Landscape-Scale Assessment of Grand Staircase-Escalante National Monument

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Natural Resource Ecology Laborator

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Introduction

The Grand Staircase-Escalante National Monument was established on September 18, 1996 by a Presidential Proclamation under the 1906 Antiquities Act. The Proclamation identifies the Monument's important ecological values, and specifically recognizes the "extraordinary opportunity to study plant speciation and community dynamics" within its boundaries. The Monument was mandated under this proclamation to protect both its natural and cultural resources while providing for diverse land uses such as recreation and grazing (Grand Staircase-Escalante National Monument 2000). To best protect the Monument's biological resources, it is important to know which resources currently exist and which management practices have been successful at protecting these resources and restoring heavily disturbed ecosystems.

In 1997, the Natural Resource Ecology Laboratory was asked to fulfill part of this mandate by providing a landscape-scale assessment of native and non-native plant diversity, soil characteristics, and cryptobiotic soil crusts within the Monument (Stohlgren et al. 1999a). Between 1998 and 2003, field sampling was conducted to create an extensive database that was used in various analyses. The case studies within this technical report are products of these efforts and provide information that fulfill the objectives initially set forth by the project (Stohlgren et al. 1999a).

Purpose of Our Research

Our primary objectives were to work closely with Monument staff to produce: (1) detailed baseline data on native and non-native plant species, cryptobiotic crust communities, rare/unique habitats, and soil characteristics; (2) geographic information system-based spatial analyses of the patterns of plant diversity, hot spots of diversity, and rare/unique habitats; and (3) the establishment of long-term study plots to monitor and evaluate the status and trends of botanical resources over time. To meet these objectives, we:

1.) Identified Hot Spots of Native Plant Diversity and Rare/Unique Habitats

We identified critical habitats by conducting a careful analysis of highly diverse and rare vegetation types within the Monument. Examples of critical habitats include those associated with rare geologic features and distinct vegetation communities (e.g., riparian areas, wet meadows). In the Monument, we hypothesized that riparian zones and small wetlands would be hot spots of biodiversity because of abundant resources not typically found in arid ecosystems. This hypothesis was supported by our findings.

2.) Determined Areas Where Cryptobiotic Crusts and Plant Vegetation Types Are Particularly Sensitive to Disturbance

Development, recreation, agriculture, and livestock grazing have all been suggested as reasons for declines in cryptobiotic soil crusts and native plant species within the western United States. Both cryptobiotic crusts and native plant species are more prevalent in some areas of the Monument than in others, while some areas also receive more disturbance It is likely that some types of crusts or specific habitats may be more resistant or resilient to disturbance than others. Our systematic surveys identified the distribution of cryptobiotic crusts, sensitive species, and sensitive habitats within the Monument and looked at how these areas might be affected by disturbance.

3.) Detected the Loss of Native Plant Diversity Caused by Non-Native Plant Species

Recent studies have shown that non-native plant species are invading hot spots of native plant diversity such as tallgrass prairie, wet meadows and aspen vegetation types in the Rockies and riparian zones throughout the west (Stohlgren et al. 1998a; Stohlgren et al. 1999b; Stohlgren et al. 2001). Some non-native species are toxic to livestock and wildlife, but their patchy distribution makes them difficult to detect and control. Others are spreading readily and have the potential to become dominant over time. Preserving native plant diversity will become more difficult as non-native plants continue to invade. Only by knowing the locations of these non-native invaders and the structure of the landscape-scale vegetation and soils matrix can predictions be made about invasions that can guide successful management actions. Summary data, distribution maps, and spatial predictive models for non-native species are provided in this report.

4.) Established Long-Term Study Plots to Monitor and Evaluate Vegetation and Soil Resources

It is critical to establish a land use history that identifies and locates past and current land use practices and quantifies impacts to vegetation and soils to better plan future developments and land use practices to minimize these impacts. Locating suitable "control" sites with minimal anthropogenic disturbance is often difficult, but not impossible. Plot selection for this project extended across 19 vegetation types including both disturbed and undisturbed habitats. By having these spatially referenced plots, temporal data can be recorded that provides a way to study processes of succession in relation to disturbance, climate, and specific management practices over time.

The database created by these sampling efforts is extensive and data analysis is on-going. Since the project was initiated, it has produced eight published papers, four papers are currently in review, two papers are in progress, and 27 presentations related to the project have been given. In addition, we have developed 19 spatial predictive models, an herbarium that includes 692 mounted plants, a project website, and a GIS databse (See Appendix A). The following chapters are a summary and review of the project's findings and how these findings will apply to the Monument's management strategies in the future.

CHAPTER 1

Methodology

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The following methods were used throughout the entire sampling period between 1998 and 2003. However, some results discussed in subsequent chapters may have required additional methods than those included here. In addition, some analyses were completed prior to the end of the sampling period and only included subsets of the entire dataset. Such changes will be noted throughout this report.

The Modified-Whittaker Plot

The multi-scale modified-Whittaker plot was the basis for all vegetation sampling. The plot measures 20 m by 50 m (1,000 m^2) and contains nested subplots of three different sizes. A 5 m by 20 m (100-m^2) subplot is placed in the plot's center, and two 2 m by 5 m (10-m^2) subplots are placed in opposite corners of the plot. There are a total of ten 0.5 m by 2 m (1-m^2) subplots. Six are arranged systematically inside and adjacent to the 1,000-m^2 plot perimeter, and four are arranged systematically outside and adjacent to the 100-m^2 subplot perimeter (Fig. 1-1). Plots were placed parallel to the major environmental gradient of the vegetation type being sampled to encompass the most heterogeneity (See Appendix B).

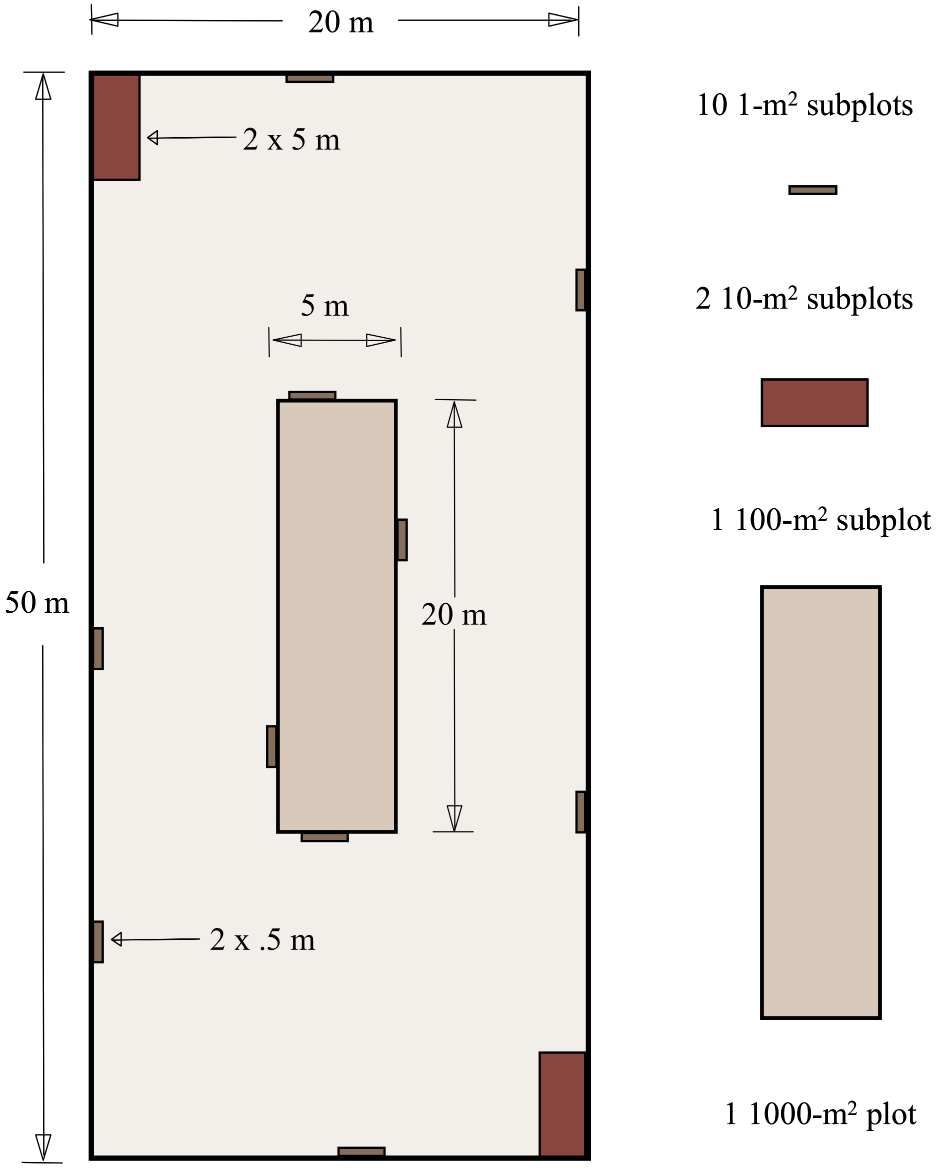

Figure 1-1. The modified-Whittaker multi-scale vegetation sampling plot.

This multi-scale plot design has several benefits. Commonly used vegetation sampling techniques such as transect and quadrat methods (Parker 1951; Daubenmire 1959) were found to significantly underestimate total species richness, the number of native and non-native species, and the number of rare species (i.e., <1% cover) in grassland vegetation types (Stohlgren et al. 1998a). This is due to small sample areas, spatial autocorrelation bias, and the tendency to miss rare species and habitats in replicated transect methods compared to a multi-scale sampling design. The modified-Whittaker plot captures twice as many plant species than replicated transect methods and is better at monitoring the spread of non-native species and evaluating range conditions and trends at local, regional, and national scales (Stohlgren et al. 1998a). In addition, species richness can also be used to construct species-area curves from the nested subplot design to estimate larger-scale richness patterns (Shmida 1984; Palmer 1990, 1991).

The shape and layout of the modified-Whittaker plot are also beneficial. Rectangular plots placed parallel to the major environmental gradient of a vegetation type encompass more heterogeneity and recover greater species richness than round or square quadrats (Bormann 1953; Stohlgren 1994). Because this holds true at all scales, this shape is kept consistent for the plot and its nested subplots (Stohlgren 1994). The systematic placement of the subplots also makes the design easy to use in the field and to use as long-term study plots because of easy relocation. The three different subplot sizes are also independent and non-overlapping, which reduces spatial autocorrelation found in other replicate transect methods (Pielou 1977).

Site Selection

During each year of the study, we focused on sampling different regions of the Monument to simplify travel logistics (Fig. 1-2). In 1998, a 29,000 ha area in the southeast corner of the Monument was selected for intensive study. We used a simple random study design with unbiased site selection to locate 50 plots. Over the course of this year, some plot names were changed to simplify analysis (See Appendix C). The majority of these plots occurred in the three most common vegetation types: blackbrush (*Coleogyne ramosissima*), sagebrush (*Artemisia tridentata*), and pinyon-juniper (*Pinus edulis, Juniperus osteosperma*). We used the locations of the 50 plots to subjectively locate 47 additional plots in close proximity that had a contrasting vegetation type or topographic position (i.e., wash versus upland). Three of the original 50 plots did not have significantly different areas nearby. These 47 new plots were chosen in this manner to reduce travel cost and to increase the sample size of habitats that are typically missed with a random sampling design because they cover a small area (e.g., drainage features in an arid environment). Unless otherwise noted for specific sub-studies, we continued to locate our plots using this method.

In 1999, plots were located in the northeast and east-central portion of the Monument as described for the 1998 field season. In 2000, plots were randomly located within stratified areas of a combination of burned, seeded, chained, and control areas of Buckskin Mountain and Skutumpah Road study areas (a few plots were also completed in the Buckskin Mountain area in 1999). In these cases, treatment plots were paired with nearby undisturbed control plots. In 2001, plots were located randomly in stratified areas of the Monument that were underrepresented in our study. A study of near-relict sites compared to grazed sites also included paired treatment plots on No Man's Mesa, Deer Spring Point, Spring Point, and Timber Mountain. In 2002, we sampled again in areas underrepresented, and in 2003 a few multi-scale plots were placed along Hackberry Canyon and in the remaining underrepresented areas.

Because plant species characteristics (e.g., biomass, establishment, reproduction) are correlated with various climatic factors, especially water in arid environments, an analysis of variance (ANOVA) was performed ($p < 0.05$) to compare climatic data across five years that the vegetation was sampled. Although complete climate records from the Monument were not available for the sampling period, climate data were available for neighboring weather stations (National Climatic Data Center 2003; Table 1-1; See Appendix D). Due to the high variability among these data, few statistically significant differences were seen across years.

Table 1-1. Climate data for towns near the Monument during the study period 1998-2002 (National Cimatic Data Center 2003). An ANOVA was performed across years. Letters indicate a significant difference within a station location for each variable.

