al.* 1964).

Winterfat is often used for reclamation (Howard *et al.* 1977; Clark and Medcraft 1986; McFarland *et al.* 1990; Stark and Redente 1990a; Stark and Redente 1990b; and Wood *et al.* 1995). Winterfat can be seeded (Wasser 1984) or propagated by stem cuttings (Everett *et al.* 1978). Seeds used in revegetation projects should be collected from sources with similar site conditions (Smoliak and Bezeau 1967; Workman and West 1969; and Slauson and Ward 1982). In Nevada, winterfat survived better on south-facing slopes than on north-facing slopes when planted from containers on arid roadcuts (Everett 1980). A winterfat cultivar, Hatch, is used in deep snow regions to provide accessible winter forage (Stevens and Monsen 1988).

Several passerine bird species breed in winterfat-dominated communities including the horned lark, Brewer's sparrow and sage thrasher in east-central Nevada (Medin 1990) and horned lark, black-throated sparrow and loggerhead shrike in Utah (Medin 1986). It is potential nesting cover for upland game birds, especially when grasses grow through its crown (Shaw *et al.* 1984).

Some Native Americans soaked the leaves of winterfat in warm water to use as a hair wash. A decoction from the leaves was also used to treat fever (Stubbendieck *et al.* 2003).

Winterfat in the Catlow Valley

There have been two recent studies involving winterfat in the Catlow Valley. In 2004 and 2005, Estes (2008) evaluated seedbed preparation techniques for squirreltail and winterfat. Secondly, Estes examined winter forage quality and preference parameters of saltbush, winterfat, squirreltail, and creeping wildrye in cattle

diets. In 2006, Leary (2008) determined annual production and seed viability of winterfat across the Catlow Valley.

Estes found that seedbed treatments were better than no seedbed treatment for winterfat. Tilling provided a superior seedbed, followed by ripping and harrowing, respectively. Estes found that many winterfat seedlings had emerged by June, but some continued to emerge later into the growing season. A year after emergence, there was little difference in seedling survival from the previous year for tilled treatments, but the total number of winterfat seedlings in ripped and harrow treatments declined one year after emergence. Estes suggested that the germination and survival of winterfat seedlings could be maximized if the following criteria were met: all plant competition is removed, the soil is highly disturbed (tilled), winterfat plants are able to germinate in early spring, and there are continued precipitation inputs during the late spring growing season. The abundance of winterfat seedlings was found to negatively correlate with the abundance of other plant species such as saltbush, mature winterfat plants and squirreltail. Abundances of winterfat seedlings were not correlated with the abundance of creeping wildrye (*Leymus triticoides* [Buckley] Pilg.). Winterfat seedlings were negatively correlated with increased frequency of clasping pepperweed and annual mustard.

When studying seedbed treatments for squirreltail, Estes found that squirreltail numbers increased when there was little soil bed treatment as opposed to treatments like tilling and ripping. Although tilling and ripping reduced saltbush density, harrowing did not. Tilling and ripping provided some control of clasping pepperweed and annual mustard. Squirreltail and saltbush numbers in treatment plots continued to increase through the second year of the study, apparently due to seed dormancy or natural recruitment. Frequency of annual mustard and clasping pepperweed did not appear to have an effect on squirreltail seedling emergence or survival. In the end, Estes suggested that seedbed preparation was not needed for squirreltail establishment.

Estes quantified biomass production of four species: winterfat, squirreltail, saltbush and creeping wildrye. He found that production of these species did not

differ across the three large pastures within the Catlow Valley, for 2004 and 2005. Average combined total production for all species in 2005 ($815 \text{ lbs} \cdot \text{acre}^{-1}$) was higher than in 2004 ($292 \text{ lbs} \cdot \text{acre}^{-1}$). Average biomass produced ($\text{lbs} \cdot \text{acre}^{-1}$) for 2005 was 416 for saltbush, 179 for squirreltail, 81 for winterfat, and 139 for creeping wildrye. On average, saltbush was the dominant plant across all three pastures. Winterfat and saltbush had higher crude protein than squirreltail and creeping wildrye. Average composition of plants in cattle diets were 24% saltbush, 13% creeping wildrye, 7% winterfat, and 5% squirreltail. Although winterfat was the preferred plant species, saltbush provided much of the forage for wintering cattle.

In 2006 Leary investigated winterfat seed viability inside and outside of cattle/jackrabbit exclosures in the Catlow Valley. Leary found that there was little difference in seed viability across exclosure treatments or by the three pasture locations in the Catlow Valley. Average percent viability of winterfat seed was approximately 11%. Unfilled seeds accounted for most of the nonviable seeds (75%). Annual production, in biomass per plant, averaged 19.5 grams across the study area. On average, plots that had higher aboveground annual production had slightly higher seed viability. Lastly, plots with higher densities of saltbush had higher numbers of nonviable seed.

CATLOW VALLEY HISTORY

The Catlow Valley was not named until 1873 (French 1964). It is located in southeastern Oregon between Steens Mountains (previously known as the Snow Mountains; Preston 1978) and Beatty's Butte. The entire valley used to be an ice age lake (Cressman 1942). Cave drawings and tool remnants indicate that the valley has been used by Native Americans for thousands of years (Cressman 1942). The most recent tribe of people occupying the area prior to European settlement (early 19th century) was the Paiute Indians.

The United States Military arrived in the Catlow Valley in 1865 (Nielson 1987) and within two years constructed a road through the valley. It was called the

Central Oregon Military Road and it bisected the Catlow Valley, traveling west to east from Beatty's Butte through Skull Creek Canyon (Nielsen 1987; Fig. 1.4). The road weaved from north of Beatty's Butte, south and east by what is known as Coyote Gap, then turning east and north to pass Garrison Lake and continuing to Skull Creek. By all accounts, the road received relatively light use (Preston 1978 and Nielsen 1987).

Outside of the military, the next person to enter the valley was a settler by the name of Peter French. Coming from California, Peter French rode through the valley during June of 1872 and remarked that the height of the winterfat and bunchgrasses were approximately 61 cm tall and stretched across the entire valley (Shirk 1956 and French 1964). Peter French brought approximately 1,200 cattle with him. Simpson (1987) wrote that French considered the Catlow Valley a summer range of indeterminate value. French setup a corral in 1872 at what is now known as the Roaring Springs Ranch headquarters and then moved on to build his home and ranch headquarters near what is now called Frenchglen (Simpson 1987). A year later, in 1873, John Catlow and Con Shea temporarily camped in the valley with their cattle, but they did not settle there. However, from that time onward the valley became known as Catlow Valley (French 1964).

In 1868, David Shirk, who managed cattle for John Catlow, arrived in the area around the Steens Mountains. Catlow had all ready claimed a homestead in the Trout Creek Mountains through David Shirk, who had driven 800 longhorn cattle into the region from Texas. Shirk foraged the cattle through the Alvord Basin country for 7 years, and by 1875, the herd of 800 had grown to 4 000. Cattle were driven around the range of Steens Mountain, including Catlow Valley. Sometime after 1873, David Shirk wintered cattle in the Catlow Valley and erected a willow pole cabin at a large creek on the eastside of the valley. David Shirk named this tributary Home Creek. Shirk remarked in his biography (1956) that the pass from what is now Fields to the Catlow Valley (Long Hollow) had grasses over 61 cm high and with 61 cm of snow on the ground in January, the cattle easily reached good forage. When he wrote his biography in the 1920's, he acknowledged that many people, even at that time, would

not have believed his claim. David Shirk's observations were in kind with Peter French's. Shirk reminisced that no matter what time cattle were taken off the range in the Catlow Valley, they were in good condition. Shirk remarked that "bunchgrass waved over the hills and plains like a meadow, while the winterfat, considered the best winter feed was growing everywhere with its height over two feet tall and was never covered by even the heaviest snow."

Shirk was the first person of European descent to own land or build a home in the Catlow Valley. Shirk established a homestead on Home Creek in 1876. Soon after, Shirk's brother, William, established a home on Three Mile Creek, named so because it is 3 miles from the Home Creek Ranch. In this fashion, Six Mile Creek was also named. By 1878, Shirk suggested that if they were to make a permanent living ranching at Home Creek, they were going to need to provide winter feed. This claim suggests that within 5 years of first cattle-use, the Catlow Valley's winterfat and bunchgrasses were significantly reduced. This was most likely due to continuous year round use by increasing numbers of cattle by both the Shirk brothers' and Peter French's operations. We know that there was basin wildrye in the area as Shirk recorded using it as both roofing material for a shack and for a temporary home on Home Creek. Today, basin wildrye is seldom found around the Catlow Valley in quantities that would allow such a use.

Cattle management in the Catlow Valley and surrounding areas was simple to the extreme in the late 1900's. Cattle were rounded up once in the spring and once in the fall. Market cattle were trailed off to market in the fall, while the rest of the cattle were free to roam. Shirk claimed that deer were bountiful in the Catlow Valley and that there were vast herds of antelope. He also noted that beaver were plentiful in Home Creek. Shirk claimed that he built his barn at Home Creek from cottonwoods, which may have been aspen, that beaver cut down off of Home Creek. In 1873, Shirk had cut several trees (species unknown) for fence material from Skull Creek to put up on the Catlow Ranch in the Trout Creek Mountains. By this time, the Shirk brothers had cornered off about 50 000 to 60 000 acres of the valley. In 1880, Andrew

Sprangenburg settled a ranch at the mouth of Skull Creek in the Catlow Valley, where he homesteaded and ranched 800 acres (WHP 1902). The Sprangenburg and Shirk ranches controlled most of the water in the Catlow Valley.

At the same time that Shirk was developing his homestead at Home Creek, Peter French was purchasing a string of 40 acre holdings around the west side of the Shirk brothers' land (from Home creek to Three Mile creek). French erected a 5 wire fence around his land, effectively shutting of the Shirks from the rest of the valley. The Central Oregon Military Road ran through French's purchased strip, with the only bypass being a gate one mile north of where the road used to cross the valley from west to east. Simpson (1987) wrote in his book that French used the Catlow primarily for summer range, but it was highly likely that the valley was also used for winter range. A report (Shirk 1956) suggested that French also paid employees to homestead in the Catlow Valley, basically surrounding it, to keep other possible trespassers and claimants from moving in. French's efforts were not entirely successful. In 1888, David Shirk killed a man in the Catlow Valley on accounts of self-defense, which exemplifies the competition between settlers in the valley over resources.

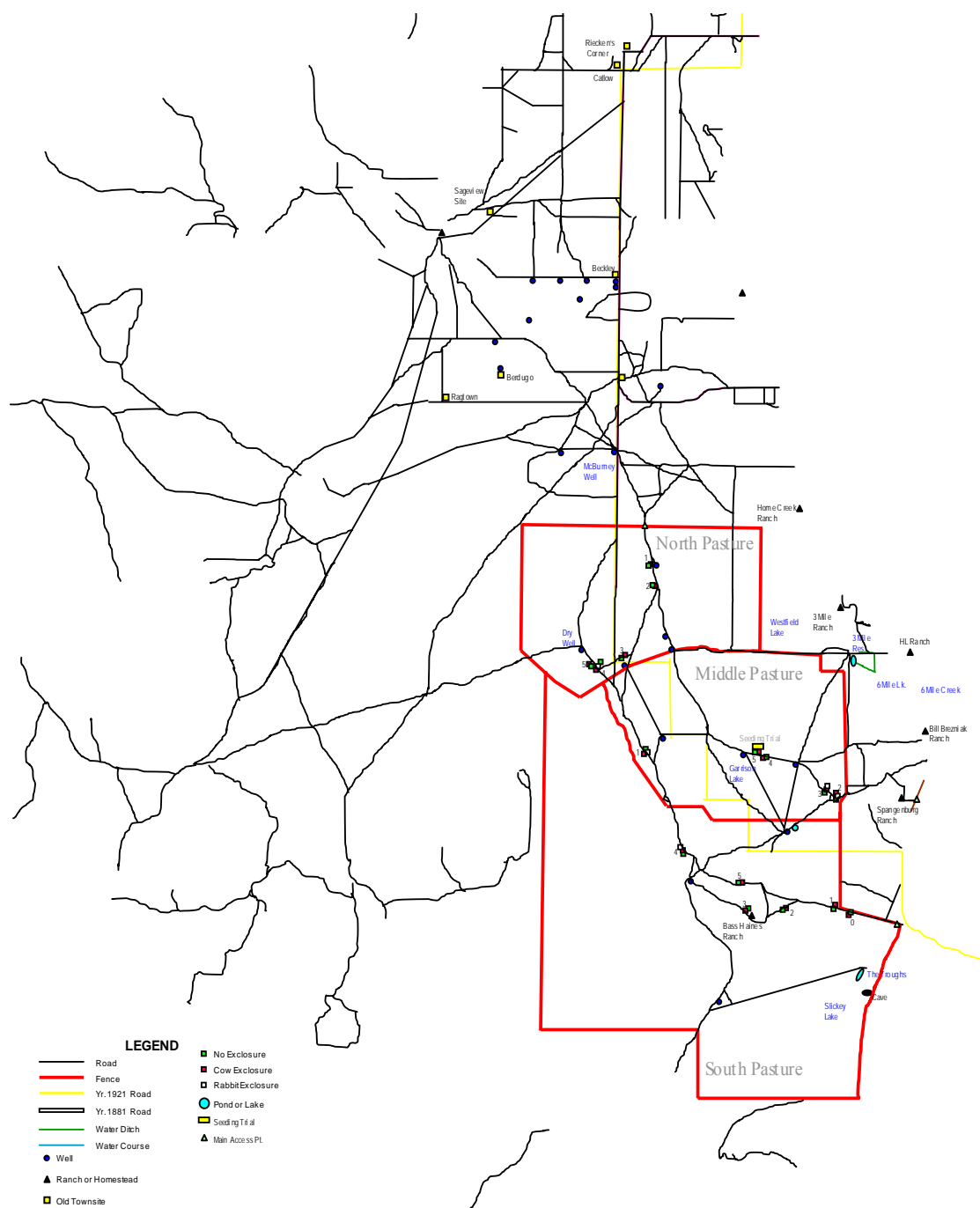


Figure 1.4. Map of the Catlow Valley and Town Sites.

In 1897, the Shirk brothers sold their Three Mile and Home Creek ranches to Peter French. This gave most of the control of Catlow Valley to Peter French, with the exception of the homesteaders. Management of the Catlow Valley went from Peter French to Bill Hanley in 1897, when Peter French was shot to death by a homesteader over a land dispute (Shirk 1956 and French 1964). During his tenure from 1897 to 1917, Bill Hanley was a supporter of homesteaders, unlike his predecessor who was staunchly against homesteading. Homesteaders often put fences up around water holes, which was problematic for rangeland ranchers (Braly 1982). By 1906, non-private lands were allotted to the Forest Service and the Bureau of Land Management. Later in 1917, management and ownership of the Catlow Valley was taken over by the Packer Swift and Company of Chicago for 3 million dollars. Swift and company sold the Blitzen River area of the French Ranch to the U.S. Government and created the Malheur Wildlife Refuge in 1935. Portions of Steens Mountain and much of the Catlow Valley remain in private ownership, including the Roaring Springs Ranch headquarters.

The Catlow Valley was a nexus of transportation that stretched from Burns, OR to Winnemucca, NV, mainly a stage coach line passing between Hart Mountain, Steens Mountain, and the Pueblo Mountains (Nielsen 1987). Over time, the Central Oregon Military Road was divided and sold on either side in 10, 20, and 40 acre tracts. Soon afterwards small hamlets sprang up in the Catlow Valley in the late 1800's and continued into the 1900's. The year of 1908 is generally considered the land boom era for the Catlow Valley. At one point in time before the 1930's, there were approximately 700 people living in the Catlow Valley. There were multiple towns that arose within the Catlow Valley: Blitzen, Ragtown, Beckley, Catlow, and Berdugo (Jackson 1978). Rock Creek Ranch was located at the northwestern extreme of the Catlow Valley. Approximately southwest of today's Roaring Springs Ranch, located in the middle of the valley, was the town of Blitzen which was granted a post office on April 27, 1888. Blitzen offered groceries, dry goods, mail, a school, a saloon,

residences, buildings, and improvements. These improvements provided goods and services for the growing population of Catlow Valley.

Homestead houses in the Catlow Valley would have ranged from 3 x 4.9 m to 4.9 x 6.1 m in size and usually would have had a privy and a barn for livestock. Houses were built with whatever material settlers could find or afford. Desert winds were notoriously strong and full of dust and sand. For many years, settlers relied upon streams, lakes and ponds for water. Some settlers dug wells that eventually became powered by windmills and gasoline pumps. Juniper, willows and shrubs were the primary source of fuel and fencing materials. Sawed lumber for building, pine and fir, was brought in from the nearest mills (Allen 1987). Homesteader use of the Catlow Valley can be generalized from information available for Silver Lake, OR. According to information for Silver Lake, much of the area around homesteads was either burnt or grubbed, and then plowed for crops. According to accounts in Christmas Valley, Oregon, by 1915 about 20-30% of the land area was plowed for crops. For the homesteaders crops were more than likely destroyed by a late frost into July, drought, and/or jackrabbits (Allen 1987).

Homesteaders primarily practiced dryland farming with rye. Rye could be sold for grain if it headed out or sold as hay if it did not. The years of 1910 to 1914 were a peak period for dryland farming in Harney County (Brimlow 1951). Black-tailed jackrabbits were a homesteader's nightmare often preying on gardens and crops. In 1915, a bounty of 5 cents per pair of rabbit's ears was offered by Harney County (Jackson 1978 and Braly 1982). Homesteaders replied in earnest and 1,029,132 pairs of ears were collected. Subsequently, 1915 was the only year a bounty was offered for the killing of jack rabbits. The Blitzen store allowed residents to pay for goods with jackrabbit ears worth 4.5 cents a pair at the store versus the 5 cent bounty offered by Harney County. In the 1930's there was a terrible jackrabbit population explosion, but the disease Tularemia eventually controlled their numbers. Drought as well as changes in sheep and cattle industries contributed to Blitzen's downfall during the 1930's. At the same time, homesteaders were starved out when attempts to farm grain

in the valley proved unfruitful. Blitzen was eventually abandoned and the post office closed on February 2, 1943 (Jackson 1978).

Severe winters were common in the late 1800's in eastern Oregon. Even the free flowing Columbia River froze over during these hard winters. In the winter of 1883-84, the worst winter on record, an estimated extreme low of -7.6°C was recorded in eastern Oregon (Brady 1982). During this same time, 2 meters of snow was deposited at Camp Polk in central Oregon. Oliphant (1968) suggested that the most severe winters were those of 1861-62, 1880-81, and 1889-90. In 1888, Dave Shirk at Home Creek, lost 1000 head of cattle due to the hard winter (Shirk 1956). Ranchers and homesteaders observed that the wildlife receded to low elevations (Brady 1982). Herbivores, including cattle, starved and froze to death, while mountain lions and bears readily pillaged the weak animals. The following winters were recorded as having severe snow and/or extreme low temperatures: 1846-1847, 1848-1849, 1849-1850, 1852-1853, 1861-1862, 1871-1872, 1873-1874, 1874-1875, 1875-1876, 1879-1880, 1881-1882, 1883-1884, 1889-1890, 1891-1892, 1899-1900, and 1901-1902 (Brady 1982). The year of 1888-1889 was recorded as a severe drought. Then, a series of droughts began in 1914, and later, the year of 1934 was a complete drought (Brimlow 1951).

In 1902, David Griffiths traveled the Great Basin country (eastside of Steens Mountain) and assessed the rangeland forage conditions. He found that most of the vegetation on Steens Mountain was grazed closely to the ground while the lowland areas where winterfat and other desert shrubs were located were not as closely grazed. Most of the foraging appeared to be from sheep. From the period of 1868 to 1902, winterfat was greatly reduced in its extent. Griffiths concluded that winterfat was very susceptible to overgrazing. Where Peter French noted winterfat up to 61 cm high, Griffiths found winterfat in the region to consist of only a stump and a few shoots ranging from 15.2 to 25.4 cm in height. The noted plant morphology indicated that the previous season's growth had been completely browsed to the stump. In places where sheep had grazed, Griffiths noted that the soil surface had been reduced to an

impalpable powder. The areas inhabited by shrubs, such as saltbush and winterfat, showed the most evidence of trampling and compaction (Griffiths 1902). He noted that this was most likely to occur in shrubby areas where the primary users were sheep. Griffiths found that sheep were wintered on the desert lowlands and fed largely on shrubs. Griffiths' conservative estimate, based on interviews with locals, was that there were 73 flocks of sheep on Steens Mountain during the summer and that the average size of a sheep flock was about 2,500 animals. The area they covered was about 400 square miles. This calculates to be approximately 456 sheep per square mile. In addition, several thousand head of cattle also roamed the range. The animals grazed this area for 4 to 5 months during the summer period. By 1902, Griffiths concluded that winterfat was of no great economic importance to ranchers due to its reduction in abundance.

RANGELAND MANAGEMENT MODELING TOOLS

According to Webster's New Collegiate Dictionary (1953) range can be defined as "that which may be ranged over, especially a sparsely populated and open region over which livestock may roam and feed." Although many definitions for range have appeared over the years, this definition is as simple and succinct as any. However livestock could be interchanged with any animal that grazes or browses, including wildlife. Recent definitions of rangeland have tried to include many parameters including soils, geology, climate and vegetation. As rangeland definitions have evolved, so have their management. In the 1900's, scientists in the U.S. recognized they needed ways to assess rangeland condition and predict rangeland responses to different purposed management strategies. Of particular importance to rangeland scientists was plant species composition. In 1919 Arthur Sampson published information on how plants and plant succession in the western states of the U.S. were affected by grazing. Sampson's work on succession was based on work done by Frederick Clements in 1916. Clements outlined the framework for how plant composition in an area changes relative to time and disturbance. Further work on the

topic of plant succession and plant composition was contributed by Henry Gleason (1926) and by Clements and John Weaver (1938). Gleason elucidated parameters that influence plant composition across regions. Clements and Weaver collaborated to construct the book “Plant Ecology”, which was written with plant succession as the overriding theme. In 1935, Sir Arthur Tansley contributed greatly to consolidating and defining the term succession, as well as related ecological terms used in studying plant succession and associations. Tansley defined succession as “a continuous process of change in vegetation which can be separated into a series of phases.” Tansley defined climax as “a relatively stable phase reached by successional change.” A climax can be reached through autogenic processes which are driven by the plants themselves, or by allogenic processes which are driven by factors completely external to the plants. Though the scientific study of plants was growing, by the mid-1900’s there was still a lack of consolidation into a framework suitable for land managers to apply to rangelands. In 1949, Dyksterhuis produced a paper that helped to fill this knowledge gap.

Dyksterhuis (1949) suggested quantitative means by which land managers could assess and predict vegetation responses to management, particularly grazing. Dyksterhuis’ model was primarily based on the successional concepts developed by Clements. The theory of succession and climax was organized into a format of rangeland condition classes based on plant composition. Condition classes were based on percentages of invader, increaser, and decreaser plant species, based on the estimated climax plant composition for the area (Figure 1.5). As a rangeland improved to excellent condition it was termed succession, while a change towards poor condition was termed retrogression. Prior to Dyksterhuis’ model, range condition was based on the relative amount of forage available or relative amount of desired forage available for grazing. Desirability was mostly linked to a plant’s forage quality for livestock. The model proposed by Dyksterhuis was put into use and remained the standard for range management for years. Over time, it became apparent that the model could not be used successfully under every situation. In fact, scientists

began to realize that in many instances they needed a new model for management because plant composition did not always follow the linear trends predicted by Dyksterhuis (Wilson 1984).

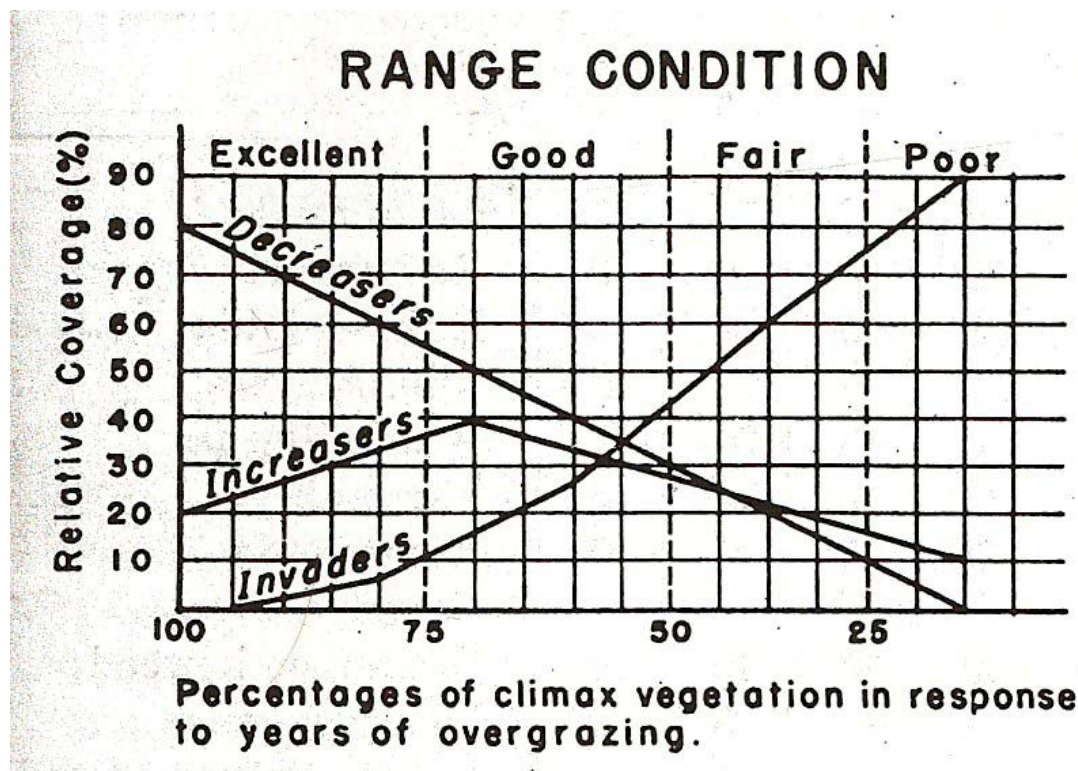


Figure 1.5. Dyksterhuis' 1949 diagram illustrating a quantitative basis for determining range condition. Excerpted from Dyksterhuis' article in the Journal of Range Management (Dyksterhuis 1949).

In 1973, Holling wrote a scientific paper discussing the ideas of stable states and ecological resilience. He defined ecological resilience as the amount of disturbance that an ecological system could absorb without changing state. He also identified a stable ecological system as a system that is maintained at or near an equilibrium state. The model Holling suggested to describe the departure from an old equilibrium state to a new state was called the adaptive cycle (non-equilibrium theory). In the adaptive cycle, when the ecological resilience of an ecological system was exceeded, it would collapse and reorganize in the direction of a new equilibrium (alternative state). The new alternative state may then persist for a period of time or collapse and reorganize in another new direction. In 1978, Joseph Connell published a

paper that listed ecological examples illustrating the concepts of non-equilibrium states in ecology. Westoby and colleagues proposed a model in 1989, state-and-transition, that explained and predicted the gray areas of rangeland response that Dyksterhuis' model could not. Dyksterhuis' model was rooted in equilibrium (climax) theory. Westoby's model was rooted in non-equilibrium theory. The main difference between the two models was that Westoby's model of rangeland trajectories could arrive at multiple alternative states, depending on the driving factors of plant community composition. In contrast, Dyksterhuis' model always suggested a single position along the linear scale between retrogression and succession. Westoby's model was more flexible and dynamic.

Westoby *et al.* (1989) suggested that botanical composition at a site could be helpful when parting site dynamics into specific states. The measurement of an ecological state could be quantitative, but the demarcation between states and the number of possible states were entirely subjective. Westoby *et al.* (1989) suggested that demarcation of states were useful for land managers to apply known management tools to reach land manager objectives. Westoby's approach to modeling has become known as state-and-transition modeling. Requirements of a state-and-transition model are 1) a defined list of alternative states, which can also be called possible states, 2) a defined list of possible transitions that would occur between all sets of possible states, and 3) a detailed list of conditions which precipitate each transition between states. A state is defined as a climate-soil-vegetation dimensional domain that can encompass a wide variation in plant species composition (Stringham *et al.* 2003). Changes that occur in any of these dimensions may cause a shift in trajectory and state.

Dyksterhuis' model framed plant composition field values relative to the estimated climax state for that site. Similarly, the state-and-transition model is framed relative to the estimated historic ecological site description for a site. The ecological site description is a quantitative estimation of the historic climate-soil-vegetation domain. In this way, all other possible alternative states for a site are derivatives from the original historical state. When a state has undergone change, the state will have

encountered some resistance to change. Resistance can be defined as the relative ability of a state to withstand a vector of change before reaching a threshold. When a threshold is crossed, the site transitions into a new state, with a new climate-soil-vegetation domain. A state may also be resilient to change, wherein the state experiences a release from a vector of change (disturbance) that has been causing deformation of the site domain towards a threshold, the state may rebound to its pre-disturbance position. A threshold is simply described as a boundary in time and space between states. Quantitatively, thresholds are difficult to describe, since transitions between states are not clear and may occur over broad timescales. Transitions are the shifts that occur between states, exceeding thresholds. Some transitions are reversible, while some are irreversible. Reversible transitions have not completely gone over a threshold, while irreversible transitions have completely cleared the threshold. The extent to which a transition can be reversible depends on the ecological process that is tied to the transition. Usually, a change to a new state is facilitated through the repair or degradation of an ecological process that affects soil or vegetation (Whisenant 1999). Some ecological processes can be repaired on short time scales, while some soil ecological processes cannot be repaired within a land manager's life time. Often, a particular event, such as fire or lack of fire, may act as a trigger to engage a transition between states (Archer 1989 and Miller *et al.* 2000).

Since 2003, Stringham *et al.*'s revised state-and-transition model has provided the best description of a working framework for modeling and predicting rangeland response using nonequilibrium theory (Figure 3). Stringham *et al.*'s model not only suggests the possibility of multiple states but also multiple phases within states. Here, changes within states (phases) follow plant community pathways and are measured in terms of changes in plant composition. These changes in plant community composition are influenced by many factors, including disturbance regime, and the corresponding ties to the ecological processes of site hydrology, energy capture, and nutrient cycling (Whisenant 1999). When a change in phase also results in the decoupling of an ecological process, the end result is a transition across a threshold to

a new state (Petersen 2005). Recently, Briske *et al.* (2008) further revised the state-and-transition model (Figure 1.6).

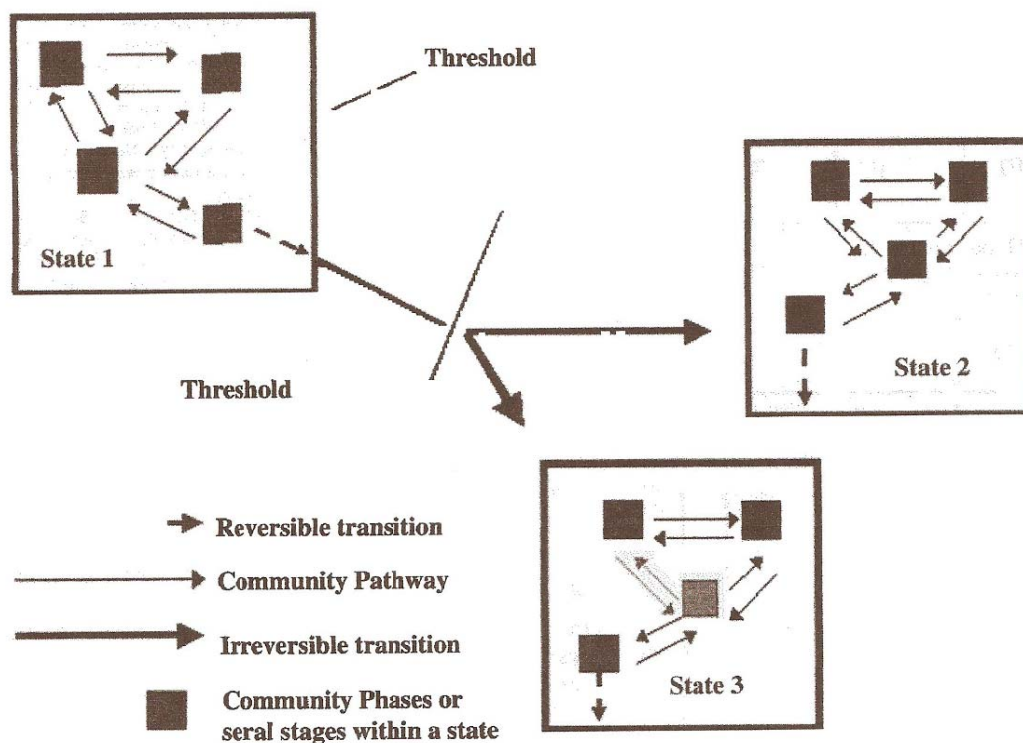


Figure 1.6. State-and-transition model display, excerpted from Stringham *et al.* (2003).

The newer state-and-transition model proposed in 2008 incorporates the concept of ecological resilience and other factors (Briske *et al.* 2008). Briske *et al.* described ecological 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.” Trends in ecological resilience, meaning movement closer to or further away from a potential state transforming event, are described as feedbacks that can either be positive or negative. Negative feedbacks reinforce the current state, while positive feedbacks move the system towards a different state. States having strong negative feedbacks are thought of as having high ecological resilience. States having weak negative

feedbacks have low ecological resilience. When one feedback becomes more dominant than the other, this change in feedback dominance is termed a “feedback switch.” When a feedback switch occurs, it is generally assumed that the system has crossed a threshold (Briske *et al.* 2008). When a system moves from a lower ecologically functioning state to a higher ecologically functioning state, reinforcing the negative feedback mechanisms of the higher ecologically functioning state, this change is considered a restoration pathway. These terms are graphically represented in the following chart (Figure 1.7). In the chart state 1 is considered the higher ecologically functioning state and state 2 is considered the lower ecologically functioning state.

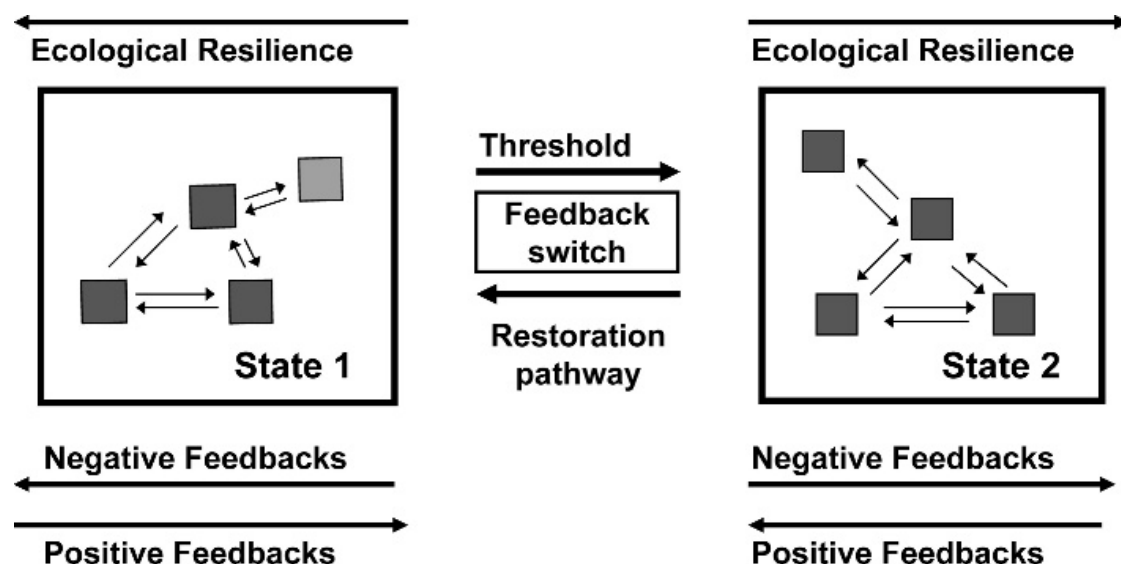


Figure 1.7. Revised state-and-transition model, excerpted from Briske *et al.* (2008).

STMs have been successfully used to describe woodlands (Archer 1989 and Petersen *et al.* 2009), shrublands (Petersen *et al.* 2009), and grasslands (Shaver 2010). Changes in vegetation and soil are measured to gauge and predict transitions between alternative states and phases within states. For each state, there is a range of values in vegetation and soil parameters that define it. In the United States each ecological site has been assigned its own unique STM, the foundation of which is based on

estimation and quantification of a historical reference plant community. Historical information, land manager experience, and scientific data are used to quantify the historical reference plant community and gauge changes towards alternative stable states. In regards to winterfat plant communities and the Silty 6-10 PZ ecological site (NRCS 2009), STMs have not been fully developed or tested, especially in the Catlow Valley of Oregon. The objective of this study was to develop, test, and refine a process-based STM for the Silty 6-10 PZ ecological site. The predominant factor of disturbance on this ecological site for the last 100 years has been grazing by livestock. To develop the process-based STM for the ecological site, we conducted a five-year grazing exclosure study to gauge the ecological resilience of the site as well as a piosphere study, to gauge changes that occurred on a continuum of cattle-use disturbance. With these two studies we were able to quantify the resilience of the ecological site when protected from disturbance (exclosure study) and also to quantify the trends of the ecological site when under continued grazing disturbance (piosphere study). We used the information gathered from both of these studies to test and refine a three state STM for the Silty 6-10 PZ ecological site.

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CHAPTER 2: A LONG-TERM HERBIVORY EXCLOSURE STUDY ON A WINTERFAT ECOLOGICAL SITE IN SOUTHEASTERN OREGON

ABSTRACT

Winterfat (*Krascheninnikovia lanata* [Pursh] A.D.J. Meeuse & Smit) communities provide valuable forage for wildlife and livestock. Many of these communities are at risk to decline due to years of heavy browse-use, especially by livestock, and co-occurring drought. The commonly proposed solution to this problem is to exclude these communities from herbivores with the assumption that over time these communities will rebound to resemble more of the presettlement native plant community. However, it is not known whether these communities possess the resilience to respond positively to protection from herbivores on a practical timescale. Even less understood is the resilience of the soil properties associated with these winterfat communities. To better understand this problem we determined the resilience of winterfat plant communities and their soil properties when protected from both cattle and wildlife (pronghorn antelope and black-tailed jackrabbits). In 2002, 16 locations were randomly selected from available winterfat communities and each received a 40 x 40 m control treatment (no enclosure) and a 40 x 40 m cattle enclosure treatment. Additionally, a subset of these 16 locations (n = 5) also received a 40 x 40 m cattle-antelope-jackrabbit enclosure treatment. Changes in plant size, frequency, and density were determined for the five-year period from the summer of 2002 to the summer of 2007. Changes were assessed for the three treatments: control (continued winter grazing by cattle and year round grazing by antelope [*Antilocapra americana*] and black-tailed jackrabbits [*Lepus californicus*]); exclusion of cattle; and exclusion of cattle, antelope, and jackrabbits. Our study suggests that continued use of winterfat communities by cattle and wildlife in the Catlow Valley resulted in increases in soil resistance to penetration and increases in gap size between perennial plants. Over the same period, winterfat cover and total perennial vegetation increased in areas protected from cattle and wildlife. We found higher aggregate stability and the

potential for higher infiltration rates in areas protected from cattle, suggesting that protection from grazing may be maintaining higher resource retention and hydrologic function. Areas protected from cattle-use offer encouragement that under no or light herbivory winterfat and sickle saltbush plants in the Catlow Valley protected from herbivory demonstrate high resilience, but that over a 5 year period managers cannot expect major changes in the plant community or soil properties. Such changes, if possible, are likely to occur over longer timescales than examined in this study and are probably greatly influenced by the timing and overall amounts of precipitation occurring at any given site.

INTRODUCTION

Winterfat provides valuable forage and habitat for many different wildlife species (Neilson 1968; Stevens *et al.* 1977; Johnson and Hansen 1979; Smith and Beale 1980; Johnson and Jorgensen 1981; Wasser 1982; Johnson and Anderson 1984; Shaw *et al.* 1984; Keting *et al.* 1985; Anderson and Shumar 1986; Medin 1986; Medin 1990; Yensen and Quinney 1992; McArthur *et al.* 1994; Ogle *et al.* 2001; and McAdoo 2002) and provides important forage for livestock (Eckert 1954; Cook and Stoddart 1963; Smoliak 1965; Cook and Child 1971; Woodmansee and Potter 1971; Hodgkinson 1975; Stevens *et al.* 1977; West 1979; Smith and Beale 1980; Blaisdell and Holmgren 1984; Clark 1986; Rasmussen and Brotherson 1986; Whisenant and Wagstaff 1991; Yorks *et al.* 1992; Chambers and Norton 1993; Dormaar *et al.* 1994; Krysl *et al.* 1994a; Krysl *et al.* 1994b; and Romo *et al.* 1995). Due to winterfat's value as forage for livestock, declines and losses of winterfat across its historic range have been linked to overuse by livestock and browsing during the growing season (Stevens *et al.* 1977 and Blaisdell and Holmgren 1984). During the late 1800's, winterfat was abundant in southeastern Oregon (Griffiths 1902; Shirk 1954; French 1964; and Simpson 1987). Oral accounts suggest early ranchers and herders allowed cattle to browse the abundant, highly palatable, and nutritious winterfat shrubs during both summer and winter (Shirk 1954; French 1964; and Simpson 1987). Continuous

season-long use along with increasing numbers of settlers and livestock (cattle, horses and sheep) resulted in precipitous losses in winterfat distribution and abundance. By the early 1900's, winterfat was either not present or in poor condition in the areas it used to occupy around Steens Mountain (Griffiths 1902). Around Steens Mountain, winterfat still persists, but only in small and highly dispersed stands that are relatively low in vigor. These winterfat plants have a long history of browse-use and continue to provide dormant season forage for cattle as part of an extensive grazing management plan.

Although many winterfat communities, including those in Catlow Valley, have been reduced in size and health, land managers would like to keep and expand existing winterfat communities. They would like to determine the ecological resilience of the winterfat plant communities including soil properties and plant community structure. Ideally managers want to allow cattle-use but maintain a certain degree of the historic presettlement plant community. Managers would also like to determine if surviving winterfat plant communities in less than optimal conditions can be restored to some semblance of the historic native plant community and determine if ecosystem function can be improved or if these sites are incapable of ecological repair.

