USE OF SATELLITE IMAGERY

TO MEASURE COVER OF PRAIRIE VEGETATION

FOR THE DETECTION OF CHANGE

by

Rebecca Jeanne Hurst

A thesis submitted in partial fulfillment
of the requirements for the degree

of

Master's of Science

in

Biological Sciences

MONTANA STATE UNIVERSITY
Bozeman, Montana

April 2006

APPROVAL

of a thesis submitted by

Rebecca Jeanne Hurst

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and is ready for submission to the Division of Graduate Education.

Dr. Theodore W. Weaver

Approved for the Department of Ecology

Dr. David Roberts

Approved for the Division of Graduate Education

Dr. Joseph J. Fedock

STATEMENT OF PERMISSION TO USE

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ABSTRACT

Adaptive resource management requires a cost effective, easily repeatable tool for measurement of vegetation quality and comparison of management treatment effects across a natural area and through time. High-resolution satellite imagery may be used as a guide for treatment by facilitating either the comparison of differentially treated units or the measurement of change in vegetation through time. Demonstration of successful application of remote sensing for measurement of vegetation to recognize trends in vegetation quality will help adaptive managers both to apply and to improve the methods for management of vegetation. Space Imaging's IKONOS satellite imagery was used first to map mixed grass prairie communities in the Missouri Coteau region of North Dakota and Montana. Vegetation was classified hierarchically into 5 classes (wetland, tree, shrub, grass/dwarf shrub, and pure grass) with an overall accuracy of 72%. The resultant map was used to sample vegetation of units with various management histories in Lostwood National Wildlife Refuge to measure differences across fire and grazing treatments. Differences across treatments were slight. Satellite imagery may provide the best tool for resampling needed in this and other systems. Sampling with remote sensing may be more expedient than ground surveying, but classifications must have higher accuracies and less bias to be useful to managers. Therefore, satellite imagery needs further development to support on-going adaptive resource management.

CHAPTER 1

INTRODUCTION

Adaptive resource management is used to manage treatment of vegetation such as is found at Lostwood National Wildlife Refuge (NWR). Managers observe condition under current management and modify treatment as necessary to guide trends toward the desired vegetation. In a heterogeneous landscape such as Lostwood NWR, topographic units may need different treatment both because topography affects presence of some species (Coupland, 1950) and because topography may also enhance or hinder the treatments. The object of current management at Lostwood NWR is to reduce woody species and encourage replacement of woody by herbaceous species. Fire and grazing, alone or in conjunction, are potential tools for managing prairie vegetation. Fire is used both to control woody species directly and to encourage herbaceous species to compete with woody species. Grazing may decrease woody species by browsing and trampling.

High-resolution satellite imagery may be used to measure differences in condition as a guide for treatment either by facilitating comparison of differentially treated units or by facilitating the measurement of change in vegetation through time. It may be the manager’s best tool for measuring differences in vegetation, because alternative methods, including standard field inventory, aerial photo interpretation, and multispectral scanner data analysis, are too costly, too time consuming, too subjective, and too processing-intensive to be reasonably used by land managers (Coulter *et al.*, 2000; Jensen *et al.*, 1991; Mehner *et al.*, 2004). Remote monitoring of effects will likely allow managers both to compare various treatments simultaneously across large reserves and to do so more frequently than can be done with ground survey methods. Demonstration of successful application of remote sensing for measurement of vegetation to recognize trends in vegetation quality will help adaptive managers both to apply and to improve the methods for management of vegetation.

Objectives

The first objective was to test the potential of multi-temporal high-resolution satellite imagery for identification and mapping of the major vegetation types of the mixed-grass prairie.

The second objective is two part: first, to measure effects of fire and grazing management of shrubby vegetation of the mixed-grass prairie and second, by doing so, to establish high-resolution imagery as a useful tool to measure management effects of prairie vegetation in the northern Great Plains.

Organization of Thesis

Chapter 2 discusses the use of multi-temporal high-resolution satellite imagery to classify and map land cover of the rolling topography of the Missouri Coteau. Chapter 3 discusses the measurement of effects of fire, cattle grazing, and habitat with this imagery on mixed grass prairie vegetation.

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CHAPTER 2

MAPPING PRAIRIE POTHOLE COMMUNITIES WITH MULTI-TEMPORAL IKONOS SATELLITE IMAGERY

Introduction

The advent of readily available high spatial resolution commercial satellite imagery (Petrie, 2001) presents important new opportunities for land managers and researchers needing classifications of landscapes that are heterogeneous at fine scales. Vegetation, for example, often varies at spatial resolutions finer than are detectable using widely available moderate resolution imagery, such as Landsat-based imagery. Traditional vegetation sampling methods use 1/10 m^2 to 1 m^2 plot sizes to inventory grassy communities (Coupland, 1950; Daubenmire, 1959). Plant communities of the Missouri Coteau, the terminal moraine of the Wisconsin Glacier, which reaches from north central Montana to Iowa were examined. The Missouri Coteau's pothole and hilltop topography provides repeated examples of a moisture gradient occupied by vegetation reaching from aquatic through aspen, snowberry, tall grass prairie, mixed grass prairie to short grass prairie (Smith, 1998). The Missouri Coteau is valuable for range, agriculture, and wildlife, including migrating waterfowl (Murphy, 1993; Rolling & Dhuyvetter, 2003).

The ability to classify accurately both grassland communities and associated wetlands, such as the prairies of the Missouri Coteau and their potholes with emergent vegetation, is of vital importance. More than one-fourth of the Earth's land surface and over 60% of the United States is classified as grassland (Williams *et al.*, 1968; Holecheck *et al.*, 1989; Laurenroth, 1979). Grasslands are critical for wildlife habitat, plant species diversity, hydrologic functions, ecosystem nutrient cycling, and grazing (Campbell and Lasley, 1969; Pearse, 1971). Worldwide decline of grasslands from pre-settlement to present due to woody plant encroachment and non-native grass introduction threatens these valuable habitats (Archer, 1990; Mast *et al.*, 1997; Murphy, 1993; Roques *et al.*, 2001; Silva *et al.*, 2001; Van Auken, 2000). The importance of discriminating among grassland types with remote sensing has been noted as particularly essential because of the vast extent of these ecosystems (Price *et al.*, 2001). Wetlands, such as are found with the potholes of the Missouri Coteau, are similarly critical habitat for species including migratory waterfowl, and mapping such features is critical to land use decisions (Muller et al., 1993; Semlitsch and Bodie, 1998).

Satellite imagery has been used extensively to map grassland vegetation. Moderate resolution imagery, however, has been almost the exclusive tool for such mapping, thereby limiting such efforts to either broad vegetation categories or areas of homogeneous cover types at the resolution of the imagery. Landsat imagery has been used to discriminate between cool- and warm-season grasses in eastern Kansas (Price *et al.*, 2002), four grassland habitat types in North Dakota (Jensen *et al.*, 2001), rough fescue grassland in western Canada (Thomson *et al.*, 1985), ten plant communities in southwestern Idaho (Clark *et al.*, 2001), and eight major grassland and shrub land groups in southwestern Idaho (Knick *et al.*, 1997). Classification accuracies ranged from 60% to over 90%, indicating that Landsat imagery has substantial potential for mapping grasslands where the vegetation communities occur in sufficiently homogeneous areas to be detectable at 30-m resolution. This resolution is inadequate for management purposes where species coverage occurs at a finer scale. Mixed grass prairie communities change with topography (Coupland, 1950). Woody species encroachment and non-native grass species invasion occur at the size of reproduction of individuals (Briggs *et al.*, 2002; Lett and Knapp, 2005). Waterfowl habitat needs are relative to the size and abundance of the nesting species (Murphy, 1993). Alternative methods, including standard field inventory, aerial photo interpretation, and multispectral scanner data analysis, are too costly, too time consuming, too subjective, and too processing intensive to be reasonably used by land managers (Coulter *et al.*, 2000; Jensen *et al.*, 1991; Mehner *et al.*, 2004).

Commercial high spatial resolution satellite-based sensors, including IKONOS and Quickbird, can provide classifications at resolutions of 4 m or less. Imagery from these sensors has been used for many applications, including monitoring prairie dog colonies (Sidle *et al.*, 2002), building extraction (Lee *et al.*, 2003), water monitoring and analysis (Huguenin *et al.*, 2004; JiQun *et al.*, 2004), site-specific agriculture (Metternicht, 2004; Vina *et al.*, 2003), documenting vegetation degradation in mountainous environments (Allard, 2003), measuring tree mortality (Clark *et al.*, 2004), estimating leaf area index (Colombo *et al.*, 2003; Johnson *et al.*, 2003), and assessing coral-reefs (Maeder *et al.*, 2002; Palandro *et al.*, 2003). Few reported studies, however, have used these sensors for classification of undeveloped land cover (but see, e.g., Carleer and Wolff, 2004; Mehner *et al.*, 2004; Quinton *et al.*, 2003; Sawaya *et al.*, 2003), and the use of these data to examine grassland communities does not seem to be well explored. One possible reason for this lack of application might be that these sensors, having sensitivity in the visible and near infrared portions of the spectrum (Goetz *et al.*, 2003; Thenkabail *et al.*, 2004), have less spectral resolution than Landsat, which also has sensitivity in the middle and thermal infrared (NASA, 2004), although many studies do not use the coarser spatial resolution thermal infrared.

