FINAL REPORT:
COMPARING EFFECTS OF MANAGEMENT PRACTICES ON RANGELAND HEALTH WITH GEOSPATIAL TECHNOLOGIES (NNX06AE47G)

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Project web-site: http://giscenter.isu.edu/research/techpg/nasa_intl/template.htm

ACKNOWLEGDEMENTS
This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation from their assistance in obtaining this grant.

Recommended citation style:
Anderson, J., J. Tibbitts, and K.T. Weber. 2009. *2007 Range Vegetation Assessment at the O'Neal Ecological Reserve, Idaho*. Pages 3-14 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

TABLE OF CONTENTS

Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies

Executive Summary

- The treatments used to manage rangelands are simple and remain unchanged over thousands of years of development and progress. These treatments or tools are grazing, fire, and rest. The application of these treatments can have profoundly different effects on an ecosystem based upon how and when they are applied. The effects of grazing, for example, vary relative to the grazing animal and the density at which they are grazed. Even more importantly, is the effect of time or the duration of grazing relative to the amount of time the plants are allowed for recovery. The results of the research conducted over the past several years indicate that 1) the effect of grazing with partial-rest is little different than total rest as both lead to varying rates of desertification (cf. chapters 8 and 14), 2) the sedenterization of once nomadic herders has led to increased rates of land degradation and accelerated desertification (cf. chapters 10 and 14), and 3) improvements in rangeland condition can be made through the use of planned grazing that minimizes animal latency (approximately 3-5 days per pasture) through high herd density and high animal impact.

- The presence of bare ground is a primary indicator of rangeland health and areas with high proportions of bare ground are nearly always associated with degraded ecosystems. While the juxtaposition of bare ground patches relative to patches of vegetation is an important consideration at fine scales, the overall percent bare ground exposure is a critical measure of rangeland health, ecosystem function, and biotic integrity at the landscape scale.

  Using remotely sensed imagery it is possible to estimate bare ground over large regions of the earth. The accuracy of such estimates is important and challenges/limitations exist relative to the ability of remote sensing techniques to reliably detect bare ground. In this study, bare ground was accurately modeled (85% overall accuracy) with classification tree techniques and SPOT-4 (20mpp) satellite imagery along with various topographic/morphometric datasets. This research (cf. chapter 8) improves upon previous results achievable only through the use of high-resolution satellite imagery (Quickbird [2.4mpp]).

- Rangelands are dynamic ecosystems experiencing multiple "green waves" each year. The first tends to occur when ephemeral grasses (typically annual grasses like cheatgrass [*Bromus tectorum*]) and forbs germinate in early spring, while the second "green wave" occurs when other grasses (perennials like Bluebunch wheatgrass [*Pseudoroegneria spicata* (Pursh) A. *Löve*]) initiate active growth later in the spring and summer. During mid-summer photosynthetic activity declines, but given sufficient autumn precipitation, a third "green wave" may occur in late summer/early fall.

Quantifying rangeland productivity has always been challenging and employing satellite imagery to address this question generates additional challenges. The observed inter-annual variation in productivity precludes the use of single-date imagery to estimate productivity. Results of this study (cf. chapter 9) indicate that multi-date imagery and composite NDVI may be much better suited to estimate rangeland productivity in semiarid ecosystems.

- Long-term/continuous, semi-extensive grazing can negatively impact arid and semiarid rangeland ecosystems through reduced productivity and changes in vegetation patch patterns. The effect of continuous animal impact, such as that seen near water holes and shelters, results in lower NDVI (cf. chapters 10, 14, and 15). These characteristics translate into areas of low vegetation cover/high bare ground resulting long-term overgrazing of plants without sufficient recovery periods.

2007 Rangeland Vegetation Assessment at the O'Neal Ecological Reserve, Idaho

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ABSTRACT

Vegetation data was collected at stratified, randomly located sample points between June 18 and July 16, 2007 (*n*=148). Data was collected through both ocular estimation and line-point intercept transects each describing the 1) percent cover of grasses, forbs, shrubs, litter and exposure of bare ground 2) dominant weed and shrub species, 3) fuel load, 4) sagebrush age, 5) GAP land cover class, 6) presence of microbial crust, 7) litter type, 8) forage availability, and 9) photo points. Sample points were stratified by grazing and total rest treatments. The three strata (simulated holistic planned grazing, rest-rotation, and total rest) had variations in the ground cover perhaps due to the different treatments.

KEYWORDS: vegetation, sampling, GIS, remote sensing, GPS, grazing treatment, land management

INTRODUCTION

Many factors influence land cover changes. Wildfire has been, and will always be, a primary source of broad scale land cover change. Also, grazing management decisions and practices has been linked to land cover change. With wildfire or grazing, a change in plant community composition, plant structure, or ecosystem function may result in increases in bare earth exposure and decreases in land sustainability. In some systems, native plants are in competition with non-native vegetation that is more aggressive. The increase of non-native vegetation can directly result in the reduction of livestock and wildlife carrying capacities. Fire frequency may also increase. An example of non-native vegetation that out competes native vegetation and increases fire frequency is cheatgrass (*Bromus tectorum*). A research project located at the O'Neal Ecological Reserve is being conducted to A) determine if planned, adaptive grazing can be used to effectively decrease bare earth exposure B) determine if ground moisture changes relative to bare earth exposure and livestock grazing and C) examine the ecological effects of livestock grazing. The approximate location of the study area is shown below (Figure 1).

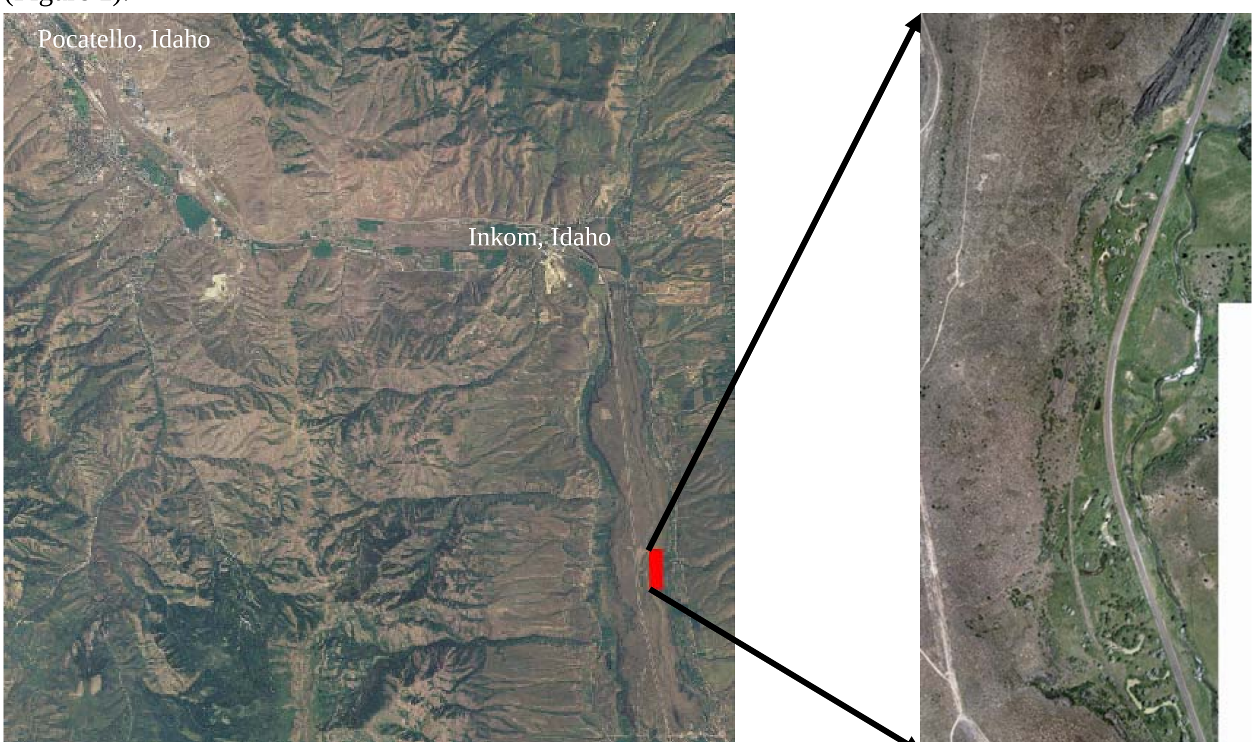

Figure 1. Research study area. The O'Neal Ecological Reserve, represented by red rectangle, is located near McCammon, Idaho.

We sampled three different grazing treatments; adaptive (simulated holistic planned grazing (SHPG)), rest-rotation (traditional), and total rest (no grazing). After comparing various traits in each of these areas we infer various generalizations which can shed light on relationships between these variables and may aid range managers in making decisions about prescribed and targeted grazing management.

METHODS

Sample points were randomly generated across the study area. Each point met the following criteria:

1) >70 meters from an edge (road, trail, or fence line)
2) <750 meters from a road.

The sample points were stratified by grazing treatment with 50 points in each treatment for a total of 150 sample points. The three grazing treatments were: 1) SHPG 2) rest-rotation and 3) total rest.

The location of each point was recorded using a Trimble GeoXH GPS receiver (+/-0.20 m after post processing with a 95% CI) using latitude-longitude (WGS 84) (Serr et al., 2006). Points were occupied until a minimum of 20 positions were acquired and WAAS was used whenever available. All points were post-process differentially corrected using Idaho State University's GPS community base station. The sample points were then projected into Idaho Transverse Mercator NAD 83 using ESRI's ArcGIS 9.2 for datum transformation and projection (Gneiting, et al., 2005).

Ground Cover Estimation

Estimations were made within 10m x 10m square plots (equivalent to one SPOT 5 satellite image pixel) centered over each sample point with the edges of the plots aligned in cardinal directions. First, visual estimates were made of percent cover for the following; bare ground, litter, grass, shrub, and dominant weed. Cover was classified into one of 9 classes (1. None, 2. 1-5%, 3. 6-15%, 4. 16-25%, 5. 26-35%, 6. 36-50%, 7. 51-75%, 8. 76-95%, and 9. >95%).

Observations were assessed by viewing the vegetation perpendicular to the earth's surface as technicians walked each site. This was done to emulate what a "satellite sees". In other words the vegetation was viewed from nadir (90 degree angle) as much as possible.

Next, transects were used to estimate percent cover of bare ground exposure, rock (>75 mm), litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed. Percent cover estimates were made along two 10 m line transects. Transects were arranged perpendicular to each other and crossing at the center of the plot at the 5 m mark of each line transect. Using the point-intercept method, observations were recorded every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm. The cover type (bare ground exposure, rock (>75 mm), litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed) at each observation point was recorded (n = 50 points for each line transect and 100 points for each plot).

The litter cover type included biomass that was on the ground and in contact with the ground. Live herbaceous species included live (i.e., green) forbs and grasses, while live shrubs included all species of shrubs.

Fuel Load Estimation

Fuel load was estimated at each sample point. Visual observations of an area equivalent to a SPOT 5 pixel, (10 mpp or approximately 100 m^2), centered over the sample point were used to estimate fuel load. These categories were derived from Anderson (1982) (Table 1).

Table 1. Fuel load classes and associated tonnage of fuels.

Fuel Load Class	Tons/acre
1	0.74
2	1.00
3	2.00
4	4.00
5	>6.0

Forage Measurement

Available forage was measured using a plastic coated cable hoop 2.36 m in circumference, or 0.44 m^2. The hoop was randomly tossed into each of four quadrants (NW, NE, SE, and SW) centered over the sample point. All vegetation within the hoop that was considered forage for cattle, sheep, and wild ungulates was clipped and weighed (+/-1g) using a Pesola scale tared to the weight of an ordinary paper bag. All grass species were considered forage. The measurements were then used to estimate forage amount in AUM's, pounds per acre, and kilograms per hectare (Sheley et al. 1995).

Microbiotic Crust Presence

Microbiotic crusts are formed by living organisms and their by-products, creating a surface crust of ground particles bound together by organic materials. Presence of microbial crust has been linked to degraded rangelands, but is still seen as being better that bare ground as they can retain water very well even against an osmotic pull helping to reduce erosion (Johnston 1997). The presence of microbiotic crust was evaluated at each sample point and recorded as either present or absent. Any trace of a microbiotic crust was defined as “presence”.

GAP Analysis

Land cover was described using a list of vegetation cover types from the GAP project (Jennings 1997). The GAP vegetation description that most closely described the sample point was selected and recorded.

Litter Type

Litter was defined as any biotic material that is no longer living. Litter decomposes and creates nutrients for new growth. For the litter to decompose it needs to be in contact with the ground in order for the microbes in the ground to break down the dead substance. If the litter is suspended in the air it turns a gray color and takes an immense amount of time to decompose through chemical oxidation. If it is on the ground it is a brownish color and decomposes biologically at a much faster rate. The type of litter present was recorded by color: either gray (oxidizing) or brown litter (decaying).

Big Sagebrush (Artemisia tridentata spp.) Age Estimation

Maximum stem diameter (up to the first 0.30 m of stem) of Big sagebrush plants was measured using calipers (+/-1cm) to approximate the age of each plant (Perryman and Olson 2000) A maximum of four samples were taken at each sample point, one within each quadrant (NW, NE, SE, and SW). The sagebrush plant nearest the plot center within each quadrant was measured using calipers (+/-1cm) and converted to millimeters. The age of each big sagebrush plant was then estimated using the following equation (AGE = 6.1003 + 0.5769 [diameter in mm]).

Photo Points
Digital photos were taken in each of 4 cardinal directions (N, E, S, and W) from the sample point.

RESULTS
Ground Cover Estimates
Based upon ocular estimates, ten percent of all 2007 field samples (n = 14) had >50 % exposed bare ground and 77 % of samples (n = 113) has bare ground exposure <=35 %. The dominant weed present in 100 % of the 2007 samples was cheatgrass. Eighty-one percent of the sample points had >5% cheatgrass cover where the majority, 82 %, were <= 25 % cover and the maximum cover of cheatgrass was 51-75 % with 1.4 % of samples (n = 2) falling within the maximum range. The majority, sixty-one percent, of the samples had <16 % grass cover.

Based upon transect estimates, the maximum bare ground exposure was 86%, the maximum cheatgrass cover was 53%, the maximum grass cover was 34%, the maximum shrub cover was 66% and the maximum forb cover was 26%.
To truly understand ground cover estimates in relation to grazing treatments, each grazing treatment was independently analyzed. The mean cover classes of each cover type were separated by grazing treatment and are summarized in Table 2.

Table 2. Mean cover class of each cover type separated by grazing treatment.

Cover Class	SHPG Mean Cover	Rest-Rotation Mean Cover	Total-Rest Mean Cover
Bare ground	16-25%	26-35%	16-25%
Shrub	26-35%	36-50%	26-35%
Grass	6-15%	1-5%	6-15%
Litter	26-35%	6-15%	6-15%
Weed	6-15%	16-25%	16-25%
Forb	6-15%	1-5%	1-5%

Ocular estimates were compared with the previous year, 2006. Compared to the 2006 mean cover class, bare-ground exposure has decreased in every grazing treatment. Mean shrub has increased in all but the total-rest treatment. Mean grass, litter, and forb have increased only in the adaptive treatement whereas mean litter decreased in both the rest-rotation and total-rest treatment. Mean weed cover has increased across each treatment.

To qualitativley visualize how the above changes in mean relate to the overall distribution of each cover class, frequency distributions of each cover class were also graphed from 2006 and 2007. The frequency distribution graphs of each grazing treatement from both 2006 and 2007 are shown in figures 2-7.

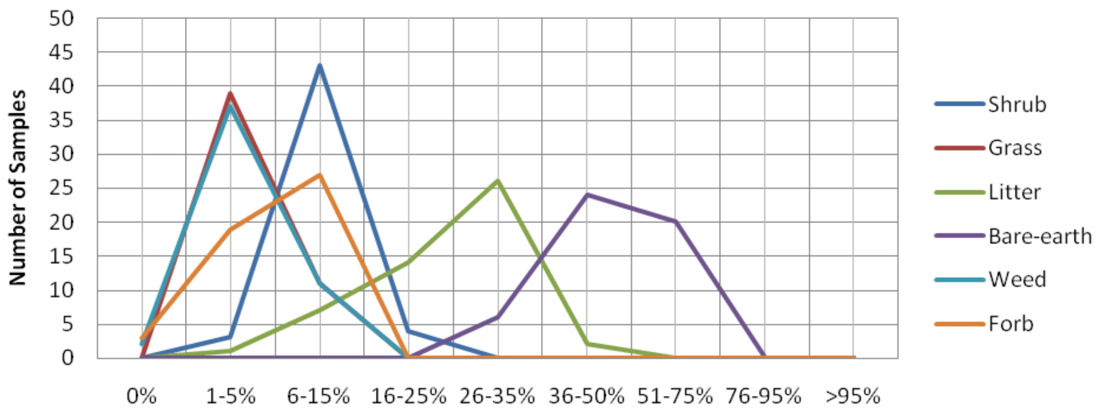

Figure 1. 2006 ground cover estimates in the adaptive grazing treatment. Cover classes are given along the horizontal (x) axis.

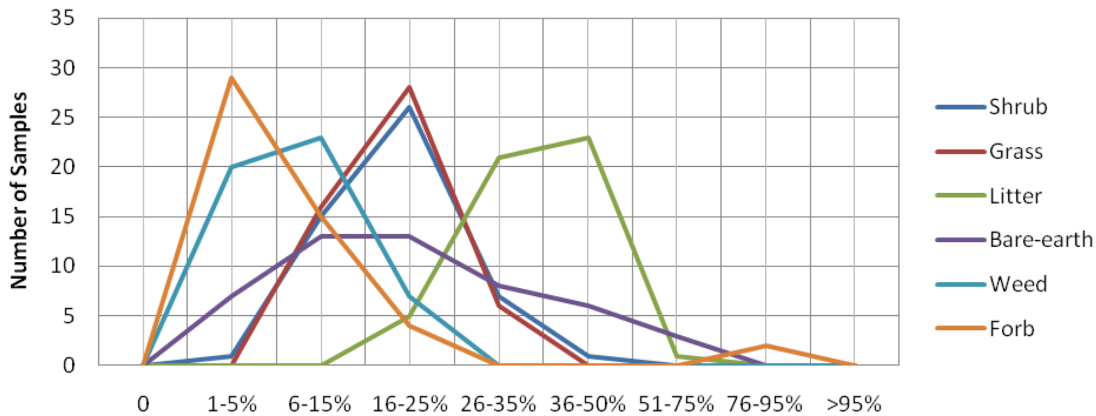

Figure 2. 2007 ground cover estimates in the adaptive grazing treatment. The cover classes are given along the horizontal (x) axis.

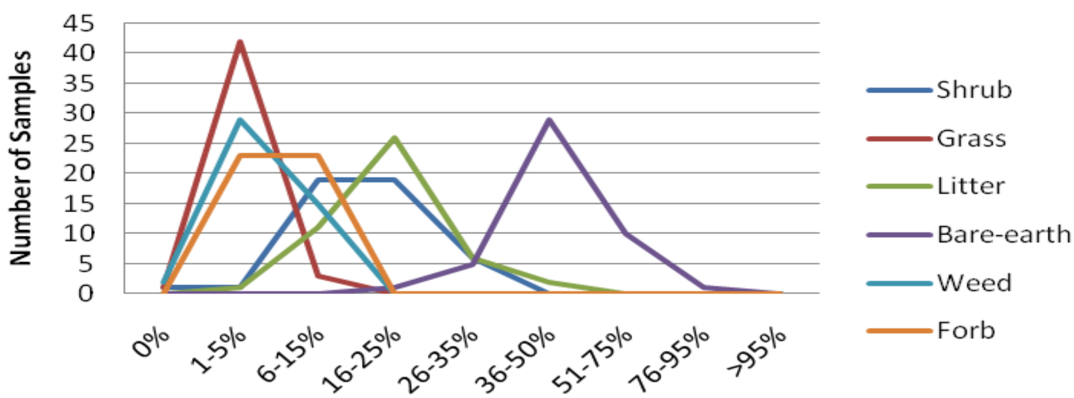

Figure 3. 2006 ground cover estimates in the rest-rotation grazing treatment. The cover classes are along the horizontal (x) axis.

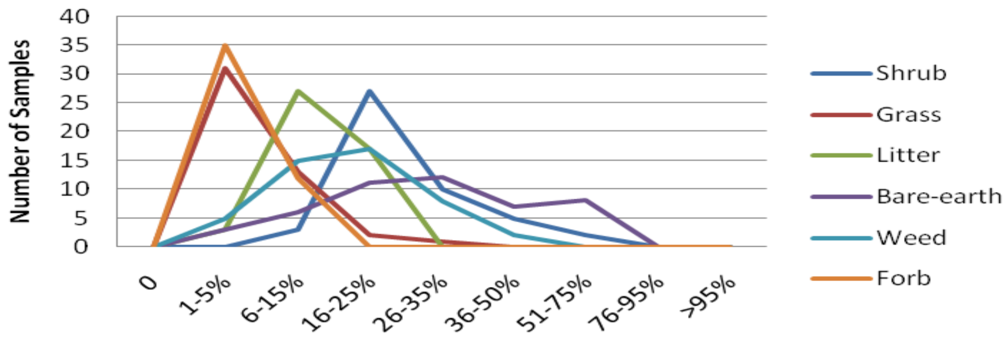

Figure 4. 2007 ground cover estimates in the rest-rotation grazing treatment. The cover classes are given along the horizontal (x) axis.

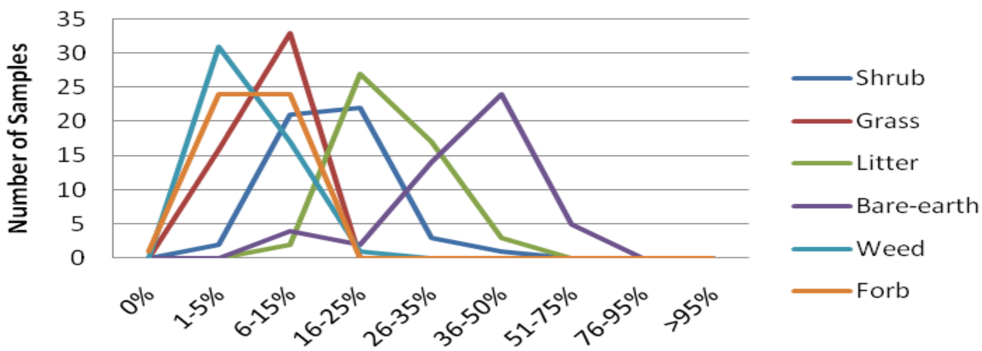

Figure 5. 2006 ground cover estimates in the total rest grazing treatment. The cover classes are given along the horizontal (x) axis.

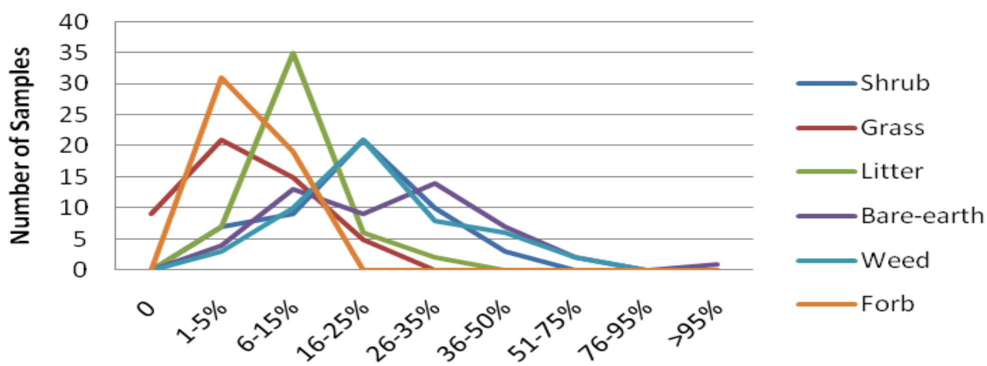

Figure 6. 2007 ground cover estimates in the total rest grazing treatment. The cover classes are given along the horizontal (x) axis.

A two-tailed Mann-Whitney U test was performed to quantify the difference between the distributions of cover classes in 2006 and 2007. The Mann-Whitney test asks if the distribution of a test statistic (ground cover) is the same across two samples. The Mann-Whitney test can be used regardless of distribution normality (mean, median, etc.) and can be used with categorical data (the type of data collected in this study). The results of the Mann-Whitney test are given in Table 3.

Table 3. Summary of two-tailed Mann-Whitney U-test results to determine if cover classes differed within treatment between years (2006 and 2007).

SHPG	**P-Value**
Bare ground	0.000002
Shrub	0.000002
Litter	0.000002
Grass	0.000002
Weed	0.000136
Forb	0.804104 *
Rest-Rotation	
Bare ground	0.000006
Shrub	0.000004
Litter	0.000112
Grass	0.013150
Weed	0.000002
Forb	0.396219 *
Total-Rest	
Bare ground	0.000004
Shrub	0.123248 *
Litter	0.000002
Grass	0.000242
Weed	0.000002
Forb	0.404594 *

Note: cover classes indicated with an asterisk (*) did not differ between years.

Fuel Load Estimation

The majority of field samples (95%; *n*=140) had fuel load estimates between 2-5 tons/acre. The remaining 5 % (*n*=7) had fuel load estimates < 2 tons/acre. The occurrence of fuel loads < 2 tons/acre in 6 of the 7 samples were in areas of high lava rock exposure (>50%) and the remaining 1 sample that was not lava rock had high bare ground exposure >50%.

Forage Measurements

Using AUM Analyzer software (Sheley et al., 1995), forage amount and available Animal Units were calculated. Mean forage available was 77.99 kg/ha with a standard deviation of 61.16. The minimum forage available was 6 kg/ha and the maximum forage available was 287 kg/ha. Grazing treatments were separated to compare available forage between them (Table 4).

Table 4. A comparison of forage estimates across grazing treatments.

Grazing Treatment	Minimum (kg/ha)	Maximum (kg/ha)	Mean (kg/ha)	Standard Deviation
Adaptive	23	141	59.53	24.92
Rest-rotation	6	124	39.47	25.72
Total-rest	17	287	132.3	70.80

A statistical test was performed on the forage estimates to check differences between grazing treatment forage estimates. A simple ANOVA was performed which determined that the difference between mean forage estimates between grazing treatments were not statistically different (p=0.05). Furthermore, each grazing treatment was individually compared to each other through a paired *t*-test and the differences again were not significantly different. The paired *t*-test results are summarized in Table 5.

Table 5. Results of two-tailed t-test of forage means between grazing treatments. No significant differences were seen (95 % CI).

Hypothesis Tested	Difference Between Means	95% CI for Difference Between Means	Two-Tailed P Value
SHPG Mean = Rest-Rotation Mean	20.06	-51.04 to 91.16	0.52
SHPG Mean = Total Rest Mean	-72.77	-221.72 to 76.18	0.33
Rest-Rotation Mean = Total Rest Mean	-92.83	-246.06 to 60.40	0.23

Microbiotic Crust Presence

In 2007, 86.4% of sample points (127 of 147) had microbial crust present. In 2006, 82.1% (119 of 149) had microbial crust. This change in presence of microbial crust is not significant within a 95% confidence interval.

GAP Analysis

Four GAP classifications were observed in 2007—vegetated lava, sagebrush grassland, big sagebrush, and bitterbrush. The majority of sample points (70%; *n*=103) were classified as sagebrush grassland, 19 % (*n*=28) as vegetated lava, 9.5% (*n*=14) as bitterbrush, and 1.4% (*n*=2) as big sagebrush.

Litter Type

Biologically decaying (brown) litter was dominant at 41% (*n*=60) of the sample points oxidizing (gray) litter was dominant at 1.4% (*n*=2) of the sample points while at 57.1% (*n*=84) of the sample points no discrimination of dominant litter type could be made and the litter type was classified as "both".

Big Sagebrush Age Estimation

The mean age of sagebrush plants sampled was 18.75 years (*n* = 142). The minimum age was 8 years and the maximum age was 36 years. The standard deviation was 6.63159. Figure 8 shows the frequency distribution of sagebrush age.

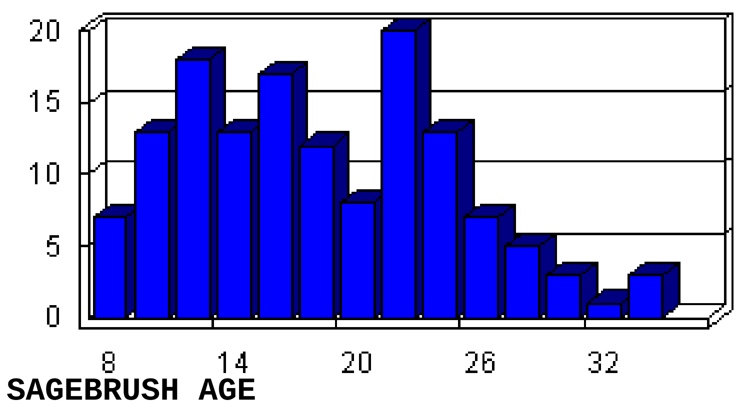

Figure 7. Cumulative frequency graph of sagebrush age estimates at the O'Neal Ecological Reserve.

CONCLUSIONS

The differences between the three treatments were interesting. Figures 2-7 are histograms of ground cover estimates comparison results from 2007 to those from 2006. There were significant differences in cover distributions that could be attributed to differing management practices. Further analysis and comparison with future sampling will hopefully provide better discrimination of these changes.

Desertification and land degradation is primarily evaluated through shifts of the keystone indicator, bare ground exposure. A land manager would want to see smaller percentages of bare ground exposure (i.e. the distribution curve shifts left) while grass, forb, shrub, and litter cover would preferably increase to higher percentages (i.e. the distribution curve shifts right). While differences in bare ground exposure and weed cover distributions (Figures 2-7) were significant in all treatments, it is the direction of the shift that is the major concern. Adaptive grazing appears to show the most promise in producing a relatively rapid shift of bare ground exposure toward smaller percentages. These early, albeit non-conclusive, trends can help to re-evaluate management decisions to correct or shift the changes toward more beneficial directions according to management goals and overall sustainability goals

It should be noted that the differences observed were most likely caused by different grazing treatments in each of the areas but observational bias and/or other environmental factors may have contributed to some of these changes. Furthermore, the sampling of the O'Neal was done only 3 weeks after grazing. Some of the changes that are shown, especially in grazed areas, could be different if sampling were done at a different time of year (i.e. pre-grazing or late Fall). However, the purpose of the total rest treatment is to infer the characteristics of the grazed treatments without grazing. But again, analyses of changes in relation to grazing are important in assessing management decisions. The primary goal should be early detection of degradation processes in order to make changes in management before it is too late or desertification thresholds are surpassed.

Regarding shrub cover, there has been an infestation of the sage defoliation moth (*Aroga coloradensis)* at the O'Neal site. In 2006, a large proportion of sagebrush was defoliated and therefore had no photosynthetically active leaves resulting in low sagebrush cover estimats. In 2007, there was a noted increase in recovering sagebrush resulting in higher leaf coverage than 2006. This information may explain the increase in shrub cover in the adaptive and rest-rotation pastures.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Anderson, J., J. Tibbitts, and K. T. Weber. 2009. *2007 Rangeland Vegetation Assessment at the O'Neal Ecological Reserve, Idaho*. Pages 3-14 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

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2007 Range Vegetation Assessment at the United States Sheep Experiment Station, Dubois, Idaho

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ABSTRACT
The rangeland vegetation of two summer pastures at the US Sheep Experiment Station (USSES) in Dubois, Idaho was assessed in the summer of 2007. Field measurements were made at 99 randomly generated point locations, with 49 and 50 sample points at the Henniger and Humphrey pastures respectively. Ground cover types, their percent cover, and available forage biomass were estimated within 10m x 10m plots at the 99 locations. Live herbaceous species had the greatest mean percent cover at the Humphrey pasture (61%), while shrubs represented the greatest mean percent cover at the Henniger pasture (35%). Available forage biomass estimates were 300 kg per hectare at the Henniger pasture, and 669 kg per hectare at the Humphrey pasture. This is the first year of data collection at either of these USSES pastures.

KEYWORDS: Field measurements, forage estimate, ground cover estimate

INTRODUCTION

The 2007 sampling effort focuses upon the Humphrey and Henniger pastures at the U. S. Sheep Experiment Station (USSES) near Dubois, Idaho (Figure 1). The Humphrey pasture consists of 2,600 acres of land near Monida, Montana and is used for spring, summer, and autumn grazing and rangeland research. The Henniger pasture consists of 200 acres of land near Kilgore, Idaho, and is used for summer grazing and rangeland research. Mean annual precipitation (1971 to 2000) at the Dubois Experiment Station (112° 12' W 44° 15'N, elevation, 1661 m) is 331 mm with 60% falling during April through September. Soils are mapped as complexes of Maremma (Fine-loamy, mixed, superactive, frigid Calcic Pachic Argixerolls), Pyrenees (Loamy-skeletal, mixed, superactive, frigid Typic Calcixerolls), and Akbash (Fine-loamy, mixed, superactive, frigid Calcic Pachic Argixerolls) soils on slopes less than 20 percent, but mostly 0 to 12 percent (NRCS 1995).

Vegetation on the study sites are sagebrush-grass communities that is dominated by mountain big sagebrush (*Artemisa tridentata* ssp. *vaseyana* [Rydb.] Beetle) and threetip sagebrush (*A. tripartita* Rydb.). Subdominant shrub species include antelope bitterbrush (*Purshia tridentata* [Pursh] DC.), yellow rabbitbrush (*Chrysothamnus viscidiflorus* (Hook.) Nutt.), and spineless horsebrush (*Tetradymia canescens* DC.). There are a few small patches of the exotic forbs leafy spurge (*Euphorbia esula* L.) and spotted knapweed (*Centaurea stoebe* L. ssp. *micranthos* [Gugler] Hayek) and trace amounts (<1% of overall plant cover) of the exotic annual cheatgrass. Lupine (*Lupinus argenteus* Pursh) is the most plentiful forb on the study sites and the graminoids present are thickspike wheatgrass (*Elymus lanceolatus* [Scribn. & J.G. Sm.] Gould ssp. *lanceolatus*), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve ssp. *spicata*), and plains reedgrass (*Calamagrostis montanensis* Scribn. ex Vasey).

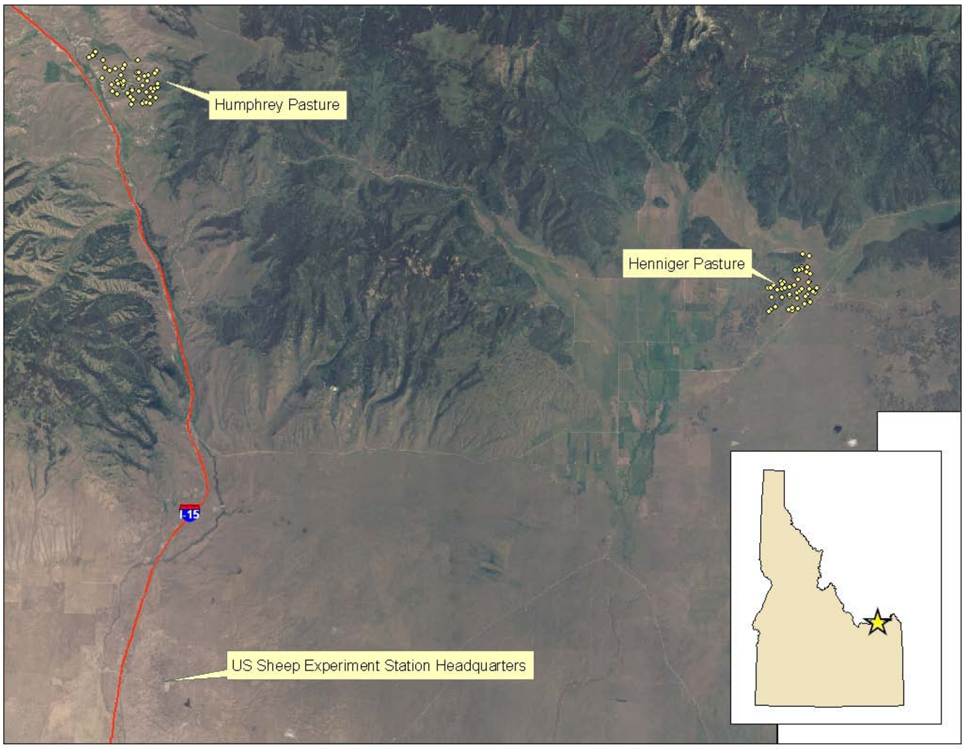

Figure 1. US Sheep Experiment Station, Henniger and Humphrey pastures.

The objectives of this study are to: 1) assess the rangeland vegetation at the Henniger and Humphrey pastures using LANDSAT and SPOT satellite imagery and field measurements and 2) compare the rangeland vegetation assessment with similar assessment performed at the Tsakhiriin tal area of the

Darkhad Valley, Mongolia. The field-based measurements of the USSES pasture vegetation assessment were performed in late July-early August of 2007. The results of the field-based measurements are presented here and will be later combined with satellite imagery analysis results.

METHODS

A total of 100 random points were generated within the Henniger and Humphrey pasture sites prior to field assessment. Each point represented a sample location, at which field measurements were made within 10m x 10m plots. The plots were centered at each random point and the edges of the plots were aligned in the cardinal directions. Four digital photographs were taken at each plot in each of the four cardinal directions. The field measurements included ground cover estimation and forage biomass measurement. Ground cover estimation included estimates of percent cover of bare soil, rock >75 mm, litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed. Percent cover estimates were made along two 10 m line transects, perpendicular to each other and crossing at the center of the plot at 5m of each line transect, using a point-intercept method (Gysel and Lyon 1980). Records were made every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm, to indicate the cover type at the point (n = 50 points for each line and 100 points for each plot).

Litter refers to biomass that is on the ground and in contact with the ground. Live herbaceous species refers to live (i.e., green) forbs and grasses, while live shrubs include all species of shrubs.

Forage biomass was measured twice at each sample plot using one 2 meter by 0.5 meter modified Daubenmire frame (Daubenmire and Daubenmire 1968). The frame was placed north of the east-west transect line, with the 0.5 meter end centered on the transect tape at 2.5 meters. The frame was then moved south of the east-west transect line, with the 0.5 meter end of the frame centered on 7.5 meters. All green and senescent herbaceous biomass was clipped, separated into forbs and grasses, and then wet-weighed in a paper bag using a spring scale (Chambers and Brown 1983). For each field day, 5 bags each of clipped forbs and grasses were labeled and kept to be air-dried and weighed again to convert wet weights to dry weights.

RESULTS

The most common ground cover type at the Henniger pasture was shrub species with a mean estimate of 35% cover (Figure 2). The second most common ground cover type was grass, which made up 19% cover on average. The most common ground cover type at the Humphrey pasture was grass species with a mean estimate of 42% cover (Figure 2). The second most common ground cover type was shrub, which made up 26% cover on average. Bare soil cover was more common at the Henniger pasture than at the Humphrey pasture, comprising 17% and 3% respectively. Forb cover class was more common at the Humphrey pasture than at the Henniger pasture, comprising 19% and 8% respectively. Other ground cover types of rock, weed, standing dead wood, and standing dead herbaceous comprised less than 1% cover (Figure 2).

Plot-level averages of wet forage biomass ranged between 31-568 gm per frame at the Humphrey pasture, and between 2-142 gm per frame at the Henniger pasture. The average wet forage biomass at Humphrey was 66 gr per frame, while the average wet forage biomass at Henniger was 18 gr per frame. These estimates translate to average forage biomass of 300 kg per hectare at the Henniger pasture, and 669 kg per hectare at the Humphrey pasture.

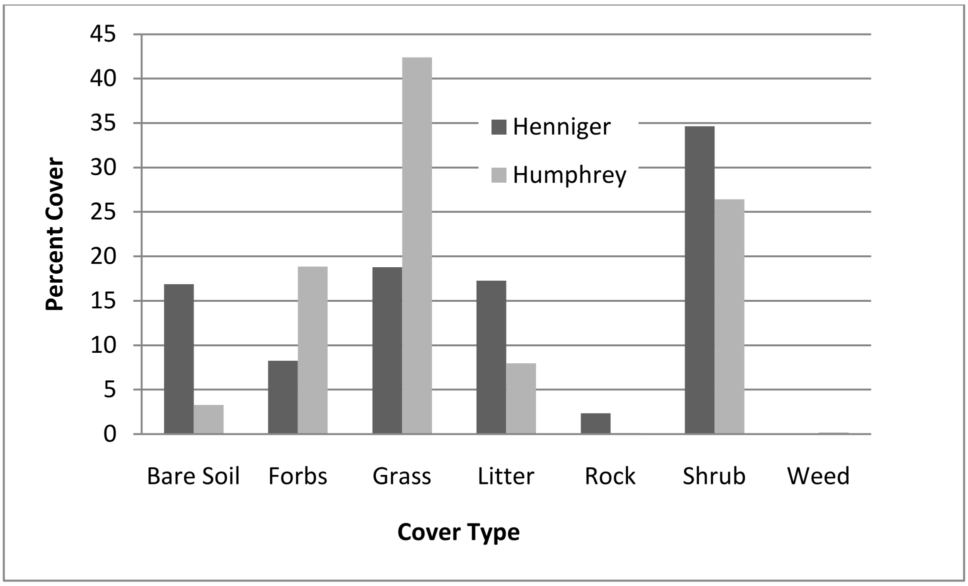

Figure 2. Mean (+SE) percent cover of all primary ground cover types at the Henniger and Humphrey pastures in the summer of 2007. Percent Standing Dead Wood, and Percent Standing Dead Herbaceous were <1%.

CONCLUSIONS

The observed ground cover types and their estimated percent cover showed distinct differences between the two pastures. While the Humphrey pasture sees sheep grazing during the spring, summer and fall, as well as a low number of cattle in the fall, forage per acre is much greater than that found at the Henniger pasture. The lower percentage of shrub at the Humphrey pasture might be attributed to the higher altitude of the site and the resultant high snow load and high winter winds. Low percent cover of bare soil and weeds, but high percent cover of live herbaceous species might suggest that the Humphrey pasture, while heavily used, is not overgrazed.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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2007 Range Vegetation Assessment in the Darkhad Valley, Mongolia

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ABSTRACT
The rangeland vegetation of the Tsakhiriin tal study site in the Darkhad Valley of north-western Mongolia was assessed in the summer of 2007. Field measurements were made at 100 randomly generated point locations throughout the study site. Ground cover types, their percent cover, and available forage biomass were estimated within 10m x 10m plots at the 100 locations. Live herbaceous species had the greatest mean percent cover, while rock, weeds, and shrub cover types were estimated have a mean of less than 0% cover. Available forage biomass estimates were 1434 kg per hectare in the Tsakhiriin tal study site. The observed patterns were consistent with the expected trends in the Darkhad Valley rangelands that are continuously grazed throughout the growing season.

KEYWORDS: Field measurements, forage estimate, ground cover estimate

INTRODUCTION

Mongolia, a continental semi-arid country, is known as one of the five most heavily grazed places in the world (Asner et al., 2004). All grazing lands in Mongolia are public lands, although the herds are privately owned by nomadic herders who migrate at least four times a year between seasonal pastures. Darkhad Valley in north-western Mongolia is grazed by several different livestock species: cattle, sheep, goats, and horses. The Tsakhiriin Tal area in the Darkhad Valley, the focus area of this study (Figure 1), is primarily used as summer pasture by approximately 30 nomadic households. Rangeland wildfires are very rare in the Darkhad Valley potentially due to low fuel accumulation associated with continuous grazing use. The isolated small forest stands along the boundary of this study site (Figure 1) are also likely excluded from fire disturbance, although the continuous, expansive forest stands surrounding the Darkhad Valley might have higher fire frequency (Sankey et al., 2006).

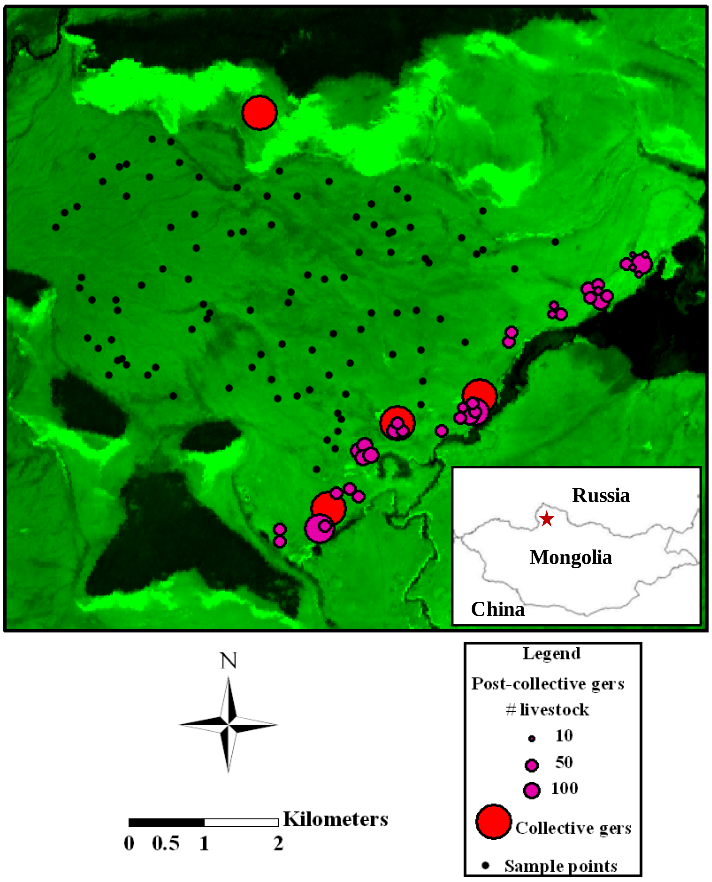

Figure 1. Mongolia and the Tsakhiriin tal study site in the Darkhad Valley.

The point locations in the Tsakhiriin tal LANDSAT imagery indicate 100 randomly generated points at which field-based measurements were made in the summer of 2007.
The objectives of this study are to: 1) assess the rangeland vegetation in the Tsakhiriin tal area of the Darkhad Valley using LANDSAT and SPOT satellite imagery and field measurements and 2) compare the rangeland vegetation assessment with similar assessment performed at the US Sheep Experimental Station in Idaho, USA. The field-based measurements of the Tsakhiriin tal rangeland vegetation assessment were performed in late June-early July of 2007. The results of the field-based measurements are presented here and will be later combined with satellite imagery analysis results.