Station Location	Year	Mean Monthly Temperature (^oC)	Precipitation (mm)	Snow (mm)
Bryce Canyon National Park Headquarters, UT	1998	4.4 a	49.6 a	228.2 a
	1999	5.5 a	24.9 a,b	95.0 a
	2000	5.3 a	42.6 a,b	196.5 a
	2001	5.7 a	27.8 a,b	227.5 a
	2002	5.6 a	17.7 b	79.0 a
Escalante, UT	1998	10.7 a	26.0 a	42.3 a
	1999	11.3 a	23.1 a	44.0 a
	2000	12.1 a	19.4 a	23.3 a
	2001	12.7 a	22.3 a	71.6 a
	2002	11.5 a	10.1 a	4.4 a
Kanab, UT	1998	11.6 a	51.2 a	92.3 a
	1999	12.7 a	25.7 a	36.8 a
	2000	13.3 a	31.6 a	45.5 a
	2001	12.8 a	31.2 a	157.7 a
	2002	12.6 a	21.8 a	40.0 a
Panguitch, UT	1998	7.9 a	25.5 a	27.5 a
	1999	14.8 a	10.1 a	0.0 b
	2000	15.6 a	12.2 a	2.1 a,b
	2001	15.3 a	12.0 a	13.3 a,b
	2002	8.2 a	12.3 a	4.9 a,b
Page, AZ	1998	14.2 a	19.6 a	7.6 a
	1999	14.8 a	10.1 a	0.0 a
	2000	15.6 a	12.2 a	2.1 a
	2001	15.3 a	12.0 a	13.3 a
	2002	15.3 a	8.5 a	5.3 a

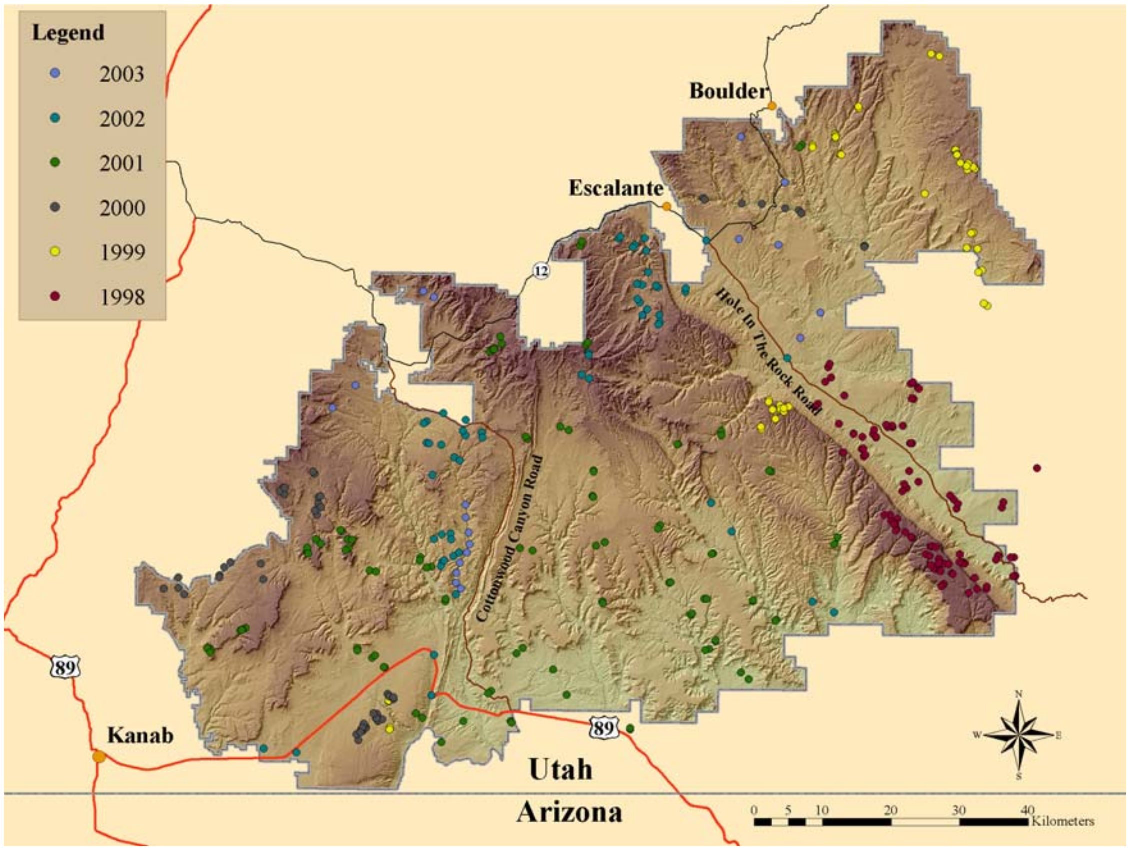

Figure 1-2. Location of all 386 plots classified by the year they were sampled.

Three hundred and eighty-six plots were located within the Monument, including 379 modified-Whittaker plots and seven smaller 100-m^2 multi-scale Intensive plots (Barnett et al. 2003). The smaller multi-scale plots were used at sites such as Hackberry Canyon, where the modifed-Whittaker plot was too large to place within the study area (See Appendix E). All the plots encompassed the range of moist/dry and disturbed/undisturbed habitats. Nineteen vegetation types were identified by dominant species, including desert shrub, blackbrush, desert shrub/grassland, sagebrush, juniper, juniper/sage, disturbed pinyon-juniper/sage, pinyon-juniper/sage, pinyon-juniper, pinyon-juniper/oak, pinyon-juniper/manzanita, pinyon pine, mountain shrub, ponderosa pine/manzanita, rabbitbrush, aspen, wet meadow, spring, and perennial riparian (Table 1-2; See Appendix F).

Table 1-2. Number of modified-Whittaker plots located within each vegetation type.

Vegetation Type	Sample Size
Desert Shrub	29
Blackbrush	27
Desert Shrub/Grassland	16
Sagebrush	31
Juniper	22
Juniper/Sage	17
Disturbed Pinyon-Juniper/Sage	35
Pinyon-Juniper/Sage	21
Pinyon-Juniper	81
Pinyon-Juniper/Oak	18
Pinyon-Juniper/Manzanita	6
Pinyon Pine	4
Mountain Shrub	11
Ponderosa Pine/Manzanita	7
Rabbitbrush	9
Aspen	6
Wet Meadow	3
Spring	2
Perennial Riparian	22

Field Sampling

At each ground truth sampling point, a modified-Whittaker nested vegetation sampling plot was established (Stohlgren et al. 1995). At the initial corner of the plot (0,0), a rebar stake with a benchmark was hammered into the ground (See Appendix B). Each benchmark had the plot name engraved on it and was often surrounded by a rock cairn to assure relocation. In the ten 1-m^2 subplots, the absolute foliar cover (%) and height (cm) by species were recorded. Species that occupied <1% cover in a subplot were recorded as 0.5%. The cover (%) of bare ground, rock, litter (i.e., detached dead plant material), duff (i.e., attached dead plant material), water, dung, wood (woody material > 0.5 cm in diameter), and cryptobiotic crusts were also recorded (See Simultaneous Surveys of Crusts and Selected Soil Characteristics). Cumulative plant species (i.e., additional species found in the subplot or plot) were recorded successively in the ten 1-m^2 subplots, the two 10-m^2 subplots, the 100-m^2 subplot, and the remaining unsampled area of the 1,000-m^2 plot.

Plants were sampled during peak biomass of most species. Plant species that could not be identified in the field were collected and later identified off-site by botanists at Brigham Young University, Utah (Drs. Stanley Welsh and Duane Atwood), Southern Utah State University (Dr. Jim Bowns), or at the herbaria at Colorado State University, Fort Collins or the University of Wyoming, Laramie. About 10% of the total specimens collected could not be identified to species due to inappropriate phenological stage or missing flower parts and were not used in analyses. Species were further classified by origin as native or non-native according to the Natural Resource Conservation Service PLANTS database (USDA 2003) and the Utah Flora (Welsh 1993).

Simultaneous Surveys of Crusts and Selected Soil Characteristics

Field Sampling Methodology

Within each 1-m^2 subplot, cryptobiotic crust cover was recorded by level of development. Determination of crust development was based on the National Park Service (NPS) Soil Crusts Condition Assessment Index, used primarily to monitor disturbance impacts (U.S. Department of the Interior 1995). This index uses a 10-point scale to classify crust structure, from 0 (i.e., bare ground with no crusts present) to 10 (i.e., well-developed crusts). With this index, the developmental stage of cryptobiotic crusts was recorded in eight classes from 1 (i.e., weakly developed) to 20 (i.e., fully developed; Belnap 1995, 1996; Table 1-3).

Table 1-3. Classification index of cryptobiotic crusts by development stages.

Class	Measurements and Descriptions
0	Bare soil, no crusts
1	Thin, flat crusts < 1 cm
2	Pedicels of 1 cm
4	Pedicels of 2 cm
6	Pedicels > 2 cm
10	Pedicels > 2 cm, formation of dark patches
15	Pedicels > 2 cm, formation of mosses and lichens
20	Pedicels > 3 cm, fully developed mosses and lichens

Cryptobiotic crusts were also collected randomly throughout the plot based on three levels: low (classes 1 and 2), medium (classes 4, 6, and 10), and high (classes 15 and 20). The top 1-2 cm of crust (in the quantity of 1/2-1 cup) were carefully removed, placed in a sealed plastic bag, and sent immediately to the USGS soils lab in Moab, Utah for chlorophyll a and other pigment analysis.

During field sampling, five soil samples were taken in the corners and the center of each modified-Whittaker plot with a 2.5 cm diameter soil increment core to depths of 15 cm and pooled into one composite sample. For each sample, surface litter (if present) was removed. If, due to rocks or other site characteristics, the core was unable to reach an approximate depth of 15 cm, two cores at depths averaging 7 cm were taken to ensure adequate volumes of soil for laboratory procedures. All soil samples were air-dried for at least 48 h. A substudy was also conducted that compared soil samples taken at two different depths and can be found in the Appendix (See Appendix G).

Laboratory Methodology

After conducting soil sampling in the field, samples were brought back to the laboratory for analysis. Soil analyses included three steps: preparation, texture analysis, and chemical analysis.

Preparation

Preparation involved sieving each sample and grinding a subset of each sieved sample. By sieving, all rock and biotic debris was removed, and a maximum particle size was attained using a standard #10 sieve (i.e., 2 mm pore size). Using a three-ball grinder, two small subsamples of each sieved sample were pulverized. These ground subsamples were later used in chemical analyses.

Texture Analysis

For texture analysis, a portion of each sieved soil sample was analyzed using the Bouyoucos hydrometer method to determine the percent sand, silt, and clay content (Table 1-4). Forty grams of soil from each sample were weighed into 250 mL square plastic bottles and 100 mL of 5% sodium-hexametaphosphate (HMP) dispersing agent was added to each. The samples were then loaded into a reciprocal shaker and shaken for 16 hr (i.e., overnight). To correct for oven-dry weight of soil, an additional 10.0 g of soil from each sieved soil sample were weighed into a pre-weighed tin and then allowed to dry overnight at 105°C. The following day, the weight of the oven-dried samples was recorded.

Texture tests were performed by pouring the solution into a 1000 mL sedimentation cylinder. The solution was brought up to the 1000 mL mark using room temperature distilled water. A control cylinder ("blank") was also filled with 100 mL of HMP solution and brought to the 1000 mL mark using room temperature distilled water. Each cylinder was mixed vigorously for 30 s using a plunger to ensure that any sediment was dislodged from the bottom and evenly mixed in the cylinder. After agitating the solution, a hydrometer (ASTM no. 152H, with Bouyoucos scale in g/L) reading was taken for each sample at 40 s and 2 h. The temperature of the control was recorded before the solution was agitated and after the final hydrometer reading was taken.

Table 1-4. Texture equations for deriving the percent sand, silt, and clay for each soil sample.

Equations
% water by weight = Pw
Pw = [(pan + wet) - (pan + dry)] / (pan + dry - pan)
Oven dried soil weight = (air dried soil) / (1 + Pw)
% Sand = ([dry soil weight)-(40 s hydrometer reading - blank)] / (dry soil weight)) * 100
% Clay = ((2 hr hydrometer reading - blank) / (dry soil weight)) * 100
% Silt = 100 - (% sand + % clay)

Chemical Analysis

For chemical analyses, ground subsamples were oven-dried at 55°C for 24 h. Samples were analyzed for percentage of total carbon and nitrogen using a LECO-1000 CHN analyzer (Carter 1993), and inorganic carbon from carbonates was determined using a volumetric method (Wagner et al. 1998). Organic carbon was then calculated using the difference between total and inorganic carbon. Soil phosphorus was determined colorimetrically from a sodium bicarbonate extraction (Kou 1996).

Samples were analyzed for the micronutrients potassium (K), magnesium (Mg), calcium (Ca), and sodium (Na) using 25 mL of ammonium acetate added to 5 g of soil and buffered to pH 8.5. The soil and solution were shaken for 5 min, centrifuged, and the supernatant was decanted off. This process was repeated three times for a total of 100 mL of extract (Sumner and Miller 1996). Cations were then determined from the filtered extract by inductively coupled plasma emission spectrometry. Using this method, the presence and concentration of each cation can be determined by measuring the intensities of wavelengths emitted by exciting the cation's electrons and returning them to their ground state.

Ancillary Data

At the plot scale, quantitative measurements in conjunction with ancillary data provide a greater descriptive power to each sample site. Ancillary data recorded for each plot included slope, aspect, and UTM location from a GPS. Elevation was later determined using the UTM's digital elevation model (DEM) or a topographic map. The azimuth of each plot's layout and the disturbance characteristics of each site were noted for each study area. Photographs were also taken for each plot to be used in the GIS database.

As requested by Monument managers, a GIS database ("pick and click") was created that showed the entire network of plots from the project. Each plot was linked to a comprehensive list of relevant data, including information about soils and cryptobiotic crusts, a vegetative species composition list and individual species characteristics. The database and all relevant documentation are exclusive of this technical report.

CHAPTER 2

Native Plant Diversity

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Utah and the Colorado Plateau

Utah is known for its complex geology, consisting of canyons, cliffs, plateaus, and mountains. Among these many geologic features exist numerous habitats, ranging from desert to alpine vegetation types. It is these characteristics that give Utah a rich native flora, consisting of many rare and endemic species (Utah Division of Wildlife Resources 1998). Of the 2,602 species and 393 subspecies or varieties of native vascular plants known to Utah, 247 (8.2%) are state endemics (Welsh 1993). Thus, Utah has one of the highest rates of endemism in the United States (McMahan 1987; Shultz 1993; Davidson et al. 1996).

Five major ecoregions are found throughout Utah, including the Great Basin, Colorado Plateau, Utah High Plains, Wasatch and Uinta Mountains, and the Uinta Basin (Bailey 1980). Of these ecoregions, the Colorado Plateau has the most endemic and rare plant species (Cronquist et al. 1972; Welsh et al. 1975; Welsh 1978, 1979; Shultz 1993). The Utah portion of the Colorado Plateau contains 85% of the Plateau's total diversity and 50% of the state's endemics (Shultz 1997). Considering that Utah has one of the highest rates of endemism in the country, the plant species diversity of this ecoregion is truly unique.