One possible solution to the lack of spectral resolution in IKONOS and Quickbird imagery compared to Landsat is to incorporate temporal information through the use of multiple images across the growing season. Use of multi-date imagery has been shown to achieve higher classification accuracies (Guo *et al.*, 2000) and has two potential advantages. First, certain cover types might be best distinguished on one date, while other cover types might be best distinguished on another date (e.g., Lawrence and Wright, 2001; Reed *et al.*, 1994). Second, difference images (those created by subtracting spectral values of one date from spectral values on another date (Coppin *et al.*, 2004)), certain components from multi-temporal principal components analysis (Fung and LeDrew, 1987; Eastman and Fulk, 1993), or other techniques can represent differences in plant phenology among different cover types (Coppin *et al.*, 2004).

The objective was to evaluate the ability of multi-temporal high spatial resolution satellite imagery to map vegetation communities in a spatially fine scale heterogeneous region. Classification success would suggest that such analysis could overcome the spatial resolution limitations of sensors such as Landsat imagery as well as the spectral resolution limitations of sensors such as IKONOS and Quickbird.

Methods

The study site was the northern half of the Lostwood National Wildlife Refuge (LNWR). LNWR is located in Burke and Montrail Counties of northwestern North Dakota, 37 km south of Canada and 113 km east of Montana (Figure 1.1) at an elevation of 675 m to 764 m. This terrain comprises non-integrated wetland drainages, filled by snowmelt and rainfall through surface runoff and subsurface seepage. The 10,888-ha refuge, thus, is dotted with 2,178 ha of prairie wetlands (Smith, 1998). The Prairie Pothole Region of the Missouri Coteau is useful for grazing and farming and, with its wetlands, is important for migratory waterfowl and upland bird species (Rolling and Dhuyvetter, 2003).

High spatial heterogeneity of vegetation types at LNWR is due primarily to diverse habitats (bottoms, slopes, and hilltops) in the rolling topography and, secondarily, to the clonal spread of grasses, shrubs, and trees. Vegetation includes native wetland communities; native prairie communities; native, but invasive, tree and shrub communities; and introduced cropland species. The pothole wetlands are diverse in size, depth, vegetation, and water quality. The primary woody communities are *Populus tremuloides* Michx. (aspen) and *Symphoricarpos occidentalis* Hook. (snowberry). The grassy prairie communities include northern phases of tall, mixed, and short grass prairie, dominated by *Stipa comata* Trin. & Rupr. (needle-and-thread), *Mulenbergia cuspidata* (Torr.) Rydb. (plains muhly), *Bouteloua gracilis* (H.B.K.) Lag. *ex* Griffiths(blue grama), *Agropyron smithii* Rydb. (western wheatgrass), *Stipa viridula* Trin. (green needlegrass), *Festuca scabrella* Torr. in Hook(rough fescue), *Mulenbergia richardsonis* (Trin.)

Rydberg. (mat muhly). Dominant introduced grasses are *Bromus inermis* Leyss. (smooth brome) and *Poa pratensis* L. (Kentucky blue grass). Both introduced grasses are highly competitive, perennial, rhizomatous, sod-formers. Plant associations described by Madden (1996) were followed. Nomenclature followed *Flora of the Great Plains* (1986).

Figure 2.1. Location of study site, Lostwood National Wildlife Refuge, North Dakota, USA, shown with an IKONOS panchromatic image.

Lostwood NWR's climate is semi-arid, with an annual mean precipitation of 43 cm and deviations of greater than 10 cm in four of ten years (Rolling and Dhuyvetter, 2003). The average annual temperatures at LNWR are between -40˚C in winter (January max/min -9/-20˚C) and 38˚C in summer (July max/min 28/11˚C). The growing season, therefore, is limited by winter storms that can occur as late as early June and frosts as early as August.

Space Imaging IKONOS multispectral imagery was acquired on 11 August 2000 and 10 October 2000, with the cooperation of the US Department of the Interior and NASA. The August imagery was received as stereo pairs, from which the supplier extracted a digital elevation model (DEM). The 12-m resolution of the DEM proved too coarse for use in this fine scale landscape, so it was omitted for use in this project. Spring, summer and fall images to capture maximum phenological variability were attempted, but clear images were unattainable prior to August due to weather and acquisition difficulties. The imagery has a spatial resolution of 4 m and included four spectral bands (blue, 0.45-0.52 µm; green, 0.52-0.59 µm; red, 0.62-0.68 µm; and near-infrared, 0.77-0.86 µm). Pixel values represented 11-bit scaled radiance values. Imagery was georeferenced to a Universal Transverse Mercator (UTM WGS 84) coordinate system and the images were registered to within one pixel. In addition to the raw IKONOS spectral bands, several derived components to represent potential changes in vegetation spectral responses between the two dates were used. These components included (1) difference images created for each of the four bands by subtracting the August spectral values from the October spectral values and (2) seven principal components from a principal components analysis (PCA) of the eight spectral bands from the two dates (the first principal component was determined to not include change data). The difference image was created and PCA performed using ERDAS Imagine software.

The reference data for classification and accuracy assessment were collected using on-ground surveys with differential GPS during summer months. A minimum of 10 circular plots were collected for each of 21 vegetation cover types (later combined unto five types for analysis), with each plot having a 6 m radius. Sites were selected for a greater than 6-m radius homogenous community type, falling entirely within the same available moisture zone, across as many management units as found to represent variability within community types. Aerial photographs were interpreted while on site, for sites where aspen canopy cover would inhibit GPS use. Reference data for each cover type were randomly divided into equal training and accuracy assessment data sets. Sets were imported into ArcView, polygon coverages were created and pixel-based signatures were extracted. The polygons were converted to ERDAS Imagine area of interest (aoi) files to develop ASCII data files for use in S-Plus for classification tree analysis (CTA) and accuracy assessment.

Data used in the classification included 8 original spectral bands, 4 difference image components, and 7 principal components. CTA in S-Plus statistical software was used to create a set of decision trees and associated classification rules for the study area. CTA (sometimes referred to as classification and regression tree analysis, CART, decision trees, or recursive partitioning) is a non-parametric classification algorithm that has been demonstrated to be effective in classifying complex data sets with multi-temporal components (Lawrence and Wright, 2000). CTA classifies hierarchically, removing distinct, broad classes and has also been shown to be a better classifier of grasslands and wooded grasslands than maximum likelihood classification (Hansen *et al.*, 1996). An initial attempt of maximum likelihood classification performed poorly, with 50-60% accuracies.

Classification was conducted hierarchically, with each of three levels representing increasing discrimination among cover types (Table 2.1). Level 1, therefore, included only the broadest differentiation of potholes (including emergent vegetation) and upland vegetation. At Level 2, upland vegetation was segregated into two functional classes, woody vegetation (consisting of aspen and snowberry) and grassy vegetation (consisting of grasses with dwarf shrubs and grasses alone). Finally, at Level 3 woody vegetation was segregated into two species classes, aspen (both tree and shrub forms) and snowberry, and grassy vegetation was segregated into two functional classes, grasses with dwarf shrubs and grasses alone. No further levels were tested, as accuracies at this last level dropped to near 70%. A reclassification on all misclassified points was attempted at each level to improve classification, but results did not improve substantially. Accuracy assessments followed traditional error atrix methods (Congalton and Green, 1999).

Table 2.1. Classification scheme for IKONOS imagery of Lostwood National Wildlife Refuge

Level 1	Level 2	Level 3
Potholes, including emergent vegetation	Potholes, including emergent vegetation	Potholes, including emergent vegetation
Upland vegetation	Woody vegetation	Aspen
		Snowberry
	Grassy vegetation	Grass with dwarf shrubs
		Grasses alone

Results

Overall accuracy for the level 1 classification was 92% (Kappa statistic 0.79). Individual class accuracies ranged from a low of 73% for producer's accuracy for potholes to 100% for producer's accuracy for upland species (Table 2.2). Most confusion was due to the classification of some pothole areas as upland vegetation. Areas of dense emergent vegetation, which had similar spectral characteristics to upland vegetation, were the primary cause of this error.

Table 2.2. Level 1 Classification Accuracy Assessment

	Potholes/ Evergent Vegetation	Upland Vegetation	Totals	User's Accuracy
Potholes/emergent vegetation	215	3	218	98.6%
Upland vegetation	78	759	837	90.7%
Totals	293	762	1055	
Producer's accuracy	73.4%	99.6%		
Overall accuracy	92.3%			
Kappa	0.79			

The decision tree for level 1 was fairly simple, with 5 terminal nodes and the incorporation of 3 of the 19 potential explanatory variables (Figure 2.2). Explanatory variables used in the classification included the near infrared and green bands from the August IKONOS image and the red band difference component. Potholes were identified primarily by lower radiance in the near infrared portion of the spectrum for water-dominated areas, as would be expected due to the high absorption of infrared by water. Other pothole areas were distinguished by high green radiance, probably because ponded water kept emergent vegetation greener than upland vegetation in August. Finally, some pothole areas were distinguished by high near infrared responses, possibly also indicating the presence of healthy vegetation in emergent vegetation zones.