We hypothesized that continued browsing by cattle in the Catlow Valley would negatively affect winterfat plant size and winterfat density. We predicted that excluding cattle from winterfat stands would result in increases in average plant size for winterfat and would either increase or maintain (no change) plant density. Concurrently, we expected the density and frequency of competing annual plants would decrease when winterfat communities were excluded from cattle-use. Additionally, we hypothesized that if winterfat communities were excluded from all use by cattle, jackrabbits, and antelope that results would be the same as the exclusion of just cattle, but that the overall positive effect on plant size and density might be greater than when only cattle were excluded.

MATERIALS AND METHODS

Study Area

The study was conducted over a 130 km² area within the Catlow Valley of southeastern Oregon. The average elevation of the valley is 1 400 m; the major landforms are lake terraces and lake plains; and soils are in the mesic temperature and xeric moisture regimes. Soils are fine textured Aridisols ranging from sandy loams to clay and primarily of the Spangenburg soil series (Appendix). The major ecological site is the Silty 6-10 PZ (NRCS 2009). Annual rainfall ranges from 127 to 254 mm. Rainfall measured on site at the study area for the water years of 2006 and 2007 were 222 mm and 129 mm, respectively. The water year 2007 was a drought year. Mean annual temperature and rainfall from 2005 to 2008 are presented (Fig. 2.1).

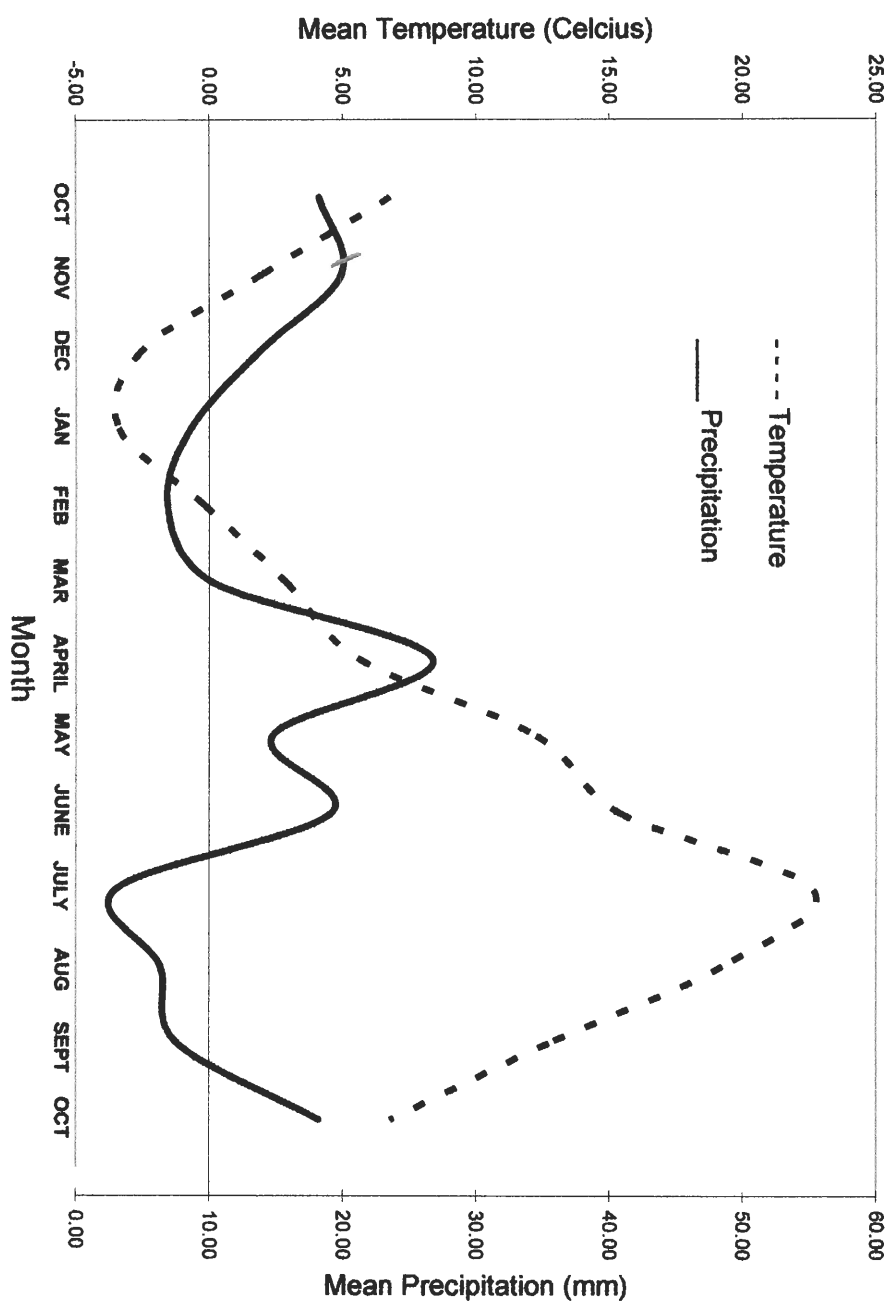


Figure 2.1. Monthly mean temperature (°C), dashed line, and mean precipitation (mm), solid line, for May 2005 through May 2008. Lines are smoothed interpolations between average monthly values.

Vegetation was composed of a mixture of shrubs, bunchgrasses, forbs, annuals, and one native rhizomatous grass species. The common native shrubs are winterfat and sickle saltbush (*Atriplex falcata* [M.E. Jones] Standl.), with the occasional Truckee rabbitbrush (*Chrysothamnus humilis* Greene). Native bunchgrasses include squirreltail (*Elymus elymoides* [Raf.] Swezey), Sandberg bluegrass (*Poa secunda* J. Presl), and Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth). The predominate native perennial forb is gooseberryleaf globemallow (*Sphaeralcea grossulariifolia* [Hook. & Arn.] Rydb. subsp. *grossulariifolia*), and the native rhizomatous grass species is creeping wildrye (*Leymus triticoides* [Buckley] Pilg.). Annuals present include cheatgrass (*Bromus tectorum* L.), clasping pepperweed (*Lepidium perfoliatum* L.), herb sophia (*Descurainia sophia* [L.] Webb ex Prantl), spreading wallflower (*Erysimum repandum* L.), and shortstem lupine (*Lupinus brevicaulis* S. Watson).

The Catlow Valley has a long history of livestock use, beginning in 1872 (Shirk 1956 and French 1964). In recent history, the Catlow Valley study area has been used as fall/winter range (October through March). Prior to 2002, movement of cattle within the valley was unimpeded. In 2002, three pastures were established. Pastures were grazed by cattle every fall/winter using an alternating North to South deferred rotation management plan. In this plan, the middle pasture was used at approximately the same time every year. In the years 2004 and 2005, 1 800 and 1 500 head of cattle, respectively, used the valley from October to March.

Treatments

From November 2001 to March 2002, most of the winterfat communities in the Catlow Valley were mapped. Soon after the winterfat communities were located, the valley was partitioned into three large pastures (North, Middle, and South). During the spring of 2002, five locations were randomly selected for study from each of the north and middle pastures, while six were selected from the south pasture. The additional location in the south pasture was selected due to the presence of a unique

area of predominately sandy soil (sandy loam). Locations were large enough to accommodate two to three 40 x 40 m experimental plots with a minimum 20 m buffer between plots. Each location was randomly assigned two treatments: 1) a cattle exclosure (n = 16) and 2) control plot (no exclosure, n = 16). A subset of these locations included a third treatment: cattle-antelope-jackrabbit (CAJ) exclosure (n = 5). One CAJ exclosure was established in each the north and south pastures, with three included in the middle pasture. CAJ exclosures were assigned to areas that were likely to have high use by black-tailed jackrabbits (*Lepus californicus*), where greasewood (*Sarcobatus vermiculatus* [Hook.] Torr.) and big sagebrush (*Artemisia tridentata* Nutt. subsp. *tridentata*) communities provided preferential cover habitat nearby to winterfat communities. Treatment exclosures were established by fall of 2002.

Design

From June through July of 2002 and 2007, vegetation species density and frequency metrics (nested contiguous quadrats) were taken from each of three to five linear transects at each study plot. Plant density for perennial bunchgrasses and perennial shrubs were recorded in contiguous 1 x 1 m frames for a total number of frames at each transect as determined by sample size calculations for winterfat. Plant density measurements for contiguous frames, frame size and number of frames, were determined for individual annual plant and perennial rhizomatous plant species according to sample size calculations in 2002. Sample size calculations were based on estimating 95% confidence intervals within 25% of the mean. Plant frequency was measured for all species in 0.2 x 0.2, 0.3 x 0.3, 0.5 x 0.5, and 1 x 1 m nested frames. The number of frames recorded for plant frequency was the same for all species. The total number of frames recorded for plant frequency at each transect was determined by the maximum number of contiguous frames used to measure plant species density.

Starting point and direction of linear transects were assigned randomly within each plot. Each transect measured a minimum length of 18 m. Within each plot was a

2 m buffer within the outer boundary of the 40 x 40 m plot. Vegetation measurements were not taken in this buffer area. In addition to density and frequency metrics, line-intercept (Elzinga *et al.* 1998) and gap-line-intercept (Herrick *et al.* 2000) methods were used along linear transects. Linear transect beginning and ending points were permanently marked with 23 cm metal stakes and 12 cm plastic fluorescent road construction tassel markers. Biological soil crust was visually identified as those soil surfaces having a darker surface and/or having lichen and/or moss (Belnap *et al.* 2001; Belnap and Lange 2003; and Rosentreter *et al.* 2007).

During September 2007, winterfat shrubs and sickle saltbush shrubs were harvested from across the Catlow Valley near exclosure treatments. Eight shrubs of each species were collected at each of the 16 exclosure treatment locations, $n = 128$ for each species. Shrubs were collected walking a random linear transect between cattle exclosures and control plots (no exclosures) at each of the 16 locations. The first 8 shrubs of each species encountered on the linear transect were harvested, regardless of size. However, shrubs with broken or fragmented crowns and shrubs that were dead were not harvested, since damaged shrubs cannot be reliably aged. Shrub age was estimated by counting annuli along the cross-section of the stem at the point of the root crown following methods by Schweingruber and Poschlod (2005) for subshrub plants.

Soil compaction was indexed by measuring soil penetration using an impact penetrometer (Herrick *et al.* 2005). Penetration was measured on the same two transects in each study plot that point intercept data was recorded for vegetation. Eight points, 2 m apart, were selected to measure soil penetration along each transect. In summary, the total number of points sampled for each plot was 16. And at each point the number of strikes it took to pass-through a given soil depth range (5 cm) was recorded. Soil depth ranges observed were 0-5, 5-10, 10-15, and 15-20 cm.

Soil nutrient content was determined for the A1 horizon for all treatments and locations in 2007. Three equidistant samples were taken along a random diagonal transect bisecting each treatment plot. Soil nutrient tests were performed by the

Oregon State University Central Analytical Lab. All samples were analyzed for total organic carbon and total organic nitrogen with a Leco CNS-2000 Macro Analyzer. Samples were determined for plant available inorganic phosphorus using the Olsen Phosphorus test (Olsen *et al.* 1954).

Soil pH samples were taken at the same three positions along line transects as soil nutrients in 2007. Soil pH was determined for all treatments and locations at the following depths: 1) A horizon, 2) thin 1 cm thick transition zone occurring between the A and Bt1 horizons (± 0.5 cm at the horizon interface), 3) Bt1 horizon, and 4) Bt2 horizon. The depths of soil layer transitions and mean layer depths were recorded. Soil pH, to the nearest 0.01, was determined using a LaMotte pH PLUS DIRECT Digital pH Meter and methods for soils that may vary due to salt content (Jones 2002). Bulk densities were determined using the volumetric ring method (NRCS 1999). Volumetric ring bulk density soil sample cores (6 cm diameter) were taken at the same locations as soil pH at three depths: A horizon, Bt1 horizon, and Bt2 horizon. Bulk density was determined for soil cores after drying at 100° C for 24 h.

One measurement of soil infiltration rate was made at each plot in 2007 using a single ring infiltrometer following protocols by Herrick and colleagues (2005). The diameter of infiltrometer rings was 50 cm. The soil was pre-wetted prior to setting infiltrometer rings. Infiltration measurements occurred at an undisturbed single random point in shrub interspaces within a 5 m radius of each treatment plot's center point. Infiltration rate was measured for 1.5 h, keeping a constant head of 2.5 cm (± 0.5 cm) in depth. The last two readings of infiltration over the 1.5 h measurement period were used to obtain average relative infiltration rates at each study plot.

Soil surface aggregate stability was measured at 18 equidistant points (1 m apart) occurring on a treatment plot line transect that was oriented perpendicular to the line transect used for soil nutrients, pH, and bulk density using methods outlined by (Herrick *et al.* 2005). Average soil surface aggregate stability was determined for all plots and treatments in 2007 only.

ANALYTICAL APPROACH

Effects on response variables between years were compared using a paired t-test for each treatment. Treatment effects on response variables within year were compared using a two-sample t-test when only the two main treatments, control and cattle enclosure ($n = 16$), were compared. CAJ enclosures were compared only to the subset of control and cattle enclosure locations that were in common ($n = 5$) using a 1-way analysis of variance (ANOVA). Statistical significance for paired t-tests between years and two-sample t-tests within year were set at $P \leq 0.05$, and significance for ANOVA's within year were set at $P \leq 0.10$. When ANOVA's were significant, mean separations between treatments were conducted using Tukey-Kramer multiple comparison procedures. All statistical analyses were performed using S-PLUS® 6.1 software (S-PLUS 2002).

Pasture locations as a factor were not included in the final analysis of enclosure parameters. In 2002, prior to the establishment of pastures and fencing, many parameters indicated a statistical difference between pastures at $P \leq 0.05$. These parameters included: winterfat density, vegetative cover, and many other metrics. These parameters were compared across pastures again, using the data collected in 2007. Results were in accordance to findings found in 2002. A further examination of these apparent differences in parameters by pasture location were compared with soils information. An analysis of metrics using soil groups, explained over twice the amount of variation, as compared to pasture locations. Data from Leary (2008) and Estes (2008) were in support of excluding pasture location as a factor in the final analysis; and winterfat seed viability, browsed shrub height in spring (open areas) following cattle-use, and standing crop production did not differ across pastures. Lack of a difference is attributed to the timing of use by cattle, rotation of pasture grazing order, and a relatively similar amount of time spent in each pasture by cattle. Further, pastures were all grazed between October and March, which is the period of the year when plants are dormant, with the assumption that cattle-use, whether it was in February or November had similar impacts on the plant community. It was also

assumed that seasonal variability in precipitation occurring during fall/winter coupled with the timing of cattle disturbance, mediated through trampling and compaction, had similar impacts on soils across the three different pastures.

RESULTS

Plant Community Structure

Between 2002 and 2007, a total of 26 plant species were encountered in the Catlow Valley study area (Table 2.1). In 2002, 25 species of plants were documented and measured for frequency and density. However, in 2007, nine of these species were totally absent, and three other species occurred so infrequently in 2007 that their numbers did not warrant a statistical test comparison with data from 2002. Russian thistle (*Salsola tragus* L.) was absent in 2002 but appeared in 2007. There were 13 species of plants that occurred in sufficient numbers (frequent/common) in both 2002 and 2007 to make statistical comparisons between years.

Plant Frequency

Plant frequency data (1 x 1 m frame size) taken from June through July of 2002 and 2007 were compared for each of 13 plant species for the three different exclosure treatments.

Control Plots

From 2002 to 2007 there were decreases in the frequency of occurrence of the annual plant species clasping pepperweed from 78% to 19% ($P < 0.0001$) and spreading wallflower from 46.7% to 12.6% ($P = 0.0008$) where no exclosures were used (Table 2.2). Over this period, shortstem lupine decreased from 4.87% to 0.55% ($P = 0.04$). There was suggestive evidence that squirreltail was reduced from roughly 47% to 35.5% frequency of occurrence ($P = 0.07$) and slight evidence that sickle saltbush decreased from 68.13% to 66.31% ($P = 0.15$).

Cattle Exclosure

Four annuals showed evidence of a reduction in frequency from 2002 to 2007: herb sophia ($P = 0.006$), spreading wallflower ($P = 0.0001$), shortstem lupine ($P = 0.04$) and clasping pepperweed ($P < 0.0001$). There was suggestive evidence that the perennial plant species sickle saltbush decreased from 70.3% to 68% ($P = 0.07$) and limited evidence that squirreltail decreased from 57.3% to 45% ($P = 0.15$). In contrast, there was some evidence that gooseberryleaf globemallow tripled in frequency from 0.6% in 2002 to 1.8% in 2007 ($P = 0.13$).

CAJ Exclosure

In CAJ exclosures, cheatgrass and spreading wallflower greatly decreased in frequency from 2002 to 2007. Cheatgrass frequency decreased from 38.6% to 15.1% ($P = 0.05$) and spreading wallflower frequency decreased from 67.8% to 1.9% ($P = 0.02$). There was convincing evidence that clasping pepperweed decreased from 77.9% to 1.6% ($P = 0.007$) and suggestive evidence that winterfat decreased from 83.2% to 81.8% ($P = 0.08$). Herb sophia was relatively abundant in 2002 but was totally absent in 2007.

Table 2.1. Mean percent frequency of occurrence for 26 species of plants across all treatments during the years 2002 and 2007 in the Catlow Valley study area. Lifespan and origin are identified.

Occurrence	Plant Species	Common Name	Lifespan	Origin	Frequency	
					2002	2007
Common/Frequent	<i>Achnatherum hymenoides</i> (Roem. & Schult.) Barkworth	Indian Ricegrass	P	N	4.4	4.4
	<i>Atriplex falcata</i> (M.E. Jones) Standl.	Sickle Saltbush	P	N	66.5	64.7
	<i>Bromus tectorum</i> L.	Cheatgrass	A	I	35.7	27.6
	<i>Chrysothamnus humilis</i> Greene	Truckee Rabbitbrush	P	N	0.7	0.6
	<i>Descurainia sophia</i> (L.) Webb ex Prantl	Herb Sophia	A	I	8.9	1.2
	<i>Elymus elymoides</i> (Raf.) Swezey	Bottlebrush Squimetail	P	N	48.5	37.3
	<i>Erysimum repandum</i> L.	Spreading Wallflower	A	I	51.7	9.7
	<i>Krascheninnikovia lanata</i> (Pursh) A. Meeuse & Smit	Winterfat	P	N	65.2	64.1
	<i>Lepidium perfoliatum</i> L.	Clasping Peppercorn	A	I	76.8	12.7
	<i>Leymus triticoides</i> (Buckley) Pilg.	Creeping Wildrye	P	N	27.6	26.7
	<i>Lupinus brevicaulis</i> S. Watson	Shortstem Lupine	A	N	5.1	0.4
	<i>Poa secunda</i> J. Presl	Sandberg Bluegrass	P	N	3.5	3.6
	<i>Sphaeralcea grossularifolia</i> (Hook. & Arn.) Rydb. subsp. <i>grossularifolia</i>	Gooseberry Globemallow	P	N	0.5	1.0
Uncommon/Infrequent	<i>Agropyron cristatum</i> (L.) Gaertn.	Crested Wheatgrass	P	I	0.03	0.04
	<i>Camelina microcarpa</i> Andr. ex DC.	Littlepod False Flax	A	I	2.7	0
	<i>Ceratocephala testiculata</i> (Grantz) Roth	Curneseed Butterwort	A	I	5.4	0.1
	<i>Chorispora tenella</i> (Pall.) DC.	Blue Mustard	A	I	0.1	0
	<i>Collinsia grandiflora</i> Lindl.	Giant Blue Eyed Mary	A	N	1.8	0
	<i>Descurainia plinnata</i> (Walter) Britton	Western Tansymustard	A	N	0.2	0
	<i>Ipomopsis congesta</i> (Hook.) V.E. Grant	Ballhead Ipomopsis	P	N	20.5	0
	<i>Leymus cinereus</i> (Schub. & Merr.) A. Löve	Basin Wildrye	P	N	0.05	0
	<i>Lomatium canbyi</i> (J.M. Coult. & Rose) J.M. Coult. & Rose	Canby's Biscuitroot	P	N	1.4	0
	<i>Mentzelia albicaulis</i> (Hook.) Torr. & A. Gray	Whitestem Blazingstar	A	N	2.4	0
	<i>Microsteris gracilis</i> (Hook.) Greene	Slender Phlox	A	N	80.4	0
	<i>Phacelia lutea</i> (Hook. & Arn.) J.T. Howell	Yellow Phacelia	A	N	1.3	0.2
	<i>Salsola tragus</i> L.	Russian Thistle	A	I	0	1.5

Lifespan: A = Annual and P = Perennial; Origin: N = Native and I = Introduced

Table 2.2. Mean percent frequency of occurrence for 13 species of plants for each of three treatments during the years 2002 and 2007 in the Catlow Valley study area.

Treatment	Species	n	df	2002	SE	2007	SE	Mean Change	P	95% CI
Control	Cheatgrass	16	15	33.4	8.9	22.6	6.1	-10.7	0.20	
	Herb Sophia	16	15	7.4	2.6	2.5	2.3	-4.8	0.19	
	Spreading Wallflower	16	15	46.7	9.5	12.6	4.3	-34.1	0.0008	(-16.7 to -51.4)
	Clasping Pepperweed	16	15	77.9	8.8	18.9	5.5	-58.9	< 0.0001	(-39.2 to -78.6)
	Shortstem Lupine	16	15	4.9	2.0	0.6	0.3	-4.3	0.04	(-0.3 to -8.3)
	Indian Ricegrass	16	15	2.5	1.5	2.4	1.1	-0.2	0.80	
	Sickle Saltbush	16	15	68.1	8.9	66.3	8.6	-1.8	0.15	
	Truckee Rabbitbrush	16	15	0.3	0.2	0.2	0.2	-0.1	0.33	
	Bottlebrush Squirreltail	16	15	47.1	6.9	35.4	6.8	-11.7	0.07	
	Winterfat	16	15	63.7	7.7	62.8	7.7	-0.9	0.29	
	Creeping Wildrye	16	15	29.5	10.7	29.0	10.4	-0.5	0.40	
	Sandberg Bluegrass	16	15	2.7	0.7	2.0	0.9	0.7	0.45	
	Gooseberry Globemallow	16	15	0.5	0.5	0.4	0.3	-0.1	0.81	
Cattle Exclosure	Cheatgrass	16	15	37.0	8.4	36.6	9.2	-0.5	0.95	
	Herb Sophia	16	15	8.4	2.4	0.4	0.3	-8.0	0.006	(-2.7 to -13.3)
	Spreading Wallflower	16	15	51.8	8.4	9.3	3.5	-42.4	0.0001	(-26.1 to -58.8)
	Clasping Pepperweed	16	15	75.5	9.0	9.8	3.5	-65.7	< 0.0001	(-47.1 to -84.3)
	Shortstem Lupine	16	15	5.0	2.2	0.1	0.1	-4.9	0.04	(-0.2 to -9.5)
	Indian Ricegrass	16	15	7.7	4.1	7.9	4.1	0.3	0.75	
	Sickle Saltbush	16	15	70.3	8.1	68.0	8.5	-2.3	0.07	
	Truckee Rabbitbrush	16	15	1.4	1.3	1.2	1.0	-0.2	0.33	
	Bottlebrush Squirreltail	16	15	57.3	8.5	45.0	8.1	-12.3	0.15	
	Winterfat	16	15	61.2	7.3	59.8	7.0	-1.3	0.26	
	Creeping Wildrye	16	15	25.5	10.3	24.6	10.0	-0.9	0.37	
	Sandberg Bluegrass	16	15	4.7	2.9	4.9	2.5	0.3	0.69	
	Gooseberry Globemallow	16	15	0.6	0.6	1.8	1.3	1.3	0.13	
CAJ Exclosure	Cheatgrass	5	4	38.6	14.9	15.1	11.4	-23.6	0.05	(-0.6 to -46.5)
	Herb Sophia	5	4	15.4	4.9	0	0	NA	NA	
	Spreading Wallflower	5	4	67.8	17.6	1.9	1.0	-65.8	0.02	(-19.0 to -112.7)
	Clasping Pepperweed	5	4	77.9	15.4	1.6	1.3	-76.3	0.007	(-34.4 to -118.1)
	Shortstem Lupine	5	4	6.0	4.5	0.7	0.7	-5.3	0.24	
	Indian Ricegrass	5	4	0	0	0	0	NA	NA	
	Sickle Saltbush	5	4	48.9	20.1	48.9	20.0	0.0	0.96	
	Truckee Rabbitbrush	5	4	0	0	0	0	NA	NA	
	Bottlebrush Squirreltail	5	4	25.0	11.1	18.7	7.0	-6.2	0.33	
	Winterfat	5	4	83.2	9.1	81.8	9.4	-1.4	0.08	
	Creeping Wildrye	5	4	28.5	16.1	25.0	14.5	-3.5	0.21	
	Sandberg Bluegrass	5	4	2.3	1.0	4.6	1.7	2.3	0.31	
	Gooseberry Globemallow	5	4	0.0	0.0	0.3	0.3	0.3	0.37	

Plant Density

Plant density data (plants·m⁻²) from 2002 and 2007 were compared for each of 13 plant species for the 3 different exclosure treatments.

Control

Of the species that did occur in both years, cheatgrass and spreading wallflower offered no evidence of a change in density. Claspings pepperweed did decrease from 33 plants·m⁻² to approximately 1 plant·m⁻² ($P = 0.003$). There was strong evidence that squirreltail and winterfat decreased in density (Table 2.3). Squirreltail decreased by 0.4 plants·m⁻² ($P = 0.01$), while winterfat decreased by 0.09 plants·m⁻² ($P = 0.03$). Basin wildrye, which was present in 2002, was absent in 2007.

Cattle Exclosure

Annual plant species were either completely absent in 2007 or were reduced in density inside cattle exclosures (Table 2.3), save cheatgrass. The evidence was strong showing that cheatgrass increased in overall density by approximately 5 plants·m⁻² ($P = 0.04$), while there was convincing evidence that claspings pepperweed decreased from 35.6 plants·m⁻² to 0.4 plants·m⁻² ($P = 0.009$). Evidence suggests that herb sophia and spreading wallflower also decreased in density, $P = 0.06$ and $P = 0.07$, respectively. In kind, there was moderate and strong evidence that winterfat ($P = 0.09$) and squirreltail ($P = 0.05$) decreased by 0.1 and 0.6 plants·m⁻², respectively. Other perennial plant species showed no evidence of change between 2002 and 2007 (Table 2.3).

CAJ Exclosure

For plant density data, all annual plants that were present in 2002, except cheatgrass, fell below values of 0.01 plants·m⁻² in CAJ exclosures during 2007. The data for cheatgrass suggests that there was no evidence of a change in density over the course of five years ($P = 0.26$). All perennial plants remained at the same density,

except there was some evidence that sickle saltbush decreased in density by 0.3 plants·m⁻² ($P = 0.12$).

Table 2.3. Mean density (plants·m⁻²) of 13 species of plants for each of three treatments during the years 2002 and 2007 in the Catlow Valley study area.

Treatment	Species	n	df	2002	SE	2007	SE	Mean Change	P	95% CI
Control	Cheatgrass	16	15	1.47	0.68	1.52	1.28	0.05	0.97	
	Herb Sophia	16	15	0.12	0.08	0	0	NA	NA	
	Spreading Wallflower	16	15	1.42	0.81	0.46	0.21	-0.96	0.27	
	Clasping Pepperweed	16	15	33.18	8.94	1.14	0.45	-32.05	0.003	(-12.81 to -51.30)
	Shortstem Lupine	16	15	0.01	0.01	0	0	NA	NA	
	Indian Ricegrass	16	15	0.03	0.02	0.03	0.02	0.00	0.49	
	Sickle Saltbush	16	15	3.03	0.77	4.09	1.63	1.06	0.34	
	Truckee Rabbitbrush	16	15	0	0	0.01	0.01	NA	NA	
	Bottlebrush Squirreltail	16	15	1.02	0.18	0.59	0.13	-0.43	0.01	(-0.11 to -0.76)
	Winterfat	16	15	2.04	0.47	1.95	0.46	-0.09	0.03	(-0.01 to -0.18)
	Creeping Wildrye	16	15	16.32	7.26	10.32	4.98	-6.00	0.41	
	Sandberg Bluegrass	16	15	0.08	0.04	0.06	0.04	-0.02	0.31	
	Gooseberry Globemallow	16	15	0.006	0.006	0.006	0.004	0.00	0.97	
Cattle Exclosure	Cheatgrass	16	15	1.70	1.08	7.04	3.05	5.35	0.04	(0.33 to 10.37)
	Herb Sophia	16	15	0.07	0.03	0	0	-0.07	0.06	
	Spreading Wallflower	16	15	2.67	1.30	0.12	0.09	-2.54	0.07	
	Clasping Pepperweed	16	15	35.64	11.71	0.44	0.22	-35.2	0.009	(-0.16 to -60.24)
	Shortstem Lupine	16	15	0.02	0.02	0	0	NA	NA	
	Indian Ricegrass	16	15	0.13	0.08	0.14	0.09	0.01	0.46	
	Sickle Saltbush	16	15	3.14	0.61	3.03	0.72	-0.11	0.54	
	Truckee Rabbitbrush	16	15	0.02	0.02	0.01	0.01	-0.01	0.33	
	Bottlebrush Squirreltail	16	15	1.56	0.31	0.97	0.23	-0.60	0.053	
	Winterfat	16	15	1.78	0.35	1.68	0.34	-0.10	0.09	
	Creeping Wildrye	16	15	8.38	3.87	7.51	3.94	0.88	0.83	
	Sandberg Bluegrass	16	15	0.26	0.22	0.21	0.15	-0.05	0.44	
	Gooseberry Globemallow	16	15	0.007	0.007	0.05	0.03	0.04	0.15	
CAJ Exclosure	Cheatgrass	5	4	2.60	1.62	0.73	0.49	-1.87	0.26	
	Herb Sophia	5	4	0.16	0.11	0	0	-0.16		
	Spreading Wallflower	5	4	5.17	2.14	0	0	-5.17	0.08	
	Clasping Pepperweed	5	4	26.98	12.99	0	0	NA	NA	
	Shortstem Lupine	5	4	0	0	0	0	NA	NA	
	Indian Ricegrass	5	4	0	0	0	0	NA	NA	
	Sickle Saltbush	5	4	1.63	0.81	1.37	0.68	-0.26	0.12	
	Truckee Rabbitbrush	5	4	0	0	0	0	NA	NA	
	Bottlebrush Squirreltail	5	4	0.47	0.22	0.24	0.09	-0.23	0.21	
	Winterfat	5	4	3.81	1.12	3.84	1.21	0.03	0.85	
	Creeping Wildrye	5	4	14.94	10.03	4.13	2.53	-10.81	0.24	
	Sandberg Bluegrass	5	4	0.07	0.05	0.13	0.07	0.06	0.26	
	Gooseberry Globemallow	5	4	0	0	0.01	0.01	NA	NA	

Population Age Structure of Shrubs

Age frequency histograms for sickle saltbush and winterfat are displayed (Fig. 2.2). Summary statistics for each species are presented (Table 2.4). On average, winterfat shrubs are 21 years older than sickle saltbush. The youngest shrubs sampled were 19 years of age for winterfat and 4 years of age for saltbush shrubs. Overall, winterfat had a longer lifespan than saltbush, with winterfat demonstrating little to no recruitment over the last 20 years. Saltbush had a much shorter lifespan and more frequent recruitment over the last 20 years.

Table 2.4. Age summary statistics for $n = 128$ shrubs of winterfat and sickle saltbush.

Species	Minimum Age	Mean Age	Median Age	Maximum Age	Variance	SD	SE	95% CI	95% CI
								Lower	Upper
Winterfat	19	42.1	42	93	174.5	13.2	1.2	39.8	44.4
Saltbush	4	20.9	21	57	61.2	7.8	0.7	19.5	22.3

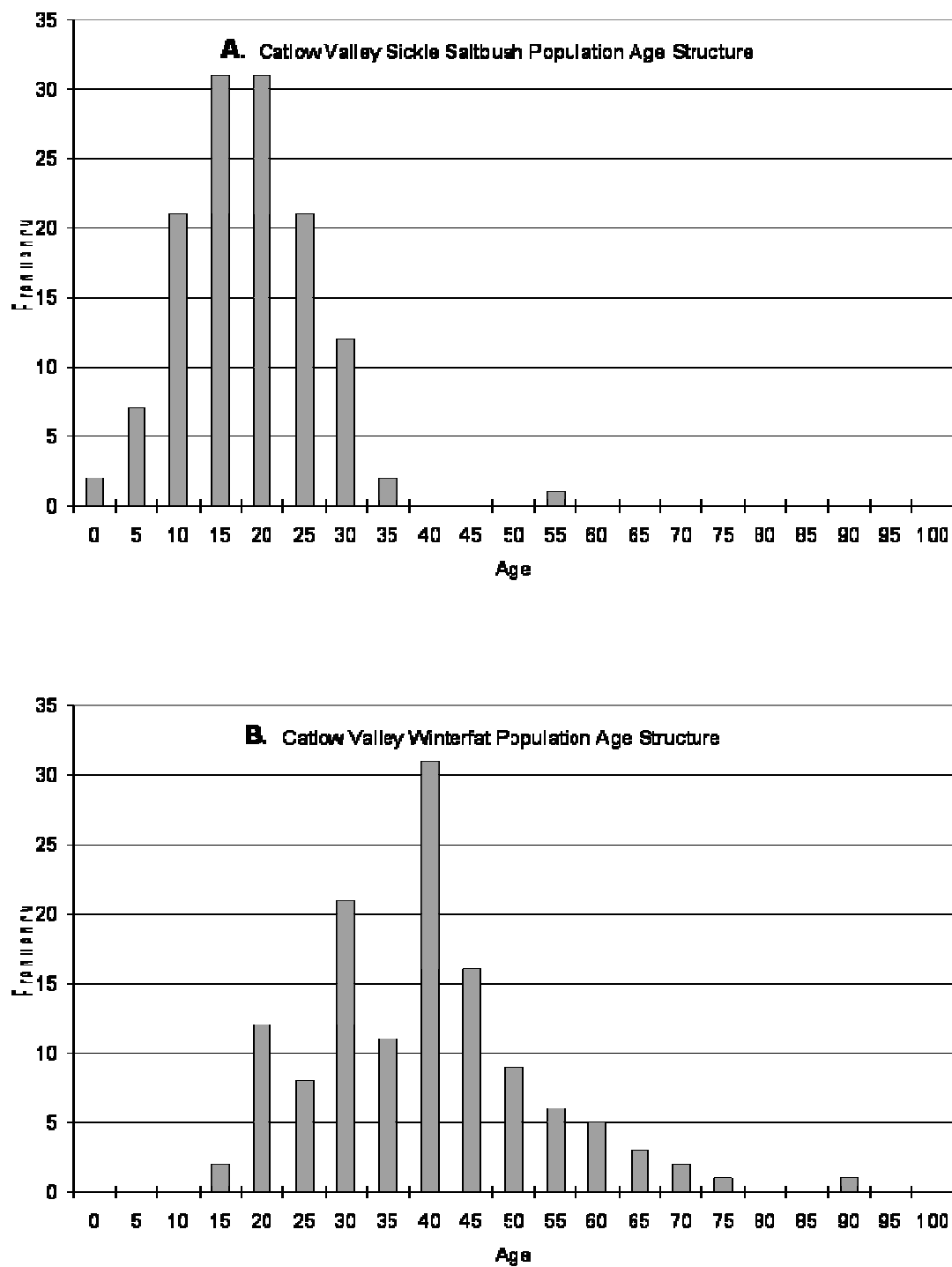


Figure 2.2. Age frequency histograms for $n = 128$ shrubs of sickle saltbush (A, top) and winterfat (B, bottom).

Soil Infiltration

There were no differences in relative infiltration rates in 2007 between control plots and cattle exclosures ($P = 0.55$; $df = 30$). Average relative infiltration rates were $3.29 \pm 0.52 \text{ cm}\cdot\text{h}^{-1}$ for cattle exclosures and $2.96 \pm 0.21 \text{ cm}\cdot\text{h}^{-1}$ for control plots. At the subset of locations where CAJ exclosures were included there were also no evidence of differences in relative infiltration rates between exclosure treatments and control plots ($P = 0.48$; $df = 2$), using a 1-way ANOVA. Mean relative infiltration rates at these three locations were 3.09 ± 0.35 , 3.13 ± 1.39 , and $4.64 \pm 0.94 \text{ cm}\cdot\text{h}^{-1}$ for control plots, cattle exclosures, and CAJ exclosures, respectively. Although there were no statistical differences in relative infiltration rates, mean relative infiltration rates were numerically greater at exclosed plots as compared to control plots, especially in regards to CAJ exclosures. Overall, standard errors of exclosed plots were two to four times greater than those of control plots.

Soil Aggregate Stability

Soil aggregate stability was examined in 2007 only. Aggregate stability scores were based on the scale of 1 (lowest aggregate stability) to 6 (highest aggregate stability; Herrick *et al.* 2005). Mean soil surface aggregate stability scores were 3.3 ± 0.1 for control plots and 4.3 ± 0.1 for cattle exclosures. Areas protected from cattle-use were 31% more stable than those in control plots ($P < 0.001$, $df = 30$). There was no evidence of differences in soil surface aggregate stability scores among exclosure treatments at the subset of locations where CAJ exclosures were included ($P = 0.36$; $df = 2$), using a 1-way ANOVA. Mean soil surface aggregate stability at these three locations were 3.5 ± 0.3 , 4.0 ± 0.3 , and 4.0 ± 0.2 for control plots, cattle exclosures, and CAJ exclosures, respectively. Numerically, exclosed plots had higher mean soil surface aggregate stability than control plots.

Soil Bulk Density

Soil bulk density was measured in 2007 only. There were no differences in bulk density values between control plots and cattle exclosures at any of the three soil depths examined ($df = 30$). Average bulk density for the A horizon was $1.10 \pm 0.02 \text{ g}\cdot\text{cm}^{-3}$ for control plots and $1.11 \pm 0.02 \text{ g}\cdot\text{cm}^{-3}$ for cattle exclosures ($P = 0.63$). Mean bulk density of the Bt1 horizon was $1.17 \pm 0.01 \text{ g}\cdot\text{cm}^{-3}$ in control plots and $1.14 \pm 0.03 \text{ g}\cdot\text{cm}^{-3}$ in cattle exclosures ($P = 0.35$). Average bulk density of the Bt2 horizon was $0.96 \pm 0.02 \text{ g}\cdot\text{cm}^{-3}$ for both control plots and cattle exclosures ($P = 0.83$).

There was no evidence of differences in mean bulk density for any of the three soil depths examined among exclosure treatments at the subset of locations where CAJ exclosures were included, using a 1-way ANOVA. Bulk density values for the three soil depths examined among the three exclosure treatments were: A horizon ($P = 0.73$), Bt1 horizon ($P = 0.70$), and Bt2 horizon ($P = 0.67$). Average bulk density and standard error for each depth and treatment are given (Table 2.5). Numerically, mean bulk density at all depths was highest in control plots. Overall, average bulk density across all treatments was highest for the Bt1 horizon, followed by the A horizon, and lastly by the Bt2 horizon.

Table 2.5. Mean soil bulk density and standard errors by horizon, for the subset of locations where CAJ exclosures were included ($n = 5$).

Treatment	Mean Soil Bulk Density ($\text{g}\cdot\text{cm}^{-3}$) by Soil Depth Layer		
	A Horizon	Bt1 Horizon	Bt2 Horizon
Control	1.12 ± 0.02	1.18 ± 0.02	0.97 ± 0.03
Cattle Exclosure	1.09 ± 0.03	1.14 ± 0.04	0.93 ± 0.04
CAJ Exclosure	1.10 ± 0.04	1.15 ± 0.05	0.96 ± 0.03

Soil Nutrients

A two-sample t-test showed no differences in the A1 horizon for total organic carbon, total organic nitrogen, or plant available inorganic phosphorus between controls and cattle exclosure treatments ($P > 0.60$; $df = 30$) in the year 2007. Mean nutrient values and standard errors for controls and cattle exclosure treatments are presented (Table 2.6). Similarly, there was no evidence of a difference in soil nutrients in 2007 among the three exclosure treatments at the subset of locations where CAJ exclosures were included ($P > 0.50$; $df = 2$), using a 1-way ANOVA. Mean nutrient values and standard errors are presented for the exclosure subset (Table 2.7).

Table 2.6. Mean values and standard errors for organic carbon, organic nitrogen, and plant available inorganic phosphorus between controls and cattle exclosure treatments ($n = 16$) in the A soil horizon.

Treatment	Mean Soil Nutrient Values in the A Horizon		
	Organic Carbon (%)	Organic Nitrogen (%)	Phosphorus (ppm)
Control	0.90 ± 0.07	0.08 ± 0.01	28.1 ± 2.8
Cattle Exclosure	0.91 ± 0.08	0.09 ± 0.01	27.3 ± 2.6

Table 2.7. Mean values and standard errors for organic carbon, organic nitrogen, and plant available inorganic phosphorus between controls, cattle exclosures, and CAJ exclosure treatments ($n = 5$) in the A soil horizon.

Treatment	Mean Soil Nutrients Values in the A Horizon		
	Organic Carbon (%)	Organic Nitrogen (%)	Phosphorus (ppm)
Control	0.73 ± 0.11	0.07 ± 0.01	26.3 ± 3.4
Cattle Exclosure	0.88 ± 0.16	0.09 ± 0.01	30.5 ± 4.9
CAJ Exclosure	0.72 ± 0.06	0.07 ± 0.01	25.6 ± 3.2

Soil pH

Data on pH were only collected in 2007; five years after exclosure treatments were established. A two-sample t-test showed no differences between controls and cattle exclosures for any of the four soil depths measured ($P > 0.30$; $df = 30$). Further, there were no differences in soil pH among the three exclosure treatments at the subset of locations where CAJ exclosures were included ($P > 0.80$; $df = 2$), using a 1-way ANOVA. Mean pH and standard errors for treatments in the two-sample t-test (Table 2.8) and 1-way ANOVA (Table 2.9) are listed. Actual depths of soil pH measurements were recorded. Overall average depths that soil pH samples were collected for soil layers across the valley were: A horizon = 1.8 cm, transition between the A horizon and Bt1 horizon = 3.6 cm, and Bt1 horizon = 6.8 cm. Measurements for the Bt2 horizon were taken at depths averaging greater than 10.8 cm. Thickness of the Bt1 horizon averaged approximately 7.2 cm across the valley, while the average thickness of the A horizon was 3.6 cm.