The high species diversity within Utah, and specifically within the Colorado Plateau, has been attributed to several factors, including climate, geology (Welsh 1978) and soil properties (Welsh 1979; Shultz 1993). The complex interactions of these factors create an abundance of habitats throughout the state and the ecoregion that are capable of sustaining a wide diversity of flora. Approximately 65% of all rare flora in Utah can be found in pinyon-juniper, desert shrub, warm desert shrub, or salt desert shrub habitats (Welsh 1979). All these vegetation types dominate the Colorado Plateau (Utah Division of Wildlife Resources 1998) and can be found throughout the Monument.

Plant Diversity within the Monument

The Monument, located within Utah's Colorado Plateau, is unique in regards to its floral diversity. There have been more than 940 plant species recorded within the Monument's boundaries of which 174 (19%) are local or regional endemics (Fertig et al. 2002). In addition, approximately 43 (45%) plant species are considered globally rare (ranked G1-G3 by the Nature Conservancy) and approximately 83 plant species are considered rare within the state (ranked S1-S3 by the Nature Conservancy; Fertig et al. 2002). The Monument also contains 50% of the Colorado Plateau's floral diversity and approximately 30% of the Utah flora (Shultz 1999). Eleven plant species are found nowhere else in the world.

Plant diversity within the Monument is high for numerous reasons. Four major floras exist and overlap within the Monument, including those of Arizona, the Great Basin, the Mojave Desert, and the Great Plains (Belnap 1999). The area also escaped glaciation during the last Ice Age, creating relict plant communities within the Monument's boundaries (Belnap 1999; Grand Staircase-Escalante National Monument 2000). Variations in topography and temperature have helped create specialized habitats such as hanging gardens (Fowler et al. 1995; May 1999), dunal pockets, tinajas, and year-round springs (Belnap 1999).

Some vegetation types exist that have been isolated from grazing and other human influences (Belnap 1999). The level of isolation provided by this remoteness and the stressful environment created by the physical and chemical properties of the region have increased the degree of speciation. With large expanses of unsuitable habitat serving as barriers for dispersal to restricted plant species, immigration occurs frequently yields new species (Belnap 1999).

Although some vegetation types in the Monument are isolated and have escaped disturbance, numerous threats to native diversity still exist. Grazing, fire suppression, road construction, mining, urbanization, and water development have all been suggested as reasons for sensitive plant species declines. These and other land use practices have also contributed to the establishment of many non-native plant species. As these non-native populations proliferate, their threat to these rare habitats and species will become a greater management concern (Monsen 1994; Rosentreter 1994).

Species Conservation Strategies

Various strategies have been developed to determine an appropriate measure for conserving species richness. These include preserving small-area hotspots of endemism and diversity (leaving more land for consumptive uses; Myers et al. 2000; Hobohm 2003), and concentrating on rare species patterns (Prendergast et al. 1993; Dobson et al. 1997), highly threatened ecosystems (Sisk et al. 1994), and even biodiversity coldspots (Kareiva and Marvier 2003). While the best strategy remains debated, few attempts have been made to accurately quantify large-scale patterns of diversity (Williams et al. 1996; Kier and Barthlott 2001). Finding local, regional, and global patterns could help benefit land managers by concentrating efforts in only a portion of available habitats. However, managers must exercise caution because protecting partial areas of a landscape could have negative consequences by potentially ignoring the contribution of species-poor or moderately rich areas to species preservation. Research on finding these patterns will help determine the extent of the landscape that needs to be preserved to maintain overall plant diversity within the Monument and elsewhere.

Case Studies

Case Study 1: *Patterns of Plant Species Richness, Rarity, Endemism, and Uniqueness*

The objectives of this case study were to document plant species rarity, patterns of high and low species richness and endemism, and to identify areas of unique species assemblages in the Monument (Stohlgren et al. 2004 *in review*).

Methods

This study included 367 plots established in 19 different vegetation types during the field seasons of 1998 through 2001 (Fig. 2-1). Prior to field sampling, we developed a simple moisture index by ranking the vegetation types from the most xeric (desert shrub = 1, blackbrush = 2) to the most mesic types (spring = 18, perennial riparian = 19; Table 2-1). This moisture gradient is somewhat subjective considering the inability to truly separate some of these vegetation types via available moisture using quantitative measures. However, the moisture index provides a way to see general trends in the data and a means to compare xeric and mesic vegetation types.

Uniqueness values were assigned to each plot by calculating the relative frequency for each plant species expressed as the proportion of total plots occupied. Relative frequencies, therefore, range from 0 if a species was not found in any of the plots to 1 if a species occurred in all of the plots. Each plot's uniqueness was calculating using the following equation:

$$\text{Uniqueness} = 1 - \frac{\Sigma \text{ species proportional frequencies on a plot}}{\text{plot species richness}}$$

These values reflect the relative rarity of species in the sample plots. Therefore, plot uniqueness values are low for plots with few species that are common to high for plots with many species that are rare.

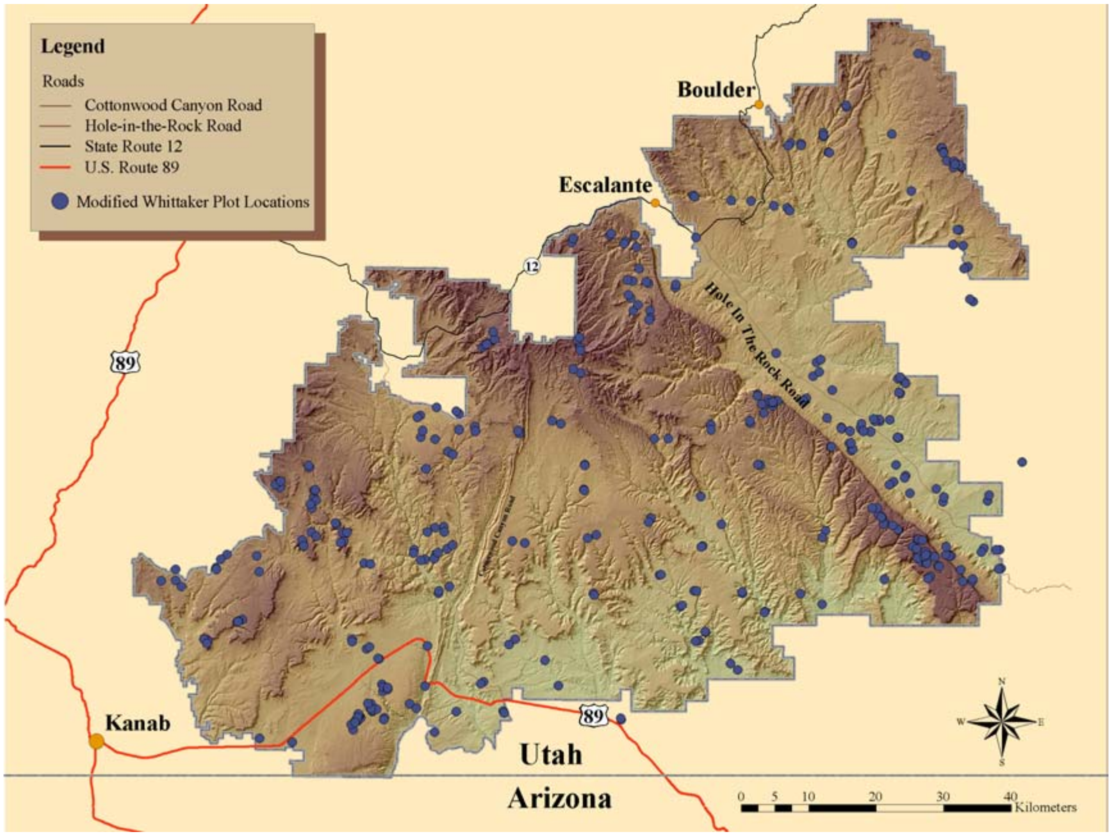


Figure 2-1. Study area for case study 1, including 367 plots established in 19 different vegetation types.

Statistical Analysis

We developed species-accumulation curves for various species groups (e.g., native, non-native, and endemic species) and for various vegetation types using Estimate-S software (Colwell 1997). To compare vegetation, soils, and cryptobiotic crust characteristics of plots with and without endemic species, t-tests were used (Systat Software Inc. 2001). Multiple stepwise regression was then used to determine what biotic and abiotic factors were most important in predicting native and non-native species richness, number of endemic species, and uniqueness at the plot scale. Inverse-distance weighting and kriging algorithms (with 10 nearest neighbors) were also used to develop trend surface maps of hotspots of native and non-native species richness, endemism, and uniqueness. Five classes were displayed for each group representing 20% increments of the plot values. The first 20% were considered primary hotspots, the second 20% were secondary hotspots, etc. Areas of the hotspots were calculated individually and collectively for primary and secondary sites. Many of the disturbed pinyon-juniper plots had been seeded with non-native species during restoration efforts. Therefore, they were excluded from regression and mapping results.

Table 2-1. Mean values (standard errors in parentheses) for plant species richness, frequency of endemic species, and uniqueness values per plot by vegetation type. Vegetation types are ordered by a moisture index, with higher values indicating more mesic sites.

Moisture Index/ Vegetation Type	N	Native Species Richness	Non-Native Species Richness	Native % Cover	Non-Native % Cover	Frequency of Endemics	Mean Uniqueness
1: Desert Shrub	29	22.4 (1.4)	1.9 (0.2)	16.2 (2.0)	2.0 (0.6)	55%	0.79 (0.01)
2: Blackbrush	27	21.2 (1.1)	1.2 (0.2)	29.2 (2.0)	2.4 (0.9)	44%	0.76 (0.01)
3: Desert Shrub/ Grassland	16	26.8 (1.8)	1.6 (0.3)	21.5 (4.3)	2.2 (0.6)	75%	0.77 (0.02)
4: Sagebrush	31	20.6 (1.7)	1.7 (0.2)	24.6 (1.9)	3.1 (0.8)	35%	0.72 (0.02)
5: Juniper-Sage	17	20.1 (1.5)	2.5 (0.4)	24.7 (3.1)	7.4 (2.0)	35%	0.72 (0.01)
6: Juniper	22	28.5 (1.8)	1.3 (0.2)	16.5 (1.9)	0.4 (0.2)	54%	0.76 (0.01)
7: Pinyon-Juniper/ Sage	21	22.8 (1.3)	1.2 (0.2)	31.2 (2.4)	0.9 (0.3)	38%	0.70 (0.01)
8: Disturbed Pinyon-Juniper	35	17.9 (1.3)	3.9 (0.4)	15.2 (2.4)	11.9 (1.7)	46%	0.74 (0.01)
9: Pinyon-Juniper	81	25.6 (0.8)	0.8 (0.1)	25.1 (1.5)	0.7 (0.4)	73%	0.75 (0.01)
10: Pinyon-Juniper/Manzanita	6	35.8 (1.9)	0.3 (0.2)	35.8 (6.2)	0.1 (0.004)	83%	0.80 (0.01)
11: Pinyon-Juniper/Oak	18	32.3 (2.1)	1.2 (0.4)	42.6 (4.4)	0.5 (0.3)	67%	0.79 (0.01)
12: Pinyon Pine	4	26.0 (4.6)	1.2 (0.5)	35.3 (10.6)	1.3 (0.8)	75%	0.80 (0.01)
13: Ponderosa Pine/Manzanita	7	27.6 (2.1)	0.6 (0.2)	38.8 (7.3)	0.8 (0.6)	28%	078 (0.01)
14: Mountain Shrub	11	30.4 (2.2)	1.2 (0.3)	34.6 (4.8)	1.9 (1.1)	82%	0.78 (0.01)
15: Rabbitbrush	9	24.9 (1.5)	3.1 (0.7)	29.7 (5.1)	5.5 (3.3)	44%	0.80 (0.01)
16: Aspen	6	33.3 (3.1)	4.7 (1.0)	57.6 (6.0)	8.7 (2.5)	50%	0.87 (0.01)
17: Wet Meadow	3	33.7 (5.0)	6.7 (0.9)	37.0 (9.5)	27.5 (5.3)	33%	0.88 (0.01)
18: Spring	2	39.5 (3.5)	6.5 (2.5)	29.8 (6.7)	5.4 (4.7)	100%	0.83 (0.01)
19: Perennial Riparian	22	26.3 (2.2)	5.8 (0.4)	35.3 (4.7)	12.3 (2.2)	32%	0.86 (0.01)

Results and Discussion

Completeness of Sampling

The 367 randomly placed plots covered only 0.004% of the Monument's entire area. Even with such a low sampling intensity, 550 of the 940 plant species known to occur in the Monument were found using our sampling design (See Appendix H). This number includes 63 of the 174 (35%) known endemic species. Those species we missed included the listed threatened Jones' waxydogbane (*Cycladenia humilis* var. *jonesii*) and listed endangered species Kodachrome bladderpod (*Lesquerella tumulosa*) and other species restricted to very rare habitats such as hanging gardens.

The species-accumulation curves began to level off, which suggests a fairly complete and balanced sampling of each group (Fig. 2-2). To capture all species in the Monument, it was estimated that 20,300 randomly selected plots (0.23% of entire area) would be needed, although such a sampling intensity would have been cost prohibitive. However, it was possible to evenly evaluate broad patterns of each group because native, non-native, and endemic species were captured at fairly similar rates.