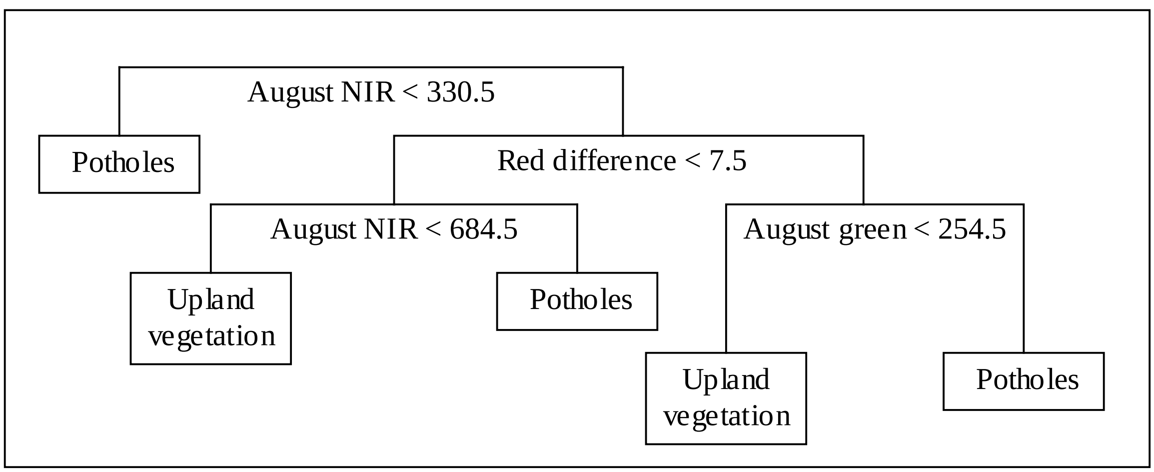

Figure 2.2. Decision tree for the level 1 classification. Splitting rules at each node indicate the left branching at the nodes. Difference components were created by subtracting the August spectral values from the October spectral values.

For the level 2 classification, which segregated upland vegetation into grassy and woody vegetation, overall classification was 80% (Kappa statistic 0.68). Individual class accuracies ranged from 68% for woody vegetation producer's accuracy to 99% for pothole user's accuracy (Table 2.3). The largest source of confusion was woody vegetation being classified as grassy vegetation, while the sum of grassy vegetation classified as woody vegetation and potholes classified as grassy vegetation accounted for an equivalent amount of error.

Table 2.3. Level 2 Classification Accuracy Assessment

	Potholes/ Emergent Vegetation	Woody Vegetation	Grassy Vegetation	Totals	User's Accuracy
Potholes/emergent vegetation	215	3	0	218	98.6%
Woody vegetation	28	199	41	268	74.3%
Grassy vegetation	50	89	430	569	75.6%
Totals	293	291	471	1055	
Producer's accuracy	73.4%	68.4%	91.3%		
Overall accuracy	80.0%				
Kappa	0.68				

The decision tree for level 2, which segregated upland vegetation into woody and grassy vegetation, was much more complex than the level 1 decision tree, had 9 terminal nodes, and incorporated 6 explanatory variables, the red band from the August image, the green, red, and near infrared bands from the October image, and principal components 5 and 6 (Figure 2.3). I interpreted principal component 5 as representing changes in green spectral radiance from August to October and principal component 6 as primarily changes in blue radiance between the two images. Most grassy vegetation was distinguished by higher values in the October red band, possibly due to earlier senescence for grasses (i.e., less absorption by photosynthetically active vegetation). For those observations with lower red radiance, much of the woody vegetation was distinguished from grassy vegetation by lower radiance in the August red band, perhaps for the same reason.

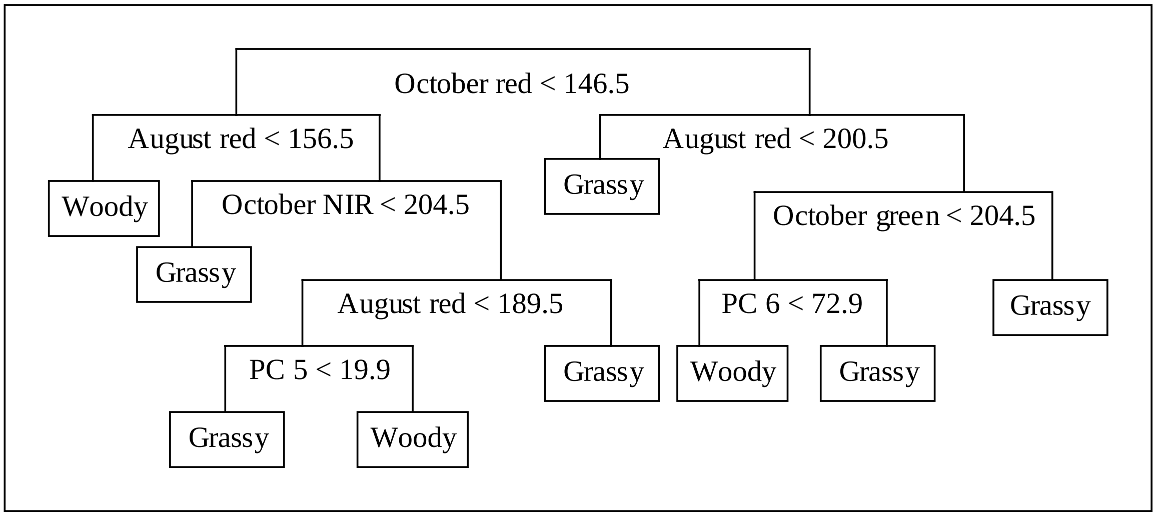

Figure 2.3. Decision tree for the level 2 classification segregating upland vegetation into woody vegetation and grassy vegetation. Splitting rules at each node indicate the left branching at the nodes. PC refers to multi-temporal principal components created by principals components analysis of all eight bands from the two dates of imagery.

For the level 3 classification, which segregated woody vegetation into aspen and snowberry and distinguished grassy vegetation into grasses with dwarf shrubs and grasses alone, overall classification was 72% (Kappa statistic 0.62). Individual class accuracies ranged from 47% for snowberry producer's accuracy to 99% for pothole user's accuracy (Table 2.4). Grasses alone were most often confused with snowberry and grasses with dwarf shrubs, while snowberry was the largest source of confusion for aspen. In addition, the grasses with dwarf shrubs were poorly distinguished and substantial confusion remained among aspen and other classes.

Table 2.4. Level 3 Classification Accuracy Assessment

	Potholes/ Emergent Vegetation	Aspen	Snowberry	Grasses with Dwarf Shrub	Grasses Alone	Totals	User's Accuracy
Potholes/emergent vegetation	215	3	0	0	0	218	98.6%
Aspen	0	88	5	0	3	96	91.7%
Snowberry	28	23	83	4	34	172	48.3%
Grasses with Dwarf Shrub	0	0	19	54	14	87	62.1%
Grasses alone	50	0	70	43	319	482	66.2%
Totals	293	114	177	101	370	1055	
Producer's accuracy	73.4%	77.2%	46.9%	53.5%	86.2%		
Overall accuracy	71.9%						
Kappa	0.62						

The decision tree segregating woody vegetation types for level 3 had 3 terminal nodes and used 2 explanatory variables, the red band from the October image and the green band from the August image (Figure 2.4). Snowberry was distinguished by having higher responses in both bands, probably because the higher leaf area in aspen resulted in more absorption in the visible bands. The aspen present at the site, however, is mostly in the shrub stage, which would have leaf areas that overlap with those of snowberry, resulting in substantial residual confusion between these classes.

The decision tree segregating grasses with dwarf shrubs and grasses alone had 5 terminal nodes and used 4 explanatory variables, the near infrared band from the October image and principal components 2, 4, 7, and 8 (Figure 2.4). I interpreted principal component 2 as change in the visible bands, principal components 4 and 7 both as change in all bands except green, and principal component 8 as change in all bands. Grasses with dwarf shrubs were distinguished from grasses without dwarf shrubs primarily by having lower values in all principal components, indicating that the grasses with dwarf shrubs had less change in spectral values from August to October. One possible explanation for this might be that the dwarf shrubs senesced later than the grasses, resulting in less change between the two dates. An alternative might be that dwarf shrubs tend to appear on drier sites with less vegetation cover and more soil exposure. The exposed soil might result in a more constant spectral signature over time. The dwarf shrubs, however, are often topped by tall grasses, which could explain the substantial confusion between these two classes.

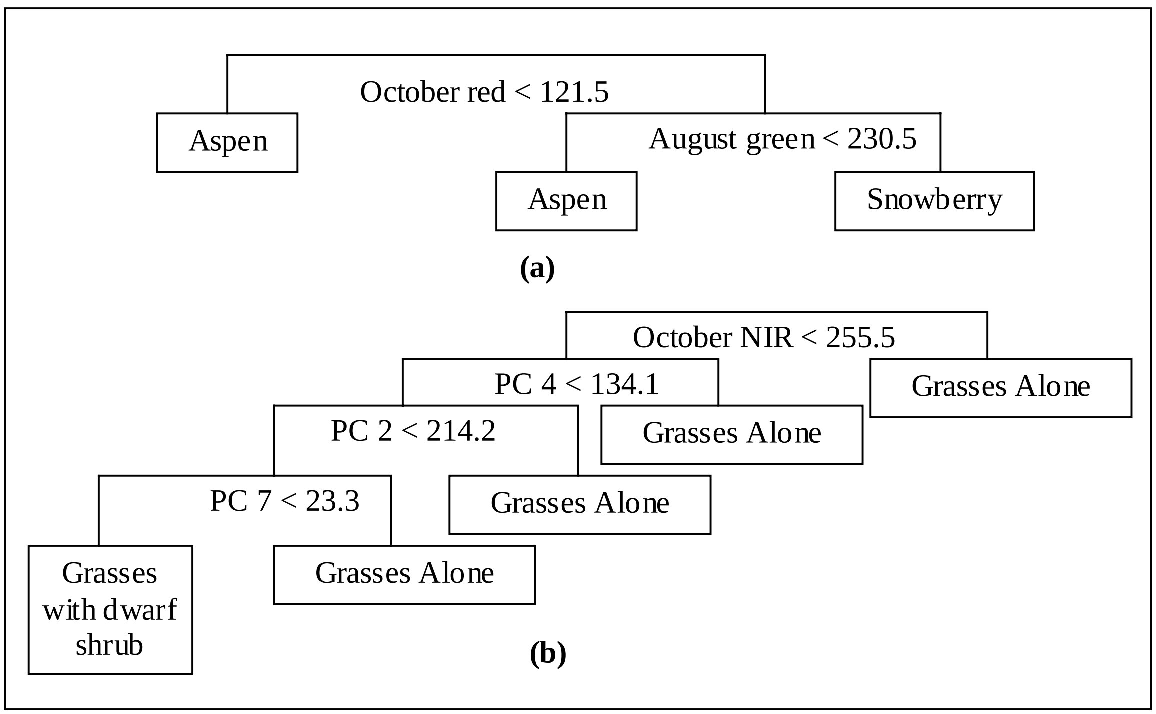

Figure 2.4. Decision trees for the level 3 classifications: (a) segregating woody vegetation in to aspen and snowberry and (b) segregating grassy vegetation into grasses with dwarf shrub and grasses alone. Splitting rules at each node indicate the left branching at the nodes. PC refers to the multi-temporal principal components created by principle components analysis of all eight bands from the two dates of imagery.