METHODS

A total of 100 random points were generated within the Tsakhiriin tal study site prior to field assessment. Each point represented a sample location, at which field measurements were made within 10m x 10m plots. The plots were centered at each random point and the edges of the plots were aligned in the cardinal directions. Two digital photographs were taken at each plot to record the general characteristics of each point at a landscape scale and at the plot scale. The field measurements included ground cover estimation and forage biomass measurement. Ground cover estimation included estimates of percent cover of bare soil, rock >75 mm, litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed. Percent cover estimates were made along two 10 m line transects, perpendicular to each other and crossing at the center of the plot at 5m of each line transect, using a point-intercept method. Records were made every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm, to indicate the cover type at the point (n = 50 points for each line and 100 points for each plot).

Litter refers to biomass that is on the ground and in contact with the ground. Live herbaceous species refers to live (i.e., green) forbs and grasses, while live shrubs include all species of shrubs.

Forage biomass was measured in four cable hoops 93 inches in circumference and 0.44 m^2 in area. The hoops were tossed randomly in each of the four quadrants of each plot. All green and senescent herbaceous biomass was clipped and wet-weighed in a paper bag using a spring scale. For each field day, 5 bags of clipped biomass were labeled and kept to be air-dried and weighed again to convert wet weights to dry weights.

RESULTS

The most common ground cover type was live herbaceous species with a mean estimate of 69% cover (Figure 2). The second most common ground cover type was litter, which made up 18% cover on average. Standing dead herbaceous and bare soil cover types fairly minimal with means of 7.8% and 4.6% cover respectively. Other ground cover types of rock, shrub, and weed comprised less than 0% cover (Figure 2).

Plot-level averages of wet forage biomass ranged between 35-225 gm per hoop. The average wet forage biomass at this study was 63 gr per hoop. These estimates translate to average forage biomass of 1434 kg per hectare.

CONCLUSIONS

The observed ground cover types and their estimated percent cover were similar to the expected trends in the Darkhad Valley grasslands. Standing dead herbaceous species and litter cover types were estimated to have low percent cover, which might be expected in such continuously grazed areas. Low percent cover of bare soil and weeds, but high percent cover of live herbaceous species might suggest that this area is not overgrazed, although it is grazed continuously throughout the growing season. Shrubs were not found in any of the plots in the Tsakhiriin tal study area. This was consistent with the observed patterns in the Darkhad Valley, where shrubs are present only in ungrazed riparian areas.

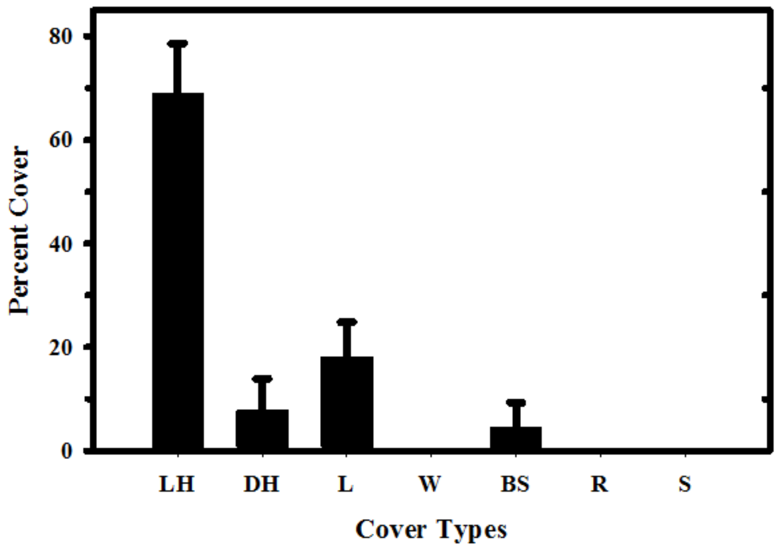

Figure 2. Mean (±SE) percent cover of all ground cover types (LH=Live Herbaceous, DH=Standing Dead Herbaceous, L=Litter, W=Weeds, BS=Bare soil, R=Rock, S=Shrubs) at the Tsakhiriin tal study site in the summer of 2007.

ACKNOWLEDGEMENTS
This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Sankey, T. T. 2009. *2007 Rangeland Vegetation Assessment in the Darkhad Valley, Mongolia*. Pages 19-22 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

2008 Rangeland Vegetation Assessment at the O'Neal Ecological Reserve, Idaho

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ABSTRACT
Vegetation data was collected at stratified, randomly located sample points during May and June, 2008 (*n*=149). Data was collected through both ocular estimation and line-point intercept transects each describing the 1) percent cover of grasses, forbs, shrubs, litter and exposure of bare ground 2) dominant weed and shrub species, 3) fuel load, 4) sagebrush plant age, 5) GAP land cover class, 6) presence of microbial crust, 7) litter type, 8) forage availability, and 9) name of collected photo point files. Sample points were stratified by grazing and rest treatments. The three strata (simulated holistic planned grazing, rest-rotation, and total rest) had variations in the ground cover due to the difference in treatments.

KEYWORDS: Vegetation, sampling, GIS, remote sensing, GPS, grazing treatment, land management.

INTRODUCTION

Many factors influence land cover changes. Wildfire has been, and will always be, a primary source of broad scale land cover change. Also, grazing management decisions and practices have been linked to land cover change. With wildfire or grazing, a change in plant community composition, plant structure, or ecosystem function may result in increases in bare ground exposure and decreases in land productivity. In some systems, native plants are in competition with non-native vegetation that is more competitive. The increase of non-native vegetation can directly result in the reduction of livestock and wildlife carrying capacities. Fire frequency may also increase and as an example, cheatgrass (*Bromus tectorum*) has been shown to alter the fire regime in a very self-perpetuating feedback cycle. Research at the O'Neal Ecological Reserve is being conducted to A) determine if Simulated Holistic Planned Grazing can be used to effectively decrease bare ground exposure B) determine if soil moisture changes relative to bare ground exposure and treatment and C) examine the ecological effects of livestock grazing. The approximate location of the study area is shown below (Figure 1).

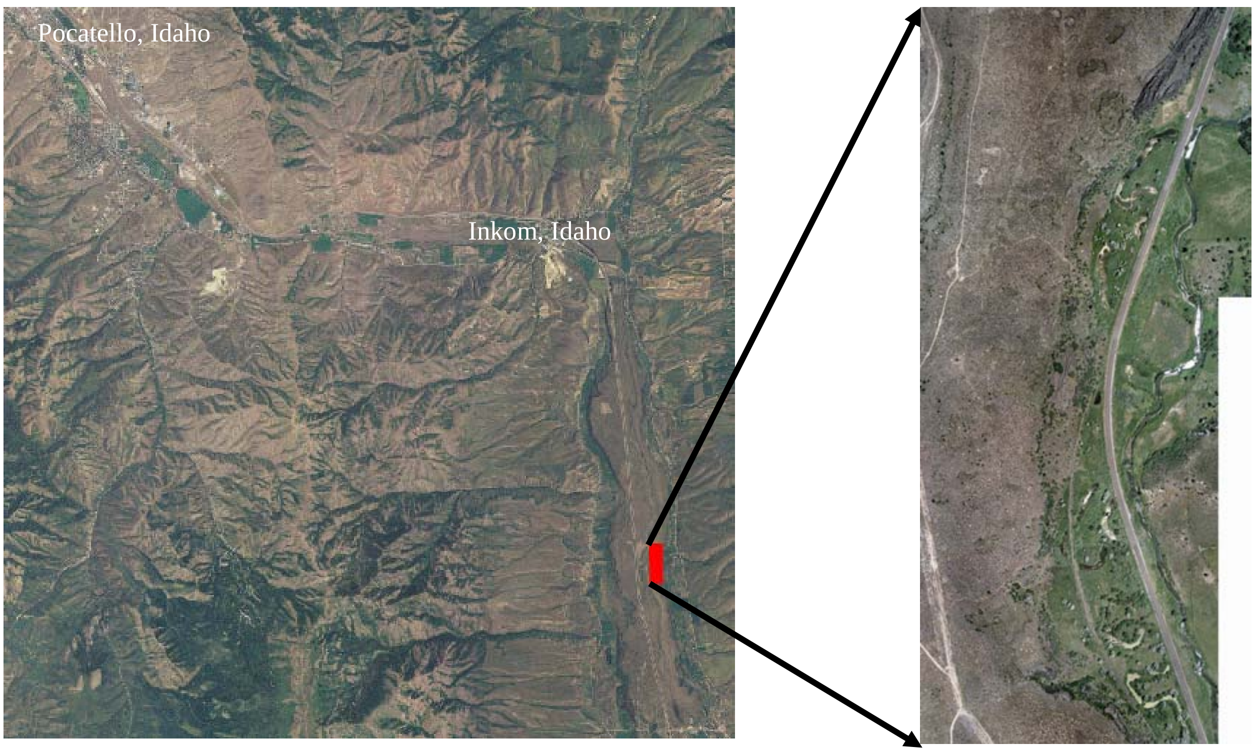

Figure 1. Research study area. The O'Neal Ecological Reserve, represented by red rectangle, is located near McCammon, Idaho.

We sampled three different grazing treatments; Simulated Planned Holistic Grazing (SHPG), rest-rotation (traditional), and total rest (no grazing). After comparing various traits in each of these areas we infer various generalizations which can shed light on relationships between these variables and may aid range managers in making decisions about prescribed and targeted grazing management.

METHODS

Sample points were randomly generated across the study area. Each point met the following criteria:

1) >70 meters from an edge (road, trail, or fence line)
2) <750 meters from a road.

The sample points were stratified by grazing treatment with 50 points placed in each treatment for a total of 150 sample points. The three grazing treatments were: 1) Simulated Holistic Planned Grazing (SHPG) 2) rest-rotation and 3) total rest.

The location of each point was recorded using a Trimble GeoXH GPS receiver (+/-0.20 m @ 95% CI after post processing) using latitude-longitude (WGS 84) (Serr et al., 2006). Points were occupied until a minimum of 20 positions were acquired and WAAS was used whenever available. All points were post-process differentially corrected using Idaho State University's GPS community base station. The sample points were then projected into Idaho Transverse Mercator NAD 83 using ESRI's ArcGIS 9.2 for datum transformation and projection (Gneiting, et al., 2005).

Ground Cover Estimation

Estimations were made within 10m x 10m square plots (equivalent to one SPOT 5 satellite image pixel) centered over each sample point with the edges of the plots aligned in cardinal directions. First, visual estimates were made of percent cover for the following; bare ground, litter, grass, shrub, and dominant weed. Cover was classified into one of 9 classes (1. None, 2. 1-5%, 3. 6-15%, 4. 16-25%, 5. 26-35%, 6. 36-50%, 7. 51-75%, 8. 76-95%, and 9. >95%).

Observations were assessed by viewing the vegetation perpendicular to the earth's surface as technicians walked each site. This was done to emulate what a "satellite sees". In other words the vegetation was viewed from nadir (90 degree angle) as much as possible.

Next, transects were used to estimate percent cover of bare ground exposure, rock (>75 mm), litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed. Percent cover estimates were made along two 10 m line transects. Transects were arranged perpendicular to each other and crossing at the center of the plot at the 5 m mark of each line transect. Using the point-intercept method, observations were recorded every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm. The cover type (bare ground exposure, rock (>75 mm), litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed) at each observation point was recorded (n = 50 points for each line transect and 100 points for each plot).

The litter cover type included biomass that was on the ground and in contact with the ground. Live herbaceous species included live (i.e., green) forbs and grasses, while live shrubs included all species of shrubs.

Fuel Load Estimation

Fuel load was estimated at each sample point. Visual observations of an area equivalent to a SPOT 5 pixel (10 mpp or approximately 100 m^2) centered over the sample point were used to estimate fuel load. These categories were derived from Anderson (1982) (Table 1).

Table 1. Fuel load classes and associated tonnage of fuels.

Fuel Load Class	Tons/acre
1	0.74
2	1.00
3	2.00
4	4.00
5	>6.0

Forage Measurement

Available forage was measured using a plastic coated cable hoop 2.36 m in circumference, or 0.44 m^2. The hoop was randomly tossed into each of four quadrants (NW, NE, SE, and SW) centered over the sample point. All vegetation within the hoop that was considered forage for cattle, sheep, and wild ungulates was clipped and weighed (+/-1g) using a Pesola scale tared to the weight of an ordinary paper bag. All grass species were considered forage. The measurements were then used to estimate forage amount in AUM's, pounds per acre, and kilograms per hectare (Sheley et al. 1995).

Microbiotic Crust Presence

Microbiotic crusts are formed by living organisms and their by-products creating a surface crust of ground particles bound together by organic materials. Presence of microbial crust has been linked to degraded rangelands, but is still seen as being better that bare ground as they can retain water very well even against an osmotic pull helping to reduce erosion (Johnston 1997). The presence of microbiotic crust was evaluated at each sample point and recorded as either present or absent. Any trace of a microbiotic crust was defined as "presence".

GAP Analysis

Land cover was described using a list of vegetation cover types from the GAP project (Jennings 1997). The GAP vegetation description that most closely described the sample point was selected and recorded.

Litter Type

Litter was defined as any biotic material that is no longer living. Litter decomposes and creates nutrients for new growth. For the litter to decompose it needs to be in contact with the ground in order for the microbes in the ground to break down the dead substance. If the litter is suspended in the air it turns a gray color and takes a long period of time to decompose through chemical oxidation. If it is on the ground, litter tends to take on a brownish color and decomposes biologically at a much faster rate. The type of litter present was recorded by color: either gray (oxidizing) or brown litter (decaying).

Big Sagebrush (Artemisia tridentata spp.) Age Estimation

Maximum stem diameter (up to the first 0.30 m of stem) of Big sagebrush plants was measured using calipers (+/-1cm) to approximate the age of each plant (Perryman and Olson 2000) A maximum of four samples were taken at each sample point, one within each quadrant (NW, NE, SE, and SW). The sagebrush plant nearest the plot center within each quadrant was measured using calipers (+/-1cm) and converted to millimeters. The age of each big sagebrush plant was then estimated using the following equation (AGE = 6.1003 + 0.5769 [diameter in mm]).

Photo Points
Digital photos were taken in each of 4 cardinal directions (N, E, S, and W) from the sample point.

RESULTS
Ground Cover Estimates
Based upon ocular estimates, only seven percent of all 2008 field samples (n = 10) had >50 % exposed bare ground and 70% of samples (n = 105) had bare ground exposure <=35 %. The dominant weed present in 100 % of the 2008 samples was cheatgrass. Sixty percent of the sample points had >5% cheatgrass cover where the majority, 98%, were <= 25 % cover and the maximum cover of cheatgrass was 26-35 % with 1.3 % of samples (n = 2) falling within the maximum cover class range.

Based upon transect estimates, the maximum bare ground exposure was 35%, maximum cheatgrass cover was 28%, maximum grass cover was 33%, maximum shrub cover was 59%, and maximum forb cover was 49%.

To truly understand ground cover estimates in relation to grazing treatments, each grazing treatment was independently analyzed. The mean cover classes of each cover type were separated by grazing treatment and are summarized in Table 2.

Table 2. Mean cover class of each cover type separated by grazing treatment.

Cover Class	SHPG Mean Cover Class	Rest-Rotation Mean Cover Class	Total-Rest Mean Cover Class
Bare ground	16-25%	6-15%	1-5%
Shrub	6-15%	6-15%	6-15%
Grass	6-15%	6-15%	6-15%
Litter	16-25%	6-15%	6-15%
Weed	1-5%	6-15%	6-15%
Forb	1-5%	6-15%	1-5%

Ocular estimates were compared with the previous year, 2007. Compared to the 2007 mean cover class, bare-ground exposure has decreased in the Rest-Rotation and the Total-Rest grazing treatments. Both treatment areas seemed to have a rather large decrease as Rest-Rotation moved from a mean cover of 26-35% to 6-15% and Total-Rest moved from 16-25% to 1-5%. Bare ground cover stayed the same in the SHPG area. The mean shrub and weed cover decreased in each treatment. Mean grass only increased in the Rest-Rotation treatement area. There was a decrease in the SHPG area for litter while the other treatment areas remained the same. Forbs decreased in the SHPG area, but had an increase in the Rest-Rotation area, and Total-Rest stayed the same.

To qualitativley visualize how the above changes in mean relate to the overall distribution of each cover class, frequency distributions of each cover class were graphed from 2007 and 2008. The frequency distribution graphs of each grazing treatement from both 2007 and 2008 are shown in figures 2-7.

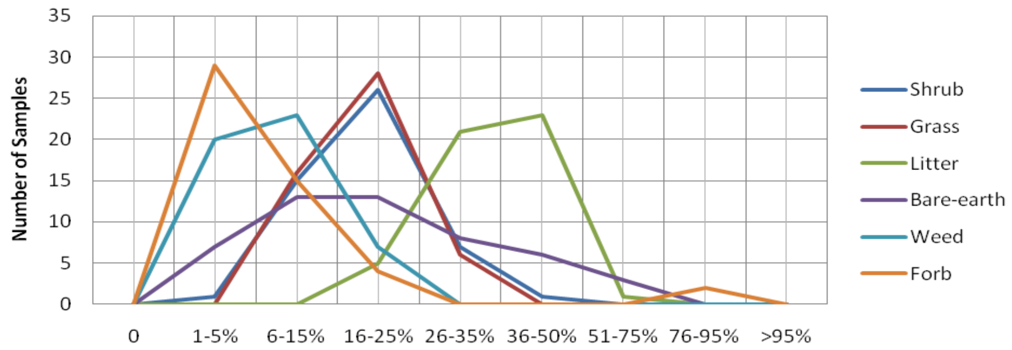

Figure 2. 2007 ground cover estimates in the SHPG grazing treatment. The cover classes are given along the horizontal (x) axis.

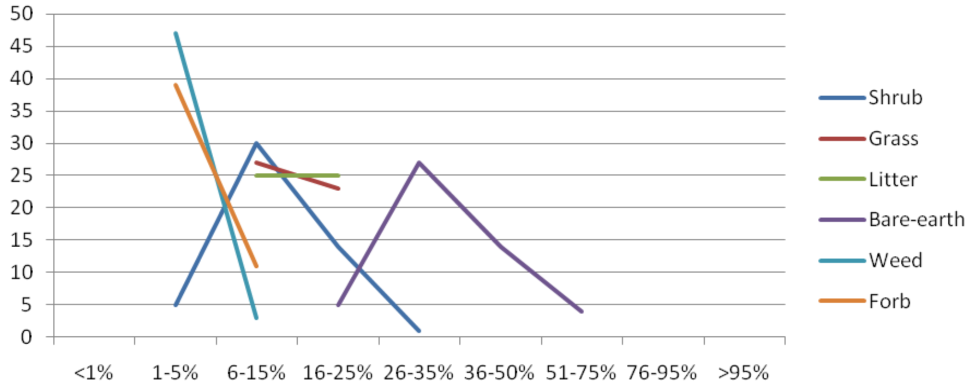

Figure 3. 2008 ground cover estimates in the SHPG grazing treatment. Cover classes are given along the horizontal (x) axis.

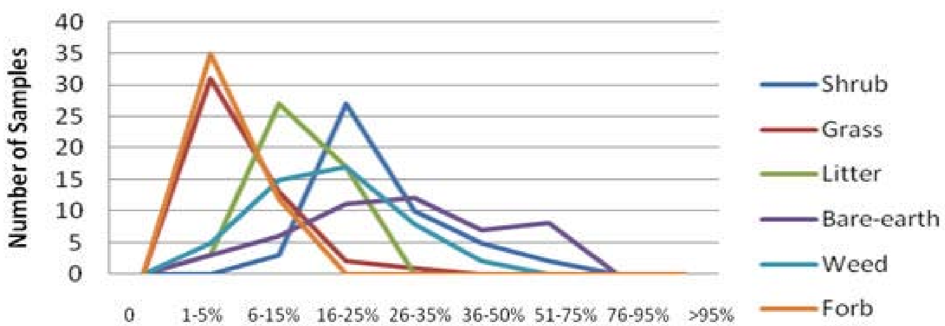

Figure 4. 2007 ground cover estimates in the rest-rotation grazing treatment. The cover classes are given along the horizontal (x) axis.

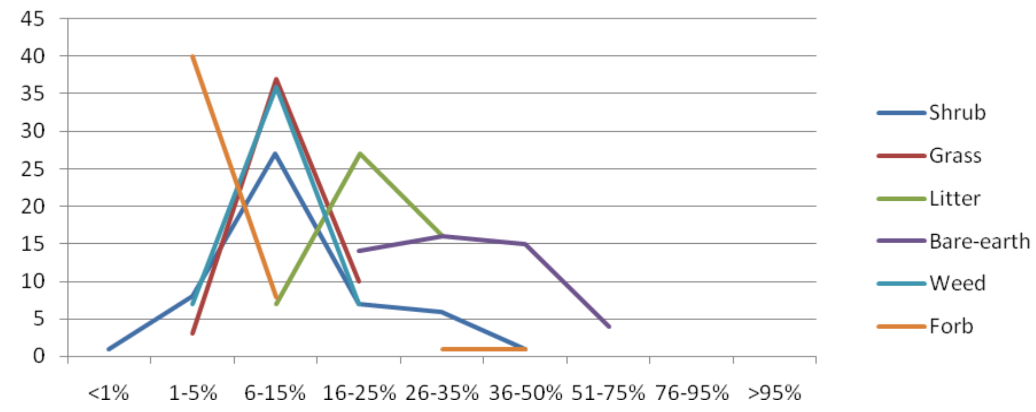

Figure 5. 2008 ground cover estimates in the rest-rotation grazing treatment. The cover classes are along the horizontal (x) axis.

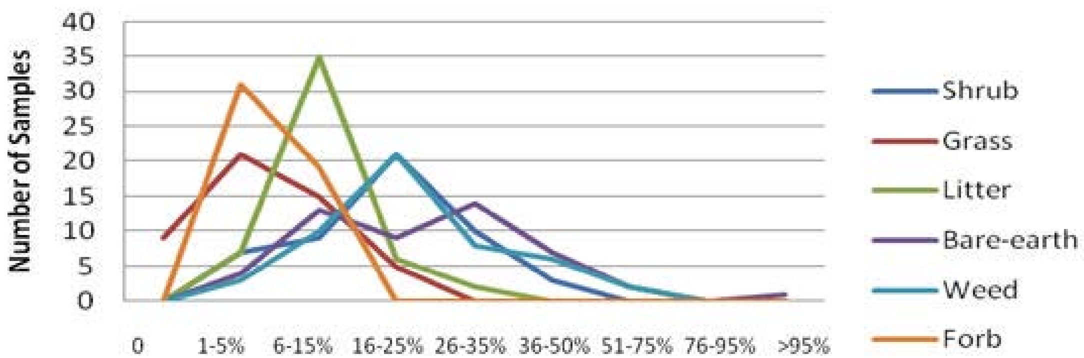

Figure 6. 2007 ground cover estimates in the total rest grazing treatment. The cover classes are given along the horizontal (x) axis.

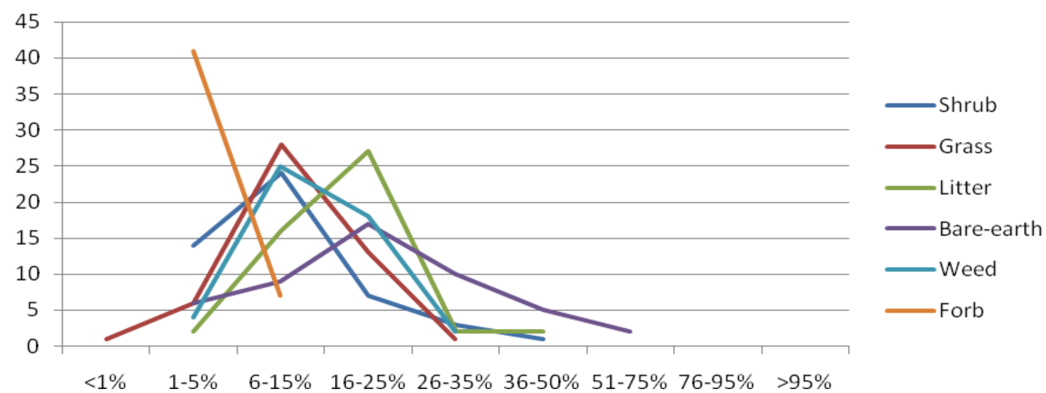

Figure 7. 2008 ground cover estimates in the total rest grazing treatment. The cover classes are given along the horizontal (x) axis.

STATISTICAL ANALYSIS

In order to better understand any differences between vegetation cover within each treatment, the ANOVA test was used. The ANOVA is a simple statistical test which compares varying observations and describes how much the observations differ from the sample mean. The ANOVA test was performed separately for each vegetation class (shrubs, grass, litter, bare ground, weed, and forbs) compared to the same class in the other treatment pastures. The P-Value is the "probability value that describes the likelihood the values tested are from the same population and therefore no different from one another". A P-Value of 1.0 would denote no difference while a P-value less than 0.001 would indicate a conservative difference in comparisons. With this in mind, shrubs, grass, and forbs did not have a significant P-value and no difference was assumed among pastures (Table 3). However, litter, bare ground, and weeds all had P-values well below 0.001. F-test results are also shown with F-value and F-critical values given (Table 3) which corroborate significance for these same comparisons. Looking at the F-critical compared to the F-value in Table 3, the difference is not significant for shrubs, grass, and forb classes. However, a difference was found in litter, bare ground, and weeds with the F-Value being much greater than the F-Critical.

Table 3. Results of Anova test between classes (F critical for this test was 3.058)

Class	P-Value	F-Value
Shrubs	0.230	1.483
Grass	0.003	6.111
Litter	$1.11\ E^{-12}$	33.437
Bare Ground	$1.99\ E^{-14}$	39.460
Weed	$7.45\ E^{-12}$	30.695
Forbs	0.087	2.4844

Included in the ANOVA test was a description of the average, or sample mean, between classes in each grazing treatment (SHPG, Rest Rotation, and Total Rest)(Table 4).

Table 4. Summary of Average (sample mean) between classes in each grazing treatments

Class	SHPG	Rest Rotation	Total Rest
Shrubs	11.1	10.8	13.8
Grass	13.8	8.9	12.2
Litter	18.6	12.1	8.4
Bare Ground	17.5	10.3	5.4
Weed	4.5	12.0	12.3
Forb	5.8	6.3	4.1

Fuel Load Estimation

The majority of field samples (87%; n=130) had fuel load estimates of 2 tons/acre. Four percent (n=6) of the field samples had a fuel load of 4 tons/acre which was primarily due to very dense areas of shrub. The remaining 8.7% (n=13) had fuel load estimates < 2 tons/acre. The occurrence of fuel loads < 2 tons/acre in 10 of the 13 samples were in areas of high lava rock exposure; (>50%) 2 of the samples were not in lava rock areas, but had high bare ground exposure with low shrub cover. The last remaining sample was in an area that was disturbed with low grass and no shrubs.

Forage Measurements

Using AUM Analyzer software (Sheley, Saunders, Henry 1995), forage amount and determined. Mean forage available was 127.44 kg/ha with a standard deviation of 61.16. The minimum forage available was 17 kg/ha and the maximum forage available was 767 kg/ha. Grazing treatments were separated to compare available forage between them (Table 5).

Table 5. A comparison of forage estimates across grazing treatments.

Grazing Treatment	Minimum (kg/ha)	Maximum (kg/ha)	Mean (kg/ha)	Standard Deviation
SHPG	28	186	79.18	24.92
Rest-rotation	17	231	71.86	25.72
Total-rest	34	767	233.41	70.80

Microbiotic Crust Presence

In 2008, 96% of sample points (143 of 149) had microbial crust present. In 2007, 86.4% of sample points (127 of 147) had microbial crust. This change in presence of microbial crust was not significant within a 95% confidence interval.

GAP Analysis

Four GAP classifications were observed in 2008—vegetated lava, sagebrush grassland, bitterbrush, and disturbed. The majority of sample points (61%; *n*=91) were classified as sagebrush grassland, 31.5% (*n*=47) as vegetated lava, 3.4% (*n*=5) as bitterbrush, and 0.6% (*n*=1) as disturbed. Five of the points did not contain data under the GAP classification.

Litter Type

Biologically decaying (brown) litter was dominant at 6.1% (*n*=9) of the sample points while oxidizing (gray) litter was dominant at 4.7% (*n*=7) of the sample points. The remaining 87.9% (*n*=131) of the sample points made no discrimination of dominant litter type and the litter type was classified as "both". Two of the points did not have any litter data recorded.

Big Sagebrush Age Estimation

The mean age of sagebrush plants sampled was 18.19 years (*n* = 149). The minimum age was 10 years and the maximum age was 47 years. Figure 8 shows a frequency distribution of sagebrush age.

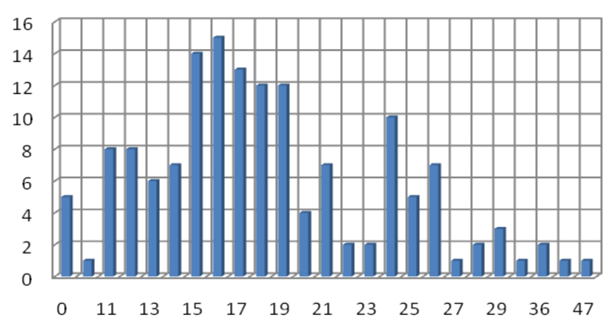

Figure 8. Cumulative frequency graph of sagebrush age estimates (X-axis) at the O'Neal Ecological Reserve, 2008.

CONCLUSIONS

The results from the 2008 field season were interesting when compared with the results from 2007. Figures 2-7 give a visual representation of changes between 2007 and 2008 for each vegetation class separated by treatment pasture. These graphs show a tendency towards a decrease in most cover classes. Weed and shrubs both saw a decrease in all grazing treatments with an increase of grass and forbs seen in the Rest-Rotation treatment area.

The mean forage estimates compared to 2007 saw a general increase especially in the Total Rest pasture. The mean increased from 132.3 kg/ha in 2007 to 233.41 kg/ha in 2008. In the Rest-Rotation pasture the mean increased from 39.47 kg/ha to 71.86 kg/ha in 2008 while the SHPG pasture had similar results increasing from 59.53 kg/ha in 2007 to 79.18 in 2008. The differences observed could be due to effective grazing treatments, but observational bias as well as environmental factors should be noted as possible influences to changes from the previous year. During the sampling process at the O'Neal rain fell consistently throughout the time spent on site. If the grass clippings had absorbed a lot of rain water at the time of weighing, the final weight would have been altered especially if the samples were not thoroughly dried prior to weighing. This factor may be the reason for the large increase in average forage weight from 2007 to 2008. Again, further comparison and sampling will better analyze this trend, and help to conclude if the grazing treatments are effective.

It is important for a land manager to see smaller percentages in bare ground exposure. The Rest-Rotation treatment area as well as the Total Rest area both saw a decrease in bare ground exposure while the Simulated Holistic Planned Grazing allotment kept the same average percent range from 2007 to 2008. Looking at the results from the 2007 study shows there was a decrease in the SHPG treatment from 2006 in overall bare ground exposure. This means the SHPG allotment is moving towards decreased bare ground exposure. On average the percentage remained the same, and it is important to note there was not an increase. If the study were to continue, it would be interesting to learn if these trends will continue towards a decrease in bare ground exposure.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:

Davis, K. and K. T. Weber. 2009. *2008 Rangeland Vegetation Assessment at the O'Neal Ecological Reserve, Idaho*. Pages 23-34 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

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2008 Range Vegetation Assessment at the United States Sheep Experiment Station, Dubois, Idaho

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ABSTRACT

The rangeland vegetation of two summer pastures at the US Sheep Experiment Station (USSES) in Dubois, Idaho was assessed in the summer of 2008. Field measurements were made at 100 randomly generated point locations with 50 sample points taken at each summer pasture. The two pastures sampled at the USSES were the Humphrey and Henniger pastures. Ground cover types, their percent cover, and available forage biomass were estimated within 10m x 10m plots at the 100 locations. Live herbaceous species had the greatest mean percent cover in both pastures. Humphrey held close to 30% average grass cover while the Henniger pasture contained about 29% average grass cover. Mountain big sagebrush was the most commonly seen shrub dominating over half of the sample points in each pasture. Weeds are having little impact on the land with less than 1% found in both the Humphrey and Henniger pastures.

KEYWORDS: Field measurements, forage estimate, ground cover estimate

INTRODUCTION

The 2008 sampling effort focuses upon the Humphrey and Henniger pastures at the U. S. Sheep Experiment Station (USSES) near Dubois, Idaho (Figure 1). The Humphrey pasture consists of 2,600 acres of land near Monida, Montana and is used for spring, summer, and autumn grazing and rangeland research. The Henniger pasture consists of 200 acres of land near Kilgore, Idaho, and is used for summer grazing and rangeland research. Mean annual precipitation (1971 to 2000) at the Dubois Experiment Station (112° 12' W 44° 15'N, elevation, 1661 m) is 331 mm with 60% falling during April through September. Soils are mapped as complexes of Maremma (Fine-loamy, mixed, superactive, frigid Calcic Pachic Argixerolls), Pyrenees (Loamy-skeletal, mixed, superactive, frigid Typic Calcixerolls), and Akbash (Fine-loamy, mixed, superactive, frigid Calcic Pachic Argixerolls) soils on slopes less than 20 percent, but mostly 0 to 12 percent (NRCS 1995). Vegetation on the study sites are sagebrush-grassland communities dominated by mountain big sagebrush (*Artemisa tridentata* ssp. *vaseyana* [Rydb.] Beetle) and threetip sagebrush (*A. tripartita* Rydb.).

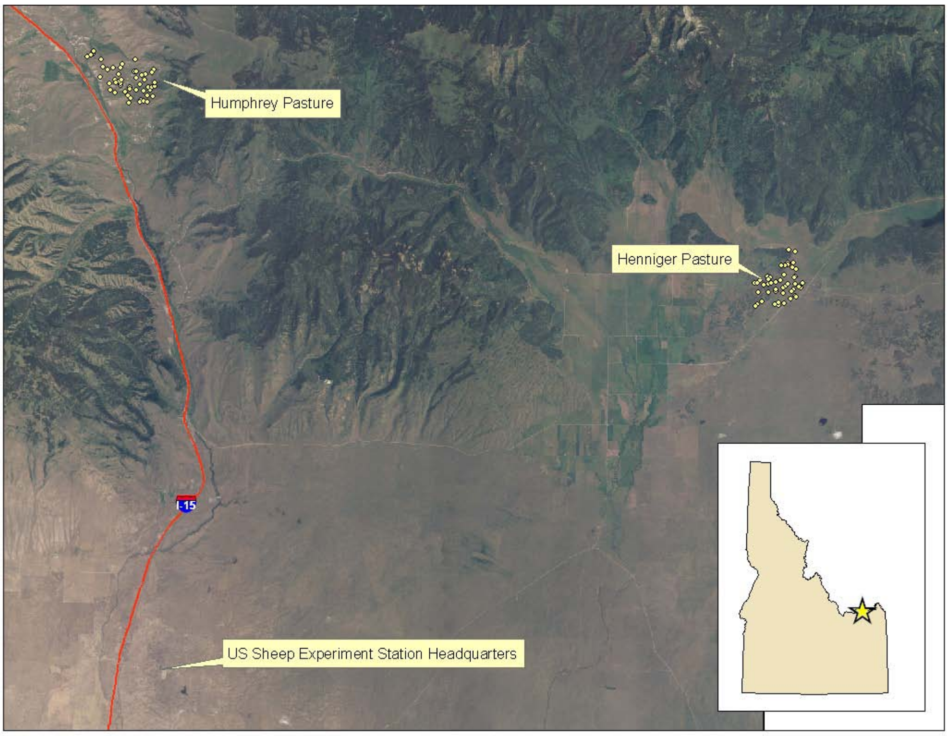

Figure 1. US Sheep Experiment Station, Humphrey and Henniger pastures.

The objectives of this study were to: 1) assess the rangeland vegetation at the Henniger and Humphrey pastures using LANDSAT and SPOT satellite imagery and field measurements and 2) compare the rangeland vegetation assessment with similar assessment performed at the Tsakhiriin Tal area of the Darkhad Valley, Mongolia. The field-based measurements of the USSES pasture vegetation assessment were performed in late July to early August of 2008. The results of the field-based measurements are presented in this document and will be later combined with satellite imagery collected during the summer of 2008.

METHODS

A total of 100 random points were generated within the Humphrey and Henniger pasture sites prior to field assessment. Each point represented a sample location, at which field measurements were made within 10m x 10m plots. The plots were centered at each random point and the edges of the plots were aligned in the cardinal directions. Four digital photographs were taken at each plot in each of the four cardinal directions. These were taken first to avoid photographs containing disturbances to the land that may have been caused while the researchers gathered the information. The field measurements included ground cover estimation and forage biomass measurement. Ground cover estimations were made describing percent cover of bare soil, rock >75mm, litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed. Percent cover estimates were made along two 10m line transects perpendicular to each other and crossing at the center of the plot at 5m of each line transect. This was done using a point-intercept method (Gysel and Lyon 1980). Records were made every 20cm along each 10m line, beginning at 10 cm and ending at 990 cm, to indicate the cover type at the point using ocular estimates (n = 50 points for each line and 100 points for each plot).

Litter refers to biomass that is on the ground and in contact with the ground. Live herbaceous species refers to live (i.e., green) forbs and grasses, while live shrubs include all species of shrubs. The dominant shrub species were noted in each sample point.

Forage biomass was measured four times at each sample plot using a plastic coated cable hoop 2.36 m in circumference, or 0.44 m^2. The hoop was randomly tossed into each of the four quadrants (NW, NE, SE, SW) that were made from the transect lines. All green and senescent herbaceous biomass was clipped and weighed in a paper bag using a Pesola scale tared to the weight of an ordinary paper bag. All grass species were considered forage. The measurements were then used to estimate forage amount in AUM's, pounds per acre, and kilograms per hectare (Sheley et al. 1995).

RESULTS

The most common ground cover type at the Henniger pasture was live herbaceous species, or grass, with a mean estimate of 29% cover (Figure 2). The second most common ground cover type was live shrubs which made up 23% cover on average. The most common ground cover type at the Humphrey pasture was also grass species with a mean estimate of 30% cover (Figure 2). The second most common ground cover type was shrub which made up 21% cover on average. Bare soil cover was only a little more common at the Henniger pasture than at the Humphrey pasture, comprising of 14% and 13% respectively. Forb cover class was again more common at the Henniger pasture, but only by a small amount. Henniger comprised of 17% mean forb cover while the Humphrey pasture contained 15% mean forb cover. Litter had an average higher percentage in the Humphrey pasture with 14% and less in the Henniger pasture with 10% average cover. Rock was seen in <1% on average in the Henniger pasture while the Humphrey pasture had just barely over 1% average cover. Standing Dead Herb and Standing Dead Wood both had a very low average cover (between 1-5%) in the Humphrey and Henniger pastures. In both pastures weed had an average cover of <1% (Figure 2).

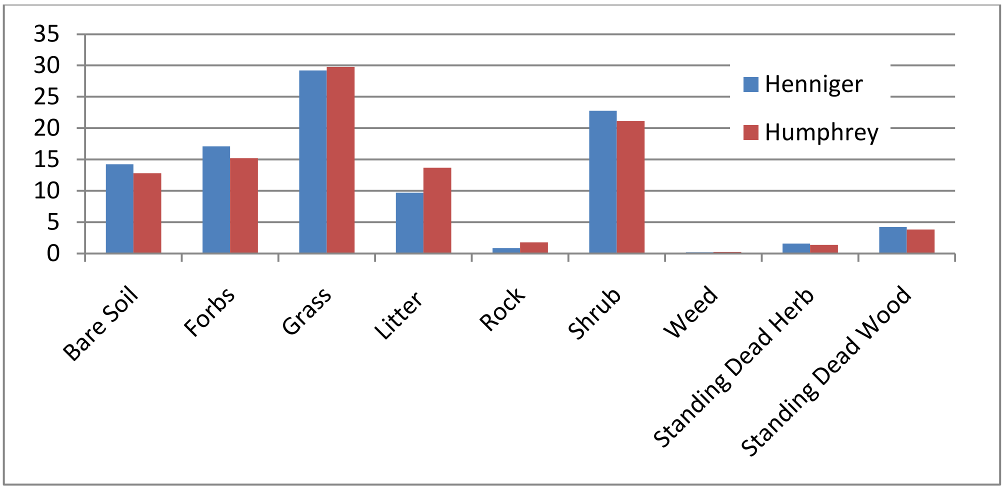

Figure 2. Mean percent cover of all primary ground cover types at the Henniger and Humphrey pastures in the summer of 2008.

CONCLUSIONS

The two pastures seemed very similar when comparing average shrub cover. Grass, litter, and rock had higher percent cover in the Humphrey pasture while bare soil, forbs, and shrub had higher percentages in the Henniger pasture. These differences in percentages were not significant though (Figure 2). In the summer of 2007, a similar assessment occurred at both the Henniger and Humphrey pastures (Figure 3). When compared with the sampling from 2008 it is easy to see differences between the two summers. Sample points were randomly generated both years, so it should be noted that the varying results may be due simply to the placement of sample points. In addition, environmental factors as well as observational bias should be noted as other possible influences to account for the observed changes from the previous year. In the Henniger pasture grass and forbs both saw a fairly significant increase while the Humphrey pasture saw an increase in bare soil and litter. Bare ground exposure is considered detrimental, so this change is a negative result of this sampling. This is only the second year spent sampling this type of data in the Humphrey and Henniger pastures, so by further sampling and comparison, a better analysis could be made to see if this trend continues.

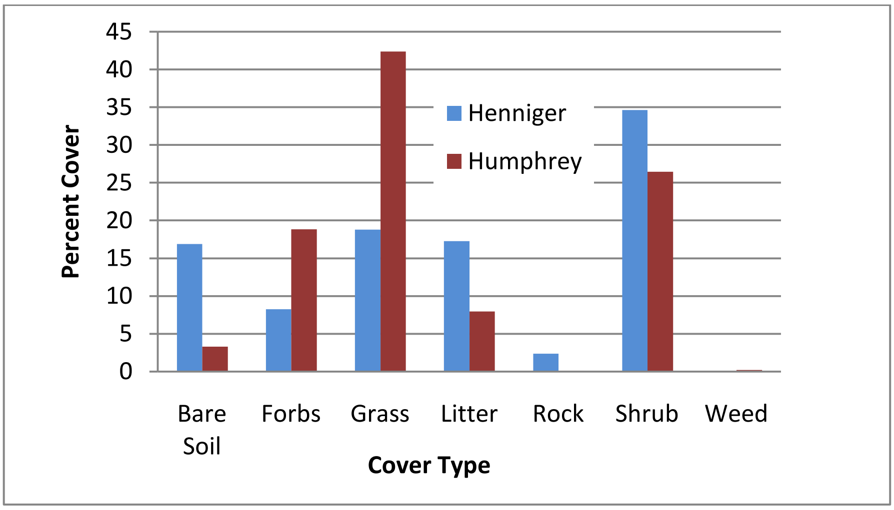

Figure 3. Mean percent cover of all primary ground cover types at the Henniger and Humphrey pastures in the summer of 2007. Percent Standing Dead Wood, and Percent Standing Dead Herbaceous were <1%.

Cheatgrass is a weed that is invasive to rangelands of the Intermountain West (Colorado State University 2008. It is good to note that none of the samples in either site contained cheatgrass. This is a positive result that also suggests the need for continued sampling. Weeds observed in the pastures tended to be Canada thistle (*Cirsium arvense* (L.) Scop) which was found at very low cover (<1%) in both 2007 and 2008. Though an increase in bare soil was seen in the Humphrey pasture from 2007 to 2008, the percentage still remains low (Figure 2 and 3). A low percent exposure of bare soil and weeds, but high percent cover of live herbaceous species (grass, forbs, and shrubs) suggests the Humphrey and Henniger pastures are currently in a good rangeland state.

ACKNOWLEDGEMENTS
This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Range Vegetation Assessment in the Big Desert, Upper Snake River Plain, Idaho 2008

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ABSTRACT

Vegetation data was collected at 99 randomly located sample points between June 10 and July 11, 2008 in the US Department of Interior, Bureau of Land Management Big Desert Region. Data was collected describing the 1) percent cover of grasses, shrubs, litter, and bare ground, 2) dominant weed and shrub species, 3) fuel load, 4) sagebrush age, 5) GAP land cover classification, 6) presence of microbial crust, 7) litter type, 8) forage availability, and 9) photo points. Sample points were stratified by fire and grazing treatments. An analysis of these data including a comparison of burned and unburned areas (based upon the 2006 Crystal Fire boundary) indicate that in the two years following the fire there has been a slight reduction in bare ground exposure with an increase in weed and litter ground cover. Biomass measurements, however, indicate a continual decline in available forage. This decline can also be seen in the reduction of grass as a ground cover. The percent cover of shrubs inside the fire boundary was equivalent to that of the previous year and continues to be less than that of pre-fire conditions.

KEYWORDS: sampling, GIS, remote sensing, GPS

INTRODUCTION

Sagebrush steppe is an extensive and important range cover type in North America extending over 400,000 km^2 of the Columbia and Snake River Plateaus (Anderson and Inouye, 2001). To conserve and manage these rangelands, it is essential to have a clear understanding of their ecological processes, functions, and the mechanisms that drive change. The mechanisms or primary drivers of land cover change in rangeland ecosystems include fire, invasive weeds, and urbanization.

Within the sagebrush steppe environment, wind erosion hazard is high, vegetation is dry, and perennial vegetation recovery rates are slow. Consequently, wildland fire is the driver considered most destructive in this ecosystem. Yet fire frequencies have increased due to the introduction of non-native, less palatable, and more readily combustible grasses such as cheatgrass (*Bromus tectorum*) (Anderson and Inouye, 2001). Following the 2006 field season, the Crystal Fire spread over the Big Desert study area and burned approximately 89,000 hectares. The data collected in 2007 and 2008 from the burned sites and immediately adjacent unburned sites describe various trends in post-fire recovery.