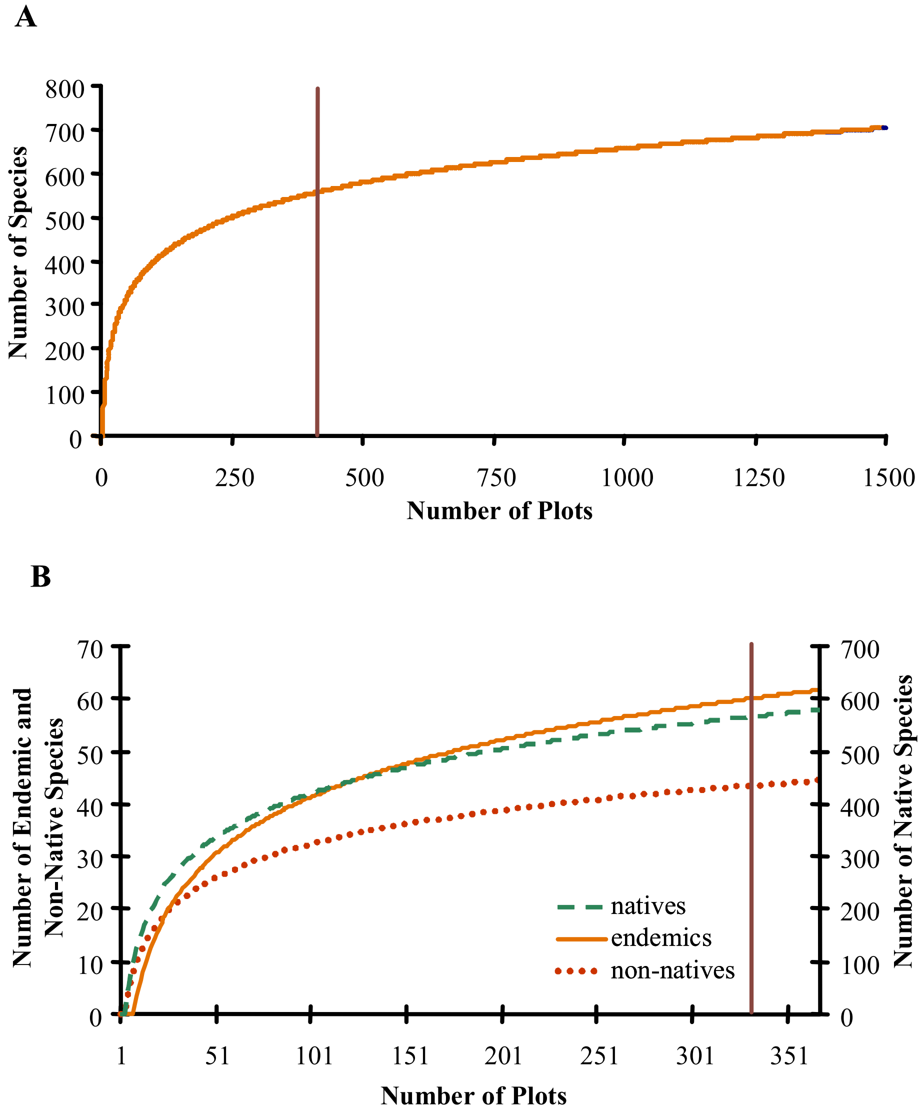


Figure 2-2. Extrapolated species accumulation curves for all species (A; vertical line indicates 367 plots) and species accumulation curves for native, non-native, and endemic plant species (B; vertical line indicates 332 plots, excluding disturbed vegetation type plots).

Species Rarity

From this study, very few plant species were found to be common, with only eight species occurring in 50% of the plots. The most common species that was found was snakeweed (*Gutierrezia sarothrae*), a native species which occurred on 80% (293) of the plots. The second most common species, found in 69% (252) of the plots, was the non-native species, cheatgrass (*Bromus tectorum*). Other frequently found non-native plant species included redstem stork's bill (*Erodium cicutarium*), found in 14% (53) of the plots, and Russian thistle (*Salsola iberica*), found in 12% (44) of the plots.

In contrast, most plant species in the Monument were found to be rare and patchily distributed. One-third of the species captured were only encountered in one or two of the 367 plots, which created a sharp inverse-J shape frequency distribution (Fig. 2-3). Suprisingly, two endemic species (roundleaf buffaloberry; *Shepherdia rotundifolia* and variable springparsley; *Cymopterus purpureus* var. *purpureus*) were found to have a wide regional range, occurring on ~40% of the 45 primary geology types found in the Monument.

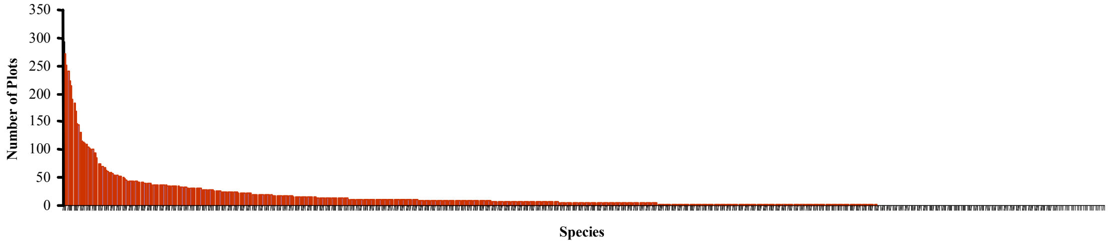


Figure 2-3. The number of plots each of 550 plant species occurs within.

The leveling off of the species accumulation curves (Fig. 2-2) and the rapidly declining frequency histogram (Fig. 2-3) suggests that the remaining uncaptured species are most likely also rare and patchily distributed on the landscape. The plants found in only one or two of the plots (N=189) plus the plants that were missed by our sampling effort (N=390) comprised approximately 62% of the flora. Thus, a pattern in plant diversity was recognized. There were few common species and many rare species found within our randomly placed plots. In the near future, this pattern may prove to be one of the most universal trends in the biological sciences (Rosenzweig 1995), requiring conservation strategies to address the commonness of plant species rarity to preserve plant diversity in complex landscapes such as the Monument.

Other Patterns of Richness, Endemism, and Uniqueness

Native and non-native species richness per plot was greatest in more mesic vegetation types (16-19), which occur rarely within the Monument (Table 2-1). Xeric vegetation types with low moisture index values (1-4) averaged about ten fewer species per plot than mesic types (Table 2-1). Non-native species richness and cover were also high in the disturbed pinyon-juniper vegetation type. This is not surprising considering that many of the disturbed areas had been seeded with non-native species during restoration efforts.

Native species richness had a mean of 25.4 ± 0.5 species per plot (N=332) and varied little across vegetation types. From the multiple regression analysis, native species richness was strongly, positively correlated to percent sand and clay in the soils, elevation, and the moisture index (Table 2-2). However, only 15% of the variation in native species richness was explained by the six independent variables used in the analysis. Positive correlations were also found between native species richness per plot and the $\log_{10}$number of endemic species per plot (r=0.36, P<0.001), the $\log_{10}$number of non-native species per plot (r=0.12, P<0.03), plot uniqueness (r=0.52, P<0.001), and total foliar cover (r=0.20, P<0.001).

Table 2-2. Summary of multiple regression results.

Variables/ Habitat Characteristics	Coefficient	Standardized Coefficient	p-value
Dependent Variable: Native Species Richness			
N=318 F=10.6 P<0.001 R^2=0.15 S.E.=7.8			
Constant	-3.955	0.000	0.549
Moisture Index	0.234	0.137	0.015
Bare Gound %	-0.042	-0.101	0.061
Sand %	0.205	0.434	0.000
Clay %	0.242	0.311	0.007
Soil K	-0.009	-0.107	0.072
Elevation	0.006	0.202	0.000
Dependent Variable: Number of Endemics ($\log_{10}$)			
N=316 F=18.5 P<0.001 R^2=0.28 S.E.=0.2			
Constant	-0.135	0.000	0.075
Clay %	0.006	0.274	0.000
Total Cover %	-0.002	-0.202	0.000
Soil P	-0.007	-0.182	0.001
Soil Mg	-0.000	-0.123	0.037
Soil Ca	0.000	0.075	0.144
Elevation	0.000	0.096	0.080
Native Species Richness	0.011	0.398	0.000
Dependent Variable: Uniqueness			
N=328 F=40.8 P<0.001 R^2=0.46 S.E.=0.05			
Constant	0.627	0.000	0.000
Moisture Index	0.004	0.266	0.000
Crust Cover	-0.001	-0.234	0.000
Sand %	0.001	0.319	0.001
Clay %	0.003	0.424	0.000
Soil N	0.010	0.048	0.267
Elevation	-0.000	-0.248	0.000
Native Species Richness	0.004	0.501	0.000

Table 2-2. (Continued) Summary of multiple regression results.

Variables/ Habitat Characteristics	Coefficient	Standardized Coefficient	p-value
Dependent Variable: Non-Native Species Richness ($\log_{10}$)			
N=312 F=21.5 P<0.001 R^2=0.40 S.E.=0.2			
Constant	0.627	0.000	0.000
Moisture Index	0.013	0.256	0.000
Bare Ground %	0.001	0.071	0.135
Highly Developed Crust	-0.009	-0.108	0.020
Total Cover %	0.003	0.234	0.000
Inorganic Carbon	-0.036	-0.112	0.025
Soil P	0.012	0.276	0.000
Soil Mg	0.000	0.149	0.002
Elevation	-0.000	-0.502	0.000
Native Species Richness	0.005	0.172	0.001
Number of Endemic Species	-0.089	-0.077	0.131

Endemic species occurred in 44% (146) of the plots, excluding those within disturbed areas, with their frequency being greatest in the more common xeric vegetation types. Xeric plots containing pinyon pines had consistently higher frequencies of endemism. In addition, endemic species were found on plots that differed significantly in biotic and abiotic characteristics (Table 2-3). Plots with endemic species had more unique species assemblages, greater cover of intermediate and well-developed crusts, greater native species richness, lower non-native species richness and cover, and tended to have higher clay content in the soils and lower percent nitrogren and percent phosphorus than plots without endemics.

Multiple linear regressions for endemic species richness found that predictor variables included native species richness and percent soil clay, and strong negative correlations existed with total foliar cover and percent soil phosphorus (Table 2-2). Because total foliar cover is generally associated with increased productivity, it can be stated that endemic species were found in the native species rich but less productive sites. Seven indpendent variables explained 28% of the variation in endemic species richness.

Uniqueness tended to increase with increasing moisture index (Table 2-1). Average values ranged between 0.70 in the pinyon-juniper-sage type to 0.88 in the wet meadow type. This suggests that plots in xeric vegetation types contain fewer total species per plot and that these species are more generalists. Plots in mesic vegetation types have high species richness but more unique species assemblages (Table 2-1). Uniqueness was positively correlated with the $\log_{10}$number of endemic species per plot (r=0.25, P<0.001).

From the multiple linear regression analysis, it was found that the primary predictor variable for uniqueness was native species richness (Table 2-2). However, although uniqueness was negatively correlated with elevation, native and endemic species richness were positively correlated with elevation. More of the variation in plot uniqueness (46%) was explained by the seven independent variables than endemic species richness.

Table 2-3. T-test results from comparison between plots with endemics occurring on them (N=199) and plots without endemics (N=168). Analysis was run on the 367 plots surveyed. Averages are reported with standard errors in parentheses.

Variable	Endemic Plots	No-Endemic Plots	p-value
Number of Endemics	1.65 (0.07)	0	N/A
Uniqueness	0.78 (0.004)	0.75 (0.007)	<0.001
Soil Cover[1]	50.8 (1.8)	45.6 (0.6)	0.007
Bare Soil[1]	23.9 (1.7)	21.5 (0.5)	0.804
Young Cryptobiotic Crust Cover[1]	22.8 (1.5)	21.4 (0.5)	0.124
Intermediate Cryptobiotic Crust Cover[1]	2.7 (0.5)	1.8 (0.1)	0.065
Well-Developed Cryptobiotic Crust Cover[1]	1.4 (0.3)	1.0 (0.8)	0.018
Non-Native Foliar Cover	2.4 (0.4)	5.4 (0.6)	<0.001
Native Foliar Cover	26.3 (1.1)	26.6 (1.4)	0.857
Non-Native Richness	1.7 (0.1)	2.3 (0.2)	0.003
Native Richness	27.3 (0.6)	21.5 (0.6)	<0.001
Cheatgrass Cover	1.3 (0.2)	3.1 (0.5)	<0.001
% Sand	71.4 (1.4)	72.6 (1.3)	0.560
% Clay	17.5 (0.9)	15.0 (0.6)	0.023
% Inorganic Carbon	0.74 (0.06)	0.645 (0.06)	0.266
% Organic Carbon	0.72 (0.05)	0.79 (0.09)	0.538
% Nitrogen	0.14 (0.02)	0.22 (0.04)	0.042
P (mg/kg)	5.9 (0.4)	8.9 (0.6)	<0.001
K + Mg + Ca + Na (ppm)	1986 (129)	1836 (84)	0.356
pH (plot average)	7.9 (0.03)	7.8 (0.03)	0.434
Elevation (m)	1835 (19)	1803 (21)	0.250

[1] Bare ground and cryptobiotic cover data are taken from 1-m^2 subplots, N=323 for subplots with endemics, and N=3,347 for subplots without endemics.

Species accumulation curves for native species were similar among all vegetation types (Fig. 2-4), but were very different among vegetation types for non-native and endemic species. For instance, mesic vegetation types quickly accumulated non-native species, while xeric vegetation types slowly accumulated them. However, xeric vegetation types quickly accumulated endemic species, while mesic types accumulated endemics more slowly. The species accumulation curve for the desert shrub sites was still steeply increasing, suggesting that many more endemic species would be found with additional sampling of this vegetation type (Fig. 2-4).