Discussion

Classification of multidate IKONOS imagery for the prairie pothole site was fairly successful, with some notable exceptions. Classes at the broad first level, pothole versus upland vegetation, were well distinguished, as were most classes at the second level, grassy versus woody vegetation. At the third level, which included species and near species classes, however, accuracies dropped substantially. The snowberry and grass with dwarf shrub classes, in particular, were not well distinguished from grasses alone. Examination of the decision trees used to create these classifications suggests the probable reason for this confusion. I believe the primary method of distinguishing grasses with dwarf shrubs and snowberry from grasses alone was differences in rates of spectral change between August and October, due either to differences in rates of senescence, soil exposure, or both.

The decision trees used for the classifications support the concept that temporal information might be valuable for separating vegetation types. Raw spectral bands from both dates were used in the decision trees, although it is unknown whether this resulted from distinctions being evident only on certain dates or slight statistical advantages from a band on one date versus another. More compelling was the prominence of derived data representing changes from one date to the other. The difference in red radiance was important to the level 1 classification. Several principal components that represented changes in radiance between the two dates were important at levels 2 and 3. These principal component data were particularly important because late in the season, when the imagery was acquired, the rates of phenological change vary substantially among species. Between image dates, for example, aspen leaves will have yellowed and most fallen, shrubs will become leafless, and cool season grasses will completely brown. The classifications, therefore, were able to exploit some of these differences among classes that most likely otherwise would not have been distinguishable because of the substantial overlap in spectral responses among classes on any single date.

The success of the classifications also might have been affected by the particular dates of imagery obtained. Although contracted for spring, summer, and fall images, Space Imaging was not able to obtain a spring IKONOS image during the study period. The failure primarily was due to unacceptable cloud conditions (one image was obtained but was returned as unacceptable for this reason, another was clear but followed a late season snow storm with substantial drifting snow still on the ground). The use of images from more phenologically varied periods, e.g. using an image from mid-April to mid-May, during initial growth starting with cool season grasses, then aspen clones, and followed closely by shrubs, but before initial growth of warm season grasses, a full month behind the green-up of cool season grasses, might have been able to detect variations in vegetation that were not evident from two relatively late season images. The use of multiple spring and autumn images might have detected variations in vegetation reflecting patterns of “green-up” and “green-down” across community types. Additional community type signatures to represent more of the variability across the vegetation, a finer-scale DEM to develop ancillary data representing moisture availability, or the use of a Normalized Difference Vegetation Index (NDVI) to enhance discrimination across large management units with differing treatments, where more or less biomass is produced by the same vegetation types, might alone or in conjunction improve classification results.

Conclusion

The objectives were to determine whether multi-temporal IKONOS imagery could successfully map prairie pothole communities, both in terms of delineating potholes with their emergent vegetation and in distinguishing among upland vegetation communities. The results indicate that these data can be highly successful for pothole and wetland determination. Results for upland communities were mixed. Broad categories were fairly well distinguished, and I found that the multi-temporal nature of the data was often important for making these distinctions. Several similar communities, however, remained confused. It is important for future research to determine whether obtaining imagery for more distinctive dates, such as spring and fall, and incorporating additional signature, ancillary, and sensor data can overcome these problems.

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CHAPTER 3

EFFECTS OF FIRE AND GRAZING IN MIXED GRASS PRAIRIE

Introduction

The abundance of woody vegetation has increased substantially in savannas and grasslands worldwide in the last century (Murphy, 1993; Archer *et al.*, 1995; Bragg and Hulbert, 1996; McPherson, 1997; Briggs, *et al*., 2002; Lett and Knapp, 2005). Invasion of woody vegetation changes composition and structure of northern mixed grass prairies (Plaggemeyer, 2003). Woody species (including aspen, *Populus tremuloides* Michx.; snowberry, *Symphoricarpos occidentalis* Hook.; and silverberry, *Elaeagnus commutata* Bernh.) are replacing native northern mixed grass communities in the Missouri Coteau region. Resource managers seek to reverse woody species encroachment.

Three primary factors, alone or in conjunction, may determine woody species abundance in northern mixed grass prairies. Woody plants may have increased with the decrease of fire. Woody plants may have increased with the decrease of grazing. With fire removal, woody plants may have increased most in moist areas. With grazing removal, woody plants may have increased more on relatively wet areas. The woody species encroachment since settlement in the managed native mixed grass prairie of the Missouri Coteau region has been clearly demonstrated at Lostwood National Wildlife Refuge (NWR) in northwestern North Dakota (Murphy, 1993).

Woody species encroachment in the mixed grass prairie is correlated with fire reduction. Pre-settlement fires were both small and frequent (Wells, 1970; Higgins, 1986). Homesteading suppressed fires – perhaps by providing barriers, reducing sets, and active firefighting - with the last major burns occurring in 1905 and 1909 (Burke County and White Earth Valley Historical Society, 1972). Since settlement, woody vegetation at Lostwood NWR increased dramatically between 1910 (5-10%) and 1970 (>50%) (Murphy, 1993).

Woody species encroachment in the mixed grass prairie is simultaneously correlated with bison extirpation. Numerous references to the abundance of bison in the Missouri Coteau region were made during the 1800s by early explorers – Lewis and Clark, Henry, and Clandening (in Coues, 1893; in Coues, 1897; Clandening, 1928). During his journey in 1863, Clandening comments "grass much eaten by buffalo" (Clandening, 1928). Bison extirpation occurred at Lostwood NWR in the 1870s (Coues, 1878, Hornaday, 1889) and aspen expansion occurred concurrently (Campbell, *et al*., 1994).

To determine which, if any, of these factors is responsible for the change in woody vegetation cover, an experiment with independent treatments is needed. Management treatments at Lostwood NWR provide a remarkable factorial management experiment (6 burning levels x 3 grazing levels) that can be used to measure the relative impacts of fire and grazing inputs.

To observe the effects of management treatments, high-resolution satellite imagery may be a useful tool. Land managers need a reliable, easily repeatable way to monitor community response to treatment, e.g., of woody species. Traditional ground surveys are inadequate because the slow progress of surveying allows confounding of treatment effects with variable (within year and between years) conditions, e.g., weather, grazing, and plant growth (Coupland, 1950; Daubenmire, 1959; Coulter *et al.*, 2000; Jensen *et al.*, 1991; Mehner *et al.*, 2004). Moderate resolution satellite imagery, e.g. Landsat, has been used to classify grassland vegetation (Thomson *et al.*, 1985; Knick *et al.*, 1997; Clark *et al.*, 2001; Jensen *et al.*, 2001; Price *et al.*, 2002). High resolution IKONOS satellite imagery has been used to classify and map vegetation (Allard, 2003; Colombo *et al.*, 2003; Johnson *et al.*, 2003; Clark *et al.*, 2004; Mehner *et al.*, 2004; Lawrence *et al*., 2006). The use of sampling with remote imaging to monitor the effects of various treatments, (e.g., burning and grazing regimes) applied in a management experiment will be tested. If possible remote sensing will provide a basis for adaptive resource management, i.e., monitoring effects of current management across time as a basis for instituting change needed to sustain grassland (e.g., reverse woody species encroachment).

The objectives are to 1) to determine whether woody vegetation declines uphill, 2) to determine whether a statistically significant negative correlation exists between increasing numbers of burns and the cover (%) of woody vegetation and reciprocally whether a statistically significant positive correlation exists between increasing numbers of burns and the cover (%) of grassy vegetation, 3) to determine whether a statistically significant positive correlation exists between years of rest after fire and the cover (%) of woody vegetation and reciprocally whether a statistically significant negative correlation exists between years of rest after fire and grassy vegetation, 4) to determine whether a statistically significant negative correlation exists between grazing and the cover (%) of woody vegetation and reciprocally whether a statistically significant positive correlation exists between grazing and the cover (%) of grassy vegetation.

Methods

Records spanning 25 years of fire and grazing management at Lostwood NWR document a remarkable management experiment for testing the hypotheses. It allowed for testing the effects of topography (four types), the burning continuum (six types), and grazing levels (three types) (Table 3.1).