Table 2.8. Mean pH values and standard errors for controls and cattle exclosure treatments ($n = 16$) at four different soil depths.

Treatment	Mean Soil pH			
	A Horizon	A/Bt1 Transition	Bt1 Horizon	Bt2 Horizon
Control	7.62 \pm 0.03	7.59 \pm 0.04	7.61 \pm 0.04	7.62 \pm 0.05
Cattle Exclosure	7.61 \pm 0.06	7.57 \pm 0.05	7.61 \pm 0.05	7.70 \pm 0.07

Table 2.9. Mean pH values and standard errors for controls, cattle exclosures, and CAJ exclosure treatments ($n = 5$) at four different soil depths.

Treatment	Mean Soil pH			
	A Horizon	A/Bt1 Transition	Bt1 Horizon	Bt2 Horizon
Control	7.54 \pm 0.03	7.55 \pm 0.07	7.56 \pm 0.04	7.60 \pm 0.08
Cattle Exclosure	7.53 \pm 0.07	7.50 \pm 0.08	7.54 \pm 0.08	7.64 \pm 0.10
CAJ Exclosure	7.54 \pm 0.03	7.55 \pm 0.07	7.58 \pm 0.05	7.64 \pm 0.08

Soil Penetrometer Resistance

Impact penetrometer data from 2002 were compared to data from 2007 for all exclosure treatments. The number of strikes needed to move the penetrometer from the soil surface to a depth of 5 cm increased in control study plots by approximately 1 strike on average ($P = 0.001$). Strikes required in CAJ exclosures did not change ($P = 0.59$), but there was moderate evidence to suggest the number of strikes needed inside

exclosures decreased by an average value of 0.34 strikes ($P = 0.05$). No study plot treatments showed evidence of a change within the 5 to 10 cm soil depth range ($P > 0.10$). However, there was moderate to overwhelming evidence that the number of strikes required to penetrate through the 10-15 cm and 15-20 cm depth ranges were reduced for all study plot treatments (Table 2.10). Penetrometer data was also compared across treatments within year, since differences in soil moisture can affect resistance to penetration and soil moisture was not accounted for when taking penetrometer measurements. Relative differences in penetration resistance were determined between exclosures in 2002 and then again within year in 2007. This was done to determine if exclosures were more similar to each other prior to the establishment of exclosures as compared to five years afterwards.

Table 2.10. Average impact penetrometer data (number of strikes) for 2002 and 2007 for three different exclosure treatments.

Treatment	Depth Range	n	df	2002	SE	2007	SE	Mean Change	P	95% CI
Control	0 cm to 5 cm	15	14	2.45	0.2	3.4	0.3	0.95	0.001	(0.44 to 1.45)
	5 cm to 10 cm	15	14	7.89	0.6	9.32	0.9	1.43	0.17	
	10 cm to 15 cm	15	14	15.74	1	11.8	0.9	-3.99	0.01	(-7.01 to -0.96)
	15 cm to 20 cm	15	14	26.49	1.3	14.6	1.1	-11.92	< 0.0001	(-15.68 to -8.17)
Cattle Exclosure	0 cm to 5 cm	15	14	2.73	0.3	2.39	0.3	-0.34	0.05	(-0.67 to -0.01)
	5 cm to 10 cm	15	14	9.04	0.8	8.07	0.6	-0.97	0.19	
	10 cm to 15 cm	15	14	17.46	1.1	11.6	0.6	-5.89	0.0006	(-8.79 to -3.00)
	15 cm to 20 cm	15	14	28.29	1.3	14.6	0.9	-13.71	< 0.0001	(-17.69 to -9.72)
CAJ Exclosure	0 cm to 5 cm	5	4	2.26	0.4	2.49	0.6	0.23	0.59	
	5 cm to 10 cm	5	4	6.88	1	6.69	1.4	-0.19	0.84	
	10 cm to 15 cm	5	4	14.1	1.1	10	1.4	-4.10	0.03	(-7.63 to -0.56)
	15 cm to 20 cm	5	4	25.03	1.7	13.8	1.3	-11.22	0.0004	(-14.13 to -8.30)

A two-sample t-test of penetrometer data from 2002 showed no differences between controls and cattle exclosures for any of the four soil depths measured ($P > 0.25$; $df = 28$). There was also no evidence of differences in penetrometer values among the three exclosure treatments at the subset of locations where CAJ exclosures were included ($P > 0.60$; $df = 2$), using a 1-way ANOVA. Mean penetrometer strikes

and standard errors for treatments in the two-sample t-test (Table 2.11) and 1-way ANOVA (Table 2.12) are listed.

Table 2.11. Mean penetrometer strikes and standard errors for controls and cattle exclosure treatments ($n = 16$) at four different soil depths in the year 2002.

Treatment	Mean Penetrometer Strikes by Soil Depth			
	0-5 cm	5-10 cm	10-15 cm	15-20 cm
Control	2.45 ± 0.21	7.89 ± 0.63	15.74 ± 0.96	26.49 ± 1.32
Cattle Exclosure	2.73 ± 0.26	9.04 ± 0.79	17.46 ± 1.12	28.29 ± 1.25

Table 2.12. Mean penetrometer strikes and standard errors for controls, cattle exclosures, and CAJ exclosure treatments ($n = 5$) at four different soil depths in the year 2002.

Treatment	Mean Penetrometer Strikes by Soil Depth			
	0-5 cm	5-10 cm	10-15 cm	15-20 cm
Control	2.59 ± 0.04	7.55 ± 1.04	15.09 ± 1.20	26.40 ± 1.66
Cattle Exclosure	2.59 ± 0.60	8.16 ± 1.48	16.23 ± 1.91	27.41 ± 2.05
CAJ Exclosure	2.26 ± 0.42	6.88 ± 1.01	14.10 ± 1.12	25.03 ± 1.65

A two-sample t-test of penetrometer data from 2007 showed that the zone from 0-5 cm differed between controls and cattle exclosures by approximately 1 strike ($P = 0.02$; $n = 16$). For the remaining deeper three soil depths measured there were no statistical differences between controls and cattle exclosures ($P > 0.28$; $df = 30$).

There was also no evidence of differences in penetrometer values for any of the four depths among the three exclosure treatments at the subset of locations where CAJ exclosures were included ($P > 0.50$; $df = 2$), using a 1-way ANOVA. However, numerically, the control required 1 more strike, on average, to penetrate the 0-5 cm zone. Mean penetrometer strikes and standard errors for treatments in the two-sample t-test (Table 2.13) and 1-way ANOVA (Table 2.14) are listed.

Table 2.13. Mean penetrometer strikes and standard errors for controls and cattle exclosure treatments ($n = 15$) at four different soil depths in the year 2007.

Treatment	Mean Penetrometer Strikes by Soil Depth			
	0-5 cm	5-10 cm	10-15 cm	15-20 cm
Control	3.40 ± 0.31	9.32 ± 0.93	11.75 ± 0.88	14.57 ± 1.12
Cattle Exclosure	2.39 ± 0.25	8.07 ± 0.63	11.57 ± 0.64	14.58 ± 0.89

Table 2.14. Mean penetrometer strikes and standard errors for controls, cattle exclosures, and CAJ exclosure treatments ($n = 5$) at four different soil depths in the year 2007.

Treatment	Mean Penetrometer Strikes by Soil Depth			
	0-5 cm	5-10 cm	10-15 cm	15-20 cm
Control	3.48 ± 0.80	8.51 ± 1.43	11.10 ± 0.98	15.66 ± 2.19
Cattle Exclosure	2.50 ± 0.65	7.48 ± 1.14	10.89 ± 0.61	14.29 ± 1.53
CAJ Exclosure	2.49 ± 0.64	6.69 ± 1.39	10.00 ± 1.35	13.81 ± 1.31

Gap-Line-Intercept

Gap-line-intercept data from 2002 were compared to data from 2007 for all exclosure treatments. Gap-line-intercept data were partitioned into two categories: 1) perennial vegetation and 2) not vegetated (bare patch). Perennial vegetation was partitioned into three additional categories: 1) percent winterfat, 2) percent sickle saltbush, 3) percent shrubs, and 4) percent perennial grasses.

In control plots the average size of bare patches increased from 1.61 m in 2002 to 2.60 m ($P = 0.01$; Table 2.15) in 2007. This was an increase in bare patch size of 61%. Over the same period, total percent of the line transect that consisted of perennial vegetation decreased from 6.6% to 3.7% ($P = 0.007$). From 2002 to 2007, the total percent of line transect consisting of winterfat decreased from 2.4% to 1.5% ($P = 0.02$) while perennial grasses decreased from 2.2% to 0.4% ($P = 0.002$). Total shrub cover decreased from 4.4% to 3.3% ($P = 0.10$). Sickie saltbush was the only category not found to differ between 2002 and 2007 in control plots ($P = 0.75$).

There did not appear to be any changes between 2002 and 2007 in bare patch size, percent bare patch, or percent perennial vegetation where exclosures were used (Table 2.15). A finer review of perennial vegetation categories however revealed that inside cattle exclosures the percentage of perennial grasses decreased from 4.0% to 1.2% ($P = 0.002$). There was also marginal evidence that perennial grasses decreased from 2.2% to 0.3% ($P = 0.13$) inside CAJ exclosures. There was moderate evidence to suggest that percentage of shrubs increased from 5.1 to 6.5% ($P = 0.06$) inside cattle exclosures, likely due to increases in sickie saltbush ($P = 0.03$) especially since percent winterfat was not found to change from 2002 to 2007 ($P = 0.67$). There was also evidence to suggest that percentage of shrubs increased from 6.3% to 9.6% ($P = 0.06$) inside CAJ exclosures. The mean increase in the percentage of shrubs from 2002 to 2007 was likely due to mean increases in winterfat (2.7%; $P = 0.13$) with an additional slight increase in mean sickie saltbush (0.6%; $P = 0.22$).

Table 2.15. Averaged gap-line-intercept data for 2002 and 2007 for 3 different enclosure treatments.

Treatment	Gap-Line-Intercept Category	n	df	2002	SE	2007	SE	Mean		
								Change	P	95% CI
Control	Mean Size of Bare Patch Between Perennial Plants (m)	16	15	1.61	0.21	2.60	0.36	0.99	0.01	(0.25 to 1.72)
	Mean Percent Bare Patch (%)	16	15	93.4	1.1	96.3	0.6	2.9	0.007	(0.93 to 4.87)
	Mean Percent Winterfat (%)	16	15	2.4	0.7	1.5	0.4	-0.9	0.02	(-0.19 to -1.75)
	Mean Percent Sickle Saltbush (%)	16	15	2.0	0.5	1.8	0.5	-0.2	0.75	
	Mean Percent Shrubs (%)	16	15	4.4	0.7	3.3	0.6	-1.1	0.10	
	Mean Percent Perennial Grasses (%)	16	15	2.2	0.6	0.4	0.2	-1.8	0.002	(-0.74 to -2.81)
	Mean Percent Perennial Vegetation (%)	16	15	6.6	1.1	3.7	0.6	-2.9	0.007	(-0.93 to -4.87)
Cattle Enclosure	Mean Size of Bare Patch Between Perennial Plants (m)	16	15	1.31	0.15	1.37	0.16	0.06	0.43	
	Mean Percent Bare Patch (%)	16	15	90.9	1.3	92.2	0.9	1.3	0.25	
	Mean Percent Winterfat (%)	16	15	2.6	0.7	2.8	0.7	0.2	0.67	
	Mean Percent Sickle Saltbush (%)	16	15	2.4	0.6	3.7	0.9	1.3	0.03	(0.14 to 2.5)
	Mean Percent Shrubs (%)	16	15	5.1	0.7	6.5	0.9	1.4	0.06	
	Mean Percent Perennial Grasses (%)	16	15	4.0	1.1	1.2	0.4	-2.8	0.002	(-1.20 to -4.27)
	Mean Percent Perennial Vegetation (%)	16	15	9.1	1.3	7.8	0.9	-1.3	0.25	
CAJ Enclosure	Mean Size of Bare Patch Between Perennial Plants (m)	5	4	1.35	0.27	1.33	0.33	-0.02	0.86	
	Mean Percent Bare Patch (%)	5	4	91.5	2.0	90.1	2.8	-1.4	0.49	
	Mean Percent Winterfat (%)	5	4	4.8	2.1	7.5	3.5	2.7	0.13	
	Mean Percent Sickle Saltbush (%)	5	4	1.5	1.1	2.1	1.5	0.6	0.22	
	Mean Percent Shrubs (%)	5	4	6.3	1.8	9.6	2.9	3.3	0.06	
	Mean Percent Perennial Grasses (%)	5	4	2.2	1.1	0.3	0.2	-1.9	0.13	
	Mean Percent Perennial Vegetation (%)	5	4	8.5	2	9.90	2.8	1.40	0.49	

Point Intercept

Point intercept data from 2002 were compared with data from 2007 for all three exclosure treatments using paired t-tests. Point intercept results for each treatment are partitioned into several categories (Table 2.16). Data collected for biological soil crust, bareground, and soil crack categories in 2002 were found to have errors, and analysis of these categories were compared between exclosure treatments using a 1-way ANOVA for 2007 only.

Control 2002 vs. 2007

Litter, sickle saltbush, winterfat, and perennial grasses displayed evidence of change from 2002 to 2007 where exclosures were not used. There was strong evidence that the amount of litter ($P = 0.0002$) and foliar cover of sickle saltbush doubled ($P = 0.03$). Over the same period, there was moderate to strong evidence that foliar cover of winterfat ($P = 0.07$) and perennial grasses ($P = 0.02$) declined by approximately half (Table 2.16).

Cattle Exclosure 2002 vs. 2007

There was evidence that cattle exclosures increased in foliar cover of perennial vegetation ($P = 0.003$), perennial shrubs ($P < 0.0001$), and litter cover ($P < 0.0001$). Increases in perennial vegetation and shrubs are evidently due to increases in both foliar cover of winterfat ($P = 0.01$) and sickle saltbush ($P = 0.0005$). In addition, the amount of cow dung was cut in half from 0.4% to 0.19% ($P = 0.05$), while rabbit dung increased from 0% to 0.35% ($P = 0.04$; Table 16). Litter increased from 17.2% to 43.4%, and perennial vegetation doubled from 5.1% to 10.1%.

CAJ Exclosure 2002 vs. 2007

There was some evidence that litter doubled from 16% to 32.6% ($P = 0.09$). Concurrently, perennial vegetation more than doubled from 4.3% to 10.2% ($P = 0.01$),

evidently due to an increase in perennial shrub cover ($P = 0.02$). Increases in shrub cover may be attributed to a doubling in winterfat foliar cover ($P = 0.09$; Table 2.16).

Table 2.16. Point-intercept data for 2002 and 2007 for three different exclosure treatments.

Treatment	Category	n	df	2002	SE	2007	SE	Mean Change	P	95% CI
Control	Cow Dung	16	15	0.3	0.1	0.3	0.1	0.0	0.95	
	Litter of Vegetation	16	15	15.0	0.4	33.0	2.8	18.0	0.0002	(10.0 to 26.0)
	Moss	16	15	0.4	0.2	0.1	0.1	-0.3	0.13	
	Rabbit Dung	16	15	0.0	0.0	0.0	0.0	0.0	NA	
	Rock	16	15	0.6	0.3	0.7	0.5	0.2	0.60	
	Sickle Saltbush	5	4	0.7	0.2	1.7	0.6	1.0	0.03	(0.1 to 2.0)
	Winterfat	5	4	2.1	0.6	1.2	0.4	-0.9	0.07	
	Annual Vegetation	16	15	0.3	0.2	0.1	0.1	-0.3	0.21	
	Perennial Grasses	5	4	1.6	0.5	0.6	0.2	-1.0	0.02	(-0.2 to -1.7)
	Perennial Shrubs	5	4	2.8	0.6	2.9	0.6	0.1	0.88	
	Perennial Vegetation	16	15	4.8	0.7	4.5	0.6	-0.3	0.75	
Cattle Exclosure	Cow Dung	16	15	0.4	0.1	0.2	0.1	-0.2	0.05	(0.0 to -0.4)
	Litter of Vegetation	16	15	17.2	1.6	43.4	3.4	26.1	< 0.0001	(16.9 to 35.4)
	Moss	16	15	0.3	0.3	0.0	0.0	-0.3	0.33	
	Rabbit Dung	16	15	0.0	0.0	0.4	0.2	0.4	0.04	(0.0 to 0.7)
	Rock	16	15	1.4	0.7	0.7	0.4	-0.7	0.28	
	Sickle Saltbush	5	4	0.5	0.2	3.2	0.7	2.7	0.0005	(1.4 to 4.0)
	Winterfat	5	4	1.7	0.5	3.1	0.6	1.4	0.01	(0.4 to 2.4)
	Annual Vegetation	16	15	0.4	0.2	0.4	0.2	0.1	0.80	
	Perennial Grasses	5	4	2.6	0.7	2.7	1.0	0.1	0.91	
	Perennial Shrubs	5	4	2.2	0.5	6.3	0.7	4.1	< 0.0001	(5.6 to 2.6)
	Perennial Vegetation	16	15	5.1	1.0	10.1	1.2	5.0	0.003	(2.7 to 7.2)
CAJ Exclosure	Cow Dung	5	4	0.0	0.0	0.0	0.0	0.0	NA	
	Litter of Vegetation	5	4	16.0	4.0	32.6	7.2	16.7	0.09	
	Moss	5	4	0.4	0.3	0.0	0.0	-0.4	0.21	
	Rabbit Dung	5	4	0.0	0.0	0.0	0.0	0.0	na	
	Rock	5	4	1.9	1.2	0.7	0.6	-1.2	0.19	
	Sickle Saltbush	5	4	0.4	0.4	2.6	1.8	2.2	0.18	
	Winterfat	5	4	3.0	1.7	6.3	2.6	3.3	0.09	
	Annual Vegetation	5	4	0.0	0.0	0.3	0.3	0.3	0.37	
	Perennial Grasses	5	4	0.7	0.4	0.6	0.3	-0.1	0.78	
	Perennial Shrubs	5	4	3.4	1.5	8.9	2.1	5.5	0.02	(9.7 to 1.3)
	Perennial Vegetation	5	4	4.3	1.3	10.2	1.8	5.8	0.01	(2.1 to 9.6)

Biological Soil Crust, Bareground, and Soil Cracks

Biological soil crust, bareground, and soil crack point-intercept data for exclosure treatments were compared in 2007 only. Means and standard errors for control plots and cattle exclosures ($n = 16$) are presented in Table 2.17. A two-sample t-test showed that the amount of bareground (bare soil devoid of biological soil crust or vegetation) in control plots was more than two times higher than in cattle exclosures ($P < 0.001$; $df = 30$). Additionally, biological soil crust was 3.5 times higher in cattle exclosures (21.8%) when compared to control plots (6.3%; $P < 0.001$; $df = 30$). There was no evidence to suggest that the amount of soil cracks differed between control plots and cattle exclosure treatments ($P = 0.54$; $df = 30$). There were also no differences in amount of soil cracks among the three exclosure treatments at the subset of locations where CAJ exclosures were included ($P = 0.55$ and $df = 2$; Table 2.18). Some evidence was found to suggest that biological soil crust differed among the subset that included CAJ exclosures ($P = 0.08$; $df = 2$). A mean separation test at the 95% confidence level confirmed that cattle exclosures had higher levels of biological crust when compared to control plots but that CAJ exclosures did not differ from either control plots or cattle exclosures. Lastly, bareground was found to be two times higher for control plots over CAJ and cattle exclosures ($P = 0.004$), and this was confirmed with a mean separations test at the 90% confidence level (Table 2.18).

Table 2.17. Point-intercept means and standard errors for controls and cattle exclosure treatments ($n = 16$) in the year 2007. Letters a and b signify mean differences between treatments.

Category	Control	SE	Cattle	SE	P
Bareground	52.1 a	2.9	20.7 b	4.1	< 0.001
Biological Crust	6.3 a	1.9	21.8 b	3.6	< 0.001
Soil Cracks	2.9	0.6	2.4	0.7	0.54

Table 2.18. Point-intercept means and standard errors for controls, cattle, and CAJ exclosure treatments ($n = 5$) in the year 2007. Letters a and b signify mean differences between treatments.

Category	Control	SE	Cattle	SE	CAJ	SE	P
Bareground	49.5 a	4.4	20.5 b	5.4	23.7 b	5.8	0.004
Biological Crust	9.3 a	4.2	32.2 b	6.7	28.6 ab	9.3	0.08
Soil Cracks	3.1	0.5	2.7	1.0	4.0	0.8	0.55

DISCUSSION

The upper 5 cm of soil in control plots became harder (more resistant to penetration) from year 2002 to 2007 as measured by cone penetrometer readings. The link between soil compaction and cattle-use is well known (Drewry and Paton 2000, Drewry *et al.* 2004, Drewry *et al.* 2008, Greenwood And Mckenzie 2001, Greenwood *et al.* 1998, Lamond 1998, Mcdowell *et al.* 2003a, Mcdowell *et al.* 2003b, Murphy *et al.* 2004, Nie *et al.* 2001, Singleton *et al.* 2000, and Stavi *et al.* 2008). The resistance to penetration for the soil depth range of 10 to 20 cm decreased for all treatments. Statistically significant increases in penetrometer readings in control plots but a general lack of differences in bulk density between exclosure treatments may be because penetrometer resistance as measured by the cone penetrometer is a finer measure of soil compaction or another explanation is that sampling effort (number of samples) of bulk density were insufficient to detect a change. The finding of reduced resistance for the deeper soil depths is probably due to higher soil moisture at those depths in year 2007 at the time of sampling versus the time of sampling in 2002. However, soil moisture was not recorded at the time of sampling in either year, so we cannot say with certainty that soil moisture explains the differences between years at deeper soil depths, but soil moisture is known to affect cone penetrometer readings (Herrick *et al.* 2005).

There were no statistical differences in soil pH, soil nutrients, soil bulk density, or relative infiltration rates between treatments, as measured in year 2007. McCune and Ponzetti (2001) also did not find differences in pH between grazed and ungrazed

plots in eastern Oregon. Although differences in total organic carbon, total organic nitrogen, and plant available inorganic phosphorus were not found to differ between exclosure treatments, other studies have linked reductions in organic carbon and organic nitrogen to long-term grazing (Nef *et al.* 2005). This loss of nutrients is induced by loss of soil due to wind erosion or water erosion (Nef *et al.* 2005, Graetz and Tongway 1986, McDowell *et al.* 2003a, and McDowell *et al.* 2003b). Since wind erosion is a well known process in the Catlow Valley (Jackman and Scharff 1967), loss of nutrients over longer timescales of cattle-use could occur but do not appear to pose a serious risk over periods as long as five years, as shown in this study. Further, early accounts of wind erosion occurring (Jackman and Scharff 1967), and the known disturbance by both farmers and cattle grazing in the area (Allen 1987, Jackson 1978, Simpson 1987, French 1964, and Shirk 1956) could have resulted in significant soil loss, many years prior to the initiation of this study. Such soil loss could have served to dampen the detection of differences in soil nutrients across the study area.

Average relative infiltration rates were numerically highest for CAJ exclosures but did not differ statistically from cattle exclosures and control plots. Larger standard errors in exclosed treatments might suggest that there were some improvements in infiltration at some exclosed areas but that they were sporadic. Infiltration rates in summer may not necessarily reflect infiltration rates in winter, since the soils in summer may contain cracks that allow for greater infiltration relative to winter months when soils are moist and have less cracks (Drewry *et al.* 2004 and Elliot *et al.* 2002). Knowing this, it is possible that relative infiltration rates measured in winter could reveal differences between exclosure treatments that were not apparent during summer.

Aggregate stability was higher for both cattle and CAJ treatments over control plots in 2007. Differences in aggregate stability may be due to increases in stability in exclosed plots and/or the lessening of stability in control plots over the five years of this study. Since there were no accurate measures of biological crust in 2002, it is difficult to determine if biological soil crust was a factor. But, increases in aggregate

stability are known to be linked to increases in biological soil crust where crusts are present (Greene *et al.* 1990 and Williams *et al.* 1995), and biological soil crust was found to be much lower in control plots in 2007 when compared to exclosed plots.

Bulk densities were slightly numerically higher in control plots. This could suggest that control plots are becoming more compact relative to exclosure treatments, when also considering the known increase in soil resistance in control plots from the cone penetrometer data. We hypothesize that insufficient time has elapsed to incur a significant change in most soil properties between treatments due to the aridity of the research area. In wetter climates, substantial improvements in saturated hydraulic conductivity and soil macroporosity have been documented in as few as 18 months following reductions in cattle-use (Drewry and Paton 2000). Another study in a wetter climate has shown that bulk density and macroporosity can be affected positively or negatively in a single season according to cattle exclusion or cattle-use, respectively (Drewry *et al.* 2004). In contrast, semi-arid areas may take several years to see decreases in bulk density following compaction by livestock (Braunack and Walker 2004). Natural processes that can improve soil bulk density include freeze and thaw cycles, wetting and drying cycles, as well as activities by soil fauna and growth/death of plant roots (Dexter 1991). These processes would likely take many years to modify the soil in the Catlow Valley where plant growth is limited by a short growing season and where annual precipitation is quite low. Most of the years in this study were relatively dry, and if there had been more precipitation, multiple years receiving normal or above normal precipitation, we might have seen more changes in soil properties.

All study treatments, exclosures and controls, showed decreases in most annual plant density and frequency, which are likely the product of drought occurring during the water year of 2007. The only evidence of increases in plant density was of cheatgrass and gooseberry globemallow inside cattle exclosures. Competition of cheatgrass (Freeman and Emlen 1995) and annual plants (West 1985) is known to reduce winterfat seedling survival and seed production. Increases in cheatgrass may

also alter the fire regime of the site by increasing fuel loads and make winterfat communities more susceptible to fire (Brooks *et al.* 2004). Cheatgrass density didn't change in control plots or CAJ exclosure treatments, but cheatgrass frequency was reduced in CAJ exclosures. This evidence suggests that cheatgrass may increase in areas that are protected from cattle grazing if cheatgrass is all ready present at the time cattle-use ceases. Subsequently, cattle grazing might suppress cheatgrass from proliferating, but it is not clear why CAJ exclosures did not follow the same response as cattle exclosures, and instead, CAJ exclosures showed evidence of decreases in cheatgrass frequency. Jackrabbit activity in cattle exclosures, as indexed by increases in jackrabbit dung, may account for some of the seed dispersal and increases in cheatgrass density. This might explain why increases in cheatgrass density did not occur in CAJ exclosures.

There was little evidence of change in winterfat density or frequency during this study for any treatment. From the data we collected, winterfat appears to be highly resistant to cattle and wildlife-use since reductions in plant cover had not yet resulted in significant changes in density or frequency when control plots were compared to exclosure plots. However, winterfat cover in control plots were reduced to lower levels, compared to when the study began five years ago. And, further use at current rates of herbivory may deplete plant size to the point that mortality will increase at some time in the near future. Control plots and cattle exclosures showed reduction of squirreltail in both frequency and density, but there did not appear to be any change in CAJ exclosures. The lack of reduction in squirreltail in CAJ exclosures might be explained by the lack of herbivory by jackrabbits. The combined effect of jackrabbit herbivory and drought may be the mechanisms of decreases in squirreltail in control plots and cattle exclosure treatments. This is very possible since squirreltail is known favored food by black-tailed jackrabbits (Anderson and Shumar 1986), an abundant grass in the valley, and jackrabbits are known to make use of new growth on grasses in the spring (Johnson and Anderson 1984). Basin wildrye was reduced in control plots and may represent its lack of resistance on these sites to

cattle/antelope/jackrabbit herbivory and/or drought. Others have found that basin wildrye responds poorly to grazing early in spring or during fall regrowth (Lesperance *et al.* 1978 and Perry and Chapman 1975), and the current grazing regime could be slightly reaching over both these periods in the Catlow Valley.

Gap-line-intercept data showed that control plots increased in bare area and decreased in perennial vegetation. A finer look at the data confirmed that the decreases in perennial vegetation were mediated by decreases in winterfat cover and perennial grass cover. These changes are not surprising, since increases in bare areas and decreases in perennial plant cover are known responses to both drought and cattle-use (Scherrer and Pickering 2005, Morgan 2004, Reynolds *et al.* 1999, and Schwinning *et al.* 2005), which occurred simultaneously during this study. Decreases in perennial plant cover could also be related to cattle hoof impact, which is known to have strong negative effects on plant growth (Cannell 1977, Carter 1988, Drewry *et al.* 2008, and Drewry *et al.* 2001). The only plant that seemed unaffected was sickle saltbush. This may have occurred from cattle and wildlife selecting winterfat preferentially over sickle saltbush or perhaps because sickle saltbush drops its leaves during winter and initiates its growth earlier than other associated plants (Goodman 1973). It could also occur because sickle saltbush is a vigorous sprouter following disturbance (Nord *et al.* 1969). The uncertainty of why sickle saltbush was unaffected in control plots is increased by the fact that Estes (2008) found that sickle saltbush comprised much of the diet of cattle in the Catlow Valley, even though winterfat was the preferred forage. Bare patch size and amount of bare patches in exclosed plots were unaffected over the five years of this study. The total amount of perennial vegetation in exclosed plots were also unaffected. What did occur were shifts in plant cover: perennial grasses decreased and shrubs increased. This shift in vegetative cover is understandable since shrubs have been found to compose much of the diet of cattle in the Catlow Valley during winter months when they are accessible (Estes 2008). The overall decrease in perennial grasses across exclosures did not appear to be

dependent on exclosure treatment but were more likely in response to drought occurring during the study.

In addition to gap-line-intercept data, point-intercept data verified that continued use by cattle and wildlife in control plots resulted in reductions in winterfat foliar cover by nearly half from year 2002 to 2007. Other studies have also documented reductions in winterfat canopy cover when browsed during winter months (Rasmussen and Brotherson 1986, Whisenant and Wagstaff 1991, and Blaisdell and Holmgren 1984). Many other studies have documented negative effects of cattle browsing on winterfat when grazed at other times of the year (Romo *et al.* 1995, Smith and Beale and 1980, Stevens *et al.* 1977, Krysl *et al.* 1994, Smoliak 1965, Dormaar *et al.* 1994, Cook and Child 1971, Eckert 1954, Cook and Stoddart 1963). Some research has noted a neutral response or tolerance to cattle browsing (Stevens *et al.* 1977, Blaisdell and Holmgren 1984, Woodmansee and Potter 1971, Smoliak 1965, West 1979, Romo *et al.* 1995, Chambers and Norton 1993, Eckert 1954, Cook and Stoddart 1963), and few have found positive effects on winterfat as a result of cattle browsing (Yorks *et al.* 1992 and Clark 1986). Winterfat plants in both of the exclosed treatments approximately doubled in foliar cover during the same period that control plots decreased. Winterfat cover increased inside cattle exclosures even though they provided access to jackrabbits and jackrabbit-use, as indexed by the increases in jackrabbit dung that occurred over the same period. Reductions in winterfat cover from year 2002 to 2007 in control plots may have been exacerbated by low precipitation occurring during the winter of 2006-2007. Eckert (1954) also found that winterfat growth was diminished during periods of drought. In all, the data collected in this study suggest that existing winterfat plants in the Catlow Valley responded positively to release from herbivory. Exclusion of jackrabbits and pronghorn antelope did not appear to result in further gains in winterfat cover when compared to cattle exclosures alone. This would seem to indicate that jackrabbit and antelope use were negligible at current levels. However, use patterns by wildlife could change with time and in accordance with changes in wildlife numbers.

Biological soil crusts in control plots during 2007 were approximately 4 times lower than protected areas. This reduction or lower levels of crust in areas used by cattle is consistent with findings from rangeland ecosystems in Oregon (Ponzetti and McCune 2001), the southwestern United States (Anderson et al. 1982, Beymer and Klopatek 1992, Johansen and St. Clair 1986), and Australia (Andrew and Lange 1986, Graetz and Tongway 1986, Hodgins and Rogers 1997). We hypothesize that low levels of biological soil crust in control plots were due to soil disturbance mediated by cattle hoof impact. Cattle hoof impact in semi-arid areas may directly damage plants and pulverize the soil surface (Greenwood et al. 1994). This could be exacerbated by the years of drought that occurred during this study as well as wind and water erosion. All treatments increased in amounts of litter cover. We attribute increases in litter to accumulations of annual plant material, which accumulated during the recent drought period, and also due to reductions in the amount of annual plant production of living plants during the year of 2007 when vegetation sampling occurred. This pattern of litter accumulation and decrease in living vegetation (herbaceous plants) is a typical response to drought (Scherrer and Pickering 2005). Cattle exclosures and CAJ exclosures increased in perennial vegetation due to increases in shrub cover of winterfat and sickle saltbush as measured by line-point-intercept. Increases in perennial shrubs were attributed to the release from herbivore disturbance, both browsing and hoof impact. Lower levels of bareground in exclosed plots were attributed to high levels of biological soil crust, litter, and perennial vegetation. McCune and Ponzetti (2001) also found plant cover of perennial plants to be higher in exclosed plots versus grazed plots in big sagebrush communities in eastern Oregon.

Changes in abundance of dung have been used to gauge use of areas by herbivores and estimate herbivore numbers (Rogers *et al.* 1958 and Wood 1988). Cattle exclosures showed decreases in amount of cow dung but showed increases in amount of rabbit dung as measured by line-point-intercept. This is explained by the lack of access by cattle and the subsequent decomposition of cattle dung over the five year period of this study. In contrast, the increase in jackrabbit dung inside cattle

exclosures may be due to the creation of preferred habitat that contained more food and cover as opposed to the areas outside the exclosures. Pronghorn antelope dung was not present in point-intercept measurements at all locations, and this lack of dung likely reflects the light-use by antelope in the winterfat communities across the valley. Further, relatively few antelope were observed frequenting the areas of research during this study.

Plant age frequency histograms reveal that most of the winterfat shrubs in the study area are over 30 years of age and have had little to no reproduction over the last 19 years. Examination of sickle saltbush shrubs shows saltbush has been actively reproducing over the last several years and that most plants are under 30 years of age. This suggests that saltbush populations are healthy, but that conditions in current and recent history have been unfavorable for winterfat reproduction. This may be a product of variable precipitation and the recent history of disturbance. It is unclear what to do to increase winterfat reproduction, since seed viability was quite low in exclosures in the Catlow Valley, even after 4 years of protection from cattle and wildlife (Leary 2004). Stimulating winterfat reproduction may have as much to do with eliminating plant competition as much as browse-use or climate. Saltbush may be excluding and lessening the ability of winterfat to grow and reproduce. A study of winterfat vs. saltbush competition seems warranted. Exclosure treatments should be reexamined in 5 to 10 years to determine if changes occur and if winterfat reproduction is positively affected. McCune and Ponzetti (2001) did not find differences in plant species composition between grazed and ungrazed plots, but they did find differences in the amount of biological soil crust. Ungrazed plots had been protected for more than 30 years, which suggests that crusts are likely to be more resilient to change than plant species composition. Quinfeng (2004) found in Arizona that it may take 50 years or more after grazing ceases for the perennial vegetative community to recover.

MANAGEMENT IMPLICATIONS

Our study suggests that continued use of winterfat communities by cattle and wildlife during the fall/winter period in the Catlow Valley increased soil compaction as indexed by soil cone penetrometer resistance and increased the amount of bare area between perennial plants and over a 5 year period from year 2002 to 2007. Areas open to cattle were also found to have lower levels of soil surface aggregate stability, winterfat cover, and biological soil crust. From 2002 to 2007, shrub cover and perennial vegetation increased in exclosed plots, but factors such as plant community dynamics and soil properties were not found to respond when released from herbivory. We did find higher aggregate stability and the potential for higher infiltration rates in areas protected from cattle, and this might suggest that protection from grazing may maintain higher resource retention and hydrologic function. Areas protected from cattle-use offer encouragement that under no or light herbivory by wildlife winterfat plants in the Catlow Valley are resilient enough to double in size and total perennial vegetative cover within 5 years. Also, protecting winterfat communities where cheatgrass is present may increase the risk of cheatgrass expansion. Over time, this could reduce winterfat establishment and increase the risk of wildfire. Increases in winterfat shrub cover and overall amounts of perennial vegetation may make the habitat more suitable for jackrabbits as both a source of food and cover. Decreases in winterfat plant cover and perennial vegetative cover along with increases in bare patch size in control plots over the 5 years of this study signify that current levels of fall/winter cattle-use may not be sustainable over the long-run. Additionally, low soil surface aggregate stability, low levels of biological soil crust, and high levels of bareground in unprotected areas, along with seasonally strong winds, may increase the risk for wind erosion. Overall, winterfat plant communities in the Catlow Valley demonstrated high resistance to cattle and wildlife herbivory, with a lack of change in plant community structure in unprotected areas over five years. Lastly, winterfat communities also demonstrated high resilience to herbivory, with winterfat plants having considerable gains in shrub cover when protected from cattle for five years.

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CHAPTER 3: SOIL DYNAMICS, VEGETATION, AND ECOLOGICAL PROCESSES IN RELATION TO DISTANCE FROM A CATTLE WATERING POINT IN THE CATLOW VALLEY OF OREGON

ABSTRACT

Congregation of herbivores in arid and semiarid areas are known to influence the vegetation and soils around watering points. Over time the effects of herbivores around watering points becomes more dramatic, affecting the surrounding soils and vegetation. This response is such that soils and vegetation are negatively impacted nearer the watering point with effects diminishing with increasing distance from the watering point. Even though effects on soil and vegetation are known to occur with watering points in dry regions, there is a lack of quantitative information regarding soil and plant metrics. In 2007, we studied a number of soil and plant metrics as well as ecological processes in relation to distance from a cattle watering point in the Catlow Valley of Oregon. The cattle watering point occurred on a winterfat (*Krashenninikovia lanata* [Pursh] A. Meeuse & Smit) ecological site and had been used by livestock for over 50 years, and our study suggests that there was a clear cattle-use disturbance gradient for many of the plant and soil metrics quantified for a 2.41 km distance from the watering point. Areas closest to the watering point were characterized by heavy soil compaction, smaller perennial plants, more annual plants, more litter, decreased soil surface aggregate stability, lack of biological soil crust, and elevated soil nutrient levels. Areas furthest from the watering point favored decreased soil compaction, higher native plant diversity, larger perennial plants, decreased annual plant abundance, less litter, higher soil surface aggregate stability, and high amounts of biological soil crust. Our data suggest that distance between watering points must be considered when planning distribution of watering areas for cattle in the Catlow Valley, since watering points placed close together (< 5 km apart) are likely to overlap in zone of use by cattle. Lastly, it suggests that distance from water

is a strong predictor for determining long-term effects of grazing on plant metrics, soil metrics, and ecological processes.

INTRODUCTION

Winterfat is used by many species of wildlife for forage and habitat (Neilson 1968; Stevens *et al.* 1977; Johnson and Hansen 1979; Smith and Beale 1980; Johnson and Jorgensen 1981; Wasser 1982; Johnson and Anderson 1984; Shaw *et al.* 1984; Keting *et al.* 1985; Medin 1986; Anderson and Shumar 1986; Medin 1990; Yensen and Quinney 1992; McArthur *et al.* 1994; Ogle *et al.* 2001; and McAdoo *et al.* 2002). It is also a valuable source of forage for livestock (Eckert 1954; Cook and Stoddart 1963; Smoliak 1965; Cook and Child 1971; Woodmansee and Potter 1971; Hodgkinson 1975; Stevens *et al.* 1977; West 1979; Smith and Beale 1980; Blaisdell and Holmgren 1984; Clark 1986; Rasmussen and Brotherson 1986; Whisenant and Wagstaff 1991; Yorks *et al.* 1992; Chambers and Norton 1993; Dormaar *et al.* 1994; Krysl *et al.* 1994a, Krysl *et al.* 1994b; and Romo *et al.* 1995). The combination of use by wildlife and livestock has led to precipitous losses of winterfat across its historic range (Stevens *et al.* 1977 and Blaisdell and Holmgren 1984). During the late 1800's, winterfat was abundant in southeastern Oregon (Griffiths 1902; Shirk 1956; French 1964; and Simpson 1987). Oral accounts suggest early ranchers and herders allowed cattle to browse the abundant, highly palatable, and nutritious winterfat shrubs during both summer and winter (Shirk 1956, French 1964; and Simpson 1987). Continuous season-long use along with increasing numbers of settlers and livestock (cattle and sheep) quickly impacted stands of winterfat. By the early 1900's, winterfat was either not present or in poor condition in the areas it used to occupy around Steens Mountain (Griffiths 1902).

During the early 1900's many people began homesteading Catlow Valley, Oregon. They established at least two concrete well sites in the valley for use by homesteaders and livestock (Matney 2010a). One water well was located at the northwest corner of the valley and the other in the middle east section of the valley

(Matney 2010a). Other sources of water at this time included shallow lakes/ponds (*ie.* Garrison Lake) and water from streams and seeps on the east rim of the valley (*ie.* Skull Creek and Three Mile Creek). In addition to these natural sources of water, in the 1960's the Bureau of Land Management (BLM) established several water wells across the Catlow Valley (Matney 2010a). These wells were specifically designed to provide water access for cattle and to better disperse grazing-use across the valley. The south and middle portion of the valley were probably the most influenced by the BLM wells, since the northern portion of the valley was the closest to natural sources of water. Initial inspections of watering points showed obvious signs of disturbance created by cattle-use. Near the wells, the soil and plants were highly disturbed, appearing heavily trampled and grazed/browsed, and the level of disturbance appeared to decrease with increasing distance from the wells. Yet, winterfat plants were present, albeit in low densities, near and around the watering points.

Livestock and wildlife are attracted to watering points in arid and semi-arid rangelands which results in a disturbance gradient with areas nearest to watering points receiving more use than areas further away. Gradients such as these are often referred to as piospheres (Forman and Godron 1981). Greater disturbance near watering points has been linked to decreases in forage production, decreases in native perennial plants, and increases in soil degradation (Valentine 1947; Holscher and Woolfolk 1953; Van Der Schijff 1957; Lange 1969; Van Wyk and Fairall 1969; Young 1970; Sneva *et al.* 1973; Western 1975; Glantz 1977; Graetz and Ludwig 1978; Forman and Godron 1981; Goodman 1982; Collinson 1983; Dregne 1983; Sinclair and Fryxell 1985; Andrew 1988; Hart *et al.* 1991; Pinchak *et al.* 1991; Hart *et al.* 1993; Pickup *et al.* 1994; Fusco *et al.* 1995; Thrash and Derry 1999; Ganskopp 2001; and Nash *et al.* 2003). Studies have shown that much of the disturbance occurs within 1 600 m of the watering point, while regions farther away are used less by livestock and wildlife (Glendening 1944; and Valentine 1947; and Fusco *et al.* 1995). Though there have been studies investigating some of the effects that livestock and wildlife disturbance have created around watering points, studies have not focused on changes

in ecological processes or on a more complete analysis of vegetation and soil properties in relation to distance from watering points.