The purpose of this study was to collect field data from the Big Desert rangeland area (managed by the USDI BLM) and compare these data to results from previous research in land cover change and rangeland health modeling from the same study area, namely the Big Desert of southeastern Idaho. This study follows seven sequential annual studies of the same area (2000 to 2007) (Anderson et al, 2008; Gregory et al., 2008; Russell and Weber, 2003; Sander and Weber, 2004; Underwood et al, 2008; Weber and McMahan, 2005). The data collected in 2008 and the previously collected data illustrate various trends in shrub, litter, bare ground, and grass cover in response to fire and or other drivers of land cover change.

METHODS

The study area, known as the Big Desert, lies in southeastern Idaho, approximately 71 km northwest of Pocatello. The center of the study area is located at 113º 4’ 18.68” W and 43º 14’ 27.88” N (Figure 1) and is managed by the Bureau of Land Management (BLM).

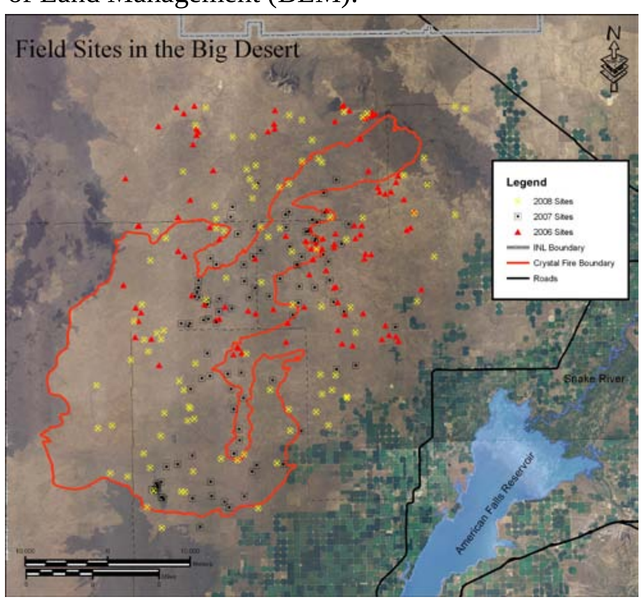

Figure 1. The crystal fire and sampling points from 2006, 2007, and 2008.

The Big Desert is a sagebrush-steppe semi-arid desert containing a large variety of native species as well as invasive species. Geologically young lava formations border the area to the south and west. Irrigated agricultural lands border the study area to its north, south and east. The area has a history of livestock grazing and wildfire occurrence.

The random point generation tool from Hawth's Analysis Tools was used to generate random sample points (*n*=99) across the study area (Figure 1) (Beyer, 2004). The limiting criteria for point selection was a distance of >70 meters from a road, trail, or fence line (to avoid edge effects), and < 750 meters from a road to aid researchers in navigating to sample points on foot.

Each point was navigated to and the location of the point was recorded using a Trimble GeoXH GPS receiver using latitude-longitude (WGS 84). Points were occupied until a minimum of 60 positions were acquired and Wide Area Augmentation System (WAAS) was used whenever available. All points were post-process differentially corrected (+/-0.20 m with a 95% CI) using an array of southeastern Idaho continuously operation reference stations (CORS) each located <80 km of the study area. All sample points were projected into Idaho Transverse Mercator NAD 83, using ESRI's ArcGIS (Gneiting, et al., 2005).

At each sample point, the area equivalent to a single SPOT5 pixel (10 mpp or approximately 100 m^2) centered over the sample point was examined to estimate the percent of bare ground exposure, vegetation ground cover, fuel load, and forage. Theses estimations and other descriptive characteristics such as the presence of microbiotic crust, the type of decaying litter, GAP land cover class, sagebrush age, and field photographs were recorded using an ArcPad Application installed on the Trimble GeoXH GPS receiver.

Ground Cover Estimation

An ocular estimate of the percent of ground cover in each 10 x 10m area was made and used to classify cover into one of nine classes (None, 1-5%, 6-15%, 16-25%, 26-35%, 36-50%, 51-75%, 76-95%, and >95%). Researchers view the vegetation perpendicular to the earth's surface, to emulate the satellite perspective, and discuss the percent cover/exposure for each of the following; bare ground, litter and duff, grass, shrub, and dominant weed.

Fuel Load Estimation

Fuel load classes at each sample point were based on the types and quantities of vegetation found in the area. These classification groups (Table 1) based on earlier works of Hal Anderson (1982) were used to estimate the fuel load in the study area.

Table 1. Fuel load classes and associated tonnage of fuels (from Anderson 1982)

Fuel Load Class	(Tons/Acre)	General Description
1	0.74	Almost bare ground, very little vegetation
2	1.00	Grasses, some bare ground, few shrubs
3	2.00	Mixture of shrubs and grasses
4	4.00	Predominantly shrubs
5	>6.00	Shrubs to trees

Forage Measurement

To determine the amount of available forage, the AUM Analyzer method was used (Sheley et al. 1995). A plastic coated cable hoop 2.36 meters in circumference, or 0.44 m^2 was randomly tossed into each of four quadrants (NW, NE, SE, and SW) centered over the sample point. All grass species within the hoop considered forage for cattle, sheep, and wild ungulates was clipped. This grass was weighed (+/-1g) using a Pesola scale tared to the weight of an ordinary paper bag. The measurements were used to estimate forage amount in AUM's, pounds per acre, and kilograms per hectare.

Microbiotic Crust Presence

Microbiotic crusts (Johnston 1997) are formed by living organisms and their by-products, creating a surface crust of soil particles bound together by organic materials. These are common in very poor rangelands and are often one of the last organisms left alive during drought conditions. The $100m^2$ area centered over the sample point is examined for the presence of microbiotic crust. Any trace of a microbiotic crust was defined as “presence” and recorded in the database as a Boolean true value.

GAP Analysis

Further description of the plant communities surrounding each sample point was made following GAP Analysis Project land cover descriptions (Jennings 1997). The GAP vegetation description that most closely described the sample point was recorded. In addition, a 30-meter radius around each point was viewed to determine if the surrounding area contained the same plant communities (homogeneous) or if there was a difference in vegetation land type (heterogeneous).

Litter Type

Litter was defined as any biotic material that is no longer living. Litter decomposes and creates nutrients for new growth. For the litter to decompose it needs to be in contact with the soil in order for the microbes in the soil to break down the dead substance. If the litter is suspended in the air, it turns a gray color and takes an immense amount of time to decompose through chemical oxidation. If it is on the ground, it is a brownish color and decomposes biologically at a much faster rate. The type of litter present was recorded as gray (oxidizing) or brown (decaying).

Big Sagebrush (Artemisia tridentata spp.) Age Estimation

Maximum stem diameter of big sagebrush plants were used to approximate the age of each plant (Perryman and Olson 2000). A maximum of four samples were taken at each sample point, one within each quadrant (NW, NE, SE, and SW) centered over the sample point. The sagebrush plant nearest the plot center within each quadrant was measured using calipers (+/-1cm) and estimated to millimeters. The age of each big sagebrush plant was then estimated using the following equation (AGE = 6.1003 + 0.5769 [diameter in mm]).

Photo Points

Digital photos were taken in each of 4 cardinal directions (N, E, S, and W) from the sample point.

RESULTS AND DISCUSSION

Percent Cover Bare Ground, Litter, Weed, Grass, and Microbiotic Crust

Only seven percent of all 2008 Big Desert field samples had >50% exposed bare ground. This is a substantial reduction from the 28% of all samples reported in 2007. This trend towards a reduction in bare ground was consistent both inside and outside the Crystal Fire Area. Of the 2006 Big Desert field samples, all samples showed < 50% exposed bare ground (Figure 2).

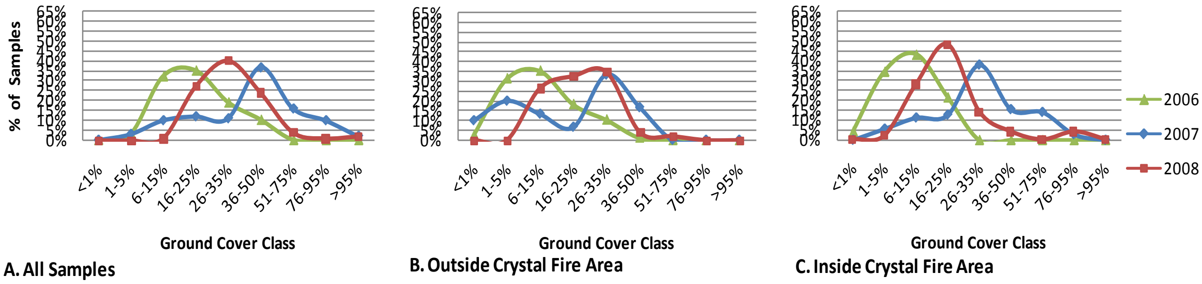

Figure 2. Bare ground exposure estimated in 2006-2008 for all samples (A), samples outside the fire perimeter (B), and samples taken inside the fire perimeter (C).

Fifty-nine percent of the samples collected in 2008 had litter in the 16-25% cover class. This is an increase in litter cover since the 2007 data collection (Figure 3).

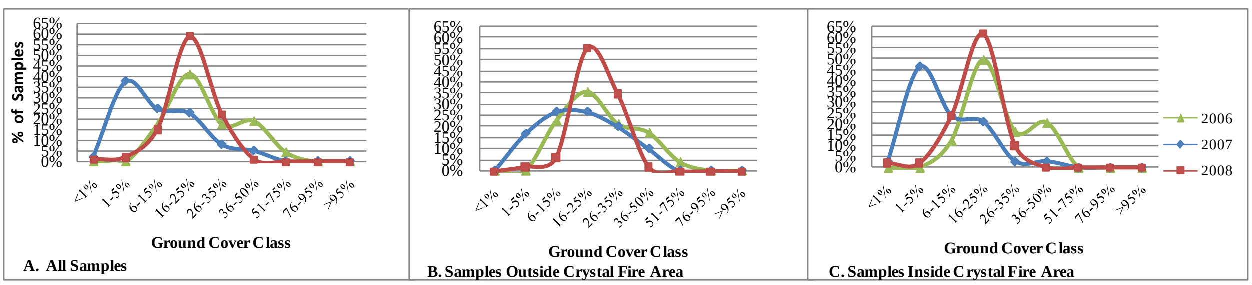

Figure 3. Litter cover estimated in 2006-2008 for all samples (A), samples outside the fire perimeter (B), and samples taken inside the fire perimeter (C).

Cheatgrass was present at 76% of all points sampled. Canada thistle (*Cirsium arvense*) was considered the dominant weed at 6% of all sample points. It is noted that Canada thistle had not been cited as a dominant weed during previous studies. Fifty-four percent of all 2008 sample points had >5% weed cover. This trend held true both inside and outside the Crystal Fire Area with >5% weed cover found at 51% of sample points outside the fire perimeter and at 56% of sample points inside the fire perimeter (Figure 4).

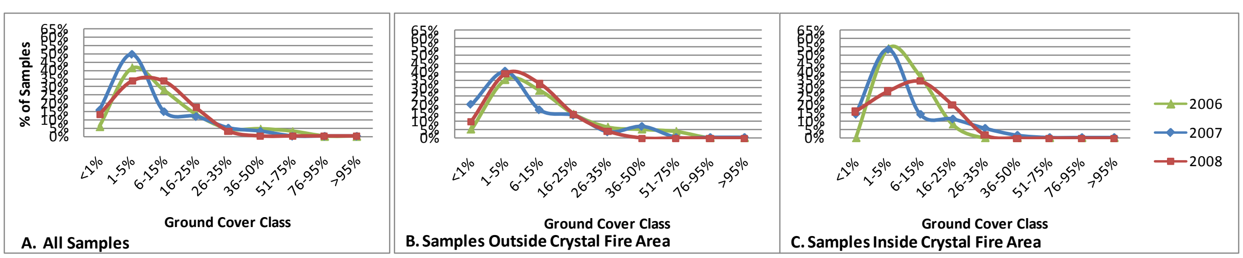

Figure 4. Weed cover estimated in 2006-2008 for all samples (A), samples outside the fire perimeter (B), and samples taken inside the fire perimeter (C).

At all 2008 sample sites, grass cover was < 36%. This was true both inside and outside the fire perimeter. Outside the fire perimeter, the most common cover class was 6 to 15%, while within the fire area the fire perimeter the most common cover class was 16 to 25% (Figure 5). The absence of samples with > 35% grass cover suggests a reduction in grass cover.

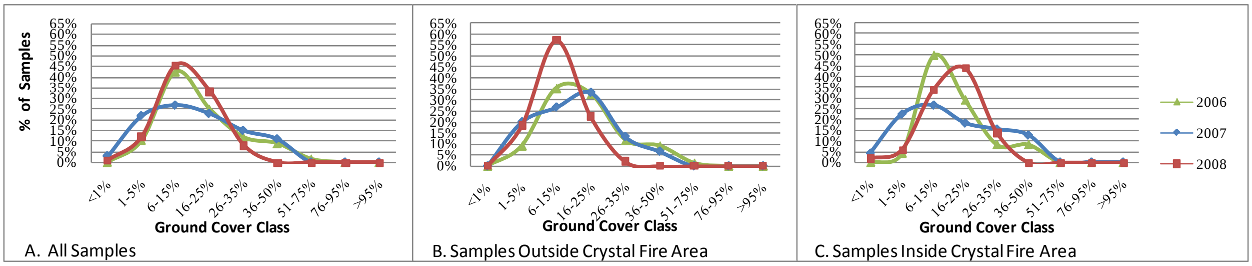

Figure 5. Grass cover estimated in 2006-2008 for all samples (A), samples outside the fire perimeter (B), and samples taken inside the fire perimeter (C).

Microbiotic crust was present at 27% of the points sampled while in 2007, microbiotic crust was present at only 10% of the points sampled.

Big Sagebrush Age Estimation

The mean age of sagebrush was 13.9 years (n = 91). The minimum age was eight years and the maximum age was 29 years.

Forage Measurements

The mean forage at the Big Desert study area was 252 kg/ha (Table 2). This mean value was lower than found both in 2006 and 2007 (461 and 362 kg/ha, respectively). Mean forage in 2008 inside the Crystal Fire perimeter was 296 kg/ha while 208 kg/ha was found outside the Crystal Fire perimeter (Table 2). The larger quantity of forage inside the Crystal Fire perimeter may be a function of the grazing restriction in place following the fire.

In 2007, sampling was done in the late spring (May 29 to June 13); but the 2008 sampling was done a little later in the summer (June 10 to July 11). Although, the sampling during a hotter and drier season may have biased the samples towards a lighter weight; the decreased forage in 2008 is consistent with the values seen for grass coverage (Figure 4).

Table 2. Forage measurements in the Big Desert 2006-2008 including a comparison of 2008 forage estimates both inside and outside the Crystal fire perimeter.

Big Desert Forage	*2006*	2007	2008	2008 Inside Crystal Fire Boundary	2008 Outside Crystal Fire Boundary
KG_HA					
Mean	460.6	361.9	251.7	296.0	208.4
Standard Error	32.3454	31.2413	22.1097	31.4591	30.1141
Median	383	259.1810	185.9342	239.4607	135.2249
Mode	208	0	124	152	124
Standard Deviation	323.4539	313.9716	217.7548	217.9548	210.7984
Sample Variance	104622.4444	98578.1696	47417.1711	47504.3086	44435.9672
Kurtosis	2.2170	0.5010	3.1396	1.5217	7.0615
Skewness	1.3496	1.1103	1.8212	1.3346	2.6119
Range	1617	1301.54	1014.19	963.48	997.28
Minimum	51	0	11.27	11.27	28.17
Maximum	1668	1301.54	1025.46	974.75	1025.46
Sum	46060	36555.79	24419.36	14209.88	10209.48
Count	100	101	97	48	49

CONCLUSIONS

Sampling results show some recovery following the Crystal fire of 2006 (i.e., estimated variables have moved closer to pre-fire conditions). However, current state and transition models suggest that a landscape may not return to pre-fire conditions (e.g., climax community) but rather a different condition that is equally stable (stable-state). Following the Crystal Fire, grazing has been restricted within the fire perimeter; however, available forage is still reduced relative to pre-fire conditions. Neither the median nor modal values for forage indicate a return to pre-fire status. Despite seeding (crested wheat grass [*Agropyron pectiniforme*]) inside the Crystal Fire perimeter, grass cover continues to be less than that of pre-fire conditions (Figure 5 and Table 2).

Shrub cover has also not returned to pre-fire status (Figure 6). The graph illustrating the samples collected inside the Crystal Fire perimeter (Figure 6C) indicates little change from 2007 to 2008; the dominant shrub in both years has been Green Rabbitbrush (*Chrysothamnus viscidiflorus*) which was expected as rabbitbrush is quick to colonize following fire.

Further comparisons of pre-fire and post-fire sampling data (Figure 2) show a reduction of bare ground both inside and outside the Crystal Fire area since 2007. Inside the Crystal Fire perimeter, the reduction of bare ground is a trend that appears to be returning to pre-fire status (Figure 2 C) with ground cover increases attributed to increases in weed and litter cover (Figure 3 and Figure 4).

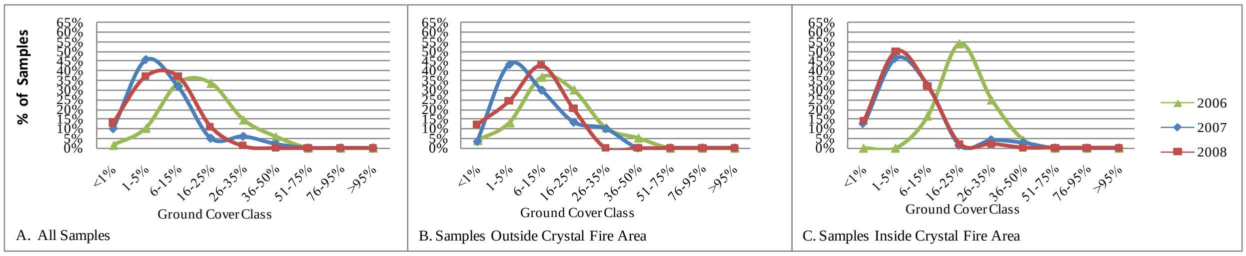

Figure 6. Shrub cover estimated in 2006-2008 for all samples (A), samples outside the fire perimeter (B), and samples taken inside the fire perimeter (C).

Analysis of the Big Desert study area vegetation sampling from 2006, 2007, and 2008 illustrates an effect of fire on shrubs and vegetation ground cover. These results suggest that following the Crystal Fire of 2006 vegetation ground cover has not yet recovered relative to pre-fire conditions.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Tedrow, L, K. Davis, and K. T. Weber. 2009. *Range Vegetation Assessment in the Big Desert, Upper Snake River Plain, Idaho 2008*. Pages 41-50 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

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2008 Range Vegetation Assessment in the Darkhad Valley, Mongolia

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ABSTRACT
The rangeland vegetation of the Tsakhiriin tal study site in the Darkhad Valley of north-western Mongolia was assessed in the summer of 2008. Field measurements were made at 100 randomly generated point locations throughout the study site. Ground cover types, their percent cover, and available forage biomass were estimated within 10m x 10m plots at the 100 locations. Live herbaceous species and litter had the greatest mean percent cover, while rock, weeds, and shrub cover types were estimated to have a mean of less than 0% cover. Available forage biomass estimates were 1086 lbs per acre and 1218 kg per hectare in the Tsakhiriin tal study site. The observed patterns were consistent with the expected trends in the Darkhad Valley rangelands that are continuously grazed throughout the growing season.

KEYWORDS: Field measurements, forage estimate, ground cover estimate

INTRODUCTION

Mongolia, a continental semi-arid country, is known as one of the five most heavily grazed places in the world (Asner et al., 2004). All grazing lands in Mongolia are public lands, although the herds are privately owned by nomadic herders who migrate at least four times a year between seasonal pastures. Darkhad Valley in north-western Mongolia is grazed by several different livestock species: cattle, sheep, goats, and horses. The Tsakhiriin Tal area in the Darkhad Valley, the focus area of this study (Figure 1), is primarily used as summer pasture by approximately 30 nomadic households. Rangeland wildfires are very rare in the Darkhad Valley potentially due to low fuel accumulation associated with continuous grazing use. The isolated small forest stands along the boundary of this study site (Figure 1) are also likely excluded from fire disturbance, although the continuous, expansive forest stands surrounding the Darkhad Valley might have higher fire frequency (Sankey et al., 2006).

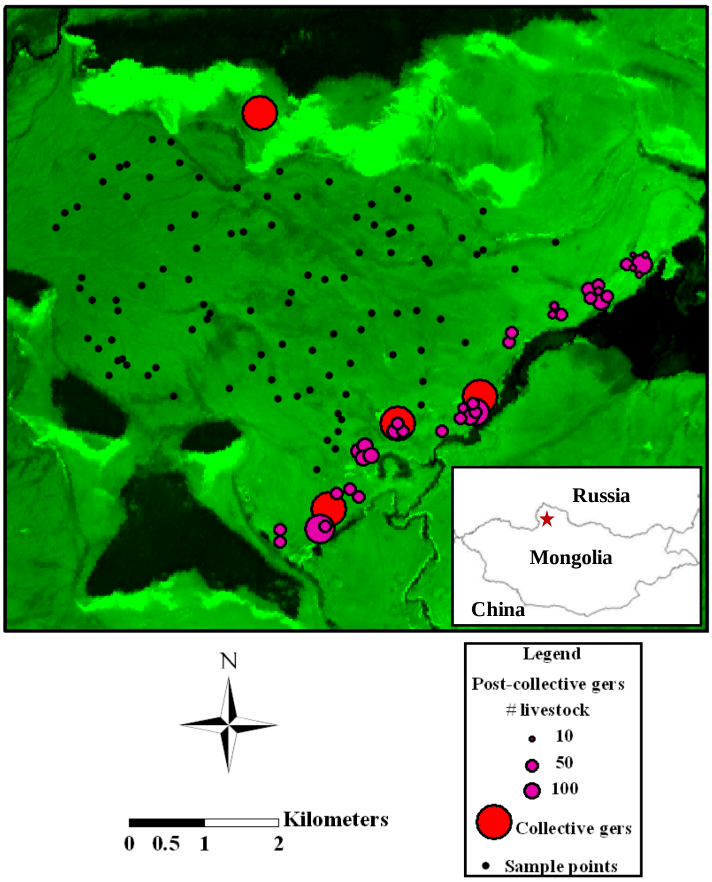

Figure 1. Mongolia and the Tsakhiriin tal study site in the Darkhad Valley. The point locations in the Tsakhiriin tal indicate 100 randomly generated points at which field-based measurements were made in the summer of 2008.

The objective of this study is to assess the rangeland vegetation in the Tsakhiriin tal area of the Darkhad Valley using SPOT satellite imagery and field measurements. The field-based measurements of the Tsakhiriin tal rangeland vegetation assessment were performed in early-mid July, 2008. The results of the field-based measurements are presented here and will be later combined with satellite imagery analysis results.

METHODS

Prior to field assessment, a total of 100 random points were generated within the Tsakhiriin tal study site using Hawth's tool in ArcMap 9.1 software. Each point represented a sample location, at which field measurements were made within 10m x 10m plots. The plots were centered at each random point and the edges of the plots were aligned in the cardinal directions. Four digital photographs (in the cardinal directions) were taken at each plot to record the general characteristics of each point at a landscape scale (Only one photograph was taken at some plots due to limited memory space on the digital camera). The field measurements included ground cover estimation and forage biomass measurement. Ground cover estimation included estimates of percent cover of bare soil, rock (coarse fragments >75 mm), litter, herbaceous standing dead, dead standing wood, live herbaceous species, live shrubs, and dominant weed. Percent cover estimates were made along two 10 m line transects, perpendicular to each other and crossing at the center of the plot at 5m of each line transect, using a point-intercept method. Records were made every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm, to indicate the cover type at the point (n = 50 points for each line and 100 points for each plot).

Litter refers to biomass that is on the ground and in contact with the ground. Live herbaceous species refers to live (i.e., green) forbs and grasses, while live shrubs include all species of shrubs. Forage biomass was measured in four cable hoops 93 inches in circumference and 0.44 m^2 in area. The hoops were tossed randomly in each of the four quadrants of each plot. All green and senescent herbaceous biomass was clipped and wet-weighed in a paper bag using a spring scale.

RESULTS

The most common ground cover type was live herbaceous species with a mean estimate of 49.04% cover (Figure 2). The second most common ground cover type was litter, which made up 41.31% cover on average. Bare soils made up 9.08%. Other ground cover types of standing dead herbaceous, rock, shrub, and weed comprised less than 0% cover (Figure 2).
Plot-level averages of wet forage biomass ranged between 26-144 gr per hoop. The average wet forage biomass at this study was 54.03 gr per hoop. These estimates translate to average forage biomass of 1086 lbs per acre and 1218 kg per hectare.

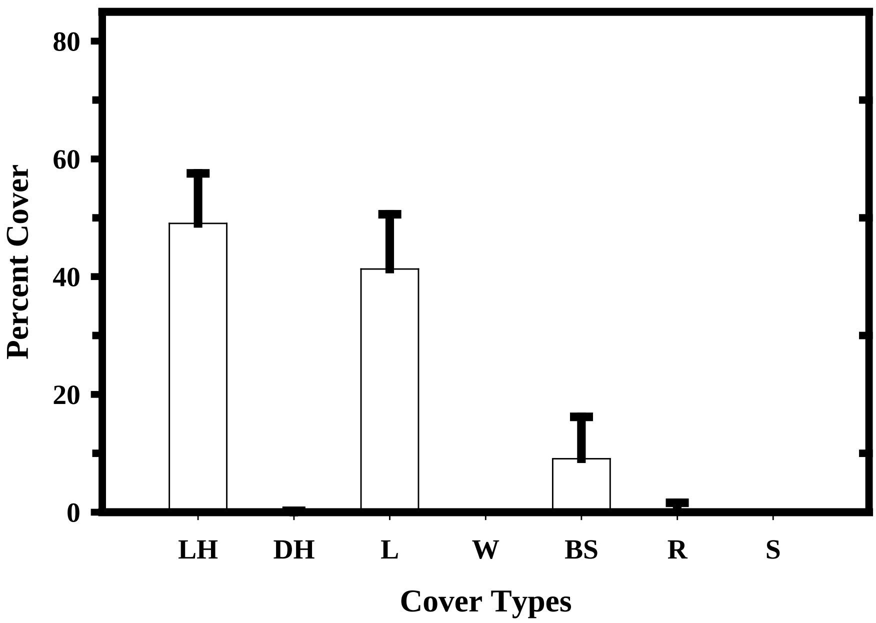

Figure 2. Mean (+SE) percent cover of all ground cover types (LH=Live Herbaceous, DH=Standing Dead Herbaceous, L=Litter, W=Weeds, BS=Bare soil, R=Rock, S=Shrubs) at the Tsakhiriin tal study site in the summer of 2008.

CONCLUSIONS

The observed ground cover types and their estimated percent cover were similar to the expected trends in the Darkhad Valley grasslands. Standing dead herbaceous species was estimated to have low percent cover, which might be expected in such continuously grazed areas. Low percent cover of bare soil and weeds, but high percent cover of live herbaceous species might suggest that this area is not overgrazed, although it is grazed continuously throughout the growing season. Shrubs were not found in any of the plots in the Tsakhiriin tal study area. This was consistent with the observed patterns in the Darkhad Valley, where shrubs are present only in ungrazed riparian areas.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:

Sankey, T. T. 2009. *2008 Range Vegetation Assessment in the Darkhad Valley Mongolia*. Pages 51-54 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

Modeling Bare Ground with Classification Trees in Northern Spain

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ABSTRACT
Bare ground abundance is an important rangeland health indicator and its detection is a fundamental part of range management. Remote sensing of bare ground may offer solutions for land managers but also presents challenges as modeling in semi-arid environments usually involves a high frequency of spectral mixing within pixels. Classification Tree Analysis (CTA) and maximum likelihood classifiers were used to model bare ground in the semi-arid steppes of the middle Ebro valley, Aragon, Spain using Satellite Pour l'Observation de la Terre 4 (SPOT 4) imagery and topographic data such as elevation, slope, aspect, and a morphometric characterization model. A total of 374 sample points of bare ground fraction from sixteen 500m transects were used in the classification and validation process. Overall accuracies were 85% (Kappa statistic = 0.70) and 57% (Kappa statistic = 0.13) from the CTA and maximum likelihood classifiers, respectively. While spectral attributes were essential in bare ground classification, the topographic and morphometric properties of the landscape were equally critical in this modeling effort. Although the specific layers best suited for each specific model will vary from region to region, this study provided an important insight on both bare ground modeling and the potential advantages of CTA.

KEYWORDS: Remote sensing, GIS, rangelands, Classification Tree Analysis, desertification

INTRODUCTION

Rangeland ecosystems cover approximately 40% of the earth's terrestrial surface (Huntsinger and Hopkinson 1996, Branson et al. 1981) and are typically dominated by grass and shrub communities. These vegetation communities exist because of the semi-arid or xeric nature of these sites. However, an effective hydrologic cycle (the capture, storage, and release of water) leads to healthy rangeland sites that produce green biomass (at least ephemerally) with minimal bare ground. The green biomass is effectively used by herbivores (e.g., livestock) which are an integral part of a functional rangeland ecosystem. When the hydrologic cycle is disturbed, rangelands desertify and as a result, exhibit increasing amounts of bare ground exposure. Chronic desertification shifts lead to a loss of ecosystem functionality, a reduction in biodiversity, and reduced livestock grazing capabilities (Daubenmire 1959, Schlesinger et al. 1990) with associated social and economic underpinnings (Savory 1999, Arnalds and Archer 2000, Griffin et al. 2001).

The degree of bare ground is a reliable indicator of rangeland health within otherwise similar regions (National Research Council 1994, Whitford et al. 1998, Pyke et al. 2002, O'Brien et al. 2003, Hunt et al. 2003, Booth and Tueller 2003). One of the consequences of sedenterization of livestock is the exceedingly high loss of plant cover and plant biomass. Although stocking rate can be relatively low, the way livestock use the landscape may have important consequences on triggering land degradation processes. Indeed, in spite of an average reduction of stocking rate in many areas of the world, recent increases in animal number per farm is leading to higher degradation around shelters (Alados et al. 2006).

Remote sensing provides a means to detect bare ground at various scales and continuous extents with multi-temporal capabilities (Booth and Tueller 2003, Palmer and Fortescue 2003, Washington-Allen et al. 2006). However, bare ground detection is challenging because of the high frequency of spectral mixing within pixels which is a function of image resolution relative to the size of the vegetation canopy and the distribution and arrangement of plants within a study area. Even when using the highest spatial resolution multispectral satellite imaging sensor (Quickbird 2.4-m pixels) pixels will nearly always be comprised of various fractions of shrub, grass, litter, and bare ground, etc. While high spatial resolution aerial imagery has been able to minimize or reduce mixed pixels (Booth and Cox 2008) it does not capture spectral reflectance data and is often fraught with georectification problems leading to numerous challenges and limitations as well (Moffet 2009).

Previous work in sagebrush-steppe rangelands suggests that bare ground can be reliably detected (overall accuracy = 87%) when bare ground is $\geq$ 50% (Gokhale and Weber 2006). Where bare ground is less common (< 25%) it becomes increasingly difficult to accurately model and classification accuracies are typically much lower.

This paper describes a study where classification tree analysis (CTA) and maximum likelihood classification were used to model bare ground fraction in northern Spain. CTA is a non-probabilistic, non-parametric statistical technique well-suited to modeling skewed, non-normal data and phenomena (Breiman et al. 1998; Friedl and Brodley 1997; Lawrence and Wright 2001; Miller and Franklin 2001). It is hypothesized that bare ground is non-normally distributed and for this reason, may be modeled more accurately with CTA relative to other supervised classification techniques. The CTA algorithms select useful spectral and ancillary data which optimally reduce divergence in a response variable (Lawrence and Wright 2001) such as bare ground exposure. CTA uses machine-learning to perform binary recursive splitting operations and ultimately yields a classification tree diagram that is used to produce a model of the response variable. Splitting algorithms common to CTA include entropy, gain ratio, and Gini. The entropy algorithm has a tendency to over-split, creating an unnecessarily complex tree (Zambon et al., 2006). The gain ratio algorithm addresses the over-splitting problem through normalization while the Gini algorithm partitions the most homogeneous clusters first using a measure of impurity while isolating the largest homogenous category from the remainder of the data (McKay and Campbell 1982; Zambon et al., 2006). As a result, classification trees developed using the Gini splitting algorithm are less complex and therefore more easily understood by the analyst. For these reasons, the Gini splitting algorithm was selected for use in this study.

A key advantage of CTA is its ability to use both spectral and non-spectral data selectively during the splitting and classification process. This allows for the use of topographic data which may be equally important in modeling bare ground. Such ancillary data can be used with other supervised classification techniques (Lillesand et al., 2008) but classifiers like maximum likelihood use all input data to arrive at a final classification. This is in contrast to the advantage of CTA noted above, which selectively applies input data in its classification process.

MATERIALS AND METHODS

Study Area

This study focuses upon the xeric-steppes of the middle Ebro valley, Aragon, Spain and is referred to as the Monegros study area (Figure 1). The dominant plant species in the area is Rosemary (*Rosmarinus officinalis*) with various gypsophile plant species over a gypsum substrate in the most xeric areas. Scattered remnants of the original Juniper woodland community (*Juniperus thurifera*) are also present. The study area covers over 300 000 ha (3 000 km^2) with the valley receiving the majority of its water from the Pyrenees Mountains, yet it is a dry area with low precipitation (< 0.30-m annually).

Grazing activity in the area consisted of various flocks of sheep grazed under a semi-extensive regimen. Specifically, livestock were led by a shepherd to graze the fallow fields and rangeland steppe continuously throughout the year. Flocks were moved daily from shelters to the surrounding grazing areas where they stayed from morning until evening. Supplementary food was provided during the driest season and for reproductive females. Livestock productivity in the area is low, with an estimated stocking rate of 0.2 head ha^{-1} yr^{-1} (Pueyo et al. 2008).

Satellite Imagery

Satellite Pour l'Observation de la Terre 4 (SPOT 4) collects data in 4 spectral bands from the visible (545 nm band center [green] and 645 nm band center [red]) through near-infrared (NIR) (840nm band center) and short-wave infrared (SWIR) (1665 nm band center) portions of the electromagnetic spectrum. These data are stored as raster imagery having a spatial resolution of 20-m x 20-m. One SPOT 4 image was acquired on May 11, 2007 for use in this study. The SPOT 4 data were processed to top-of-the-atmosphere reflectance using the Cos(t) image-based correction method (Chavez 1988) in Idrisi Andes software (Clark Labs, Worcester, MA). The imagery was then georectified (RMSE = 8.3 m) using 0.5-m x 0.5-m aerial photography and projected into Universal Transverse Mercator (zone 30N, European datum 1950) using a first order affine transformation and nearest neighbor resampling.

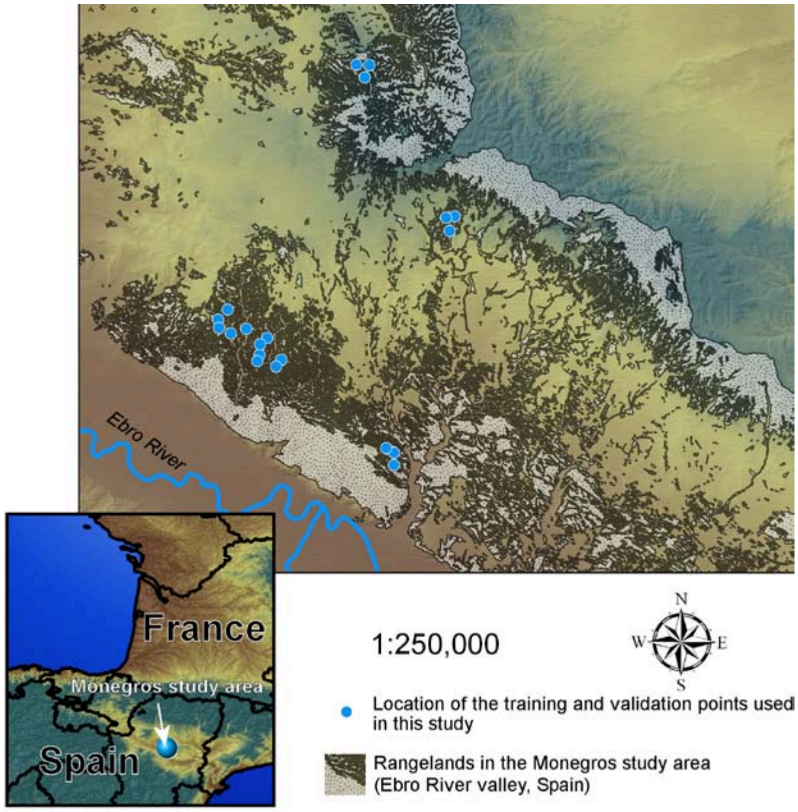

Figure 1. The Monegros study area in northern Spain. Note: due to scale, each individual sample point cannot be shown.

In addition to the atmospherically corrected SPOT 4 bands (1- 4), a normalized difference vegetation index (NDVI), moving standard deviation index (MSDI) (Tanser 1997, Tanser and Palmer 1999), principal components anlaysis (PCA) layers, and biomass estimates (Mirik et al. 2005) were also calculated within Idrisi Andes using SPOT reflectance data to develop a predictive model of bare ground for the Monegros study area.

The biomass layer is a simple ratio-type vegetation index where reflectance values from the short-wave infrared region (band) are divided by reflectance values from the green band. The resulting layer is an index and pixel values were not expressed in physical units. While Mirik et al. (2005) demonstrated a strong empirical relationship (R^2 = 0.87) between this index and actual standing crop biomass on rangelands, the relationship of the biomass index with actual above ground rangeland biomass at the Monegros study area was not performed as part of this study.

Topographic Data

A digital elevation model (20-m x 20-m pixels; RMSE = 7.42 [Pueyo 2005]) for the Monegros study area was acquired from the Confederación Hidrográfica Del Ebro (http://oph.chebro.es/ContenidoCartografico.htm). Slope (expressed in degrees) and aspect models were calculated in Idrisi Andes and a model of morphometric characterization (i.e., valley, ridge, pass, or flat) was developed using LandSerf software (Wood 1996). These topographic data (elevation, slope, aspect, and morphometry) were used to develop a predictive model of bare ground exposure.

Field Sampling

To estimate bare ground at the Monegros study area, sixteen 500-m transects were acquired between May 17 and May 24, 2004. The start location of each transect was recorded using GPS with eight transects located on north facing slopes and eight transects located on south facing slopes. Observations were made every 0.2m along each transect which described the cover type (plant species or bare ground) at that point (Gysel and Lyan 1980, Herrick et al. 2005). Percent bare ground was calculated for each 20-m segment of each transect and X- and Y-coordinates determined for the location of each segment. As each transect was oriented in an east-west direction the Y-coordinate remained constant along each transect line. The X-coordinate for each segment was determined by incrementing the beginning X-coordinate (+/- 10-m to shift the point to the center of the first line segment) by 20-m and repeating this process until the end of each transect was reached. Percent bare ground for each 20-m segment was subsequently represented as a point feature (n = 397) in all future analyses.

In May 2008, an additional 42 points were collected using GPS (+/- 0.3m @ 95% CI) which described bare ground only. Three bare ground classes were used: minimal (~ ≤10%), moderate (~ 10-50%), and high (~ ≥50%) with percent bare ground determined ocularly. All GPS locations were differentially corrected to minimize positioning error and improve coregistration among the data used in this study (Weber et al. 2008).

While two methods were used to collect field sample data these methods were considered complementary by the authors. Similarly, both McMahan et al. (2003) and Norton (2008) reported that these methods are applicable for ground truthing purposes especially where estimates are made at nadir and categorical cover classes are used to support image processing of remotely sensed data.

Data Preparation

All field sample locations (n = 439) were classified as either a 1) bare ground site (having ≥50% bare ground fraction [n = 129]), 2) non-bare ground site (having ≤10% bare ground [n = 65]), or 3) an intermediate site with 10-50% (n = 245) bare ground. Only bare ground and non-bare ground sample locations (n = 194) were used to develop the model as they effectively represented pure end-members. Sixty field sample locations were randomly selected using Hawth's tools in ESRI's ArcGIS and reserved as validation sites with 50% of the points selected from each class (bare ground and non-bare ground). The remaining locations were used as training sites (n = 134). The training and validation point shape files were imported into Idrisi Andes and rasterized using the same spatial parameters as the satellite imagery and topography layers described above (e.g., 20 x 20m pixels).

Image Processing and Accuracy Assessment

Spectral signatures for bare ground and non-bare ground training sites were extracted from all satellite imagery layers and examined for signature seperability. Most layers indicated some potential for separation between bare ground and non-bare ground sites save for PCA bands 2 and 3 which were subsequently removed from future analysis.

CTA was performed in Idrisi Andes using the Gini splitting algorithm (Zambon et al. 2006) with twelve input layers available for the classification process: green, red, near-infrared (NIR), and shortwave-infrared (SWIR) reflectance bands, NDVI and biomass band-ratios, MSDI band filter, PCA band one, and elevation, slope, aspect, and morphometry topography layers. Output included the resulting tree and a classified predictive model of bare ground with all pixels assigned one of two values; 1) bare ground site and 2) non-bare ground site. For comparison, a maximum likelihood classification was performed using spectral signatures from the same twelve input layers. Accuracy was assessed using a standard error matrix (Congalton 1991, Congalton and Green 2009) which reported user's accuracy, producer accuracy, overall accuracy, and the Kappa index of agreement statistic (Cohen 1960, Titus et al. 1984, Foody 1992, Monserud and Leemans 1992). Both error matrices were compared using Kappa and the variance of Kappa following Congalton and Green (2009) by calculating a pairwise Z-statistic (Equation 1).

$$Z_{\text{pairwise}} \quad \frac{|K_1 - K_2|}{\sqrt{\text{var}(K_1) + \text{var}(K_2)}} \qquad (1)$$

Where K_1 and K_2 are the Kappa statistics for error matrices 1 and 2 and var(K_1) and var(K_2) are estimates of variance for matrices 1 and 2. The $Z_{pairwise}$ critical value at the 95% confidence interval is 1.96.

RESULTS AND DISCUSSION

CTA classification yielded an overall accuracy of 85%, user's accuracy of 79%, and producer accuracy of 97% for the bare ground class (Table 1). The bare ground model had an overall Kappa of 0.70 and a Kappa Index of Agreement of 0.91 for the bare ground class alone. The Kappa scores indicate that the classification performed far better than a chance classification.

Table 1. CTA results for bare ground modeling in the Monegros study area in northern Spain

	Known validation sites			
Model results	Bare ground	Non-bare ground	Total	User accuracy
Bare ground	29	8	37	0.79
Non-bare ground	1	22	23	0.96
Total	30	30	60	
Producer's accuracy	0.97	0.74	Overall accuracy = 0.85	

Overall Kappa index of agreement = 0.70

Results of the maximum likelihood classification yielded an overall accuracy of 57%, user's accuracy of 54%, and producer accuracy of 83% for the bare ground class (Table 2). The Kappa score (0.13) indicates this classification performed only marginally better than a chance classification. While the same input layers, training sites, and validation sites were used for both classifications, CTA performed much better than the more traditional maximum likelihood classifier ($Z_{pairwise}$ = 4.43; $Z_{critical}$ = 1.96). The observed difference in performance is likely attributable to the way in which maximum likelihood functions with respect to the input layers the software is provided by the user. Maximum likelihood uses the spectral signature from all input layers to determine the output class of each pixel. As a result, some input layers may confuse the classifier and result in poor overall performance. This confusion is suggested in table 2 by the model over-committing pixels to the bare ground class.

Table 2. Maximum likelihood results for bare ground modeling in the Monegros study area in northern Spain

	Known validation sites			
Model results	Bare ground	Non-bare ground	Total	User accuracy
Bare ground	25	21	46	0.54
Non-bare ground	5	9	14	0.64
Total	30	30	60	
Producer's accuracy	0.83	0.30	Overall accuracy = 0.57	

Overall Kappa index of agreement = 0.13

In contrast, CTA can be given many input layers initially, but after running its splitting algorithm the final model may be based upon only a fraction of those layers. Subsequently, classification tree (Figure 2) can offer insight into the classification process by allowing the analyst to study what was identified as an indicator layer. In this instance, none of the raw imagery bands were selected for use in the classification with the exception of the SWIR band. In addition, the principal components layer was not used as well as the slope layer. The initial split chosen by the Gini algorithm was based upon elevation (~ 300m) where the elevation in the Monegros study area ranged from 137-805m ($\underline{x}$ = 354m). Within the lower elevation areas, moving standard deviation index (MSDI) was used but was not selected for use in the higher elevation areas. In the lower elevation areas, higher MSDI values were more indicative of a bare ground site than a non-bare ground site which agrees with Tanser and Palmer (1999) who reported that degraded or unstable areas exhibited higher MSDI values. SWIR reflectance was used to make two splits in the tree with the selected threshold values occurring at relatively low values (approximately 0.16 and 0.13, where the minimum value in the layer was 0.003 and the maximum value was 0.347) and below the mean (0.18). The biomass layer was also used by the Gini algorithm but was selected only within the low elevation branch of the tree. Here, low biomass values (< 6.2) were indicative of bare ground sites while all higher values higher were indicative of non-bare ground sites (x = 6.9).

Apart from the initial split which used the elevation layer, no topographic layers were used to arrive at a final classification for the lower elevation sites (38.8% of the Monegros study area). Instead, spectral information was used to finalize the classification of these areas. In contrast, the Gini algorithm used numerous topographic layers along with two spectral layers to classify the higher elevation areas (61.2% of the Monegros study area) including aspect (where westerly and northwesterly sites were more indicative of non-bare ground areas) and morphometry layers. One explanation for the increased number of variables used to classify bare ground above 300-m is the gradual increase in patch heterogeneity found in these areas. This is related to a higher proportion of residual forest and shrub land patches along the elevational gradient. The upper elevation areas were traditionally less used by local inhabitants as more favorable farming and grazing areas were found at lower elevations closer to the Ebro River. In most parts of the study area human activities such as timber harvesting, farming, and grazing, have been intensively developed for centuries (Pueyo and Alados, 2007). The result is these long-term anthropic disturbances has led to fragmented secondary communities which are very sensitive to aridity, and more directly related to past human activities than environmental factors (Pueyo and Alados, 2007).