Table 3.1. Fire and grazing histories of management units at Lostwood NWR

	Number of Burns						
	0	1	2	3	4	5	6
Ungrazed	0	0	3	3	3	1	0
Grazed, rested	2	0	0	3	1	1	0
Grazed, current	0	0	0	2	1	3	1

Study Site

Vegetation communities were examined at Lostwood NWR, Kenmare, North Dakota. Lostwood NWR is home to 10,888 ha of mixed grass prairie vegetation, including 2,178 ha of prairie wetlands. The flora is typical of the mixed grass prairie (Küchler, 1964) of the northern Great Plains. Dominant components include *Agropyron smithii* Rydb. (western wheatgrass), *Bouteloua gracilis* (H.B.K.) Lag. *ex* Griffiths (blue grama), *Festuca scabrella* Torr. in Hook(rough fescue), *Mulenbergia cuspidata* (Torr.) Rydb. (plains muhly), *Mulenbergia richardsonis* (Trin.) Rydberg. (mat muhly), *Stipa* *comata* Trin. & Rupr. (needle-and-thread grass), and *Stipa viridula* Trin. (green needlegrass) (Küchler, 1964). Invasive, native components include *P. tremuloides* (aspen), *S. occidentalis* (western snowberry), *E. commutata* (silverberry), *Poa pratensis* L. (Kentucky bluegrass), and *Bromus inermis* Leyss. (smooth brome). Nomenclature followed *Flora of the Great Plains* (1986).

Lostwood is located in the Missouri Coteau, a rolling landscape deposited by the terminal moraine of the Wisconsin Glacier. The Missouri Coteau's rolling topography provides habitat for communities along a moisture gradient from wetland vegetation, through mesic (tall, warm-season grasses, and woody species clones around potholes), tall, cool-season grasses with shrub clones on slopes, to xeric, short grasses on hilltops (Smith, 1998). The climate of the area is semi-arid with a mean annual precipitation of 42 cm. During the 2000 growing season, when images were acquired, the average monthly precipitation was 11.8 cm (less than the normal 14.5 cm). The Missouri Coteau region is important for agriculture and wildlife. Lostwood NWR is especially important to migratory wildlife (Murphy, 1993).

Data Collection

Lostwood NWR was divided into management units with fire, grazing, and fire with grazing histories providing a factorial management experiment to measure these treatment effects on woody vegetation. From the management units, a set of 24 environmentally similar units were chosen. Fire effects were measured by comparing units which were burned 0 to 6 times, with variable years of recovery since burning (0-2 years, 3-8 years, and more than 90 years). Burned units were equally divided among three grazing treatments, i.e., a 6 x 3 treatment was selected (Table 3.1). Units, managed to return a dominant herbaceous plant community, were treated in sequence phases – renovation, renovation-maintenance, or maintenance. Under renovation, the target plant community was short, woody shrubs with little or no grass understory. For the first three to four treatments, prescribed fire alone was used. The cycle was usually burn, rest two years, burn, rest two years, burn. Once sufficient herbaceous cover developed, cattle were added in years not burned (renovation-maintenance), e.g., burned, rested one year, grazed three years, rested one year, burned, and the cycle continued. The objectives were to decrease exotic grasses and to increase native herbaceous species, to continue to reduce woody plants, and to improve nutrient cycling. In the grazing phases, each pasture was grazed three years and divided into three units of equal size using portable electric fences. Cattle were used at a rate of 0.5 animal unit per month (AUM)/acre from the first of June through mid August. They were rotated through the three units, staying on each unit for 14 days, then rotated again through the first and second units for another 14 days each, a twice-over. Usually only the first two units received the "twice-over" but if ample moisture occurred in July and early August, the third unit was also grazed again for 14 days. During the second and third year of grazing, the 14-day grazing periods were rotated so each unit was grazed a different 14-day period. Maintenance will allow fire and grazing, but no units have reached the vegetation composition required by the comprehensive conservation plan for the refuge to be in the maintenance phase.

The rolling topography of the refuge has varying habitat types (bottoms, north slopes, hilltops, and south slopes) in each unit. This allowed for comparison of treatment effects across major habitats of the refuge. In addition, the results could be expanded both from south slopes to environmentally similar drier areas to the west (e.g., eastern Montana) and from north slopes to environmentally similar moister areas to the east (e.g., central North Dakota). The sampling design included four microclimates, six burn levels, three post-fire recovery periods, three grazing treatments, and 500 plots stratified by topography (i.e., “sites”). All sites were field examined during 2001 and 2002 and remotely imaged in 2000.

The vegetation was sampled with Space Imaging’s IKONOS multispectral imagery, acquired on August 11, 2000 and October 10, 2000. To capture maximum phenological changes, image acquisition was attempted for May (spring green-up), August (summer growth), and October (autumn senescence), but a clear image for spring was unobtainable due to weather and acquisition scheduling. The imagery has a spatial resolution of 4 m and included four spectral bands (blue, 0.45-0.52 µm; green, 0.52-0.59 µm; red, 0.62-0.68 µm; and near-infrared, 0.77-0.86 µm). Pixel values represented 11-bit scaled radiance values.

A classified image was created using classification tree analysis. For classification, one composite image included two 4-m resolution satellite images (consisting of four wavelength bands each for a total of eight bands), seven principle components analysis layers of the two images, and four difference images (calculating the difference between values of each wavelength between each image). This image was overlaid with a map of the management blocks, provided by Lostwood NWR, to distinguish the burning, burn-graze, and grazing treatment areas.

The resultant vegetation map was sampled to compare the vegetation of four habitats (bottomlands, north slopes, south slopes, and hilltops) in each of 24 units differing in fire and grazing treatments. As digital technology was not available to stratify the vegetation map by habitat type for a complete sample, the sample was coordinated with ground sampling points in a concurrent study. Random points were generated using a 16-m digital elevation model. Space Imaging built the DEM with a grant from NASA to T. Weaver, 1999. Random points were located using GPS technology (e.g., Trimble Pro-X, Garmin eTrex Vista) and slope and aspect verified for each point (personal communication, J. Rubin, 2003). Points were buffered using ERDAS Imagine software to correlate with 6-m radius plots on-site. Vegetation community types were recorded at each site (personal communication, J. Rubin, 2003). Data were collected from 176 sites in 2001 and 324 sites in 2002 on Lostwood NWR. Vegetation for each site was remotely sampled with 4-m pixels within a homogenous known community type and slope position. These areas were used to extract vegetation data from the classified image and correlate vegetation type with known treatment histories. Vegetation types recorded on-site were pooled by groups (e.g., 4 snowberry communities and 3 silverberry communities were combined to represent the shrub type). Groups were defined in the third (and final) level of hierarchical classification (see Chapter 2). Cover (%) of each vegetation type was averaged across each slope position within the same management unit. Using the remote sample, a matrix was constructed containing the following data: management block unit, topographic position, number of burns, years rest since last burn, grazing occurrence, years rest since last grazing rotation, and percent cover of vegetation, grouped as tree, shrub, dwarf shrub, and grass.

Because the glacial topography is heterogeneous, bottoms, north slopes, south slopes, and hilltops were separately sampled for a mesic to xeric continuum. This stratification provided three benefits: it reduced variance and retained response independence, it allowed separate conclusions for different habitats, and it allowed the expansion of results to short grass prairie (xeric), mixed-grass prairie (slopes), and tall grass prairie (mesic).

Data Analysis

Community differences between treatments were compared using quantitative analysis of classified image. All points were stratified by habitat type within management units. The number of pixels classified into each vegetation type at every site was averaged to determine cover (%) of a vegetation type by habitat type within a management unit. Separate statistical models were made for each habitat to ensure the responses were independent. The number of burns for a management unit was used to scale sampled units along a gradient from 0 (no fire in more than 90 years) to 6 (many burns). Recovery time since burning for a management area was used to categorize sampled units as unburned (more than 90 years rest since burning), long recovery (3-8 years rest since burning), and short recovery (0-2 years rest since burning). Grazing treatment for a management unit was used to categorize sampled units as ungrazed (no grazing in 14 or more years), grazed and rested (grazed, then rested 2-3 years), or currently grazed (grazed during the 2000 growing season). Vegetation data from classified images were analyzed using regression analysis to quantify the relationships between treatments and the cover of woody and herbaceous vegetation. The first regression was produced for cover (%) against topography. After stratification by habitat, separate univariate regressions were produced for cover (%) of vegetation against burning, post-burn recovery of vegetation, and grazing categories.

Results

Potential prairie dominance in the Missouri Coteau may be determined by climate. Within this prairie landscape, topography seems to determine the distribution of local potential vegetation. Within topography-determined habitats, the vegetation may be modified finally by fire and grazing.

Effect of Topography

Water becomes increasingly scarce on the gradient from bottoms through north and south slopes to hilltops (Plaggemeyer, 2003). Analysis of satellite imagery showed vegetation also changed along this gradient (Table 3.2, Figure 3.1), probably in response to the decline in water availability. Trees, mostly aspen, were rare in all habitat types (1% - 14%). Shrubs (snowberry and silverberry) were most dominant in bottomlands (36%) and north slopes (44%) and declined upslope to south slopes (16%) and hilltops (20%). Grass/dwarf shrub (prairie rose and young/low snowberry) was least common in bottoms (5%) and became more common toward hilltops (17%). Pure grass vegetation (warm- and cool-season native grasses, cool-season exotic grasses, and native and exotic forbs) was found throughout (33% - 74%, bottom to hilltop). Thus, woody vegetation (tree and shrub) was found less uphill and herbaceous vegetation was found correspondingly more uphill.

Table 3.2. Effect of topographic position on mean cover (%) of vegetation. Averages were calculated across fire and grazing treatments.