The main objective of this study was to determine the relationship between distance from a watering point and the soil metrics, plant metrics, and ecological processes on the Silty 6-10 PZ ecological site(s) (NRCS 2009) located in southeastern Oregon. To accomplish the above objective, we measured plant metrics, soil metrics, and ecological processes at fixed distances from a water point. We hypothesized that soil surface aggregate stability and other soil characteristics would improve with distance from water and that distances furthest from water would favor perennial native plants, especially winterfat.

MATERIALS AND METHODS

Study Area

The study was conducted within the Catlow Valley of southeastern Oregon. The average elevation of the valley is 1 400 m; the major landforms are lake terrace and lake plains; and soils are in the mesic temperature and xeric moisture regimes. Soils are fine textured Aridisols, primarily of the Spangenburg soil series (Appendix) and range from sandy loams to clay. The major ecological site is the Silty 6-10 PZ. Annual rainfall ranges from 127 to 254 mm. Rainfall measured on site at the study area for the water years of 2006 and 2007 were 222 mm and 129 mm, respectively. The water year 2007 was a drought year. Mean annual temperature and rainfall from 2005 to 2008 are presented (Fig. 3.1).

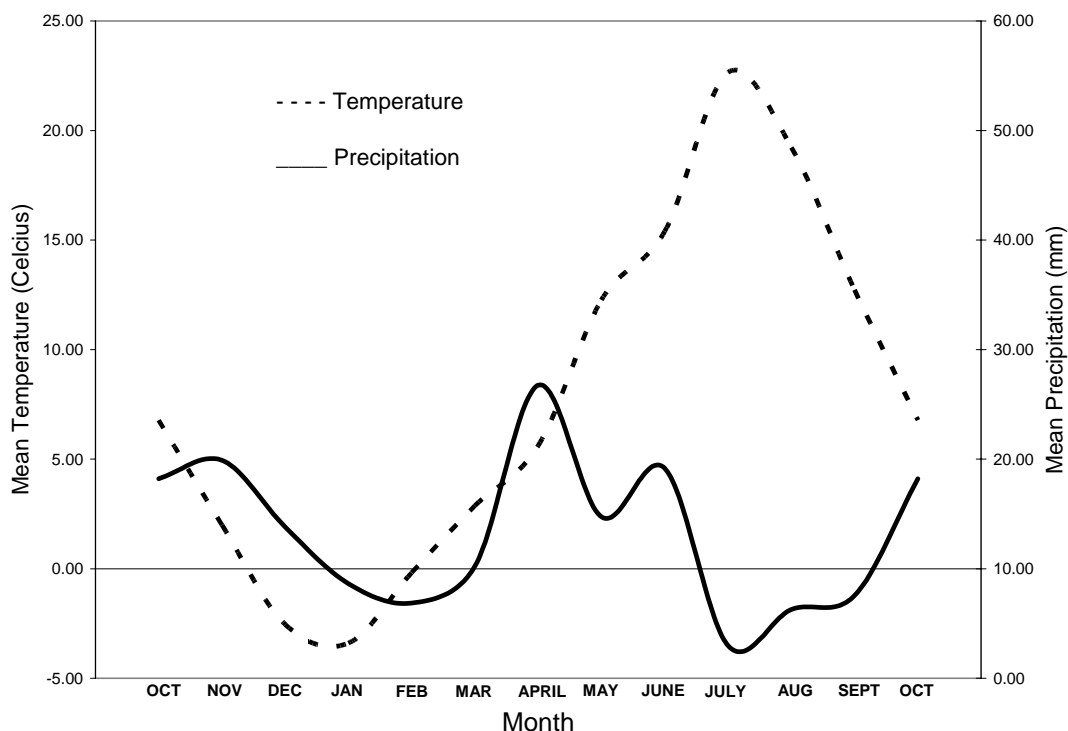


Figure 3.1. Monthly mean temperatures (°C; dashed line) and mean precipitation (mm; solid line) for May 2005 through May 2008. Lines are smoothed interpolations between average monthly values.

Vegetation was composed of a mixture of shrubs, bunchgrasses, forbs, annuals, and one native rhizomatous grass species. The common native shrubs were winterfat and sickle saltbush (*Atriplex falcata* [M.E. Jones] Standl.), and the occasional green rabbitbrush (*Chrysothamnus humilis* Greene). Native bunchgrasses included squirreltail (*Elymus elymoides* [Raf.] Swezey), Sandberg bluegrass (*Poa secunda* J. Presl), and Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth). The predominate native perennial forb was gooseberryleaf globemallow (*Sphaeralcea grossulariifolia* [Hook. & Arn.] Rydb. subsp. *grossulariifolia*), and the native rhizomatous grass species was creeping wildrye (*Leymus triticoides* [Buckley] Pilg.). Annuals present included cheatgrass (*Bromus tectorum* L.), clasping pepperweed (*Lepidium perfoliatum* L.), herb sophia (*Descurainia sophia* [L.] Webb ex Prantl),

spreading wallflower (*Erysimum repandum* L.), and shortstem lupine (*Lupinus brevicaulis* S. Watson).

The Catlow Valley has a long history of livestock use, beginning in 1872 (Shirk 1956 and French 1964). The Catlow Valley study area has been used as fall/winter range (October through March) for the past decade. Prior to 2002, movement of cattle within the valley was unimpeded. In 2002, 3 pastures were established. Since that time, pastures have been used every fall/winter using an alternating North to South deferred rotation management plan. In this plan, the middle pasture has been used at approximately the same time every year. In the years 2004 and 2005, 1 800 and 1 500 head of cattle, respectively, used the valley from October to March. This study was conducted at a single watering point located in the northernmost pasture.

Design

One watering point was selected in the Catlow Valley that had been in operation for approximately 50 years and had a long history of cattle-use. Watering point selection was based on the following criteria: 1) the watering point had been used by cattle for at least the last 40 years, 2) the watering point occurred in an area where winterfat plants were present and where there was not a high heterogeneity of soil type or landform type, 3) the watering point was not closely grouped with another watering point or natural standing body of water, 4) drastic changes in geography did not occur on more than half of the area surrounding the watering point or within 2 400 m from the watering point, and 5) cattle supplement locations or fences were not likely to be a confounding factor on cattle distribution around the watering point. At the watering point that met these criteria, a compass sector between 270° and 360° being 2.41km in length was determined as the only suitable area for study due to the other sectors being confounded by fencing, roads, landform, and proximity to other watering points. Within this sector, five azimuths rounded to the nearest 0.5° were randomly chosen for each of 6 radii distances from the water well: 50 m, 0.48 km, 0.97 km, 1.45

km, 1.93 km meters, and 2.41 km (Fig. 3.2). At each point of interest (total of 30) a line transect of at least 36 meters in length was established parallel to the watering point. Vegetation metrics, soil metrics, and ecological processes were measured along each transect. Vegetation metrics measured were plant frequency, plant density, and percent foliar cover by plant species and plant type. Soil metrics measured were resistance to penetration, bulk density, and soil pH. Other metrics measured related to ecological processes included: soil surface conditions (soil surface aggregate stability, percent bareground, bare patch size, perennial plant size, percent perennial vegetative cover, percent rock, percent soil crack, and percent biological soil crust); hydrologic cycle (relative infiltration rate); and nutrient cycle (soil organic carbon, soil organic nitrogen, plant available inorganic phosphorus in the soil, and percent litter ground cover). Bare patch size was calculated by the gap size between perennial plants determined from gap-line-intercept data. Perennial plant size (size of perennial plants based on canopy and basal measurements) was also determined from gap-line-intercept data. Percent bareground was defined as exposed bare soil void of biological soil crust, litter, vegetation, or other cover.

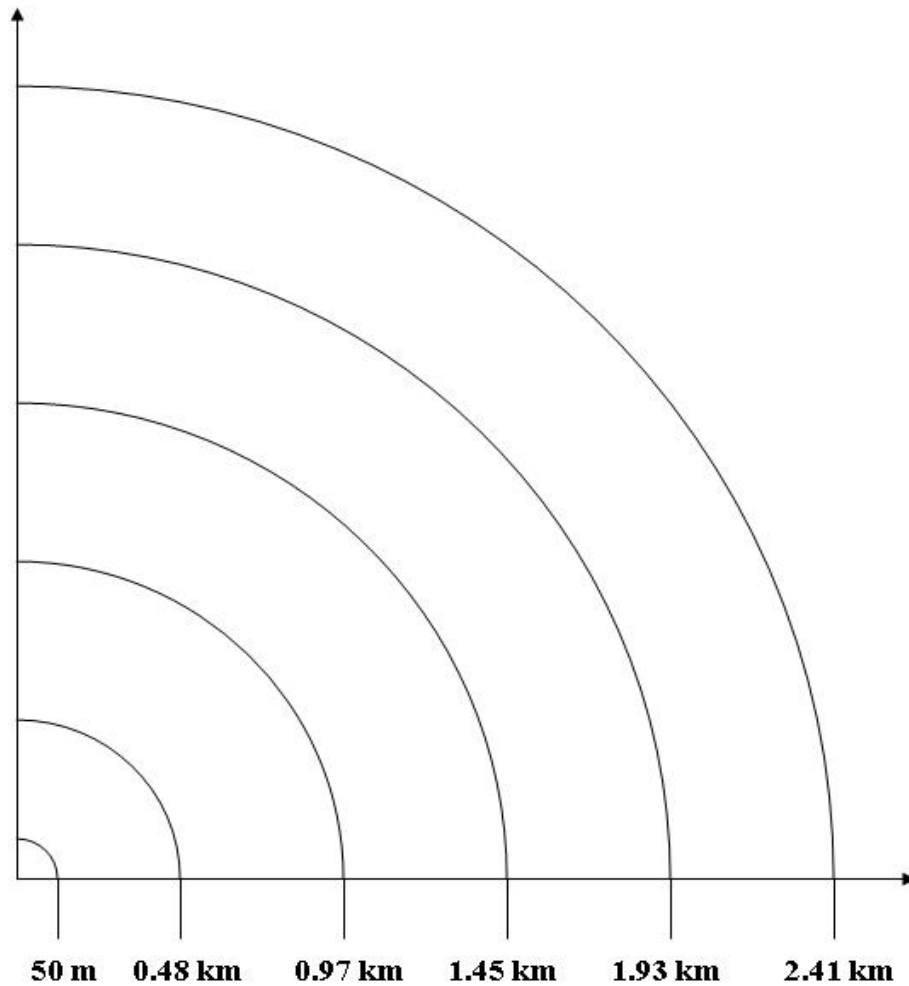


Figure 3.2. Layout of radial sampling distances relative to watering point location in the Catlow Valley.

Soil surface aggregate stability was measured at each transect (NRCS 1999 and Herrick *et al.* 2005). Samples were taken from the first 18 consecutive 1-m increments along each transect. Soil compaction was indexed by measuring soil penetration using an impact cone penetrometer (Herrick *et al.* 2005). Resistance to penetration was measured at the 10, 20, and 30 m increment mark along each transect. At each point the number of strikes it took to pass-through the 0-5 cm, 5-10 cm, 10-15 cm, and 15-20 cm soil depths were recorded.

Soil pH was determined on each transect at the following depths: 1) A horizon, 2) 1 cm thick transition zone occurring between the A and Bt1 horizons (± 0.5 cm at

the horizon interface), 3) Bt1 horizon, and 4) the Bt2 horizon. The depths of soil horizons and mean horizon depths were recorded. Soil pH, to the nearest 0.01, was determined in the laboratory using a LaMotte pH PLUS DIRECT Digital pH Meter ® and methods for soils that may vary due to salt content (Jones 2002). Soil bulk densities were determined using the volumetric ring method (NRCS 1999), and the soil sample cores (6 cm diameter) were taken at the same locations as soil pH at three depths: A horizon, Bt1 horizon, and Bt2 horizon. Bulk densities were determined for soil cores after drying at 100° C for 24 h.

A 1 quart sample of soil was taken from the A horizon to be measured for soil nutrients. All samples were analyzed for total organic carbon and total organic nitrogen with a Leco CNS-2000 Macro Analyzer ®. Samples were determined for plant available inorganic phosphorus using the Olsen phosphorus test (Olsen *et al.* 1954).

Within 5 m of each line transect, one measurement of relative infiltration rate was made at interspaces between shrubs using a pre-wetted, single ring infiltrometer test following the protocols by Herrick *et al.* (2005). The diameter of infiltrometer rings were 50 cm. Infiltration was measured for 1.5 h, keeping a constant head of 2.5 cm (± 0.5 cm) in depth. The last two readings of infiltration over the 1.5 h measurement period were used to obtain average relative infiltration rates at each study plot.

Point-intercept cover data were recorded every 30 cm along each line transect for a total distance of 36 m or 121 points (Elzinga *et al.* 1998). Biological soil crust was visually identified as those soil surfaces having a darker surface and/or having lichen and/or moss (Belnap *et al.* 2001; Belnap and Lange 2003; and Rosentreter *et al.* 2007). Plant basal cover (gap-line-intercept) was recorded along this same 36 m section of transect (Herrick *et al.* 2005). Plant density for perennial bunchgrasses and perennial shrubs were recorded in contiguous 1 x 1 m frames for a total number of frames at each transect as determined by sample size calculations for winterfat, when present, or for saltbush when winterfat was not present (Elzinga *et al.* 1998). Plant

density measurements for contiguous frames, frame size and number of frames, were determined for individual annual plant and perennial rhizomatous plant species according to sample size calculations. Sample size calculations were based on estimating 95% CIs within 25% of the mean. Plant frequency was measured for all species in 0.2 x 0.2, 0.3 x 0.3, 0.5 x 0.5, and 1 x 1 m nested frames. The number of frames recorded for plant frequency was the same for all species. The total number of frames recorded for plant frequency at each transect was determined by the maximum number of contiguous frames used to measure plant species density.

ANALYTICAL APPROACH

Simple linear regression and multiple regression analyses were used to make planned comparisons of plant metrics, soil metrics, and ecological processes in relation to distance from the cattle watering point. Sometimes, a combination of both regression and other parametric statistical tests such as 2-sample t-tests and/or 1-way analysis of variance (ANOVA) were used to describe differences among the six radii of distances from 50 m to 2.41 km attenuating away from the watering point. P-values (*P*) approximately ≤ 0.10 were considered ecologically important. All statistical analyses were performed using S-PLUS[®] 6.1 software (S-PLUS 2002).

RESULTS

Many plant metrics, soil metrics, and ecological processes demonstrated an association with distance from the cattle watering point. Some response variables demonstrated a simple linear association with distance, but the natural logarithm (ln), square, and cube of distance were more appropriate for explaining other response variables. Models were cataloged in tables. Soil penetrometer resistance models are displayed in Table 3.1, and all other metrics are displayed in Table 3.2.

Table 3.1. Regression models for impact penetrometer data (number of strikes) for five different depth zones, in relation to distance from a cattle watering point. Penetrometer resistance for the depths 0 to 5 cm, 5 to 10 cm, 10 to 15 cm, 15 to 20 cm, and 20 to 25 cm in the Catlow Valley, OR.

Depth	Regression Model	Mean	95% CI	R^2	2-sided P	df
0 to 5 cm	$-0.7263(\ln \text{ Distance}) + 2.7$	0.5	0.66 to 0.35	62	< 0.0001	1,28
5 to 10 cm	$-2.0693(\ln \text{ Distance}) + 4.3625$	1.4	1.79 to 1.08	71	< 0.0001	1,28
10 to 15 cm	$-2.0355(\ln \text{ Distance}) + 6.5284$	1.4	1.74 to 1.08	78	< 0.0001	1,28
15 to 20 cm	$-1.6649(\ln \text{ Distance}) + 10.7982$	1.2	1.82 to 0.49	31	0.001	1,28
20 to 25 cm	$3.3634(\text{Distance})^2 - 8.7773(\text{Distance}) + 19.381$	NA	NA	17	* 0.09	1,26

* Signifies overall model P and not 2-sided P .

Table 3.2. Regression for several metrics in relation to distance from a cattle watering point in the Catlow Valley, OR.

Metric	Regression Model	R^2	2-sided P	df
Bulk Density of A Horizon ($\text{g}\cdot\text{cm}^{-3}$)	$= 0.0376(\text{Distance}) + 1.0595$	0.12	0.06	1,28
Bulk Density of Bt2 Horizon ($\text{g}\cdot\text{cm}^{-3}$)	$= -0.0344(\text{Distance}) + 0.0537(\text{Distance}) + 0.9615$	0.29	0.01	2,26
Thickness of the A Horizon (cm)	$= -0.4403(\ln \text{Distance}) + 3.4977$	0.45	< 0.0001	1,28
Mean Soil Surface Aggregate Stability	$= 0.6986(\text{Distance}) + 2.2936$	0.85	< 0.0001	1,28
Percent Organic Soil Carbon of A Horizon	$= 0.2025(\text{Distance}) - 0.587(\text{Distance}) + 1.1277$	0.35	* 0.003	2,27
Percent Organic Soil Nitrogen of A Horizon	$= 0.0151(\text{Distance}) - 0.0448(\text{Distance}) + 0.1018$	0.32	* 0.005	2,27
Plant Available Inorganic Phosphorous (ppm)	$= 4.0906(\text{Distance}) - 11.6672(\text{Distance}) + 27.5048$	0.31	* 0.007	2,27
Percent Biological Soil Crust	$= 4.0393(\text{Distance}) + 1.3772(\text{Distance}) - 0.1724$	0.80	* < 0.0001	2,27
Percent Litter Ground Cover	$= -5.021(\text{Distance}) + 2.6481$	0.22	0.010	1,28
Percent Bareground	$= -12.969(\text{Distance}) - 28.5225(\text{Distance}) + 47.3492$	0.45	* 0.0003	2,27
Mean Perennial Plant Size (m)	$= 0.006(\ln \text{Distance}) + 0.0709$	0.20	0.03	1,28
Mean Bare Patch Size (m)	$= 0.2439(\ln \text{Distance}) + 1.4$	0.32	0.001	1,28
Density of Bunchgrasses ($\text{plants}\cdot\text{m}^{-2}$)	$= -1.6591(\text{Distance}) + 7.5928(\text{Distance}) - 9.5497(\text{Distance}) + 3.9446$	0.43	* 0.002	3,26
Density of Winterfat ($\text{plants}\cdot\text{m}^{-2}$)	$= 0.1066(\ln \text{Distance}) + 0.3513$	0.16	0.03	1,28
Density of Sickle Saltbush ($\text{plants}\cdot\text{m}^{-2}$)	$= 1.2249(\text{Distance}) - 4.3081(\text{Distance}) + 7.2478$	0.51	* 0.0006	2,21
Density of Annuals ($\text{plants}\cdot\text{m}^{-2}$)	$= -1.8465(\text{Distance}) + 5.5106$	0.36	0.001	2,23
Density of Perennials ($\text{plants}\cdot\text{m}^{-2}$)	$= 1.3651(\text{Distance}) - 3.5774(\text{Distance}) + 7.2153$	0.23	* 0.03	2,27
Relative Infiltration Rate (cm h ⁻¹)	$= -0.9622(\text{Bare Patch Size}) + 4.4859$	0.21	0.010	1,28
Soil Surface Aggregate Stability	$= 0.4776(\text{Bare Patch Size}) + 2.5099$	0.19	0.02	1,28

* Denotes the overall model P

An impact cone penetrometer was used to measure soil penetrometer resistance. Soil penetrometer resistance for the depths 0 to 5 cm, 5 to 10 cm, 10 to 15 cm, and 15 to 20 cm were all found to diminish linearly in relation to increases in log distance from the cattle watering point. Confidence intervals (CI), percent of variance explained (R^2), P , and linear models for all soil penetrometer depths are displayed in Table 3.1. At the depth of 0 to 5 cm, with each doubling in distance from the watering point there was an average decrease of 0.5 strikes in penetrometer resistance (95% CI = 0.66 to 0.35 strikes; Fig. 3.3). Distance from the watering point explained 62% of the variability in soil penetrometer resistance from 0 to 5 cm. For the soil depth zones of 5 to 10 cm, 10 to 15 cm, and 15 to 20 cm there was a respective decrease of 1.4, 1.4, and 1.2 strikes on average for each doubling in distance from the watering point. At the soil depth zone of 20 to 25 cm, penetrometer resistance decreased by an average of 5 strikes out to a distance of 1.45 km from the watering point. However, from 1.45 km to 2.41 km, there was an average increase of 4 strikes in penetrometer resistance. This trend was explained best by a regression model including distance and square distance as explanatory factors (2-sided $P = 0.09$, df 1,26). One outlier occurring at 1.45 km, with a penetrometer resistance higher than 30 strikes, was excluded from the analysis. Distance from the watering point explained a total of 17% of the variability in soil penetrometer resistance from 20 to 25 cm. Overall, with the exception of the 20 to 25 cm depth zone, soil penetrometer resistance decreased with increasing distance from the watering point.

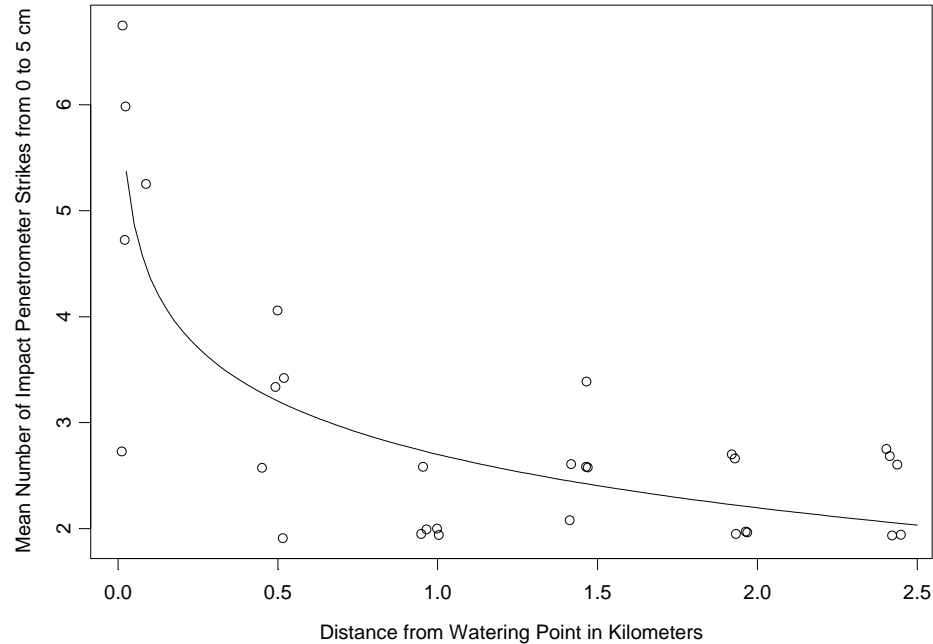


Figure 3.3. Mean number of impact penetrometer strikes for the 0 to 5 cm depth zone in relation to distance from a cattle watering point in the Catlow Valley, OR.

There was moderate evidence that the distribution of bulk density of the A soil horizon was associated with distance from the cattle watering point (2-sided $P = 0.06$; $R^2 = 0.12$). Associated with each 1 km increase in distance from the watering point there was a $0.04 \text{ g}\cdot\text{cm}^{-3}$ increase in bulk density. In contrast, bulk density of the Bt1 horizon was not associated with distance from the watering point (2-sided $P = 0.6$). Bulk density for the Bt1 soil horizon averaged $1.24 \text{ g}\cdot\text{cm}^{-3}$ across all distances. Bulk density of the Bt2 soil horizon was associated with distance from the watering point (overall model $P = 0.01$, Fig. 3.4). Associated with increasing distance from the watering point there was an average overall increase in bulk density of the Bt2 soil horizon of $0.02 \text{ g}\cdot\text{cm}^{-3}$ out to 1 km. From 1 km to 2.41 km in distance from the watering point there was an average decrease of $0.07 \text{ g}\cdot\text{m}^{-2}$. One data outlier occurring at the distance of 2.41 km with a bulk density value greater than $1 \text{ g}\cdot\text{m}^{-2}$ was excluded

from the analysis. Bulk density of the A and Bt2 soil horizons are displayed in figure 3.4.

Thickness of the A soil horizon was found to decrease linearly in relationship to increases in log distance from the watering point (2-sided $P < 0.0001$; Fig. 3.5). Associated with each doubling in distance from the watering point there was a decrease of 0.31 cm on average in the thickness of the A soil horizon (95% CI is 0.43 to 0.18 cm). Figure 3.5 depicts the variability in mean soil horizon depth out to 2.41 km from the watering point. Lastly, no differences were found for pH levels in relation to distance from the watering point for the A, Bt1, or Bt2 horizons. Average pH and SE for the A, Bt1, and Bt2 horizons regardless of distance from the watering point were 7.7 ± 0.04 , 7.7 ± 0.03 , and 7.7 ± 0.04 , respectively.

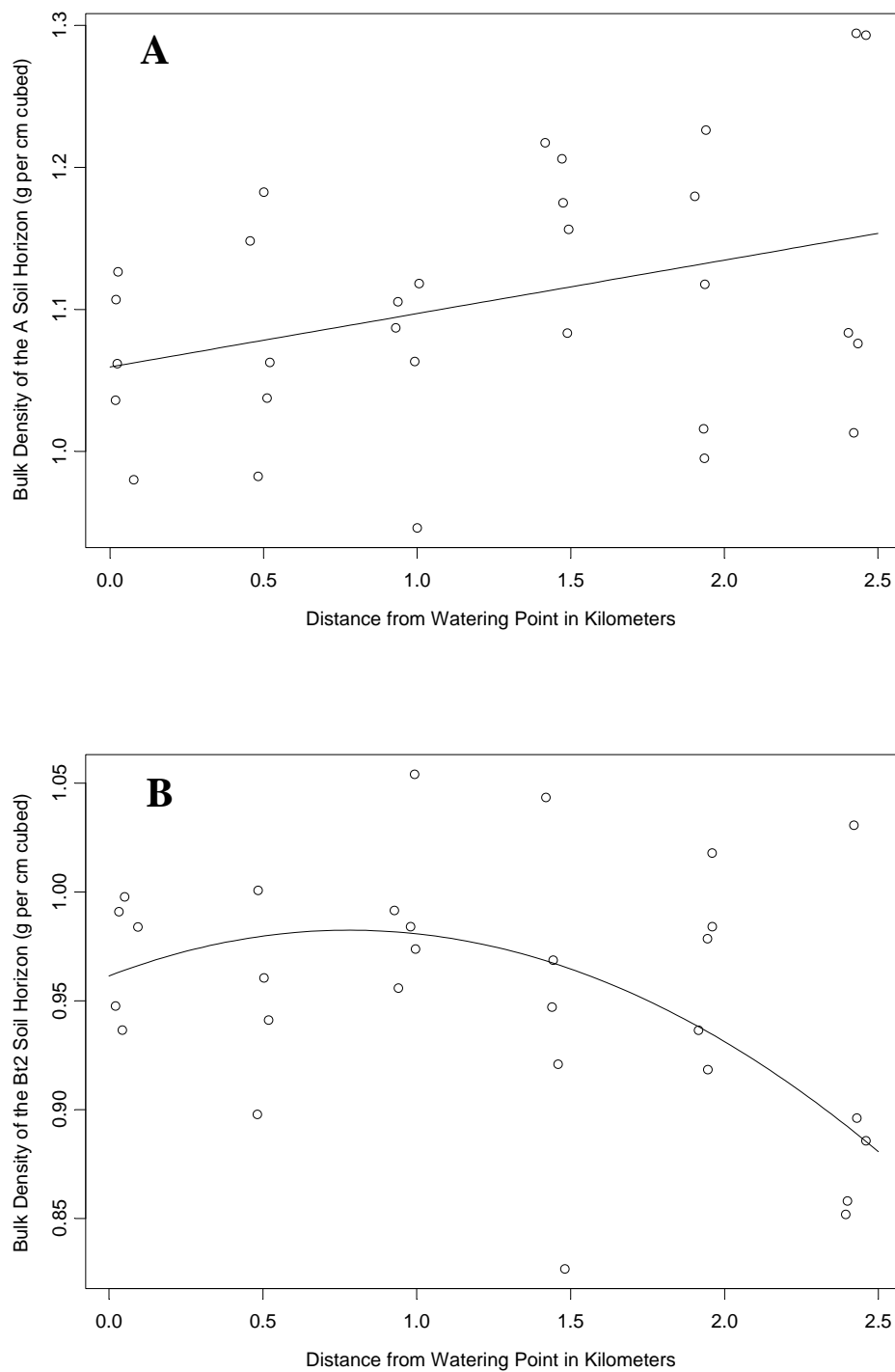


Figure 3.4. Bulk density ($\text{g}\cdot\text{cm}^{-3}$) of the A (A) and Bt2 (B) soil horizons in relation to distance from the cattle watering point in the Catlow Valley, OR.

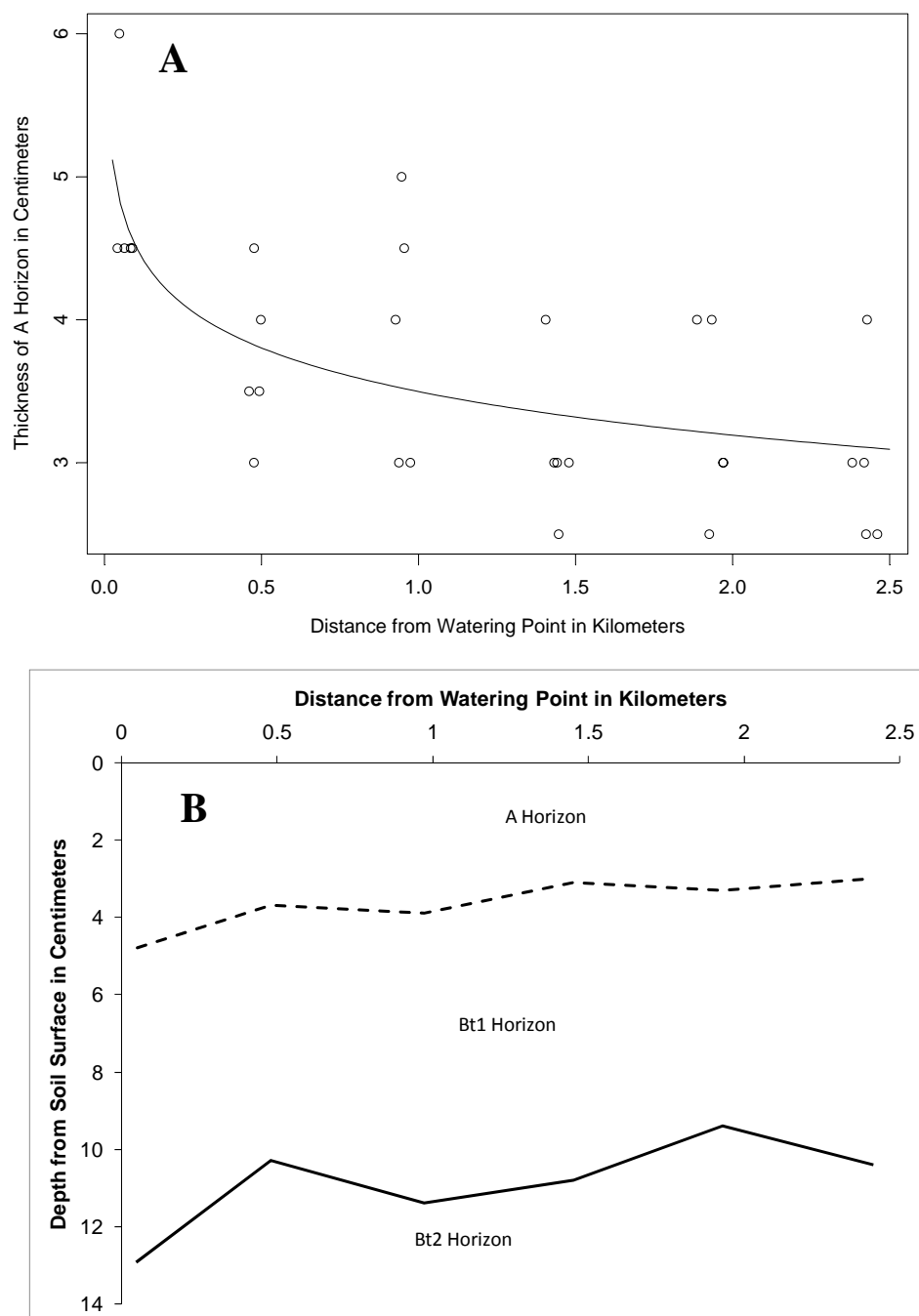


Figure 3.5. Thickness of the A soil horizon (A), and mean depths (cm) of the A, Bt1, and Bt2 soil horizon interface (B) in relation to distance from the cattle watering point in the Catlow Valley, OR.

Average soil surface aggregate stability was linearly associated with distance from the cattle watering point (2-sided $P < 0.0001$; Fig. 3.6). Distance from the watering point explained 84.5% of the variability in aggregate stability scores. With each 1 km increase in distance from the watering point there was a 0.7 on average increase in soil surface aggregate stability score (95% CI = 0.58 to 0.81). Soil surface aggregate stability was also highly associated with percent biological soil crust ($P < 0.0001$; df 1,28). For every 10% increase in biological soil crust, there was a 0.5 increase in soil surface aggregate stability score ($R^2 = 0.68$; Table 3.2).

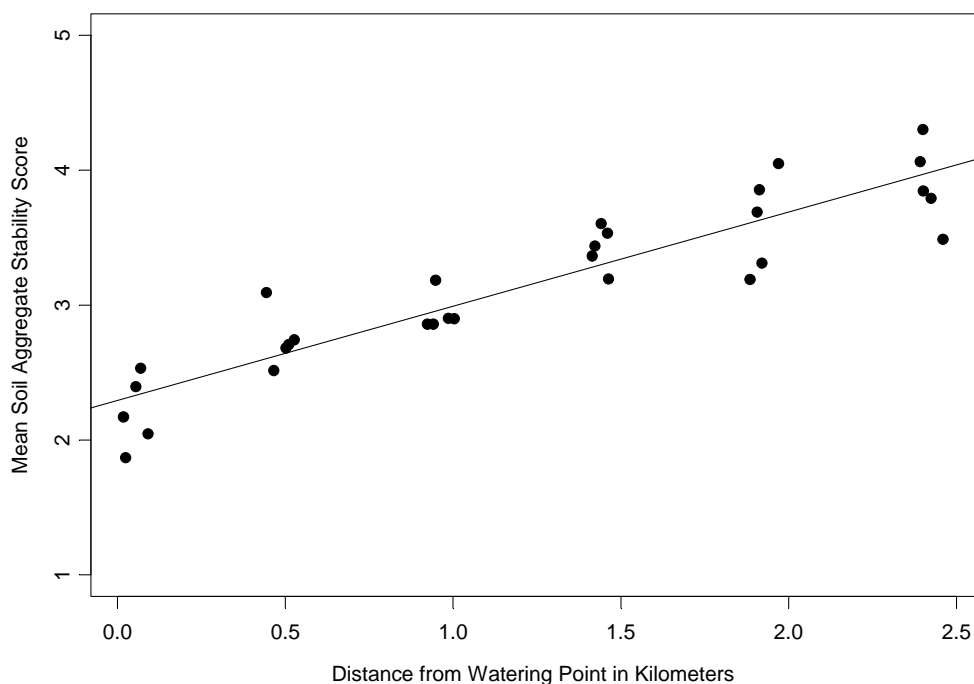


Figure 3.6. Mean soil surface aggregate stability score in relation to distance from a cattle watering point in the Catlow Valley, OR.

Carbon and nitrogen both displayed an inverse parabolic relationship to distance from the watering point, with their levels being highest at the watering point and lowest in the zone approximately 1 km from the watering point. There was overwhelming evidence that the percent organic soil nitrogen of the A1 horizon was

associated with distance from the watering point (overall model $P = 0.005$; Fig. 3.7). Associated with increasing distance from the watering point there was an average overall decrease in percent organic soil nitrogen of 0.034% out to 1.45 km. From 1.45 km to 2.41 km in distance from the watering point there is an average increase of 0.016%. Similarly, there was overwhelming evidence that the percent organic soil carbon was associated with distance from the watering point (overall model $P = 0.003$; Fig. 3.7). Associated with increasing distance from the watering point there was an average overall decrease in organic soil carbon of 0.42% out to 1.45 km. From 1.45 km to 2.41 km in distance from the watering point there was an average increase of 0.22%.

Plant available inorganic phosphorous of the A soil horizon displayed an inverse parabolic relationship with distance from the watering point in a similar fashion to soil carbon and nitrogen (overall model $P = 0.007$; Fig. 3.8). Associated with increasing distance from the watering point there was an average overall decrease in plant available inorganic phosphorous of 9 ppm out to 1.45 km whereas from 1.45 km to 2.41 km in distance from the watering point there was an average increase of 5 ppm.

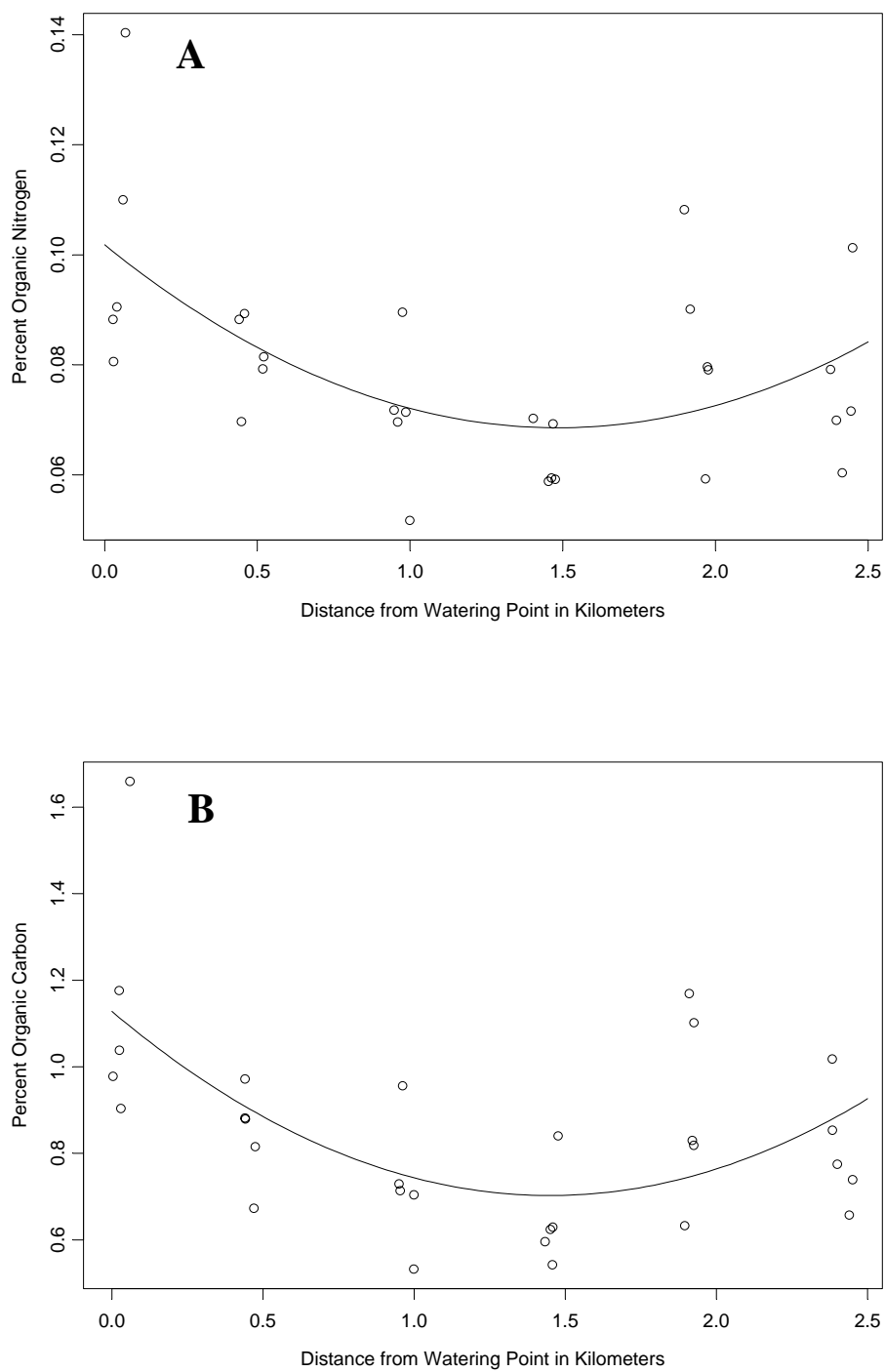


Figure 3.7. Mean percent organic nitrogen (A) and carbon (B) of the A soil horizon in relation to distance from the cattle watering point in the Catlow Valley, OR.

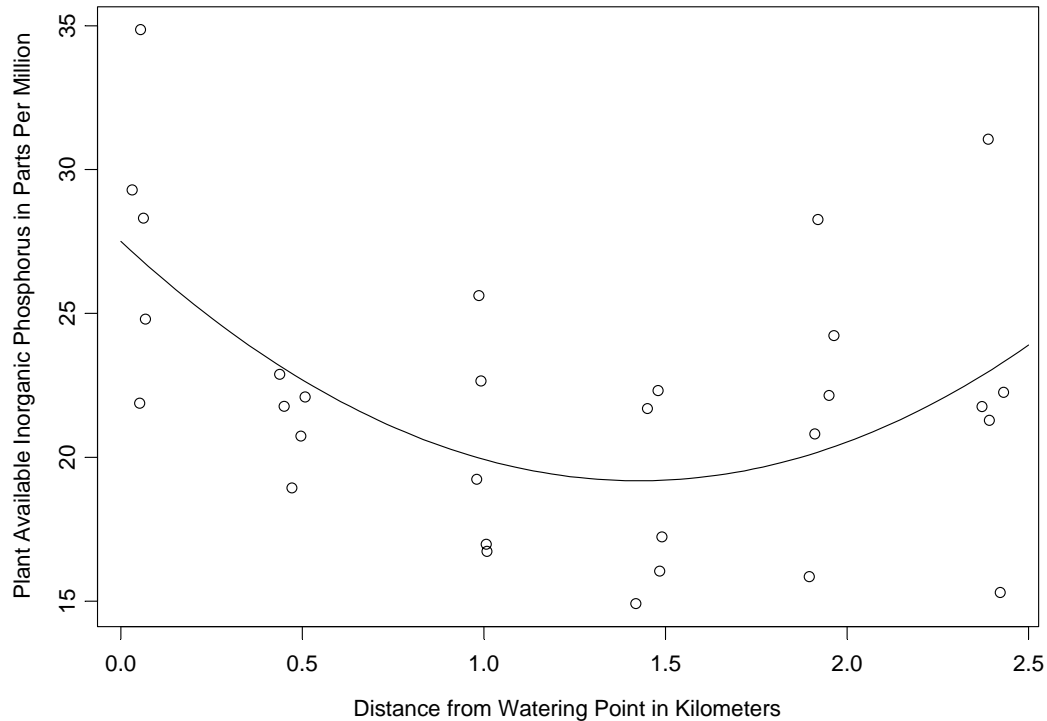


Figure 3.8. Mean plant available inorganic Phosphorous (ppm) of the A soil horizon in relation to distance from the cattle watering point in the Catlow Valley, OR.