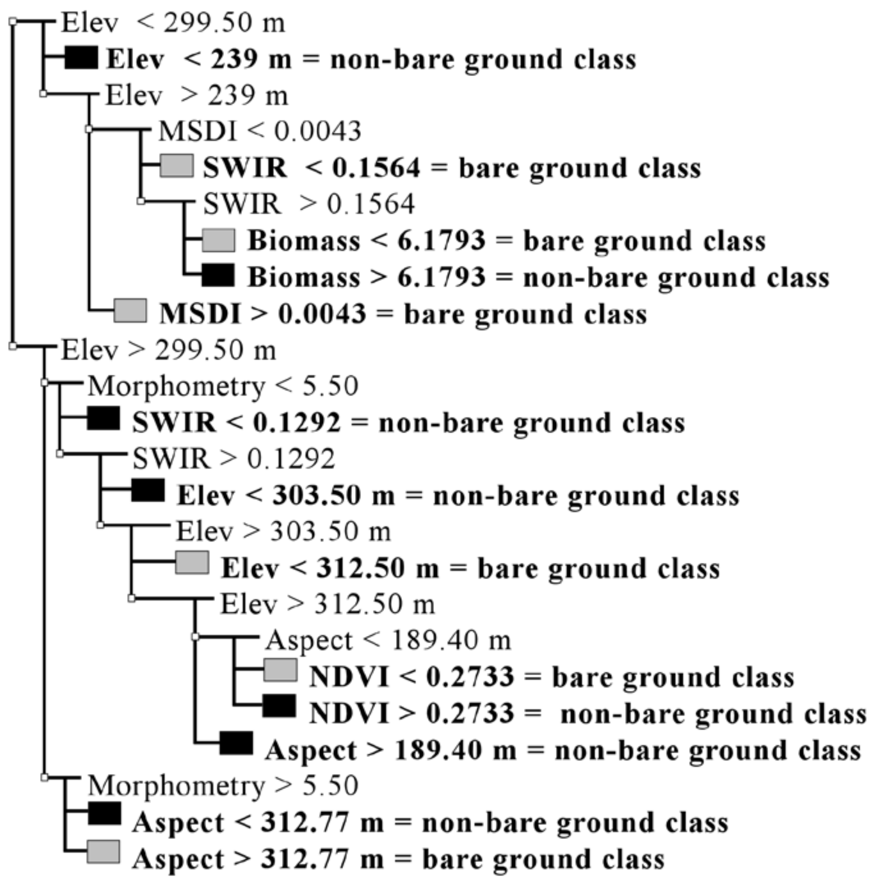

Figure 2. Classification tree produced for the bare ground model. Bold text is used to indicate where a final class decision was made: gray boxes = bare ground class and black boxes = non-bare ground class (bare ground sites were defined as having ≥50% bare ground).

The morphometry layer played an important role in the classification of higher elevation sites. Albeit a simple model, the morphometry layer described each pixel in the study area as either: valley (2 [23%]), pass (3 [3%]), ridge (4 [24%]), or flat (6 [5%]). During the classification, all values > 5.5 (i.e., flat areas) were differentiated from non-flat areas and then further split and classified using other layers. This corroborates well with field observations (Figure 3) where it was noted that the least amount of bare ground tended to be found in the flat areas between or at the foot of hills. These areas are sink sites and the result of where sediment and litter were exported from the hill top to the foot of the hill (Bilbro and Fryrear 1994; Belnap and Gillette 1998). As a result, soil fertility has increased, which favors the growth of a vegetation community dominated by rhizomatous grasses (Guerrero-Campo et al. 1999). In contrast the slopes have been more desiccated by wind (Aguiar and Sala, 1999) yielding more xeric conditions. In these higher elevation sites, NDVI and SWIR were the only spectral layers used with lower NDVI values (< 0.27) indicative of bare ground sites.

Figure 3. A photograph of the Monegros study area illustrating the effect of landscape morphometry on bare ground exposure. Very little bare ground exists in the flat areas (morphometry = 6) whereas much higher proportions of bare ground were found on the adjacent hilly sites. This phenomena was captured by the classification tree and used to improve the final model (cf. figure 2).

To further interpret the model, the 245 sample points previously removed from the classification process because they did not represent pure end-members (i.e., bare ground ranged from 10-50%), were cross-tabulated with the bare ground model. Similar in process to that described for the preparation of training and validation points, this shape file was rasterized for use in Idrisi Andes. As a result, 190 pixels were used in the cross-tabulation with 114 pixels (60%) falling into areas considered bare ground and 76 pixels (40%) falling into areas considered non-bare ground. Based upon field transect data, the mean bare ground at these sites was 31% suggesting that bare ground detection may be possible at levels below 50%. However, when additional CTA iterations were performed using training sites with bare ground ≥ 33%, classification accuracy decreased to 49% overall accuracy with a Kappa of only 0.03. This result suggests a bare ground detection threshold exists and a minimum of 50% bare ground is required to produce a model with reliable accuracies (i.e., > 75% overall accuracy; Goodchild et al., 1994).

CTA outperformed maximum likelihood (85% and 57% overall accuracy, respectively) in this study and produced classification accuracy results equivalent to those reported by Gokhale and Weber (2006) (87% overall accuracy). The previous study however, used Quickbird imagery (2.4-m pixels) while the present study accomplished comparable accuracies using 20-m pixels (SPOT 4). This provides a distinct advantage relative to both cost-effectiveness and the aerial extent covered by a single scene (~16.5-km x

16.5-km Quickbird; ~60-km x 60-km SPOT 4). These results suggest a need for additional research to learn more about the effect of spatial resolution on classification accuracy.

The results of this research indicate that CTA can be a valuable technique for the detection of bare ground in semi-arid rangelands where bare ground is ≥50%, especially when applied at landscape scales. Semi-arid ecosystems like the Monegros study area frequently exhibit plant cover <60 % (Aguiar and Sala, 1999) and the plant cover/bare ground fraction can change rapidly in response to disturbance. In these areas, detection of bare ground exceeding 50% can be beneficial to land managers as an early detection technique for land degradation and unsustainable use. While livestock grazing is common in the Monegros, stocking rate was considered relatively low (Pueyo 2005). However, the existing grazing management predisposes the areas near shelters to overuse as flocks frequent those pastures every day both before and after movement to/from the grazing areas. While daily movements of animals were typically < 3 km from shelters, the detection of bare ground in these areas is important for the management of critical water resources, which may otherwise trigger serious desertification processes.

CTA may have performed better than more traditional classifiers like maximum likelihood, because each branch and each leaf of the classification tree can use raster layers that may or may not have been used to finalize other branches or leaves of the same tree. This gives CTA the capability to fit a solution to each unique classification problem. In addition, while numerous input layers are available to the classifier, the classifier is not programmatically required to use each available layer. Rather, CTA will use only those layers offering optimal splitting. The user can then study the resulting tree to learn more about the landscape he/she is analyzing and in this way, CTA becomes a highly interactive human-machine learning system.

The results presented here do not imply that the best way to model bare ground is with those layers selected for this classification. Rather, one important result presented in this paper is the application of CTA for bare ground modeling and potentially other complex detection applications.

MANAGEMENT IMPLICATIONS

Where bare ground exceeds 50%, CTA appears to be a classification technique appropriate for modeling bare ground in semi-arid rangelands. The results presented in this paper are similar to those reported by Gokhale and Weber (2006) where Quickbird imagery and maximum likelihood classification was used for bare ground detection.

While spectral data were essential to this model, of equal importance were the topographic and morphometric characteristics of the landscape. This finding lends insight to both bare ground modeling and the potential capabilities of CTA. The results presented here should not be interpreted as the only way to model bare ground, but rather, CTA should be viewed as a powerful and flexible classification technique applicable to bare ground modeling with potential for application to other complex detection applications.

ACKNOWLEDGEMENTS
This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant. In addition, the study would not have been possible without the collaboration and cooperation of Consejo Superior de Investigaciones Cientificas (CSIC) and in particular, the Instituto Pirenaico de Ecologia (IPE).

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Recommended citation style:
Weber, K. T., C. L. Alados, C. G. Bueno, B. Gokhale, B. Komac, and Y. Pueyo. 2009. *Modeling Bare Ground with Classification Trees in Northern Spain*. Pages 55-70 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

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Application of Composite-NDVI in Semiarid Rangelands

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ABSTRACT

Ecosystem productivity is an important yet difficult metric to accurately measure. Satellite remote sensing has been used to arrive at broad-scale estimates of productivity but no single algorithm has been developed which is well suited across all ecosystems and biomes. Vegetation indices were some of the earliest estimates of productivity with the normalized difference vegetation index being the most commonly applied. Semiarid rangelands account for approximately 40% of the earth's terrestrial surface and typically exhibit spatially heterogeneous and seasonally dynamic land cover. For these reasons a single measure of productivity will nearly always underestimate the total annual productivity of rangeland sites. To address this potential problem, the composite NDVI (cNDVI) was developed which describes peak photosynthetic activity over a selected time series. In this study, cNDVI was used to compare two biophysically similar semiarid rangelands (the O'Neal Ecological Reserve in Idaho, USA and the Monegros study area in Aragon, Spain) under different grazing regimes (total rest, semi-extensive rest-rotation and continuous grazing, and intensive holistic planned grazing) to 1) equitably compare season-long productivity estimates and 2) better understand the spectral signature of the human decision-making process. Results reveal no difference in season-long cNDVI across all study areas and treatment types save for the holistic planned grazing treatment which exhibited higher cNDVI values ($P < 0.001$). This suggests there is little ecological difference between traditional semi-extensive grazing regimes but substantial, and apparently positive, effects resulting from holistic planned grazing.

KEYWORDS: season-long NDVI, growing season, holistic planned grazing, GIS, remote sensing, Idaho, Spain

INTRODUCTION

Rangeland ecosystems cover approximately 40% of the earth's terrestrial surface (Huntsinger and Hopkinson 1996, Branson et al. 1981) and are typically dominated by grass and shrub communities. These vegetation communities exist because of the semiarid or xeric nature of these sites. However, an effective hydrologic cycle (the capture, storage, and release of water) leads to healthy rangeland sites that produce green biomass (at least ephemerally) with minimal bare ground exposure. When the hydrologic cycle is disturbed, rangelands desertify and as a result, exhibit increasing amounts of bare ground exposure. Chronic desertification shifts lead to a loss of ecosystem functionality and a reduction in biodiversity (Daubenmire 1959, Schlesinger et al. 1990) with associated social and economic underpinnings (Savory 1999, Arnalds and Archer 2000, Griffin et al. 2001).

Ecosystem productivity is a related and important metric to evaluate and monitor, especially when desertification and the potential effects of global climate change are concerned (Tian et al 2000; Weber et al 2009). Measures of productivity are less direct however, than measures of bare ground exposure as the latter exists along a horizontal plane and –for the most part—can be measured and expressed as a unit of area or percent exposure. Unlike bare ground exposure, the definition of ecosystem productivity tends to be vague and open to interpretation. Further, measures of productivity tend to be more difficult to quantify with numerous methods available including above ground biomass (Chambers and Brown 1983), percent cover (Canfield 1941; Daubenmire and Daubenmire 1968), and canopy coverage (Gysel and Lyon 1980) to name but three. In most ecosystems, productivity measures are confounded by the fact that herbivores consume vegetation (the product of ecosystem productivity) while one is trying to measure productivity. Productivity estimates are typically made over large landscapes and for this reason, satellite remote sensing has been used to arrive at broad-scale estimates of ecosystem productivity. Similar to field based measures; remote sensing estimates are varied with no single algorithm being considered universally applicable. Some of the earliest and most common productivity algorithms are simple band ratios (SBR) which express an index of photosynthetically active vegetation. These vegetation indices (VI's) are varied also, but typically leverage a ratio of reflectance in the red band of a sensor to that of the near infra-red band of the sensor. Perhaps the best known and most widely applied VI is the normalized difference vegetation index (NDVI) (Rouse et al. 1973; Tucker 1979).

More recently, advanced algorithms have been developed in an attempt to systematically estimate ecosystem productivity. For instance, the Moderate Resolution Imaging Spectroradiometer (MODIS) provides an array of products that estimate vegetative productivity. The MODIS algorithms use photosynthetically active radiation (PAR) and its relationship with net primary productivity (NPP) to develop a variety of products. As some PAR is absorbed by the vegetation it is known as absorbed photosynthetically active radiation (APAR). APAR is a function of the spatial and seasonal variability of photoperiod, potential incident radiation, and the amount and geometry of displayed leaf material. It is similar to green leaf area index (LAI) but accommodates the fraction of absorbed photosynthetically active radiation (FPAR) which helps define the relationship of APAR and PAR as $APAR = PAR * FPAR$.

Gross Primary Productivity (GPP) describes the total light energy that has been converted to plant biomass. As some energy is lost during plant respiration, this fraction can be derived from GPP by subtracting leaf maintenance respiration and fine root mass maintenance respiration from GPP (Running et al 1999) to arrive at net photosynthesis (PsnNet). The fraction of photosynthetically active radiation (FPAR) is the least processed productivity estimate and is positively related to NDVI (Sellers 1992; Walko and Tremback 2005). NDVI can be considered a basic ecosystem productivity estimate from which other estimates can be calculated. For this reason, NDVI was chosen for use in this study. However, rangeland ecosystems exhibit strong seasonal dynamics and the use of a single NDVI may result in an incorrect assessment of ecosystem productivity. For this reason, a composite NDVI (cNDVI) can be used to better capture seasonal variability and the flush of grasses and forbs throughout an entire growing season.

This study uses cNDVI to enable basic ecosystem productivity comparisons between two biophysically similar study sites, the O'Neal Ecological Reserve in southeastern Idaho, USA, and the Monegros Study area in northern Spain. The goal of this comparison was to provide a better understanding of 1) ecosystem dynamics in semiarid rangelands and 2) cNDVI as an ecosystem productivity estimator relative to single-date NDVI.

MATERIALS AND METHODS

Study Area

The O'Neal Ecological Reserve, USA

The O'Neal Ecological Reserve is an area of sagebrush-steppe rangelands in southeastern Idaho approximately 30 km southeast of Pocatello, Idaho (42° 42' 25"N 112° 13' 0" W), where many local-scale rangeland studies are being conducted (Figure 1). The O'Neal is relatively flat with an elevation of approximately 1400 m. This 50 ha site is composed of typical sagebrush steppe upland areas located on lava benches and receives < 0.38 m of precipitation annually (primarily in the winter). The dominant plant species include big sagebrush (*Artemisia tridentata*) with various native and non-native grasses and forbs, including Indian rice grass (*Oryzopsis hymenoides*) and needle-and-thread (*Stipa comata*).

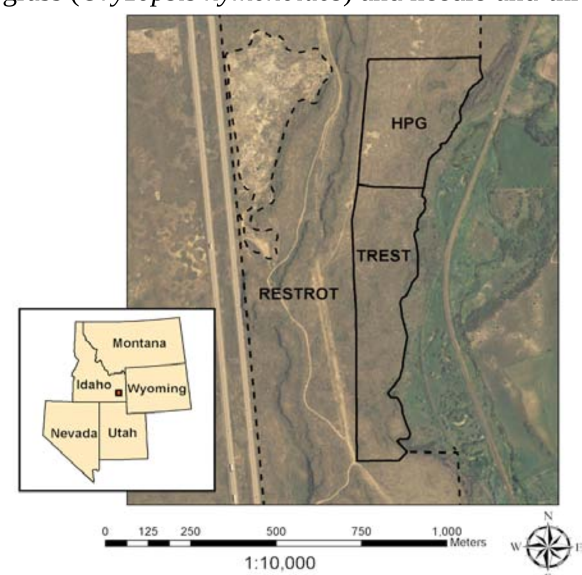

Figure 1. Location and general characteristics of the O'Neal Ecological Reserve in southeastern, Idaho.

The study area was divided into three treatment pastures (Table 1). The first was a simulated holistic planned grazing pasture where cattle graze at high density (66 Animal Units [AU]/ 11 ha) for a short period of time (6 days) during the first week of June each year (2006-2008) (cf. intensive grazing). The second treatment was a rest-rotation pasture where cattle graze at low density (300 AU/ 1467 ha) for long periods of time (30 days) during the late spring (May) of each year. This treatment is a type of extensive or semi-extensive grazing. The third treatment was a total rest pasture (13 ha) where no livestock grazing has occurred since June 2005.

Table 1. Stocking information and grazing details for the treatment pastures used in this study.

Treatment	Animal Days/ha
Holistic planned grazing (O'Neal)	36
Semi-extensive rest-rotation grazing (O'Neal)	6
Semi-extensive continuous grazing (Monegros)	11
Total rest (O'Neal)	0

The Monegros Study Area, Spain

The Monegros is a semiarid steppe region of the middle Ebro valley, Aragon, Spain (Figure 2) (41° 40' 18"N 0° 33' 51" W). The study area covers over 300 000 ha (3 000 km^2) with the valley receiving the majority of its water from the Pyrenees Mountains. It is a dry area with low precipitation (< 0.30 m annually). The dominant plant species is Rosemary (*Rosmarinus officinalis*) with various gypsophites found over a gypsum substrate in the more xeric areas. Scattered remnants of the original Juniper woodland community (*Juniperus thurifera*) are also present.

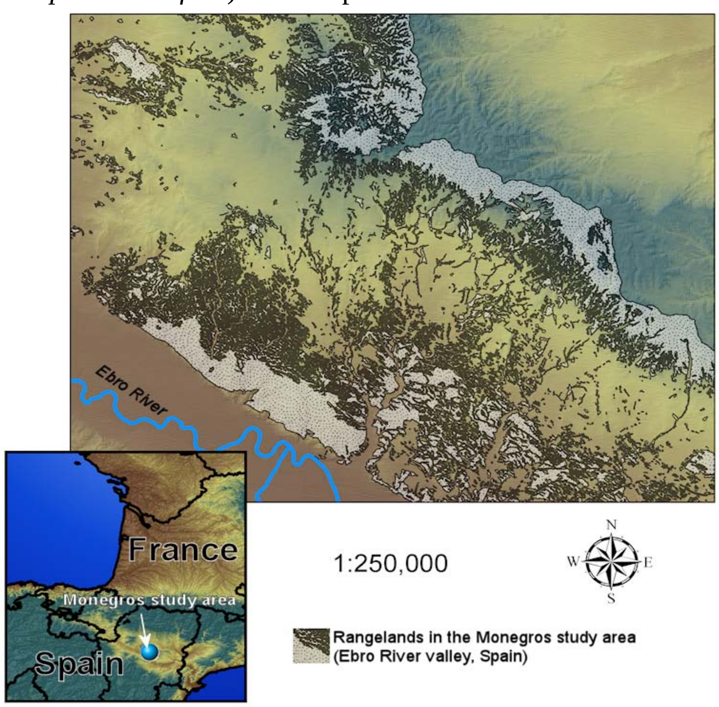

Figure 2. Location and general characteristics of the Monegros study area, northern Spain.

Grazing activity in the area consisted of various flocks of sheep grazed under a semi-extensive continuous regimen. Specifically, livestock were led by a shepherd to graze the fallow fields and rangeland steppe continuously throughout the year. Flocks were moved daily from shelters to the surrounding grazing areas where they stayed from morning until evening. Supplementary food was provided during the driest season and for reproductive females. Livestock productivity in the area is low, with an estimated stocking rate of 0.23 head ha^{-1} yr^{-1} (Pueyo et al. 2008) (Table 1).

Satellite Imagery

Satellite Pour l'Observation de la Terre (SPOT) collects data in 4 spectral bands from the visible (545 nm band center) through near-infrared (NIR) (840nm band center) and short-wave infrared (SWIR) (1665 nm band center) portions of the electromagnetic spectrum. These data are stored as raster imagery having a spatial resolution of 20 m x 20 m (SPOT 4) or 10 x 10m (SPOT 5). SPOT 5scenes were acquired on April 28, June 29, and September 15, 2007 for the O'Neal study area and SPOT 4 scenes were acquired for the Monegros study area on May 11, August 3, and August 19, 2007. These three scenes generally correspond with the early-growing season, mid-growing season, and late-growing season (senescence) for the respective study areas. All data were processed to reflectance by performing an atmospheric correction using the Cos(t) image-based absolute correction method (Chavez 1988) in Idrisi Andes software (Clark Labs, Worcester, MA). The imagery was then georectified (RMSE < 3.2 for O'Neal imagery and RMSE < 8.3 m for Monegros imagery) using high resolution aerial photography and projected into Idaho Transverse Mercator (O'Neal study area) or Universal Transverse Mercator (zone 30N, European datum 1950) (Monegros study area) using a first order affine transformation and nearest neighbor resampling.

A normalized difference vegetation index (NDVI) was calculated for each scene-date (n = 6) using Idrisi Andes with SPOT reflectance data (i.e. imagery that has been corrected for atmospheric effects). These NDVI data were then used to calculate a single composite NDVI (cNDVI) layer for each study area where the output value of each pixel represented the maximum value of each pixel from the set of input layers. Maximum NDVI was used as it has been linked to species richness by Bailey (1994) and Ivits et al. (2009). cNDVI has the potential to better characterize the vegetation of dynamic study sites where distinct vegetation communities (grasses, forbs, and shrubs) experience divergent periods of peak biomass and/or greenness. For instance, at the O'Neal study area cool season grasses like *Bromus tectorum* germinate and "green-up" early in the growing season (e.g., April and May) and then quickly senesce (June and July). During the senescent period of *Bromus tectorum*, native grasses such as Indian rice grass (*Oryzopsis hymenoides*) and needle-and-thread (*Stipa comata*) are actively growing and achieving peak biomass. Using a single NDVI layer to characterize vegetation would likely result in an underestimation of productivity whereas a single cNDVI captures the maximum NDVI value for each pixel over an entire growing season, thereby improving the characterization of vegetation within dynamic, semiarid ecosystems.

Analysis

Three distinct treatments were identified across these study areas: 1) semi-extensive (cf. rest-rotation) grazing typified by long periods of herbivory and even longer periods of rest, 2) intensive (holistic planned) grazing where plants are grazed quickly by dense herds of herbivores and 3) total rest where no livestock grazing is used. The former (semi-extensive) was the only treatment found in both study areas thereby allowing direct comparisons to be made. The latter two treatments were unique to the O'Neal study area but still offer interesting insights relative to the effect of semi-extensive grazing on cNDVI values, as well as various inferences regarding the productivity at each of these study areas.

One-hundred sample points were randomly generated across each treatment pasture at each study area (n = 400) and the cNDVI value at each point was extracted using ESRI's ArcGIS software (Sample tool). Single factor analysis of variance (ANOVA) was used to compare cNDVI values following a pair-wise approach. Specifically, cNDVI values from the semi-extensive grazing pasture of the O'Neal study area were compared to cNDVI values from the semi-extensive pastures of the Monegros study area, cNDVI values from the semi-extensive grazing pasture (O'Neal) were compared to cNDVI values from the total rest pasture (O'Neal), and cNDVI values from the total rest pasture (O'Neal) were compared to cNDVI values from semi-extensive grazing pastures of the Monegros study area. In total, six comparisons were made to test all possible pair-wise combinations. In all cases, P-values < 0.001 were considered significant.

To better understand ecosystem dynamics and compare cNDVI to single-date NDVI, a season long NDVI curve was created and cNDVI loadings determined by finding the difference in pixel values between cNDVI and each single-date NDVI layer. As a result of this calculation, pixels with a value of zero indicate a location where cNDVI was equal to the single-date NDVI. The proportion of zero-value pixels in each output layer was used to determine relative loading of the cNDVI layer and thereby better understand seasonal ecosystem dynamics of semiarid rangelands.

To interpret the results of image processing within an ecological context, 2007 field observations of land cover derived from point-intersect vegetation transects (n = 150; n = 50 per treatment pasture) were used. Percent cover estimates (Tibbits et al., 2007) for shrubs, grasses, litter, and bare ground were compared between pastures as each of these functional groups contributed substantially to the NDVI and cNDVI values under analysis. Land cover functional groups were compared using ANOVA

RESULTS AND DISCUSSION

cNDVI values of the semi-extensive grazing pastures for the O'Neal ($\underline{x}$ = 0.44) and Monegros ($\underline{x}$ = 0.42) study areas were not different (P = 0.08, and F = 3.06 [$F_{critical} = 3.88$]) (Table 2). While biophysically similar, these sites experienced very different land use and land tenure. The rest-rotation pasture at the O'Neal study area (USA) was part of a large grazing allotment administered by the USDI Bureau of Land Management and were grazed by cattle under a semi-extensive regimen. Likewise, the pastures of the Monegros study area (Spain) were grazed by sheep under a semi-extensive regimen. The similarity in grazing regimen may explain the corresponding cNDVI values since the primary treatment (temporally) impacting the land in both cases was rest and relatively low stocking rates.

If long periods of rest result in a similar signature upon the landscape regardless of the myriad differences of use applied by the rancher or shepherd during the grazing period, then one would expect to see no difference in cNDVI when comparing any semi-extensive grazing pasture with a total rest pasture. Indeed, the cNDVI values of the semi-extensive grazing pasture at the O'Neal ($\underline{x}$ = 0.44) and the total rest pasture at the O'Neal ($\underline{x}$ = 0.46) were not different (P = 0.02). Likewise, the cNDVI values of the semi-extensive grazing pastures from the Monegros study area ($\underline{x}$ = 0.42) and the total rest pasture at the

O'Neal study area ($\underline{x}$ = 0.46) were not different (P = 0.004). The latter inference is less well defended however, as the F-statistic (8.31) might indicate a significant difference between these treatment pastures. For reasons explained below, the author has chosen to be conservative in any conclusions drawn in these comparisons and readers are encouraged to review the assessment of error and bias to better understand the conservative approach taken.

Table 2. Results of single-factor ANOVA comparisons among all treatments.

			Semi-extensive		Intensive
	Treatments	Total rest	Monegros	O'Neal	O'Neal
	Total rest	X			
Semi extensive	Monegros	P = 0.004 (F = 8.30)	X		
	O'Neal	P = 0.018 (F = 5.61)	P = 0.081 (F = 3.06)	X	
Intensive	O'Neal	P = 0.041 (F = 4.20)	P = 0.000 (F = 13.74) **	P = 0.000 (F = 17.42)**	X

F-critical = 3.89

** indicates statistically significant results

The comparison of cNDVI values between the holistic planned grazing pasture ($\underline{x}$ = 0.47) and semi-extensive grazing pastures at both the O'Neal and Monegros study areas were different (P < 0.001; F = 17.42 and F = 13.75 for the O'Neal and Monegros comparisons, respectively). To understand this result, one must revisit the inference drawn earlier. Temporally, the holistic planned grazing pasture received six days of grazing and 359 days of rest annually. In contrast, the semi-extensive grazing pasture at the O'Neal received 30 days of grazing and 335 days of rest. Meanwhile in Monegros the animals graze all year, with only sporadic rest periods in some cases of two or three months where the sheep are moved to other areas. Following the logic established earlier to understand the results of comparisons between semi-extensive grazing (i.e., partial rest) and total rest treatments one would be lead to infer that rest was the primary treatment again where the holistic planned grazing pasture is concerned. However, upon closer examination the stocking rate applied to this intensively grazed pasture was found to be six- fold higher (36 animal days/ha) than the semi-extensive rest-rotation pasture (6 animal days/ha) at the O'Neal. In light of these figures, it appears the intensity of grazing has been sufficient to overwhelm the effect of rest thus yielding a treatment that is statistically unique when compared to the other production pastures. However, this alone does not explain all differences as cNDVI values for the holistic planned grazing pasture was not different from the cNDVI values for the adjacent total rest pasture.

Initially, this similarity appears to confound the inferred results, but in fact, after careful investigation of land cover the results become even more meaningful. Statistical analysis of land cover functional groups (grasses, shrubs, forbs, and litter) within each treatment pasture were compared using single-factor ANOVA. This comparison used percent cover calculations derived from vegetation transects (n=150 with 50 20-m transects collected in each treatment pasture) collected during the summer of 2007(Tibbitts et al. 2007). No difference was found in percent indicates no difference in percent grass cover between the treatment pastures (P > 0.269) and no difference in percent cover of shrubs between the holistic planned grazing and semi-extensive rest-rotation pastures (P = 0.687). Similarly, no difference in percent shrub cover was found between the total rest and rest-rotation pastures ($P = 0.249$). However, a difference in percent shrub cover was found between the holistic planned grazing and total rest pastures ($P = 0.002$). The ANOVA tests comparing percent litter cover also revealed statistically significant differences among all three treatments ($P < 0.001$) and pair-wise comparisons revealed differences between the holistic planned grazing and semi-extensive rest-rotation pastures ($P < 0.001$), as well as between the holistic planned grazing and total rest pastures. No statistical differences in litter cover were found between the total rest and rest-rotation pastures ($P > 0.001$).

These differences in land cover (specifically shrub and litter functional groups) help explain the differences observed in cNDVI. Specifically, one must recall that a reflectance value (and the indices derived from these values) represents the measured reflectance of the earth's surface within each ground resolution cell (i.e., pixel, Lillesand and Kiefer 2000). The pixel may be comprised of homogenous or heterogeneous land cover and when the latter is the case, a mixed pixel results. Within semiarid rangelands, mixed pixels are the norm and while one pixel may contain higher proportions of shrubs than another which contains a higher proportion of litter, the reflectance values of these pixels may be indistinguishable especially where broad wavebands are used. For this reason, no difference was found in cNDVI values between the total rest (with higher percent cover of shrubs) and holistic planned grazing pastures (with higher percent cover of litter).

The higher cNDVI values observed for the holistic planned grazing pasture at the O'Neal study area relative to all other production pastures was attributed to higher proportions of litter cover ($P < 0.001$) found in the holistic planned grazing pasture. Litter, while not photosynthetically active, affects all simple band ratio vegetation indices which rely upon the near-infrared band (780nm – 890nm) (Roberts et al. 1993; van Leeuwen and Huete, 1996; Asner et al. 1998; Nagler et al. 2000). This band is sensitive to the cellulose structure of plants, including the cellulose found non-photosynthetic vegetation (litter). According to Nagler et al. (2000), dry grass litter has high reflectance within the NIR band and low reflectance in the red band. This pattern of reflectance is quite similar to that seen for photosynthetically active vegetation and as a result, pixels comprised primarily of litter can have NDVI values identical to green vegetation (e.g., NDVI ~ 0.55).

The higher proportions of litter cover found within the holistic planned grazing pasture is a function of high animal impact. In comparison, total rest cannot provide the same amount of litter for two reasons. First, under long periods of rest not as many plants will grow (Savory 1999) and secondly, because the dead plant material tends to remain standing to breakdown gradually through aerial oxidation and physical weathering rather than in contact with the soil where biological decomposition occurs much more rapidly. In this case, a much more gradual chemical/physical breakdown replaces rapid biological decay (Savory 1999). Partial rest, as seen in the semi-extensive grazing pastures, exhibits nearly the same effect as total rest, i.e., fewer individual plants grow and chemical/physical breakdown predominates while some individual plants are overgrazed due to prolonged presence of grazing animals. Subsequently, less litter is laid upon the ground both because less plant material is present and also because of lower animal impact.

The ecological significance of this difference should not be overlooked as biologically degrading litter (i.e., litter in contact with the soil) adds organic matter to the soil and reduces the soil's surface temperature which, in turn, allow a higher percent volumetric water content in the active root zone of the soil (Weber and Gokhale 2009).

Season-long NDVI for both the O'Neal and Monegros study areas exhibited an interesting curve (Figure 3). In both cases, high NDVI values were achieved at the end of the growing season with an NDVI trough exhibited in mid-summer. The majority (>60%) of values loaded into the cNDVI layer were contributed by the end of growing season NDVI layers (Figure 4a, b). This suggests that the rate of photosynthetic activity during satellite overpass in mid-summer is very low, with peak photosynthetic rates occurring during the cooler parts of the year in semiarid rangelands such as the O'Neal and Monegros study areas.

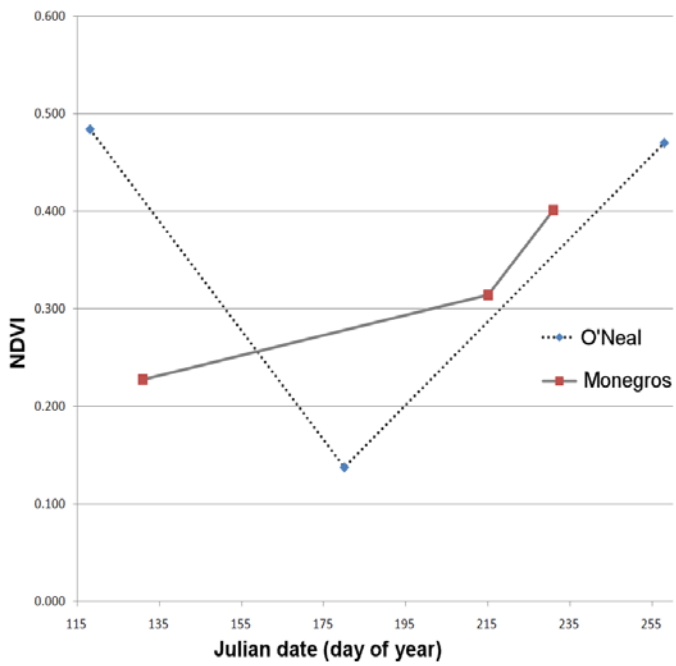

Figure 3. Season-long NDVI curves for the O'Neal and Monegros study areas.

While biophysically similar to the O'Neal, the growing season in the Monegros study area appears to be shifted to the left (i.e., there is a slightly earlier growing season). It is hypothesized that NDVI values from early April would be relatively high and similar to those observed for the O'Neal in late April. In both cases, however, the majority of pixels (>60%) constituting the cNDVI were derived from late season imagery (Figure 4). Still, substantial data was contributed from other portions of the growing season (40%) which supports continued use of cNDVI instead of single-data imagery. In addition, the contribution loadings observed in this study mirror the phenology of these study areas quite well as forbs and grasses tend to "green-up", mature, and senesce at different times throughout the growing season. Future research should be directed toward developing a better understanding of these results, and specifically understanding the composition and characteristics of late season vegetation relative to observed spectral reflectance patterns.

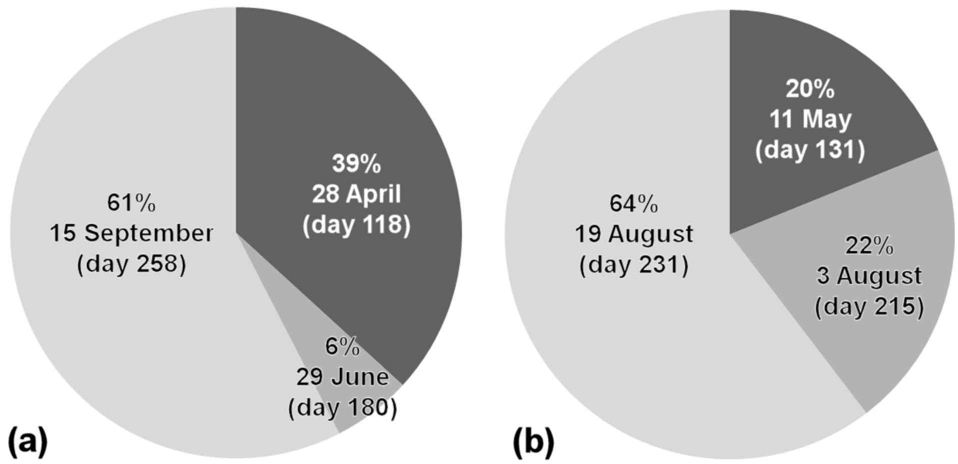

Figure 4. Contribution of single-date NDVI for the cNDVI layers in both the O'Neal (a) and Monegros (b) study areas. Note: contribution totals exceed 100% as single-date maximum NDVI values for some pixels remained consistently high throughout the year. These pixels represents roads and other static NDVI features within the study areas.

While no difference in cNDVI was noted among similar treatments within environmentally and biophysically similar sites, the single statistically significant difference reported in this study relates to a substantial difference in grazing treatment. This treatment resulted in higher cNDVI values compared to most other treatments examined in this study. The specific grazing treatment followed holistic planned grazing principles and used a relatively high density of livestock (36 animal days/ha) grazed for short time periods (6 days). This difference in grazing treatment represents a difference in land management and the manifestation of the human decision making process upon the landscape. As a result, the impact of anthropic effects upon the ecosystem is evident.

Assessment of Error and Bias

This study used SPOT 5 (O'Neal study area) and SPOT 4 (Monegros study area) imagery to calculate cNDVI values for the 2007 growing season. The cNDVI values were compared by treatment (total rest, semi-extensive grazing [e.g., rest-rotation], and intensive holistic planned grazing) within study areas and among study areas. While Theau et al. (2010) demonstrated that direct comparison of various vegetation indices across sensors is not valid, this study made exclusive use of the SPOT sensor with the only difference being the type of SPOT sensor used (i.e., SPOT 5 or SPOT 4). For this reason, direct comparison of cNDVI values was considered valid.

Another concern related to the SPOT imagery is spatial resolution. All SPOT 4 imagery has a spatial resolution of 20 m while SPOT 5 imagery has a spatial resolution of 10 m. Consequently, each SPOT 4 pixel contains four SPOT 5 pixels. As each pixel can contain only one index value it is understood that SPOT 4 pixels are more highly generalized than the SPOT 5 pixels. As a result, one would expect higher variance in all SPOT 5 derived products (NDVI and cNDVI). This was not the case however, as the standard deviation of cNDVI values for the O'Neal study area was 0.121 compared to a standard deviation of 0.244 for the Monegros study area. This may be attributable to more homogeneous land cover at the O'Neal which is likely a function of its much smaller extent (1500 ha compared with 300000 ha).

The dates of imagery acquisition representing early-growing season, mid-growing season, and late-growing season conditions were not ideal for the Monegros study area. Optimally, imagery representing early-growing season conditions could have been collected in April (instead of May) and mid-growing season imagery could have been collected in mid-May (instead of early August). In addition, the fact that the mid-growing season imagery was collected within two weeks (16 days) of the late-growing season imagery also poses some problems. As a result, cNDVI values may have been underestimated for the Monegros study area.

While the authors have made every attempt to ensure consistency across all experimental variables, the fact remains that numerous slight differences exist (e.g., SPOT 5 imagery was compared with SPOT 4 imagery and the dates of image acquisition were not ideal) in this comparative study. It is for these reasons that the significance threshold of $P < 0.001$ was selected. With this decision, it is hoped that false inferences will be avoided and the reported results will be received with confidence.

CONCLUSIONS

Composite NDVI values were calculated throughout the 2007 growing season and compared for two biophysically similar (i.e., located at similar latitudes and having similar growing seasons, precipitation regimes, range of elevation, and the presence of similar vegetation functional groups) semiarid rangeland sites: the O'Neal Ecological Reserve (Idaho, USA) and Monegros study area (Aragon, Spain). In general, no difference was found between these geographically distant study areas, substantiating the hypothesis that biophysically similar areas will exhibit similar spectral signatures over a landscape. As the primary land use (Cummins 2009) found in both study areas was livestock grazing with partial rest it was not surprising that no difference was found in cNDVI when these pastures were compared. However, comparison of a total rest pasture with partial rest pastures also failed to reveal differences in cNDVI, suggesting that semi-extensive grazing with partial rest manifests no detectable difference on the landscape relative to total rest.

The most significant result of this research was the difference in the cNDVI values observed between the holistic planned grazing pasture and all other production pastures included in this study. This suggests that holistic planned grazing 1) can have considerable effect on the landscape and 2) the result of the some human decision making processes (i.e., application of holistic planned grazing) are detectable through satellite image processing techniques. In this case, the effect of holistic planned grazing was a positive one for the ecosystem. As a result of high animal impact, the holistic planned grazing pasture exhibited a higher percent cover of litter, which subsequently resulted in higher cNDVI values.

Ecologically, the cumulative effect of holistic planned grazing led to significantly higher soil moisture levels (Weber and Gokhale 2009) most probably as a consequence of higher litter cover, manure deposition, and a higher degree of trampling/hoof action which both breaks the crust of the soil and introduces organic matter into the soil, which in turn improves the soils ability to capture and retain moisture (Savory 1999).

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). ISU would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant. In addition, field research in the Monegros study area was supported by projects CGL2008-00655/BOS from the Ministry of Science and Innovation of Spain.

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Recommended citation style:
Weber, K. T., F. Chen, B. Gokhale, C. G. Bueno, and C. L. Alados. 2009. *Application of Composite-NDVI in Semiarid Rangelands*. Pages 71-84 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

Changes in Pastoral Land Use and their Effects on Rangeland Vegetation Indices

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ABSTRACT

Extensive livestock production has been Mongolia's major industry for centuries and traditional nomadic herding lifestyle and Mongolia's expansive rangelands sustain this industry. After the democratic election and economic liberalization in 1992, formerly state-owned collectives were disbanded and Mongolia's livestock population was privatized. There was no longer a state institution to formally regulate pasture use and herders became responsible for pasture use management. We studied the changes in pastoral land use management in Tsahiriin tal of northwestern Mongolia and their effects on rangeland Normalized Difference Vegetation Index (NDVI), a remote sensing satellite-based estimate of rangeland vegetation productivity. We estimated NDVI from the collective (pre-1992) and post-collective (1992-present) period using six different Landsat Thematic Mapper satellite images and compared the mean NDVI estimates from the two periods. Our results indicate that three major changes occurred in pastoral land use management. First, grazing distribution changed from localized clusters to a more evenly distributed pattern. Secondly, the grazing animal species changed from predominantly sheep to herds of sheep, goats, cattle, and horses. Third, grazing intensity increased by over 800 animal units. Our results also indicated that NDVI values from the post-collective period are significantly lower than the NDVI values from the collective era indicating that rangeland vegetation productivity might be declining in Tsahiriin tal. This decline in NDVI might be largely associated with the increased grazing intensity from the collective era to the post-collective period.

KEYWORDS: Grazing, pastoralism, GIS, remote sensing

INTRODUCTION

Mongolia is one of the five most heavily grazed places in the world (Asner et al., 2005). Extensive livestock production has been Mongolia's major industry for centuries and traditional nomadic herding lifestyle and Mongolia's expansive rangelands sustain this industry. Mongolia's current livestock population is over 40 million and consists of cattle (2.4 million), sheep (16.9 million), goats (18.3 million), horses (2.2 million), and camels (0.2 million) (Mongolian Statistics Review Book, 2007). The livestock population was substantially smaller at the beginning of the twentieth century, but continually increased throughout the last century despite the dramatic institutional, political, and economic changes that took place as Mongolia transitioned from feudal to socialist and then democratic socio-political system (Sankey et al., 2006). The livestock population has more than doubled since Mongolia became a democratic country in 1992 and began its transition into market economy (Mearns, 2004).

After the economic transition in 1992, the livestock collectives or *negdels* of the socialist regime were disbanded and Mongolian herders were no longer employed by the state collectives forcing them to become economically self-sustaining. Transportation, access to markets, and veterinary and social services were no longer provided by the government. As a result, many herders migrated to areas near settlements and urban centers for better access to market, services, and goods. These changes lead to dramatically increased use of rangelands near urban centers (Mearns, 2004). Increased grazing pressure on rangelands has been most apparent around Ulaanbaatar (FAO Crop and Grassland Service, 2008). Most rangeland studies, therefore, have focused on areas surrounding Ulaanbaatar, which is not representative of the entire country. The remaining rangelands are largely unstudied. Local-scale studies in rural, less populated areas have not been common. Effects of the grazing land use changes in rural areas are not well understood, even though land use management changes also occurred in these areas (Sankey et al., 2006). Several remote sensing studies have described rangeland productivity throughout Mongolia (Purevdorj et al., 1998; Kogan et al., 2004; Erdenetuya and Khudulmur, 2008), but these studies were conducted at a nationwide, coarse scale without site-specific analysis of land use changes.

We studied a summer pasture in the Darkhad Valley of northern Mongolia using Global Positioning System (GPS) and remote sensing techniques. Our objectives were to: 1) document grazing land use patterns during the collective (pre-1992) and post-collective (1992-present) periods at a local scale using GPS mapping methods, 2) evaluate changes in grazing land use from the collective era to the post-collective period, and 3) assess the effects of land use changes on rangeland vegetation productivity using Landsat satellite images from the two periods. We selected the decade of 1980 (1981-1990) to represent the collective era and the current decade of 2000 (2001-2008) to represent the post-collective period. We chose these two decades due to the absence of major regime shifts within them, the presence of a major shift between them in the decade of 1990, and the availability of satellite imagery during the growing seasons (images prior to 1980 were not available).

Changes in pastoral land use in Mongolia

Traditionally, herders camp near rivers, lakes, and springs in the summer season for access to water and use pastures far from water in the winter months due to the availability of snow as a water source (Fernendez-Gimenez, 2002). When the state livestock collectives were established in the 1960s, they followed this general seasonal pastoral land use pattern. Collectives provided well-funded infrastructure including transportation for moving camps, development of wells and water tanks in waterless pastures, veterinary services, supplemental feed supplies, and monthly salary for the herders

(Fernandez-Gimenez, 1999). This allowed better distribution of grazing land use including the use of pastures more distant from water sources and community centers.

Herding households were aggregated into units known as *suuri* which consisted of 1-2 households (Fernandez-Gimenez, 1999). Each *suuri* was responsible for a fixed-sized, medium to large herds of animals of one species. Managers of collectives, namely the collective leader who was also typically the *sumiin darga* or county governor, made decisions regarding the timing and location of all movements, and coordinated all herders (Mearns, 2004). Each *sum* or county had one collective.