	Bottoms	North Slopes	South Slopes	Hilltops
Tree	5	14	1	5
Shrub	36	44	16	20
Grass/Dwarf Shrub	5	9	9	17
Pure Grass	54	33	74	58

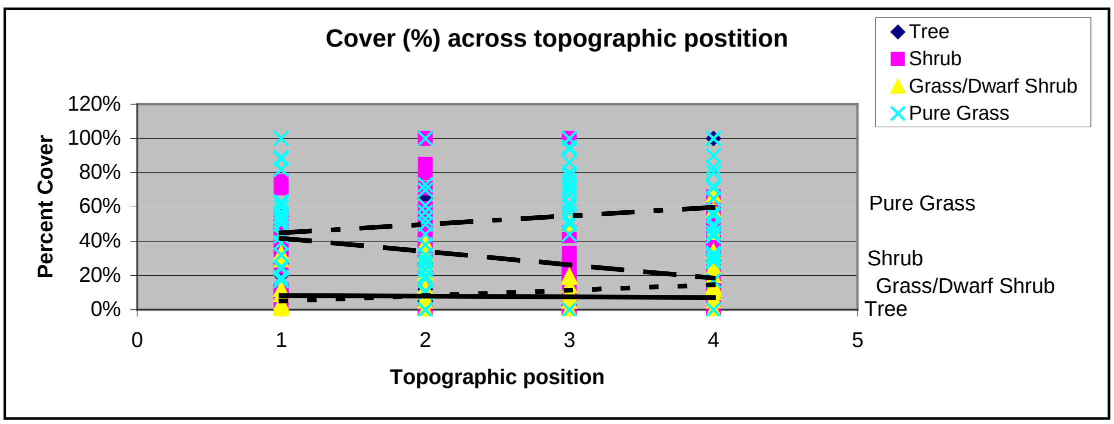

Figure 3.1. Cover (%) of four vegetation types across topography - bottoms (1), north slopes (2), south slopes (3), and hilltops (4).

<u>Effect of Fire</u>

A Poisson regression of cover (%) against number of burns was produced (Table 3.3). Effects of fire were examined within habitats, but comparisons were not made across habitats. With increasing number of burns, tree cover was less on slopes and hilltops (1 – 2% per burn, $p<0.001$), but analysis failed to detect a difference in bottoms. Shrub cover was greater on north slopes and hilltops (1% per burn, $p<0.001$), but analysis failed to detect a difference in bottoms and south slopes. Grass/dwarf shrub cover was less on all habitat types (1 – 2% per burn, $p<0.001$). Pure grass cover was greater on hilltops (1% per burn, $p=0.01$), but analysis failed to detect a difference in any other habitat types. The trends shown in regression are illustrated in graphs of cover (%) against number of burns (Figure 3.2). This analysis showed little, if any, removal of woody plants and little, if any, replacement of woody vegetation by herbaceous vegetation. Any effect of fire on vegetation cover is small and change over time is slow.

Table 3.3. Poisson regression of burning treatment. Variable is continuous with values of 0 to 6 number of burns. Shading indicates values without significant probabilities.

		Bottom	North Slope	South Slope	Hilltop
Tree	y-intercept	1.75, p<0.001	3.27, p<0.001	2.04, p<0.001	2.95, p<0.001
	coefficient	-0.07	-0.2	-0.6	-0.47
	probability	p=0.25	p<0.001	p<0.001	p<0.001
Shrub	y-intercept	3.48, p<0.001	3.50, p<0.001	2.66, p<0.001	2.83, p<0.001
	coefficient	0.03	0.08	0.04	0.2
	probability	p=0.18	p<0.001	p=0.27	p<0.001
Grass/Dwarf Shrub	y-intercept	2.80, p<0.001	2.66, p<0.001	2.72, p<0.001	3.41, p<0.001
	coefficient	-0.4	-0.15	-0.19	-0.18
	probability	p<0.001	p<0.001	p<0.001	p<0.001
Pure Grass	y-intercept	3.89, p<0.001	3.39, p<0.001	4.19, p<0.001	3.91, p<0.001
	coefficient	0.03	0.03	0.03	0.05
	probability	p=0.11	p=0.25	p=0.06	p=0.01

Effect of Rest After Fire

The Poisson regression of cover (%) against time since burning was produced (Table 3.4). Effects of rest after fire were examined within habitats, but comparisons were not made across habitats. Tree cover was greater on south slopes with rest since burning (1% per year, $p<0.001$), but analysis failed to detect a difference in any other habitat types. Shrub cover was greater in bottoms (1% per year, $p=0.01$), but analysis failed to detect a difference in any other habitat types. Grass/dwarf shrub cover was greater in bottoms, south slopes and hilltops (1% per year, p<0.001), but analysis failed to detect a difference on north slopes. Pure grass vegetation was likely out competed and therefore was less in bottoms, south slopes, and hilltops (1% per year, p=0.005-<0.001), but analysis failed to detect a difference on north slopes. The rise in woody species appeared less than complementary to the decrease in herbaceous species. The trends shown in regression are illustrated in graphs of cover (%) against rest since burning (Figure 3.3).

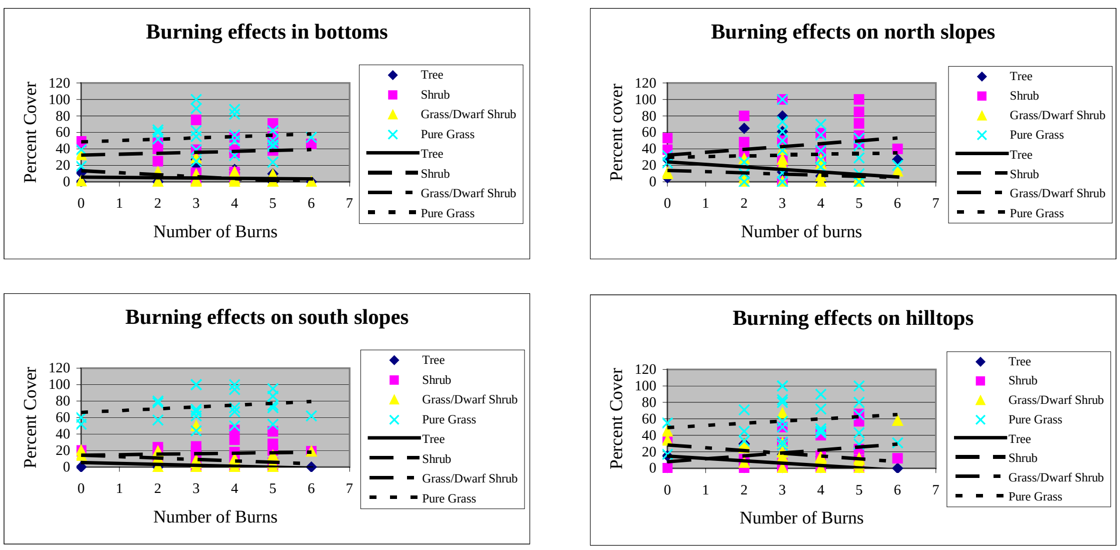

Figure 3.2. Repeated burning on topographic positions. 120 sites were examined for each topographic position across 24 managed units. Each point is the average of sites (5) within a managed unit for each topographic position.

Effect of Grazing

Analysis of grazing effects paralleled the analysis of fire effects. Plots were either grazed on a rotation or left ungrazed. Three grazing treatments were compared with declining recent grazing impact: currently grazed units, units with 2-3 years rest since grazing, and ungrazed units.

Table 3.4 Poisson regression of rest since burning. Variable is continuous with 0 to 8 years of rest after fire and the unburned condition represented as 15 years rest after fire. Shading indicates values without significant probabilities.

		Bottom	North Slope	South Slope	Hilltop
Tree	y-intercept	1.45, p<0.001	2.64, p<0.001	-0.01, p=0.97	1.60, p<0.001
	coefficient	0.004	0	0.02	0
	probability	p=0.26	p=0.94	p<0.001	p=0.52
Shrub	y-intercept	3.55, p<0.001	0, p<0.001	2.77, p<0.001	3.00, p<0.001
	coefficient	0.003	0	0	0
	probability	p=0.01	p=0.99	p=0.37	p=0.49
Grass/Dwarf Shrub	y-intercept	1.32, p<0.001	2.17, p<0.001	2.00, p<0.001	2.66, p<0.001
	coefficient	0.02	0	0.01	0.01
	probability	p<0.001	p=0.48	p<0.001	p<0.001
Pure Grass	y-intercept	4.06, p<0.001	3.50, p<0.001	4.33, p<0.001	4.12, p<0.001
	coefficient	-0.01	-0.001	-0.003	-0.01
	probability	p<0.001	p=0.73	p=0.005	p<0.001

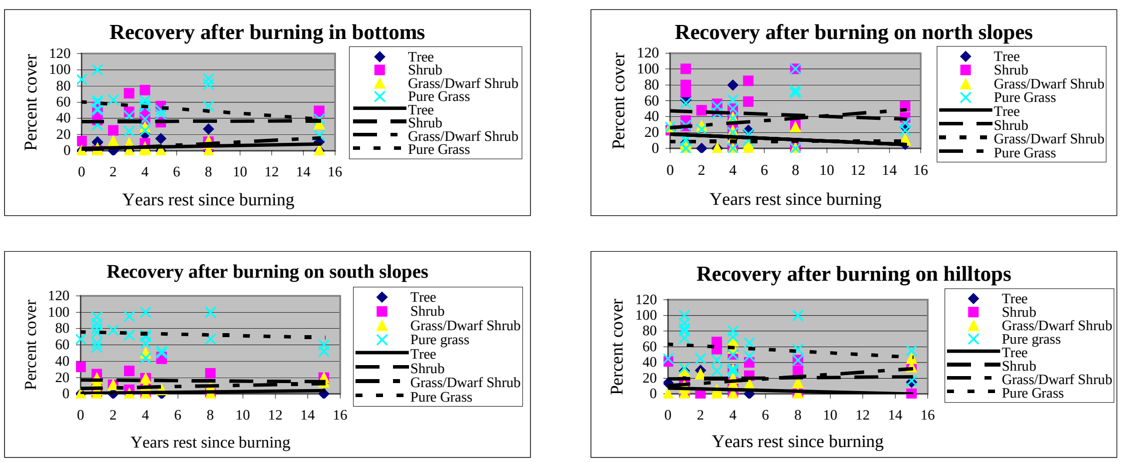

Figure 3.3. Recovery (years rest since burning) on topographic positions. 120 sites were examined for each topographic position across 24 managed units. Each point is the average of sites (5) within a managed unit for each topographic position.