There was strong evidence that the average percent litter cover was associated with distance from the cattle watering point (2-sided $P = 0.01$; Fig. 3.9). Associated with each 1 km increase in distance from the watering point there was an average decrease of 5% litter cover (95% CI is -8.7 to -1.3%). Over the same distance, percent biological soil crust displayed an almost exponential relationship to distance from the cattle watering point but was best explained by a multiple regression model incorporating both distance and square distance ($R^2 = 0.80$; overall model fit $P < 0.0001$). Biological soil crust showed an average overall increase of 28% percent in the range from 0.48 to 2.41 km (Fig. 3.9). Biological soil crust was absent from 0.05 to 0.48 km in distance from the watering point.

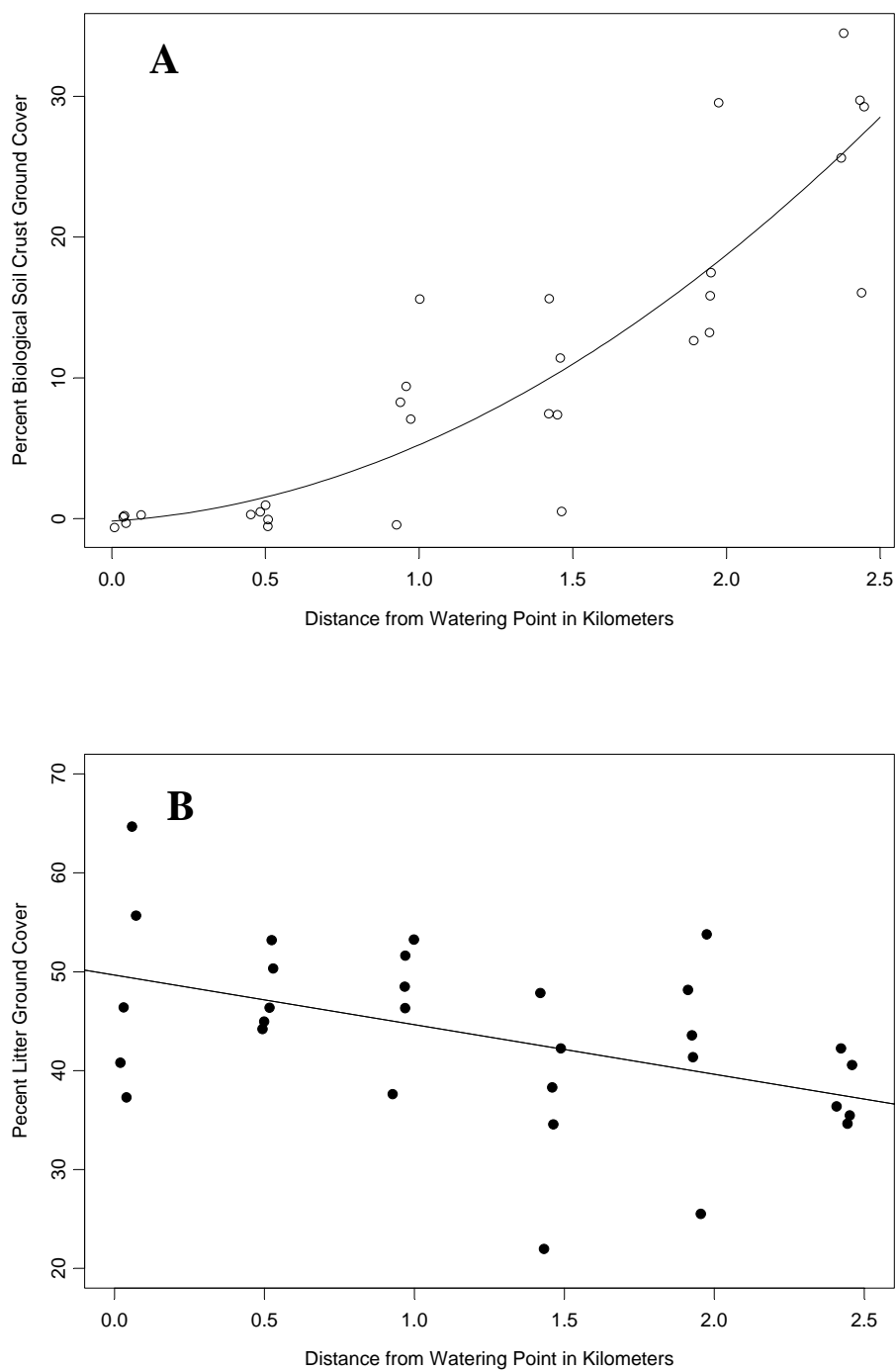


Figure 3.9. Percent biological soil crust (A) and litter cover (B) in relation to distance from the cattle watering point in the Catlow Valley, OR.

There was overwhelming evidence that the distribution of percent bareground had a parabolic relationship with distance from the watering point (overall model $P = 0.0003$). Associated with increasing distance from the watering point there was an average overall increase in bareground of 15% out to 0.97 km (Fig. 3.10). From 0.97 km to 2.41 km in distance from the watering point there was an average decrease of 25% in bareground cover. Percent bareground was linked strongly to the amount of organic soil carbon in the A horizon across all distances from the cattle watering point ($P = 0.0008$; Fig. 3.10). Associated with a 10% increase in bareground cover there was a 0.088% decrease in organic soil carbon of the A soil horizon (95% CI = 0.013 to 0.004%). One data outlier with an extraordinarily high amount of organic soil carbon compared to other samples (1.6%) was excluded from the analysis. Changes in bareground cover explained 35% of the variability in organic soil carbon (Table 3.2).

Bare patch size (gap size from gap-line-intercept data) between perennial plants was strongly associated with distance from the watering point (2-sided $P = 0.001$; df 1,28). With every doubling in distance, the average bare patch size increased by an average of 17 cm (95% CI = 7.3 to 26.5 cm). Mean perennial plant size (plant size, canopy and basal, from gap-line-intercept data) was also strongly associated with distance from the watering point (2-sided $P = 0.03$; df 1,28). With every doubling in distance, mean perennial plant size increased by 0.4 cm (95% CI = 0.05 to 0.8 cm). The relationship between mean bare patch size and mean perennial plant size with distance from the cattle watering point is displayed in figure 3.11. Average perennial plant size and mean bare patch size were strongly associated with each other ($P = 0.0002$; Fig. 3.12). With every 1 m increase in mean bare patch size, there was a 2.2 cm increase in mean perennial plant size (95% CI = 1.1 to 3.2 cm; Table 3.2).

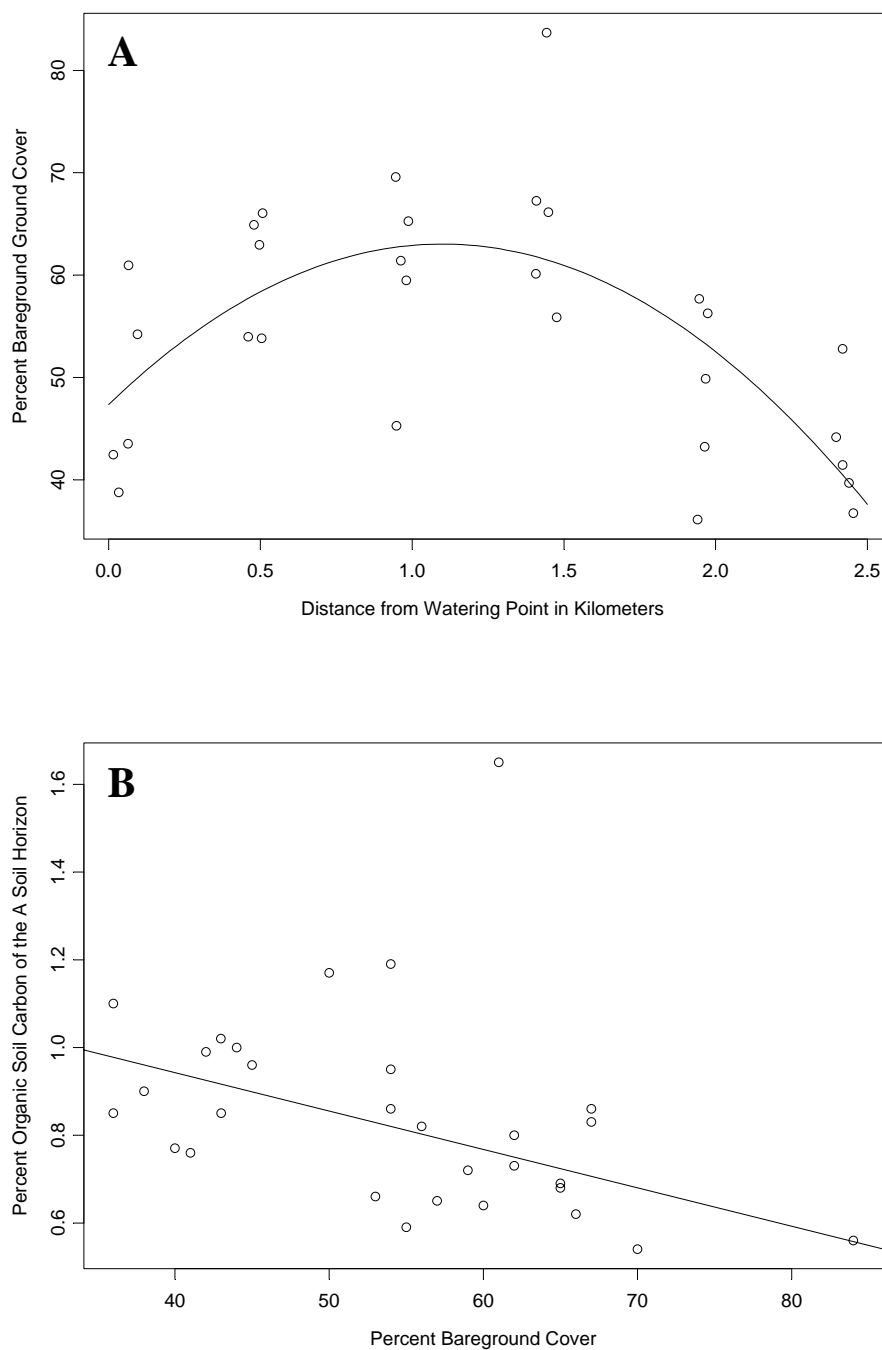


Figure 3.10. Percent bareground cover (A) in relation to distance from the cattle watering point; and percent organic soil Carbon of the A horizon (B) in relation to percent bareground cover for the area out to 2.41 km from a cattle watering point in the Catlow Valley, OR.

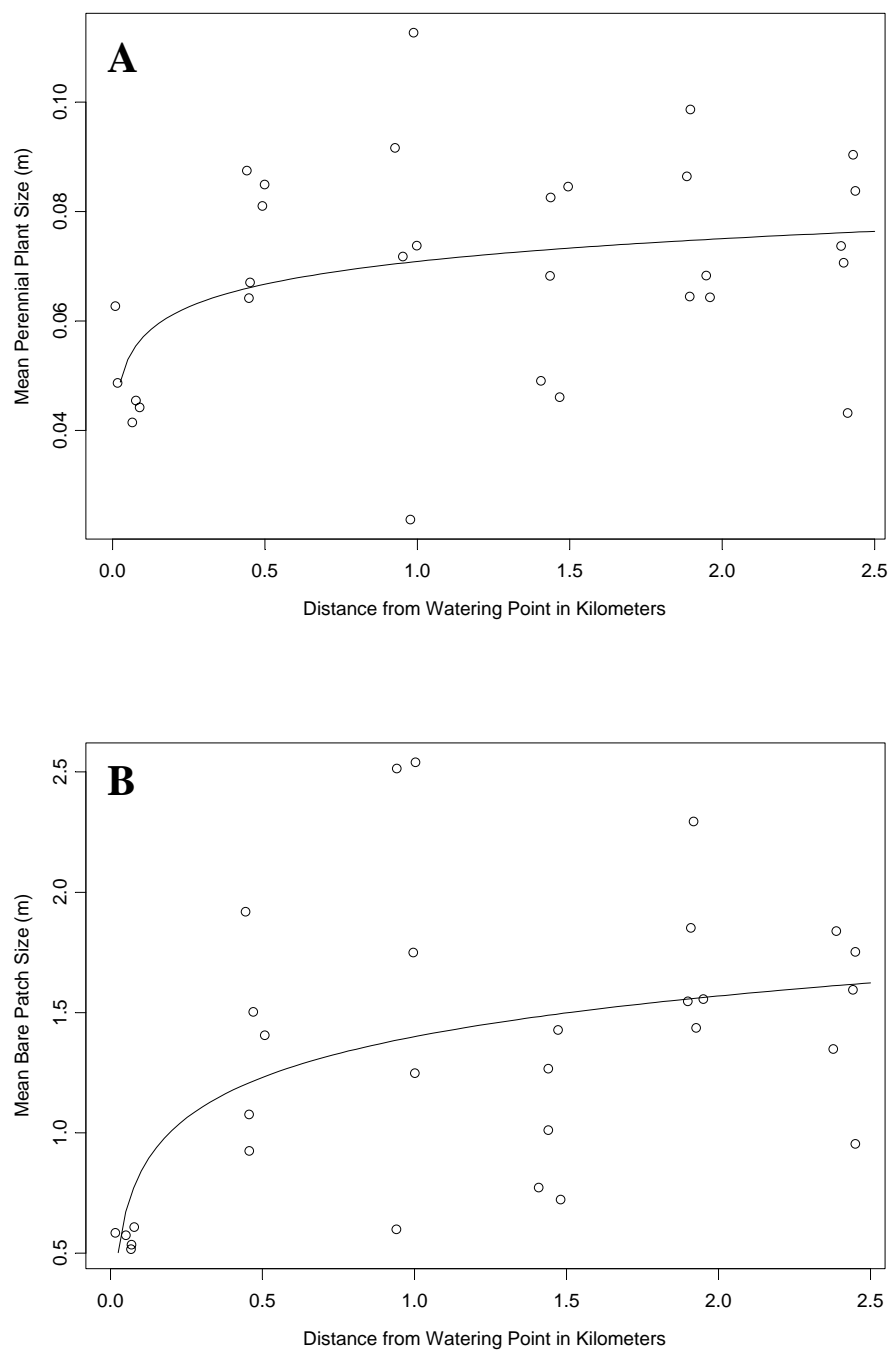


Figure 3.11. Mean bare patch size (A) and mean perennial plant size (B) in relation to distance from the cattle watering point in the Catlow Valley, OR.

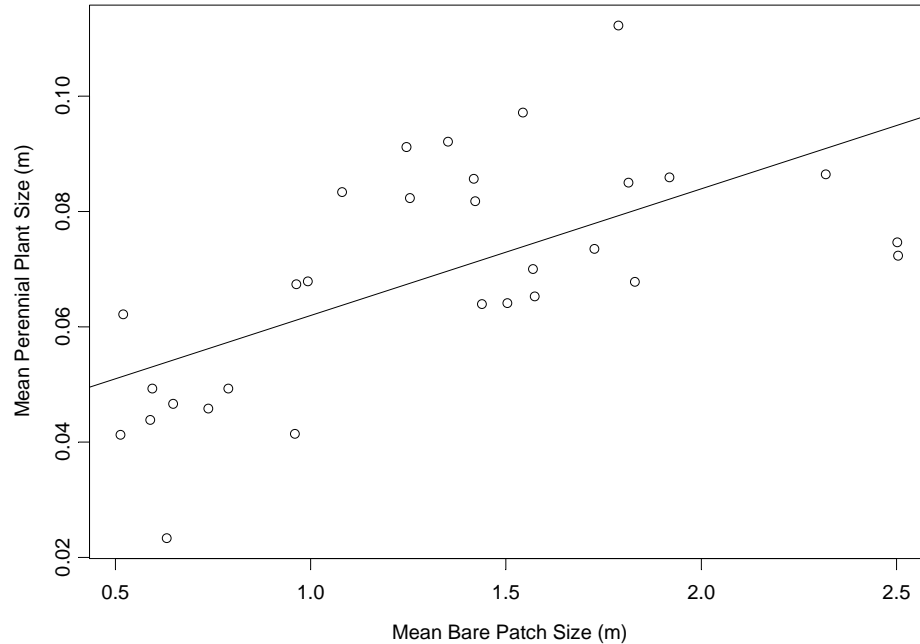


Figure 3.12. Mean perennial plant size (m) in relation to mean bare patch size (m) for the area out to 2.41 km from a cattle watering point in the Catlow Valley, OR.

Relative infiltration rate was not associated with distance from the cattle watering point. However, infiltration did not appear to be static across all distances. A planned comparison of infiltration rates across all distances by means of 1-way ANOVA revealed some differences ($P = 0.002$; $df\ 5,24$). Specifically, pairwise comparison of means using Tukey's method at the 95% confidence level showed differences between the distances 0.48 km vs. 1.45 km and 1.93 km vs. 2.41 km. Means and standard errors (SE) for infiltration are displayed in figure 3.13. There did not appear to be a reasonable explanation for differences in mean infiltration. Relative infiltration rate was then tested for an association with mean bare patch size. Relative infiltration was found to be highly associated with mean bare patch size (2-sided $P = 0.01$; Fig. 3.14), and for every 1 m increase in mean bare patch size there was a $0.96\text{ cm}\cdot\text{h}^{-1}$ decrease in relative infiltration rate (95% CI = 0.25 to $1.67\text{ cm}\cdot\text{h}^{-1}$). Soil surface aggregate stability was also tested for correlation with mean bare patch size

(Fig. 3.14), and the soil surface aggregate stability score was found to increase 0.5 for every 1 m increase in bare patch size ($P = 0.02$; df 1,28). Mean bare patch size explained 19% of the variability in soil surface aggregate stability (Table 3.2). However, relative infiltration rate and aggregate stability were not correlated ($P = 0.82$, df 1,28).

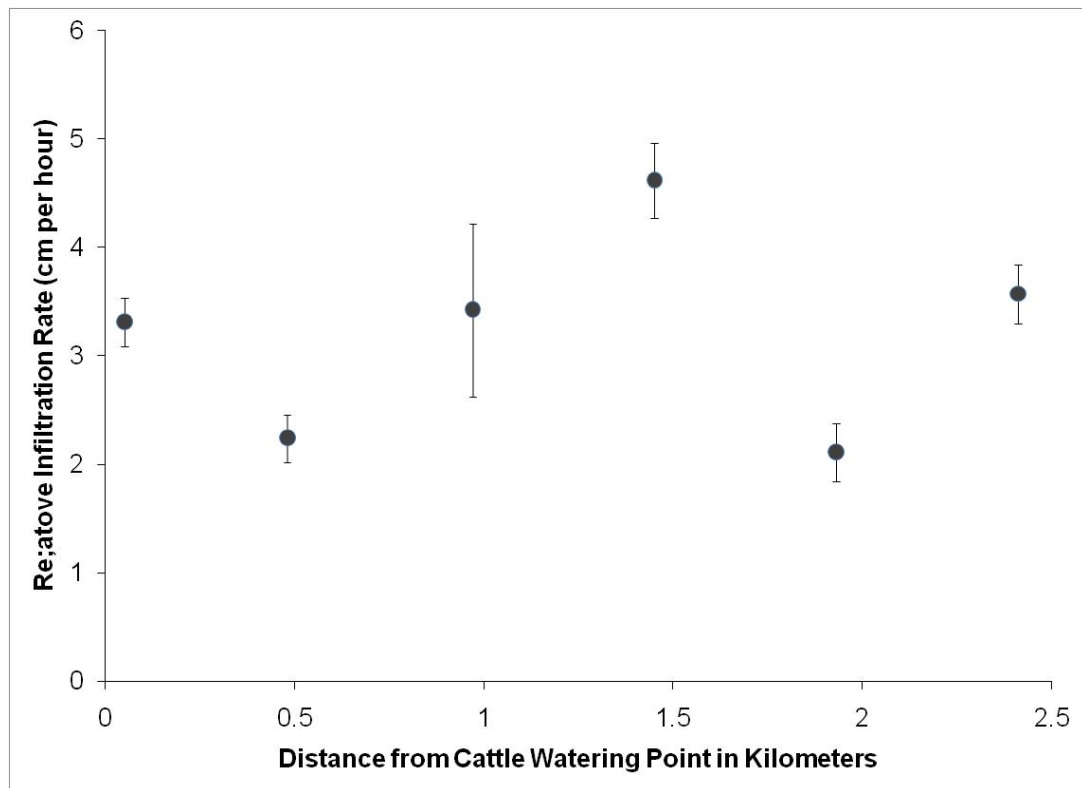


Figure 3.13. Mean relative infiltration rate ($\text{cm}\cdot\text{h}^{-1}$) and SE in relation to distance from a cattle watering point

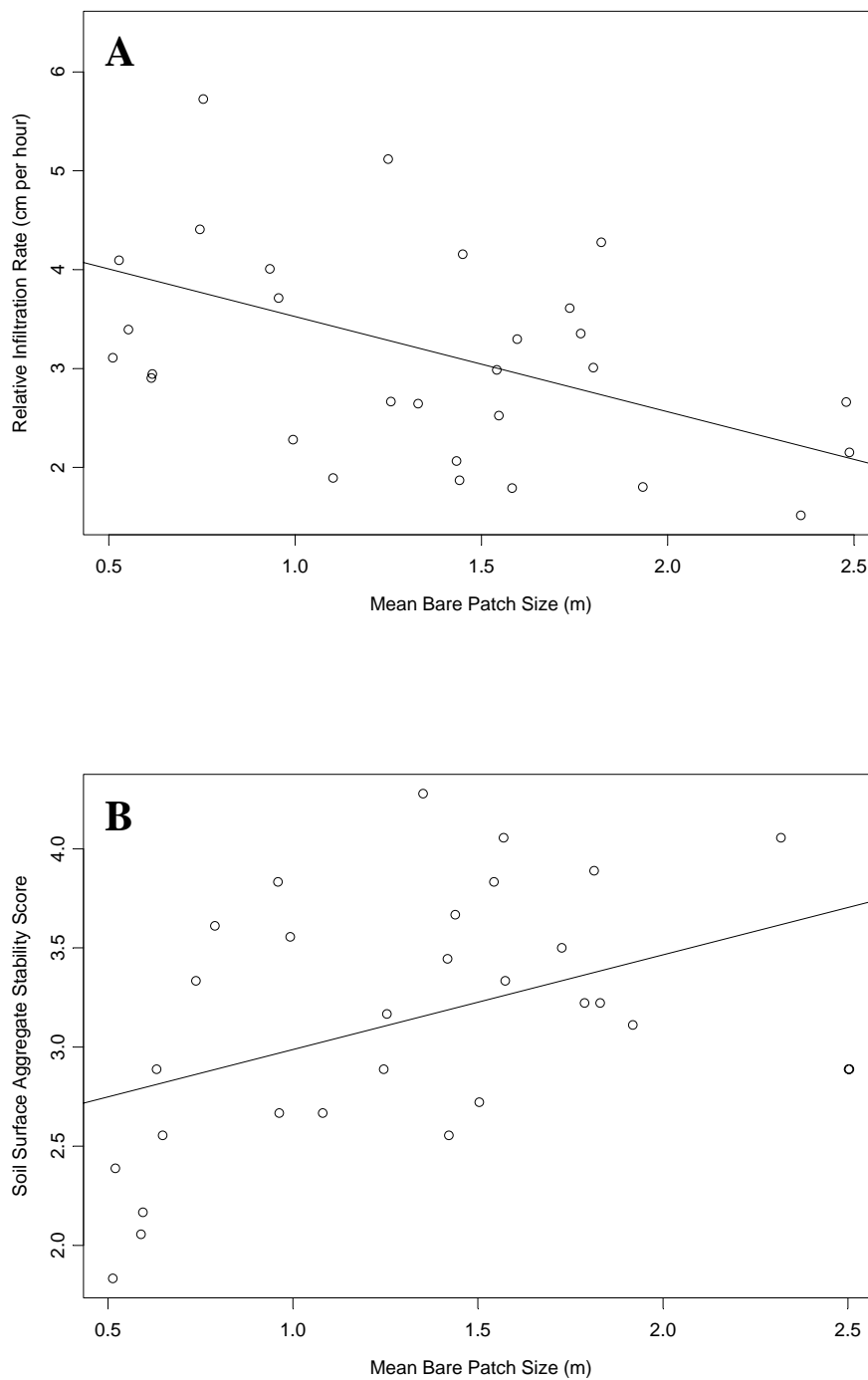


Figure 3.14. Relative infiltration rate ($\text{cm}\cdot\text{h}^{-1}$; A) and soil surface aggregate stability (B) in relationship to the mean bare patch size (m) for the area out to 2.41 km from a cattle watering point in the Catlow Valley, OR.

Density of creeping wildrye, on average, was numerically six times higher at the distance of 50 m from the watering point (Fig. 3.15), as compared to other distances from the watering point, but there was no statistical difference between means ($P = 0.4$; $df\ 5,24$) using a 1-way ANOVA. Numerically on average, creeping wildrye densities decreased with increasing distance from the watering point. Indian ricegrass, Sandberg bluegrass, and gooseberry globemallow were infrequently observed, but on average they were found to numerically increase in frequency with increasing distance from the cattle watering point (Fig. 3.15). The species Indian ricegrass did not occur until the distance of 1.93 km and was most prevalent at the distance of 2.41 km. Gooseberry globemallow occurred at 50 m from the watering point, but like Indian ricegrass, it was most prevalent at 1.93 and 2.41 km. Sandberg bluegrass numerically increased with distance from the watering point and was present across all distances.

Bunchgrass density (Indian ricegrass, Sandberg bluegrass, and bottlebrush squirreltail) was strongly associated with distance from the watering point (overall model $P = 0.002$). Associated with increasing distance from the watering point there was an average overall decrease in bunchgrass density of $3.5\ \text{plants}\cdot\text{m}^{-2}$ out to 0.97 km (Fig. 3.16). From 0.97 to 2.41 km in distance from the watering point there was an average increase of $1.5\ \text{bunchgrass plants}\cdot\text{m}^{-2}$. Distance from the watering point explained 43% of the variability in bunchgrass density.

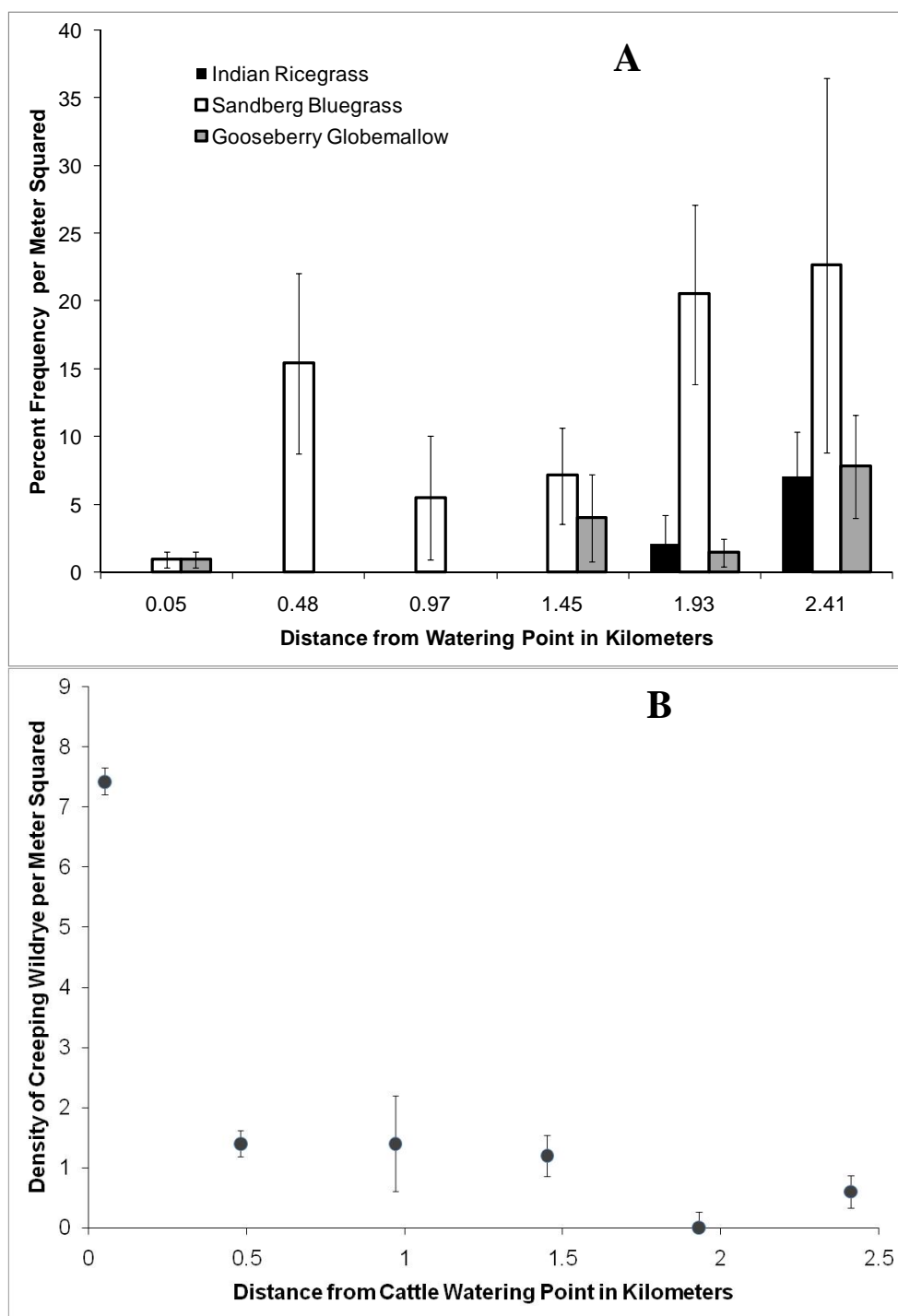


Figure 3.15. Frequency of Indian ricegrass, Sandberg bluegrass, and gooseberry globemallow (plants·m⁻²; A) as well as density of creeping wildrye (plants·m⁻²; B) in relation to distance from the cattle watering point in the Catlow Valley, OR.

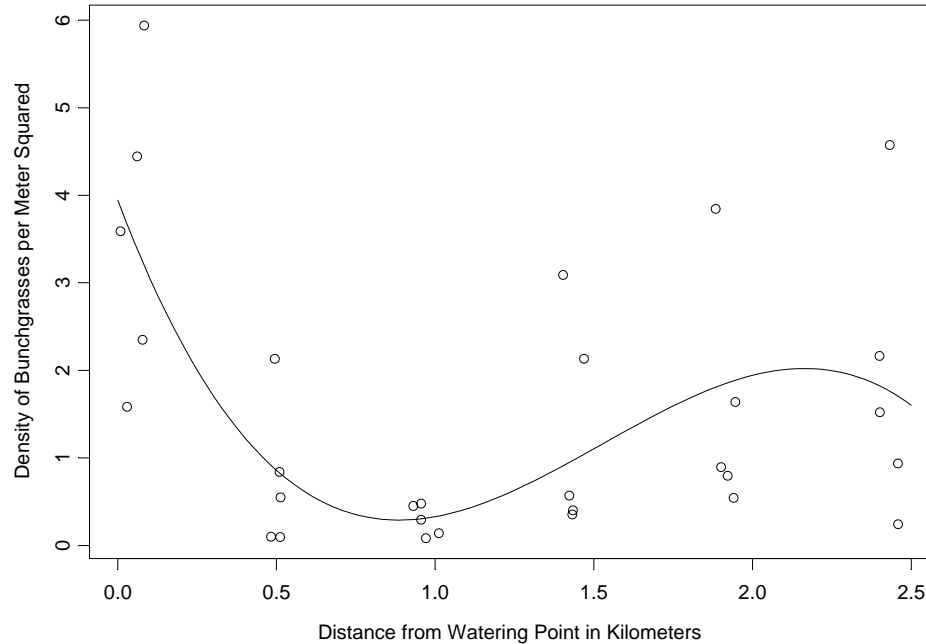


Figure 3.16. Density of bunchgrasses ($\text{plants} \cdot \text{m}^{-2}$) in relation to distance from the watering point in the Catlow Valley, OR.

Average density of winterfat $\cdot \text{m}^{-2}$ was associated with distance from the watering point (2-sided $P = 0.03$; Fig. 3.17). Associated with each doubling in distance from the watering point there was a $0.07 \text{ plants} \cdot \text{m}^{-2}$ increase in average density of winterfat (95% CI = 0.008 to $0.14 \text{ plants} \cdot \text{m}^{-2}$). Distance from the watering point explained 16% of the variability in average density of winterfat. Density of sickle saltbush $\cdot \text{m}^{-2}$ was also associated with distance from the watering point (overall model $P = 0.0006$; Fig. 3.17); there was an average overall decrease in density of sickle saltbush of $2.1 \text{ plants} \cdot \text{m}^{-2}$ in the range from 0.48 to 1.45 km . Sickle saltbush increased in density by $0.7 \text{ plants} \cdot \text{m}^{-2}$ from 1.45 to 2.41 km from the cattle watering point. From 0.05 km to 0.48 km in distance from the watering point the density of sickle saltbush increased from 3.3 to $5.5 \text{ plants} \cdot \text{m}^{-2}$. One data outlier at the distance of 0.97 km from the watering point, having no shrubs, was excluded from the analysis.

Distance from the watering point explained 51% of the variability in density of sickle saltbush.

Annual plant density and perennial plant density were both linked to distance from the cattle watering point (Fig. 3.18). Density of annual plants·m⁻² were associated with distance from the watering point (overall model $P = 0.001$). Associated with increasing distance from the watering point there was an average decrease in annual plant density of 1.85 plants·m⁻² in the range from 0.48 to 2.41 km (95% CI is 2.9 to 0.8). From 0.05 km to 0.48 km in distance from the watering point the density of annuals increased from 3.1 to 5.2 plants·m⁻². With increasing distance from the watering point there was an average overall decrease in perennial plant density of 2.5 plants·m⁻² out to 1.45 km. From 1.45 km to 2.41 km in distance from the watering point there was an average increase of 2 plants·m⁻².

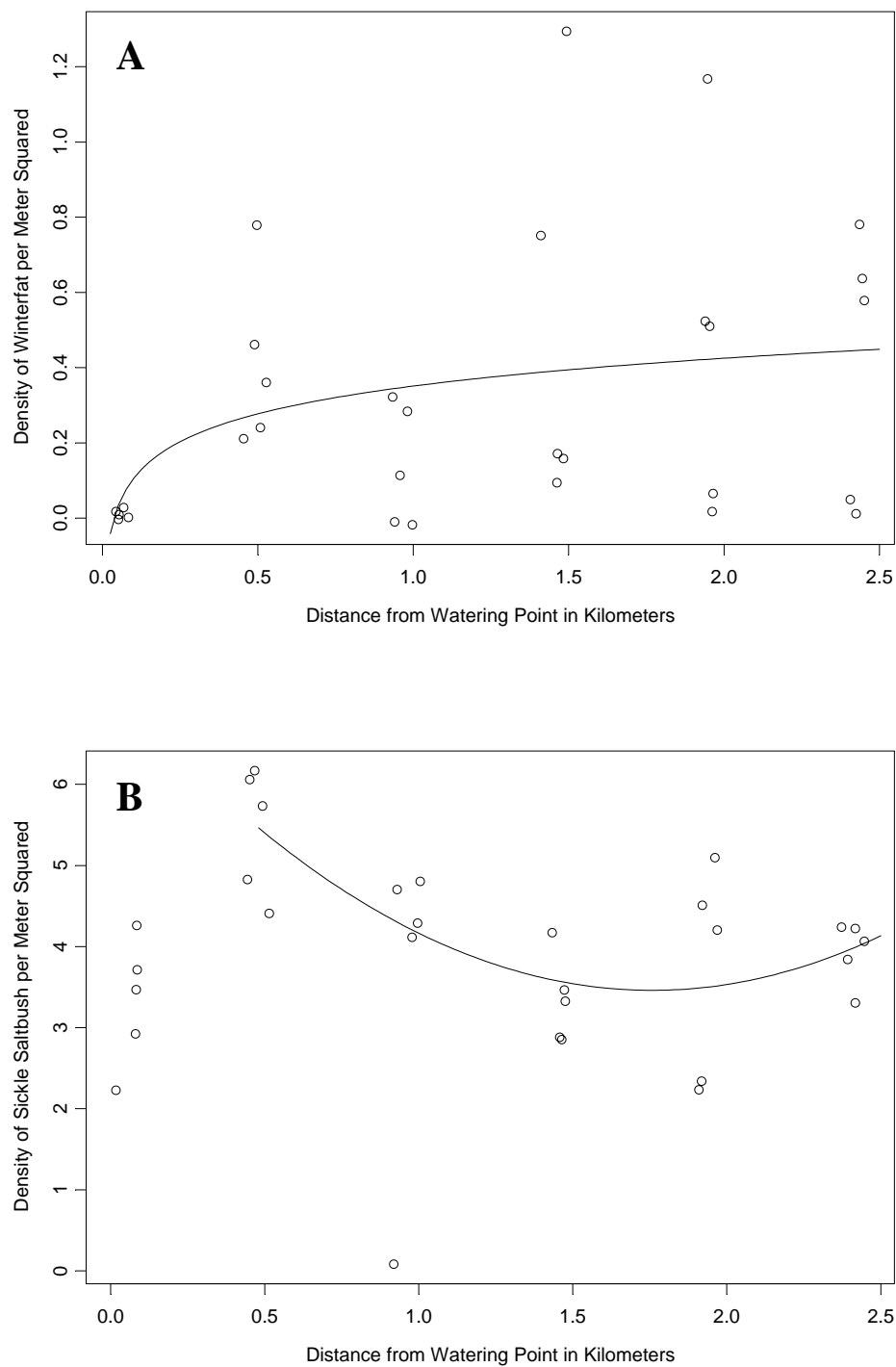


Figure 3.17. Density of winterfat (A) and sickle saltbush (B) shrubs ($\text{plants} \cdot \text{m}^{-2}$) in relation to distance from the cattle watering point in the Catlow Valley, OR.

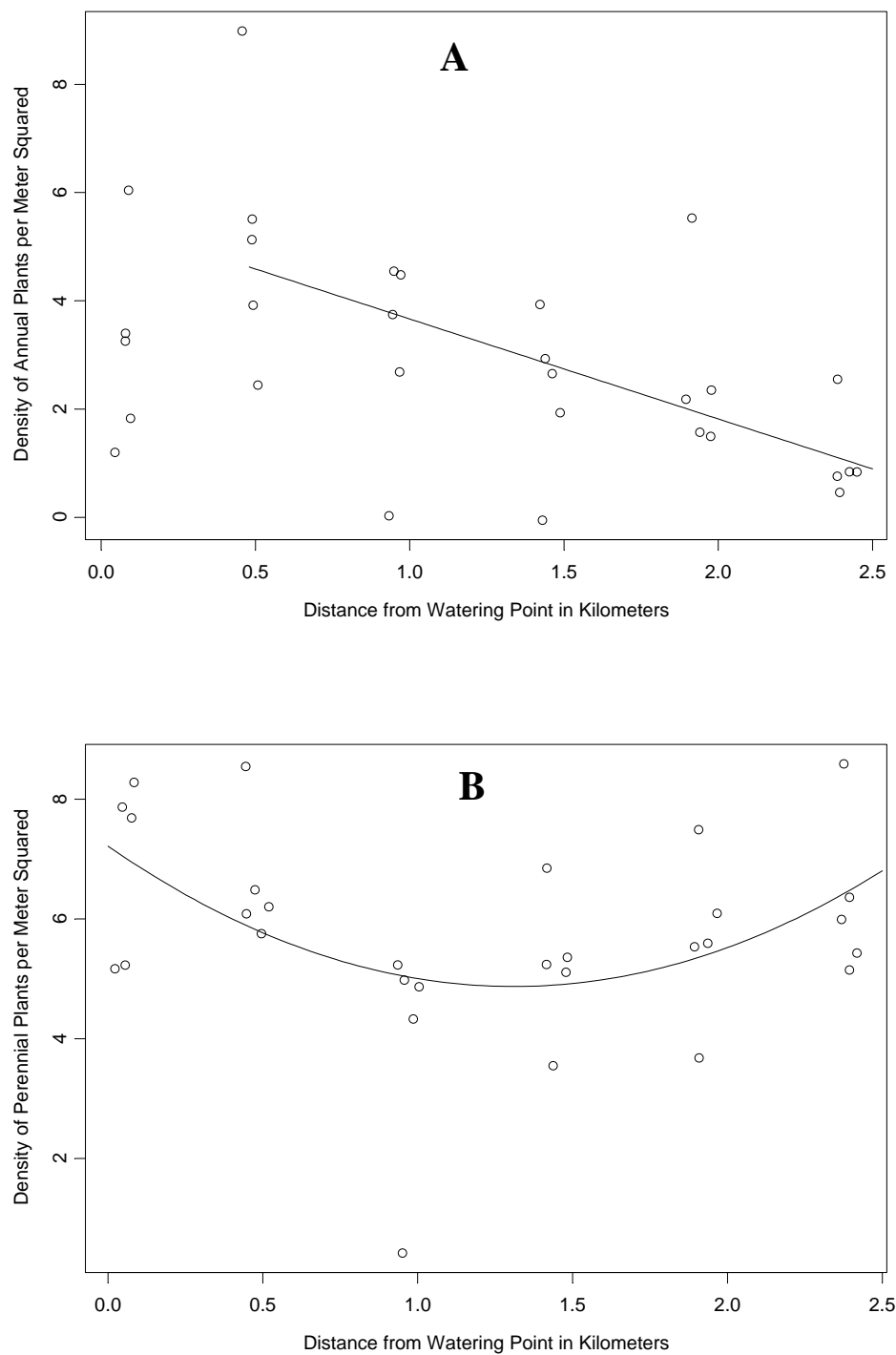


Figure 3.18. Mean density ($\text{plants} \cdot \text{m}^{-2}$) of annual plants (A) and perennial plants (B) in relation to distance from the cattle watering point in the Catlow Valley, OR.