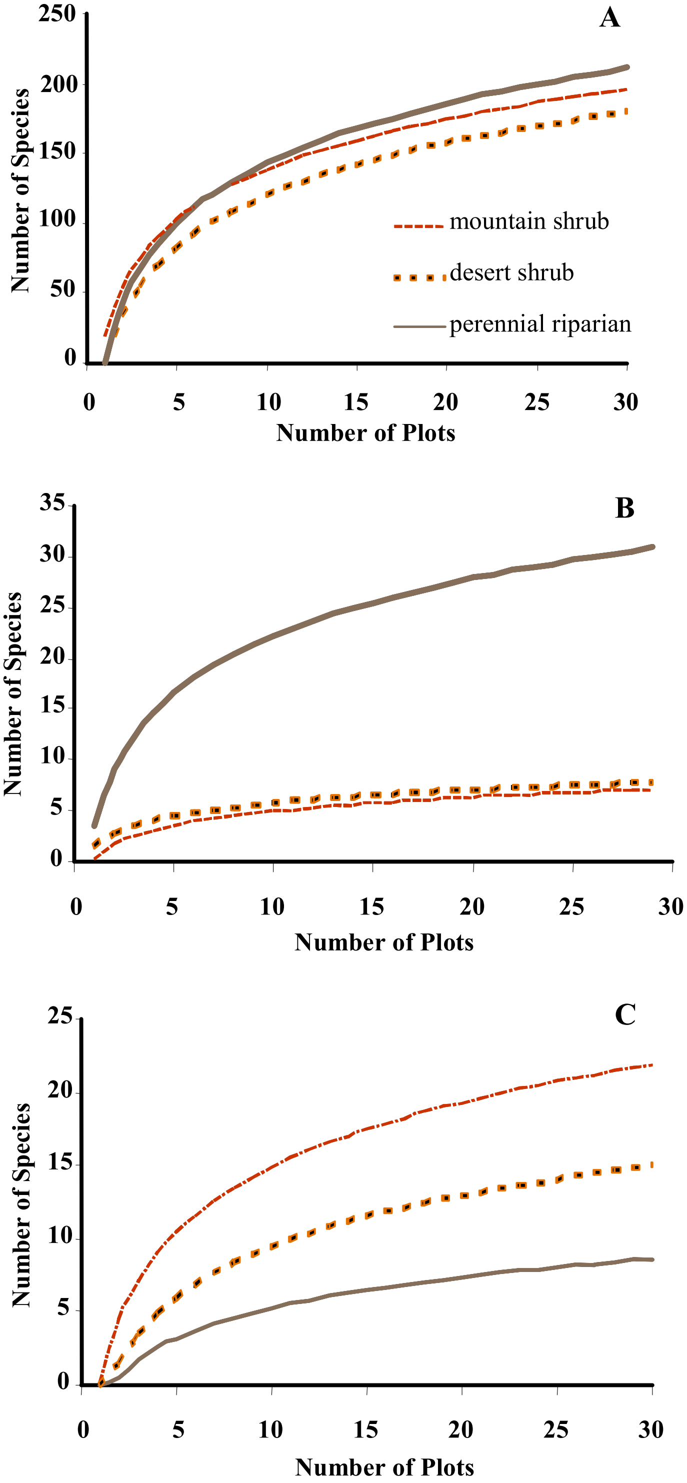
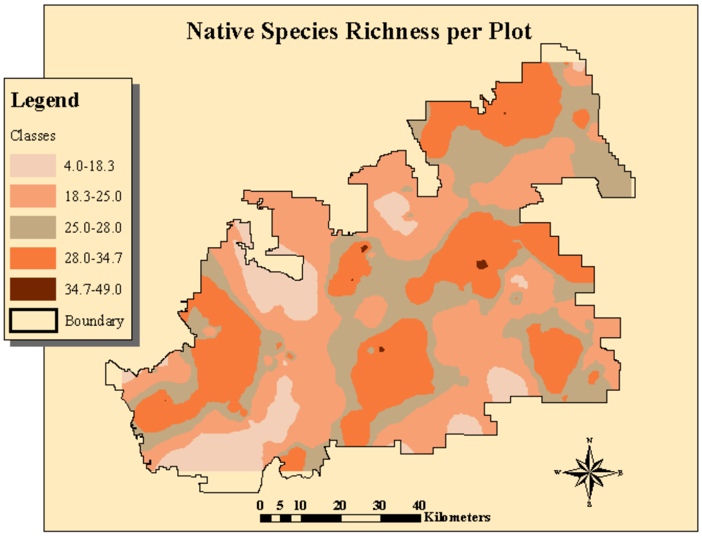
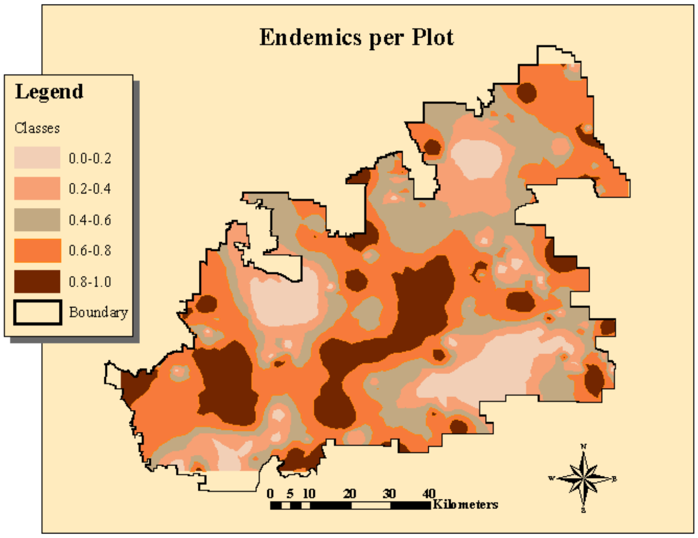


Figure 2-4. Examples of vegetation-type contributions to (A) native species richness, (B) non-native species richness, and (C) endemic species richness as calculated by species accumulation curves.

Landscape Patterns of Richness, Endemism, and Uniqueness

This study found that 16% of the Monument supported the highest concentrations of endemic species (Table 2-4), while a global-scale study found that 12% of the Earth's land mass supported primary areas of endemism (Myers et al. 2000). These comparable findings may have resulted for two reasons. First, concentrated hotspots might be associated with site-specific evolutionary patterns, processes, and unique environments at most spatial scales (Davis et al. 1994; Davis et al. 1995; Davis et al. 1997). In addition, a strong positive correlation between endemic species richness and total native species richness often exists (Hobohm 2003). This case study found concentrated hotspots of native species richness (0.1% of landscape) and endemic species (16% of landscape), and a positive correlation between native species richness per plot and log$_{10}$number of endemic species per plot (r=0.36, P<0.001).





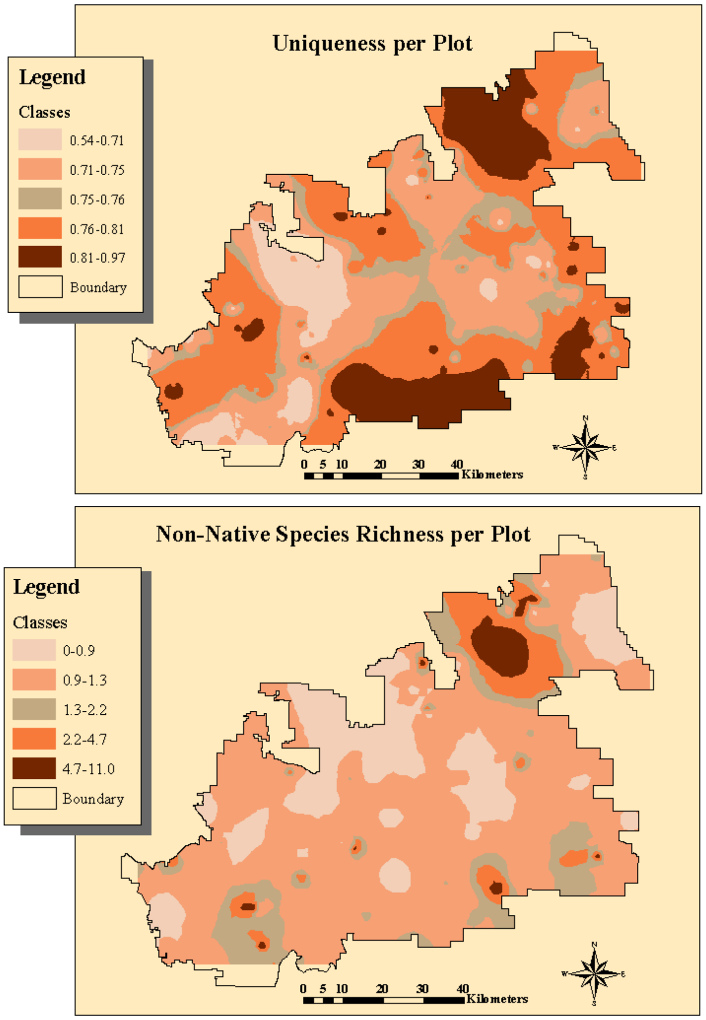


Figure 2-5. Primary (top 20% of plot values) and secondary hotspots (next 20% of plot values) of native species richness, endemism, uniqueness, and non-native species richness in the Monument.

However, little overlap existed among the primary hotspots of native species richness, endemism, and uniqueness on the landscape (Table 2-4; Fig. 2-5). Overlap among the groups totaled 11% of the landscape; whereas, total cover of the three groups was 74% of the Monument. The different biotic and abiotic variables that were correlated with patterns of richness, endemism, and uniqueness (Table 2-4) further suggests that different evolutionary and ecological mechanisms may be maintaining these patterns.

Table 2-4. Percent of the Monument occupied by primary (top 20%) and secondary (top 40%) hotspots of native species, endemic species, and unique species assemblages.

Hotspot Type	Primary Hotspot Area	Primary + Secondary Hotspot Area
Hotspots of Native Species	0.1%	16%
Hotspots of Endemic Species	16%	29%
Hotspots of Uniqueness	16%	37%
Cumulative Hotspots	32%	74%

Case Study 2: *Example of a Perennial Riparian Vegetation Type - The Escalante River*

A study was conducted to establish baseline ecological conditions for the riparian vegetation type of the Escalante River to monitor vegetation changes over time. This study was specifically established to monitor the responses of riparian vegetation to altered grazing regimes, flood regimes, and the increasing presence of invasive plant species (most notably species of the *Tamarix* and *Elaeagnus* genera). Specifically, the objectives of this study were to (1) establish baseline conditions by sampling ten modified-Whittaker plots along the Escalante River to compare with future post-grazing years, (2) ensure re-location of these plots by reporting accurate GPS locations, and (3) report total, native, and non-native species richness along with total, native, and non-native cover, cheatgrass cover, cryptobiotic crust development, and baseline soil information for the ten plots.

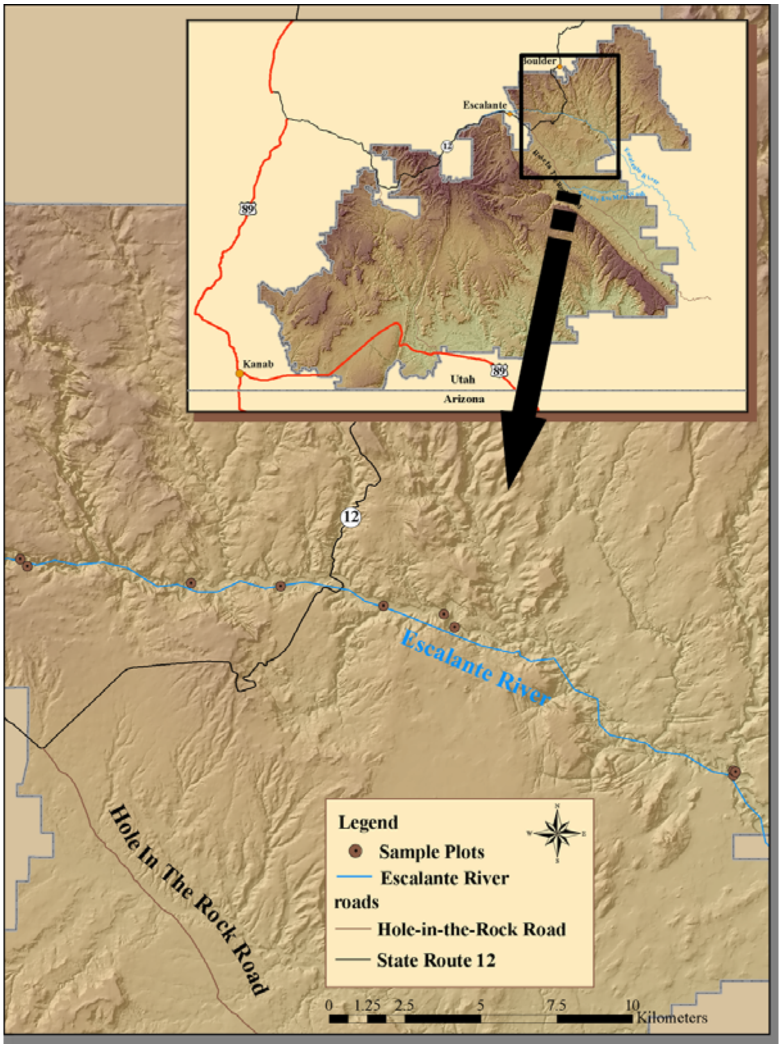


Figure 2-6. Study area for the Escalante River case study, including plot locations.

Site Description

The Escalante River basin in south-central Utah is nestled between the Aquarius Plateau to the north, the Kaiparowits Plateau to the west, the Circle Cliffs to the east, and its drainage destination of Lake Powell to the south. The Escalante River is the major drainage river for this basin (Fig. 2-6). It originates in the Escalante Mountains of west-central Garfield County at elevations of 3,735 m and flows southeast past the town of Escalante, through the Escalante Canyons across sections of Grand Staircase-Escalante National Monument and Glen Canyon National Recreation Area, and finally into the Colorado River where it becomes an arm of Lake Powell at an elevation of 1,160 m (Birkeland 1999).

The extensive riparian vegetation now present along the Escalante River and its tributaries is known to be a fairly recent development. Historical photographs (Woolsey 1964) and written accounts (Webb 1985) document little or no riparian vegetation communities along the river channel from 1875, when the town of Escalante was established, to 1940. However, this scarce vegetation became more developed since 1940 as the hydrological regime of the river changed from large, frequent flood events to a lower flow regime due to increased European human occupancy and subsequent anthropogenic hydrological changes (Birkeland 1999). Such changes include the construction of diversion dams and irrigation canals (late 1800s) and the water storage reservoirs of North Creek reservoir built in 1938 and Wide Hollow reservoir built in 1951 (Woolsey 1964).

The altered post - 1940 flow and flood regimes of the Escalante River and its tributaries encouraged the establishment of woody riparian vegetation communities consisting of native Fremont Cottonwood (*Populus fremontii*), coyote willow (*Salix exigua*), Goodings willow (*Salix gooddingii*), and box elder (*Acer negundo*) and the non-native tamarisk (*Tamarix ramossissima*) and Russian olive (*Elaeagnus angustifolia*). Prior to about 1930, tamarisk and Russian olive were nearly absent along the Escalante River according to both historical photos and written accounts (Lambrechtse 1985; Webb 1985). By the 1980s, both species had established and proliferated to greater numbers (Lambrechtse 1985; Webb 1985). It is this proliferation for which the study plots of this study were established to monitor into the future as other unforeseen changes occur.