After the first democratic election and the pastoral economic liberalization in 1992, collectives were dismantled and all formerly state-owned animals were privatized. Although pasture land remained, and still is publicly owned, there was no longer a state institution to formally regulate pasture use. Herders were left to regulate their own pasture use management and were responsible for all production inputs and costs as the infrastructure and salary collectives provided were no longer available (Fernandez-Gimenez, 2002). At the same time, economic conditions in urban areas declined and many formerly non-herding state employees moved to the countryside to become herders with animals they acquired through privatization (FAO Crop and Grassland Service, 2008). The limited economic opportunities outside the livestock industry doubled the number of herding households (Mearns, 2004). This meant that the animals were re-distributed amongst a greater number of households initially following the regime shift in 1992. The number of animals steadily increased throughout the decade, however, at least partially due to several consecutive winters in the decade with relatively mild weather.

Today, many herders prefer to camp near settlements to take advantage of the veterinary and social services, as well as access to markets, schools, shops, and telecommunications in remote areas. Towns and settlements (*sumiin tuv* and *aimgiin tuv*) are the only places where such services are available. Herders also tend to stay close to major roads to be able to deliver their goods to markets or trade with traveling stores (Fernandez-Gimenez, 1999). Herders currently make their own decisions regarding how many and what type of animals to herd. There is no limit on the number of animals each household can own. Most herders now have several different species of livestock rather than a single species (Sankey et al., 2006).

Remote sensing of rangeland productivity

Remote sensing satellite images have been commonly used to study rangelands. Different image classification approaches and band ratios have been used to assess rangeland conditions through estimates of biomass, productivity, or percent vegetative ground cover (Jensen, 1996). The amount of total green vegetation can be estimated using Normalized Difference Vegetation Index (NDVI) (Jensen, 1996; Montandon and Small, 2008). This index is calculated using the spectral properties of vegetation reflectance in the red (R) and near-infrared (NIR) wavelengths (Rouse et al., 1974). Green vegetation typically has low reflectance in the red portion (630-690nm) of the electromagnetic spectrum due to the absorption of radiation by chlorophyll pigments, and high reflectance of the near-infrared portion of the spectrum (760-900nm) by leaf mesophyll (Jensen, 1996). NDVI is expressed as (Rouse et al., 1974):

$$NDVI = \frac{NIR\ band - R\ band}{NIR\ band + R\ band}$$

NDVI values range between -1 and 1. Higher values represent greater amounts of photosynthetic vegetation (Jensen, 1996). In semi-arid grasslands, NDVI has been successfully correlated with field based measurements of grassland biomass and some of the previously published correlation coefficients (R^2) have ranged 0.74-0.96 (Anderson et al., 1993; Fukuo et al., 2001; Wylie et al., 2002; Zha et al., 2003; Kensuke et al., 2005). In Mongolia, several coarse-scale studies have estimated the nationwide rangeland productivity using NDVI (Purevdorj et al., 1998, Bayarjargal et al., 2000, Bayarjargal et al., 2006, Erdenetuya and Khudulmur, 2008). NDVI has not been commonly used for land use change detection purposes in Mongolia, although NDVI has been widely used for change detection purpose in other regions of the world (Jin and Sader, 2005; Cakir et al., 2006; Numata et al., 2007).

METHODS

Study site description

Our study site is Tsahiriin tal valley located within the southern portion of the Darkhad Valley in northwestern Mongolia (Figure 1). Tsahiriin tal is within Renchinlhumbe *sum* of Khuvsgul *aimag* and was within the Renchinlhumbe collective territory. Tsahiriin tal is approximately 5km x 6km in dimension (~30,000 m^2). It is at 1650m elevation and experiences extreme continental climate with cold winters, short summers, and a summer-wet, winter-dry annual precipitation pattern (Figure 2). Mean annual precipitation is less than 300 mm with more than half of the yearly total falling during the months of June-August. Monthly average temperatures range from less than -30 C° in winter to close to 15 C° in summer.

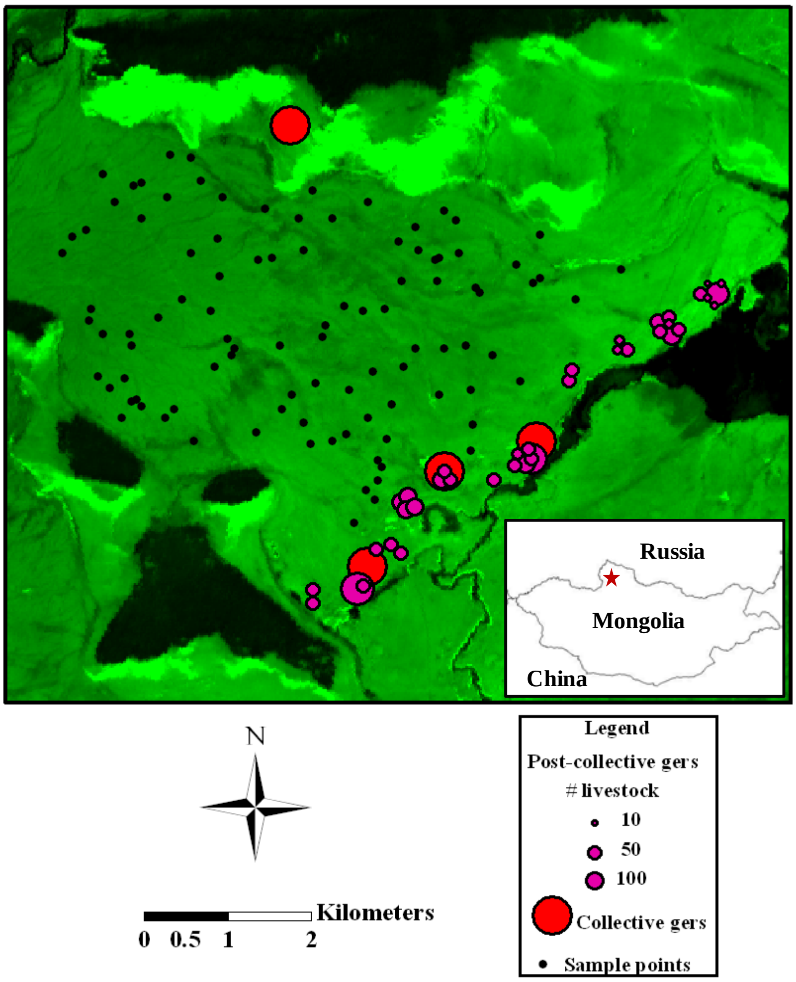

Figure 1. The Tsahiriin tal study site and documented ger or household distribution during the collective and post-collective period.

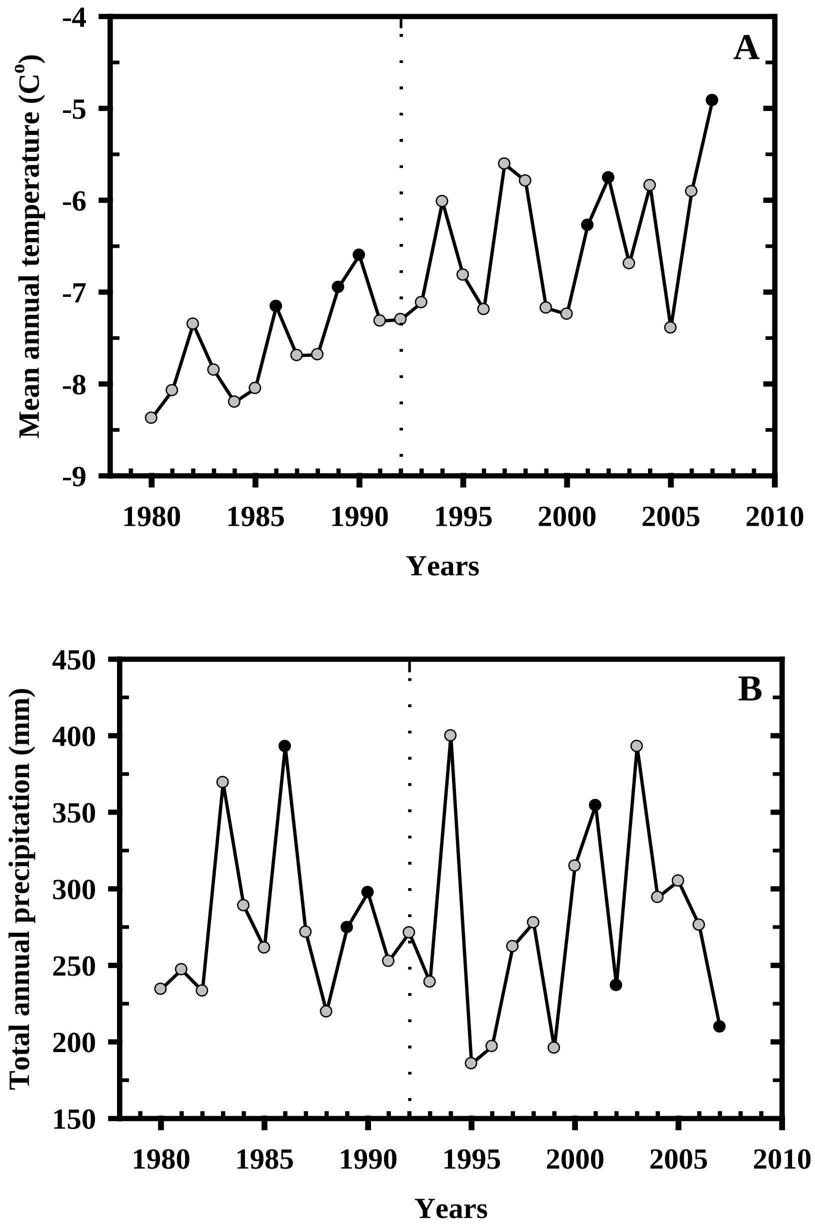

Figure 2. Mean annual temperatures (A) and total annual precipitation (B) in 1980-2007 for Renchinlhumbe county, Khuvsgul province, Mongolia. The six years selected for this study are marked with black circles. Dashed line marks the regime shift in 1992 from collective to post-collective periods.

Tsahiriin tal is bordered to the north and south by small bedrock-controlled hills with exposed limestone outcrops and herbaceous vegetation on the southerly aspects, and Siberian larch (*Larix sibirica*) forests on the northerly aspects (Figure 1). The valley is bordered by Hugiin gol river to the west, and by Tsagaan nuur lake to the east (Figure 1). Common plant species are *Poa pratensis* L., *Artemisia mongolica* (Fisch. ex Bess) Nakai, *Artemisia frigida* Willd., *Potentilla acaulis* L., and *Stipa krylovii* Roshev.

The valley floor within Tsahiriin tal consists of relic alluvial channels, terraces, and plains, as well as areas with closed depressions and hummocky rises. Ten to twenty meters of topographic relief spans the highest landscape positions (terraces and hummocks) to the lowest (channels and depressions). Soil parent materials are predominantly alluvial and lacustrine sediments. Soils associated with the alluvial features include calcareous grassland soils with organic-rich surface horizons in the more well-drained positions, and similar soils with more strongly developed subsurface clay-rich horizons in the lower (and sometimes wetter) landscape positions. These soils would include Typic Calcicryolls and Ustic (or Oxyaquic) Argicryolls, respectively, as classified by the United States soil classification system (Soil Survey Staff, 1998). Soils associated with the hummock/depression features include frost-churned (cryoturbated) permafrost and weakly developed non-permafrost soils. These soils would be classified as Aquic Haploturbels and Ustic Eutrocryepts (Soil Survey Staff, 1998).

Field study

Tsahiriin tal was visited in the summers of 2007 and 2008 to map current grazing land use and to interview local herders, veterinarians, and government officials regarding grazing land use in the collective era. Current grazing land use was documented by mapping the geographic location of each household's summer camp in the summer of 2007 using a Trimble GeoXT GPS receiver with ±3m real-time horizontal accuracy. The name of every household camping in Tsahiriin tal in the summer of 2007 was acquired and their livestock numbers were obtained from the local government records. In the summer of 2008, the former veterinarian from Tsahiriin tal during the collective period was interviewed regarding the grazing intensity and distribution during the collective period and the collective-period household locations were mapped with associated herd sizes.

Image analysis

Landsat 4 Thematic Mapper (one image) and Landsat 5 Thematic Mapper (five images) images from the peak of six different growing seasons were selected. Landsat 4 and 5 Thematic Mapper images have 30m x 30m spatial resolution and six spectral bands spanning 0.45-2.35 µm of the electromagnetic spectrum. Three of the images represent the collective era (acquisition dates: July 23, 1986, August 17, 1989, and July 19, 1990) and the other three represent the post-collective period (acquisition dates: August 9, 2001, July 20, 2002, and July 17, 2007). All images were corrected for atmospheric effects using Idrisi's ATMOSC module (based on Chavez (1996) cos(t) model) and projected in UTM Zone 47 North, WGS 1984 projection and datum. Each image was co-registered to a georectified July 9, 2007 SPOT4 image with 20m x 20m resolution (root mean squared error ranged between 0.43-0.96) using ENVI software (ENVI Version 4.3, ITT Industries Inc, 2006, Boulder, CO). All images were then subset to the Tsahiriin tal area and NDVI was estimated in each image subset using ENVI software.

Statistical analysis

Using Hawth's tool in ESRI® ArcMap™ 9.2 software (ESRI Inc, 1999-2006), 150 random points were generated within the study area and NDVI values from each image date was extracted to these points. The extracted NDVI values were then used as samples. The 1986, 1989, and 1990 NDVI values at each sample point were averaged to produce a mean value for the collective era at each point location, whereas 2001, 2002, and 2007 NDVI values were averaged to produce a mean value for the post-collective era at each point location. The NDVI values from the two periods were then compared for a statistically significant difference using analysis of variance (ANOVA) test (SPSS 14.0 for Windows, 2005).

RESULTS

Both during the collective era and now, the valley has been used as summer pasture. During the collective era, the valley was largely grazed by sheep only, totaling in 360 animal units (all species were converted to a common unit, a cow and calf combination) for 3 months a year. There were 4 collective-owned sheep herds herded by 4 households. Each herd included 450 animals of which 20-30 were goats (Table 1). The Renchinlhumbe collective was dismantled in 1992. The valley is currently used as summer pasture by 34 households (Figure 1) for 3 months a year and is grazed by 1191 animal units consisting of cattle, sheep, goats, and horses (Table 1) which are distributed in numerous small herds. In addition, there are 3 spring camps and 3 others that were being built in the summer of 2007. Total grazing intensity in Tsahiriin tal increased by approximately 830 animal units between the two time periods, which has more than tripled the grazing pressure from the collective period. The ANOVA test indicated that the Landsat image-derived NDVI values from the collective era was significantly greater than the post-collective NDVI values (p-value<0.0001) (Figure 3), suggesting that greater quantities of photosynthetic vegetation were present during the three years analyzed from the collective versus the contemporary period, respectively.

Table 1: Summary of livestock population in Tsahiriin tal during the collective and post-collective period

Total number	Collective period	Post-collective period
Sheep	1680	1169
Goats	120	755
Cattle	0	613
Horses	0	161
Total livestock	1800	2698
Total Animal Units	360	1191
Households	4	34

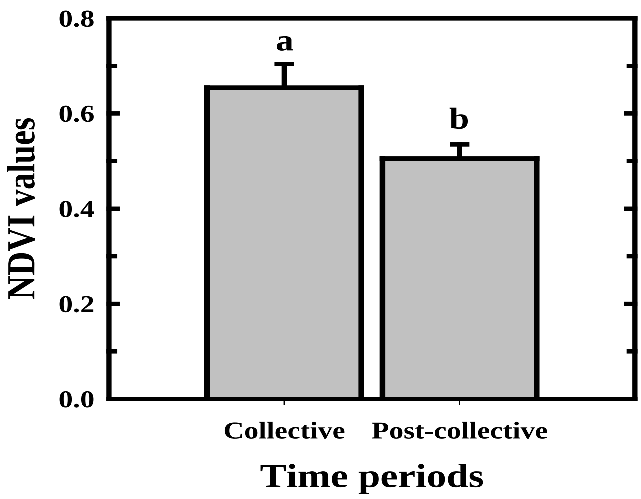

Figure 3: Landsat-derived mean (with standard error) Normalized Difference Vegetation Index (NDVI) values from the collective and post-collective periods. Different letters indicate statistically significant differences at a significance level of 0.05.

DISCUSSION

Pastoral land use changes

Three major changes were observed in pastoral land use in Tsahiriin tal during our study period (Table 1 and Figure 1). First, during the collective era, livestock grazing pressure was distributed in a few localized clusters with equally-sized, larger herds, while it is now distributed more evenly throughout the valley with numerous smaller herds. Similar to other areas of Mongolia (Bedunah and Schmidt, 2004), this change in Tsahiriin tal is associated with increased number of herding households and might be expected to result in a substantially different effect on the rangeland. The presumed cause of the difference relates to the length of recovery time for the plants between different grazing events (Voisin, 1988; Savory, 1999). Numerous smaller herds represent a continuous grazing system, in which plants are frequently grazed with little recovery time between grazing events leading to depletion of root reserves in plants. Fewer, larger herds, on the other hand, emulate a high-intensity, short-duration grazing system, in which plants receive a relatively longer recovery period between more intense grazing events.

Secondly, the grazing animal species composition changed in Tsahiriin tal from herds of predominantly a single species of sheep to four different species of cattle, sheep, goats, and horses. Although sheep remains a proportionally large component of the current herds, our livestock survey from Tsahiriin tal indicates that the number of goats is now fairly close to the number of sheep. This might be associated with the increased price of goat cashmere in Mongolia due to a more direct market in China. During the centrally planned economy prior to 1992, herders did not have direct access to the market and the Mongolian government traded goat cashmere and paid herders fixed rates of monthly salary (Agriteam, 1997). Herders, therefore, did not have the economic incentive to herd large numbers of goats that they presently have. Cashmere currently continues to increase at the domestic and foreign markets (Mongolian Statistics Book, 2007). Herders can sell goat cashmere to travelling stores, if they camp nearby major roads, or they can ship their goat cashmere to major cities to sell for higher prices. Herders can also have up-to-date information on marketing and cashmere prices through the national public radio (Bedunah and Schmidt, 2004).

The number of cattle has also increased in Tsahiriin tal. This might be due to meat and dairy consumption. Majority of the milk consumed on a daily basis during the summer season comes from cattle. Although sheep might have been milked during the collective period, it’s more time-efficient to milk cows. Cows produce greater amount of milk per animal compared to sheep and only a few cows can produce the same amount of milk as a whole herd of sheep. Cattle also produce greater amount of meat per animal. Having some cattle in the herd, therefore, help increase one’s herd size without consuming many animals in a given year to meet the meat requirement. The increased number of cattle has the greatest proportional impact on the changes in total animal units from the collective to contemporary period. When livestock numbers are converted to grazing animal units, the current number of cattle translates to more than twice as many animal units as does the current number of sheep. Such change in grazing animal species is known to have substantially different effects on the grazed vegetation community because different grazing animal species prefer different plant species (Vallentine, 2001). This might suggest that the plant species present in the Tsahiriin tal valley now receive more evenly distributed grazing pressure compared to the collective period when only plant species palatable to sheep were grazed.

Third, the grazing intensity at our study site increased by over 800 animal units resulting in more than three times greater grazing pressure in the Tsahiriin tal valley during the present decade compared to the collective era. During the collective era, herders camped in the Tsahiriin tal valley for three months a year. Currently, the valley is still used as summer pasture. However, different families can now spend varying amount of time at the summer pasture. Due to the ambiguity in current pasture management regulation and the lack of formal institution to coordinate herders in Mongolia, out-of-season pasture use and trespassing in customary grazing lands have become more common (Mearns, 2004; Fernandez-Gimenez, 2002). Furthermore, the Tsahiriin tal valley is at a fork of two major roads, one leading to Renchinlhumbe town and the other leading to Tsagaan Nuur town via a major bridge across the Hugiin gol river. Renchinlhumbe is the nearest town to Tsahiriin tal and is approximately 17km from Tsahiriin tal. This proximity to towns and the two major roads provides a convenient place for herders to stay. Herders can travel to either or both towns easily to take advantage of the veterinary and social services, as well as access to markets, schools, shops, and telecommunications in Renchinlhumbe and Tsagaan Nuur towns. Furthermore, the major roads in Tsahiriin tal provide opportunities for the herders to deliver their goods to markets or trade with traveling stores. Traveling stores tend to stop by gers close to the road more often and trade with those households rather than traveling long distances to visit individual households that are camped in remote areas.

Effects of land use change on rangeland productivity

Our Landsat image analysis results indicate that the observed changes in pastoral land use might have had significant effects on the rangeland productivity in Tsahiriin tal. The NDVI values from the post-collective period are significantly lower compared to the collective era, when livestock grazing intensity was lower in Tsahiriin tal. This indicates that rangeland productivity might have decreased in Tsahiriin tal compared to that during the collective era. Among the changes discussed above, the grazing intensity increase might have contributed most to this decrease in NDVI values. Our results of low biomass productivity in Tsahiriin tal are consistent with nationwide trends documented in Mongolia (Damdinsuren et al., 2008). The United Nations Environment Programme statement on Mongolia's environmental health (2002) indicates that over 70% of Mongolia's pastureland is degraded due to overgrazing. Furthermore, it states that the diversity of plant species has decreased by 80% near urban centers due to overgrazing. In contrast, however, other rangeland assessments continue to suggest that Mongolian rangelands are currently healthy and can support an even greater number of animals than the current population of 65 million animals in sheep units (a conversion, used in Mongolia, of all livestock species into a single species) (Mongolian Statistics Book, 2007). Tserendash's review (2008) of Mongolian rangeland assessment indicates that it can support 86 million animals in sheep units.

In addition to the grazing land use changes, we explored climate variables from the two time periods to determine if precipitation and temperature were confounding variables that could have contributed to the observed decrease in NDVI values. Nationwide trends indicate an increase in the annual mean temperature and a decrease in the annual mean precipitation (Asian Development Bank and the Clean Air Initiative for Asian Cities Center , 2006). Similarly, the long-term climate data (1974-2007) from the local weather station in Renchinlhumbe indicates that annual average temperatures have continually over the last 30 years, although precipitation fluctuated (Figure 2). This might suggest that if temperatures continue to increase in northern Mongolia in the face of global climate change, it might have an important impact on rangeland productivity. The effects of the current grazing management system combined with this increase in temperature might further accelerate the observed decline in rangeland productivity.

CONCLUSIONS

Our results indicate that rangeland productivity has declined in the rural, remote valley of Tsahiriin tal in northern Mongolia. This decline appears to be associated with the changes in grazing land use management over the last twenty years. In particular, increased number of livestock might be associated with this decrease in rangeland productivity. Such patterns could continue and further reduce rangeland productivity in Tsahiriin tal if current rangeland use is to continue without formal rangeland management institution or organized, well-structured efforts by the herding households.

Herders can use seasonal pastures close to urban settlements for shorter periods of time or camp with fewer animals to sustain healthy rangeland productivity. Herders, however, need to be better coordinated at a local scale for such management changes. Some nationwide, coarse-scale rangeland assessments continue to suggest that Mongolian rangelands are healthy and can support even greater numbers of livestock than the current size. However, our local-scale study suggests that there are areas where such recommendations should not apply.

ACKNOWLEDGEMENTS
This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Sankey, T. T., J. Sankey, K. T. Weber, and C. Montagne. 2009. *Changes in Pastoral Land Use and their Effects on Rangeland Vegetation Indices*. Pages 85-96 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

Geospatial Assessment of Grazing Regime Shifts and Socio-political Changes in a Mongolian Rangeland

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ABSTRACT

Drastic changes have occurred in Mongolia's grazing land management over the last two decades, but their effects on rangelands are ambiguous. Temporal trends in Mongolia's rangeland condition have not been well documented relative to the effects of long-term management changes. This study examined changes in grazing land use and rangeland biomass associated with the transition from the socialist collective to the current management systems in the Tsahiriin tal area of northern Mongolia. Grazing lands in Tsahiriin tal that were formerly managed by the socialist collective are now used by numerous nomadic households with their privately-owned herds, although the lands remain publicly owned. Grazing pressure has more than tripled and herd distribution has changed from a few, spatially-clustered large herds of sheep to numerous smaller herds of multiple species. Landsat image-derived Normalized Difference Vegetation Index (NDVI) estimates suggest that rangeland biomass significantly decreased (p-value <0.001) from the collective to the post-collective periods. The observed decrease was significantly correlated with changes in the grazing management system and increased stocking density (p-values < 0.001), even when potential climate-induced changes were considered. Furthermore, field- and SPOT satellite imagery-based rangeland assessments in 2007 and 2008 indicate that current rangeland biomass is low. Spatial pattern analyses show that the low biomass is uniform throughout the study site. The observed decrease in rangeland biomass might be further accelerated, if current grazing land use continues with no formal rangeland management institution or organized, well-structured efforts by the local herding households.

KEYWORDS: grassland biomass, remote sensing, GPS, GIS, NDVI

INTRODUCTION

Extensive livestock production has been Mongolia's major industry for centuries. Mongolia is one of the most heavily grazed places in the world (Asner et al., 2005). Mongolia's livestock population continually increased throughout the 20th century, despite dramatic transitions from feudal to socialist and then democratic socio-political systems (Sankey et al., 2006), and pulses of large-scale animal losses due to severe winters and drought (Angerer et al., 2008; Tachiri et al., 2008). Most notably, the livestock population more than doubled after Mongolia became a democratic country in 1992 and began its transition into market economy (Mearns, 2004; Bohannon, 2008). The trend of increasing livestock population currently continues (Figure 1a). In the year 2007 alone, Mongolia's livestock population increased 15 percent and reached over 40 million animals (Mongolian Statistics Book, 2007). During the same time period, the total number of herding households in Mongolia also doubled (Mearns, 2004) and is currently increasing again after a short period of decline associated with increasing migration of herders to urban areas as a result of large-scale animal losses (Figure 1b) (Mongolian Statistics Book, 2007). In addition, the herding households make their own decisions regarding how many and what type of animals to herd. Mongolia has no regulatory limit on the number of animals each household can own. Taken together, these conditions make Mongolia's rangelands potentially susceptible to overgrazing.

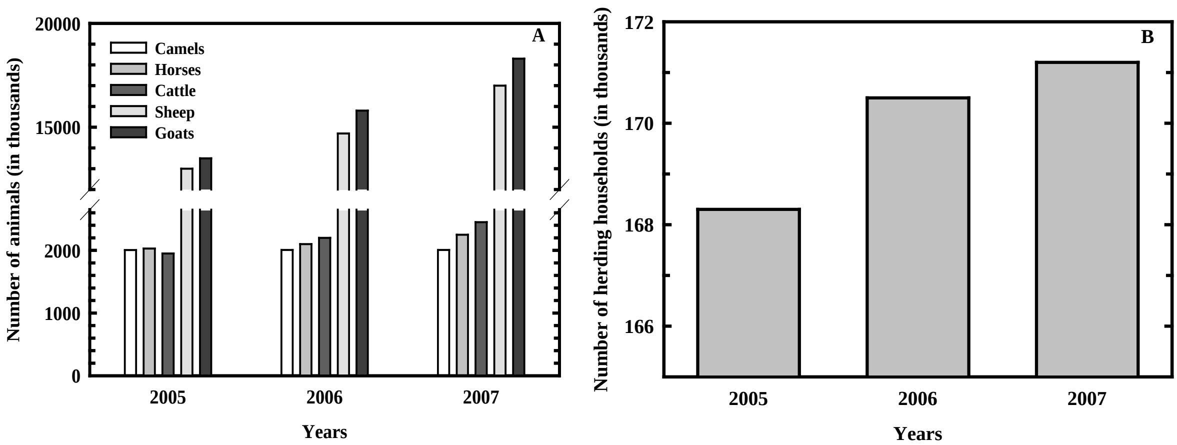

Figure 1. Livestock numbers (A) and numbers of herding households (B) from the last three years in Mongolia as examples of increasing numbers of animals and herding households since 1992 (adapted from the Mongolian Statistics Book, December 2007)

The current condition of Mongolia's rangelands and trends since the disbandment of the socialist collectives has attracted recent attention (Havstad et al., 2008), yet these issues remain largely unstudied, especially at local scales. The few nationwide studies of rangeland productivity in Mongolia (Purevdorj et al., 1998, Kogan et al., 2004, Bayarjargal et al., 2006, Erdenetuya and Khudulmur, 2008) have thus far focused on current rangeland condition only, without the analysis of long-term changes. Rangeland assessments relative to grazing land use changes are necessary to understand the recent trends in Mongolia's rangeland productivity. Moreover, some national-scale rangeland studies in Mongolia continue to suggest that rangelands are currently healthy and can support even further increase in the livestock population (Tserendash, 2008). Such recommendations are based on blurred national averages that lack the detailed documentation of correlation between grazing management changes and rangeland condition. Site-specific studies with quantified geospatial data on grazing intensity and rangeland productivity are necessary to complement national-scale studies.

This study analyzed a typical northern Mongolian rangeland using field methods and geospatial analysis tools. The objectives were to: 1) document changes in grazing land use from the collective period (pre-1992) to the post-collective period (1992-present) using GPS mapping, 2) evaluate the effects of the observed land use changes on rangeland biomass using Landsat satellite imagery acquired during the collective and post-collective periods, and 3) assess current rangeland biomass and its spatial distribution using field data and Satellite Pour l'Observation de la Terre (SPOT) satellite imagery. The decade of 1980 (1981-1990) was selected to represent the collective period and the current decade of 2000 (2001-2008) was selected to represent the post-collective period. These decades were chosen due to: 1) the absence of major socio-political and economic regime shifts during the decades, 2) the presence of a major regime shift between these decades during the decade of 1990, and 3) the availability of satellite imagery during the peak of the growing seasons (digital images prior to 1980 were not available).

The northern Mongolian rangeland examined in this study is called the Tsahiriiin tal valley. It was selected because it provides a rare opportunity with natural pastoral boundaries that limit the extent of movement by grazing animals during the typically 3-month summer season. Pasture land is publicly owned in Mongolia and not fenced or delineated for individual household use, which allows free range for all animals. Spatial boundaries in Mongolian pasture use have been described as “fuzzy, permeable, and overlapping” (Mearns, 2004 (pp 139)) and can change from year to year depending on precipitation and forage growth. This makes it difficult to delineate replicated study area boundaries in much of Mongolia’s rangelands. In this study, randomly-generated 100 point locations are used as the replicated sampling unit.

Grazing Regime Changes in Mongolia

Mongolian socialist livestock collectives were established in the 1960s and herders were paid monthly salary by the government to herd the state-owned livestock. The livestock collectives followed the traditional seasonal pastoral land use pattern. The herds grazed near rivers, lakes, and springs in the summer season for access to water and used pastures far from water in the winter months due to the availability of snow as a water source (Fernendez-Gimenez, 2002). The collectives provided well-funded infrastructure including transportation for moving camps, development of wells and water tanks in waterless pastures, supplemental feed supplies, veterinary services, travelling stores with household goods and supplies (Fernandez-Gimenez, 1999). This allowed, at a nationwide and coarser scale, better distribution of grazing land use including the use of pastures more distant from water sources and community centers. Each county had one collective with evenly distributed, large herds of a fixed size which did not vary between years or among households. Managers of the collectives made decisions regarding the timing and location of all herd movements, and coordinated all nomadic herders (Mearns, 2004). Each herd consisted of a single animal species, although a limited number of privately-owned animals of other species were allowed.

Collectives were dismantled and all formerly state-owned animals were privatized after the first democratic election in 1992 and subsequent *pastoral economic liberalization* (Fernandez-Gimenez, 1999). Although pasture land remained, and still is, publicly owned, there was no longer a state institution to formally regulate pasture use (Mearns, 2004). Herders were left to regulate their own pasture use and to pay for all expenses as the infrastructure and salary collectives provided were no longer available (Fernandez-Gimenez, 2002). At the same time, economic conditions in urban areas declined and many formerly non-herding state employees moved to the countryside to become herders with animals they acquired through privatization (FAO Crop and Grassland Service, 2008). Most herders now own a mix of cattle (includes yaks), sheep, goats, and horses, which are four of the five species of livestock traditionally found in Mongolia (with the fifth being camel) (Sankey et al., 2006).

Geospatial Tools for Rangeland Assessment

Remote sensing satellite images have been commonly used to study rangelands. Different image classification approaches and band ratios have been used to assess rangeland conditions through estimates of biomass, productivity, or vegetative ground cover (Jensen, 1996). The relative abundance of total green vegetation can be estimated using Normalized Difference Vegetation Index (NDVI) (Jensen, 1996, Montandon and Small, 2008). This index is calculated using the spectral properties of vegetation reflectance in the red (R) and near-infrared (NIR) wavelengths (Rouse et al., 1974). Green vegetation typically has low reflectance in the red portion (630 - 690 nm) of the electromagnetic spectrum due to scattering and the absorption of radiation by chlorophyll pigments, but high reflectance of the near-infrared portion of the spectrum (760 - 900 nm) by leaf mesophyll (Jensen, 1996). NDVI is expressed as (Rouse et al., 1974):

$$NDVI = \frac{NIR\ band - R\ band}{NIR\ band + R\ band} \qquad \text{(Eq. 1)}$$

NDVI values range between -1 and 1. Higher values represent greater amounts of photosynthetic vegetation (Jensen, 1996). In semi-arid grasslands, NDVI has been successfully correlated with field-based measurements of grassland biomass and some of the previously published correlation coefficients (R^2) have ranged between 0.74-0.96 (Anderson et al., 1993, Fukuo et al., 2001, Wylie et al., 2002, Zha et al., 2003, Kensuke et al., 2005). In Mongolia, several coarse-scale studies have estimated the nationwide or regional rangeland productivity using NDVI (Purevdorj et al., 1998, Bayarjargal et al., 2000, Yu et al., 2003, Yu et al., 2004, Bayarjargal et al., 2006, Erdenetuya and Khudulmur, 2008, Tachiri et al., 2008, Iwasaki, 2009). NDVI has not been commonly used for land use and land cover change detection purposes in Mongolia, although NDVI has been widely used for change detection purpose in other regions of the world (e.g., Jin and Sader, 2005, Cakir et al., 2006, Numata et al., 2007, Karnieli et al., 2008).

In addition to remote sensing, accurate GPS-based mapping of nomadic herding household distribution along with field-based vegetation and soil measurements can provide baseline data for analysis of spatial patterns of grazing use and rangeland conditions. Such spatial analysis can be used to determine whether rangelands are deteriorating or degrading (Koppel et al., 2002, Pearson, 2002, Zhong Su et al., 2006, Kefi et al., 2007, Roder et al., 2008). GPS mapping-based analyses of nomadic grazing management have not been common in Mongolia, although spatial pattern analyses of fine-scale vegetation and soil distribution have been performed (Zemmrich et al., 2007: Sasaki et al., 2008). Such geospatial analyses are crucially important in understanding rangeland health in spatially-dynamic nomadic grazing systems.

METHODS

Regional Setting and Study Area

The Tsahiriin tal valley is within Renchinlhumbe county of Khuvsgul province in northwestern Mongolia (Figure 2) and was within the Renchinlhumbe collective territory. Tsahiriin tal is approximately 5 km x 6 km in dimension (~30,000 m^2). It is at approximately 1650 m elevation and experiences extreme continental climate with cold winters, short summers, and a summer-wet, winter-dry annual precipitation pattern. Mean annual precipitation is less than 300 mm with more than half of the yearly total falling during the months of June-August. Monthly average temperatures range from less than -30 Cº in winter to close to 15 Cº in summer. Common plant species are *Poa pratensis* L., *Artemisia mongolica* (Fisch. ex Bess) Nakai, *Artemisia frigida* Willd., *Potentilla acaulis* L., and *Stipa krylovii* Roshev. The valley floor within Tsahiriin tal consists of relic alluvial channels, terraces, and plains, as well as areas with closed depressions and hummocky rises. Soil parent materials are predominantly alluvial and lacustrine sediments. Ten to twenty meters of topographic relief spans the highest landscape positions (terraces, plains, and hummocks) to the lowest (channels and depressions). Soils associated with the alluvial features include calcareous grassland soils with organic-rich surface horizons in the more well-drained positions, and similar soils with more strongly developed subsurface clay-rich horizons in the lower (and sometimes wetter) landscape positions. These soils include Typic Calcicryolls and Ustic (or Oxyaquic) Argicryolls, respectively, as classified by the United States soil classification system (Soil Survey Staff, 1998). Soils associated with the hummock/depression features include frost-churned (cryoturbated) permafrost and weakly developed non-permafrost soils. These soils are classified as Aquic Haploturbels and Ustic Eutrocryepts (Soil Survey Staff, 1998).

Tsahiriin tal is bordered to the north and south by bedrock-controlled hills with exposed limestone outcrops and herbaceous vegetation on the southerly aspects, and Siberian larch (*Larix sibirica*) forests on the northerly aspects (Figure 2). The Hogiin gol river and the Tsagaan nuur lake border the valley on the west and east, respectively. The valley is used as summer pasture only. *Gers* (traditional Mongolian tents used by herders) are located beyond the natural borders of Tsahiriin tal during the summer. However, animals from these *gers* cannot normally graze into the Tsahiriin tal valley, just as animals do not often graze out of the valley. Tsahiriin tal, therefore, encompasses an area for which stocking density can be quantified. Although a greater geographic extent might be more desirable, grazing boundaries at such scales are not feasible to determine, which makes it difficult to estimate grazing effects.

Field Methods

To assess current rangeland biomass, two seasons of field work were completed during the month of July in 2007 and 2008. Prior to field work, 100 random points were generated across the Tsahiriin tal area using Hawth's tool in ESRI® ArcMap™ 9.2 software [ESRI Inc, 1999-2006]. The same set of points were visited each year by navigating with a Trimble GeoXT GPS receiver with $\pm$ 3 m real-time horizontal accuracy. At each point, estimates of percent cover of litter, herbaceous cover, bare soil, and rock (coarse fragments > 75 mm) were made within a 10 m by 10 m plot centered on the point and aligned in the cardinal directions. Point-intercept method was used along two, 10 m line transects that were oriented perpendicular to each other and intersected at the center of the plot at 5 m along each transect. Observations were recorded at every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm, to indicate the cover type at the point. This resulted in 100 point measurements for each plot. All herbaceous plants within a 0.44 m^2 cable hoop randomly tossed within each quadrant of each plot were clipped and weighed to estimate average standing plant biomass (henceforth referred to as biomass) for each plot. A total of 108 bags of biomass samples were randomly selected from the set of all samples across the study site. These samples were dried to estimate the weight difference between wet and dry biomass samples. On average, 49.96 % (SD$\pm$5.02) of the weight was lost during drying. This difference was subtracted from all wet weights to convert the wet biomass estimates to dry biomass estimates. At each plot, a soil profile was described to evaluate the surface and first subsurface horizon thickness, color, and structure.

Topography was classified into one of three possible classes at each plot: convex (water-shedding), level, or concave (water-collecting).

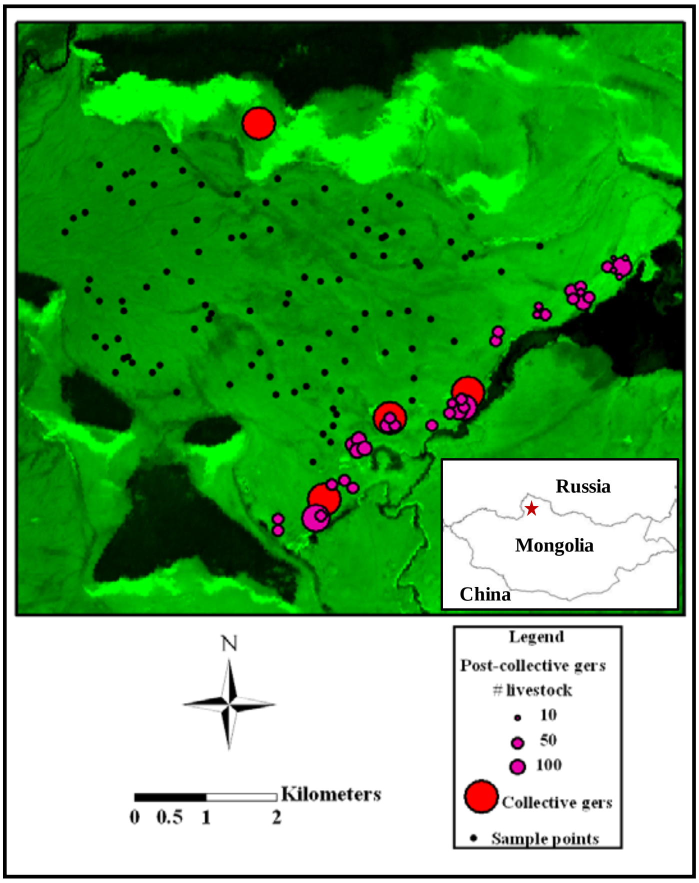

Figure 2. The documented *ger* or household distribution during the collective and post-collective periods at the Tsahiriin tal study site in Mongolia (inset).

The location of the households currently camped in the Tsahiriin tal valley and their grazing distribution was documented by mapping the summer camps or *gers* in the summer of 2007 using a Trimble GeoXT GPS receiver. The name of each household was acquired during mapping. Their livestock numbers were then obtained from local government tax records. In the summer of 2008, a collective-period veterinarian from Tsahiriin tal was interviewed regarding the herd size and distribution during the collective period (Maruush, July 15, 2008, personal communication). A map of the collective-period household locations with associated herd sizes was produced with the veterinarian's assistance.

Image Analysis

Landsat-4 Thematic Mapper (one image) and Landsat-5 Thematic Mapper (five images) images from the peak of six different growing seasons were acquired to assess changes in rangeland biomass (objective 2). Three of the images represent the collective period (dated July 23, 1986; August 17, 1989; and July 19, 1990) and three represent the post-collective period (dated August 9, 2001; July 20, 2002; and July 17, 2007). In addition, SPOT-4 satellite imagery (acquired on August 8, 2007) and SPOT-5 imagery (acquired on August 9, 2008) were used to assess current rangeland biomass (objective 3). All images were corrected for atmospheric effects using Idrisi's ATMOSC module (based on Chavez (1996) cos(t) model) and were projected in UTM Zone 47 North with WGS 1984 datum. Each image was co-registered to a georectified SPOT-4 image with 20 m x 20 m resolution (root mean squared error ranged between 0.43 - 0.96 meters) using ArcMap 9.2 software. All images were then subset to the Tsahiriin tal area. NDVI was estimated in each image subset using ENVI software (ENVI Version 4.3, ITT Industries Inc, 2006, Boulder, CO). NDVI values at the 100 random points were then extracted for statistical analysis.

GIS Data Sets

A shapefile of the 100 random points was created using ArcMap 9.2 software and each point was assigned attributes of: field-based estimates of biomass and percent cover of green vegetation in 2007 and 2008, SPOT image-derived NDVI values from 2007 and 2008, and Landsat image-derived NDVI values from 1986, 1989, 1990, 2001, 2002, and 2007. In addition, attributes describing the current stocking density as well as the collective-period stocking density were created using the *ger* maps from the two periods. Stocking density attributes were derived by generating six concentric buffer rings around each *ger*. The buffer rings were each 1 km wide and increased in circumference with increasing distance from each *ger*. The ring closest to each *ger* (i.e. the innermost ring) was classified as having the greatest stocking density, while the remaining rings were classified with decreasing stocking density as distance from the *ger* increased. The assumption that stocking density was greatest within the rings closest to the *gers* and decreased with increasing distance away from the *gers* was made, because all animals, except for horses, are brought to camp every night for milking, shelter, and protection from predators. Animals also spend a portion of each morning grazing adjacent to the camp, before herders herd them to farther reaches of the valley for the day. Next, the area of each buffer ring was calculated and the number of animals owned by each household was divided by this area to estimate the animal density per square km within each buffer ring. The concentric buffer rings radiating away from each *ger* eventually overlap with other buffers from the neighboring *gers.* Therefore, the animal densities from all overlapping buffer rings of all neighboring *gers* were added to estimate the total animal density per square km throughout the entire Tsahiriin tal valley. The resulting zonal attributes were converted to a raster format with 28.5 m resolution. The estimated stocking density at the 100 random points were then extracted for statistical analysis.

STATISTICAL ANALYSIS

Collective versus Post-collective Change Analysis

The 1986, 1989, and 1990 Landsat NDVI values at each sample point were averaged to produce a mean value for the collective period at each point location. Means were similarly calculated for the post-collective period using the 2001, 2002, and 2007 Landsat NDVI values. The mean NDVI values at the 100 sample locations from the two periods were then compared using analysis of variance (ANOVA) test (SPSS 14.0 for Windows, 2005) to assess changes in rangeland biomass between the collective and post-collective periods. In addition, a simple regression model was developed using all Landsat NDVI values from the six growing seasons as a response variable and the grazing management systems from the two periods as a categorical predictor variable. A separate regression model was also developed using all Landsat NDVI values as a response variable and the estimated stocking densities at the 100 random locations during the two periods as a predictor variable.

A climate dataset from our study region since 1980 indicates that mean annual temperatures have increased from 1980-present, albeit with substantial inter-annual variability (Figure 3a), while total annual precipitation has fluctuated without a substantial positive or negative trend during the same time period (Figure 3b). The increasing temperatures and fluctuating precipitation probably had some effects on the observed Landsat NDVI values in addition to the effects of grazing management changes. Propastin et al. (2007) report strong positive correlation between AVHRR NDVI data and temperature and precipitation at all scales in Central Asian rangelands in Kazakhstan. We acquired AVHRR Pathfinder NDVI time-series data (NOAA/NASA EOS-WEBSTER) from August of 1982-2002 (with 1995, 1996, and 1997 missing). We selected a 1225 km^2 area (35 km x 35 km pixel) centered over the study site from each year to construct an annual NDVI time-series dataset for the peak of the growing season from 1982-2002 (Figure 3c). A linear regression trendline was fit to the AVHRR Pathfinder NDVI dataset (R^2=0.10). The regression slope indicated a 0.0019 increase in NDVI per year (Figure 3c), which was assumed to reflect changes in NDVI due to climate effects. The observed Landsat NDVI values from the six years were adjusted to remove the climate-related trend (i.e., regression slope) observed in the AVHRR time-series. The adjusted NDVI values were then averaged to produce an adjusted mean value for the collective and post-collective periods at each point location. These adjusted mean values from the two periods were again compared using an ANOVA test (SPSS 14.0 for Windows, 2005) to examine the effects of grazing management changes.