DISCUSSION

It has long been considered a fact that large herbivores congregate around watering points in arid and semi-arid areas where access to water is limited (Valentine 1947; Holscher and Woolfolk 1953; Van Der Schijff 1957; Lange 1969; Van Wyk and Fairall 1969; Young 1970; Sneva *et al.* 1973; Western 1975; Glantz 1977; Graetz and Ludwig 1978; Forman and Godron 1981; Goodman 1982; Collinson 1983; Dregne 1983; Sinclair and Fryxell 1985; Andrew 1988; Hart *et al.* 1991; Pinchak *et al.* 1991; Hart *et al.* 1993; Pickup *et al.* 1994; Fusco *et al.* 1995; Thrash and Derry 1999; and Ganskopp 2001), and congregation of these animals creates a disturbance gradient that is primarily mediated by herbivory as well as trampling and compaction of soil (Valentine 1947; Andrew 1988; and Thrash and Derry 1999). Changes linked to this disturbance gradient include wind and water erosion, shifts in plant community structure, and deposition of cattle dung and urine (Thrash and Derry 1999). In the Catlow Valley, we were interested in determining if changes in soil metrics, plant metrics, and ecological processes were evident around a watering point that had been used by cattle and wildlife for over 50 years. Vegetation metrics we measured were plant frequency, plant density, and percent foliar cover by plant species and plant type. Soil metrics we measured were resistance to penetration, bulk density, and soil pH. Other metrics measured related to ecological processes included: soil surface conditions (soil surface aggregate stability, percent bareground, bare patch size, perennial plant size, percent perennial vegetative cover, percent rock, percent soil crack, and percent biological soil crust); hydrologic cycle (relative infiltration rate); and nutrient cycle (soil organic carbon, soil organic nitrogen, plant available inorganic phosphorus in the soil, and percent litter ground cover). We hypothesized that soil pH, soil resistance to penetration, and soil compaction would be highest near the watering point. Secondly, we hypothesized that annual plants would be most abundant near the watering point and that perennial plant cover and perennial plant abundance would increase with increasing distance from the watering point. Thirdly, we hypothesized that soil surface conditions and hydrologic cycling (relative infiltration rate) would

improve with increasing distance from the watering point. Lastly, we hypothesized that soil nutrients would be highest near the watering point and that litter cover would increase with increasing distance from the watering point. If evidence of a disturbance gradient were lacking, our null hypothesis, we assumed that plant metrics, soil metrics, and ecological processes would be homogenous across all distances radiating from the cattle watering point. To the contrary, our study found a strong gradient in relation to distance from the cattle watering point for most of the soil metrics, plant metrics, and ecological processes measured.

Soil penetrometer resistance and thickness of the A horizon decreased in relation to the natural logarithmic increase in distance from the watering point in a fashion very similar to the percent forage utilization findings found by Valentine in 1947. The decrease in soil penetrometer resistance seemed to suggest that the greatest amount of soil compaction occurred within 0.48 km from the watering point and that beyond that point there was a slight but consistent decrease in level of compaction. Visual inspection of the A and Bt1 horizons, nearest the watering point revealed that the two horizons were mixed and that soil pore spaces were squashed. Compaction has been documented to be the most extreme near herbivore congregation points (*ie.* water or salt) in other studies (Mphinyane 2001 and Walters and DeLuca 2007). In similar fashion, thickness of the A horizon decreased substantially from 50 m to 0.48 km and then thinned at a moderate rate out to 2.41 km from the watering point. Changes in soil horizon depth, soil structure, and soil morphology in relation to watering points have been scantily mentioned in other studies to present date (Mphinyane 2001 and Walters and DeLuca 2007). Therefore, we have little to compare our results to. We hypothesize that increases in thickness of the A horizon were not likely the result of enhanced soil building processes nearer the watering point but were a result of increased levels of trampling and compaction which affected soils to the greatest depth in the area near the watering point. This was likely mediated by pulverization of the soil surface during dry months (Griffiths 1902; Senzota and Mtahko 1990; and Greenwood *et al.* 1998), mixing and compression of the A1 and A2

horizon during wet months (Lason 1979, Walker 1980, Riney 1982, Van Wijngaarden 1985, Andrew and Lange 1986a., Rauzi 1963, and Van Den Berg *et al.* 1976), as well as deposition of litter and dung in close proximity to the watering point (Weir 1971, Georgiadis 1987, Tolsma *et al.* 1987, Perkins and Thomas 1993a, and Mphinyane 2001).

We found that biological soil crust increased at nearly an exponential rate while aggregate stability increased at a linear rate in relation to increases in distance from the watering point. Biological soil crust was all but excluded from distances closer than 0.93 km. We surmise that trampling was so great nearer the watering point that it prevented biological soil crust from forming. Biological soil crust has been found to be sensitive to trampling in other cattle grazing studies (Anderson *et al.* 1982; Andrew and Lange 1986a; Graetz and Tongway 1986; Johansen and St. Clair 1986; Beymer and Klopatek 1992; Heshmatti 1997; Hodgins and Rogers 1997; and Ponzetti and McCune 2001). Aggregate stability increased at a constant amount from 0 to 2.41 km from the watering point and aggregate stability and biological soil crust were highly correlated to each other. Others have also found that higher levels of aggregate stability were linked to increases in biological soil crust (Greene *et al.* 1990 and Williams *et al.* 1995). This again suggests that soil surface disturbance through trampling and pulverization were major factors affecting soil around the watering point. In agreement with this position, we surmise that the overall depth to the Bt1 horizon was highest near the watering point due to additions of dung and litter over decades that were mixed and pulverized with the A horizon overlying the Bt1 horizon. For the most part, penetrometer resistance values suggest that soil compaction was greatest at distances closer than 1.5 km. Although studies have found increases in level of compaction near to watering points (Rauzi 1963; Van Den Berg *et al.* 1976; Lason 1979; Walker 1980; Riney 1982; Van Wijngaarden 1985; and Andrew and Lange 1986a) there has been no information discussing changes in impact cone penetrometer data in combination with bulk density data surrounding watering points. In disagreement with the penetrometer resistance data we collected we found that the

bulk density of the A horizon actually slightly increased, while there was no change in the Bt1 horizon, and a slight reduction in the Bt2 horizon in association with increases in distance from the watering point. We suggest that the A horizon was unconsolidated over most of the range near the watering point and that increases in aggregate stability and biological soil crust at distance from the watering point were responsible for increases in bulk density. Maestre *et al.* (2002) also found increases in biological soil crust to be linked to higher bulk density. Although there were no discernible differences in bulk density within the Bt1 horizon it was apparent given the penetrometer data that it was much more resistant to penetration nearer the watering point. Decreases in bulk density of the Bt2 soil horizon at the farthest distances from the watering point were indicative of a reduction of compression from cattle hoof impact penetrating down through the A and Bt1 horizons. This would be expected as cattle-use would be more diffuse and less severe at the farthest distances from the watering point (Valentine 1947; Holscher and Woolfolk 1953; Van Der Schijff 1957; Lange 1969; Van Wyk and Fairall 1969; Young 1970; Sneva *et al.* 1973; Western 1975; Glantz 1977; Graetz and Ludwig 1978; Forman and Godron 1981; Goodman 1982; Collinson 1983; Dregne 1983; Sinclair and Fryxell 1985; Andrew 1988; Hart *et al.* 1991; Pinchak *et al.* 1991; Hart *et al.* 1993; Pickup *et al.* 1994; Fusco *et al.* 1995; Thrash and Derry 1999; and Ganskopp 2001). It is unclear whether or not the bulk density data we collected were robust enough in sample size to accurately determine trends in bulk density across the different distances from the watering point. This may be the case since the impact penetrometer data we collected were very consistent and demonstrated a clear trend across all distances from the watering point that the soil nearer the watering point was harder and more compact than distances farther away.

Our data indicate that soil nutrients were transported and deposited nearer the cattle watering point via foliage consumption at further distances by herbivores and subsequent defecation and urination near the watering point. We submit that these changes are due to many years of use by cattle (Georgiadis 1987). Similar trends in soil nutrient content around watering points have been found in other studies (Weir

1971; Georgiadis 1987; Tolsma *et al.* 1987; Perkins and Thomas 1993a; and Mphinyane 2001). At the further distances, beyond 1.93 km, there was a rise in soil nutrient levels of the A horizon which suggests that the transportation of nutrients offsite was diminished. This decrease in transportation of nutrients offsite may be mediated by decreases in grazing by cattle. Other studies have found that foraging by cattle decreases substantially at distances greater than 1.6 km from water (Valentine 1947; Holscher *et al.* 1953; Fusco *et al.* 1995; and Ganskopp 2001).

Low densities of winterfat occurring between the watering point and 50 m are likely explained by the heavy trampling and browsing that occur by cattle. The increase in density of winterfat at 0.48 km may be in response to a decrease in browse pressure but also, a very high soil nutrient content which may compensate for the heavy herbivory occurring at this distance from the watering point. We also submit that the increase in winterfat density from 0.97 to 2.41 km was due to increases in soil nutrients that occurred over the same distance as well as a decrease in browse pressure from cattle. Sickle saltbush demonstrated a much different response to herbivory and distance from watering point than winterfat. Although herbivory was high and trampling pressure was also high near the watering point, sickle saltbush densities were much higher than winterfat. With casual observation of excavated sickle saltbush and winterfat plants, we found that sickle saltbush appeared to allocate less resources into developing its roots than winterfat. Sickle saltbush had a large taproot and few fibrous roots while winterfat had both extensive fibrous roots as well as a large taproot. The higher densities of sickle saltbush could also be attributed to the fact that they are a shorter lived species than winterfat (Matney 2010a), or because they have more woody and sparse above ground stems (Driese and Reiners 1997 and Matney 2010a), produce lateral meristems that occur below the soil surface (Foiles 1974), have longer lived seed (Nord *et al.* 1969 and Foiles 1974), and have larger seed (Wein and West 1971; Springfield 1973; and Abouguendia 1995). Density of sickle saltbush was greatest at 0.48 km and decreased all the way to 1.45 km from the watering point. This decrease in density with distance appears to follow the trend in

reductions in soil nutrients as well as decreases in theoretical grazing pressure. The resurgence in sickle saltbush density at 1.92 km through 2.41 km from the watering point may be due to further decreases in grazing pressure and the associated increases in soil nutrients. In general, this seems to indicate that sickle saltbush thrives under moderate to heavy cattle disturbance as long as soil nutrients are also high. Densities of sickle saltbush were four to five times higher than winterfat regardless of distance from the watering point and this suggests that sickle saltbush had a greater influence on the current ecosystem at the time of this study, rather than winterfat.

Nutrient accumulation around watering points may become toxic to plants (Heshmatti 1997). Although nutrient content did not appear to be at toxic levels in our study, using annual plant abundance as an index, we found that conditions near watering points, less than 0.48 km, were not conducive for the establishment and growth of perennial plants, save creeping wildrye. However, at 0.48 km from water, annual plant abundance peaked. At further distances, annual plant abundance steadily declined with distance from the watering point. This suggests to us that annual plants were favored with increased levels of disturbance, but only to a point. A high soil seedbank near the watering point (Navie *et al.* 1996), high soil nutrient levels (Weir 1971; Georgiadis 1987; Tolsma *et al.* 1987; Perkins and Thomas 1993a; and Mphinyane 2001), and an annual life cycle that occurs outside the grazing season could partially explain why density of annuals did well at 0.48 km from the watering point. Grazing, beginning in October, after annual plants have set and/or dispersed seed is likely to enhance the seedbed for annuals by disturbing the soil surface and working seeds into the soil. Of special note, at the highest levels of disturbance, 50 m from the watering point, even annuals were not successful in reproduction and establishment. We conclude that this is likely due to poor physical soil conditions and high levels of cattle hoof impact, which creates a poor seedbed environment that is characterized by poor soil surface aggregate stability, higher soil compaction, and little to no vegetation (Van Der Schijff 1959 and Graetz and Ludwig 1978). Perennial plant

density displayed a different association with distance from the watering point than did annual plant density.

The association of perennial plant density, with distance from the watering point resembled a parabolic shape, creeping wildrye excluded, where perennial plant density was least at the medial distances and highest nearer and further away from the watering point. This association was very similar to the relationship that soil nutrients displayed in conjunction with distance from the watering point. The numbers of perennials near the watering point were composed mostly of bottlebrush squirreltail and sickle saltbush. These plants at close distances to the watering point were observed to be smaller and younger, representative of plants that are establishing but not thriving. This makes sense since Navie *et al.* (1996) found that the soil seedbank may be highest near the watering point, supporting our conclusion that sickle saltbush and bottlebrush squirreltail have the best seed characteristics for establishing on poorly prepared seedbeds. The number of perennials decreased toward the middle distances from the watering point possibly due to greater competition with annual plants and lower levels of soil nutrients. At greater distances the numbers of perennial plants increased due to increases in other bunchgrasses such as Indian ricegrass and *Poa nevadensis* as well as the native shrub winterfat. Indian ricegrass appeared to be quite susceptible to disturbance, appearing at distances only greater than 1.93 km. Meanwhile winterfat seemed to be quite resilient and occurred in moderate densities as close as 0.48 km from the watering point. The distance of 50 m from the watering point was inhospitable for all plant species except creeping wildrye. Creeping wildrye was highest in density at 50 m from the watering point, suggesting it may be an indicator of extreme disturbance. The success of creeping wildrye under great disturbance may be due to its vegetative growth via rhizomes and the fact that grazing occurs seasonally during its dormant period (October through March) – limiting its exposure to grazing.

Bunchgrass density decreased out to 1 km from the watering point, hypothetically due to decreases in bottlebrush squirreltail, which closely tracked the

trend for perennial plant density. We expect that trends in bunchgrass density were heavily influenced by changes in bottlebrush squirreltail due to the overall low amount of Indian ricegrass and Nevada bluegrass. While Nevada bluegrass occurred at several distances from the watering point, Indian ricegrass was relegated to distances greater than 1.93 km. Indian ricegrass had been found to decline under heavy grazing use (NRCS 2009). Bottlebrush squirreltail occurred across all distances – small and young plants near the watering point with older and larger plants at distances farther from the watering point. Bottlebrush squirreltail's success without seedbed preparation (Estes 2008) and its generally large seed size may account for its success across all distances, especially near the watering point.

The amount of litter ground cover decreased steadily outward from the watering point. This relationship is inconsistent with our hypotheses, whereas we expected greater accumulation of resources such as litter at distances farther away from the watering point. This observed relationship may be partly explained by accumulations of annual plant material and creeping wildrye nearer the watering point, and a higher incidence of trampling nearer the watering point. The parabolic relationship of bareground cover to distance from the watering point was likely due to the high levels of creeping wildrye, bottlebrush squirreltail, sickle saltbush, and percent litter near the watering point and again due to the higher levels of biological soil crust at the distances of 1.93 km and 2.41 km. Additionally, trends in bareground seemed to follow trends in soil nutrient content. Although bareground and soil nutrient content were parabolic in their relationship to distance from the watering point, they were linear in relationship to each other. As bareground increased, the relative amount of soil nutrients, for instance organic Carbon, decreased in the A horizon. Therefore higher amounts of bareground may indicate lower soil productivity. This might be linked to having low amounts of biological soil crust and the loss of soil through wind and water erosion (Young 1970; Lason 1979; Edroma 1981; O'Connor 1985; Perkins and Thomas 1993a; and Venter 1990).

Although the density of perennials is parabolic in relation to distance from the watering point, the basal bisect area of perennial plants, as indicated by gap-line-intercept data, demonstrated increases with distance from the watering point. The greater sized plants with increasing distance from the watering point may be explained by plants gaining age and biomass (Andrew and Lange 1986b; Perkins and Thomas 1993; Thrash 1998). Along with perennial plant size, bare patch size increased with increasing distance from the watering point. This is contrary to our original hypothesis. Originally we had surmised that bare patch size would decrease with distance from the watering point, with the assumption being that high levels of disturbance would result in high plant mortality and that decreases in disturbance would allow more plants to establish. However our assumptions were incorrect. At the same time that bare patch size was increasing, average perennial plant size increased. This suggest to us that native perennial bunchgrasses and shrubs became larger in size thereby increasing their zones of influence around themselves which resulted in increasing plant competition in plant interspaces with interspaces occupied not by plants, but biological soil crust. This phenomenon has not been discussed in the literature. Overtime, we predict that increased distance between perennial plants along with increased size of perennial plants has the potential to affect the distribution of nutrients, with nutrients pooling around the base of established plants but separated by interspaces of biological soil crust – creating heterogeneity of soil nutrients spatially across the landscape. Additionally, we have observed winterfat plant communities protected from cattle grazing for 20 years in the Mickey Basin, 20 miles to the east, to consist of very large winterfat and sickle saltbush plants with very large interspaces occurring between plants. Interspaces between plants consisted of thick biological soil crust. The primary disturbance that was apparent in these protected areas was that of pocket gophers. These observations suggest that decreases in cattle disturbance may not yield more perennial plants, but it may yield larger and older perennial plants that are separated by larger interspaces which are occupied by biological soil crust.

Increases in bare patch size have been found or hypothesized in other studies to decrease rates of infiltration. In our study, bare patch size was linked to relative rates of infiltration and soil surface aggregate stability. As average bare patch size in between plants increased, relative rates of infiltration decreased and soil surface aggregate stability increased. Even though soils with large bare patch areas were more stable than those with smaller bare patch areas closer to the watering point (higher aggregate stability), rates of infiltration may have been depressed due to higher levels of biological soil crust. Biological soil crust may have limited infiltration rates due to the presence of cyanobacterial filaments and a relatively smooth soil surface (Maestre *et al.* 2002). This explanation seems plausible for the soils in the Catlow Valley, since on average, the soil surface is also quite smooth.

MANAGEMENT IMPLICATIONS

A watering point that had been used for over 50 years in the Catlow Valley, Oregon was examined for evidence of a piosphere affect in relation to longtime use by cattle. We found that there was a clear cattle-use disturbance gradient surrounding the watering point. Many plant metrics, soil metrics, and ecological processes demonstrated a relationship with distance from the watering point. The nature of this relationship was such that areas closer to the watering point were heavily trampled and foraged and areas farther from the watering point were less affected. Zone of influence from cattle use was apparent from 0 to 2.41 km from the watering point. The area closest to the watering point, 50 m, was characterized by soil compaction, little vegetation, and accumulations of litter and soil nutrients. Areas furthest from the watering point favored larger native perennial plants, higher diversity of native perennial plants, higher aggregate stability, and abundances of biological soil crust and lower infiltration rates. These observations are of importance to land managers in that the spatial influence of watering points may extend to at least 2.41 km. This would be important to keep in mind when planning the distribution of watering points and their proximity to sensitive habitats. Overlapping the zone of influence between

watering points by placing watering points too close to each other, within 4 or 5 km of each other, could exacerbate negative effects on soil, plants, and ecological processes.

When grazing areas around watering points it can be assumed that areas closer than 0.5 km are going to be greatly disturbed by trampling and herbivory. At distances closer than 0.5 km it will be quite difficult to maintain native species on Silty 6-10 PZ ecological sites in the Catlow Valley, especially winterfat and Indian ricegrass. Land managers can assume that compaction will be at its highest in areas that are within 0.5 km from the watering point and that soil surface degradation will occur all the way out to 2.41 km from the watering point. Soil biological crust is an important component of this ecosystem in that it comprises much of the soil surface when disturbance is reduced and it is linked to soil stabilization. Distances closer than 1 km from watering points are not likely to support biological soil crust. The presence of soil biological crust and levels of soil surface aggregate stability are indicators of areas that receive less disturbance from cattle use. Distribution of watering points on the landscape is a key factor to consider when planning grazing in the Catlow Valley.

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CHAPTER 4: STATE-AND-TRANSITION MODELS FOR A WINTERFAT ECOLOGICAL SITE IN SOUTHEASTERN OREGON

ABSTRACT

State-and-transition models (STMs) are commonly used to predict and describe changes in ecological attributes and plant community composition and structure. To-date the majority of STMs are conceptual because data and analyses to construct and support models are limited. We collected data in the Catlow Valley of Oregon to develop and test a conceptual three state model for the Silty 6-10 PZ ecological site. This ecological site is comprised of winterfat (*Krashenninikovia lanata* [Pursh] A. Meeuse & Smit), Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth), sickle saltbush (*Atriplex falcata* [M.E. Jones] Standl.), other native bunchgrasses, and creeping wildrye (*Leymus triticoides* [Buckley] Pilg.). Our models and testing incorporated 19 different plant and soil variables, however biological soil crust, soil surface aggregate stability, density of winterfat, and frequency of Indian ricegrass were the most helpful in classifying sites into different states and plant community phases. Models were based on a piosphere study and a long term grazing exclosure study. Hierarchical cluster analysis and a review of group means, standard errors, and 95% confidence intervals were useful in determining different states. Data from this study supported the initial proposed STM for the Silty 6-10 PZ ecological site. States defined were a Reference Winterfat State (State 1), Sickle Saltbush State (State 2), and a Creeping Wildrye State (State 3). Each state corresponds to a site characterized by high, moderate, and low ecological functioning, respectively.

INTRODUCTION

In 1919, Arthur Sampson published information about how plants and plant succession in the western states of the U.S. were affected by grazing. Sampson's work on succession was predicated on work done by Clements in 1916. Clements outlined the framework for how plant composition in an area changed over time relative to time

and disturbance. Further work on the topic of plant succession and plant composition was contributed by Gleason (1926) and by Clements and Weaver (1938). Gleason elucidated parameters that influence plant composition across regions. Clements and Weaver collaborated to construct the book “Plant Ecology”, which was written with plant succession as the overriding theme. In 1935, Tansley contributed greatly to consolidating and defining the term succession, as well as related ecological terms used in studying plant succession and plant associations. Tansley defined succession as “a continuous process of change in vegetation which can be separated into a series of phases.” And, Tansley defined climax as “a relatively stable phase reached by successional change.” A climax can be reached through autogenic processes, processes driven by the plants themselves, or by allogenic processes, processes driven by factors completely external to the plants. Although the scientific study of plants was growing and building, by the mid-1900s a framework suitable for land managers to use as decision making tool on rangelands was lacking. In 1949, Dyksterhuis produced a seminal paper that filled this gap.

Dyksterhuis suggested quantitative means by which land managers could assess and predict plant responses to management, particularly grazing. Dyksterhuis’ model was driven primarily on successional concepts developed by Clements. The theory of succession and climax was parted-out into a format of rangeland condition classes based on plant composition. Condition classes were based on percentages of invader, increaser, and decreaser plant species, as measured against the estimated climax plant composition for the area (Fig. 4.1). As the rangeland approached excellent condition it was termed succession, while changes towards poor condition were termed retrogression. The quantitative climax model proposed by Dyksterhis assumed plant community response to disturbance was linear and predictable thereby allowing management to manipulate the direction of change by increasing or decreasing grazing pressure. This model was utilized as the standard decision making tool on rangelands for over 50 years. However, over time, it became apparent that the

model often failed to predict plant community response to disturbance especially in the semi-arid and arid ecosystems of the West (Wilson 1984).

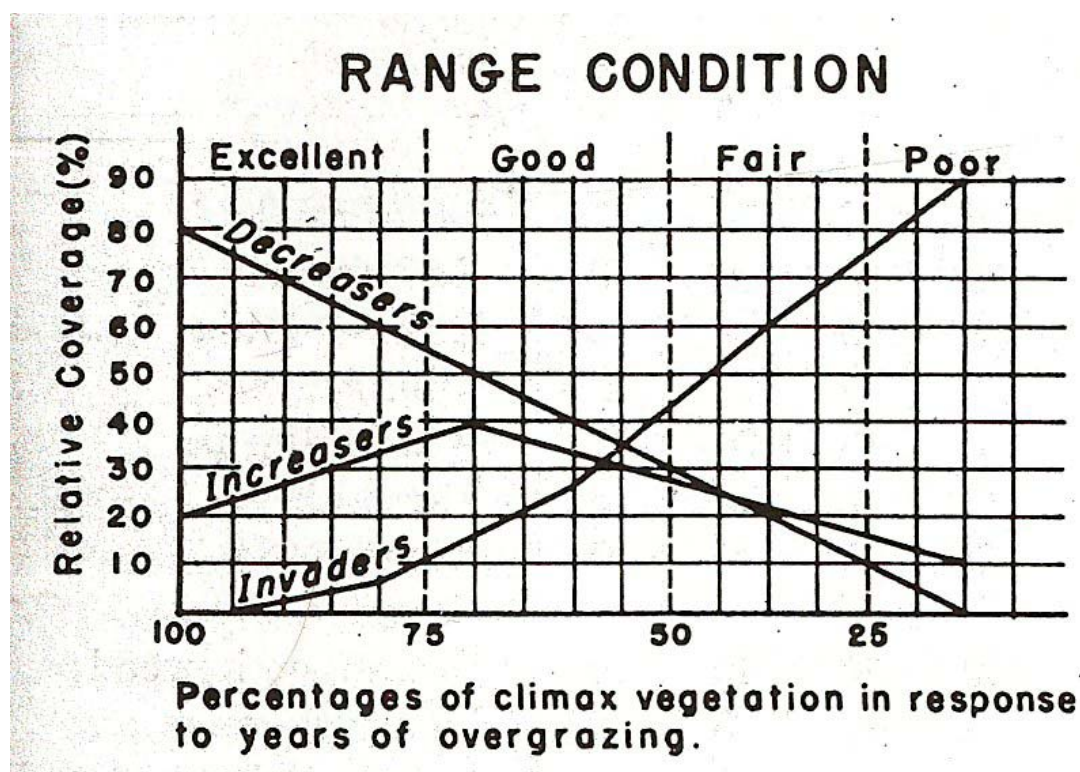


Figure 4.1. Dyksterhuis' 1949 diagram illustrating a quantitative basis for determining range condition. Excerpted from Dyksterhuis' article in the Journal of Range Management (Dyksterhuis 1949).

In 1973, Holling wrote a scientific paper discussing the ideas of stable states and ecological resilience. He defined ecological resilience as the amount of disturbance that an ecological system could absorb without changing state. He also identified a stable ecological system as a system that is maintained at or near an equilibrium state. The model Holling suggested to describe the departure from an old equilibrium state to a new state was called the adaptive cycle (non-equilibrium theory). In the adaptive cycle, when the ecological resilience of an ecological system was exceeded, it would collapse and reorganize in the direction of a new equilibrium (alternative state). The new alternative state may then persist for a period of time or

collapse and reorganize in another new direction. In 1978, Connell published a paper that listed ecological examples illustrating the concepts of non-equilibrium states in ecology. Westoby *et al.* (1989) proposed a model, state-and-transition, that explained and predicted the gray areas of rangeland response that Dyksterhuis' model could not. Dyksterhuis' model was rooted in equilibrium (climax) theory. Westoby's model was rooted in non-equilibrium theory. The main difference between the two models was that Westoby's model of rangeland trajectories was a compartmentalization of plant community composition into states, depending on the driving factors of plant community composition. In contrast, Dyksterhis' model always suggested a single position along the linear scale between retrogression and succession. Westoby's model was an adaptation to Dyksterhuis' model.

Westoby and others (1989) suggested that botanical composition at a site could be helpful when parting site dynamics into specific states. The measurement of an ecological state could be quantitative, but the demarcation between states and the number of possible states were entirely subjective. Westoby *et al.* (1989) suggested that demarcation of states were useful for land managers to apply known management tools to reach land manager objectives. Westoby's approach to modeling has become known as STM. Requirements of a STM are 1) a defined list of alternative states, which can also be called possible states, 2) a defined list of possible transitions that would occur between all sets of possible states, and 3) a detailed list of conditions which precipitate each transition between states. Further refinement of the STM concept was developed by Stringham *et al.* 2003. Stringham suggested that ecological drivers responsible for state changes functioned by modifying the basic ecological processes (energy capture, hydrologic cycle, and nutrient cycle) responsible for state stability and resiliency. She defined a state as a climate-soil-vegetation dimensional domain that can encompass a wide variation in plant species composition supported through a defined set of ecological processes (Stringham *et al.* 2003). Changes that occur in any of these ecological processes may cause a shift in trajectory and a threshold event leading to a new state. Dyksterhuis' model framed plant composition

field values relative to the estimated climax state for that site. Similarly, the STM is framed relative to the estimated historic ecological site description for a site. The ecological site description is a quantitative estimation of the estimated historic climate-soil-vegetation domain. In this way, all other possible alternative states for a site are derivatives from the original historical state. The climate-soil-vegetation relationship varies by ecological site therefore each site has a unique inherent ability to resist change and recover from disturbance. However, when site capacity to recover has been exceeded, an ecological threshold is crossed and the site transitions to a new state with a new climate-soil-vegetation domain. A threshold is simply described as a boundary in time and space between states. Quantitatively, thresholds are difficult to describe, since transitions between states are not clear and may occur over broad timescales. Transitions are the shifts that occur between states, exceeding thresholds. Some transitions are reversible, while some are irreversible. Reversible transitions occur within a state boundary and are associated with at-risk plant community phases. Release from the disturbance causing the at-risk community phase allows for stabilization of ecological processes and reverse of the transition toward a threshold. Irreversible transitions imply a threshold has been crossed, autogenic repair is not possible, and human inputs are required to restore the site to a previous state. The extent to which a transition can be reversed depends on the magnitude of disruption to the ecological process driving the transition. Usually, a change to a new state is facilitated through the repair or degradation of an ecological process that affects soil or vegetation (Whisenant 1999). Some ecological processes can be repaired on short time scales, while some soil ecological processes cannot be repaired within a land manager's life time. Often, a particular event, such as fire or lack of fire, may act as a trigger to engage a transition between states (Miller *et al.* 2000 and Archer 1989).

Since 2003, Stringham and colleague's revised processed based STM has provided the best description of a working framework for modeling and predicting rangeland response using non-equilibrium theory (Fig. 4.2). Stringham's model not only includes the possibility of multiple states but also multiple vegetation phases

within states. Here, changes within states (phases) follow plant community pathways and are measured in terms of changes in plant composition. These changes in plant community composition are influenced by many factors, including disturbance regime, and the corresponding ties to the ecological processes of site hydrology, energy capture, and nutrient cycling (Whisenant 1999). When a disturbance results in the decoupling of an ecological process, the end result is a transition across a threshold to a new state (Petersen 2005). Recently, Briske and others (2008) have further revised the STM (Fig. 4.3).

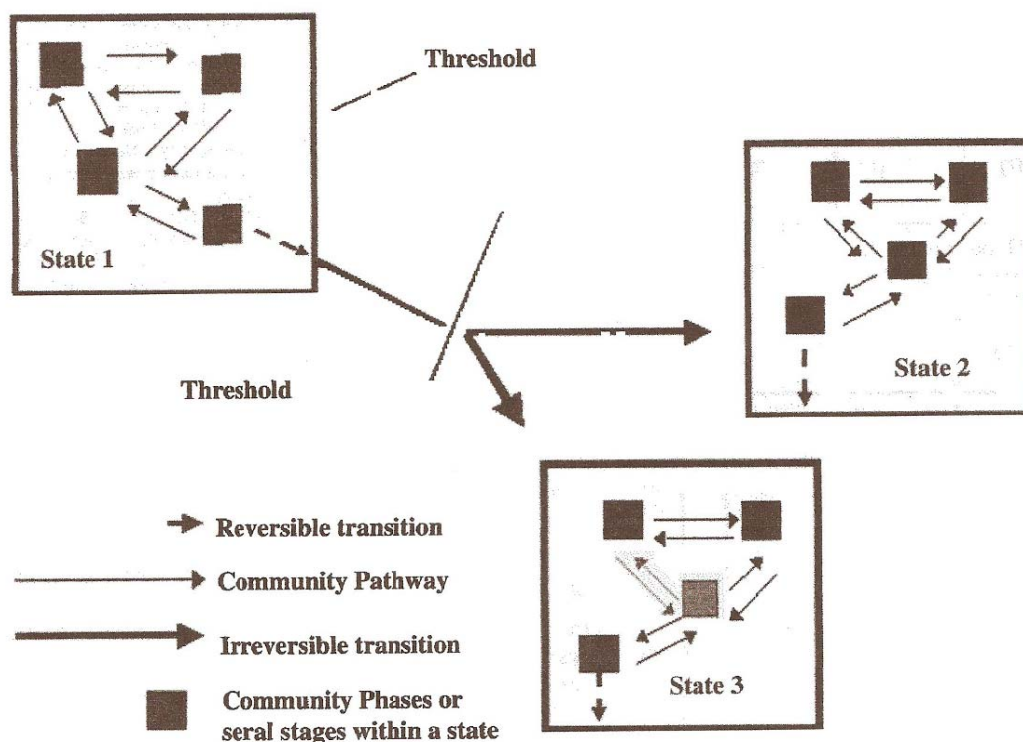


Figure 4.2. STM display, excerpted from Stringham and others (2003).

The newer STM proposed in 2008 incorporates the concept of ecological resilience and other factors (Briske *et al.* 2008). Briske and colleagues described ecological 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.” Trends in ecological resilience (movement closer to or further away from a potential state transforming event) are described as feedbacks that can either be positive or negative. Negative feedbacks reinforce the current state, while positive feedbacks move the system towards a different state. States having strong negative feedbacks are thought of as having high ecological resilience. States having weak negative feedbacks have low ecological resilience. When one feedback becomes more dominant than the other, this change in feedback dominance is termed a “feedback switch.” When a feedback switch occurs, it is generally assumed that the system has crossed a threshold (Briske *et al.* 2008). Restoration pathways describe the techniques required to repair the negative feedback mechanisms (ecological processes) associated with the desired state. These concepts are graphically represented in Figure 4.4. Where State 1 is considered the higher ecologically functioning state and State 2 is considered the lower ecologically functioning state.

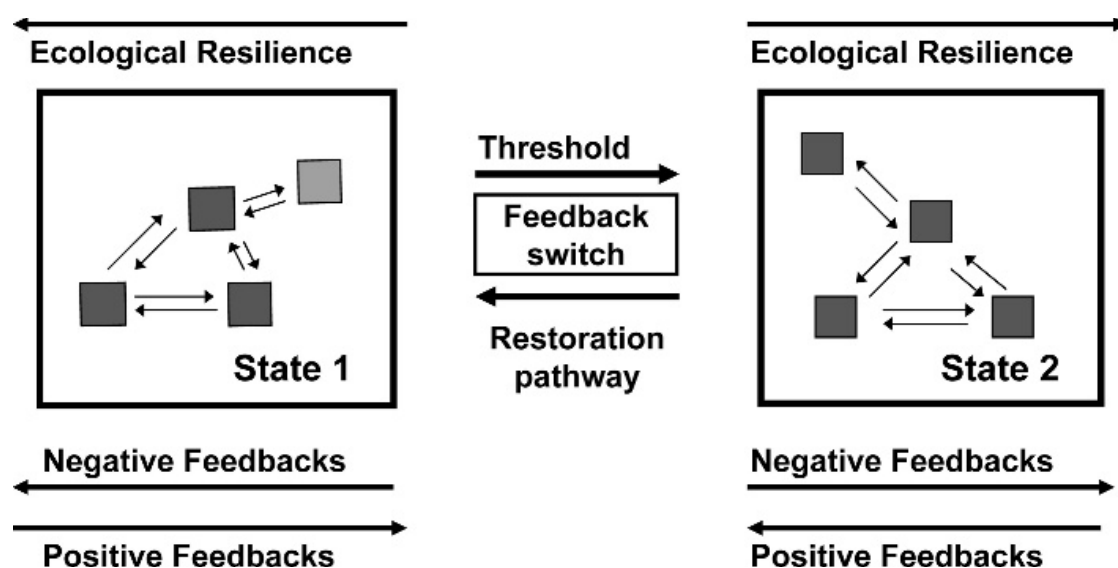


Figure 4.3. Revised STM, excerpted from Briske and others (2008).

STMs have been successfully used to describe woodlands, shrublands, grasslands, and several other ecological systems. Changes in vegetation and soil are

measured to gauge and predict transitions between alternative states and phases within states. Each ecological site warrants its own unique STM, the foundation of which is based on estimation and quantification of a historical reference plant community. Historical information, land manager experience, and scientific data are used to quantify the historical reference plant community and gauge changes towards alternative stable states. The objectives of this study were to:

1. Develop a STM for the Silty 6-10 PZ ecological site(s) (R024XY011OR) in the Catlow Valley of Oregon using the ecological site description, historical information, experience, and scientific knowledge.
2. Test the hypothesized STM using data collected from a piosphere study and a long-term (five year) grazing exclosure study.
3. Refine the STM using test results and interpretation of expert knowledge of the ecological site(s) and onsite ecological processes.

MATERIALS AND METHODS

Study Area

The study was conducted over a 130 km² area within the Catlow Valley of southeastern Oregon. The average elevation of the valley is 1,400 m with the major landforms being lake terraces and lake plains. Soils are fine textured Aridisols, primarily of the Spangenburg soil series (Appendix) ranging from sandy loams to clay. The major ecological site is the Silty 6-10 PZ (NRCS 2009). Annual rainfall for the ecological site ranges from 127 to 254 mm (NRCS 2009). Rainfall measured on site at the study area for the water years of 2006 and 2007 were 222 mm and 129 mm, respectively. Mean annual temperature and rainfall from 2005 to 2008 are presented (Fig. 4.4).

Vegetation was composed of a mixture of shrubs, bunchgrasses, forbs, annuals, and one native rhizomatous grass species. The common native shrubs are winterfat (*Krashenninikovia lanata* [Pursh] A. Meeuse & Smit) and sickle saltbush (*Atriplex falcata* [M.E. Jones] Standl.), and the occasional green rabbitbrush (*Chrysothamnus*

humilis Greene). Native bunchgrasses include bottlebrush squirreltail (*Elymus elymoides* [Raf.] Swezey), Sandberg bluegrass (*Poa secunda* J. Presl), and Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth). The predominate native perennial forb is gooseberryleaf globemallow (*Sphaeralcea grossulariifolia* [Hook. & Arn.] Rydb. subsp. *grossulariifolia*), and the native rhizomatous grass species is creeping wildrye (*Leymus triticoides* [Buckley] Pilg.). Annuals present include cheatgrass (*Bromus tectorum* L.), clasping pepperweed (*Lepidium perfoliatum* L.), herb sophia (*Descurainia sophia* [L.] Webb ex Prantl), spreading wallflower (*Erysimum repandum* L.), and shortstem lupine (*Lupinus brevicaulis* S. Watson).

The Catlow Valley has a long history of livestock use, beginning in 1872 (Shirk 1956 and French 1964). In recent history, the Catlow Valley study area has been used as fall/winter range (October through March). Prior to 2002, movement of cattle within the valley was unimpeded, however, in 2002, three pastures were established. Since that time, pastures have been used every fall/winter using an alternating north to south deferred rotation management plan. In this plan, the middle pasture has been used at approximately the same time every year. In the years 2004 and 2005, 1 800 and 1 500 head of cattle, respectively, used the valley from October to March.

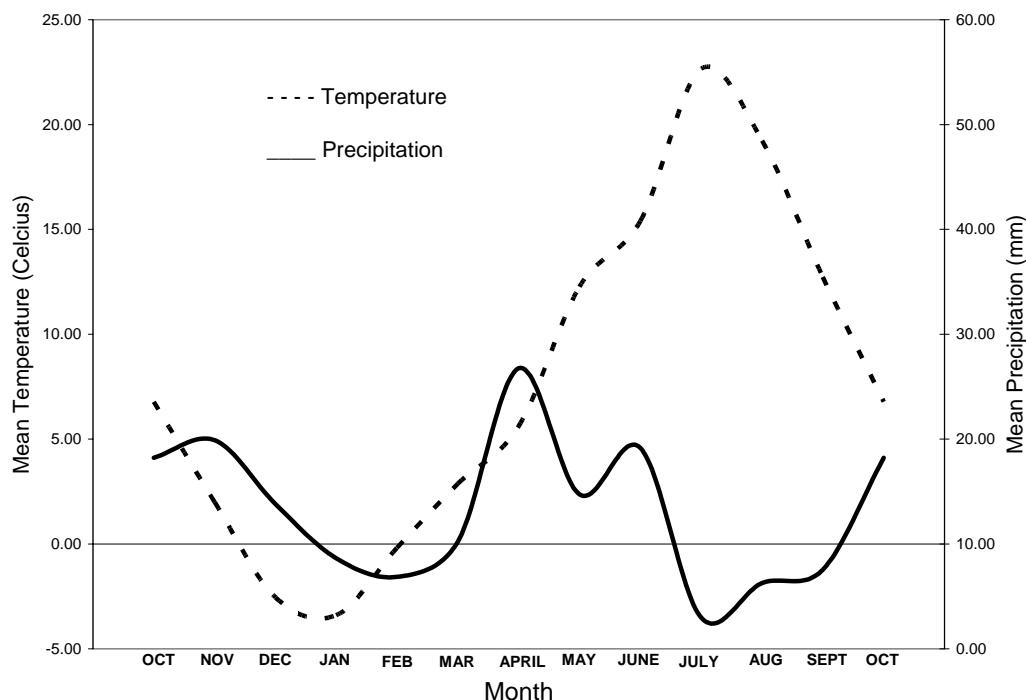


Figure 4.4. Monthly mean temperature (°C), dashed line, and mean precipitation (mm), solid line, for May 2005 through May 2008. Lines are smoothed interpolations between average monthly values.

Model Development

The strategy used for the development of the STM followed examples by Petersen (2005), Bestelmeyer *et al.* (2009), Petersen *et al.* (2009), and Shaver (2010). The ecological site description, historical information, experience, and scientific knowledge were used to develop the proposed STM.