Little is known about the land use history of the Escalante River area. However, it is known that prehistoric peoples inhabited the area prior to the 1800s (Birkeland 1999), white settlers established the town of Escalante in 1875 (Woolsey 1964), logging and vegetation clearing practices occurred for shelter construction immediately after settlement (Birkeland 1999), lumbering within the Pine Creek area became an industry (Woolsey 1964), and the areas' water diversions were built in the mid 1900s (Woolsey 1964). These practices led to significant erosion throughout the upper Escalante Basin during this time period (Birkeland 1999).

Livestock grazing of cattle (up to 12,500 head) and sheep (up to 75,000) occurred on the high plateaus in summer and lower elevations in winter from 1900 to 1922 and then decreased through 1963 (Woolsey 1964; Birkeland 1999). In 1957, the Bureau of Land Management restricted livestock grazing in riparian areas to seasonal use from November 1 to June 15 and then changed this seasonal period in 1978 to September 1 through March 31 (Birkeland 1999). The allotments on the Glen Canyon National Recreation Area below where Harris Wash joins the Escalante River were closed in 1988 (Stewart 2003; pers. comm.). Above that point, all allotments were closed by 1998. Some areas such as the Big Browns Bench and Lower Horse Creek areas were not closed until 2002, but haven't been grazed since 1998. Some trespass grazing has occurred along the river from time to time, but is relatively rare (Stewart 2003; pers. comm.).

Thus, there has been a diverse set of past land use practices throughout the Escalante River basin. However, detailed scientific studies of these land use practices have not been conducted. The ten plots established in this study will hopefully provide solid scientific observations of baseline conditions at this time from which comparisons can be made upon future visits to these plots. Such time series data may provide insight about the potential effects of land use changes on the riparian vegetation of the Escalante River.

Methods

In May 2000, ten modified-Whittaker plots were located along the Escalante River within two grazing allotments, Phipps and Deer Creek. Seven plots were located within the Phipps allotment, and three plots were located within the Deer Creek allotment. The plots ranged in elevation from 1,475 m to 1,682 m. Plots were sampled for total, native, and non-native species richness; mean total, native, non-native, and cheatgrass cover; tamarisk presence/absence, cryptobiotic crust development; ancillary soil parameters; and soil chlorophyll content.

Results & Discussion

Native species richness averaged 24.5 ± 3.0 species and ranged between 6 (plot 104b) and 39 species for the ten plots. Non-native species richness averaged 5.0 ± 0.7, with values ranging from 1 species (plot 104b) to 8. Plot number 104b (distinctively different due to the presence of a salt lick) only contained 7 species; 6 native and 1 non-native. These values were the lowest of all ten plots sampled. Excluding this plot, native species richness ranged from 15 to 39 species and non-native species ranged from 2 to 8 species.

Table 2-5. Summary vegetation data for the Escalante River plots (standard errors are in parentheses). Unknowns were not included so the addition of native and non-native values do not equal total values.

Plot	Total Species Richness	Native Species Richness	Non-Native Species Richness	Total Cover	Native Cover	Non-Native Cover	Cheatgrass Cover
101a	44	39	2	36.8	36.6	0.0	0.0
101b	30	25	4	19.6	13.4	6.2	6.1
102a	27	22	5	21.1	18.7	2.5	0.0
102b	34	25	8	37.3	26.5	10.9	0.0
103a	35	29	5	37.4	33.5	3.9	1.3
103b	28	19	8	55.9	42.9	13.0	1.0
104a	38	33	5	26.5	26.2	0.3	0.3
104b	7	6	1	0.8	0.1	0.7	0.0
105a	37	32	5	19.9	17.0	3.0	0.7
105b	25	15	7	24.5	13.8	10.4	2.9
Average	**30.5 (3.2)**	**24.5 (3.0)**	**5.0 (0.7)**	**28.0 (4.7)**	**22.8 (4.0)**	**5.1 (1.5)**	**1.2 (0.6)**

Eighty percent of the ten 1000-m^2 plots recorded the presence of tamarisk. Where this species was recorded in the 1-m^2 subplots, its percent cover ranged from 2.1% to 9.5%. Russian olive occurred in 50% of the ten 1000-m^2 plots and ranged from 0.05% cover to 0.10% cover. Cheatgrass cover within the plots was low compared to that typical of other vegetation types in the Monument. Mean cheatgrass cover averaged 1.2 $\pm$ 0.6 and 40% of the plots sampled had no cheatgrass cover. A study by Waters (2003) found greater cheatgrass biomass within more mesic vegetation types, but the perennial riparian vegetation type had the least cheatgrass biomass of all the other mesic vegetation types. It was suggested that the level of water availability at these sites is high enough that cheatgrass may lose its competitive advantage over native species.

Out of the 100 total 1-m^2 subplots sampled, only one cottonwood sapling (< 100 cm) was found, indicating little recent cottonwood regeneration along the Escalante River. There were more non-native species present in the ten Escalante River plots than typically found in other plots established throughout the Monument, but there was less cheatgrass cover compared to the Monument average (2.01%). The additional non-native species richness along the Escalante River can be attributed to tamarisk, Russian olive, Kentucky bluegrass (*Poa pratensis*), alfalfa (*Medicago sativa*), common dandelion (*Taraxacum officinale*), and other moisture loving non-native invaders that might otherwise be less abundant in other areas of the Monument due to less water availability. Past altered flow and flood regimes of the Escalante River appear to have rendered the riparian channel more susceptible to present and future invasion by non-native species, and this area must be monitored closely to prevent further invasion.

Management Implications

To properly develop strategies for preserving various hotspots and species from extinction, threats to biodiversity must be considered (Sisk et al. 1994). The first case study found that approximately 80% of the plots sampled contained at least one non-native species and that, on average, non-native plant species comprise 7.5% of the species and 12.4% of the total cover on a given plot. The primary predictors of non-native species richness included positive correlations with soil phosphorus, moisture, and total foliar cover and a negative correlation with elevation (Table 2-2). Forty percent of the variation in non-native species richness was explained by the ten factors used in the multiple regression analysis. Similar to other studies (Stohlgren et al. 1998b; 1999b; Bashkin et al. 2003), we found that non-native plant invasions have been most successful in mesic habitats high in native species richness and soil nutrients (Table 2-1; Fig. 2-4). Managers may find protection of these highly diverse areas more difficult as non-native species continue to establish and gain cover in these rare vegetation types.

Non-native species may also be negatively affecting broad areas of unique species assemblages (Table 2-2). The $\log_{10}$number of non-native species per plot was negatively associated with the $\log_{10}$number of endemic species per plot (r=-0.16, P=0.004), and positively associated with the plot uniqueness (r=0.37, p=0.0001). Positive correlations that exist between uniqueness and non-native species richness and cover show that non-native species have successfully invaded areas where native plant species have very low frequencies of occurrence.

Similar to other studies (Harrison 1999), endemic species within the Monument seem to flourish in arid habitats low in soil nutrients and foliar cover (i.e., lower productivity sites, but moderately high in native species richness). These areas may help preserve other isolated populations of endemic and other native species by serving as refugia for these species because non-native species are unable to survive in these harsh environments (Fig. 2-4).

The Escalante River plots provide a useful baseline for future studies. The vegetation community is susceptible to non-native species invasions and activities which limit natural, episodic, and large flood events are likely to exacerbate these invasions. Control efforts should be focused on containing current tamarisk and Russian olive populations at or downstream of their current locations to help protect upstream areas and tributaries from further invasion.

CHAPTER 3

Soils and Cryptobiotic Crusts

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The Monument can be divided into three regions: the Grand Staircase , the Kaiparowits Plateau, and the Escalante Canyons, each having it own distinct characteristics. The Grand Staircase region ascends 1,676 m from the rim of the Grand Canyon north to Bryce Canyon National Park. Its “steps” include the Chocolate Cliffs, Vermilion Cliffs, White Cliffs, Gray Cliffs, and Pink Cliffs, which consist of large expanses of exposed, virtually undeformed rock strata. The Kaiparowits Plateau has the highest elevations within the Monument and covers approximately 2,575 km^2. The soils are rich in salts and acids, making the area inhospitable to many plants. The Escalante Canyons consist of Navajo sandstone which has been carved by the Escalante River and its tributaries. Within these canyons are riparian vegetation types and hanging gardens (Grahame and Sisk 2002). Because erosional processes of each region produce soils that retain properties of their parent material, the soils within each of these geologic regions are as distinct as the regions themselves.

The Monument contains a mosaic of soil types with varying chemical and textural characteristics that provide suitable conditions for a wide range of native plant species (Belnap 1999). These properties can combine to form unique substrates that are very isolated and limited in area. The environmental conditions at these microsites are such that only several narrowly restricted plant communities can occupy their niche (Welsh 1979). Therefore, soil type can be a major factor in determining the distribution of the Monument’s plant species.

Cryptobiotic Soil Crusts

One of the most unique properties of soils within the Monument is the presence of cryptobiotic soil crusts (Grand Staircase-Escalante National Monument 2000). These crusts are found throughout semiarid and arid regions of the Colorado Plateau and play a critical role in maintaining the sustainability of these ecosystems. They function in increasing soil stability, erosion control from wind and water, atmospheric nitrogen fixation, nutrient uptake to plants, soil-plant-water relations, infiltration processes, germination enhancement, and plant growth (Anderson et al. 1982a; Anderson et al. 1982b; Belnap and Gardner 1993; Belnap and Harper 1995; Belnap and Gillette 1998).

Cryptobiotic soil crusts can range from minor ridges (<1 cm high) to highly developed pedicles (3-6 cm high) and contain a combination of blue-green algae, lichens, green algae, and fungi. Specifically, crusts within the Monument are composed of predominantly cyanobacteria (*Microcoleus vaginatus*), algae (>40 species), lichens (*Collema tenax* and *C. coccophorum* dominate), and mosses (*Syntrichia caninervi* and *S. ruralis* dominate), but are also known to contain taxa of liverworts, fungi, and bacteria (Johansen et al. 1984; Evans and Johansen 1999; Belnap and Lange 2001).

These organisms are extremely vulnerable to compressional disturbances such as trampling by livestock, people, and off road vehicle use (Belnap 1996; Belnap 1997; Fig. 3-1). Such disturbances could impede the nitrogen fixing capability of these organisms by reducing nitrogenase activity (Belnap 1996). This could potentially affect the nitrogen budget in these systems and possibly result in shifts to species composition (Belnap 1995; Evans and Belnap 1999). The loss of crusts can also result in decreased water availability to plants, accelerated soil loss through wind and water erosion, decreased diversity and abundance of soil biota, and slowed decomposition of soil organic matter (Belnap 1995, 1996). Following such disturbances, a study has shown that it could take these crusts 14 to 18 years to regenerate 15% of their original areas and only an additional 1% during the following 20 years (Anderson et al. 1982a).

Figure 3-1. Pictures of undisturbed and disturbed cryptobiotic soil crusts.

Case Studies

Case Study 1: *Soil Characteristics and Non-Native Plant Species Invasions*

A study was conducted to examine correlations among native and non-native plant species and soil characteristics within a 29,000 ha area in the southeast corner of the Monument

(Bashkin et al. 2003). This study included only a subset of the plots (N = 97) sampled, and included 11 vegetation types specific to this study (Fig. 3-2). The objectives were to quantify patterns of non-native plant species and cryptobiotic crusts and determine the effects of soil characteristics on invasion by non-native plant species.

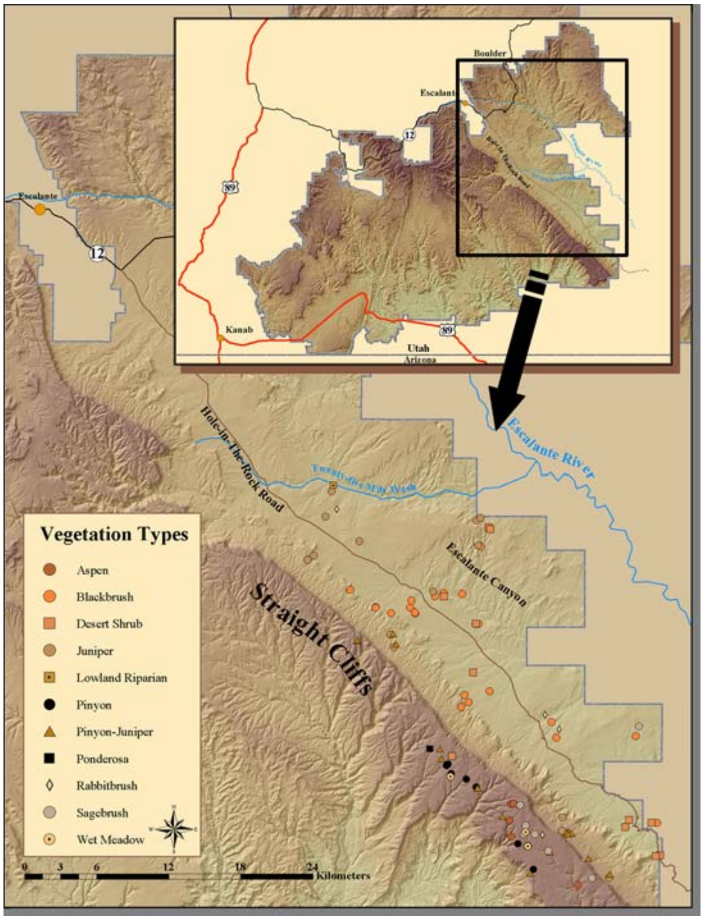

Figure 3-2. Study area for case study 1, including plot locations by vegetation type.

Methods

Field sampling, soil sampling, and soil analyses were conducted as described in the methods chapter, but some additional statistical analyses were performed to address the specific objectives of this study (See Methodology).

Statistical Analyses

Canonical correspondence analysis (CCA; ter Braak 1986; 1987) was used to characterize relationships between native and non-native species and soil characteristics. CCA is a widely used direct gradient analysis technique that constrains the extracted pattern to linear combinations of explanatory variables (i.e., measured environmental variables; ter

Braak 1986; ter Braak 1987). CCA has been proven a robust method for describing species-environment relationships (Palmer 1993; Reed et al. 1993).