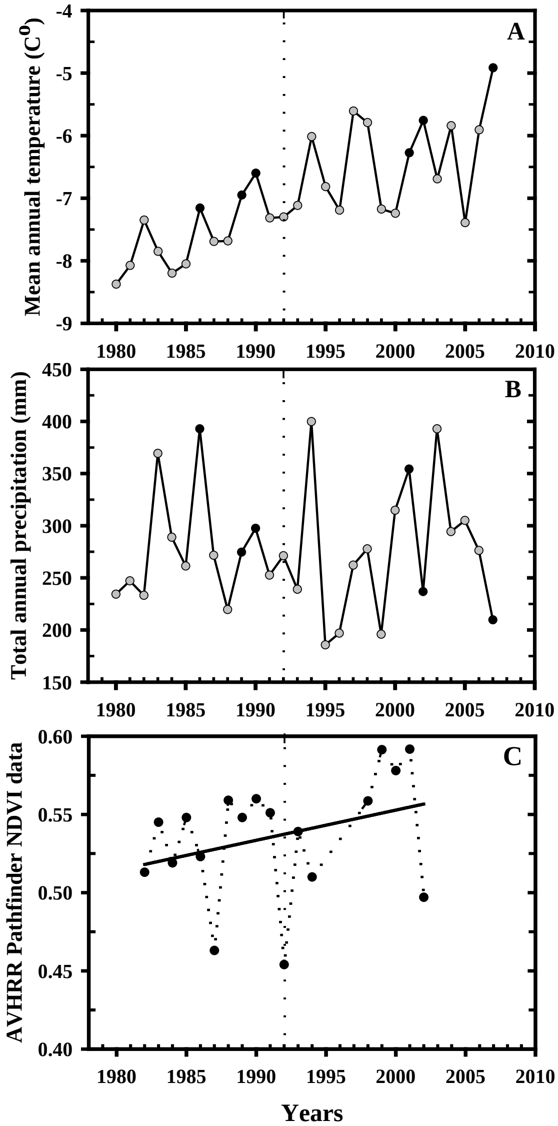

Figure 3. Mean annual temperatures (A) and total annual precipitation (B) in 1980-2007 for Renchinlhumbe county, Khuvsgul province, Mongolia. The six years selected for this study are marked with black circles. Dashed line marks the regime shift in 1992 from collective to post-collective periods. The AVHRR NDVI time-series data from the study region beyond the Tsahiriin tal valley and its long-term trend (C) was used to adjust the Landsat NDVI values for potential climate-induced effects.

Current Rangeland Biomass

We used field-based biomass estimates and SPOT NDVI estimates individually as the response variables to represent rangeland biomass in separate regression models. Field-based biomass estimates were predicted as a function of stocking density, topographic classes, and surface soil horizon thickness. SPOT NDVI values were predicted as a function of the same predictor variables using separate linear regression models. Field-based biomass estimates from 2007 and 2008 were not strongly correlated with SPOT NDVI estimates (p-value = 0.432 in 2007 and p-value <0.0001 and adjusted R^2=0.13 in 2008). Field-based and image-based estimates, therefore, could not be used to predict one another.

Exploratory spatial pattern analysis was performed to evaluate the spatial distribution of the field biomass estimates and SPOT NDVI values. Field biomass measurements and SPOT NDVI estimates from 2007 and 2008 were examined using Moran's I to determine whether their distribution was spatially clustered, random, or uniform. Moran's I index was estimated using the Euclidian distance method with inverse distance relationship in ArcMap 9.2 software. A Z-score was also estimated to determine the statistical significance of the estimated I. Moran's I values close to -1 indicate a uniform pattern, values close to 0 indicate a random pattern, and values close to 1 indicate a clustered pattern (O'Sullivan and Unwin, 2003). Getis-Ord general G with a Z score (significance level of 0.01) was additionally used to determine if high and low field biomass estimates and NDVI estimates were spatially clustered across the study site. In Getis-Ord analysis, a Z score close to 0 indicates that there is no clustering, a positive Z score indicates clustering in the high values, and a negative Z score indicates clustering in the low values.

RESULTS

Changes in Grazing Land use and Rangeland Biomass

The Tsahiriin tal valley has been used as summer pasture during the collective and post-collective periods. During the collective period, the valley was predominantly grazed by sheep with 360 Animal Units (AU) (each AU equals one cow and calf pair) for three months a year. There were four collective-owned sheep flocks herded by four households. Each herd included 450 animals of which 20-30 were goats (Table 1). The collective was dismantled in 1992. The valley is currently used by 34 households (Figure 2) for approximately three months a year and is grazed by 1191 AU consisting of cattle (includes yaks), sheep, goats, and horses (Table 1).

Table 1. Summary of livestock population in Tsahiriin tal during the collective and post-collective period

Total number	Collective period	Post-collective period
Sheep	1680	1169
Goats	120	755
Cattle	0	613
Horses	0	161
Total livestock	1800	2698
Total Animal Units	360	1191
Households	4	34

The first ANOVA model (comparing the observed NDVI values) indicated that the post-collective, observed Landsat NDVI values were significantly lower than the observed Landsat NDVI values from the collective period (p-value <0.0001) (Figure 4a). The second ANOVA model (comparing the adjusted NDVI values) indicated that the adjusted Landsat NDVI values from the post-collective period were also significantly lower than those from the collective period (p-value < 0.0001) (Figure 4b). The simple regression models both indicated statistically significant negative effects of grazing management changes and increasing stocking densities on NDVI (p-values <0.001), although the coefficients of determination were low (adjusted R^2 of 0.14 and 0.03, respectively).

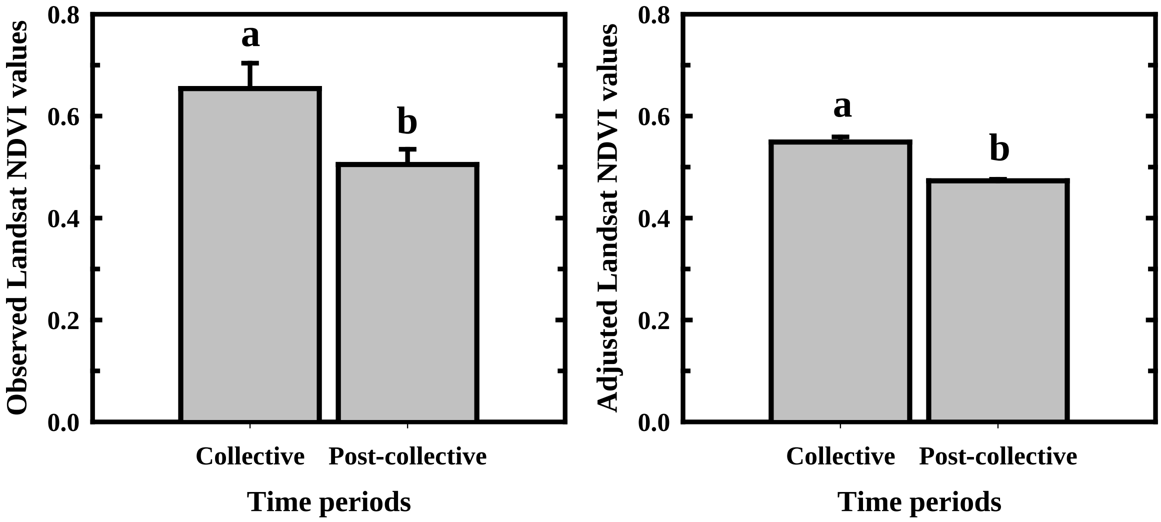

Figure 4. Estimated (A) and adjusted (B) Landsat-derived mean (with standard error) Normalized Difference Vegetation Index (NDVI) values from the collective and post-collective periods. Different letters indicate statistically significant differences at a significance level of 0.05.

Current Rangeland Biomass

Mean field-based green vegetation cover was 68 percent (SD±11.8) in 2007 and 49 percent (SD±7.6) in 2008. Field-based estimates of average dry forage was 712 kg/ha in 2007 and 605 kg/ha in 2008 in Tsahiriin tal. Field-based biomass estimates were not significantly correlated, in both years, with topography (*p*-values = 0.28 and 0.42) or thickness of the surface and first subsurface soil horizons (*p*-values = 0.283 and 0.789). In 2007, field-based biomass estimates were not significantly correlated with stocking density (*p*-value = 0.858), but the correlation was significant in 2008 (*p*-value = 0.035) with a low adjusted R^2 of 0.035. Moran's *I* for both years indicated a random spatial pattern (*I* = 0.004 and 0.017, Z-score = 0.01 and 0.10, for 2007 and 2008, respectively). The Getis-Ord general *G* index indicated no clustering in both years (0.0006 and 0.009 with a Z-score of -0.61 and 0.005, respectively).

The estimated mean SPOT NDVI values were 0.193 (SD±0.06) and 0.406 (SD±0.05) in 2007 and 2008, respectively. SPOT NDVI was not significantly correlated with topography (*p*-value = 0.650) or stocking density (*p*-value = 0.787) in 2007, but was significantly correlated with topography (*p*-value = 0.027) and stocking density (*p*-value = 0.054) in 2008 with an adjusted R^2 of 0.11. SPOT NDVI values were not correlated, in either year, to surface soil horizon thickness (*p*-value 0.098 and 0.56). Moran's *I* indicated a completely random pattern for SPOT NDVI values for both years (*I* = 0.25 with a Z-score of 0.05, and *I* = 0.0001 with a Z-score of 0.000, for 2007 and 2008, respectively). The Getis-Ord general *G* index also indicated a random pattern for SPOT NDVI values (0.0005 with a Z-score of -0.37 in 2007 and 0.004 with Z-score of 0.001 in 2008.

DISCUSSION

Grazing Land use Changes and their Effects on Rangeland Biomass

Three major changes were observed in Tsahiriin tal when the collective-period grazing land use was compared to the current grazing land use (Table 1 and Figure 2). First, during the collective period, livestock grazing was distributed in a few localized clusters of equally-sized large herds within the geographic extent of our study site, while it is now distributed more evenly throughout the valley with numerous smaller herds. The collective management maintained a small group of four households in the valley, whereas nomadic herders can now freely migrate to Tsahiriin tal resulting in a much larger number of households. Similar to other areas of Mongolia, this change in Tsahiriin tal is associated with increased number of herding households (Bedunah and Schmidt, 2004). One possible effect of this change on the rangeland might be a decrease in length of recovery time for the plants between grazing events (Voisin 1988, Savory 1999). Numerous smaller herds represent a continuous grazing system, in which plants are frequently grazed with little recovery time between grazing events. In contrast, fewer, larger herds, such as during the collective period, more closely emulate a high intensity grazing system, in which plants receive a relatively longer recovery period between more intense grazing events.

Secondly, the grazing animal species composition changed in Tsahiriin tal from herds of predominantly a single species of livestock (sheep) to four different species of livestock (cattle, sheep, goats, and horses). Although sheep remains a proportionally large component of the current herds, our livestock survey from Tsahiriin tal indicates that the number of goats is now fairly close to the number of sheep due to increased cashmere prices in Mongolia and China. Furthermore, the number of cattle has increased, which has the greatest proportional impact on the changes in total Animal Units from the collective to the post-collective period. Such changes in herd composition are known to have substantially different effects on the grazed vegetation community because different grazing animal species prefer different plant species (Vallentine, 2001). Lastly, the stocking density in the Tsahiriin tal valley has increased by over 800 Animal Units, which has more than tripled the grazing pressure from the collective period. This trend is similar to the observed patterns in other areas of Mongolia (UNEP, 2002, Bedunah and Schmidt, 2004, Bohannon, 2008) as well as the national trend over the last 15 years (Damdinsuren et al., 2008).

These changes appeared to correspond with a decrease in rangeland biomass as measured by a significant decrease in Landsat NDVI values even when potential climate-induced effects were taken into consideration. In particular, the decrease in rangeland biomass was significantly correlated with the changes in grazing management and increased stocking density in the Tsahiriin tal valley. This trend of decreased rangeland biomass might be occurring at many other locations in Mongolia where increased livestock numbers are documented (UNEP, 2002, Bedunah and Schmidt, 2004, Bohannon, 2008). Furthermore, a similar trend might have dominated across the entire country over the last two decades since the livestock population has doubled nationwide with the socio-economic and political changes (Figure 1). However, long-term trends since the collective period have not been examined at the national scale. Only the deteriorating conditions in areas surrounding major urban areas have been documented (Mearns, 2004, FAO Crop and Grassland Service, 2008), while less populated rural areas are mostly unstudied.

Current Rangeland Biomass

Tsahiriin tal had less than half of the average biomass in an ungrazed enclosure (35 years of no grazing, 17 km from Tsahiriin tal), which was sampled as a potential reference site (1,876 kg per ha), although with no formal statistical comparison because of limited sample size within the enclosure. The observed, relatively low rangeland biomass in Tsahiriin tal was not strongly correlated to any of the other local variables measured. Most importantly, current biomass was not correlated to the estimated stocking density. Rangeland biomass was expected to increase with increasing distance away from camps (Kensuke et al., 2005), where stocking density was estimated to be lower. This pattern was not found, however, which might indicate that grazing pressure was high not only near camp sites, but throughout the entire Tsahiriin tal valley. Furthermore, greater biomass was expected in the small, wet depressions and swales which were common across the study site. These water-collecting landscape positions tended to have slightly thicker soil A-horizons, suggesting that they might have historically been locations of greater biomass. However, results indicated these locations to be equally grazed relative to the others. Spatial pattern analysis also showed that the current low biomass is evenly distributed throughout the valley, with no spatial clustering of low or high biomass estimates, and no directional increase or decrease in biomass with distance from camps. Taken together, our results indicate that the drastic increase in grazing pressure might have overwhelmed the effects of other local factors resulting in uniformly heavily grazed rangelands with little variability in biomass. This lack of variability in biomass might have contributed to the low correlation between field-based biomass estimates and NDVI values. NDVI correlation with field biomass has been low (R^2 ranges 0.05-0.4) in other studies in heavily grazed areas (Numata et al., 2007, Yang et al., 2009).

The current low biomass in Tsahiriin tal is consistent with nationwide trends documented in Mongolia (Damdinsuren et al., 2008). The United Nations Environment Programme statement on Mongolia's environmental health (2002) indicates that over 70% of Mongolia's rangeland is degraded due to overgrazing. Interestingly, there are rangeland assessments which continue to suggest that Mongolian rangelands are currently healthy and can support an even greater number of animals than the current population of 65 million animals in sheep units (a conversion, used in Mongolia, of all livestock species into a single species) (Mongolian Statistics Book, 2007). Tserendash's review (2008) of Mongolian rangeland assessment, for example, indicates that it can support 86 million animals in sheep units. Results from Tsahiriin tal, however, clearly indicate that the changes in grazing pressure and grazing management since disbandment of the socialist collectives have already had significant impact on rangeland biomass.

CONCLUSIONS

Major changes in grazing land use management have had significant effects on rangeland biomass in Tsahiriin tal of northern Mongolia. Rangeland biomass has significantly decreased in the post-collective period relative to the collective period, and low biomass appears currently wide spread and predominant throughout the valley. The Tsahiriin tal rangeland biomass might further decline, if current rangeland use continues without either formal government-led management or organized, well-structured efforts by the local herding households. Some nationwide, coarse-scale rangeland assessments continue to suggest that Mongolian rangelands are healthy given the current grazing regime and can support even greater numbers of livestock than the current size. This study provides evidence from one northern Mongolian rangeland where such recommendations should not apply. Mongolian national-level rangeland management might benefit from more studies that examine local, site-specific effects on rangelands of the regime shift that has occurred with the transition from socialist to democratic socio-political systems.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:

Sankey, T. T., J. Sankey, K. T. Weber, and C. Montagne. 2009. *Geospatial Assessment of Grazing Regime Shifts and Socio-Political Changes in a Mongolian Rangeland.* Pages 97-112 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

Rangeland Assessments Using Remote Sensing: Is NDVI Useful?

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ABSTRACT

Two semi-arid rangeland sites were chosen to assess the applicability of NDVI as a predictor of vegetation cover and biomass. While geographically distant, both sites shared many traits and were considered biophysically similar environments. These sites, one in northern Mongolia and one in western USA, were the focus of field based vegetation studies and repeated remote sensing acquisitions between 2007 and 2008. Atmospherically corrected Satellite Pour l'Observation de la Terre (SPOT) imagery was used to develop NDVI models representing early, middle, and late segments of the growing season. Field-based biomass and percent cover of green vegetation were correlated with SPOT NDVI data for each imagery date at 100 sample locations for each study site using simple linear regression models. The resulting correlations were weak ($R^2 \leq 0.184$) and only five of the 18 relationships tested demonstrated statistical significance. When bare soil reached or exceeded 20%, NDVI was no longer statistically significant as a predictor variable for any vegetation characteristic tested in this study. These results suggest that NDVI might not be a useful estimate of vegetation cover or biomass in semi-arid rangelands, especially when bare soil cover is >20 percent.

KEYWORDS: biomass, vegetation cover, NDVI, remote sensing

INTRODUCTION

Rangelands around the world can have drastically different grazing management systems depending on the political, social, economic, and cultural settings. To study the effects of two contrasting traditional grazing systems on rangelands, we conducted rangeland assessments in two biophysically-similar rangelands of northern Mongolia and western USA. The grasslands of northern Mongolia are used by nomadic herders with their multiple livestock species at a greater grazing intensity, while the shrubland steppe of the western USA is grazed by sheep only at a lower grazing intensity. A core indicator of plant cover (Pellant et al., 2000) is used to provide information on the functioning of the two systems (Havstad and Herrick, 2003). Remote sensing assessment is used along with field data to enhance sampling and site representation (Booth et al., 2005). Current and anticipated future capability of moderate-resolution multispectral satellite systems do not provide the level of detailed identification of species and community type and productivity measurements required for similarity index calculations (Hunt et al., 2003), a desired choice of method for comparative studies such as ours. Band ratios including Normalized Difference Vegetation Index (NDVI) are among the possible options available. NDVI, the most commonly used band ratio, however, has important limitations (Philips et al., 2008), although it has been widely used in rangeland studies (Anderson et al., 1993, Purevdorj et al., 1998, Bayarjargal et al., 2000, Fukuo et al., 2001, Wylie et al., 2002, Zha et al., 2003, Kensuke et al., 2005, Bayarjargal et al., 2006, Erdenetuya and Khudulmur, 2008) with varying levels of success (Maynard et al., 2007). As vegetation cover decreases, NDVI becomes increasingly sensitive to the effects of bare soil (Richardson and Wiegand, 1977, Gao et al., 2000). Rangelands often have some amount of bare soil, especially in semi-arid environments such as our two study sites. NDVI is, therefore, expected to be impacted. Exactly how much bare soil can be present to warrant the successful use of NDVI in rangelands, however, is not well documented. The current literature lacks quantitative estimates of how much bare soil should be present in rangelands for NDVI to be useful. Here we present estimates of NDVI correlation with plant cover and biomass at point locations with varying amounts of bare soil exposure at the two study sites over two growing seasons. We evaluate the statistical significance and the portion of the variability in vegetation cover that NDVI can explain in three bare soil cover classes of up to 30 percent bare soil exposure.

METHODS

Study site description

Two sites were selected for this study, one representing the grasslands of northern Mongolia and one representing the shrub steppe of western USA. While latitude, elevation, topography, climate (e.g. extreme continental climate with cold winters and short summers), and some of the plant species are similar between the two sites, there are differences between the two sites in some variables (e.g. patterns of precipitation) in addition to the grazing systems (Figure 1).

Tsahiriin tal, northern Mongolia

Tsahiriin tal is a small valley located within the southern portion of the Darkhad Valley (51º2’17”N, 99º19’42”E) within Renchinlhumbe county of Khuvsgul province (Figure 1). The valley is used by 34 nomadic herding households as a summer pasture for 3 months at 1.191 AUM/ha grazing intensity with multiple livestock species of sheep, goats, cattle (includes yaks), and horses. Mean annual precipitation is less than 300 mm and monthly average temperatures ranges from less than -30 Cº in winter to close to 15 Cº in summers. Common plant species are *Poa pratensis* L., *Artemisia mongolica* (Fisch. ex Bess) Nakai, *Artemisia frigida* Willd., *Potentilla acaulis* L., and *Stipa krylovii* Roshev. The Tsahiriin tal valley floor consists of relic alluvial channels, terraces, and plains, as well as areas with closed depressions and hummocky rises. Soil parent materials are predominantly alluvial and lacustrine sediments. Calcareous grassland soils with organic-rich surface horizons are dominant throughout the site.

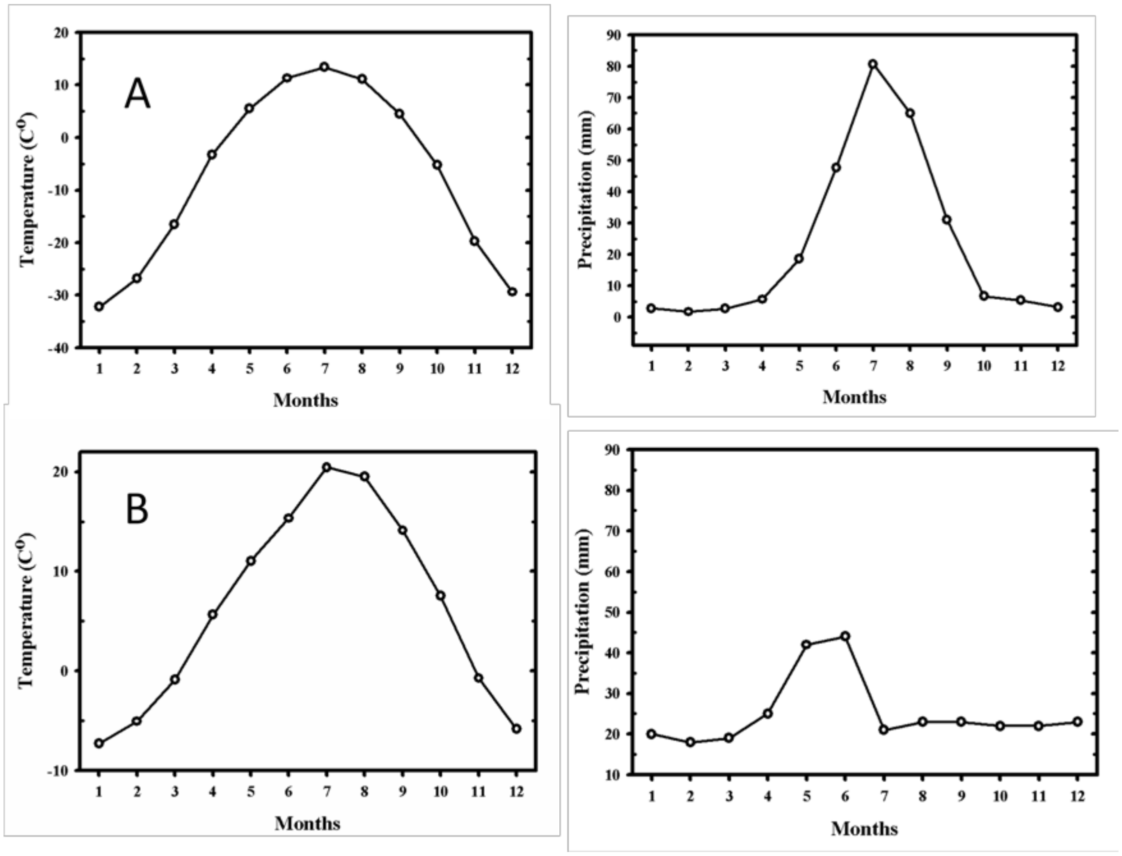

Figure 1. Long-term monthly averages of temperature and precipitation in Tsahiriin tal, northern Mongolia and the USSES, western USA.

US Sheep Experiment Station, western USA

This study site is the northwest portion of the US Sheep Experiment Station (USSES) headquarters (44º14’44”N, 112º12’47”E) rangeland. The USSES is grazed by only sheep during the spring and fall seasons at <0.62AUM/ha grazing intensity. The area is dominated by mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle) with subdominant shrubs of antelope bitterbrush (*Purshia tridentata* [Pursh] DC.), spineless horsebrush (*Tetradymia canescens* DC.), and yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.). The understory is cool season grasses and forbs including bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve), Sandberg bluegrass (*Poa secunda* J. Presl), and arrowleaf balsamroot (*Balsamorhiza sagittata* [Pursh] Nutt.). Mean annual precipitation is approximately 326 mm. The soils are a complex of sandy loam aeolian deposits of varying depth over lava flows.

Field methods

At each site, two seasons of field work were completed during the months of July and August in 2007 and 2008. Prior to field work, 100 random points were generated across each study area using Hawth’s tool in ESRI® ArcMapTM 9.2 software (ESRI Inc, 1999-2006). The same set of points was visited each year by navigating with a Trimble GeoXT GPS receiver with ± 3 m real-time horizontal accuracy. At each point, estimates of percent cover of shrub, litter, herbaceous cover, bare soil, and rock (coarse fragments > 75 mm) were made within a 10 m by 10 m plot centered on the point and aligned in the cardinal directions. Point-intercept method was used along two 10 m line transects that were oriented perpendicular to each other and intersected at the center of the plot at 5 m along each transect. Observations were recorded at every 20 cm along each 10 m line, beginning at 10 cm and ending at 990 cm, to indicate the cover type at the point. This resulted in 50 point measurements for each line and 100 point measurements for each plot. All herbaceous plants within a 0.44 m^2 cable hoop randomly tossed within each quadrant of each plot were clipped and weighed (without oven drying) to estimate total plant biomass. An average of the four biomass measurements was estimated for each plot. A subsample of the biomass samples were randomly selected from the set of all samples across each study site. These samples were dried to estimate the weight difference between wet and dry biomass samples. On average, 49.96 % (SD+5.02) of the weight was lost during drying. These differences were subtracted from all wet weights to convert the wet biomass estimates to dry biomass estimates.

Image analysis

To assess plant cover and biomass productivity, SPOT4 and SPOT5 images from early, middle, and late growing seasons in 2007 and 2008 were acquired (Table 1). All images were corrected for atmospheric effects using Idrisi's ATMOSC module (based on Chavez (1996) cos(t) model) and were projected in UTM Zone 47 North with WGS 1984 datum and Idaho Transverse Mercator with NAD 1927 datum for Tsahiriin tal and USSES, respectively. All images were co-registered to a georectified image source and then subset to the study sites. NDVI was estimated in each image subset using ENVI software (ENVI Version 4.3, ITT Industries Inc, 2006, Boulder, CO). NDVI values at the 100 random points within each study site were then extracted from each image for statistical analysis.

STATISTICAL ANALYSIS

Field-based biomass and green vegetation percent cover estimates were correlated with SPOT NDVI estimates from each date at the 100 sample locations at each study site using a simple linear regression model. To evaluate the SPOT NDVI prediction of rangeland biomass and green vegetation percent cover, coefficient of determination and p-values were summarized. Next, the 100 sample locations were subdivided into bare soil cover classes: 0-10%, 10-20% and 20-30% bare soil. There were only a few point locations (<10 at each site) where bare soil cover exceeded 30% and these points were excluded. The correlation between SPOT NDVI and vegetation percent cover was then evaluated by estimating coefficient of determination and p-values for each bare soil cover class to determine if NDVI is increasingly sensitive with increasing bare soil cover and becomes statistically insignificant with greater bare soil.

RESULTS

At the Tsahiriin tal study site, field-based mean green vegetation cover was 68 (SD±11.8) percent in 2007 and 49 (SD±7.6) percent in 2008. Field-based estimate of average dry forage was 712 kg/ha in 2007 and 605 kg/ha in 2008. There was no correlation between Aug 08, 2007 and June 02, 2008 NDVI estimates and field-based biomass estimates. Correlations were poor, when statistically significant correlation was found between May 10, 2008 and Aug 09, 2008 NDVI estimates and field-based biomass estimates (Table 1). Similar pattern was observed in the correlations between NDVI estimates and green vegetation percent cover estimates. The highest coefficient of determination was a low 0.20 (p-value <0.001) (Table 1).

At the USSES study site, field-based mean green vegetation cover was 74.8 (SD±18.3) percent in 2007 and 67.6 (SD±21.4) percent in 2008. Field-based estimate of average dry forage was 243.3 (SD+259.8) kg/ha in 2007 and 182.6 (SD±214.5) kg/ha in 2008. There was no or little correlation between NDVI and field-based biomass estimates (Table 1) and no correlation between NDVI and green vegetation percent cover, except for June 29, 2007 NDVI which explained 47 percent of the variability in vegetation cover.

Table 1. SPOT NDVI correlation with rangeland biomass and green vegetation percent cover at 100 sample points at each study site

Images	Associated timing in the growing season	Correlation with rangeland biomass (R^2 (p-value))	Correlation with vegetation percent cover (R^2 (p-value))
Tsahiriin tal study site, Mongolia			
SPOT4, Aug 08, 2007			
SPOT5, May 10, 2008	Late growing season	0.006 (p=0.432)	0.020 (p=0.163)
SPOT5, June 02, 2008	Early growing season	0.110 (p=0.001)	0.145 (p<0.001)
SPOT5, Aug 09, 2008	Mid growing season	0.018 (p=0.185)	0.043 (p=0.040)
	Late growing season	0.143 (p<0.001)	0.205 (p<0.001)
USSES study site, USA			
SPOT5, April 28, 2007	Early growing season	0.044 (p=0.038)	0.004 (p=0.557)
SPOT5, June 29, 2007	Mid growing season	0.182 (p<0.001)	0.474 (p<0.001)
SPOT5, Sep 15, 2007	Late growing season	0.001 (p=0.791)	0.001 (p=0.771)
SPOT5, June 28, 2008	Mid growing season	0.041 (p=0.044)	0.005 (p=0.478)
SPOT5, Aug 18, 2008	Late growing season	0.030 (p=0.087)	0.000 (p=0.973)

When correlations between green vegetation percent cover and NDVI were examined in different bare soil classes in Mongolia, the Aug 08, 2007 NDVI estimates were not statistically significant as a predictor variable in all bare soil classes (Figure 2, panel A). The NDVI estimates from summer 2008 were statistically significant as a predictor variable in all bare soil classes of 0-10% and 10-20%, except for the June 2 NDVI in 10-20% bare soil class. The 2008 NDVI estimates explained 3.8-36.6 percent of the variability in vegetation cover (Figure 2, panel A). All 2008 NDVI estimates, however, were no longer statistically significant as a predictor variable in the bare soil class of 20-30%.

At the USSES study site, all 2007 NDVI estimates were statistically significant as a predictor variable in 0-10% and 10-20% bare soil classes, except for the April 26 and Sep 15 NDVI in 10-20% bare soil class. The 2007 NDVI estimates, however, were not significant in all 20-30% bare soil classes, except for the June 29 NDVI. None of the 2008 NDVI estimates was statistically significant as a predictor variable for any of the bare soil classes (Figure 2, panel B).

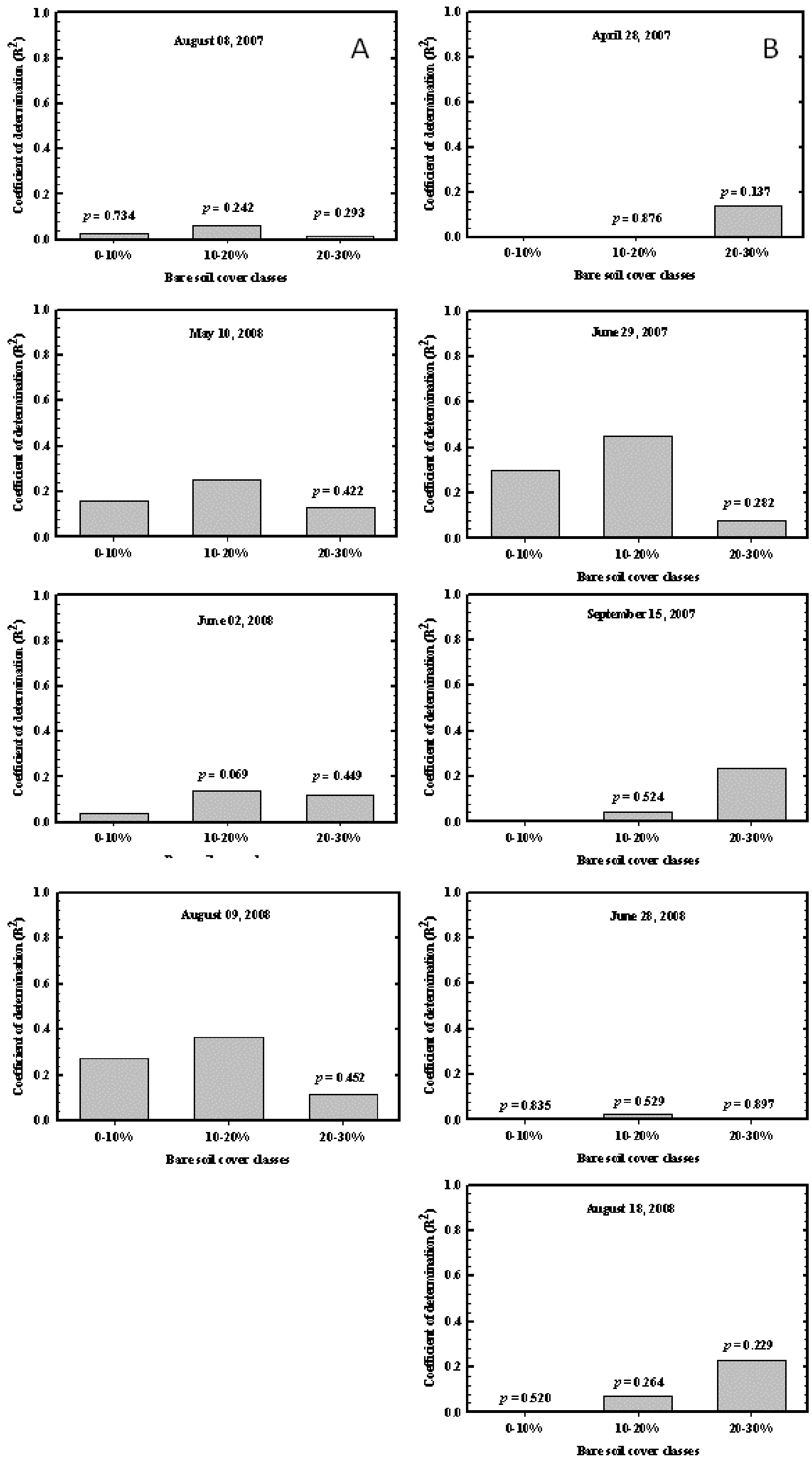

Figure 2. SPOT NDVI prediction of green vegetation cover in Tsahiriin tal, northern Mongolia (panel A) and the USSES, western USA (panel B). P-values are provided in cases where NDVI estimates were not statistically significant as a predictor variable.

DISCUSSION

The results of this study in the Tsahiriin tal valley of northern Mongolia and the USSES in the western USA indicate two patterns in NDVI correlation with vegetation cover and biomass. Firstly, NDVI correlation with vegetation percent cover and biomass in both years was absent in many cases at both sites. When NDVI was statistically significant as a predictor variable, the correlation was very poor. Our best NDVI correlation with vegetation cover during the two years yielded a coefficient of determination of 0.47 at the USSES and 0.20 at the Tsahiriin tal valley. The best coefficient of determination between NDVI and biomass was a low 0.14 in Tsahiriin tal and a low 0.18 at the USSES during this two year study. While NDVI estimates from the two sites were not expected to be similar due to differences in grazing management as well as other potential factors, NDVI was expected to be correlated with either field-based vegetation cover estimates or biomass estimates at each site. Despite careful georegistration and simultaneous or nearly simultaneous timing of field data collection and image acquisition, NDVI and field-based plant cover and biomass estimates produce poor correlation at both rangeland sites. Although NDVI has been successfully used in other studies, our study results indicate that NDVI might be poorly correlated with vegetation cover and biomass in semi-arid rangeland sites with little local-scale variability. NDVI correlation with vegetation cover and biomass might be greater in areas with various biomes and community types. However, our rangeland sites each represent a single biome with little variability in life-form and species distribution.

Secondly, NDVI appears to be increasingly impacted by the amount of bare soil present. NDVI estimates at our study sites were mostly significant as a predictor variable of vegetation cover in 0-10% bare soil cover class, and sometimes significant in 10-20% bare soil cover class, and almost never significant in 20-30% bare soil cover class. These results indicate that in semi-arid rangeland sites NDVI sensitivity increases with increasing bare soil cover. At our study sites, when bare soil cover reaches 20-30%, NDVI appears to reach a threshold where it is no longer statistically significantly correlated to vegetation cover. This result has important implications for future use of NDVI in semi-arid rangeland sites. NDVI might not be a useful estimate of vegetation cover at sites with greater than 20% bare soil.

CONCLUSIONS

The results from two biophysically similar semi-arid rangelands over two years with different grazing management systems indicate that NDVI is not well correlated with field-based estimates of green vegetation cover and biomass. The statistical significance of NDVI as a predictor variable of vegetation cover decreases as bare soil cover increases. When bare soil reached or exceeded 20% at our study sites, NDVI was no longer statistically significant as a predictor variable of any vegetation characteristic tested in this study. These results suggest that NDVI might not be a useful estimate of vegetation cover in semi-arid rangelands, especially when bare soil cover is >20 percent.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Sankey, T. T.and K. T. Weber. 2009. *Rangeland Assessments Using Remote Sensing: Is NDVI Useful?* Pages 113-122 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

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Woody-Herbaceous-Livestock Species Interaction

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ABSTRACT
Woody-herbaceous-livestock species interactions have attracted a great deal of research attention due to global land cover changes and increasing livestock production systems around the world. It has been well recognized that many local and regional factors, physical and biological, influence the interaction between woody and herbaceous species. While impacting the dynamics and tipping the balance, between woody and herbaceous species, livestock grazing effects also interact with these factors leading to various ecosystem states and woody/herbaceous ratios. The result is a complex set of multiple interacting factors that are difficult to experimentally control in long-term studies at large spatial scales. Ecological processes and empirical relationships observed in woody-herbaceous-livestock interactions, therefore, have largely been developed based on site-specific, local-scale studies emphasizing limited number of factors, processes, and relationships. Many of the proposed processes and empirical relationships have not been explicitly tested outside of the areas where they were developed. Future studies need to use such site-specific data in quantitative models and simulation-based approaches and test the validity of empirical models that are based on local data and relationships.

KEYWORDS: woody species, herbaceous species, livestock grazing, interaction

INTRODUCTION

Woody and herbaceous species interaction in disturbed and natural environments has attracted a great deal of research attention due to its implications for land cover change, land surface-atmosphere interaction, global carbon cycle (House et al., 2003), biodiversity, primary and secondary productivity, and the associated land use management (Archer, 1994). Ecosystems of mixed woody and herbaceous plants comprise 15-35% of the terrestrial surface area and are distributed from hot tropical to cold temperate climates across varying topography and soils (House et al., 2003). Mixed woody-herbaceous ecosystems are often heavily impacted by natural and anthropogenic factors such as fire and grazing (House et al., 2003).

Livestock grazing is known as one of the major factors that influence woody and herbaceous species interaction (Werner, 1990) throughout the world, as livestock grazing occupies 25% of the global land surface (Asner et al., 2004). Over the last 300 years, livestock grazing systems have increased 600% in extent and are projected to continue to increase with growing global human population and the associated increase in demand for meat and dairy products (Asner et al., 2004). A recent review of livestock grazing effects and ecosystem responses by Asner et al. (2004) identifies three major responses of ecosystems to livestock grazing observed at regional scales: desertification, woody species encroachment, and deforestation. In Asner et al.'s definition, desertification refers to grassland and steppe conversion to desert shrubland in arid regions of the world, while woody encroachment refers to grassland conversion to savanna and woodland in semiarid regions. In this review, both ecosystem responses are discussed and combined into a single term woody encroachment.

Livestock grazing interacts with multiple other physical and biological factors at various spatial and temporal scales while influencing woody-herbaceous species balance. The complex interaction of livestock grazing with other factors such as climate, topography, fire, and soils has often made it difficult to quantitatively assess livestock grazing effects on woody-herbaceous species interaction at decadal and centennial scales (Archer, 1994). To fully understand livestock grazing effects, all variables need to be controlled simultaneously in various environments and at a range of spatial and temporal scales. Due to the logistics involved in such a study and the lack of quantitative historical data, most ecological research is not able to do this. The current knowledge of woody-herbaceous-livestock species interaction is largely based on short-term studies and ecological investigations of only one or two variables with limited control on other potential factors.

When woody-herbaceous species balance is disturbed, one of the two life-forms is likely to dominate the other and a shift occurs in the density of woody and herbaceous plants and the location of woody-herbaceous boundaries, known as ecotones. Proximate causes of shifts in woody-herbaceous ecotones have been studied in many different parts of the world to understand the dynamics and balance between woody and herbaceous species. A few of these causes have been widely agreed upon to be the main driving factors. Most conceptual models of woody-herbaceous balance shifts acknowledge the interactive effects of multiple factors rather than a single driving force (Daly et al., 2000; House et al., 2003; Kupfer and Miller, 2005). Among them are climate change, increased CO_2, nitrogen pollution, drought, fire suppression, and grazing (Dando and Hansen, 1990; Archer, 1994; Bachelet et al., 2000; Bartolome et al., 2000; Asner et al., 2004). Topographic slope and aspect, snow accumulation, and soil texture and depth further influence changes in the balance and determine spatial patterns of the balance shift (Brown, 1994; Walsh and Butler, 1994; Kupfer and Cairns, 1996).

Woody-Herbaceous Species Interactions and Associated Models

Mixed woody-herbaceous communities are diverse in composition, structure, functional forms, and spatial patterns due to their wide-spread distribution across the world (House et al., 2003). The co-existence of woody and herbaceous species and the key driving factors that facilitate the co-existence have been well studied (Figure 1), although different conceptual models emphasize different driving factors (Belsky, 1990). Woody and herbaceous species can influence each other in many different ways and the effects can be expressed in various forms.

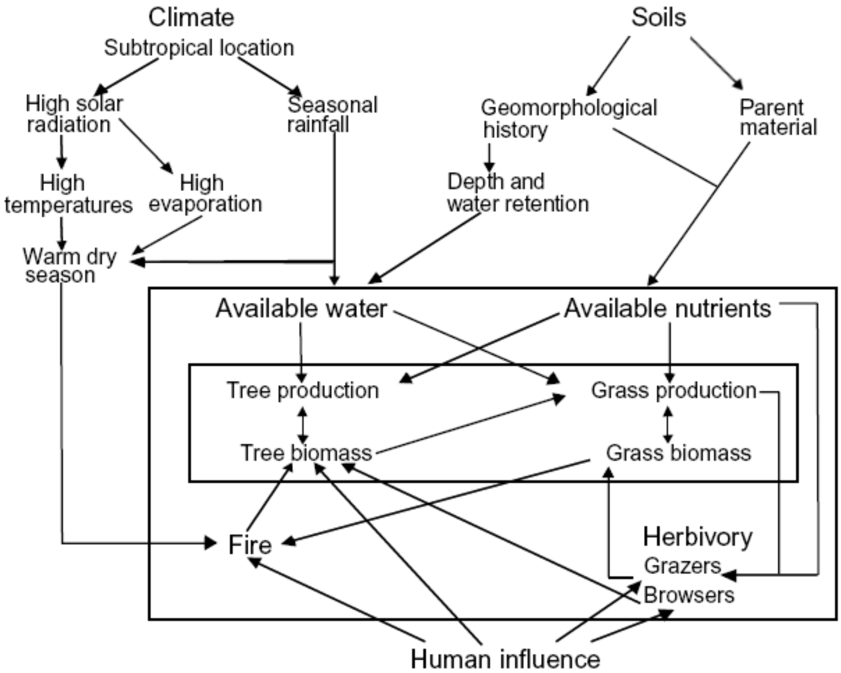

Figure 1. Key driving factors for mixed woody-herbaceous systems (from House et al., 2003). Water, nutrients, fire, and herbivory are defined as the determinants of structure and function in woody-herbaceous systems and they collectively affect the innermost level, the balance between tree and grass.

The effects of woody plants on herbaceous species can be positive, neutral, or negative depending upon the characteristics of the woody and herbaceous growth-forms, ecophysiological features, photosynthetic pathway (C_3 versus C_4) and habit (deciduous versus evergreen), and water and nutrient requirements (Scholes and Archer, 1997 and references therein). The effects of woody plants can also be expressed in varying forms. Firstly, woody plants can affect herbaceous species composition (Burrows et al., 1990). In mixed woody-herbaceous communities, herbaceous species composition under a tree canopy might be very different compared to that in the inter-tree space. C_3 grasses might be found mostly under the tree canopy, while C_4 grasses might dominate in-between trees in subtropical and temperate regions. Furthermore, herbaceous species composition can vary under the canopy from the tree trunk to the edge of the canopy (Scholes and Archer, 1997). Secondly, woody species can influence herbaceous species production, biomass allocation, and phenology. Trees can often reduce herbaceous species biomass production (Burrows et al., 1990). However, herbaceous biomass production under tree canopies can also increase (Burrows et al., 1990) due to improved nutrient supply, reduced evapotranspiration (Reid and Ellis, 1995), and increased water availability (Walker et al., 1981). Alteration of the geologic parent material and soil characteristics and the improvement of harsh environmental conditions are considered other facilitation effects of woody species for herbaceous plants. These facilitation effects might not be observed for many years after tree establishment, because the effects are dependent on tree size, age, and density and are not obvious until woody species reach a critical size and age (Scholes and Archer, 1997). In other cases, trees/shrubs can have facilitation effects for herbaceous plants only when they are young. As the trees/shrubs grow bigger, the facilitation effects might be outweighed by their competition effects on herbaceous plants. This competition often results in a strong, negative correlation between tree density or cover and grass cover or biomass (Stuart-Hill and Tainton, 1989). Herbaceous production and diversity, therefore, might be low at high tree/shrub density (Burrows et al., 1990). The negative correlation might be due to the tree/shrub litter accumulation (which might increase soil acidity), canopy shading, reduced rainfall under the canopy, and root competition.