A Poisson regression of cover (%) against three grazing treatments was produced (Table 3.5). Effects of grazing were examined within habitats, but comparisons were not made across habitats. When current grazing is compared to no grazing, tree cover was greater in bottoms (6%, p<0.001) and north slopes (22%, p=0.008), was less on south slopes (2%, p<0.001), and hilltops (1 %, p<0.001). Shrub cover was greater on all habitat types (23-51%, p=0.009 or p<0.001). Grass/dwarf shrub cover was greater in bottoms (2%, p=0.004), north slopes (3%, p<0.001), and south slopes (5%, p<0.001), but analysis failed to detect a difference on hilltops. Pure grass cover was greater in bottoms (43%, p<0.001), north slopes (40%, p<0.001), and hilltops (43%, p<0.001), but analysis failed to detect a difference on south slopes. The trends shown in regression are illustrated in graphs of cover (%) against grazing treatment (Figure 3.4).

Table 3.5. Poisson regression of grazing. Variable is categorical with the ungrazed condition as the first category, represented by the y-intercept. Coefficients and probabilities are given for each grazed condition. Shading indicates without significant probabilities.

		Bottom	North Slope	South Slope	Hilltop
Tree	y-intercept	0.01, p=0.75	2.77, p<0.001	0.75, p<0.001	2.31, p<0.001
	Grazed-rested	1.98, p<0.001	-1.25, p<0.001	0.01, p=0.97	-1.55, p<0.001
	Grazed current	1.80, p<0.001	0.31, p=0.008	-1.59, p=0.01	-2.47, p<0.001
Shrub	y-intercept	3.53, p<0.001	3.67, p<0.01	2.83, p<0.001	2.22, p<0.001
	Grazed-rested	-0.22, p=0.02	0.16, p=0.03	-0.61, p<0.001	0.50, p<0.001
	Grazed current	0.37, p<0.001	0.27, p<0.001	0.30, p=0.009	1.47, p<0.001
Grass/Dwarf Shrub	y-intercept	1.57, p<0.001	2.61, p<0.001	2.41, p<0.001	2.90, p<0.001
	Grazed-rested	0.65, p=0.001	-0.68, p<0.001	-0.33, p=0.049	-0.17, p=0.17
	Grazed current	-0.96, p=0.004	-1.51, p<0.001	-0.74, p<0.001	-0.07, p=0.55
Pure Grass	y-intercept	4.10, p<0.001	3.95, p<0.001	4.25, p<0.001	4.14, p<0.001
	Grazed-rested	-0.08, p=0.26	0.31, p<0.001	0.14, p=0.01	0.07, p=0.23
	Grazed current	-0.35, p<0.001	-0.26, p=0.008	0.02, p=0.69	-0.39, p<0.001

When grazed-rested is compared to ungrazed (Table 3.5), tree cover was greater in bottoms (7%, p<0.001), north slopes, (4%, p<0.001), and hilltops (5%, p<0.001), but analysis failed to detect a difference on south slopes. Shrub cover was greater on all habitat types (9-46%, p=0.02-<0.001). Grass/dwarf shrub cover was greater in bottoms (9%, p=0.001), north slopes (7%, p<0.001), and south slopes (8%, p=0.049), but analysis failed to detect a difference on hilltops. Pure grass cover was greater on north slopes

(71%, p<0.001) and south slopes (81%, p=0.01) but analysis failed to detect a difference in bottoms and hilltops. The trends shown in regression are illustrated in graphs of cover (%) against grazing treatment (Figure 3.4).

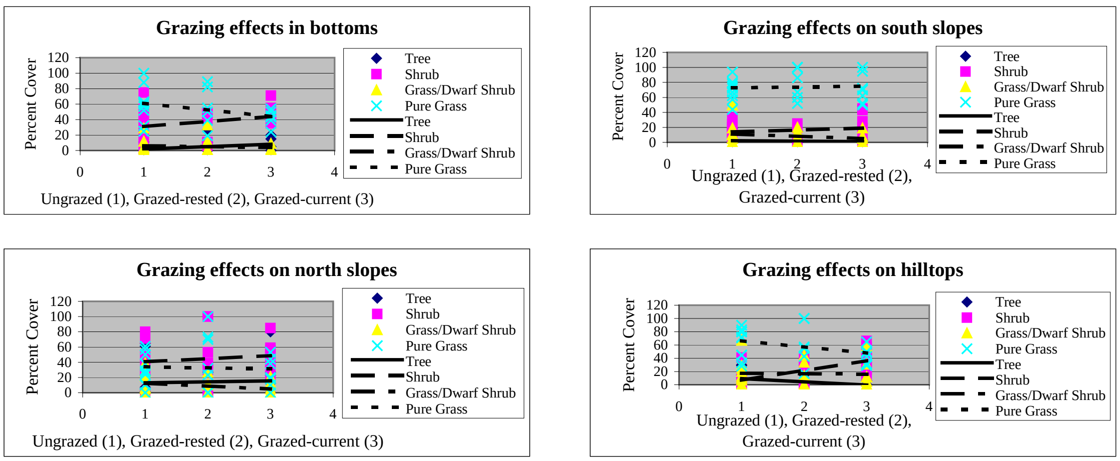

Figure 3.4. Comparison of community cover across grazing treatment, ungrazed (1), grazed, then rested (2) and currently grazed (3). The comparison is made separately for four habitats – bottoms, north slopes, south slopes, and hilltops. 120 sites were examined for each topographic position across 24 managed units. Each point is the average of sites (5) within a managed unit for each topographic position.

Discussion

Managers observe vegetation to determine how it relates to environments in its landscape and how it changes in time, due either to recovery from disturbance (autogenic succession) or to changes in the environment (allogenic succession).

Vegetation and Habitat

Vegetation at Lostwood NWR varied with changes in topography and associated environmental quality. With more available moisture in bottoms around potholes, woody species, mesic grasses, and wet meadow communities dominate. Moving up the slope, trees, shrubs, and lush tall grasses give way to mixed grasses and dwarf shrubs on slopes to short grasses on hilltops. While stratification of vegetation by topography did exist (Table 3.2, Figure 3.1), stratification was not as strong as expected. Coupland (1950) also observed mixed grass prairie communities of the Missouri Coteau and southern Canada change with topography.

Whether observed on-site or sampled from classified images, woody vegetation present today at Lostwood NWR is more than in pre-settlement records. Early explorers saw a rolling grassland without either shrubs or trees (in Coues, 1893; in Coues, 1897; Clandening, 1928). Since variation in the physical environment (i.e., bottoms and ridges with presumed changes in water availability between each) has not changed, some factor eliminating woody plants from moister sites must have vanished. This could be changes in fire and grazing regimes. Fire was reduced by settlement ~1880 to 1930 and then excluded by management until 1978, when current management treatments began. Reduction of grazing may have also contributed to the increase of woody vegetation. With settlement in the late 1800s, bison were removed from the area and replaced by cattle until refuge management limited grazing from 1930 to 1978.

Response of Vegetation to Repeated Burning

The oblique viewer at Lostwood NWR sees reduction of woody vegetation in units repeatedly burned. The horizon of burned units is unmarred by woody stems and vegetation appears to be comprised only of herbaceous mixed-grass prairie communities.

Tree presence in all habitats across the refuge was low (Figure 3.2). Cover (%) of trees was less with repeated burning, but the difference was slight (Table 3.3). Reduction in the shrub type (snowberry and silverberry) with fire was expected, but was not seen in analysis. Fire was also expected to remove shrubs from grass/dwarf shrub vegetation. Grass/dwarf shrub vegetation may have been both damaged mildly by fire on drier sites, and simultaneously encouraged by release from competition with woody vegetation on moister sites. The small change in the extent of this type may result from the integration of these two forces. Fire was expected to encourage pure grass communities. Pure grass was greater on hilltops, but analysis failed to detect a change in any other habitat types. However, subjective evaluation of true color images showed a trend in color and texture with increasing numbers of burns (Figure 3.5). With increasing number of burns, the vegetation generally became lighter in color and less patchy. The apparent smoothing and lightening of color may be due to replacement of shrub savanna with grasses.

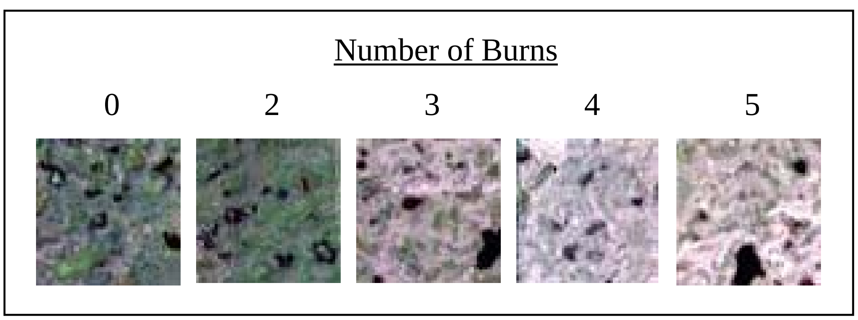

Figure 3.5. Images of prairie burned 0 – 5 times. Selections isolated from satellite image of Lostwood National Wildlife Refuge representing typical units for each number of burns.