Data used in all analyses were checked for normality and transformed using $\log_{10}$ (x+1) or square root transformations if needed. CCA was run on two disparate data sets. The first data set contained species richness data for dominant (>1% cover) and sub-dominant (<1% cover) native and non-native plant species and cryptobiotic class. The second data set consisted of cover data for native and non-native species, cheatgrass, and cryptobiotic crusts. Both data sets were evaluated against 11 environmental variables that included total carbon, inorganic carbon, nitrogen, phosphorus, potassium, sodium, calcium, magnesium, clay, elevation, and bare ground. All environmental variables were tested for correlation among each other, but multicollinearity was not a problem. Monte Carlo permutation tests were used to test for significance in the ordination and showed that it was significant (ter Braak 1991). The environmental variables were then used as independent variables in stepwise forward multiple regressions (SPSS Inc. 2001) to predict native and non-native species cover and richness, cheatgrass cover, and cryptobiotic cover and class.

Results and Discussion

Patterns of Native Species Richness and Cover

From the canonical correspondence analysis, the centroid (i.e., the weighted mean of a multivariate data set) for dominant native species richness was located at the center of the diagram. This ordination space corresponds to habitats with low elevation and less fertile soils (Fig. 3-3). The stepwise forward multiple regression showed that soil characteristics were able to explain only 25% and 13% of the variation in native species richness and cover, respectively (Table 3-1). Native plant species have adapted to this arid landscape making them able to establish and succeed in a broad array of environments (i.e., xeric lowlands to rare mesic habitats). Therefore, it is not surprising that native species richness was found near the center of the ordination diagram, suggesting an equal preference to all habitats. This may also explain why little variation in native species richness and cover could be explained by soil characteristics. Because native plant species are able to grow in a wide range of soil types (Belnap 1998), it is unlikely that soil characteristics would be a strong predictor of native species presence.

The centroid for native species cover was near the end of canonical axis 2, corresponding to habitats with higher elevations and moderately high soil nutrients (Fig. 3-3). Plant species are known to accumulate foliar cover and gain dominance in areas of high resource availability, so it is not unexpected that native species cover was greatest in habitats of high elevation (i.e., greater water availability) and nutrient availability.

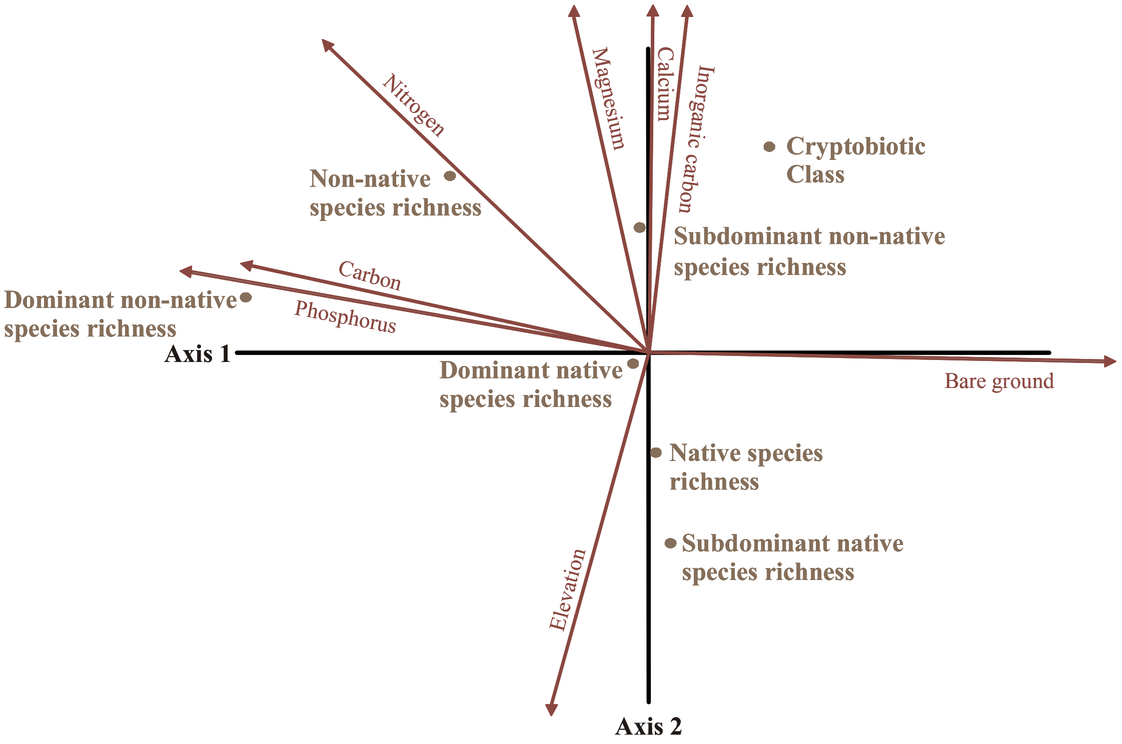

Figure 3-3. Canonical correspondence analysis ordination for native and non-native species richness and cryptobiotic class. The "dominant" designation refers to the number of species with >1% cover. The subdominant designation refers to the number of species with <1% cover. Total richness is the total number of species regardless of cover.

Patterns of Non-Native Species Richness and Cover

Unlike native plant species, non-native species are not capable of growing and succeeding in all vegetation types. From the canonical correspondence analysis, the centroid for non-native plant species richness was located almost directly on the soil nitrogen gradient, and the centroid for dominant non-native species richness was located near the increasing portions of the soil carbon and soil phosphorus gradients. When looking at cover, the centroid for non-native species corresponded to habitats high in soil phosphorus. The centroid for cheatgrass cover was located between axes 1 and 2. This ordination space is positively correlated with soil phosphorus and negatively correlated with bare ground. Therefore, non-native species richness was closely associated with habitats of greater resource availability.

In general, soil characteristics were better able to explain non-native species richness (41%) and cover (46%) than native species richness (24%) and cover (14%), thus increasing our ability to predict patterns of non-native plant species invasions. Certain vegetation types, such as the aspen and wet meadow, have more fertile soils and more moisture availability and are particularly vulnerable to invasion. These vegetation types have a high number of non-native species and high non-native cover (Stohlgren et al. 2001; Bashkin et al. 2003).

In addition, soil phosphorus might prove to be a good indicator of non-native species establishment and success. However, because linear models were used to assess relationships between species composition and soil characteristics, further research may suggest that these relationships are non-linear (Fig. 3-4). A threshold level needs to be defined to determine the concentration of soil phosphorus at which non-natives may establish and succeed.

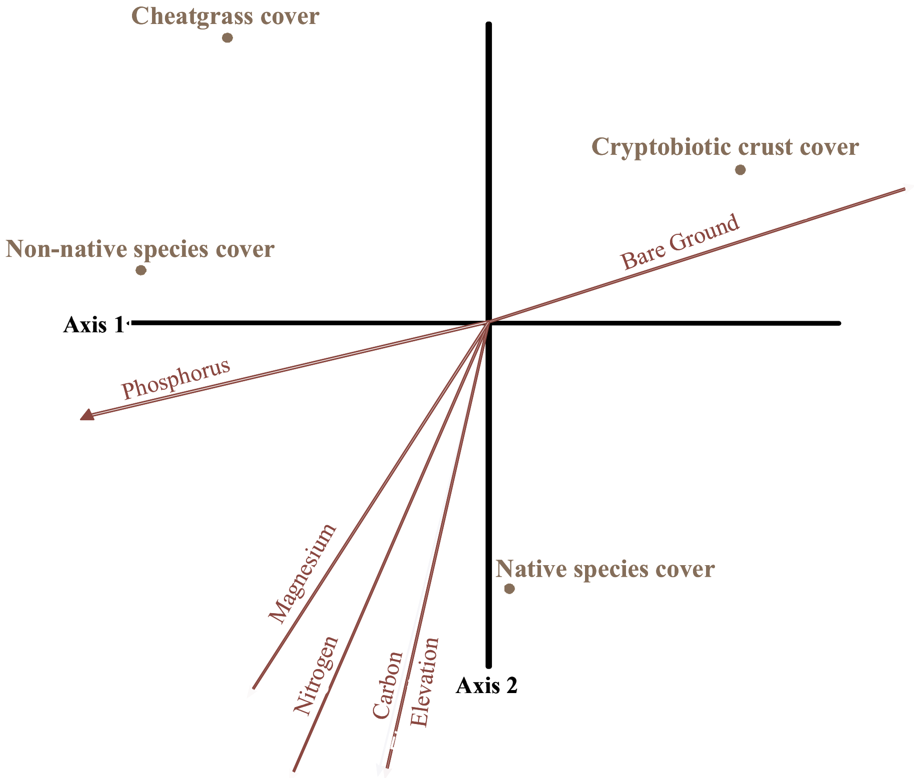

Figure 3-4. Canonical correspondence analysis ordination for cover of native and non-native species, cryptobiotic crusts, and cheatgrass.

Patterns of Cryptobiotic Crust Class and Cover

Native cryptobiotic crusts appear to fill in soil habitats typically lower in soil nutrients. This is not surprising considering the role of these crusts in arid landscapes. However, the interpretation of the cryptobiotic data is preliminary and disturbance history needs to be considered (Belnap 1995). The distribution of cryptobiotic crusts and their complex interactions with non-native plant species requires further research.

Predictors of Native, Non-Native and Cryptobiotic Richness and Cover

The results of the multiple regression analysis revealed that few of the environmental variables that were measured could predict native and non-native species richness and cover, and cryptobiotic class and cover (Table 3-1). Elevation and bare ground were found to be the most commonly significant variables, and soil phosphorus was found to be a significant ($P < 0.001$) predictor of non-native species cover and richness. The variation explained by the environmental variables ranged from 13% for native species cover to 53% for non-native species richness (Table 3-1). The addition of biotic variables to the multiple regression models increased R^2 values only slightly, which is an indication that more environmental variables need to be added to the model to explain patterns of native and non-native species more fully.

Table 3-1. Stepwise forward multiple regressions for species composition (richness and cover), cheatgrass, and cryptobiotic cover.

Dependent Variable/Predictor	Coefficient	*t*	*P*	Model *F*, R^2, *P*
Number of Native Species				
Constant	3.016	17.488	<0.001	F=16.2, R^2=0.25, P <0.001
Elevation	<0.001	5.595	<0.001	
Mg	-0.247	-3.161	0.001	
Native Species Cover				
Constant	4.247	16.691	<0.001	F=7.9, R^2=0.13, P=0.001
Bare Ground	-0.010	-2.664	0.009	
Cryptobiotic Class	-0.116	-1.617	0.110	
Cryptobiotic Cover				
Constant	2.817	4.222	<0.001	F=17.5, R^2=0.44, P <0.001
Ca	<0.001	3.290	0.002	
C	-1.659	-5.165	<0.001	
Bare Ground	0.012	1.722	0.089	
Number of Non-Native Species	-0.542	-2.173	0.033	
Cryptobiotic Class				
Constant	2.469	3.325	0.001	F=13.7, R^2=0.22, P <0.001
Elevation	-0.001	-2.979	0.004	
Bare Ground	0.015	2.627	0.010	
Number of Non-Native Species				
Constant	0.119	0.200	0.842	F=16.2, R^2=0.53, P <0.001
Elevation	-0.001	-4.161	<0.001	
Mg	0.344	2.601	0.011	
P	0.275	3.813	<0.001	
Bare Ground	-0.008	-3.005	0.004	
Cryptobiotic Cover	-0.093	-2.464	0.016	
Number of Native Species	0.528	3.654	<0.001	
Non-Native Species Cover				
Constant	2.671	3.556	0.001	F=23.1, R^2=0.43, P <0.001
Elevation	-0.001	-1.465	0.147	
P	0.622	4.204	<0.001	
Bare Ground	-0.026	-4.505	<0.001	
Cheatgrass Cover				
Constant	3.086	3.768	<0.001	F=10.3, R^2=0.33, P <0.001

Table 3-1. (Continued) Stepwise forward multiple regressions for species composition (richness and cover), cheatgrass, and cryptobiotic cover.

Dependent Variable/Predictor	Coefficient	*t*	*P*	Model *F*, R^2, *P*
Elevation	-0.001	-1.906	0.061	
Na	-0.003	-1.610	0.112	
P	0.569	3.491	0.001	
Bare Ground	-0.022	-3.523	0.001	

Case Study 2: Fire Effects on Cryptobiotic Soil Crusts

The objective of this study was to compare the response of cryptobiotic soil crusts within burned sites and burned sites with post-burn mechanical seeding treatments to unburned sites over time (Evangelista et al. 2004a).

Methods

Study Areas

Four burned areas and associated controls were chosen for this study. All sites were predominantly Utah juniper (*Juniperus osteosperma*) mixed with pinyon pine (*Pinus edulis*), big sagebrush (*Artemesia tridentata*) and cliffrose (*Purshia mexicana*) prior to the burn. All burns examined, except for the Crawford Bench Fire, were ignited by lightning strikes and occurred in July. The Crawford Bench site was also geographically different (i.e., elevation, soil types, precipitation, etc.) than the other sites, which may have influenced fire impacts and crust recovery rates.

Crawford Bench Fire

The Crawford Bench Fire occurred in the early 1950's and burned approximately 200 ha. Prior to the fire, the site had a mixed composition of Utah juniper, pinyon pine and Gamble's oak (*Quercus gambellii*), but the area is currently comprised of Gamble's oak, big sagebrush, and manzanita (*Arctostaphylos patula*). The site is grazed by livestock during the winter months, and limitations on recreational and other activities exist due to restricted access and the existence of privately owned land within the area.

Five Mile Mountain Fire

The fire at Five Mile Mountain occurred in 1989, ten years prior to our sampling, and burned approximately 300 ha. The area is currently dominated by Utah juniper with interspersed patches of big sagebrush and cliffrose. Soils are from limestone parent materials (Table 3-2) and described as sandy loam (Chapman 1997). Land-use on Five Mile Mountain includes livestock grazing during winter months, firewood cutting, and minimal recreational use.

Table 3-2. Site descriptions and average soil characteristics for each of the study sites. Substrate was determined by Doelling and Davis (1989).