Model Testing

The proposed STM for the Silty 6-10 PZ ecological site(s) was tested using data collected from a long-term grazing enclosure study (Matney 2010a) and a piosphere study (Matney 2010b). The long-term grazing enclosure study determined the effects on plant metrics, soil metrics, and ecological processes from protection from cattle and wildlife herbivory from 2002 to 2007. The grazing enclosure study was conducted across 16 locations within a 50 km² area in the Catlow Valley of

Oregon (Fig. 4.5). Additionally, a piosphere study was conducted in 2007 in order to better understand how cattle grazing affected soils, plants, and ecological processes as a function of distance from a cattle watering point. Results from the study showed that at least 19 variables (Table 4.1) changed as a function of distance from the watering point, out to a distance of 2.41 km (Matney 2010b). The 19 variables measured in the piosphere were also measured in the grazing exclosure study. Results from both studies led to the further development of the proposed process based STM.

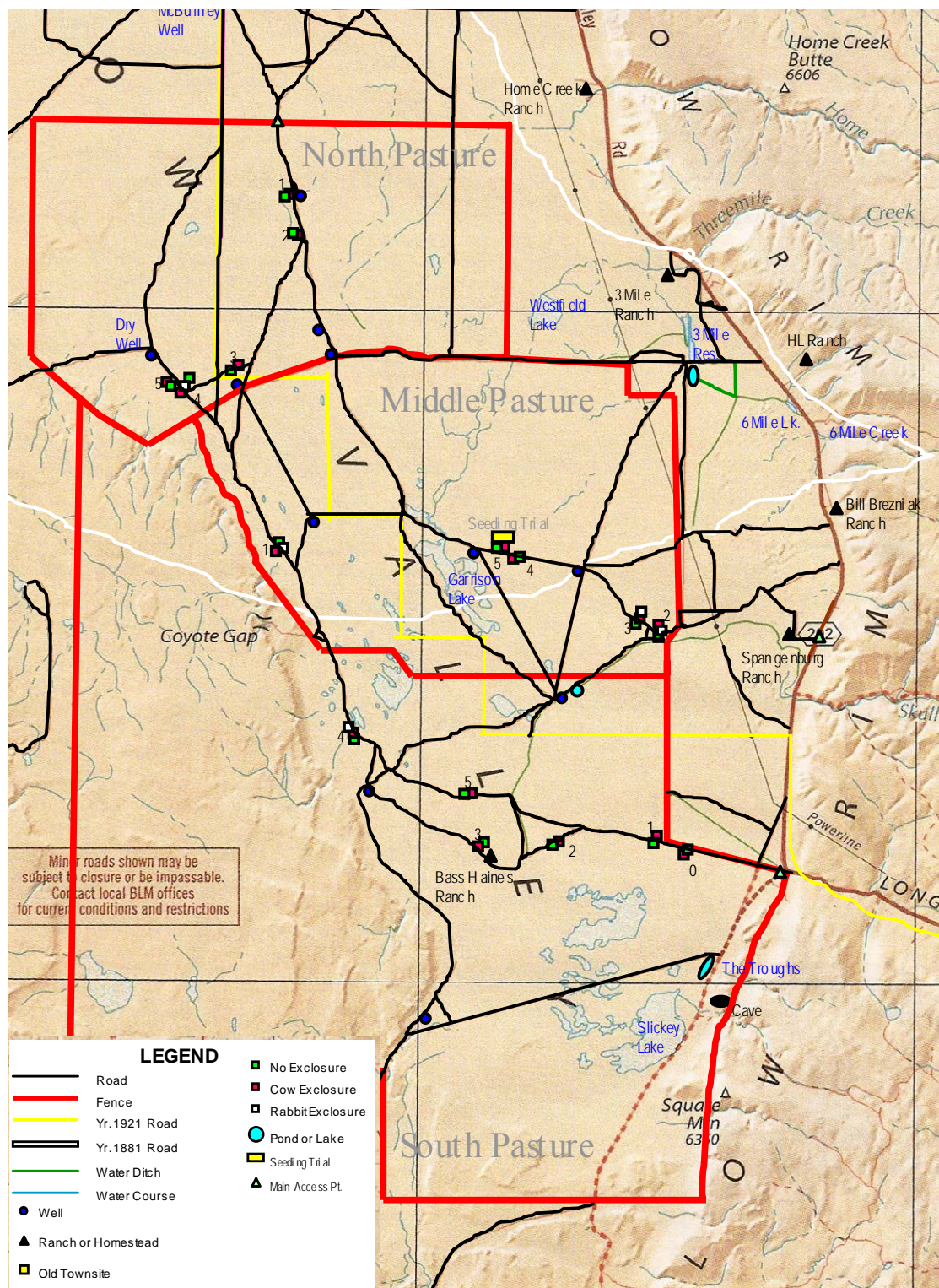


Figure 4.5 Map of the Catlow Valley Study Area.

Table 4.1. Nineteen variables included in the hierarchical cluster analysis for determining differences between sites in the Catlow Valley, OR.

Variables Included in Hierarchical Cluster Analysis
Presence/Absence of Creeping Wildrye
Presence/Absence of Sandberg Bluegrass
Presence/Absence of Gooseberry Globemallow
Presence/Absence of Indian Ricegrass
Mean Density of Sickle Saltbush
Mean Density of Winterfat
Mean Density of Annual Plants
Mean Soil Surface Aggregate Stability
Mean Infiltration Rate
Percent Organic Carbon of the A Soil Horizon
Thickness of the A Soil Horizon
Percent Bareground
Percent Biological Soil Crust
Percent Litter
Soil Penetrometer Resistance from 0 to 5 cm in Soil Depth
Soil Penetrometer Resistance from 5 to 10 cm in Soil Depth
Soil Penetrometer Resistance from 10 to 15 cm in Soil Depth
Mean Bare Patch Size between Perennial Plants
Mean Perennial Plant Size (Canopy and Basal)

Model Refinement

The proposed STM was refined using test results from the grazing enclosure study (Matney 2010a), piosphere study (Matney 2010b), historical information, and current scientific knowledge.

Design

The objective of the grazing enclosure study was to determine the ecological resilience of the Silty 6-10 PZ ecological site(s) in the Catlow Valley when protected from cattle and wildlife herbivory for five years. The objective of the piosphere study was to determine the ecological differences that were apparent in relation to distance from a cattle watering point that has been in use since the 1960's. Data from these two studies were used to test the proposed STM and delineate state and community phase characteristics. The 19 variables quantified in both studies (Table 4.1) were

used in this analysis. These 19 variables included vegetation metrics, soil metrics, and ecological processes. These variables were included due to their importance in the piosphere study (Matney 2010b), their use in the proposed STM, and their importance in resilience feedback mechanisms.

Grazing Exclosure Study

From June through July of 2002 and 2007, vegetation species density and frequency metrics (nested contiguous quadrats) were taken from each of three to five linear transects at each study plot. Plant density for perennial bunchgrasses and perennial shrubs were recorded in contiguous 1 x 1 m frames for a total number of frames at each transect as determined by sample size calculations for winterfat. Plant density measurements for contiguous frames, frame size and number of frames, were determined for individual annual plant and perennial rhizomatous plant species according to sample size calculations in 2002. Sample size calculations were based on estimating 95% confidence intervals within 25% of the mean. Plant frequency was measured for all species in 0.2 x 0.2, 0.3 x 0.3, 0.5 x 0.5, and 1 x 1 m nested frames. The number of frames recorded for plant frequency was the same for all species. The total number of frames recorded for plant frequency at each transect was determined by the maximum number of contiguous frames used to measure plant species density.

Starting point and direction of linear transects were assigned randomly within each plot. Each transect measured a minimum length of 18 m. Within each plot was a 2 m buffer within the outer boundary of the 40 x 40 m plot. Vegetation measurements were not taken in this buffer area. In addition to density and frequency metrics, line-intercept (Elzinga *et al.* 1998) and gap-line-intercept (Herrick *et al.* 2000) methods were used along linear transects. Linear transect beginning and ending points were permanently marked with 23 cm metal stakes and 12 cm plastic fluorescent road construction tassel markers. Biological soil crust was visually identified as those soil surfaces having a darker surface and/or having lichen and/or moss (Belnap *et al.* 2001; Belnap and Lange 2003; and Rosentreter *et al.* 2007).

Soil penetration was determined using an impact penetrometer (Herrick *et al.* 2005). Penetration was measured on the same two transects in each study plot that point intercept data was recorded for vegetation. Eight points, 2 m apart, were selected to measure soil penetration along each transect. In summary, the total number of points sampled for each plot was 16. And at each point the number of strikes it took to pass-through a given soil depth range (5 cm) was recorded. Soil depth ranges observed were 0-5, 5-10, 10-15, and 15-20 cm.

Soil nutrient content was determined for the A horizon for all treatments and locations in 2007. Three equidistant samples were taken along a random diagonal transect bisecting each treatment plot. Soil nutrient tests were performed by the Oregon State University Central Analytical Lab. All samples were analyzed for total organic carbon and total organic nitrogen with a Leco CNS-2000 Macro Analyzer. Samples were determined for plant available inorganic phosphorus using the Olsen Phosphorus test (Olsen *et al.* 1954).

Soil pH samples were taken at the same three positions along line transects as soil nutrients in 2007. Soil pH was determined for all treatments and locations at the following depths: 1) A horizon, 2) thin 1 cm thick transition zone occurring between the A and Bt1 horizons (± 0.5 cm at the horizon interface), 3) Bt1 horizon, and 4) Bt2 horizon. The depths of soil layer transitions and mean layer depths were recorded. Soil pH, to the nearest 0.01, was determined using a LaMotte pH PLUS DIRECT Digital pH Meter and methods for soils that may vary due to salt content (Jones 2002). Bulk densities were determined using the volumetric ring method (NRCS 1999). Volumetric ring bulk density soil sample cores (6 cm diameter) were taken at the same locations as soil pH at three depths: A horizon, Bt1 horizon, and Bt2 horizon. Bulk density was determined for soil cores after drying at 100° C for 24 h.

One measurement of relative soil infiltration was made at each plot in 2007 using a single ring infiltrometer following protocols by Herrick and colleagues (2005). The diameter of infiltrometer rings was 50 cm. The soil was pre-wetted prior to setting infiltrometer rings. Infiltration measurements occurred at an undisturbed single

random point in shrub interspaces within a 5 m radius of each treatment plot's center point. Infiltration was measured for 1.5 h, keeping a constant head of 2.5 cm (± 0.5 cm) in depth. The last two readings of infiltration over the 1.5 h measurement period were used to obtain average relative infiltration rates at each study plot.

Soil surface aggregate stability was measured at 18 equidistant points (1 m apart) occurring on a treatment plot line transect that was oriented perpendicular to the line transect used for soil nutrients, pH, and bulk density using methods outlined by (Herrick *et al.* 2005). Average soil surface aggregate stability was determined for all plots and treatments in 2007 only.

Piosphere Study

One watering point was selected in the Catlow Valley that had been in operation for approximately 50 years and had a long history of cattle-use. Watering point selection was based on the following criteria: 1) the watering point had been used by cattle for at least the last 40 years, 2) the watering point occurred in an area where winterfat plants were present and where there was not a high heterogeneity of soil type or landform type, 3) the watering point was not closely grouped with another watering point or natural standing body of water, 4) drastic changes in geography did not occur on more than half of the area surrounding the watering point or within 2 400 m from the watering point, and 5) cattle supplement locations or fences were not likely to be a confounding factor on cattle distribution around the watering point. At the watering point that met these criteria, a compass sector between 270° and 360° being 2.41km in length was determined as the only suitable area for study due to the other sectors being confounded by fencing, roads, landform, and proximity to other watering points. Within this sector, five azimuths rounded to the nearest 0.5° were randomly chosen for each of 6 radii distances from the water well: 50 m, 0.48 km, 0.97 km, 1.45 km, 1.93 km meters, and 2.41 km (Fig. 4.6). At each point of interest (total of 30) a line transect of at least 36 meters in length was established parallel to the watering point. Vegetation metrics, soil metrics, and ecological processes were measured along

each transect. Vegetation metrics measured were plant frequency, plant density, and percent foliar cover by plant species and plant type. Soil metrics measured were resistance to penetration, bulk density, and soil pH. Other metrics measured related to ecological processes included: soil surface conditions (soil surface aggregate stability, percent bareground, bare patch size, perennial plant size, percent perennial vegetative cover, percent rock, percent soil crack, and percent biological soil crust); hydrologic cycle (relative infiltration rate); and nutrient cycle (soil organic carbon, soil organic nitrogen, plant available inorganic phosphorus in the soil, and percent litter ground cover). Bare patch size was calculated by the gap size between perennial plants determined from gap-line-intercept data. Perennial plant size (size of perennial plants based on canopy and basal measurements) was also determined from gap-line-intercept data. Percent bareground was defined as exposed bare soil void of biological soil crust, vegetation, or other cover.

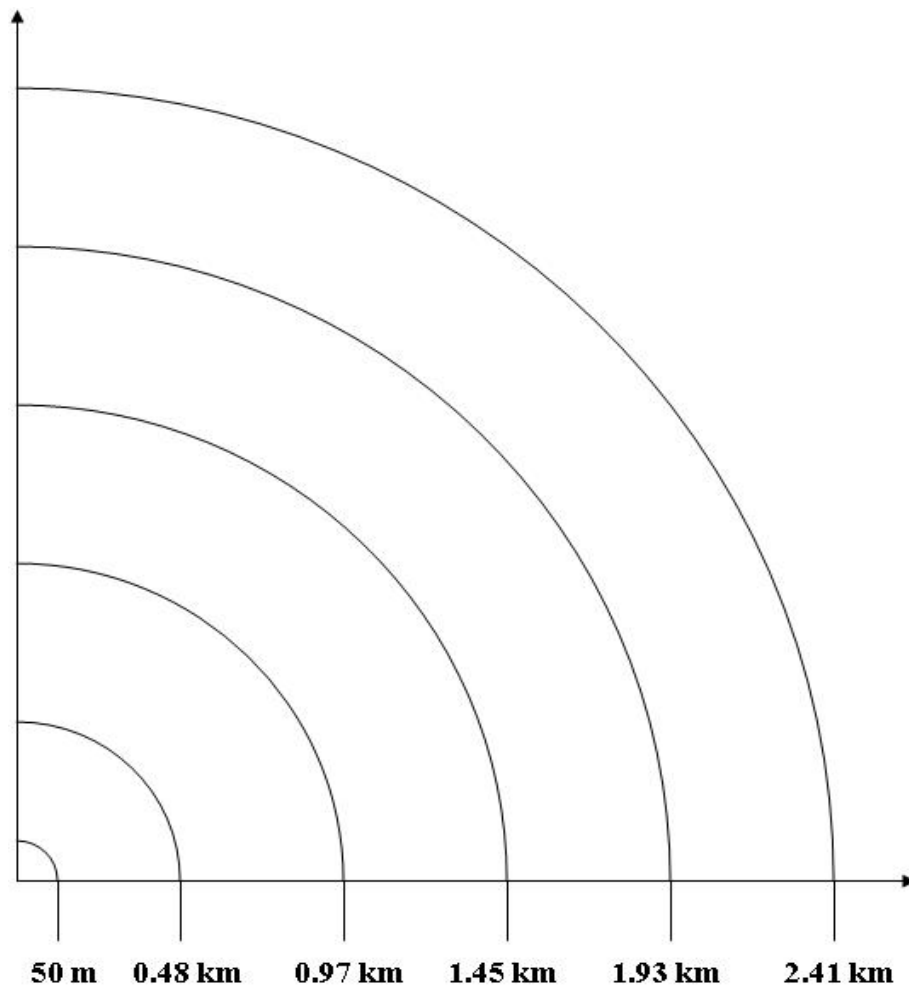


Figure 4.6. Layout of radial sampling distances relative to watering point location in the Catlow Valley.

Soil surface aggregate stability was measured at each transect (NRCS 1999 and Herrick *et al.* 2005). Samples were taken from the first 18 consecutive 1-m increments along each transect. Soil compaction was indexed by measuring soil penetration using an impact cone penetrometer (Herrick *et al.* 2005). Resistance to penetration was measured at the 10, 20, and 30 m increment mark along each transect. At each point the number of strikes it took to pass-through the 0-5 cm, 5-10 cm, 10-15 cm, and 15-20 cm soil depths were recorded.

Soil pH was determined on each transect at the following depths: 1) A horizon, 2) 1 cm thick transition zone occurring between the A and Bt1 horizons (± 0.5 cm at

the horizon interface), 3) Bt1 horizon, and 4) the Bt2 horizon. The depths of soil horizons and mean horizon depths were recorded. Soil pH, to the nearest 0.01, was determined in the laboratory using a LaMotte pH PLUS DIRECT Digital pH Meter ® and methods for soils that may vary due to salt content (Jones 2002). Soil bulk densities were determined using the volumetric ring method (NRCS 1999), and the soil sample cores (6 cm diameter) were taken at the same locations as soil pH at three depths: A horizon, Bt1 horizon, and Bt2 horizon. Bulk densities were determined for soil cores after drying at 100° C for 24 h.

A 1 quart sample of soil was taken from the A horizon to be measured for soil nutrients. All samples were analyzed for total organic carbon and total organic nitrogen with a Leco CNS-2000 Macro Analyzer ®. Samples were determined for plant available inorganic phosphorus using the Olsen phosphorus test (Olsen *et al.* 1954).

Within 5 m of each line transect, one measurement of relative infiltration rate was made at interspaces between shrubs using a pre-wetted, single ring infiltrometer test following the protocols by Herrick *et al.* (2005). The diameter of infiltrometer rings were 50 cm. Infiltration was measured for 1.5 h, keeping a constant head of 2.5 cm (± 0.5 cm) in depth. The last two readings of infiltration over the 1.5 h measurement period were used to obtain average relative infiltration rates at each study plot.

Point-intercept cover data were recorded every 30 cm along each line transect for a total distance of 36 m or 121 points (Elzinga *et al.* 1998). Biological soil crust was visually identified as those soil surfaces having a darker surface and/or having lichen and/or moss (Belnap *et al.* 2001; Belnap and Lange 2003; and Rosentreter *et al.* 2007). Plant basal cover (gap-line-intercept) was recorded along this same 36 m section of transect (Herrick *et al.* 2005). Plant density for perennial bunchgrasses and perennial shrubs were recorded in contiguous 1 x 1 m frames for a total number of frames at each transect as determined by sample size calculations for winterfat, when present, or for saltbush when winterfat was not present (Elzinga *et al.* 1998). Plant

density measurements for contiguous frames, frame size and number of frames, were determined for individual annual plant and perennial rhizomatous plant species according to sample size calculations. Sample size calculations were based on estimating 95% CIs within 25% of the mean. Plant frequency was measured for all species in 0.2 x 0.2, 0.3 x 0.3, 0.5 x 0.5, and 1 x 1 m nested frames. The number of frames recorded for plant frequency was the same for all species. The total number of frames recorded for plant frequency at each transect was determined by the maximum number of contiguous frames used to measure plant species density.

Statistical Analysis

Data was collected from many locations (sites) during both the grazing exclosure study and piosphere study. Experimental plots accounted for the sites sampled in the grazing exclosure study, and randomly chosen sector locations accounted for the sites sampled in the piosphere study. Each of the two studies were independently clustered into ecologically meaningful groups using 19 data variables (Table 4.1). This was accomplished using agglomerative hierarchical cluster analysis in PC-ORD[®] (McCune and Grace 2002). Sorensen distance measures and Flexible Beta linkage methods were used to determine the groups. Sorensen distance is a proportion coefficient used to calculate intergroup dissimilarities, and it is also known as the Bray-Curtis coefficient (Faith *et al.* 1987). The Flexible Beta linkage method is more compatible with Sorensen distance measures than the Ward method (McCune and Grace 2002). Groups from each study were identified from the hierarchical cluster analysis and were independently displayed in dendrograms to test the proposed STM. Each dendrogram was scaled by Wishart's objective function and converted to a percentage of information remaining. In general, clustering of groups with greater amounts of information remaining signifies that the similarities between groups are stronger than if clustering had occurred with less information remaining. For instance, as the amount of information remaining nears zero, all groups become clustered together into one large group. Therefore, the strongest clustering of groups (*ie.* having

the greatest similarity within groups) are those that occur with the most information remaining. After groups were identified from the two different dendrograms, apparent differences between groups were verified using nonmetric multi-response permutation procedures (MRPP) with Sorenson distance measures. The chance-corrected within-group agreement (A) and p -values (P) were reported. When groups derived from the dendrograms were verified using MRPP, they were compared to the proposed STM. Means and standard errors for variables were used to graphically describe and delimit differences between groups. Petersen (2005) and Petersen *et al.* (2009) used means and standard errors along with agglomerative hierarchical cluster analyses to define membership of sites to different ecological states. Shaver (2010) also used cluster analysis to assess the validity of a proposed STM. Once sites from each study were classified into groups and compared to the proposed STM, they were used to further develop state and community phase characteristics. When state and community phases were determined, confidence intervals (95%) were computed for each of the 19 variables measured in order to describe the range of variability of each state or plant community phase. Shaver (2010) also used 95% confidence intervals to define ecological states and plant community phases.

RESULTS

Model Development

A three state model was developed for the Silty 6-10 PZ ecological site based on the ecological site description, historical information, experience, and scientific knowledge was developed. State characteristics, at risk community phases, thresholds, resilience feedback mechanisms, and transitions were included. The proposed STM is displayed in Fig. 4.9. The entire Catlow Valley used to be an ice age lake (Cressman 1942). By the time settlers arrived in the late 1800's, the valley was no longer a lake, but was dominated by shrubs and bunchgrasses (Shirk 1956 and French 1964). At this time, settlers remarked that winterfat and bunchgrasses in the valley were approximately 60 cm tall and that the plant community stretched across

the entire valley (Shirk 1956 and French 1964). When settlers arrived, the area had already been inhabited by Native Americans for thousands of years (Cressman 1942). Fossils found in the valley and local region indicate that bison (*Bos bison* spp. *oregonus*; Allen 1942) were present in the valley over the last several centuries prior to arrival of European settlers (Cressman 1942 and Anan 1994). Herbivores present today and at the time of settlement of the Catlow Valley (Verts and Carraway 1998) in order of abundance include: black-tailed jackrabbits (*Lepus californicus*), pronghorn antelope (*Antilocapra americana*), mule deer (*Odocoileus hemionus* spp. *hemionus*), and bighorn sheep (*Ovis canadensis*). It is not known if herbivory by bison prior to European settlement was a significant factor affecting vegetation and soils in the Catlow Valley. The majority of evidence indicates that black-tailed jackrabbits and pronghorn antelope were the dominant herbivores in the Catlow Valley and local region (Cressman 1942, Anan 1994, and Verts and Carraway 1998).

Beginning in the 1873, cattle and horse herbivory was introduced to the Catlow Valley (Shirk 1956 and French 1964). At this time, cattle were not provided supplemental winter feed. By 1878, settlers remarked that if they were to make a permanent living ranching in the valley they were going to need to provide winter feed. This remark suggests that within 5 years of the first cattle-use the Catlow Valley's winterfat and bunchgrasses were significantly reduced. This was most likely due to continuous year round use by increasing numbers of cattle. By the early 1900's, Griffiths (1902) concluded in his assessment of forage conditions in the northern Great Basin that winterfat was of no great economic importance to ranchers due to its reduction in abundance. Griffiths found winterfat in the region to consist of only a stump and a few shoots ranging from 15-25 cm in height. Plant morphology indicated that the previous season's growth had been completely browsed to the stump. In places where sheep had grazed, Griffiths noted that the soil surface had been reduced to an impalpable powder. Griffiths stated that this was most likely to occur in shrubby areas where the primary users were sheep. Griffiths found that numbers of cattle and sheep ranging in the region ranged into the thousands and that

sheep wintered on the desert lowlands and fed largely on shrubs (Griffiths 1902). In 1942, Cressman observed that healthy stands of native vegetation in the Catlow Valley and the greater northern Great Basin region were typically found in areas too far for livestock to travel from water. At the time of settlement, water available for cattle in the Catlow Valley occurred at stream mouths on the eastside of the valley, in seasonal ponds in winter/spring months, and in Garrison Lake located in the south central portion of the valley. Since that time there have been many water developments (wells) for livestock in order to allow livestock greater access to all parts of the valley, especially in the 1960's when the Bureau of Land Management established several wells across the valley. Our evaluation of winterfat communities of the Catlow Valley during 2002 found that winterfat plants were short in stature, occurred in few areas, and appeared to be low in vigor. Observation of soils in 2002 indicated that soils were disturbed by cattle hoof impact, had low amounts of biological soil crust, and appeared to be windblown in some areas.

Wind movement of soils in the Catlow Valley were observed by Cressman (1942) and John Scharff (Conkling 1967). During our investigations of the Catlow Valley from 2002 to 2007, we also observed movement of soil by wind during thunderstorms (Fig. 4.7) and wind events. Soil erosion occurred in patches across the valley, evidence of which was observed by exposed roots of shrubs and blowouts (Fig. 4.8).

Historical information, the Silty 6-10 PZ ecological site description (NRCS 2009), and existing data have led to conditions that have produced an ecological site characterized by winterfat, sickle saltbush, and perennial bunchgrasses. Sickle saltbush and perennial bunchgrasses were the subdominant plants on this ecological site. Information on forbs is lacking but due to the aridity of the site and variability in precipitation both annually and seasonally – abundance and extent of forbs is assumed to be a function of available moisture. We assume fire has not been a major driver of the ecological site, especially during contemporary times, due to the short stature of plants, relatively large interspaces that occur between plants, and low fuel loads. We

do submit, however, that the primary drivers for the ecological site are herbivory by pronghorn antelope and black-tailed jackrabbits, periodic drought, and frequent high winds. We surmise that the ecological resilience of the site was maintained by the continued input of organic matter from annual turnover of foliage and roots of herbaceous plants as well as leaf and stem shedding of both winterfat and sickle saltbush. The development and maintenance of soil aggregate stability was a function of biological soil crust development and organic matter input from native shrubs and grasses. We submit that the aggregate stability of soil on the ecological site is important to the negative feedback mechanisms responsible for site resilience. Higher soil aggregate stability promotes higher soil water holding capacity, aeration, and mineral cycling. This in turn promotes overall plant production and organic matter inputs. Increases in these variables, especially grass production, on other sites has been shown to promote a uniform distribution of soil nutrients and available soil water for plants. In some cases, this uniform distribution has been found to stabilize organic matter inputs (Kemper and Koch 1966; Tisdall and Oades 1982; Goldberg *et al.* 1988; and Bird *et al.* 2007). We surmise that this stabilization of organic matter inputs and maximized soil aggregate stability strengthened the resilience of the site to herbivory disturbance by native herbivores, periodic drought, and the frequent occurrence of high winds.

Upon the arrival of domestic herbivores to the Catlow Valley, the total amount of herbivory immediately elevated to many times above historic levels. These increases in herbivory depressed the production and cover of native shrubs and grasses (Matney 2010a). Simultaneously, it caused the breakdown of biological crust through hoof impact (Matney 2010b). These changes precipitated lower organic matter inputs and decreased soil aggregate stability (Tisdale and Oades 1982). Through time, plant production and cover was reduced, which increased the amount of bareground and wind mediated soil erosion. Resources such as soil nutrients, soil water, and organic matter diminished in plant interspaces and were relegated to points under shrubs or the bases of surviving bunchgrasses. Declines in bunchgrasses and organic matter

between shrubs caused further reductions in soil aggregate stability, water holding capacity, and mineral cycling. Over-time, with continued herbivory, loss and redistribution of resources weakened the negative feedback mechanisms that maintained the ecological site, which resulted in declines in site resilience.

As time passed, bunchgrasses between shrubs decreased to scarcity and so did biological soil crust. Biological soil crust was relegated to small zones beneath and around shrubs and was all but eliminated in plant interspaces. This resulted in more resources being lost and further concentrated the remaining resources below winterfat and sickle saltbush. With winterfat and sickle saltbush remaining, herbivores focused foraging efforts on the remaining winterfat plants, which shifts the plant composition to sickle saltbush. This loss of winterfat causes further losses of resources and develops abiotic feedback mechanisms controlled primarily by wind erosion, that leads to desertification which in turn develops strong resilience feedback mechanisms and resistance to change (Whisenant 1999).



Figure 4.7. Photo taken in August of 2007 showing windblown soil during a thunderstorm in the Catlow Valley, Oregon.

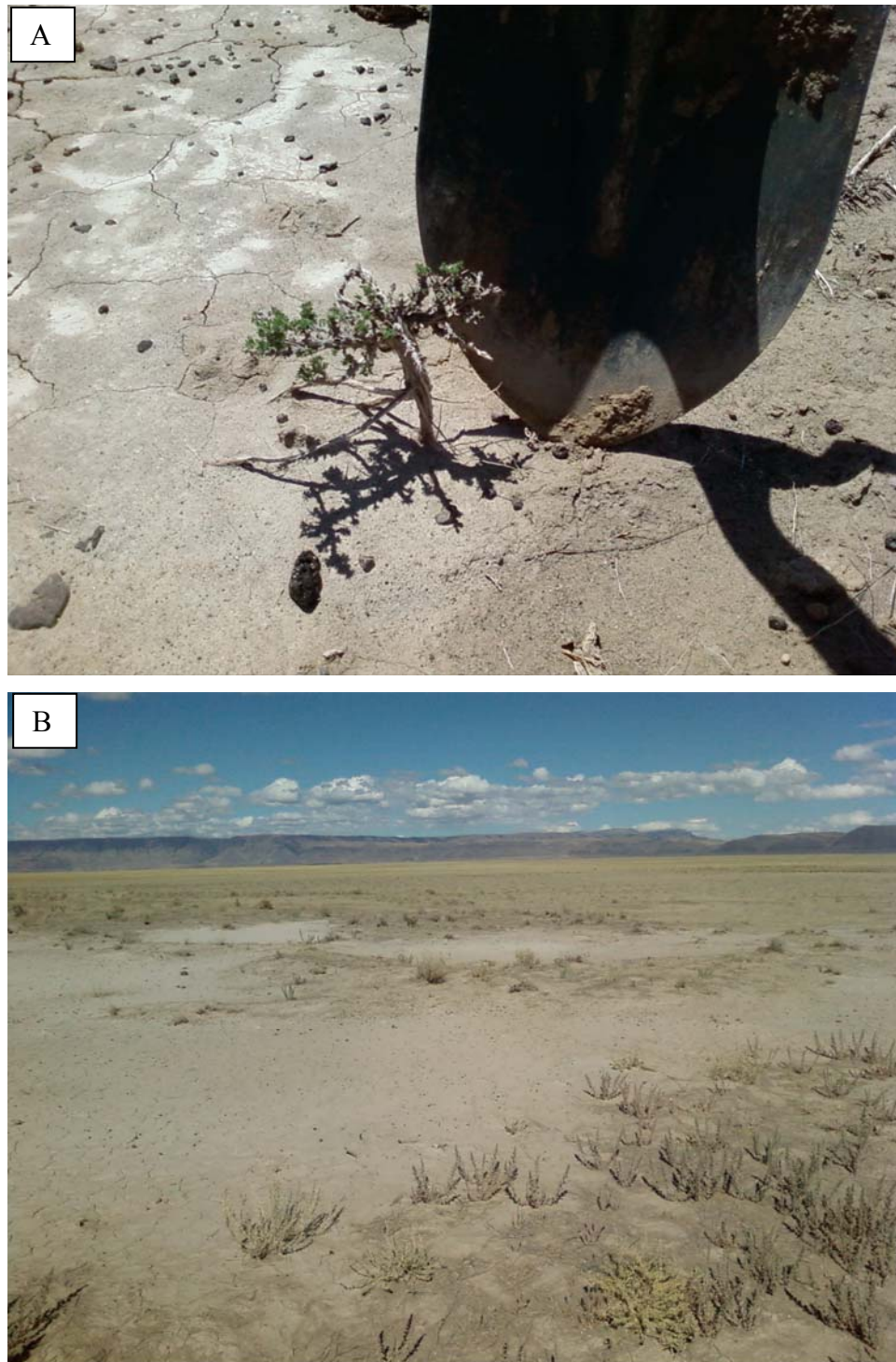


Figure 4.8. Photo (A) taken in May of 2007 showing budsage (*Picrothamnus desertorum* Nutt.) with exposed roots due to wind erosion and photo (B) taken in August of 2007 showing a blowout in the Catlow Valley, Oregon.

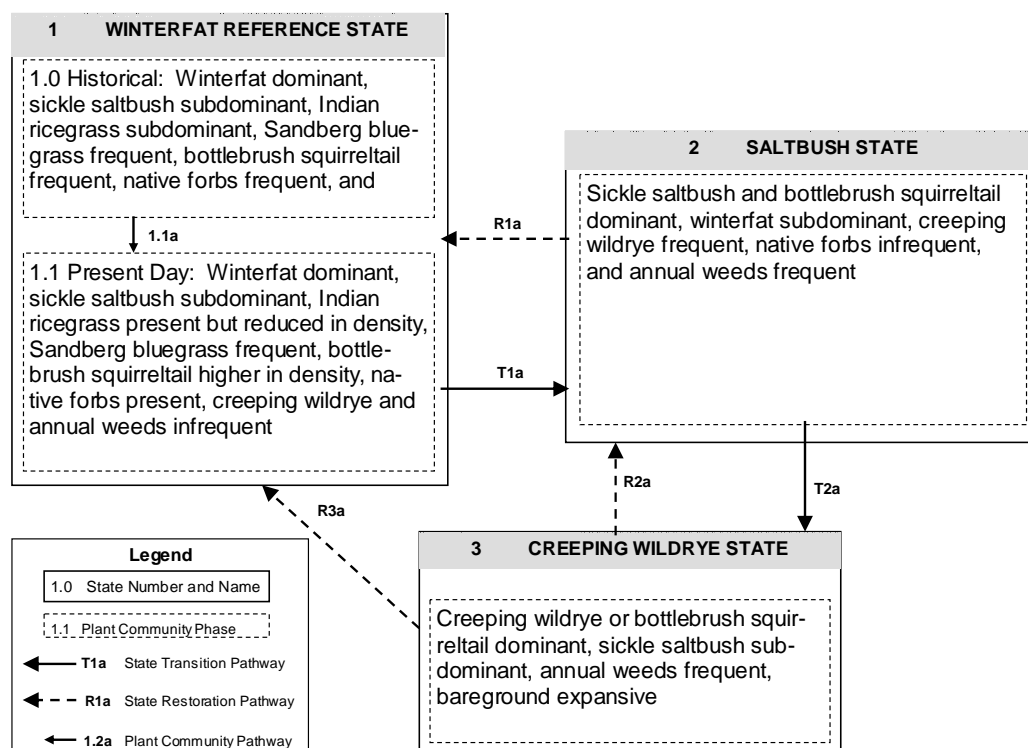


Figure 4.9. Proposed STM for the Silty 6-10 PZ ecological site(s).

Model Testing

A cluster analysis was first run on the piosphere study data set to determine similarities between sites. There were 19 variables included in the analysis (Table 4.1). The cluster analysis method used was Flexible Beta (-0.7) – Sorensen (2.9% chaining; Fig. 4.10). This method provided the best distinction between sites and the least amount of chaining. Groupings on the cluster dendrogram with 60% information remaining indicated three zones in distance from the watering point: 0 to 50 m; 0.48 km to 1.45 km; and 1.93 km to 2.41 km. Differences between groups were verified using MRPP ($A = 0.271$ and $P < 0.00001$). The 1.93 to 2.41 km zone included one site each from the distance of 0.97 km and 1.45 km. Inclusion of these sites in the farthest zone is due to their high aggregate stability and high biological soil crust. Each distance zone grouping represents one of three states, and different states on the dendrogram are defined by a dashed line indicating an ecological threshold (Fig. 4.8). The 1.93 to 2.41 km zone is represented by State 1. Vegetative and soil conditions of

State 1 are most like the historical native plant community for the Silty 6-10 PZ ecological site while States 2 and 3 correspond to an increasingly greater departure in condition from the historic conditions, respectively. The 0.48 to 1.45 km zone is represented by State 2 and is bounded from State 1 by a biotic threshold. The 50 m zone is represented by State 3 and is bounded from State 2 by both a biotic and abiotic threshold.

A second cluster analysis was run for the long-term grazing exclosure study data set using the same methodology that was used for the piosphere study. The cluster analysis method used was Flexible Beta (-0.7) – Sorensen (3.8% chaining; Fig. 4.11). This method provided the best distinction between sites and the least amount of chaining. Our cluster dendrogram of the grazing exclosure data suggests 4 primary groupings with a split in the dendrogram at 75% information remaining. Differences between groups were verified with MRPP ($A = 0.382$ and $P < 0.00001$). The 4 groups depicted in the dendrogram represent 2 states in the state-and-transition framework (States 1 and 2). Within State 1 there were three plant community phases represented as (1.1, 1.2, and 1.3). State 2 was comprised of entirely control plots. Phase 1.1 is comprised of three plots occurring all at a single site. Phase 1.2 consisted of plots that were all protected from cattle, while phase 1.3 was comprised by a mix of all treatments. The threshold defining shift between State 1 and 2 was both biotic and abiotic in nature. Values for the 19 variables included in the analysis were summarized for the 2 States in Tables 4.2 and 4.3. Distinction between states were most apparent when reviewing the means and standard errors for biological soil crust, soil surface aggregate stability, winterfat density, and frequency of Indian ricegrass. Evaluation of variables within a two-dimensional space for groups determined from both the piosphere and grazing exclosure study succinctly revealed the divergence between states (Figs. 4.12 and 4.13).

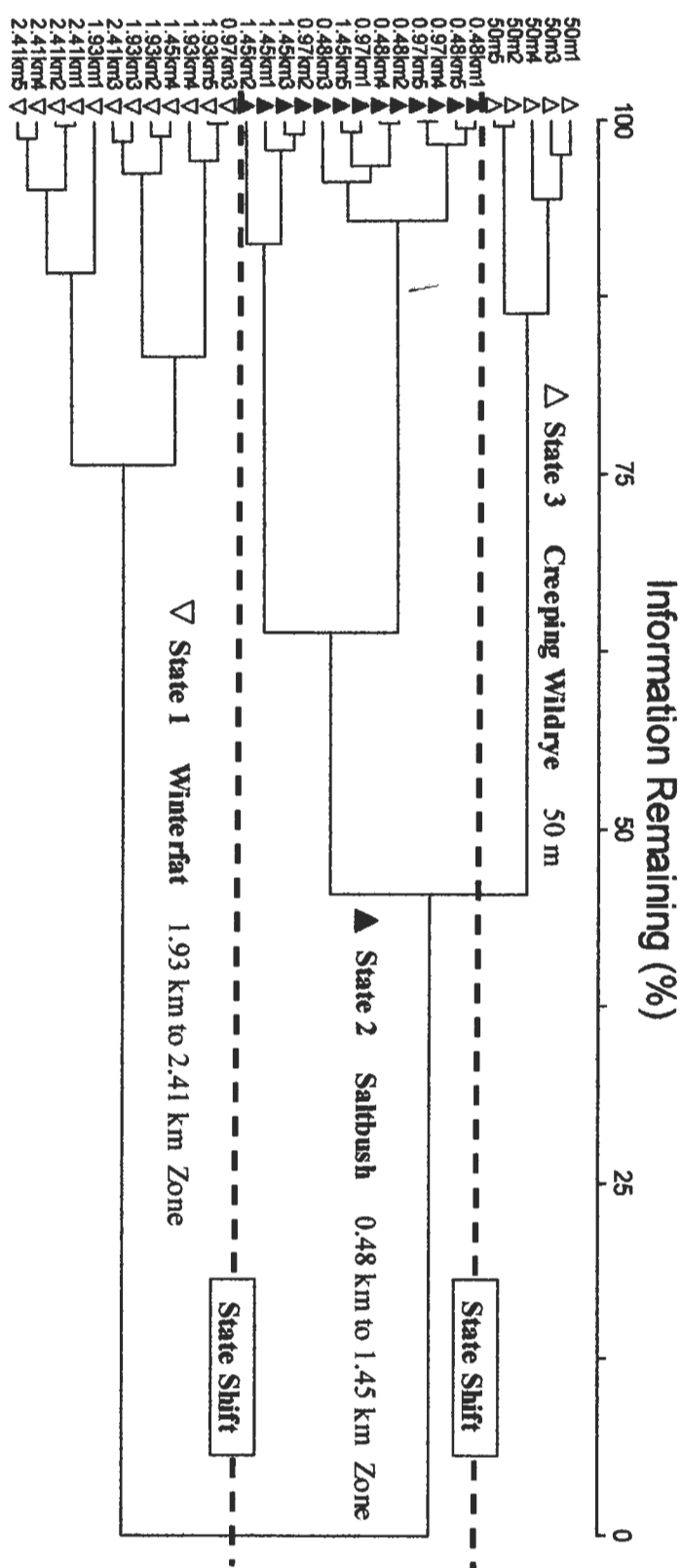


Figure 4.10. Cluster analysis dendrogram for piosphere sites in relation to distance from a cattle watering point on the Silty 6-10 PZ ecological site. Unit labels at the left margin depict each site's distance from the watering point and specific location.