Any transformation of vegetation by burning was slow. Slopes seen in plots of vegetation cover against number of burns were low (Figure 3.2, Table 3.3). If the slight change in vegetation per burn slopes (0.5 – 2 % rise or decline per burn) were maintained over time, recovery of the pre-settlement condition would require 20 to 100 years of regular burning.

The slow rate of change might be explained in three ways. 1) Fire cannot eliminate woody plants. 2) Intense burning may be required to initiate conversion. The area lies in a tension zone between nearby (within 100 km) aspen parkland to the north (Bird, 1961) and short to mixed grass prairies to the west and south and tall grass prairies to the east (Küchler, 1964). In a tension zone, stable alternate communities (woody & grassy) may exist. Under these conditions, it may take a large increase in burning pressure to convert woody communities to pre-settlement grassland (Bragg and Hulbert, 1976; Briggs and Gibson, 1992; Harnett and Fay, 1998). Thereafter, less frequent fire may maintain grass communities. 3) Fire effects may be slight at first, but may compound as repetition of burning increases. Alternatively, if increased burning reduced woody presence exponentially, as negative compound interest in finance would, the rate of future decline may increase with number of burns and bring about a more rapid decline in woody species than predicted. The latter seems likely because, while the canopy of trees and shrubs may recover in 1 or 2 years (as seen on-site, in subjective analysis of imagery, and in classified images), vegetative reproduction depends upon large root reserves, which may not be replenished between frequent burns.

<u>Recovery of Vegetation After Fire</u>

While woody species invasion was expected after long periods without fire, sampling of classified images did not support this expectation. Analysis (Table 3.4, Figure 3.3) failed to detect recovery of woody vegetation with rest after fire.

In contrast, subjective evaluation of true color images for recovery after burning (years an area was allowed to rest since the previous burn) showed a difference in the color (Figure 3.6) as number of burns also showed. Units with a short recovery (0 to 2 years) were greener than units with longer recovery (3 to 8 or more than 90 years). This greenness could result from either an increase green leaf area (stimulated by fire) or by reduced masking by woody stems and litter removed by recent fire. Conversely, the longest unburned units (more than 90 years) were greener than units with moderate recovery (3 to 8 years) and appeared similar to some of the recently burned units. This suggests the greening was not due to masking and therefore due to relatively high amounts of green growth.

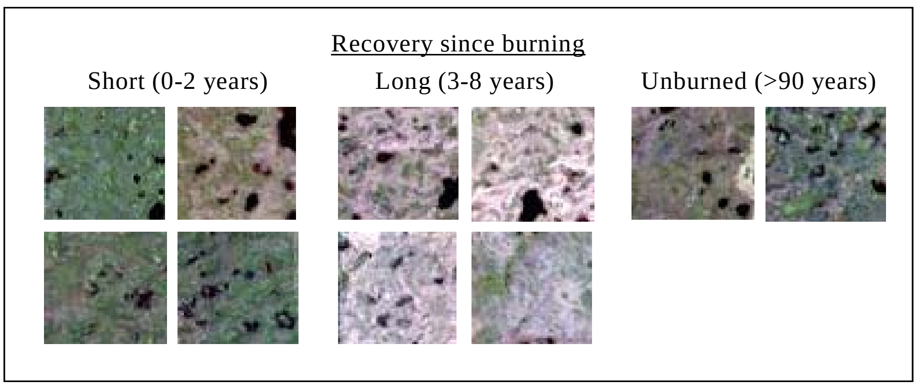

Figure 3.6. Recovery of vegetation after burning on sites burned 0-2, 3-8, or more than 90 years ago. Selections isolated from satellite image of Lostwood National Wildlife Refuge representing typical units for each number of burns.

With infrequent fires, therefore, woody plants tend to persist possibly at the expense of herbaceous vegetation. While aspen may be contained in extent by fire, stimulation of leafy growth following burning maintained aspen clones. Under a regime with three or more years of recovery between burns, snowberry retains its hold on hospitable slopes. Silverberry may respond similarly to aspen or snowberry, since it is also a rhizomatous shrub. After burning, grasses tended to decline as woody vegetation increased. Grasses may have been overtopped and gradually replaced by invading woody vegetation.

Vegetation and Grazing

The management experiment was less well suited for examination of grazing than fire effects. Many combinations of burning and grazing treatments were unavailable. No units with 1 or 2 burns were also grazed and no units were unburned and ungrazed. The apparent rise in cover of woody species with grazing probably indicated removal of standing crop (current growth) of grass and exposure of underlying shrubs to the sensor, but due to the short time available for woody growth, not a change in amount of woody cover. Subjective evaluation of true color images for grazing treatments showed a difference in the color of true color images (Figure 3.7). Ungrazed units were darkest, grazed, rested units were lighter, and currently grazed units were lightest. The color change was likely due to removal of standing crop of grasses.

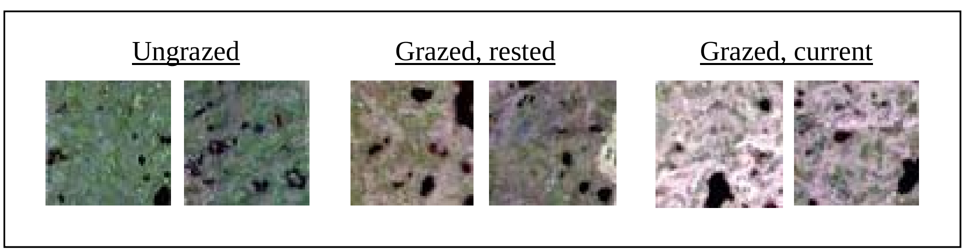

Figure 3.7. Images of prairie currently grazed, grazed then rested and ungrazed. Selections isolated from satellite image of Lostwood National Wildlife Refuge representing typical units for each number of burns.

In the longer term, grass might replace a thinning woody canopy browsed or trampled by herbivores. At other grazed sites, grass communities did replace woody plants under ungulate pressure, for example, aspen expansion increased with decline of ungulates, then aspen cover decreased with ungulate population increase in aspen parkland ecosystem in Elk Island National Park, Alberta (Blithe and Hudson, 1987).

Limitations

Lostwood's management plan purposely confounded repeated burning with grazing (Table 3.l). Most repeatedly burned units (12 of 18) were also grazed. Of these, half (7 of 12) were grazed at the time of imaging. Only one unit had been burned since the last grazing activity. Collinearity of the study probably inhibited the ability of the analysis to detect change.

The spatial accuracy of the imagery may have mislocated some sites. The spatial accuracy of multi-spectral IKONOS imagery was 6-m. Due to the field sampling design, some sites (all of which were less than 6-m in radius) were close to unit boundaries and were located on the image in a unit with a different treatment history. Cover (%) averages of vegetation within management units were altered.

Some geographical variation may have existed where glaciation variously affected the area. The terminal moraine of the Wisconsin Glacier moved down from the northwest corner into what is now Lostwood NWR leaving scattered depressions and ridges from scouring and deposition. The larger lakes and sloughs across the east and south of the study area are what remain of the drainage system of the receding edge of the glacier. This variation in topography may affect presence of vegetation and effectiveness of treatments.

Results from a concurrent ground-based study using the same points showed less shrub cover on all slopes (personal communication, J. Rubin, 2002) than shrub cover measured from imagery. In bottoms, only one third (30%) of the shrub-classified pixels were actually woody species (Table 3.6). The remaining pixels were grassy, i.e., mesic and wet meadow grasses, exotic grasses, and forbs. On north slopes, only half (56%) of the shrub-classified pixels were actually dominated by woody species. The remaining pixels were predominantly exotic grasses. On south slopes and hilltops, one third (28% and 29%) of the shrub-classified pixels were actually woody species. The remaining pixels were predominantly upland and exotic grasses. Shrub presence was overemphasized across the refuge and in all treatments. If occurrence of misclassification was more frequent within any treatment component (e.g., current grazing), the misclassification may have inhibited the ability of the analysis to detect change.

Table 3.6. Actual vegetation cover of pixels classified as shrub. Classified pixels were counted for woody versus herbaceous vegetation.

	Bottoms	North Slopes	South Slopes	Hilltops
Woody vegetation	30%	57%	29%	30%
Herbaceous vegetation	70%	43%	71%	70%
Total number of pixels	188	177	91	117

Conclusions and Management Implications

Fire will immediately make woody vegetation less visible. Analysis at Lostwood

NWR suggests, however, cover (%) of woody vegetation will be slowly reduced over the long term, i.e., 1 to 5 % per burn. Over the longer term the rate of this decline may increase exponentially due to gradual exhaustion of woody plant root reserves.

The management experiment has yielded some information on fire effects, less on grazing effects, and still less on joint effects. Because a scientific experiment designed to investigate fire and grazing effects more rigorously will last for decades or a century, fire and grazing effects should be investigated by continued application of likely treatments with periodic review of progress and revision of treatment as necessary

Satellite imagery may provide the best tool for resampling needed in this and other systems. Sampling with remote sensing may be more expedient than ground surveying, but classifications must have higher accuracies and less bias to be useful to managers. Therefore, satellite imagery needs further development to support on-going adaptive resource management (Allard, 2003).

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