The effects of herbaceous species on woody plants are most critical during woody seedling establishment stage, although the effect can be variable. Firstly, herbaceous species can impact woody seedling establishment and recruitment directly by effectively competing for light, water, and nutrients (Knoop and Walker, 1985). The competition can prevent woody seedling emergence, increase the mortality of newly established woody seedlings, and reduce woody seedling growth and recruitment. Even the growth of mature woody plants can be reduced by herbaceous species competition for water in wetter years, when herbaceous biomass is high (Knoop and Walker, 1985). Secondly, herbaceous species can influence woody seedling recruitment indirectly (Scholes and Archer, 1997 and references therein). Herbaceous species biomass can increase fine fuel loads, which increases fire frequency and intensity, leading to increased mortality of small woody seedlings that are especially vulnerable to fire (Dando and Hansen, 1990; Archer, 1994). However, the direct and indirect influences of herbaceous species on woody plants are often not enough to completely exclude woody plants and to prevent woody encroachment. Woody plants still might be able to expand into adjacent grassland with a wide range of herbaceous species composition and production (Scholes and Archer, 1997). Woody plants can establish during wet periods, when competition from herbaceous species are limited. Once woody seedlings establish and grow beyond the height of the herbaceous layer, they can establish vertical dominance and herbaceous species might have little or no influence on them. Scholes and Archer (1997) summarize that experimental studies in savanna environments largely found no significant effects on woody species, when herbaceous plants were cut and cleared. Only on fine textured soils with greater clay content herbaceous species appeared to limit water recharge from rainfall deeper in the soil profile where tree roots uptake water.

In mixed woody-herbaceous communities, the interaction between woody plants themselves has been considered important. Tree-tree interaction or shrub-shrub interaction can lead to competition for belowground resources such as water and nutrients as well as competition for light. This intraspecific competition is often assumed to lead to self-thinning and ultimately a regular spatial pattern of woody plants. Clumped and random spatial patterns are also possible in savanna tree distribution due to fire effects, topography, soils, and resource patchiness (Scholes and Archer, 1997 and references therein). Clumped and random spatial patterns can be associated with some level of facilitation effects such as increased seed dispersal and improved environmental conditions under canopies and nearby existing trees/shrubs.

Three different types of models describe woody-herbaceous species interaction and coexistence, particularly in savanna ecosystems: niche separation models, balanced competition models, and disequilibrium models (Scholes and Archer, 1997 and references therein). Niche separation models are based on the assumption that a variable, such as water, is a limiting factor and woody and herbaceous species, therefore, have to use resources at different times or places (House et al., 2003). For example, grasses and shrubs can have different root systems at different depths in the soil profile so that they can use water at different soil depths to coexist (Walker et al., 1981; Knoop and Walker, 1985).

Balanced competition models are based on the concept of intraspecific competition, which is assumed to be stronger than interspecific competition (House et al., 2003). In other words, competition between herbaceous species is assumed to be stronger than competition between woody and herbaceous species. Likewise, competition between woody species is assumed to be stronger than competition between woody and herbaceous species. The result would be woody species that outcompete herbaceous species and establish dominance or herbaceous species that outcompete newly establishing woody seedlings and prevent woody establishment and encroachment. Balanced competition models, therefore, predict two stable states: woodland and grassland. Similar to the balanced competition models, Walker and Noy-Meir's 1982 model predicts the two stable states after adding grazing as a factor to a niche separation model centered on soil water, which was initially proposed by Walter in 1971 (Jeltsch et al., 2001).

Following the simpler models that predict stable equilibrium of woody and herbaceous vegetation, newer concepts emerged modeling and predicting non-equilibrium dynamics in the woody-herbaceous interactions (Jeltsch et al., 2000). Equilibrium models might explain the co-existence of woody and herbaceous species at smaller scales, whereas disequilibrium models are more appropriate for describing landscape- and decadal-scale dynamics (Sharp and Whittaker, 2003). Disequilibrium models predict cycles and oscillations in the relative abundance of woody and herbaceous species at larger scales (Sharp and Whittaker, 2003). They suggest that mixtures of woody and herbaceous species only exist due to disturbances such as fire and grazing and, therefore, represent a transitional state between the possible stable states (Jeltsch et al., 2000). Further development of the disequilibrium models predicts multiple stable states with varying tree-grass ratio (House et al., 2003). Disequilibrium models have also been extended to include a spatial aspect and a concept of patches of disequilibrium which result from stochastic processes such as gap dynamics (Jeltsch et al., 2000).

Woodland and Grassland Stables States and Conceptual Models

Ecosystems respond in different ways when external conditions change over time. Some ecosystems might respond gradually in a smooth, continuous manner, while others respond abruptly, especially after a certain threshold is passed in external conditions (Figure 2). Many different ecosystem studies have demonstrated the existence of alternative stable states and multiple stable states in different environments (Werner, 1990; Scheffer et al., 2001 and references therein). Studies of woody-herbaceous interactions have been common among such research demonstrating two possible alternative states: woodland and grassland. Scheffer et al. (2001) term the changes between the two alternative stable states "catastrophic shift", because shifts occur very rapidly and there is often no "early warning signals". Moreover, it is extremely difficult to recover an ecosystem after such shifts and many ecosystems remain in the new alternative state (Walker et al., 1981; Sharp and Whittaker, 2003), even if previous environmental conditions are restored.

Numerous studies documented a shift from a grassland stable state to a woodland stable state (Figure 3) in Africa, south-western USA, drier parts of India, and in Australia (Walker et al., 1981), while shifts from a woodland state to a grassland state have also been observed (Burrows et al., 1990). African grasslands, for example, were kept open by herbivory and fire until herbivore numbers drastically declined allowing successful establishment of woody plants. Once successfully established and recruited, the woody species were no longer kept in balance by herbivores and their canopy shade reduced herbaceous biomass accumulation, which then reduced fire frequency. Reduced fire frequency further increased successful woody establishment and encroachment (Scheffer et al., 2001). Returning this ecosystem to the grassland stable state would require drastic measures taken at substantial spatial and temporal scales. In contrast, conditions in dry environments can enable a shift from a woodland stable state to a grassland stable state (Scheffer et al., 2001 and references therein). In dry environments, if well-established tree populations are heavily disturbed and killed due to fire and other factors, conditions might be too harsh to allow woody seedlings to establish in the absence of nurse trees and herbaceous species might dominate. Restoring the woodland stable state might require a rare combination of adequate precipitation and reduced grazing effects.

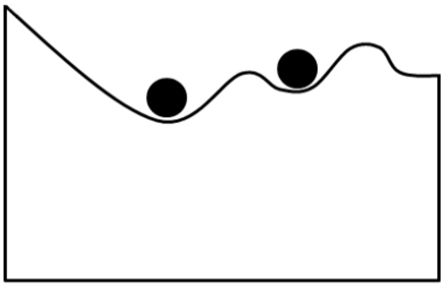

Figure 2. Alternative stable states and their basins of attraction (modified from Scheffer et al., 2001). Stable equilibria correspond to the valleys or attraction basins, while unstable transitional periods correspond to the hill between the valleys. If the size of the attraction basin is small, ecosystem resilience is small and even small changes in the external conditions might move the system into an alternative stable state.

The dynamics and shifts between woodland and grassland are dependent upon processes and mechanisms that influence the resistance, resilience, and persistence of the associated woodland and grassland ecosystems. Both ecosystems create positive feedbacks to persist. Such positive feedback mechanisms in woodlands include tree suppression of grass through shading (Menaut et al., 1990), increased seed input within areas around tree patches (Archer, 1990), and reduced fire frequency (Archer, 1990; Menaut et al., 1990). Woody-herbaceous ecotones exist as a result of a balance in such feedback mechanisms. Equally strong persistence and feedback mechanisms of the two ecosystems create a stable ecotone that does not change rapidly in space over time. In contrast, imbalance in the feedback systems might result in unstable conditions over time (Werner, 1990) and constant fluctuations in ecotones.

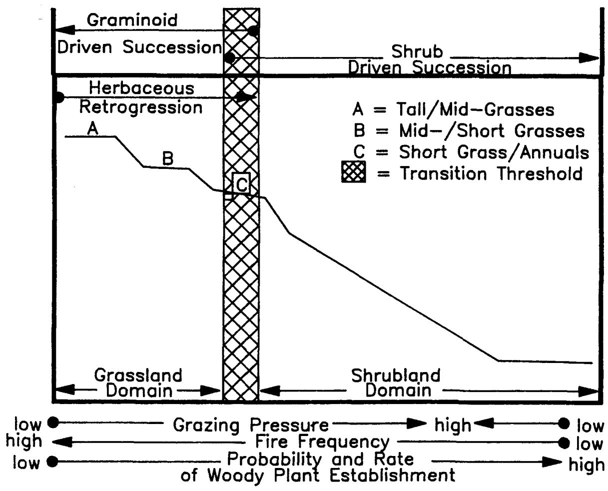

Figure 3. A conceptual model of shrubland and grassland stable states and a transition between the two states (from Archer, 1994). This model demonstrates the conversion from a grassland stable state to a shrubland stable state and the existence of a threshold in livestock grazing pressure.

Woody-Herbaceous Ecotones

Ecotones play a vital role in understanding the interaction between woody and herbaceous species. Clements defined ecotone in 1905 as "the junction zone between two communities, where the processes of exchange or competition between neighboring formations might be readily observed" (Holland and Risser, 1991). The current definition of ecotone is a "zone of transition between adjacent ecological systems, having a set of characteristics uniquely defined by space and time scales, and by the strength of the interactions between adjacent ecological systems" (Holland and Risser, 1991). The ecological importance of ecotones and their roles in understanding global environmental changes have long been recognized (Holland and Risser, 1991). Ecotone characteristics, including their location, size, shape, and composition are more sensitive to global environmental changes than those of homogenous landscape units (Turner et al., 1991). Ecotones thus provide good early indicators of such changes (Gosz, 1991).

Biophysical characteristics of forest-grassland ecotones are defined by a complex interaction of biotic and abiotic factors, including plant interactions, disturbance regime, physiography, topography, geologic parent materials, soil properties, and climate variables (Alverson et al., 1988; Tilghman, 1989; Smit and Olff, 1998; Carmel and Kadmon, 1999; Mast and Veblen, 1999; Zald, 2002). Changes in these factors can have a substantial impact on woody-herbaceous ecotones and cause a shift in their location (Camarero et al., 2000, Taylor, 1995). Shifts in forest-grassland ecotones impact carbon sequestration and land surface-atmosphere interactions and have important implications for biodiversity, primary and secondary productivity, soil development, and populations and carrying capacity of both domestic and wild animals (Archer, 1994).

Rates and Patterns of Woody-Herbaceous Ecotone Shift

Previous studies of forest-grassland ecotones have demonstrated varying rates and patterns of forest encroachment into the adjacent grassland. Sankey et al. (2006) documented rates and patterns of forest encroachment using dendrochronological data with individual tree maps along a lower ecotone in southwestern Montana of western USA. The aspen (*Populus tremuloides*) and Douglas-fir trees (*Pseudotsuga menziesii*) along this ecotone appeared to encroach into the adjacent grassland at different rates and patterns (Sankey, 2007). A similar study was conducted in northwestern Mongolia (Sankey et al., 2006) and the results indicated that the dominant tree species along the northern Mongolian forest-grassland ecotones, Siberian larch (*Larix sibirica*), shifted into the adjacent grassland at different rates and patterns compared to aspen and Douglas-fir observed in the previous study. Neither of these studies documented a shift into the forest or a retreat in the forest boundary location during the 20th Century. The results of these studies indicate that three general patterns of tree encroachment into the adjacent grassland are evident (Figure 4).

Type I change is a shift in the forest-grassland boundary location into the adjacent grassland. This type of ecotone change results from a mechanism where new trees establish in the adjacent grassland advancing the ecotone location towards the grassland (Sankey et al., 2006). Type I ecotone change might mostly occur in systems where the dominant tree species regenerates through seed dispersal, although it can be observed in systems with vegetatively-reproducing species. For example, Douglas-fir is a seed-dispersed species. Its seeds are dispersed through wind, animals, and birds (Hermann and Lavender, 1965). Seeds usually fall within 100 meters from a seed tree or a stand edge, but they can fall 1-2 km away from the seed sources (Hermann and Lavender, 1965).

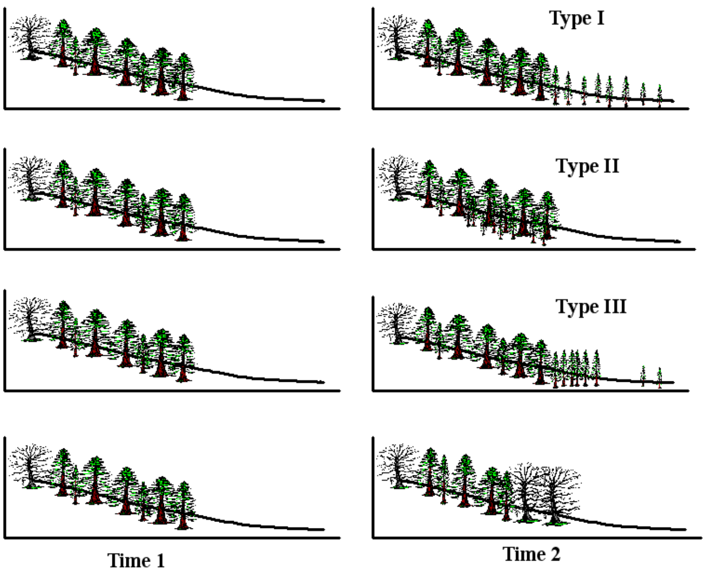

Figure 4. Patterns of tree encroachment into the adjacent grassland observed in grazed areas. Trees might encroach into the adjacent grassland in three different patterns. These patterns are not mutually exclusive and can occur simultaneously. Type I pattern is a shift in the forest-grassland ecotone location over time into the adjacent grassland. Type II pattern is tree density increase within the same forest-grassland boundary location. Type III pattern is fairy ring establishment that advances the forest-grassland boundary into the adjacent grassland. Forest-grassland ecotones might also retreat and herbaceous species might expand into the adjacent forest over time (bottom panel).

Type II change is an increase in tree density at the forest-grassland boundary (Arno and Gruell, 1986). During this change, new trees establish within the same boundary location and do not advance the boundary towards the adjacent grassland. Type II ecotone change occurs in systems where tree establishment sites are available under the forest canopy, but no establishment occurs outside of the forest boundary due to unfavorable site conditions and disturbance. Type II ecotone change can occur with Type I ecotone change at the same time, if conditions outside of the forest boundary change allowing Type I ecotone change. Type II change can be observed in systems with both tree species that regenerate vegetatively and through seeds. For example, Douglas-fir, aspen, and Siberian larch all can result in Type II ecotone change.

Type III change is an establishment of a fairy ring in the grassland along the edge of the forest (Sankey, 2007). This change occurs when new trees establish as a fringe in the grassland, adjacent to the forest boundary. Fairy rings consist of new stands of densely distributed new stems of similar age. Fairy ring establishment is often associated with wave regeneration mechanisms. During this mechanism, new regenerations occur as pulses advancing the forest boundary into the adjacent grassland. Both seed-dispersed and vegetatively-reproducing species can regenerate in pulses. For example, aspen regenerates in pulses after a fire disturbance event (DeByle and Winokur, 1985), whereas Siberian larch can regenerate in pulses following reductions in grazing disturbance forming a fringe (Didier, 2001).

Woody-Herbaceous-Livestock Species Dynamics

Most ecological processes in woody-herbaceous species dynamics can be impacted by herbivory, an important local control over vegetation. Vertebrate and invertebrate herbivores can regulate plant cover types, their composition, structure, and productivity (Alverson et al., 1988; Tilghman, 1989; Mast et al., 1997; Carmel and Kadmon, 1999; Mast and Veblen, 1999; Bachelet et al., 2000; Bartolome et al., 2000; Scheffer et al., 2001; Wahungu et al., 2002). Herbaceous and woody plant species react to herbivory differently due to their differences in tolerance to grazing and palatability (Archer, 1994). This makes their interaction in grazed environments more complex than in undisturbed environments. This complexity has generated abundant interest in the interaction between herbaceous and woody plant species and, in particular, the changes from herbaceous vegetation cover to woody species cover due to grazing. Some studies show that the processes of woody species seedling emergence, growth, and survival are facilitated by grazing (Walker et al., 1981; Dando and Hansen, 1990; Reid and Ellis, 1995; Archer, 1994; Sharp and Whittaker, 2003), while others suggest the seemingly conflicting result that these processes are inhibited by grazing (Reid and Ellis, 1995; Carmel and Kadmon, 1999; Bartolome et al, 2000). In most cases, the assumption is that co-existing herbaceous species and woody species within an ecosystem show the opposite trends under livestock grazing effects. If woody species increase with increasing grazing effects, herbaceous species are assumed to decrease and vice versa.

Woody species encroachment due to grazing has been demonstrated throughout the world, including southern Asia, Australia, Africa, South America, and North America (Walker et al., 1981; Archer, 1989). Ecological processes described in studies that proposed woody species encroachment due to grazing are: (1) Grazing decreases seed production, seedling establishment, biomass, and basal area of palatable herbaceous species and increases their mortality; (2) Reduced herbaceous species ground cover increases sunlight levels on the ground, which increases seed germination and early establishment of woody species seedlings; (3) Reduced herbaceous species biomass decreases fine fuel accumulation and reduces fire frequency, which increases woody species invasion; (4) Invading woody species are less palatable than herbaceous species and are not browsed enough to be eliminated; (5) Grazing makes herbaceous species less able to compete for resources and unable to limit woody species growth and their seedling establishment; and (6) Livestock disperse woody species seeds across the landscape, which facilitates woody species expansion (Archer, 1994).

The opposite effects of grazing on woody species establishment have also been demonstrated (Carmel and Kadmon, 1999; Bartolome et al., 2000). Studies of these effects suggest that grazing can inhibit tree seedling establishment, survival, and growth. Grazing, therefore, might be expected to control woody species encroachment into grasslands. Ecological processes described in studies of negative effects of grazing on woody species include: (1) Slow growth rate of most woody species allows repeated grazing in their seedling stage when they are most vulnerable to grazing (Alverson et al., 1988; Tilghman, 1989); (2) Intense grazing causes shoot loss, tissue damage, and biomass loss for woody species (Hjalten et al., 1993), which decreases their seedling growth (Alverson et al., 1988; Tilghman, 1989) and increases seedling mortality (Hjalten et al., 1993); (3) Increased seedling mortality reduces recruitment into the tree population (McInnes et al., 1992; Rooney et al., 2002); and (4) Trampling and rubbing against the bark by grazing animals damage woody species and their seedlings (Kay and Bartos, 2000).

The current literature indicates two seemingly conflicting linear relationships between grazing effects and woody species (i.e., increasing woody species with increasing grazing effects and thus decreasing herbaceous species or vice versa). The vast majority of grazing impact studies in the current literature, however, has compared only two or three levels of grazing intensity. Such comparisons of limited number of grazing levels might provide only a simple linear relationship between woody species establishment and grazing effects. The potential variability in woody establishment due to varying levels of grazing intensity needs to be investigated across a wider gradient of multiple levels of grazing intensity.

Sankey et al. (2006) examined aspen and Douglas-fir tree encroachment into the adjacent grassland in Montana, USA under ten different levels of livestock grazing intensity using non-experimental observational data of 60 years (Table 1). The objective of this study was to determine if tree establishment-livestock grazing relationship always had a simple linear trend as suggested by previous studies or if patterns were different at decadal time scales. When aspen tree establishment was analyzed with a gradient of all ten grazing levels, increasing in intensity from 1 to 10, grazing levels 2, 8, and 10 had significantly greater aspen establishment than all other grazing levels (p-value <0.001) (Figure 5 (a)). Grazing level 3 had significantly greater aspen establishment than grazing levels 4, 5-7, and 9 (p-value <0.001). There were no other significant differences in aspen establishment along this gradient. A similar test indicated that grazing level 1 had significantly greater Douglas-fir establishment compared to grazing levels 4 and 7 (p-value of 0.002) (Figure 5 (b)). There were no other significant differences in Douglas-fir establishment. There was no apparent trend of linear increase or decrease in aspen and Douglas-fir establishment with increasing grazing intensity along this gradient. A regression model of all 10 grazing levels (AUM ha^{-1}) and aspen establishment was built with a significant squared term (Figure 5 (c)). The statistically significant squared term might suggest a possibility of a curvilinear relationship between aspen establishment and grazing levels (Sankey et al., 2006). The regression model of all ten grazing levels and Douglas-fir establishment did not have a statistically significant squared term, but indicated a linear relationship between Douglas-fir and grazing levels with a trend of decrease (Figure 5 (d)). However, the regression model produced a low correlation coefficient and did not suggest a strong relationship.

Table 1. Long-term averages of grazing pressure estimated in AUM ha^{-1} in the Sankey et al. (2006) study in Montana, USA

Grazing levels	AUM ha^{-1}
Grazing level 1	0.00
Grazing level 2	0.11
Grazing level 3	0.12
Grazing level 4	0.17
Grazing level 5	0.28
Grazing level 6	0.41
Grazing level 7	0.50
Grazing level 8	0.79
Grazing level 9	1.00
Grazing level 10	2.00

The collective results of this study indicated varying relationships between tree establishment and livestock grazing intensity. The relationships were not always simple linear increase or decrease in tree establishment with increasing livestock grazing intensity, although linear relationships were observed in some cases. At decadal time scales, simple linear trends of inhibition and facilitation effects as suggested by previous studies did not appear to hold across varying gradients of grazing levels and two different tree species (Sankey et al., 2006). Indeed, complex curvilinear trends might be possible at decadal time scales across wider gradients of grazing intensity (Sankey et al., 2006). This is consistent with other studies that suggest that mixed woody-herbaceous systems can have non-linear trends in woody plant abundance and rates of change in tree abundance (McPherson, 1992; Archer et al., 1988; Miller and Wigand, 1994).

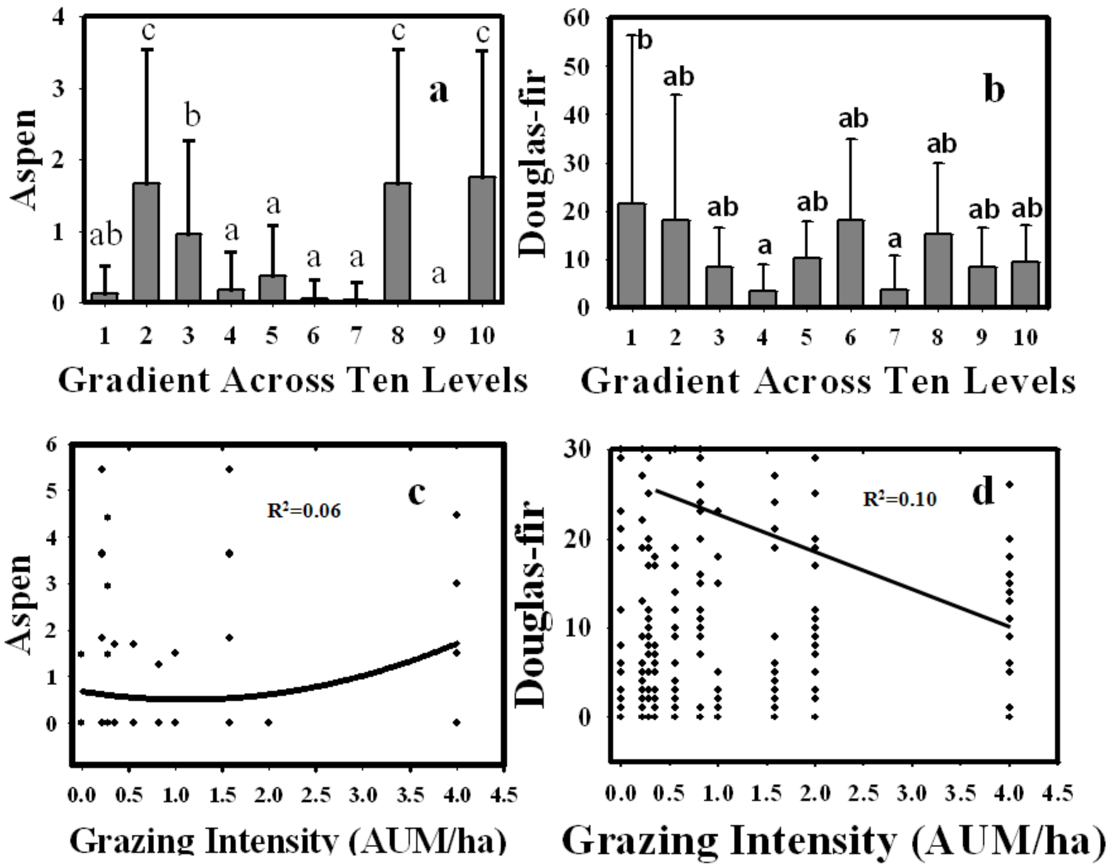

Figure 5. Aspen (*Populus tremuloides*) and Douglas-fir (*Pseudotsuga menziesii*) tree establishment with varying grazing pressure in Sankey et al. (2006) study in Montana, USA. Number of new tree stems established (expressed in percent) under each grazing level is on the Y-axis. The ten grazing levels on the X-axis in figures a and b correspond with categorical classes of grazing pressure, while the grazing levels in figure c and d are livestock grazing pressure estimated in Animal Unit Months per hectare (AUM ha^{-1}).

Previous studies collectively indicate that multiple trajectories might be observed in livestock grazing effects on woody species. Most of these trajectories might be difficult to discern in long-term studies with limited control on grazing treatments and other potential factors that influence woody-herbaceous species interaction. Better controlled experimental designs and process-based studies might allow conclusive tests of the possible trajectories that have been hypothesized in the current literature. Currently proposed, but not exclusively tested, trajectories in the relationship between woody establishment and livestock grazing effects include four possible trends: 1) facilitation effect or a simple linear increase in woody species with increasing grazing intensity (Figure 6 (a) (Archer, 1994), 2) inhibition effect or a simple linear decrease in woody species with increasing grazing intensity (Figure 6 (b)) (Carmel and Kadmon, 1999; Bartolome et al, 2000), 3) a curvilinear relationship in which inhibition effects dominate at low and high grazing intensities, but facilitation effects dominate at medium grazing intensities (Figure 6 (c)) (Cairns and Moen, 2004), and 4) a curvilinear relationship in which facilitation effects dominate at low and high grazing intensities, but inhibition effects dominate at medium grazing intensities (Figure 6 (d)) (Sankey et al., 2006).

The first two trajectories can be explained by the ecological processes described in the positive and negative effects of livestock grazing discussed earlier in this section. The first trajectory, facilitation effects, predicts decreasing herbaceous species and increasing woody species with increasing grazing intensity. This trend might be observed in systems where herbaceous species are palatable and are preferred by the livestock species over the woody species. The second trajectory, inhibition effects, predicts increasing herbaceous species and decreasing woody species with increasing grazing intensity. This trend might be observed in systems where woody species are palatable and are commonly grazed or browsed by livestock. In this case, woody establishment can be greatest under low grazing intensity. In the third trajectory, woody and herbaceous species are balanced at medium grazing intensity, but woody species are expected to decrease at low and high grazing intensities. This trend is expected to occur where grazing effects largely include woody species trampling, seed dispersal, and seed predation, but woody species foliage consumption is relatively low (Cairns and Moen, 2004). It is also expected to occur in systems where multiple grazing animal species are present (Cairns and Moen, 2004).

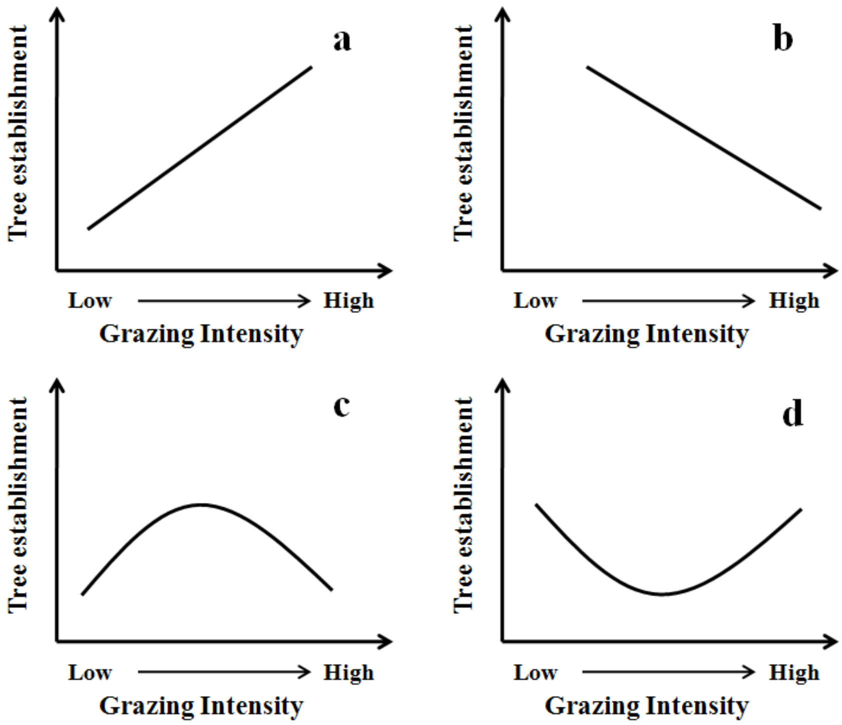

Figure 6. Currently proposed trajectories in the relationship between tree establishment and livestock grazing effects. Herbaceous species compete for similar resources and are assumed to show the opposite trend compared to trees. Livestock grazing can have facilitation or inhibition effects on tree establishment resulting in simple linear trends. Facilitation and inhibition effects can also dominate at varying levels of grazing pressure resulting in complex curvilinear trends.

In the fourth trajectory, woody and herbaceous species are hypothesized to be balanced at medium grazing intensity, but woody species are expected to increase at low and high grazing intensities. This trajectory might be explained by the grazing optimization hypothesis. The grazing optimization hypothesis, developed for herbaceous species, states that herbaceous species productivity increases with increasing grazing intensity at low grazing levels due to overcompensation (McNaughton, 1979). This trend continues up to a point called “the level of optimal grazing” and then declines with increasing grazing intensity at high grazing levels. Since co-existing herbaceous and woody species compete for largely similar resources, woody species might be expected to show the opposite trend. Therefore, at lower levels of grazing intensity, we might expect tree establishment to be relatively high. Even if woody species are grazed/browsed at this intensity, they can show increasing competitiveness due to stimulatory effects of grazing (Stuart-Hill and Tainton, 1989). Tree establishment might then decrease with increasing grazing intensity as herbaceous species productivity increases due to overcompensation. At medium levels of grazing intensity, tree establishment might reach its minimum because herbaceous species productivity is highest at these levels and, thus, tree competition with herbaceous species is greatest. At higher levels of grazing intensity, however, we expect tree establishment to increase because herbaceous species at these levels show a trend of decrease.

Walker et al. (1981) proposed conceptual models of woody-herbaceous-livestock interaction that the four trajectories in Figure 6 don't fully describe. Walker et al.'s conceptual models describe the stability of woody-herbaceous species balance under livestock grazing effects in semi-arid savanna ecosystems. In their definition, semi-arid savanna includes regions in which scattered to numerous trees/shrubs are distributed across continuous grass cover. They first describe the effects of livestock grazing on grass recruitment curve (Figure 7 (a)) based on McNaughton's (1979) grazing optimization hypothesis and suggest that grazing has the greatest stimulating effect at intermediate values of grass. They also describe the effects of woody vegetation on grass recruitment curve (Figure 7 (b)), because woody and herbaceous species compete for the same water resource in the top-soil in semi-arid regions (Walker et al., 1981). Their conceptual model suggests that small and medium amounts of woody species (W) have greater effects on high amounts of grass (G) than small amounts of grass. Walker et al. (1981) then describe the effects of grass on woody vegetation (Figure 7 (c)) and suggest that high values of grass (G) do not have strong effects on the woody vegetation recruitment curve, because woody species have exclusive access to subsoil water. Lastly, Walker et al. (1981) describe the zero-isocline or zero recruitment of grass as related to grass, woody species, and the effects of grazing (H) (Figure 7 (d)). The zero grass recruitment curve indicates the equilibrium between woody plants and grass under grazing effects. Similar to some of the previously discussed trajectories, this model suggests a curvilinear relationship in the woody-herbaceous-livestock species interaction. However, this model suggests varying curves with increasing grazing pressure. The equilibrium always indicates a similarly-shaped curve at varying grazing pressure, but the curve falls at varying amounts of total vegetation and varying woody-grass ratios. Unlike the other models, which largely assume unvarying total vegetation amount, this model assumes decreasing amounts of total vegetation with increasing grazing pressure. The assumption might depend on the relative palatability and tolerance of the plant species involved to grazing as well as the forage preferences of the grazing animal species. Total vegetation amount might decrease with increasing grazing pressure in some systems, while in other systems only the palatable species might decrease and unpalatable species might remain constant or increase with increasing grazing pressure.

Sankey et al. (2006) examined the relationship between Siberian larch forest-grassland ecotone shift and a gradient of five different livestock grazing regimes dominated by different livestock species (Table 2) in northern Mongolia (Sankey et al., 2006). The five grazing regimes varied in overall grazing intensity from 3.3 AUM ha^{-1} to 5.9 AUM ha^{-1}. They varied in species composition such that each site represented either sheep-dominance, sheep-goat mix, sheep-goat-cattle mix, or cattle-dominance (Table 3). Forage preferences between these animal species are known to be substantially different (Vallentine, 2001). Cattle are grazers and consume mostly graminoids. Sheep are intermediate feeders and consume grasses, forbs, and woody species. Goats are browsers and prefer leaves and tender twigs of new growth on trees and shrubs. The results indicated that Siberian larch forest-grassland ecotone response to grazing varied among different grazing regimes and tree establishment varied statistically significantly. The number of new trees established varied significantly between Site 2 (goat-sheep-dominated and low overall grazing intensity) and Site 4 (cattle-dominated and medium overall grazing intensity) and between Site 2 (goat-sheep-dominated and low overall grazing intensity) and Site 5 (sheep-dominated and high overall grazing intensity) (Figure 8). The number of new stems established also varied between Site 3 (cattle-sheep-goat mix and medium overall grazing intensity) and Site 4 (cattle-dominated and medium overall grazing intensity). There was no statistical difference between sheep-dominance at low and high overall grazing intensities. This might indicate that sheep, in general, do not have substantial negative effects on tree establishment regardless of sheep grazing intensity, which might be explained by their lower consumption of woody species compared to herbaceous species. The sites with high numbers of goats had lower tree establishment than all other sites with lower numbers of goats, regardless of overall grazing intensity. The implications of this study are important for future studies of woody-herbaceous-livestock species interaction and future land resource management. It is not only the overall grazing intensity that researchers and land managers should be concerned with, but also the types of grazing animal species and the different combinations of varying grazing intensities and livestock species (Sankey et al., 2006).

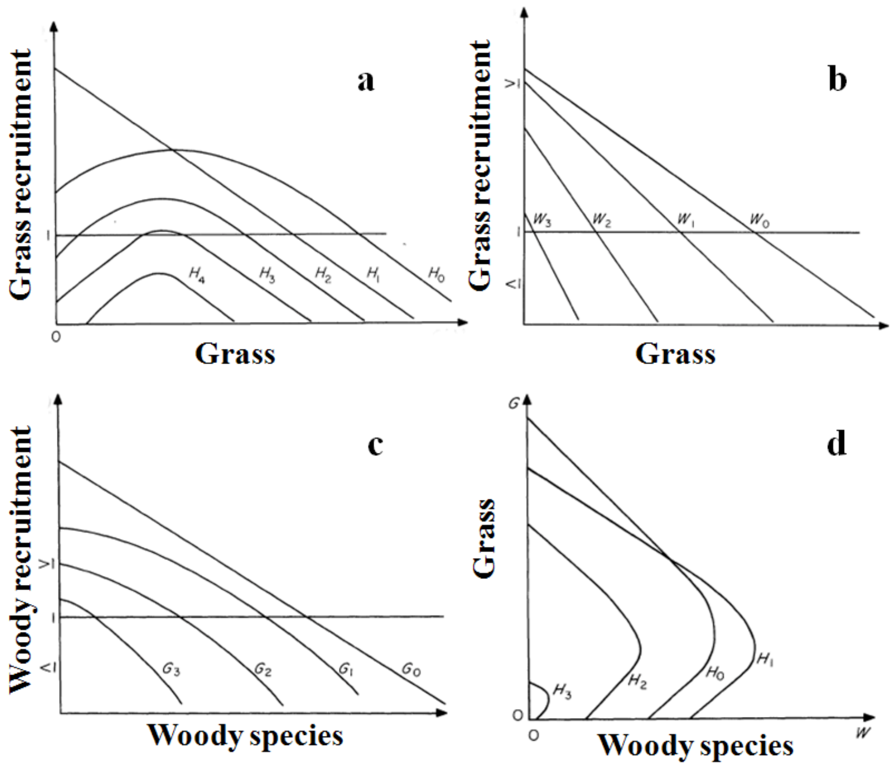

Figure 7. Livestock grazing effects, grass and woody species recruitment, and their interactions (from Walker et al., 1981). Figure a shows the hypothesized effects of increasing grazing pressure (H_0=none, H_4=heavy) on grass recruitment. Figure b shows the hypothesized effects of woody species (W_0= no woody plants, W_3=dense woody plants) on grass recruitment, while figure c demonstrates the potential effects of grass (G_0= no grass, G_4=dense grass) on woody species recruitment. Figure d shows the zero grass recruitment in relation to increasing grazing pressure, grass, and woody species simultaneously.

Sankey et al. (2006) also suggested that many new trees established during the decades when livestock were distributed in numerous small herds in northern Mongolia. Prior to these decades (1930-1950), livestock were distributed in a few large herds owned by a few religious leaders. After a revolution, many socio-economic changes occurred and consequently the large herds were re-distributed into small herds. A pulse of tree regeneration appears to have established following the herd re-distribution (Sankey et al., 2006). This indicates that herd distribution at the landscape scale and the driving socio-economic changes and policy changes are important variables to consider when studying woody-herbaceous-livestock interaction. Human legacies can have lasting effects on this interaction at varying spatial and temporal scales.

Table 2. Long-term averages of grazing intensity observed in the Sankey et al. (2006) study in northern Mongolia.

Sites	Overall Grazing Intensity	Overall Grazing Intensity (AUM ha^{-1})	Dominant livestock species	Number of households at the site
Site 1	Very low	3.3	Sheep	20
Site 2	Low	4.2	Goat-sheep	32
Site 3	Medium	4.8	Cattle-sheep-goat	56
Site 4	Medium	4.9	Cattle	36
Site 5	High	5.9	Sheep	43

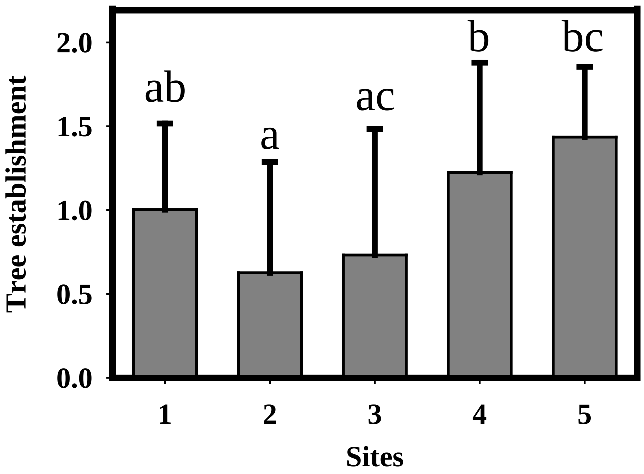

Figure 8. Siberian larch (*Larix sibirica*) tree establishment with different grazing regimes in Sankey et al. (2006) study in northern Mongolia. Site 2 had significantly lower tree establishment than Site 5. Sites 2 and 3 also had significantly lower tree establishment than Site 4.

Previous studies indicate that several variables are important to consider in livestock grazing effects on woody-herbaceous dynamics. They include overall grazing intensity, grazing animal species and their forage preferences (herbaceous vs. woody species), spatial distribution of grazing pressure (few large herds vs. many small herds), and temporal distribution of grazing pressure (winter grazing vs. summer grazing), tree and herbaceous species composition, their palatability, and tolerance to grazing (Figure 9). Accurate understanding of these variables has important implications for modeling and managing woody-herbaceous-livestock species interaction. Different combinations of varying grazing intensities and livestock species composition, for example, can be used to either facilitate or inhibit directional changes in the woody-herbaceous species balance. Forest-grassland ecotone shift can be influenced by both overall grazing intensity and different grazing animals, if a shift is occurring. Different gradients of overall grazing intensity and grazing animals might correspond with different trajectories of change in woody-herbaceous balance. Grazers, for example, might facilitate tree encroachment, while browsers might inhibit tree encroachment and facilitate increased herbaceous species distribution. Furthermore, different levels of grazing effects might be observed in different tree species due to their differences in palatability and their response to grazing. Douglas-fir, for example, is unpalatable to most livestock species, while aspen is highly palatable to livestock species. Siberian larch can also be highly palatable to livestock species. Changes in woody-herbaceous species balance, therefore, might be facilitated or inhibited to different levels depending upon the dominant tree species.

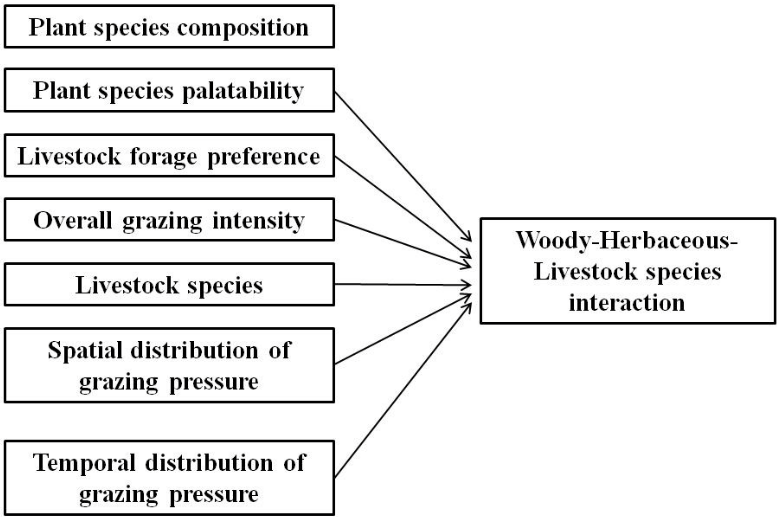

Figure 9. Potential factors influencing grazing effects on woody-herbaceous dynamics.

Other Potential Factors Influencing Woody-Herbaceous Species Dynamics

In addition to livestock grazing, several other variables have been proposed as potential factors that tip the balance in woody-herbaceous dynamics and cause a shift in the ecotone (Figure 10). In savanna environments, four factors have been acknowledged as the main determinants that create and maintain the co-existence. These four determinants are water, nutrients, herbivory, and fire (Werner, 1990). In other mixed woody-herbaceous systems, atmospheric CO_2 increase has been suggested as one of the potential factors that can cause woody-herbaceous shift. Carbon dioxide affects plant photosynthetic rates, stomatal conductance, water-use efficiency, resource allocation, growth, and architecture (Bazzaz, 1996). An increase in CO_2 from 270 ppm to 370 ppm in the last 200 years has been proposed as a possible cause of woody species expansion (Archer, 1994 and references therein). Increased CO_2 is expected to change competitive abilities of different plant species, altering the interaction among species and, consequently, species composition in the community (Bazzaz, 1996). Archer (1994) summarizes that increased atmospheric CO_2 is hypothesized to favor woody plants over herbaceous species due to the following specific reasons: 1) woody plants have C_3 photosynthetic pathway, while many grass species have C_4 photosynthetic pathway, 2) C_3 species have greater advantage for growth and competition with increased CO_2, 3) C_4 grasses evolved at lower CO_2 concentrations (~200ppm) and woody encroachment into C_4 grasslands occurred during a 30% increase in atmospheric CO_2 over the last 200 years. However, Archer (1994) argues that C_3 grasses would also be expected to invade C_4 grasses, if atmospheric CO_2 increase was a probable cause of C_3 woody species encroachment. C_3 grasses, however, have not invaded C_4 grasses. Furthermore, Archer (1994) suggests that C_3 cold desert and temperate grasses have also been invaded by woody species.

Changes in climate variables such as mean annual temperature, rainfall, and evapotranspiration are expected to influence the balance between herbaceous and woody species, because the distribution of many grasslands and savannas throughout the world are closely related to these variables (Archer, 1994 and references therein). Potential changes that might lead to woody encroachment at the expense of herbaceous species include increased or decreased rainfall, shifts in the seasonality of rainfall, shifts in the distribution of precipitation events, increased temperature, and drought (Archer, 1994 and references therein). Increased rainfall might facilitate increased woody species establishment in mixed herbaceous-woody communities. Under normal precipitation conditions, herbaceous and woody species would utilize soil moisture at different depths and herbaceous species would not competitively eliminate woody species. However, when precipitation increases, woody species have the advantage to utilize deeper and more abundant soil moisture. This might increase successful establishment of more woody plants, after which woody and herbaceous species would continue to exploit soil moisture from different depths and herbaceous species would not eliminate newly established woody species. When precipitation decreases, herbaceous species mortality might increase leading to gaps that can be colonized by woody species that are more drought tolerant (Archer, 1994).

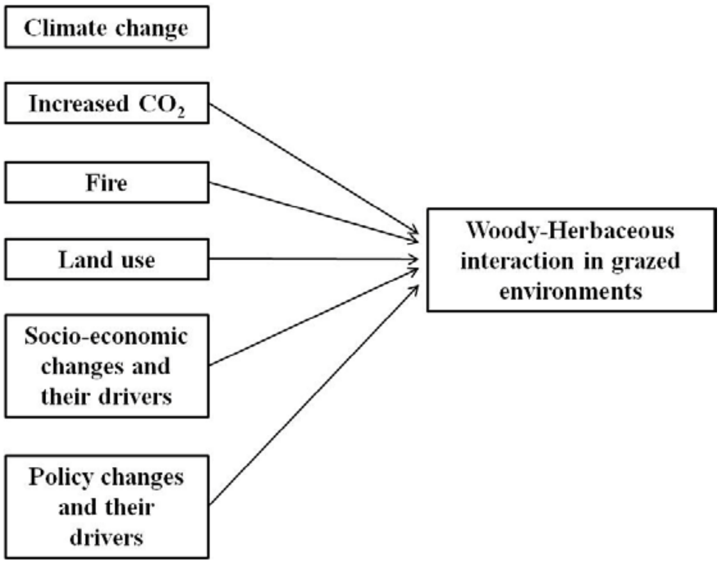

Figure 10. Multiple interacting factors that are commonly proposed as causes of shifts in the woody-herbaceous species balance. In grazed woody-herbaceous systems, the effects of these factors might reduce or increase the effects of livestock grazing.