Table 4.2. Mean, standard deviation, and standard errors for data collected from a piosphere study and a long-term grazing enclosure study on the Silty 6-10 PZ ecological site(s) located in the Catlow Valley, Oregon (NP = Not Present). Letter and numerator for state denotes group and state/phase (*ie.* A-1 = Piosphere State 1 and B-1.2 = Enclosure State 1, Phase 1.2)

Variable	Piosphere Study					Enclosure Study				
	State	Mean	Standard Deviation	Standard Error	<i>n</i>	State	Mean	Standard Deviation	Standard Error	<i>n</i>
Aggregate Stability Score	A-1	3.7	0.40	0.12	12	B-1.1	4.5	0.4	0.2	3
	A-2	3.0	0.34	0.10	13	B-1.2	4.5	0.3	0.1	9
	A-3	2.2	0.28	0.13	5	B-1.3	3.8	0.6	0.1	16
						B-2	3.0	0.5	0.2	7
Relative Infiltration Rate (cm·h ⁻¹)	A-1	2.9	0.98	0.28	12	B-1.1	2.0	0.7	0.4	3
	A-2	3.5	1.52	0.42	13	B-1.2	3.9	2.6	0.9	9
	A-3	3.3	0.50	0.22	5	B-1.3	3.4	1.5	0.4	16
						B-2	3.1	0.9	0.3	7
Organic Carbon of A Soil Horizon (%)	A-1	0.85	0.18	0.05	12	B-1.1	0.83	0.13	0.08	3
	A-2	0.73	0.12	0.03	13	B-1.2	0.92	0.41	0.14	9
	A-3	1.15	0.30	0.13	5	B-1.3	0.90	0.28	0.07	16
						B-2	0.77	0.20	0.07	7
A Soil Horizon Thickness (cm)	A-1	3.3	0.8	0.2	12	B-1.1	3.6	0.4	0.2	3
	A-2	3.5	0.6	0.2	13	B-1.2	3.9	0.7	0.2	9
	A-3	4.8	0.7	0.3	5	B-1.3	3.6	0.6	0.1	16
						B-2	3.3	0.8	0.3	7
Frequency of Creeping Wildrye (%)	A-1	8.1	28.1	8.1	12	B-1.1	NP	NP	NP	3
	A-2	30.1	40.4	11.2	13	B-1.2	13.3	29.6	9.9	9
	A-3	98.1	2.0	0.9	5	B-1.3	27.4	37.4	9.3	16
						B-2	32.4	44.3	16.8	7
Bareground (%)	A-1	46.3	7.6	2.2	12	B-1.1	29.6	23.1	13.4	3
	A-2	64.2	7.7	2.1	13	B-1.2	6.8	6.7	2.2	9
	A-3	47.6	9.6	4.3	5	B-1.3	37.1	5.8	1.4	16
						B-2	62.5	3.0	1.1	7
Biological Soil Crust (%)	A-1	20.4	8.7	2.5	12	B-1.1	47.3	20.1	11.6	3
	A-2	4.5	5.3	1.5	13	B-1.2	26.9	8.7	2.9	9
	A-3	NP	NP	NP	5	B-1.3	12.1	7.7	1.9	16
						B-2	1.8	1.6	0.6	7
Litter (%)	A-1	41.3	7.9	2.3	12	B-1.1	11.7	5.0	2.9	3
	A-2	43.5	8.6	2.4	13	B-1.2	50.4	12.4	4.1	9
	A-3	49.0	11.5	5.2	5	B-1.3	39.5	8.4	2.1	16
						B-2	27.2	3.2	1.2	7
0 to 5 cm - No. of Penetrometer Strikes	A-1	2.3	0.3	0.1	12	B-1.1	5.4	1.1	0.6	3
	A-2	2.7	0.6	0.2	13	B-1.2	2.3	0.9	0.3	9
	A-3	5.1	1.5	0.7	5	B-1.3	2.6	1.0	0.2	16
						B-2	3.0	0.9	0.3	7
5 to 10 cm - No. of Penetrometer Strikes	A-1	3.1	0.9	0.3	12	B-1.1	12.0	1.1	0.6	3
	A-2	4.5	1.2	0.3	13	B-1.2	6.5	2.6	0.9	9
	A-3	11.1	3.5	1.6	5	B-1.3	8.1	2.3	0.6	16
						B-2	10.2	4.0	1.5	7

Table 4.3. Mean, standard deviation, and standard errors for data collected from a piosphere study and a long-term grazing enclosure study on the Silty 6-10 PZ ecological site(s) located in the Catlow Valley, Oregon (NP = Not Present). Letter and numerator for state denotes group and state/phase (*ie.* A-1 = Piosphere State 1 and B-1.2 = Enclosure State 1, Phase 1.2)

Variable	Piosphere Study					Enclosure Study				
	State	Mean	Standard Deviation	Standard Error	<i>n</i>	State	Mean	Standard Deviation	Standard Error	<i>n</i>
10 to 15 cm - No. of Penetrometer Strikes	A-1	5.0	1.0	0.3	12	B-1.1	11.7	0.7	0.4	3
	A-2	6.9	1.7	0.5	13	B-1.2	9.8	3.0	1.0	9
	A-3	13.1	1.9	0.8	5	B-1.3	11.4	2.6	0.6	16
						B-2	13.4	3.6	1.3	7
Bare Patch Size (m)	A-1	1.670	0.418	0.120	12	B-1.1	1.179	0.837	0.483	3
	A-2	1.294	0.533	0.147	13	B-1.2	0.900	0.179	0.059	9
	A-3	0.573	0.056	0.025	5	B-1.3	1.681	0.836	0.209	16
						B-2	2.622	1.703	0.643	7
Size of Perennial Plants (cm)	A-1	7.5	1.5	0.4	12	B-1.1	11.5	3.6	2.0	3
	A-2	7.1	2.2	0.6	13	B-1.2	9.6	3.1	1.0	9
	A-3	4.8	0.8	0.3	5	B-1.3	7.8	2.8	0.7	16
						B-2	6.1	1.3	0.4	7
Frequency of Sandberg Bluegrass (%)	A-1	18.6	21.9	6.3	12	B-1.1	1.1	1.9	1.1	3
	A-2	10.3	12.0	3.3	13	B-1.2	7.5	13.0	4.3	9
	A-3	1.0	1.3	0.6	5	B-1.3	2.5	2.8	0.7	16
						B-2	3.3	5.3	2.0	7
Frequency of Goosberry Globemallow (%)	A-1	3.9	6.4	1.8	12	B-1.1	NP	NP	NP	3
	A-2	1.5	4.6	1.3	13	B-1.2	1.3	2.7	0.9	9
	A-3	1.0	1.3	0.6	5	B-1.3	0.2	0.8	0.2	16
						B-2	NP	NP	NP	7
Frequency of Indian Ricegrass (%)	A-1	3.8	6.1	1.8	12	B-1.1	12.2	10.7	6.2	3
	A-2	NP	NP	NP	13	B-1.2	8.5	21.0	7.0	9
	A-3	NP	NP	NP	5	B-1.3	2.0	5.0	1.2	16
						B-2	NP	NP	NP	7
Density of Sickle Saltbush (plants·m ⁻²)	A-1	3.78	0.85	0.24	12	B-1.1	NP	NP	NP	3
	A-2	4.17	1.62	0.45	13	B-1.2	4.26	3.08	1.03	9
	A-3	3.24	0.77	0.34	5	B-1.3	2.57	3.13	0.78	16
						B-2	5.72	8.88	3.36	7
Density of Winterfat (plants·m ⁻²)	A-1	0.47	0.45	0.13	12	B-1.1	5.31	0.84	0.49	3
	A-2	0.29	0.25	0.07	13	B-1.2	1.16	0.90	0.30	9
	A-3	NP	NP	NP	5	B-1.3	2.29	1.76	0.44	16
						B-2	1.63	2.27	0.86	7
Density of Annuals (plants·m ⁻²)	A-1	2.09	1.59	0.46	12	B-1.1	0.04	0.08	0.04	3
	A-2	3.52	2.39	0.66	13	B-1.2	13.12	14.97	4.99	9
	A-3	3.11	1.87	0.84	5	B-1.3	3.12	5.85	1.46	16
						B-2	0.86	1.14	0.43	7

Model Refinement

Clusters of groups for both the piosphere study and the grazing enclosure study were charted in two-dimensional charts (Figs. 4.12 and 4.13) in order to view the

similarities among groups for these two studies. When viewing mean values and standard errors of groups simultaneously for percent biological soil crust and soil surface aggregate stability score it was apparent that states independently identified from the piosphere study and the grazing exclosure were very similar in range. State 1 and State 2 membership for groups are depicted in the chart with dashed ovals. State 1 occupied a range of biological soil crust $> 10\%$ and a range of soil surface aggregate stability score > 3.5 . State 2 occupied a mean range of biological soil crust of approximately 2-8% and mean score of soil surface aggregate stability that was approximately 3. Lastly, State 3 had no biological soil crust and a mean aggregate stability score of less than 2.5 (Fig. 4.12).

Means and standard errors for frequency of Indian ricegrass and density of winterfat shrubs independently identified from the piosphere study and the grazing exclosure were also very similar in range (Fig. 4.11). State 1 membership for groups are depicted in the chart with a dashed oval. State 1 occupied a range of frequency of Indian ricegrass approximately $> 2\%$ and a density of winterfat shrubs of 0.5 to 6 plants $\cdot\text{m}^{-2}$. State 2 was void of Indian ricegrass and had a mean density of winterfat shrubs of approximately 0.25 to 2 plants $\cdot\text{m}^{-2}$. Lastly, State 3 had no Indian ricegrass and no winterfat plants (Fig. 4.13). In contrast, the remaining 15 variables quantified for each state and group for the piosphere and grazing exclosure study were quite variable.

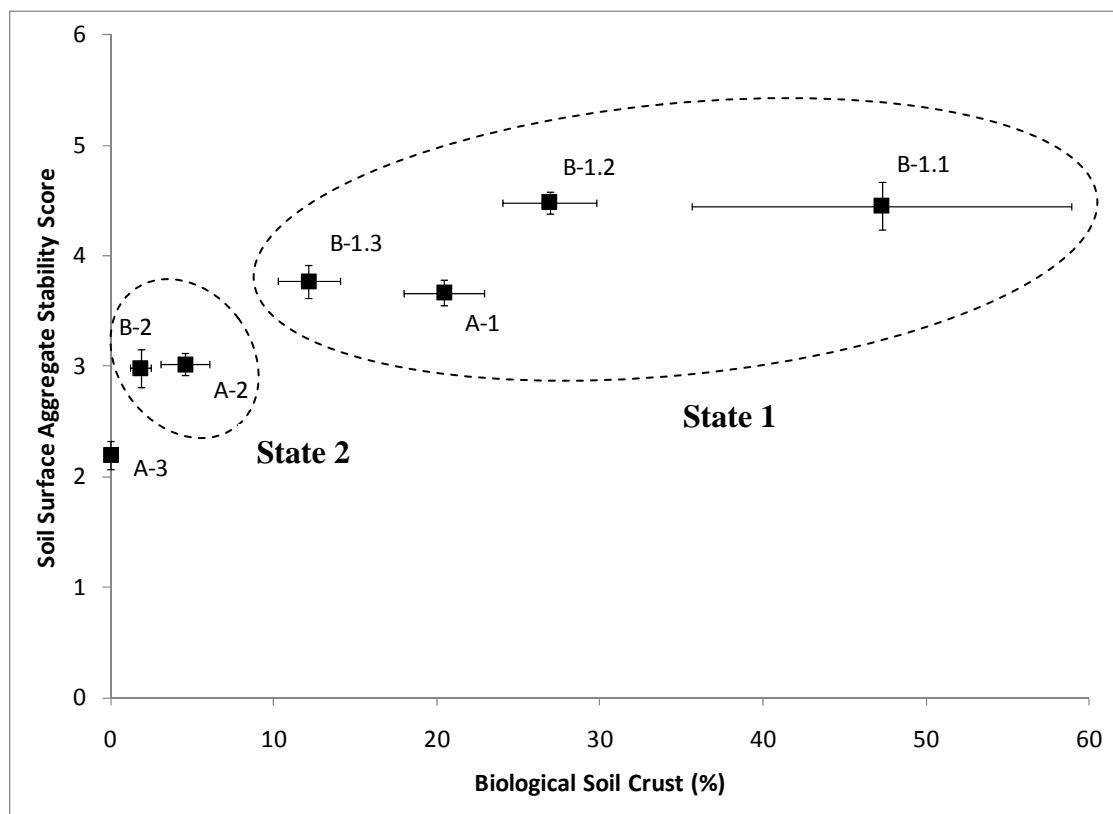


Figure 4.12. Mean percent biological soil crust and soil surface aggregate stability score with SE bars for groups from a piosphere study (Group A) and a long-term grazing exclosure study (Group B) on the Silty 6-10 PZ ecological site(s) located in the Catlow Valley, Oregon. Dashed line ovals indicate classification to different states.

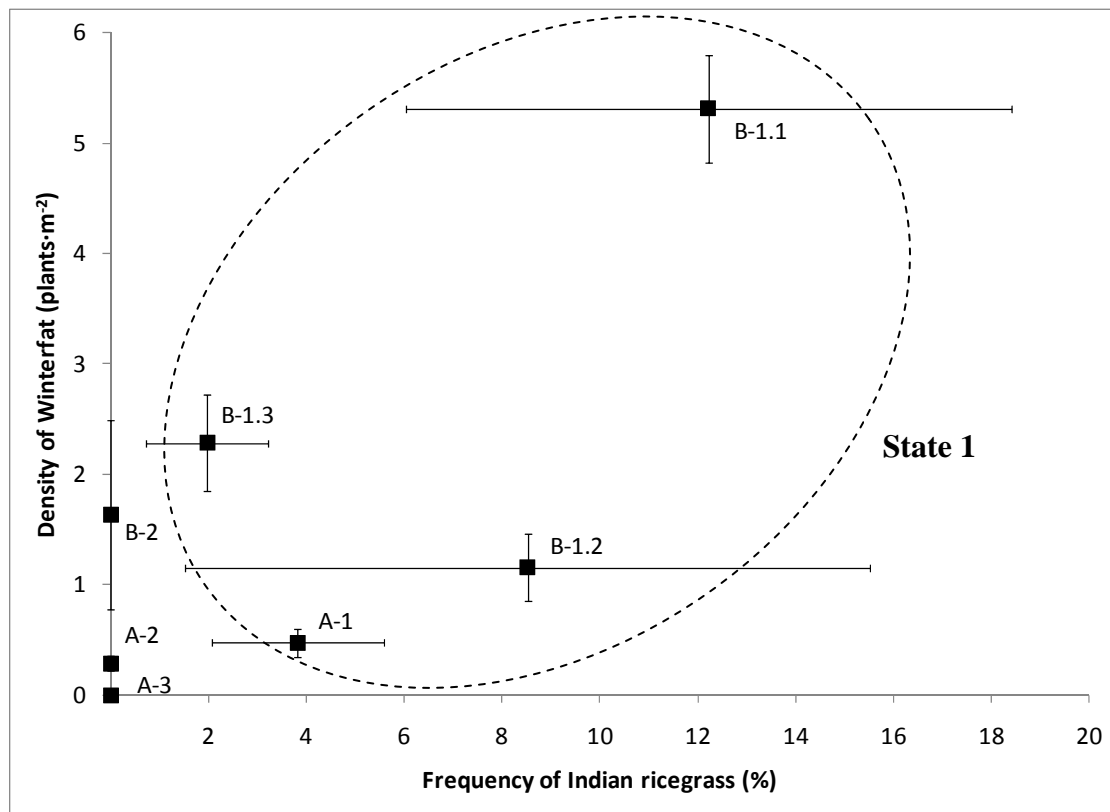


Figure 4.13. Mean frequency of Indian ricegrass and density of winterfat shrubs with SE bars for data collected from a piosphere study (Group A) and a long-term grazing exclosure study (Group B) on the Silty 6-10 PZ ecological site(s) located in the Catlow Valley, Oregon. Dashed line ovals indicate classification to different states.

DISCUSSION

Hierarchical agglomerative cluster analysis found three ecologically different states across two different studies (piosphere study and grazing exclosure study). Four variables of 19 used were found to be excellent predictors of group membership into states: soil surface aggregate stability, percent biological soil crust, frequency of Indian ricegrass, and density of winterfat. These four variables provided good measures of ecological site resilience and characterization of states. These data supported the proposed three state STM based on ecological processes. The four variables used to describe the three STM states are presented with 95% confidence intervals for each state (Table 4.4).

Table 4.4 Confidence intervals (95%) for the three state STM for the Silty 6-10 PZ ecological site.

Variable	Winterfat State 1	Saltbush State 2	Wildrye State 3
Aggregate Stability Score	> 3.8	3.2 - 2.8	< 2.6
Biological Soil Crust (%)	> 16	6 - 1	NP
Density of Winterfat (plants·m ⁻²)	> 1.1	1.4 - 0.1	NP
Frequency of Indian Ricegrass (%)	> 1	NP	NP
Frequency of Sandberg Bluegrass (%)	> 4	13 - 2	< 3
Frequency of Goosberry Globemallow (%)	> 1	< 3	< 3
Density of Sickie Saltbush (plants·m ⁻²)	2.3 - 4.0	2.3 - 7.2	2.3 - 4.2
Density of Annuals (plants·m ⁻²)	2.0 - 7.7	1.5 - 3.7	0.8 - 5.4
Frequency of Creeping Wildrye (%)	6 - 27	12 - 50	> 95
0 to 5 cm - No. of Penetrometer Strikes	2.3 - 3.0	2.5 - 3.1	3.2 - 7.0
5 to 10 cm - No. of Penetrometer Strikes	5.4 - 7.5	4.7 - 8.2	6.7 - 15.4
10 to 15 cm - No. of Penetrometer Strikes	8.0 - 10.3	7.3 - 11.0	10.8 - 15.4
Bare Patch Size (m)	1.2 - 1.7	1.2 - 2.3	0.5 - 0.6
Size of Perennial Plants (cm)	> 7.5	7.7 - 5.8	< 5.9
Bareground (%)	27 - 38	61 - 67	36 - 59
Litter (%)	36 - 45	33 - 43	35 - 63
Organic Carbon of A Soil Horizon (%)	1.0 - 0.8	0.8 - 0.7	1.5 - 0.8
A Soil Horizon Thickness (cm)	3.8 - 3.4	3.8 - 3.1	5.6 - 4.0
Relative Infiltration Rate (cm·h ⁻¹)	2.7 - 3.8	2.7 - 4.0	2.7 - 3.9

The range of values for the 19 variables used; especially biological soil crust, soil surface aggregate stability, density of winterfat, and frequency of Indian ricegrass; allowed for the identification of plant community phases within State 1 of the STM. The range of values for State 1 as demonstrated in Figs. 4.12 and 4.13 showed that groups B-1.1 and B-1.2 of the grazing exclosure data set were located farthest away from themselves and the other groups. In contrast, groups A-1 and B-1.3 occupied a very similar range of values within the data dimensions of State 1. This observation led to the formation of three plant community phases within State 1: 1.1 (group B-1.1), 1.2 (group B-1.2), and 1.3 (groups A-1 and B-1.3). Confidence intervals for these

phases were determined (Table 4.5). Separation between groups showed very little overlap for biological soil crust, soil surface aggregate stability, winterfat density, and frequency of Indian ricegrass. Shaver (2010) also found that aggregate stability was a good predictor of group membership for sites within STM states and community phases. Phase 1.1 was comprised of a small sample size ($n = 3$), and standard errors used to quantify its variability in the 19 data variables, instead of 95% confidence intervals. The remaining groups for States 2 and 3 were investigated for potential demarcation of plant community phases. However, due to the coarseness and similarity of the data, it did not warrant separation of sites in these states into different plant community phases. The 95% confidence intervals for plant community phases and states is displayed in Table 4.5. Examination of State 1 indicated that community phase 1.3 was an at-risk community phase within State 1 and that it is approaching an ecological threshold that could lead to State 2. A chart of the refined STM for states and phases of the Silty 6-10 PZ ecological site are displayed in Figure 4.14.

Table 4.5 Confidence intervals (95%) for the three state STM with plant community phases included for the Silty 6-10 PZ ecological site. Standard errors are displayed for phase 1.1 instead of the 95% confidence interval due to a small sample size (n). NP = Not Present and T = Trace.

Variable	Winterfat State Phase 1.1	Winterfat State Phase 1.2	Winterfat State Phase 1.3	Saltbush State 2	Wildrye State 3
Aggregate Stability Score	4.7 - 4.3	4.7 - 4.3	3.9 - 3.5	3.2 - 2.8	< 2.6
Biological Soil Crust (%)	35 - 59	34 - 20	19 - 12	6 - 1	NP
Density of Winterfat (plants·m ⁻²)	4.8 - 5.8	1.9 - 0.5	2.1 - 0.9	1.4 - 0.1	NP
Frequency of Indian Ricegrass (%)	18 - 6	< 25	5 - 1	NP	NP
Frequency of Sandberg Bluegrass (%)	T	< 17	16 - 3	13 - 2	< 3
Frequency of Goosberry Globemallow (%)	NP	< 3	< 4	< 3	< 3
Density of Sickle Saltbush (plants·m ⁻²)	NP	1.9 - 6.6	2.1 - 4.0	2.3 - 7.2	2.3 - 4.2
Density of Annuals (plants·m ⁻²)	T	1.6 - 25.6	0.9 - 4.4	1.5 - 3.7	0.8 - 5.4
Frequency of Creeping Wildrye (%)	NP	< 36	6 - 32	12 - 50	> 95
0 to 5 cm - No. of Penetrometer Strikes	4.8 - 6.0	1.6 - 2.9	2.2 - 2.8	2.5 - 3.1	3.2 - 7.0
5 to 10 cm - No. of Penetrometer Strikes	11.4 - 12.6	4.4 - 8.5	4.7 - 7.1	4.7 - 8.2	6.7 - 15.4
10 to 15 cm - No. of Penetrometer Strikes	11.3 - 12.1	7.5 - 12.1	7.2 - 10.2	7.3 - 11.0	10.8 - 15.4
Bare Patch Size (m)	0.7 - 1.7	0.8 - 1.0	1.4 - 1.9	1.2 - 2.3	0.5 - 0.6
Size of Perennial Plants (cm)	14 - 10	12.1 - 7.2	8.6 - 6.8	7.7 - 5.8	< 5.9
Bareground (%)	16 - 43	2 - 12	38 - 44	61 - 67	36 - 59
Litter (%)	9 - 15	41 - 60	37 - 44	33 - 43	35 - 63
Organic Carbon of A Soil Horizon (%)	0.7 - 0.9	0.6 - 1.2	0.8 - 1.0	0.8 - 0.7	1.5 - 0.8
A Soil Horizon Thickness (cm)	3.8 - 3.4	4.5 - 3.4	3.7 - 3.2	3.8 - 3.1	5.6 - 4.0
Relative Infiltration Rate (cm·h ⁻¹)	1.6 - 2.4	1.9 - 5.9	2.7 - 3.7	2.7 - 4.0	2.7 - 3.9

Plant community phases 1.0 through 1.3 depict the feedback mechanisms that maintain the resilience of the winterfat reference state (State 1). With continued grazing of plant community phase 1.3; Indian ricegrass, biological soil crust, and winterfat declined; which reduced the negative feedback mechanisms that maintained site resilience. Over time, this reduced plant production and organic matter inputs resulted in diminished soil surface aggregate stability and site resilience, which in turn promoted positive feedbacks that furthered a loss of resources and site degradation. Due to these dynamics, phase 1.3 is considered the at-risk plant community phase in State 1. Biological soil crust for phase 1.3 was 12 - 19%, which is the low end of the range for State 1. Soil surface aggregate stability score (3.5 – 3.9) was also at the low end of the range. Density of winterfat stayed within a tolerable range (0.9 to 2.1 plants·m⁻²), however frequency of Indian ricegrass (1 - 5%) indicated that this species was at-risk of being eliminated from the site, indicating that this phase may indeed be crossing an ecological threshold and shifting towards a sickle saltbush dominated state (State 2). The ecological threshold (T1a) from the Winterfat Reference State to the Sickle Saltbush State is biotic (Whisenant 1999) and indicated by a loss of Indian ricegrass from the plant community and also by declines in soil surface aggregate stability score and percent biological soil crust that fall below 3.5 and 12%, respectively. Crossing of the ecological threshold causes the positive feedbacks on the site to now become negative feedbacks that strengthen the resilience of the Sickle Saltbush State (State 2).

The Sickle Saltbush State was characterized as having no Indian ricegrass and being dominated by sickle saltbush. Winterfat was still present but occurred in densities of 1.4 to 0.3 plants·m⁻². Biological soil crust ranged from 1 to 6% and soil surface aggregate stability score ranged from 2.8 to 3.2. Data to determine plant community phases within the Sickle Saltbush State were lacking. Our observations indicate that sickle saltbush increases with moderate to heavy grazing while winterfat declines. Winterfat is the preferred forage plant, and it receives greater browse-use by livestock and wildlife than sickle saltbush. Sickle saltbush vigorously resprouts from the crown as well as laterally spreading roots. It also reproduces well by seeds.

Winterfat resprouts from the crown only and has had poor seedling success over the last 20 years. Continued grazing in the Sickie Saltbush State may allow the Sickie Saltbush State to persist, maintaining limited biological soil crust, limited winterfat shrub density, and limited plant cover to maintain organic inputs at levels high enough to keep soil surface aggregate stability at around a score of 3. However, under heavy grazing, winterfat is removed from the system and biological soil crust is lost.

Remaining sickie saltbush plants and bottlebrush squirreltail decrease in size and cover. This yields lower organic matter inputs and decreased soil surface aggregate stability. This in turn leaves the soil susceptible to wind and water erosion which are the feedback mechanisms that build resilience in the Creeping Wildrye State (State 3). Matney (2010b) found that this state occurred under only the most extreme grazing disturbance around cattle watering points. Indication of crossing an ecological threshold from State 2 to State 3 is found when winterfat and biological soil crust are removed from the plant community and soil surface aggregate stability declines to 2.6 or less. This threshold from the Sickie Saltbush State to the Creeping Wildrye State is both biotic (loss of winterfat and biological soil crust) and abiotic (physical processes of erosion become dominant forces). Conditions of the Creeping Wildrye State are characterized as having little vegetative cover and much bareground. The plant community is dominated by creeping wildrye, small sickie saltbush plants, and small bottlebrush squirreltail plants. In some areas, creeping wildrye may be subdominant to bottlebrush squirreltail. Soil surface aggregate stability is low and disturbed soil surface conditions are evident from both cattle hoof impact and the forces of wind/water erosion.

CONCLUSIONS

A three state model for the Silty 6-10 PZ ecological site based on the ecological site description, historical information, experience, and scientific knowledge was developed and tested. State characteristics, at risk community phases, thresholds, resilience feedback mechanisms, and transitions were characterized. Hierarchical agglomerative cluster analysis was used to partition sites into groups of

similar ecological states and phases. Nineteen variables were used to partition sites among three states. The 19 variables used were presence or absence (P/A) of creeping wildrye; P/A of Sandberg bluegrass; P/A of goosberry globemallow; P/A of Indian ricegrass; mean density of sickle saltbush; mean density of winterfat; mean density of annual plants; mean aggregate stability; relative infiltration rate; percent organic carbon of the A1 soil horizon; thickness of the A1 soil horizon; percent bareground; percent biological soil crust; percent litter; impact penetrometer data for the soil depth zones of 0 to 5 cm, 5 to 10 cm, and 10 to 15 cm; mean bare patch size between perennial plants; and mean basal size of perennial plants. The four strongest factors indicating differences between states were amount of biological soil crust, soil surface aggregate stability, density of winterfat, and frequency of Indian ricegrass. Models were based on a piosphere study and a longterm grazing exclosure study. Data from this study supported the initial proposed STM for the Silty 6-10 PZ ecological site. States defined were a Reference Winterfat State (State 1), Sickle Saltbush State (State 2), and a Creeping Wildrye State (State 3). Each state corresponds to a site characterized by high, moderate, and low ecological functioning, respectively.

Our data suggest that periodic rest from winter grazing may be needed to maintain the resilience of the ecological site when in State 1 to retain the native bunchgrass Indian ricegrass and keep levels of biological soil crust high. Periodic rest of a year or more would allow bunchgrasses and winterfat to make gains in size as well as allow soils to make gains in biological soil crust. Additionally, shifting the timing of grazing from October through the end of December may decrease the negative effects on soil compaction and winterfat plants, since soils are drier during this time period and winterfat plants have completed their annual reproductive cycle and have not yet initiated spring growth. However, with increases in grazing, Indian ricegrass is likely to be lost, biological soil crust will decline, and soil surface aggregate stability will decline which will cause an ecological threshold to be crossed causing a shift to State 2. Areas that are heavily grazed for long periods of time or areas very close to watering points and salt licks may be disturbed enough to cause a shift to State 3. Overall, soil surface aggregate stability, biological soil crust, density

of winterfat, and frequency of Indian ricegrass were all reliable indicators of state status within the three state STM. The relationship between biological soil crust and aggregate stability were strong and offer the opportunity as a readily identifiable indicator of ecological site status within the STM. Land managers could evaluate these two variables with relative ease in order to determine ecological site status in order to make management decisions. Having this information would allow managers to develop a strong understanding of site resilience and increase their ability to manage for a higher functioning state and more desirable plant community.

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CHAPTER 5: GENERAL CONCLUSION

State-and-transition models (STM) have been successfully used to describe woodlands, shrublands, grasslands, and several other ecological systems. Changes in vegetation and soil are measured to gauge and predict transitions between alternative states and phases within states. For each state, there is a range of variability in vegetation and soil parameters that define it. In the United States each ecological site has been assigned its own unique state and transition model, the foundation of which is based on estimation and quantification of a historical reference plant community. Historical information, land manager experience, and scientific data are used to quantify the historical reference plant community and gauge changes towards alternative stable states. In regards to winterfat plant communities and the Silty 6-10 PZ ecological site, state and transition models have not yet been tested, especially in the Catlow Valley of Oregon. The objective of this study was to develop a process-based state and transition model for the Silty 6-10 PZ ecological site. The predominant factor of disturbance on this ecological site for the last 100 years has been grazing by livestock. To develop the process-based STM for the ecological site, we conducted a five-year grazing exclosure study to gauge the ecological resilience of the site as well as a piosphere study. With these two studies we were able to quantify the resilience of the ecological site when protected from disturbance (exclosure study) and also to quantify the trends of the ecological site when under continued grazing disturbance (piosphere study). We used the information gathered from both of these studies to develop a STM for the Silty 6-10 PZ ecological site.

Our study suggests that continued use of winterfat communities by cattle and wildlife during the fall/winter period in the Catlow Valley negatively affected perennial plant size, soil compaction, and biological soil crust over a 5 year period from year 2002 to 2007. Over the same period, only perennial plant size seemed to benefit from protection from cattle and wildlife, and this suggests that factors such as plant community dynamics and soil properties are slow to respond when released from herbivory. We did find higher aggregate stability and the potential for higher

infiltration rates in areas protected from cattle, and this might suggest that protection from grazing may maintain higher resource retention and hydrologic function. Areas protected from cattle-use offer encouragement that under no or light herbivory by wildlife winterfat plants in the Catlow Valley are resilient enough to double in size and total perennial vegetative cover within 5 years. Also, protecting winterfat communities where cheatgrass is present may increase the risk of cheatgrass expansion. Over time, this could reduce winterfat establishment and increase the risk of wildfire. Increases in winterfat shrub size and overall amounts of perennial vegetation may make the habitat more suitable for jackrabbits as both a source of food and cover. Decreases in winterfat plant size, perennial vegetative cover, and biological soil crust along with increases in bare patch size and bareground over the five years of this study signify that current levels of fall/winter cattle-use are not likely to be sustainable over the long-run. Low aggregate stability and increased bareground in unprotected areas, along with seasonally strong winds, may increase the risk for wind erosion. Continued reductions in winterfat plant size and perennial vegetative cover may increase the risk of extirpation of winterfat and reduce their resilience if they were to be protected from cattle-use in the future.

Overall winterfat communities in the Catlow Valley demonstrated low to moderate resistance to cattle-use. Further they demonstrated low resilience when protected from cattle for 5 years. Except, individual winterfat plants demonstrated high resilience by persisting in unprotected areas. Individual winterfat plants also demonstrated high resilience by making considerable gains in size when protected from cattle. In addition to studying winterfat response to protection from herbivory we investigated the effects of distance from watering point on winterfat and soil.

We found that there was a clear cattle-use disturbance gradient surrounding the watering point. Many plant, soil, and ecological attributes demonstrated a relationship with distance from the watering point. The nature of this relationship was such that areas closer to the watering point were heavily trampled and foraged and areas farther from the watering point were less affected. Zone of influence from cattle use was apparent all the way to 2.41 km from the watering point. The area closest to the

watering point, 50 m, was characterized by soil compaction, little vegetation, and accumulations of litter and soil nutrients. Areas furthest from the watering point favored larger native perennial plants, higher diversity of native perennial plants, higher aggregate stability, and abundances of biological soil crust. These observations are of importance to land managers who want to maintain ecological function around cattle watering points and to minimize risks to native plants.

When grazing areas around watering points it can be assumed that areas closer than 0.5 km are going to be greatly disturbed and it will be quite difficult to maintain native species there, especially winterfat and Indian ricegrass. Land managers can assume that compaction will be at its highest in areas that are within 0.5 km from the watering point and that soil surface degradation will occur all the way out to 2.41 km from the watering point. Soil biological crust is an important component of this ecosystem, and that distances closer than 1 km will not be able to support biological soil crust. The presence of soil biological crust and levels of soil surface aggregate stability are indicators of areas that receive less disturbance from cattle use. Distance from water is a key factor to consider when planning grazing in the Catlow Valley.

A three state model for the Silty 6-10 PZ ecological site based on the ecological site description, historical information, experience, and scientific knowledge was developed and tested. State characteristics, at risk community phases, thresholds, resilience feedback mechanisms, and transitions were characterized. Hierarchical agglomerative cluster analysis was used to partition sites into groups of similar ecological states and phases. Nineteen variables (Table 5.1) were used to partition sites among three states (Figure 5.1). The four strongest factors indicating differences between states were amount of biological soil crust, soil surface aggregate stability, density of winterfat, and frequency of Indian ricegrass. Models were based on a piosphere study and a longterm grazing exclosure study. Data from this study supported the initial proposed STM for the Silty 6-10 PZ ecological site. States defined were a Reference Winterfat State (State 1), Sickie Saltbush State (State 2), and a Creeping Wildrye State (State 3). Each state corresponds to a site characterized by high, moderate, and low ecological functioning, respectively.

Table 5.1. Nineteen variables included in the hierarchical cluster analysis for determining differences between sites in the Catlow Valley, OR.

Variables Included in Hierarchical Cluster Analysis
Presence/Absence of Creeping Wildrye
Presence/Absence of Sandberg Bluegrass
Presence/Absence of Gooseberry Globemallow
Presence/Absence of Indian Ricegrass
Mean Density of Sickle Saltbush
Mean Density of Winterfat
Mean Density of Annual Plants
Mean Soil Surface Aggregate Stability
Mean Infiltration Rate
Percent Organic Carbon of the A Soil Horizon
Thickness of the A Soil Horizon
Percent Bareground
Percent Biological Soil Crust
Percent Litter
Soil Penetrometer Resistance from 0 to 5 cm in Soil Depth
Soil Penetrometer Resistance from 5 to 10 cm in Soil Depth
Soil Penetrometer Resistance from 10 to 15 cm in Soil Depth
Mean Bare Patch Size between Perennial Plants
Mean Perennial Plant Size (Canopy and Basal)

Periodic rest from winter grazing may be needed to maintain the resilience of the ecological site when in State 1 to retain the native bunchgrass Indian ricegrass and keep levels of biological soil crust high. Periodic rest of a year or more would allow bunchgrasses and winterfat to make gains in size as well as allow soils to make gains in biological soil crust. Additionally, shifting the timing of grazing from October through the end of December may decrease the negative effects on soil compaction and winterfat plants, since soils are drier during this time period and winterfat plants have completed their annual reproductive cycle and have not yet initiated spring growth. However, with increases in grazing, Indian ricegrass is likely to be lost, biological soil crust will decline, and soil surface aggregate stability will decline which will cause an ecological threshold to be crossed causing a shift to State 2. Areas that are heavily grazed for long periods of time or areas very close to watering points and salt licks may be disturbed enough to cause a shift to State 3. Overall, soil

surface aggregate stability, biological soil crust, density of winterfat, and frequency of Indian ricegrass were all reliable indicators of state status within the three state STM. The relationship between biological soil crust and aggregate stability were strong and offer the opportunity as a readily indentifiable indicator of ecological site status within the STM. Land managers could evaluate these two variables with relative ease in order to determine ecological site status in order to make management decisions. Having this information would allow managers to develop a strong understanding of site resilience and increase their ability to manage for a higher functioning state and more desirable plant community.

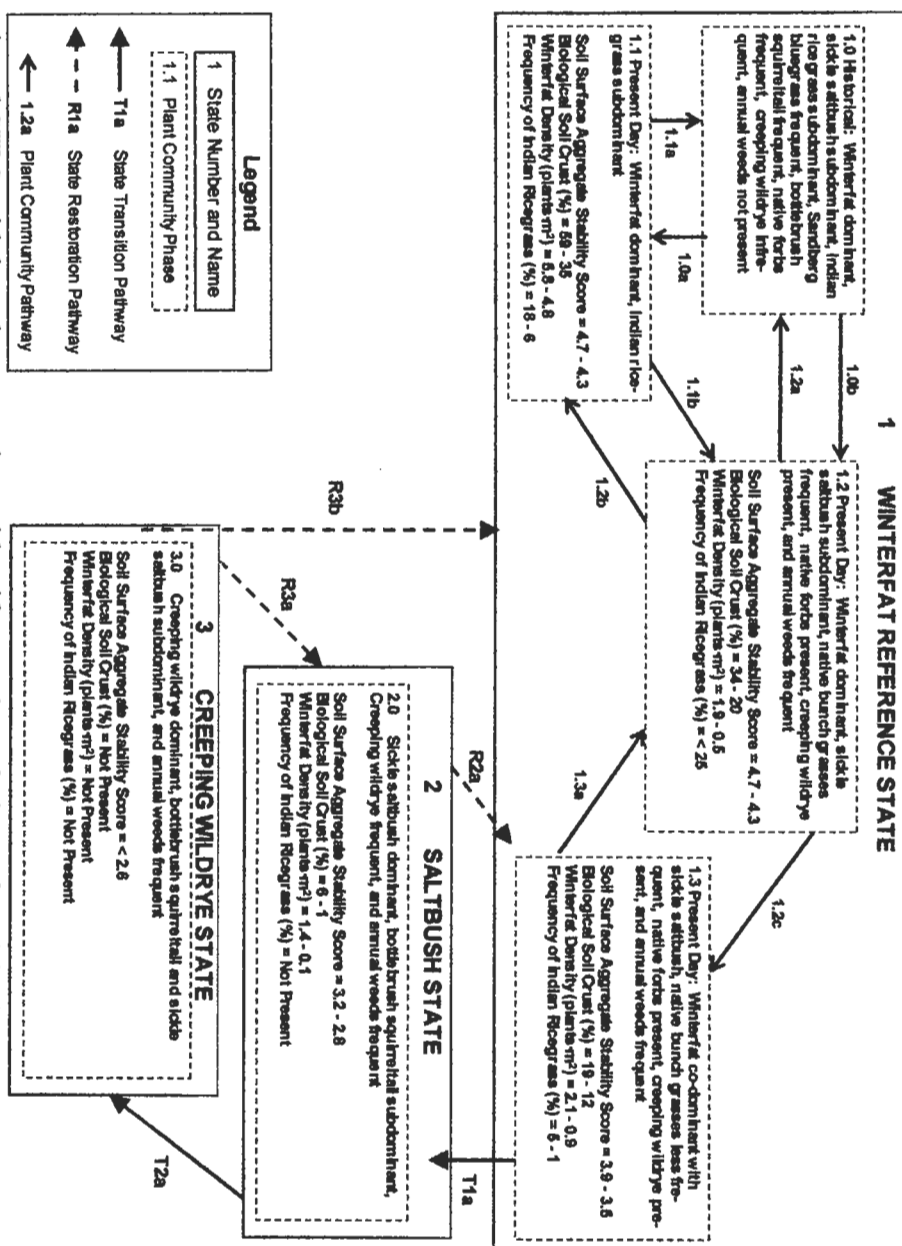


Figure 5.1 Developed STM, with key characteristics to identify state boundaries and at-risk plant community phases, for the Silty 6-10 PZ ecological site.

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APPENDIX

Spangenburg Soil Series

The Spangenburg series consists of very deep, well drained and moderately well drained soils that formed in lacustrine sediment. The soils are on lake terraces and lake plains. Slope is 0 to 2 percent. Elevation is 4,300 to 5,300 feet. The mean annual precipitation is 8 to 10 inches, and the mean annual air temperature is 45 to 49 degrees F.

Taxonomic classification: Fine, montmorillonitic, mesic Xeric Paleargids
 Typical pedon of Spangenburg silty clay loam, 0 to 1 percent slopes, about 300 feet south and 1,400 feet west of the northeast corner of sec. 26, T. 34 S., R. 31 E.; Blitzen quadrangle.

A—0 to 2 inches; light brownish gray (10YR 6/2) silty clay loam, very dark grayish brown (10YR 3/2) moist; weak medium platy structure; slightly hard, very friable, very sticky and moderately plastic; many fine roots; many fine and medium vesicular pores; neutral (pH 7.2); abrupt wavy boundary.

2Bt1—2 to 6 inches; pale brown (10YR 6/3) silty clay, brown (10YR 4/3) moist; moderate medium prismatic structure parting to strong medium and fine subangular blocky; hard, firm, very sticky and moderately plastic; many fine roots; many very fine tubular pores; common faint clay films lining pores; slightly alkaline (pH 7.4); clear wavy boundary.

3Bt2—6 to 15 inches; yellowish brown (10YR 5/4) silty clay, dark yellowish brown (10YR 4/4) moist; moderate fine angular blocky structure; slightly hard, friable, very sticky and moderately plastic; many fine roots; many very fine tubular pores; few faint clay films lining pores; moderately alkaline (pH 8.0); clear wavy boundary.

3Btk—15 to 34 inches; pale brown (10YR 6/3) silty clay loam, brown (10YR 4/3) moist; massive; slightly hard, friable, slightly sticky and moderately plastic; common fine roots; many very fine tubular pores; few faint clay films lining pores; 20 percent disseminated white ash; slightly effervescent; moderately alkaline (pH 8.0); abrupt wavy boundary.

4C—34 to 60 inches; very pale brown (10YR 7/3) loam, brown (10YR 4/3) moist; massive; slightly hard, friable, nonsticky and nonplastic; few fine roots; many very fine interstitial pores; laminar platy lacustrine sediment; moderately alkaline (pH 8.0).

Bedrock is at a depth of more than 60 inches. The solum is 10 to 30 inches thick. The content of clay in the 2Bt horizon is 15 to 25 percent more than that of the A horizon. Occasional ponding occurs in spring in some years. A high water table is

present late in winter and in spring in some years.

The A horizon is 2 to 10 inches thick. It has value of 5 to 7 dry and 3 or 4 moist, and it has chroma of 1 to 3 dry and 2 moist. It is neutral or slightly alkaline. It is silty clay loam or silt loam.

The Bt horizon has value of 5 or 6 dry and 3 or 4 moist, and it has chroma of 3 or 4 dry and 2 to 4 moist. The upper part is clay or silty clay with 45 to 60 percent clay.

The lower part is silty clay loam or silt loam with 25 to 40 percent clay. The horizon is slightly alkaline or moderately alkaline.

The C horizon has value of 6 or 7 dry and 3 to 5 moist, and it has chroma of 3 or 4 dry or moist. It is stratified loam, silt loam, or silty clay loam. It is slightly alkaline or moderately alkaline.