Archer (1994) summarizes that shifts in seasonality of rainfall in the last century might have contributed to shrubland expansion in southwestern USA and future shifts from summer to winter precipitation associated with increased atmospheric CO_2 might make the current grasslands vulnerable to woody encroachment. In arid and semi-arid environments, cool season moisture favors woody plants, while warm season precipitation favors grasses. When precipitation falls during the cool season, soil moisture percolates down the soil horizons and accumulates at deeper depths. Woody species with deeper root systems are able to utilize such soil moisture. Grasses, however, are unable to exploit such soil moisture and can be especially vulnerable during summer drought. Small frequent precipitation events would favor herbaceous species with shallow root systems, while larger precipitation events would benefit woody species with deeper roots. Therefore, a shift in precipitation distribution would have important implications for herbaceous-woody species balance. In arid environments, changes in extreme climatic events might have more profound effects on vegetation than the gradual change in mean conditions.

Increased atmospheric CO_2-increased temperature models predict that woodland distribution would increase in extent in tropical, subtropical, and cool temperate regions of the world under increasing temperature conditions, potentially because woody species are more stress tolerant than herbaceous species (Archer, 1994 and references therein). Similarly, during drought periods woody species are better able to persist, while herbaceous species decline. The resulting gaps can be occupied by woody species. Periodic droughts, therefore, might be associated with episodic woody plant establishment.

Fire suppression is another explanation that has been proposed for increased woody species distribution. Fire has been shown to be a primary factor that creates and maintains grasslands (DeByle, 1981; Arno and Gruell, 1986; Dando and Hansen, 1990; Covington and Moore, 1994; Mast et al., 1997). When fire is suppressed, woody species encroach into grasslands (DeByle, 1981; Arno and Gruell, 1986; Dando and Hansen, 1990; Covington and Moore, 1994) through increased woody seedling establishment (Mast et al, 1997; Archer, 1994) and survival (Dando and Hansen, 1990; Archer, 1994). Once seedlings reach a sufficient size and age, they are able to tolerate fires and dominate grasslands (Archer, 1994). When fire was suppressed, many different grasslands of varying composition were encroached by woody species of *Juniperus*, *Artemisia*, and *Prosopis* in western and southwestern USA (Burkhardt and Disdale 1969; Blackburn and Tueller, 1970; Young and Evans, 1981; Johnson 1987, Brown and Archer, 1989; Miller and Wigand, 1994; Miller and Rose, 1995; Miller and Rose, 1999; Baker and Shinneman, 2004), and by *Pseudotsuga* and *Pinus* in other parts of the USA (Arno and Gruell, 1986; Dando and Hansen, 1990; Mast et al, 1997; Mast and Veblen, 1999).

Mixed woody-herbaceous ecosystems are sensitive to land use changes (Werner, 1990). However, land use in many savanna environments are intensifying around the world (Werner and Stott, 1990) and the extent of area modified by human use is continually increasing (House et al., 2003). The above described factors, especially changing climate and increasing carbon dioxide accelerate the effects of land use and changes in land use on woody-herbaceous-livestock species interaction (House et al., 2003). Land use policies and socio-economic interests of the pastoral livestock industry, the most common land use in mixed tree-herbaceous ecosystems, are increasingly focused on improved pasture production through additions of fertilizers and supplements, improved breeds and types of livestock, and manipulations of the plant species composition through introducing new species (e.g. legumes) and removing trees (Werner, 1990). Lastly, many different local factors have been proposed, in addition to the proximate factors discussed above, as important variables influencing the woody-herbaceous interaction and leading to a transition into grassland or a transition to woodland. Jeltsch et al. (2000) reviewed these local factors from studies around the world (Table 3) to propose ecological buffering mechanisms as their unifying theory that explains long-term tree-grass co-existence.

Current and Future Research on Woody-Herbaceous-Livestock Species Interaction

Current studies provide detailed, field-based observations of woody-herbaceous-livestock species interactions (Sankey et al., 2006, Sankey et al., 2006). Tree/shrub age and distribution are often characterized with livestock grazing information and the data are used to make inferences regarding woody-herbaceous-livestock species interaction and to build empirical models (Burrows et al., 1990). However, models of interactions have not been explicitly tested outside of the regions and sites for which they were developed (House et al., 2003). Further studies need to use such data in quantitative models and simulation-based approaches (McKeon et al., 1990) and test the validity of empirical models that are based on site-specific data and relationships. Future research can also include process-based studies with carefully designed experiments. Such studies, although they are likely to be short-term, would provide important details on ecological processes involved in the woody-herbaceous-livestock interaction. Using the detailed understanding of the processes involved, accurate empirical relationships and simulation models could be built to observe potential patterns and changes at longer time-scales (Daly et al., 2000). This would further enhance our understanding of woody-herbaceous-livestock species dynamics and their changes at different spatial and temporal scales. Process-based studies should also have more controlled experiments, where effects of different grazing intensities and the effects of varying grazing animals can be statistically separated. This can further improve our understanding of the effects of overall grazing intensities and different grazing animals and allow an understanding of the importance of overall grazing intensity versus grazing animal species.

Table 3. Local factors that impede the transition savanna to woodland or to grassland in different regions of the world (from Jeltsch et al., 2000)

Buffering mechanism impeding transition to Woodland	Buffering mechanism impeding transition to Grassland	Functioning	Location	Reference
Fire		Fires reduce woody plant densities and maintain them at low levels, primarily by killing or suppressing individuals in the smaller size classes	General	McNaughton 1992; Skarpe 1992; Frost & Robertson 1987; Furley et al. 1992; Sarmiento 1992
			Africa	Trollope 1982; Teague & Smit 1992; Huntley 1982; Jones 1992; Jeltsch et al. 1996; Menaut & Cesar 1982; Gignoux et al. 1997; Hochberg et al. 1994
			South America	Coutinho 1982; Eiten 1982; Eden & McGregor 1992; Butcher 1982
			Central America	Rebertus & Burns 1997
			Australia	Lacey et al. 1982; Walker & Gillison1982; Kershaw 1992
Elephant		Felling, pushing over or uprooting of trees	East and central Africa	Belsky 1990; Ben-Shahar 1996; Cumming 1982
Browsers (incl. prairie-dog (*Cynomys ludovicianus*))		Heavy browsing pressure: reduction in the density, growth and regeneration	General	Cumming 1982
			East Africa	Belsky 1992
			Central America	Weltzin et al. 1997
			Australia	Lacey et al. 1982; Walker & Gillison 1982
Fire+elephant; Fire+browser; Fire+elephant +browsers		Elephant may facilitate the entrance of fire into dense stands of woody plants; fire maintains woody plants at an accessible height for browsers; combined effects	Africa	Frost & Robertson 1987 Belsky 1990; Cumming 1982; Furley et al. 1992; Belsky 1992; Barnes 1982; Teague & Smit 1992; Trollope 1984; McNaughton 1992
Wood cutting		Reduction of tree density	Brazil	Eiten 1982
			Southern Africa	Tietema et al. 1991; Jones 1992
Seed predators		Reduction of reproduction success	Southern Africa	Tietema et al. 1991; Miller 1996
Tsetse fly		Control of grazers	East and central Africa	Cumming 1982
	Micro sites with favourable conditions for tree establishm. and survival (incl. microelevations or depressions, ant or termite mounds, tree seed patches in herbivore dung, fire protected sites e.g. termite mounds or swamps)	Enable tree seedlings to become established in the grass layer which is otherwise more competitive e.g. improved moisture conditions; increasing the number of establishment opportunities; protection from frequent fires	General	Solbrig 1996; Jeltsch et al. 1998; Scholes & Walker 1993
			South America	Bucher 1982; Dubs 1992
			Africa	Jones 1992; Abbadie et al. 1992; Reid & Ellis 1995; Leistner 1961; Jeltsch et al. 1998; Menaut & Cesar 1982

Another trend in current studies of woody-herbaceous species dynamics with or without livestock grazing effects is the use of digitally available data such as satellite imagery and digital aerial photography to map and monitor changes in woody-herbaceous plants. In northwestern USA, for example, several studies have used digital imagery to detect the commonly field-observed expansion of juniper and pinyon-juniper woodlands into adjacent shrublands and grasslands (Strand et al., 2006; Weisberg et al., 2007). Field-based approaches for detecting woody cover increase provide highly accurate and valuable results, but they can be labor-intensive, time-consuming, and limited in the spatial extent they can cover. In comparison, the application of remote sensing methods can be more cost-effective and timely due to the large areal extent they cover. Digital satellite imagery also provides opportunities for more robust and comprehensive analysis of change, as the imagery can be easily integrated with other sources of digital data, such as digital maps of grazing lands and topography. Moreover, data from satellite platforms, such as Landsat, can be acquired in retrospect to examine past changes or past vegetation distribution and to compare with current distribution in order to quantify the extent and rates of change. Such analysis of remote sensing data along with detailed field data could provide information on indicators of global environmental changes and enhance our understanding of processes, signals, extent, and rates of woody-herbaceous vegetation changes under herbivory effects. The information would also be useful in grazing management and land use decision making regarding desired vegetation patterns across the landscape.

Quantitative models woody-herbaceous-livestock species interaction could also include data of other important factors that contribute to changes in woody-herbaceous species such as climate change, CO_2 increase, and fire suppression. This would enhance our ability to quantitatively describe the combined effects of multiple interactive factors on ecotone shift. This would also improve our predictive ability and forecasting skills regarding when, where, and under what conditions changes occur leading to a regional environmental change. Finally, currently proposed hypotheses and empirical models should be quantitatively tested in varying regions with different plant and livestock species.

ACKNOWLEDGEMENTS
This study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Sankey, T. T. 2009. *Woody-Herbaceous-Livestock Interaction.* Pages 123-148 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

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Spatial Pattern of NDVI in Semiarid Ecosystems of Northern Spain

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ABSTRACT

Understanding the distribution of landscape variables presents several advantages to researchers focusing on ecological studies. The Normalized Difference Vegetation Index (NDVI) is a relatively common productivity variable used to assess, monitor, and compare landscapes. In this study, an NDVI layer was created using Satellite Pour l'Observation de la Terre 4 (SPOT 4) satellite imagery to compare the pattern of NDVI values at waterholes/shelters relative to the pattern of NDVI values at random locations in semiarid rangelands of northern Spain. Single factor ANOVA revealed that NDVI values at waterholes/shelters were significantly lower than those at random locations. Analysis of topographic variables (slope) revealed that NDVI values at gentler slopes ($\leq 5°$) were significantly higher ($P < 0.001$) than values at steeper slopes ($>5°$) within grazed areas, while NDVI values in areas with steeper slopes ($>5°$) were significantly higher ($P < 0.001$) under total rest. Land use, specifically semi-extensive livestock grazing, appears to be the main factor governing the observed NDVI distributions.

Keywords: rangelands, spatial pattern, ANOVA, NDVI

INTRODUCTION

Human activities and climatic conditions impact ecosystems and may involve continuous or discontinuous transitions from one stable state to another. Discontinuous transitions result in the most catastrophic changes to ecosystems and these changes are often abrupt and irreversible (Kefi et al. 2007). It is important to detect early signs of potential transitions that can negatively alter ecosystem services and cause the loss of ecological and economic resources. Vegetation patchiness can be used as a signature of imminent discontinuous transition (Kefi et al. 2007). It is especially important within arid and semiarid ecosystems as they are more vulnerable to desertification processes (Savory 1999, Kefi et al. 2007). Spatial pattern analysis can be used to quantify vegetation patchiness and various efforts have already been made to better understand the spatial pattern of vegetation and its connection with desertification (Bergkamp et al. 1996, von Hardenberg et al. 2001). Rietkerk et al. (2002) have developed a model using plant density, soil water, and surface water to explain observed spatial patterns of vegetation whereas von Hardenberg et al. (2001) developed a similar model using biomass density and availability of groundwater to explain the observed spatial patterns of vegetation. In spite of these efforts, researchers are still searching for comprehensive techniques to explain the variety of observed spatial patterns and understand if these patterns are the result of pre-existing environmental heterogeneity, spatial self-organization (Rietkerk et al 2002), applied land use practices, or a combination of these factors. In addition, it is important to investigate the relationship between spatial pattern and biodiversity, ecosystem resilience, and desertification.

Normalized Difference Vegetation Index (NDVI)

A landscape productivity parameter such as the Normalized Difference Vegetation Index (NDVI) can be used to understand various characteristics of a vegetation community. Comparisons between vegetation communities can be used to understand the distribution and structure of vegetation within a study area. NDVI has been applied in numerous studies and used to evaluate the conversion of one land cover type to another (Deyong et al. 2009, Lunetta et al. 2006, Martinez and Gilabert 2009) and ecological responses to environmental change (Pettorelli et al. 2005). These authors found NDVI was effective to monitor habitat degradation and fragmentation, and the ecological effects of climatic disasters such as drought or fire. The spatial patterns of NDVI values were studied by Wang et al. (2001) in response to changes in temperature and precipitation. They reported a strong correlation between the general spatial distribution of NDVI values and the pattern of average annual precipitation while the influence of temperature on NDVI values was observed only during the early and late parts of the growing season (Wang et al 2001). Roberts et al. (1997) also used NDVI to assess spatio-temporal patterns of vegetation relative to various atmospheric properties derived from AVIRIS hyperspectral data. Based upon these applications, NDVI was chosen to assess the spatial pattern of vegetation in semiarid rangelands of northern Spain relative to land use and land tenure (Cummins 2009).

This study uses a combination of point analysis and topographic analysis performed using NDVI values and focuses on the spatial distribution of NDVI values in the middle Ebro valley (i.e. Monegros study area) of northern Spain. The distribution of NDVI values at waterholes/shelters were analyzed using single-factor ANOVA to better understand the impact of land use and potential long-term degradation of the ecosystem. In addition, the effect of topography (specifically slope) on the pattern of NDVI values was also evaluated. These data were important to study the impact of land use with respect to ecosystem stress and potential desertification.

The hypotheses tested in this study include 1) lower NDVI values were expected in response to increased bare soil and the loss of vegetation in those sites experiencing overgrazing of plants, 2) NDVI values were expected to be different in response to topographic effects (i.e., slope) due to both differential livestock use and runoff patterns.

MATERIALS AND METHODS

Study area

This study focuses upon the xeric-steppes of the middle Ebro valley, Aragon, Spain and is referred to as the Monegros study area (Figure 1). The dominant plant species in the area is Rosemary (*Rosmarinus officinalis*) with various gypsophile plant species occurring over a gypsum substrate in the most xeric areas. Scattered remnants of the original Juniper woodland community (*Juniperus thurifera*) are also present. The study area covers over 300000 ha (3000 km^2) with the valley receiving the majority of its water from the Pyrenees Mountains, yet it is a dry area with low precipitation (< 0.30-m annually). Grazing activity in the Monegros study area consisted of various flocks of sheep grazed under a semi-extensive regimen. Specifically, livestock were led by a shepherd to graze the fallow fields and rangeland steppe continuously throughout the year. Flocks were moved daily from shelters to the surrounding grazing areas where they stayed from morning until evening. Supplementary food was provided during the driest season and for reproductive females. Livestock productivity in the area is low, with an estimated stocking rate of 0.2 head ha^{-1} yr^{-1} (Pueyo et al. 2008).

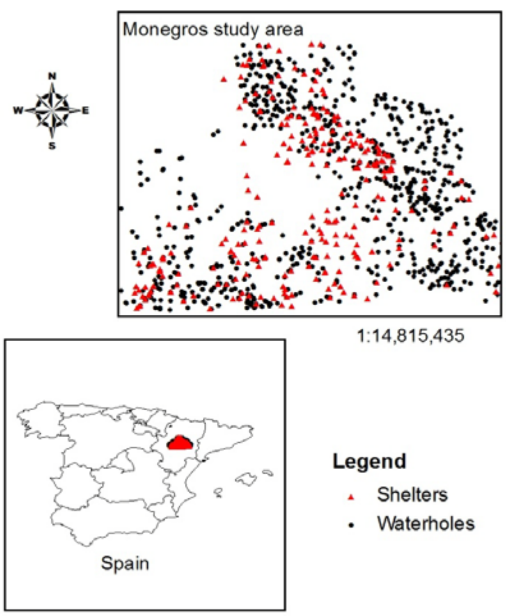

Figure 1. Monegros study area in northern Spain and locations of waterholes and shelters used in this study.

Data acquisition and preparation

Satellite Pour l'Observation de la Terre 4 (SPOT 4) collects data in 4 spectral bands from the visible (545 nm band center [green] and 645 nm band center [red]) through near-infrared (NIR) (840nm band center) and short-wave infrared (SWIR) (1665 nm band center) portions of the electromagnetic spectrum. These data are stored as raster imagery having a spatial resolution of 20-m x 20-m. One SPOT 4 image was acquired on May 11, 2007 for use in this study. The SPOT 4 data were processed to reflectance by performing an atmospheric correction using the Cos(t) image-based absolute correction method (Chavez 1988) in Idrisi Andes software (Clark Labs, Worcester, MA). The imagery was then georectified (RMSE

= 8.3 m) using 0.5-m x 0.5-m aerial photography and projected into Universal Transverse Mercator (zone 30N, European datum 1950) using a first order affine transformation and nearest neighbor resampling. NDVI was calculated using the VEGINDEX module of Idrisi Andes and the red and near- infrared bands of SPOT 4 imagery.

A point shapefile was created describing the location of all known waterholes and shelters within the study area (*n* = 755). An equal number of random points (*n* = 755) were generated using Hawth's tools for ArcGIS and stored within a second shapefile. One constraint placed upon the random points was that they could not be located within 100 m of a waterhole or a shelter to avoid regions of overlap during future proximity analysis.

Point analysis

The "Sample" tool within ESRI's ArcGIS 9.3 was used to extract NDVI values at each waterhole/shelter location as well as at each random location. The resulting data were exported to a MS Excel spreadsheet for further analysis. Single-factor ANOVA was used to determine whether mean NDVI values at waterholes/shelters were statistically different from those at the random points.

Topographic analysis

A digital elevation model (DEM) (20-m x 20-m pixels; RMSE = 7.42 [Pueyo 2005]) available for a portion of the study area (total area approximately 1800 km^2 [i.e. 40 % of the study area]), was used for topographic analysis. Using this DEM, a slope model was generated in Idrisi Andes with slopes expressed in degrees (°). The slope model was used to determine 1) if livestock favored specific slopes, and 2) if slope affected the spatial distribution of vegetation as indicated by the spatial distribution of NDVI values.

Within the area covered by the slope model, a sub-region where livestock grazing had been excluded since 2003 (area ~ 55 km^2) was selected for further analysis. Two sets of random points were created in the total rest region representing two slope classes (i.e. $\leq$5.0° [77% of the entire study area] and >5.0° [23% of the entire study area]) (*n* = 124, *n* = 62 within each slope class). Similarly, two sets of waterholes/shelters points were selected using the same two slope classes (*n* = 124, *n* = 62 within each slope class). These classes were selected as the slope threshold (~5°) has been cited as the point where livestock use is reduced due to slope (Ganskopp and Vavra 1987; Holechek et al. 2000). The waterholes/shelters area was considered a grazing area and later compared with the total rest area. Single-factor ANOVA was used to determine the effect that grazing stratified by slope had upon NDVI values.

RESULTS AND DISCUSSION

Point analysis

The results of single factor ANOVA (Table 1) comparing NDVI values at waterholes/shelters with NDVI values at random points indicate that NDVI values were statistically higher ($P < 0.001$) at random locations. This suggests that vegetation characteristics differ between these areas and may be the result of higher percent land cover, higher productivity, or differences in biodiversity at the random locations relative to that found at the waterholes/shelters.

Table 1. Results of single-factor ANOVA comparing NDVI values between waterholes/shelters (*n* = 755) and random points (*n* = 755).

Group	Mean NDVI	Variance	P-value
Waterholes / shelters	0.220	0.030	0.000
Random points	0.305	0.051	

Vegetation type, phenology**,** and distribution, as well as soil type, climatic conditions, and land use are all factors that affect NDVI values (Pettorelli et al. 2005, Bounoua et al. 2000, Nagler et al. 2000, Verhulst et al. 2009). Alados et al. (2005) have shown that grazing can favor biodiversity and heterogeneity of plant species. Furthermore, Pueyo and Alados (2006) have demonstrated that the availability of the gypsum substrate is an important factor governing plant community patterns in semiarid Mediterranean landscapes such as the Monegros study area.

Topographic analysis

The results of single-factor ANOVA examining the cumulative effects of slope (≤5.0° and >5.0°) and land treatment (grazing or total rest) on NDVI values indicate NDVI values were statistically higher in areas with gentler slope ($\underline{x}$ = 0.34 and $\underline{x}$ = 0.16 in ≤5.0° and >5.0 slope areas, respectively [$P < 0.001$]) where grazing was allowed, suggesting an effect of either grazing, slope, or a combination of these factors (Table 2). However, there was no difference in NDVI values within the slope classes ($P = 0.98$) in the total rest area ($\underline{x}$ = 0.22 and $\underline{x}$ = 0.22 for ≤5.0° and >5.0 slope areas, respectively). This suggests that slope alone has little effect on observed NDVI values.

Table 2. Results of single-factor ANOVA comparing NDVI values within two slope categories (≤ 5.0° and > 5.0°) and two treatments (grazing and total rest) (*n* = 62)

	Grazing $_{\leq 5°}$	Grazing $_{> 5°}$	TR $_{\leq 5°}$	TR $_{> 5°}$
Grazing $_{\leq 5°}$	-	*	*	-
Grazing $_{> 5°}$	*	-	-	*
TR $_{\leq 5°}$	*	-	-	0.98
TR $_{> 5°}$	-	*	0.98	-

* $P \leq 0.001$

The results of single factor ANOVA tests performed on the waterholes/shelters and random points comparing grazed areas with ≤5° slope to total rest areas with ≤5° slope indicate that the NDVI values were statistically different (Table 2). These results indicate that land management decisions (i.e. grazing) have very tangible effects on the landscape. Furthermore, these results imply that sheep may show a preference for gentler slopes but also make use of steeper slopes to cause a detectable difference in NDVI when compared with areas of total rest. Holechek et al. (2000) report that sheep and goats use rugged terrain better than cattle because of their smaller size, more surefootedness, and stronger climbing instinct. Holechek et al. (2000), Cook (1966), Gillen et al. (1984), Ganskopp and Vavra (1987), and Pinchak et al. (1991) estimated there can be 30% or more reduction in grazing activity when slope exceeds 10% (5.7 °). This factor seems to have been borne out by this study and helps explain the differences seen in NDVI values. Further, this demonstrates the effect of topography on the spatial pattern of NDVI-values in response to land use and animal-specific use of the available landscape.

MANAGEMENT IMPLICATIONS

The results of this study describe a difference in NDVI values between areas of semi-extensive continuous grazing and total rest. In addition, NDVI differences were noted due to the interactive effect of slope. These results may be interpreted to imply that grazing is detrimental, however, such a conclusion would be premature and ill-founded. A more correct interpretation of the results presented here is that overgrazing of plants is detrimental. Grazing by itself, however, does not necessarily lead to overgrazing of plants. Rather, sedentary grazing systems tend to lead to overgrazing (Weber and Horst 2009) while more highly mobile grazing practices have been demonstrated to effectively improve semiarid rangelands (Voisin 1988, Savory 1999, Weber and Gokhale 2010).

CONCLUSIONS

NDVI values at waterholes/shelters were different from NDVI values at random points. The difference was attributed primarily to grazing. However, other factors affecting NDVI are important to consider. The interactive effect of grazing and topography was explored and an indirect relationship with slope was observed. Livestock selection for gentler slopes was shown to affect NDVI-values differentially by slope. However, livestock use of steeper slopes (>5°) was still sufficient to cause a difference in NDVI-values compared to areas with similar slopes (>5°) but managed under a total rest treatment. The results of this study demonstrate the tangible effects of the human decision-making process and the role of anthropic forces forming and changing the landscapes and ecosystems of the world.

ACKNOWLEDGEMENTS

The study was made possible by a grant from the National Aeronautics and Space Administration Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Gokhale, B. and K. T. Weber. 2009. *Spatial Pattern of NDVI in Semiarid Ecosystems of Northern Spain*. Pages 149-156 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.

Applying Knowledge of Traditional Pastoralists to Current Range Management

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ABSTRACT
Pastoralism is an ancient form of subsistence that is still in wide use today throughout the world. While many traditional pastoral regions are the focus of current desertification studies, the long history of sustainability by these cultures is of great interest nonetheless. Numerous studies suggest that the land degradation observed in these areas today is a recent phenomenon attributable to changes in land tenure, management, and treatment, in addition to changes in the environment. This paper explores the suggested causes of land degradation and focuses upon applied land management and grazing treatments common to traditional pastoral cultures. Comparisons are made with western livestock ranching and numerous similarities noted. Historical observations suggest that desertification is the result of both climatic and anthropic factors with specific emphasis recently placed upon the effect of sedenterization and the subsequent negative feedback cycle initiated through partial-rest and total rest found across nearly all continents, societies, and grazing cultures today.

KEYWORDS:Grazing, pastoralism, desertification

INTRODUCTION

Pastoralism is an ancient craft which on the surface, appears to demand only minimal skills. The shepherd or herdsman is simply tasked with keeping his stock alive so that he may subsist on the animals' milk, blood, wool, meat, and value in trade. Just beneath this thin veneer however, rests a myriad of complexities involving forage, animal health, reproduction, predation, weather, and the social and cultural fabric within which the pastoralist functions. Over time, pastoral cultures have developed and these complexities have been mastered, with learned animal husbandry skills and the wisdom of experienced pastoralists handed down through generations[1].

It is not without debate that pastoralism developed after agriculturalism (Khazanov 1994). By 7000 BCE pastoralism was well established (Flannery 1965) and most likely developed as people migrated into areas of low productivity and/or regions of unreliable rainfall (i.e., the arid and semiarid regions of the world). As a result, these people came to rely upon domesticated animals for subsistence instead of agricultural crops (Salzman 2004; Cummins 2009). Over time, three unique forms of pastoral production took hold: 1) sedentary production, 2) transhumance, and 3) nomadism (Yalcin 1986). Sedentary pastoralism involves keeping livestock near farms and villages year-round while transhumance includes the seasonal movement of animals and people from valley bottoms to mountain pastures (Yalcin 1986; Ott 1993; Cummins 2009). Nomadic pastoralism may have developed in response to recurring and wide-spread drought (Salzman 2004) or widespread and erratic rainfall and is typified by livestock being moved in constant search of forage. Nomadism differs from transhumance in that no permanent base (home or village) is developed and likewise, no pre-defined series of movements is used. Of all the forms of pastoralism, nomadism is least systematic.

Much of what is considered rangeland today (Bedell 1998) falls within the arid or semiarid regions of the world. These areas support grasses, forbs, and shrubs which are managed without cultivation, irrigation, herbicides, pesticides, or fertilizers. The primary management tools of the traditional pastoralist are his livestock (principally sheep, goats, cattle, horses, donkeys, and camels) and fire (Savory 1999).

Within arid and semiarid rangelands, water is the limiting factor (Niamir-Fuller and Turner 1999; Hill 2006) and precipitation is highly variable both spatially and temporally. In seasons of increased precipitation, forage availability improves dramatically (Niamir-Fuller and Turner 1999; Gregory et al. 2008) whereas in years of drought grass becomes scarce. Unfortunately, contemporary grazing systems can create less effective water cycles (cf. rain use efficiency) resulting in increasingly frequent and severe droughts events (Savory 1999). As a result, some pastoral cultures (e.g., the Herero of Namibia and the Samburu of Northern Kenya) have degraded their environments to the point where temporary abandonment was required (Hill 2006), and all have altered their environment to some degree (Wilson 2007). Still, numerous pastoral cultures (e.g., Rashayada Bedouin of the Sudan, Mongolian and Chinese herdsman, and Pyrenean herders) (Figure 1) have survived for thousands of years despite various complexities, hardships, and challenges. Herein lies an important point for consideration and an equally important question; that is, how have these traditional pastoral cultures managed to sustain themselves for thousands of years? This is not meant to imply that the landscapes used by all pastoral cultures are pristine as many are desertifying. There is evidence to suggest that pastoral landscapes were in better

[1] This is not to imply that ancient pastoralists developed a utopian society as perfect long-term ecological sustainability of arid and semiarid landscapes has yet to be achieved (Khazanov 1994).

condition throughout the 1800's and early 1900's (Niamir-Fuller and Turner 1999) and that the observed rapid degradation is a relatively recent phenomenon that has accelerated during the latter parts of the 20th and current centuries (Waller 1985; Gritzner 1988; Smith 1992). This raises a second, interrelated and perhaps more intriguing question; what changed to cause these declines?

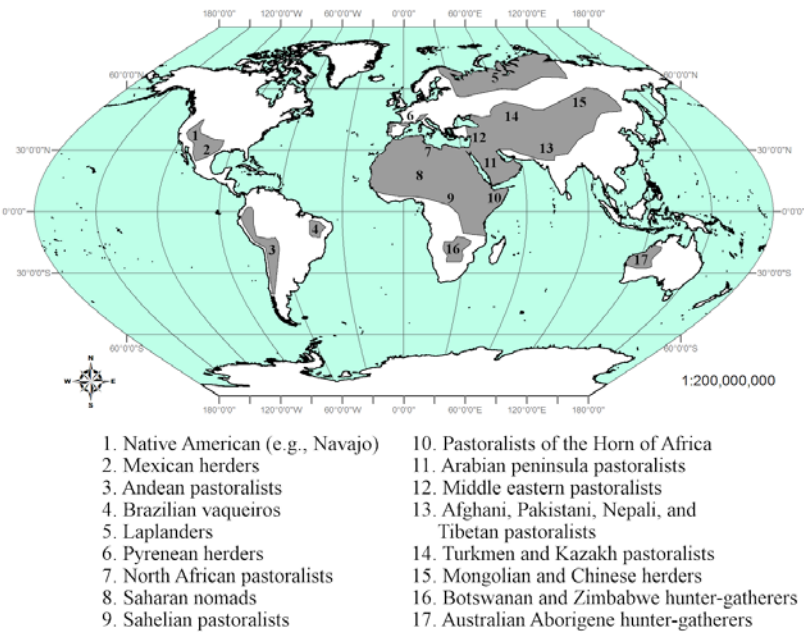

Figure 1. Map of general distribution of traditional pastoralists worldwide (note: the terms pastoralist, herders, and hunter-gatherers and general region map from Niamir-Fuller 1999).

To address these questions, one must first understand what is meant by the term desertification. Desertification is a term first used by Auberville (1949) which refers to the severe degradation of the arid, semiarid, and sub-humid areas of the world due principally to climatic and anthropic forces (UNCCD 1995; Arnalds 2000). The term implies a nearly irreversible condition (Dougill and Cox 1995; Niamir-Fuller and Turner 1999) of the landscape in contrast to a less severe perturbation reserved for the term degradation. Desertification was also used by Savory (1999) to refer to the manifested symptom of biodiversity loss in arid and semiarid environments. The universal remedy for degraded rangelands has been the removal of livestock (i.e., de-stocking). Under the most systematic grazing regimes, *rest* is deliberately used as a temporary de-stocking that serves as much as a pre-determined scheduling process as it is a land management technique. Under less systematic regimes, the term *recovery* is applied, inferring an active management decision that allows plants to recuperate before additional grazing is allowed. The length of the *recovery* period is not pre-determined (Voisin 1988) but rather, decided upon by the pastoralist based upon his/her knowledge, experience, and goals. Rest then, as part of a grazing system, may or may not have any relationship to actual leaf and root recovery.

The most extreme form of de-stocking is *abandonment*. In western cultures, *abandonment* is equated with failure, while in other pastoral cultures, *abandonment* is viewed as part of the normal process of good management (Stone 1993; Hill 2006). In essence then, all pastoral cultures have applied intervals of no-grazing (rest, recovery, and abandonment) along with periods of grazing as part of their historic and traditional grazing practices. The only real difference --apart from semantics-- is the duration of the abandonment (cf. rest or recovery) which is a function of the particularities of the season (Voisin 1988), brittleness of the environment (Savory 1999), and so forth. Regardless of the term used, rest, recovery, and abandonment all involve periods of total or near total absence of grazing throughout a growing season or grazing cycle.

It may seem a logical conclusion then, that the period of rest or recovery constitutes an entirely positive influence on the environment. Such a conclusion however, is paradoxical, as just like too brief a recovery period degrades the environment, so too does a prolonged recovery period. This is because arid and semiarid grass species have co-evolved with herbivores and the prolonged absence of herbivory tends to lead to excessive standing litter accumulations called moribund grass. Moribund grass breaks down through a gradual physical weathering process rather than rapid biological decay and is particularly detrimental to grazing-dependent bunchgrasses. With sufficient time, this condition can kill individual plants leaving only exposed soil in its stead (Savory 1999; Figure 2). Savory (1999) draws a clear distinction between the recovery period required by individual plants --to minimize or avoid overgrazing-- and the episodic, yet high levels of disturbance the plants and soil surface requires to maintain the health of its biological communities through the trampling of moribund material to ensure rapid biological decay, increase soil organic matter, and provide soil-covering litter to promote improved rain use efficiency. Furthermore, Savory observed that while livestock are grazing, much of the range is essentially rested as the livestock are scattered and produce inadequate disturbance-- to describe this effect, Savory used the term *partial rest*.

Figure 2. An example of excessive litter accumulation degrading through oxidative rather than biological means.

Under conditions of partial rest, livestock are grazed at low density (i.e., few animals graze a large pasture in an unbunched manner) and when herds remain relatively sedentary over long periods of time (e.g., a month or more) overgrazing of plants occurs. This, combined with the adverse effects of partial rest, exacerbates an already declining rain use efficiency trend through both increased run-off and soil surface evaporation (Savory 1999; Huxman et al. 2004). While some plants will be grazed repeatedly others may remain un-grazed and over time, moribund grass accumulations form just as they do in over-rested areas. The moribund grasses present a less palatable option to the herbivore, which tend to select the same individual grass plants resulting in over-grazing of these plants. As a result, over-grazing damages or kills grazed plants while un-grazed plants are weakened, and the rangeland enters a negative feedback cycle of slow but progressive degradation. Recent studies support these observations and suggest that partial and total rest have remarkably similar affects on arid and semiarid grassland environments (Gomez-Ibanez 1975; Cummins 2009; Weber et al. 2009a; Weber et al. 2009b).

The cause of rangeland desertification has been attributed repeatedly to a combination of climatic and anthropic factors (UNCCD 1995; Geist and Lambin 2004; Hill 2006; Lambin et al. 2009) with specific emphasis placed on overgrazing and drought (Bedell 1998; Puigdefabregas 1998). Climate theories have focused upon changes that have occurred over the past ten thousand years of the current Holocene and note several periods of increased aridity (drought) and still other periods of increasing humidity. In addition, some changes were localized (Stebbing 1935; Niamir-Fuller and Turner 1999) while others were global in nature. Some changes persisted over long time periods while others were much shorter in duration (Brooks 1949; Khazanov 1994). In essence, changes in the earth's climate since the last Ice age have not been progressive in any sense but rather oscillatory. Indeed, it has been suggested that the periods of increased aridity have led to the emergence and increased prevalence of nomadic pastoralism and not the inverse, nor a global increase in desertification due to pastoralism (Khazanov 1994). This is because nomadic and transhumant pastoralism is a successful adaptation for survival within highly variable semiarid and arid environments (Niamir-Fuller 1999; Khazanov 1994; Salzman 2004; Cummins 2009).

One reason for the success of nomadic and transhumant pastorlism in semiarid and arid ecosystems in contrast to cultivated agriculture relates to effective rainfall, rain-use efficiency or soil moisture storage capacity. Thurow (2000) described various hydrologic effects on rangelands and noted that soil structure, soil texture, and organic matter content are key factors governing soil moisture storage capacity. While the particular soil type or soil association does not change with treatment, a soil's structure and organic matter content can be affected. In the absence of large herbivores, organic matter inputs will be dramatically reduced and the surface of soils tends to become capped (Khazanov 1994). Both of these factors degrade a soil's ability to retain water (Thurow 2000) and lead to a reduction of plant production. Similar to, and often compounded upon the effects of prolonged rest, these rangeland ecosystems enter a negative feedback cycle which ultimately leads to desertification (Le Houerou 1984; Thurow 1991).

While literature from the 1980's and early 1990's repeatedly linked livestock to the degradation and desertification of rangelands (Lamprey 1983; Sinclair and Frywell 1985; Wolfson 1990) more recent studies have refuted this by suggesting that prolonged rest leads to even more serious degradation than overgrazing (Seligman and Perevolotsky 1994; Olaizola et al., 1999; Cummins 2009). And so it seems that neither climatic or anthropic factors are solely to blame for the desertification of the earth's rangelands. It stands to reason then, that some interactive or combinatory explanation may be most agreeable. Indeed Hill (2006) arrived at a similar conclusion when he examined the arid rangelands of the Transjordan plateau. His conclusion was that climate change was a major factor explaining the disappearance of surface water and changes in vegetation due to increased aridity (Bar-Matthews et al. 1999; Hill 2006). This, however may also be attributed to reduced soil moisture storage capacity, increased surface runoff and increased soil surface evaporation because too *few* animals were present on the rangelands for too *long* a period of time (Savory 1999).

A second major factor cited by Hill was human ignorance regarding the consequences of mismanagement (McGovern et al. 1988) (i.e., land use decisions and practices). The third causal factor was the role of politics (i.e., land management or land tenure [Lundsgaard 1974]) and the hypothesis that environmental sustainability is inversely related to the levels of hierarchy and dissociation present in the governing/managing body (Hill 2006).

What is most interesting amongst all these studies is the clear admission of the substantial role played by humans (albeit not a solitary role) in shaping and altering the environment and the inseparability of humans and nature (Goldman and Schurman 2000). It seems reasonable then, to consider what humans may be able to do to improve the environment instead of focusing solely upon what they have done to degrade it or on oscillating climatic conditions.

Land use, and specifically pastoral land use is highly variable both temporally and spatially across the rangelands of the world (Niamir-Fuller 1999). To enable modern scientific inquiry, some means of quantifying and classifying land use is required (Funtowicz and Ravetz 2003). In range science, various specific types of grazing are recognized and in terms of management, grazing is typically classified as either intensive or extensive relative to the degree of management effort involved (Bedell 1998). A second set of terms (stocking density or stocking rate) describes the number of animals grazing an area relative to the size of the area (density) or the amount of time allocated to an area (rate). While a plethora of terms are applied to specific styles of grazing (rest-rotation, deferred-rotation, high intensity-low frequency, short-duration, etc.[Holechek et al. 2001]) they differ in the proportion of time spent grazing relative to the proportion of time spent for recovery of the plants in that same area and in how each views and applies disturbance or a lack thereof. In western societies, extensive or semi-extensive management has become the norm, and graziers typically apply a single grazing system for their herd/herds which is repeated on an annual cycle. A problem with this approach is that it places the focus of livestock management upon the herd and in essence, the "herd" *is* the management unit. In contrast, the "season" is the management unit for transhumant pastoralists and as a result, the latter is less systematized and more variable. In neither case, however, is "time" (the period over which plants are exposed to a grazing animal and the range experiences a disturbance through the effect of the herd) the focal management unit even though numerous studies have stressed its importance to ensuring long-term sustainability (Voisin 1988; Savory 1999). Voisin, for instance, points out that promoters of the rotational method "overlooked the necessity for the periods of occupation being sufficiently short" and instead emphasized "dividing the pasture into a greater or smaller number of paddocks...and then shifting the herd from one paddock to the next".

Range scientists have recommended and tested a great many “grazing systems” varying from continuous grazing through a plethora of rotational grazing practices designed without taking into account the full complexity of cultural/social issues, wildlife, alternative uses, market forces, etc. Both pastoralists and ranchers attempt to address these complexities using a myriad of grazing systems. To effectively address complexity requires a planning process that embraces complexity, rather than a pre-determined management system designed for simplicity (Savory 1999).

Niamir-Fuller and Turner (1999) note the importance of mobility within highly variable environments (i.e., arid and semiarid areas) and while they opt to focus upon *mobility* itself, the reason why mobility is so important is intimately tied to Voisin's emphasis on time. Behnke (1999) echoes these same concerns and the importance of highly mobile herds in his study of the Etanga pastoralists of Namibia. In both cases, mobile pastoralism (e.g., transhumant and nomadic pastoralism) is considered an ideal adaptation within arid and semiarid rangelands especially in contrast to the alternative, sedenterization (Salzman 2004). Sedenterization is the process by which once highly mobile pastoral cultures are converted to less mobile ones and concentrated near major trade routes, villages, and other communities. As a result, the pastoralist no longer needs to rely upon himself and his livestock for subsistence, but upon his ability to purchase goods and services using money gained through the sale of his livestock. In such emerging market economies lessons in business acumen are quickly learned and the adage of "location, location, location" is proven true again. The consequence of such change is that the pastoralist's herd may spend nearly the entire year within a relatively small area and in response to market demands --instead of personal needs or the carrying capacity of the land-- may increase the number of animals in his flock or herd placing further stress upon a brittle arid or semiarid environment.

In a study of nomadic cultures, Khazanov (1994) describes a worldwide trend in which nomadism is being replaced by market-oriented ranching (cf. sedenterization). In these cases, the result is the prolonged occupation of livestock within a given area and the subsequent impoverishment and desertification of the landscape. Keohane (2008) reports a similar transition of Bedouin tribes where livestock were traditionally moved every three to five days to one of increased sedenterization around settlements. Again, the result was an observed decline in rangeland condition.

If sedenterization leads to the overgrazing of plants, a loss of biodiversity, and ultimately desertification, it seems reasonable to expect the opposite treatment (nomadism) to yield opposing results upon the landscape. However it does not (Savory 1999) and what has been observed is that both nomadism and more sedentary grazing practices can lead to desertification, albeit at different rates of degradation. Pastoralism, given adequate land area and freedom to move, simply leads to more gradual desertification than sedentary practices.

Hence, mobility alone is not the key and simply describing nomads as mobile does not adequately capture the essence of the grazing practices followed by the nomadic pastoralist. To look at it another way, would a grazier who moves his livestock to fresh pasture twice each year be considered a nomadic pastoralist? What if he moved his herd or flock 12 times per year, or 150 times per year covering hundreds of kilometers in the process? Only in the latter example would one consider the hypothetical grazier a nomadic pastoralist. In terms of land management, the effective difference between the former examples of punctuated sedenterism and nomadism is the amount of time spent grazing one area before moving to another and the amount of time allowed for recovery of the plants (Voisin 1988).

While Voisin (1988) advocated that overgrazing of plants was the greatest influence in land degradation and desertification, only more recently have the effects of partial-rest and total rest been more fully understood (Behnke 1999; Niamir-Fuller and Turner 1999; Cummins 2009) as factors that tend to override the influence of overgrazing and may consequently be the principle factors driving rangelands toward desertification. In addition, it was Savory (1999) who observed that "Since about two-thirds of the earth's land surface is brittle [e.g., arid or semiarid rangelands]... and since the dawn of agriculture it has carried livestock under management that paradoxically produces both partial-rest and overgrazing of plants, the remorseless growth of deserts is no mystery".

The western rangelands of North America are little different than many rangelands where traditional pastoralism has been practiced for thousands of years. Both are typically arid or semiarid environments dominated by grasses and shrubs, grazed by domesticated cattle, sheep, and goats. The primary and perhaps only difference is that traditional pastoralism is a means of subsistence whereas ranching is a market-oriented business (Cummins 2009). As noted earlier, shifts towards market-oriented grazing leads to sedenterization (cf. partial-rest of rangelands) which in turn leads to a more rapid overgrazing of plants, loss of biodiversity, and accelerated desertification. This market-oriented shift has also changed land tenure as significant acreages are now held in "public lands" all of which are managed, by policy, under regimes of partial-rest or total rest. Is this the future of the world's rangelands? Could a change be made to reduce the latency of livestock within a pasture or paddock while eliminating the negative impact of partial-rest to thereby improve rangeland ecosystems?

The latter is a very large and important question and certainly some will argue that the suggested change will not yield the expected results in spite of the historical observations referenced throughout this paper indicating otherwise. This then becomes both a dilemma and a challenge for the future of rangeland ecosystems, range science, range managers, and graziers across the globe.

SUMMARY

While numerous pastoral cultures have subsisted for thousands of years and continue to survive today, nearly all are facing great difficulties as their landscapes deteriorate. Historical observations suggest that desertification is the result of both climatic and anthropic factors with specific emphasis recently placed upon the effect of sedenterization and the subsequent negative feedback cycle initiated through partial-rest and total rest found across nearly all continents, societies, and grazing cultures today. As a result, it is suggested that "management systems" be re-considered and supplanted by more inclusive planning processes focusing upon improving arid and semiarid rangeland ecosystems through the use of livestock as a solution to the problem of desertification. Savory (1999) made the point that for all of human history, mankind has tried to manage the environment using only three "tools" (technology, fire and resting land). He further pointed out that none of these tools can achieve what is required to reverse desertification, and that as long as humans continue to use fire as a surrogate for grazing animals and the management of moribund grass, desertification will only continue to worsen across the globe.

ACKNOWLEDGEMENTS

This study was made possible by a grant from the National Aeronautics and Space Administration (NASA) Goddard Space Flight Center (NNX06AE47G). Idaho State University would also like to acknowledge the Idaho Delegation for their assistance in obtaining this grant.

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Recommended citation style:
Weber, K. T. and S. Horst. 2009. *Applying Knowledge of Traditional Pastoralists to Current Range Management*. Pages 157-168 in K.T. Weber and K. Davis (Eds.), Final Report: Comparing Effects of Management Practices on Rangeland Health with Geospatial Technologies (NNX06AE47G). 168 pp.