Study Site	Elevation	Substrate	% Sand	% Silt	% Clay	% Soil C	% Soil N	Soil P (mg/kg)
Crawford Bench	2,165 m	Quaternary alluvial gravel	59	27	14	0.09	1.1	10.2
Five Mile Mt. 1989	1,805 m	Triassic limestone	52	28	20	1.2	0.1	14.8
1997 Buckskin	1,795 m	Triassic limestone	54	24	23	0.6	1.3	20.5
1998 Buckskin	1,750 m	Triassic limestone	44	28	25	0.6	0.7	15.6

Buckskin Fire (1997)

In 1997, approximately 140 ha burned on Buckskin Mountain. Pre-burn vegetation of the site was mostly Utah juniper, big sagebrush and cliffrose. Three months after the fire, resource managers mechanically seeded (i.e., rangeland drill and dribbler pulled by a bull-dozer) the burn site with native annual and perennial seeds. Additionally, 100 cliffrose seedlings were planted in areas inaccessible by the drill (Chapman 1997). Soils are from limestone parent materials (Table 3-2) and described as sandy loam with gravelly loam and clay loam sub-surfaces (Chapman 1997). Land-use on Buckskin Mountain includes winter livestock grazing, firewood cutting, and minimal recreational use.

Buckskin Fire (1998)

The site description for the 1998 fire is similar to the 1997 burn (Table 3-2). This fire occurred a few miles east of the 1997 burn, but received no post-fire seeding treatments. The fire burned 445 ha, including 120 ha that previously burned in 1996. Only the area burned in 1998 was sampled.

Field Sampling

Four to six pairs of plots were randomly established at each burn site and nearby control site (Table 3-3; 3-4). However, the Crawford Bench site had restricted access and was limited to two sets of paired plots, consisting of one plot within the disturbed area and one plot in the control area adjacent to the burn perimeter. Plot data and ancillary data were recorded as described in the methods chapter (See Methodology).

Statistical Analyses

Data were checked for normality and transformed using $\log_{10}$ transformations if needed. T-tests were used to compare cryptobiotic crust by stage of development between the burned and paired control plots, while an analysis of variance (ANOVA) was used to test these same parameters to compare the 1997 Buckskin burn, the 1998 Buckskin burn, and their unburned controls.

Results and Discussion

Comparison of Burned Plots to Paired Control Plots

When comparing the 1998 Buckskin Fire (3 year-old burn) site to the paired unburned control, the cover of bare ground was 59% greater and young cryptobiotic crust cover was 42% less for the burned area. Of particular interest, older crusts were significantly less (93% difference) in the burned site (Table 3-3). When comparing the Five Mile Mountain Fire (10-year old burn) area to its neighboring control, it was found that bare ground was significantly reduced by 81% for the burned site. Although percent cover of young cryptobiotic crust was not very different, the difference in cover for older well-developed crusts was 86% less in the burned plots (Table 3-3). For the Crawford Bench Fire (50 year-old burn) and control plots, the cover of bare ground was significantly lower (75% difference) for the burned area. While young crust cover was 42% higher in the burned area than the control, well-developed crust cover was 86% less (Table 3-3).

Table 3-3. Mean percent cover of cryptobiotic crusts on untreated burn sites and controls with standard errors in parentheses and p-values. Significance at α = 0.05 is highlighted by (*).

Indices	n	Bare Ground 0	Young Crusts 1, 2, 4	Old Crusts 6, 10, 15, 20
3 yr old burn	40	28.6 (4.5)	14.0 (3.6)	0.3 (0.2)
Control	40	18.0 (3.5)	24.2 (4.0)	4.2 (1.3)
P value		0.067	0.06	0.005*
10 yr old burn	40	1.2 (0.3)	17.6 (1.3)	2.2 (1.8)
Control	40	6.6 (1.3)	18.7 (2.3)	10.3 (1.4)
P value		<0.001*	0.35	<0.001*
50 yr old burn	20	8.9 (1.5)	26.7 (4.9)	1.0 (0.5)
Control	20	35.0 (5.7)	18.8 (4.4)	7.8 (3.3)
P value		0.005*	0.09	0.14

From these results, the predictive trend analysis suggests that cover of both young and well-developed cryptobiotic crusts will significantly decrease following a fire, and the cover of bare ground will increase (Fig. 3-5). Using data from the 1998 Buckskin fire (3 years following fire), it can be predicted that cover of bare ground will decrease as cryptobiotic crusts begin to redevelop. Fifty years following a fire (Crawford Bench Fire), it appears that young cryptobiotic crust cover remains high compared to unburned controls. Due to the lack of data between the ten and fifty post-burn plots, it is difficult to determine if the levels of cryptobiotic crust cover have stabilized or are continuing to change either positively or negatively.

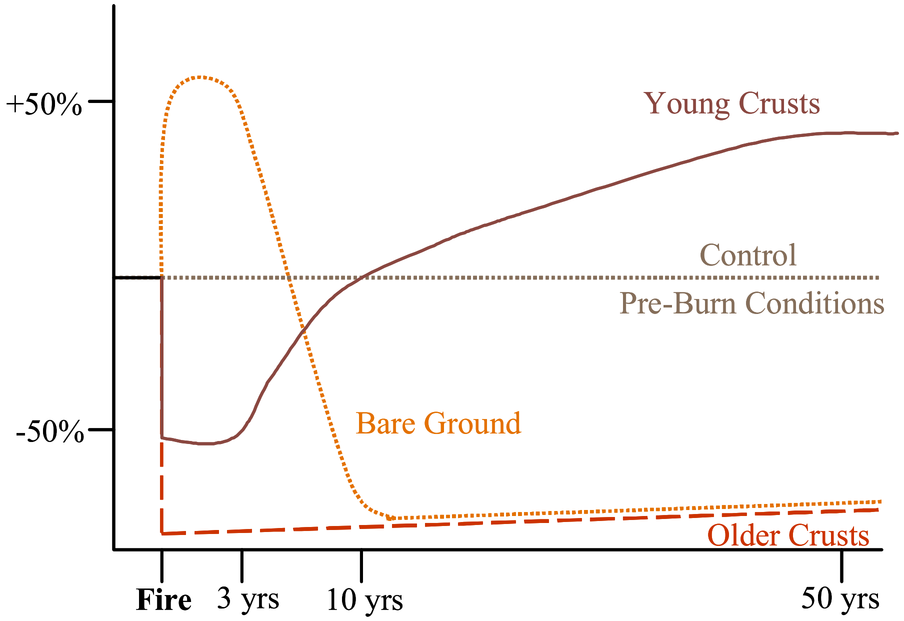

Figure 3-5. Predicted trends of younger and older developed crusts and bare ground following fire.

All the study sites showed little or no sign of regeneration of older, well-developed cryptobiotic crusts. It's possible that older crusts have more exposed pedicles (> 2 cm above soil surface) that are more vulnerable to the extreme temperatures present during a fire. This could potentially result in a greater mortality to these microorganisms. However, field observations have found that older well-developed crusts will keep their structure even if living organisms are absent. The presence of these crust remnants continue to contribute to soil stability and may increase moisture and nutrient retention (Belnap and Gardner 1993).

These findings are consistent with other research studies on crust recovery following fire (Johansen et al. 1982; Johansen et al. 1984; Ponzetti et al. 1999). However, most studies have a short temporal duration considering the extended recovery time needed for most crusts to develop. Therefore, it is essential that long-term studies be conducted to understand the overall dynamics of crust recovery following fire.

Comparison of Buckskin Burn Sites

When comparing the 1997 Buckskin burn (burn/seeded) with the 1998 Buckskin burn (burn/unseeded) and control sites, cover of bare ground was not significantly different (Table 3-4). Young crust cover was not significantly different between the 1998 burn (burn/unseeded) plots and the unburned sites, but crusts were significantly lower (89% difference) in the 1997 burn area (burn/seeded). Well-developed crust cover was significantly lower in both burned sites when compared to the unburned control sites.

These results suggest that mechanical seeding is not appropriate as a restoration tool following fire. This process reduces crust cover and inhibits regeneration of young crusts. Therefore, the beneficial roles of these crusts in this ecosystem are being depleted by such a management action. Loss of crust cover could facilitate additional changes that include non-native weed invasions and shifts in fire regimes. Because many of the planted seeds are species palatable to livestock, an increase in livestock densities may occur which would further disrupt crust development (Beymer and Klopatek 1992) and promote the invasion of non-native plants (Knapp 1996; Stohlgren et al. 2001a).

Table 3-4. Mean percent cover of cryptobiotic crusts on burn/seeded, burn/unseeded, and unburned control sites on Buckskin Mountain. Standard errors are in parentheses and significance of ANOVA tests are signified by supercripts (a,b,c) within columns.

Indices	n	Bare Ground 0	Young Crusts 1, 2, 4	Old Crusts 6, 10, 15, 20
3 yr old burn	40	28.6 (4.5)^a	14.0 (3.6)^a	0.3 (0.2)^a
4 yr old burn and seeded	60	26.6 (1.8)^b	1.6 (0.3)^b	0.3 (0.2)^a
Control	60	12.8 (2.5)^c	13.6 (3.1)^a	1.4 (0.7)^b

Case Study 3: Comparisons of Pigment Levels in Cryptobiotic Soil Crusts

Chlorophyll *a* levels can be used as a surrogate for the biomass of autotrophs (cyanobacteria and green algae), and this can be equated to carbon fixation. Therefore, higher levels of chlorophyll *a* indicate higher levels of productivity (J. Belnap, *pers. comm.*). Other pigments produced by cryptobiotic crusts aid in decreasing mortality from increased UV radiation by avoidance, repair, or protection (Bowker et al. 2002). When the crusts are disturbed or under stress, their ability to produce these pigments becomes compromised.

There were two objectives for this study: 1.) to determine if our visual assessments of cryptobiotic crust development levels (low, med, and high) could be used as an accurate surrogate for assessing productivity, and 2.) to determine if there was a relationship between disturbance stress and the amounts or presence/absence of supplemental pigments found in the cryptobiotic crusts.

Methods

Chlorophyll *a* levels in the cryptobiotic crusts were measured at the USGS laboratory in Moab, Utah for all plots sampled in 1999, 2000, and 2001 using a flourometer method. In addition, nine other pigment levels (chlorophyll *b*, scytonemin, myxoxanthopyll, violaxanthin, lutein, canthaxanthin, echinenone, beta-carotene, and pheophytin) were measured for only a sub-set of plots in 2000 and 2001 using High Performance Liquid Chromatography (HPLC). Only selected plots received the HPLC analysis due to the higher cost. See Bowker et al. (2002) for complete methodology.

Variables that were examined for each set of data included the following (if possible due to adequate sample sizes and/or pigment analyses performed): 1.) amount of chlorophyll *a* (mg/g of soil sample), 2.) total amount of pigments (mg of all pigments added together/g of soil), and 3.) presence/absence of pigments in the soil sample (a plot could have a maximum score of 27, since we examined 9 pigments at three different cryptobiotic crust development levels).

Four subsets of data from this project were examined: 1.) All plots from 1999, 2000, and 2001 with chlorophyll *a* data (n=174), 2.) Plots from the grazed (Deer Spring Point) versus near-relict site (No Man's Mesa) study (n=10), 3.) Buckskin Mountain burn area plots (n=32), and 4.) Skutumpah area disturbed sites (n=17, seeded plots were not included due to small sample sizes).

Results and Discussion

For all of the plots for which we had chlorophyll *a* data, we ran an analysis of variance (ANOVA) to determine if there were significant differences in the amount of chlorophyll *a* in each of the cryptobiotic crust collection categories (low = development levels 1 and 2, medium = development levels 4, 6, and 10, and high = development levels 10 and 15). A graph of chlorophyll *a* pigment levels by crust development stage demonstrated that there were indeed significant differences between the collection categories (P = <0.001, F = 15.1; Fig. 3-6). This suggests that we can visually assess biomass and carbon fixation of cryptobiotic crusts by crust development level. Because cryptobiotic crusts are often the only source of nutrients in some areas of the Monument, this finding has important implications.

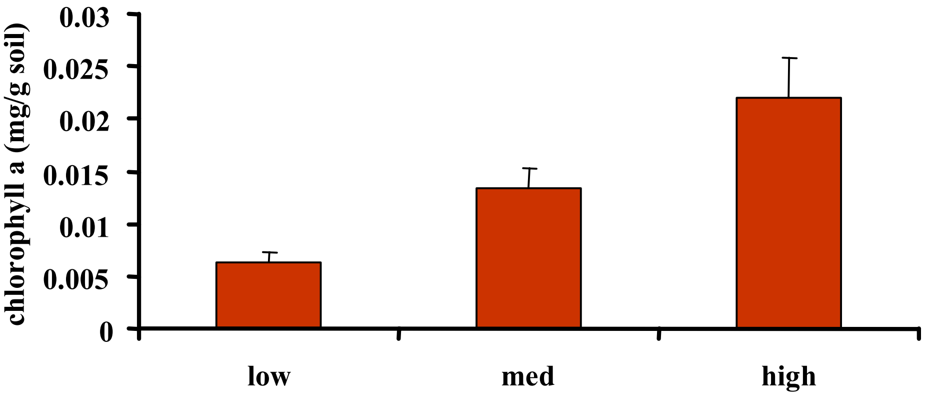

Figure 3-6. Chlorophyll a pigment levels by crust development level low (n=174), medium (n=113), and high (n=77).

We used a t-test to compare No Man's Mesa (a near-relict site, n=5) to Deer Spring Point (a grazed site, n=5), and there were no significant differences in chlorophyll *a* (p=0.51, all three levels summed) or in the total amount of pigments (p=0.62). There was also no significant difference in the presence or absence of pigments between the two sites (p = 0.75). Thus, at Deer Spring Point, grazing did not appear to greatly damage the cryptobiotic crust pigments.

At the Buckskin Mountain burn area, t-tests were used to examine the difference between burned sites (n= 21) and adjacent unburned control sites (n=11). Note that burned and seeded sites were included all together as burned sites; consequently the additional effects of seeding were not examined. A significant difference was found between the amount of chlorophyll *a* in the low cryptobiotic crust development level of the burn versus control plots (p-value = 0.02), with three times the amount of chlorophyll *a* found in the controls (mean = 0.009 mg chl *a*/g soil, SE = 0.002) than in the burned areas (mean = 0.0035 mg chl *a*/g soil, SE = 0.001). There was no significant difference in the amount of chlorophyll *a* in the medium development category (p=0.93) and not enough samples to make a comparison for the high development category. These data support our hypothesis that fires greatly damage cryptobiotic crust and cause a slow recovery. There were not enough samples to compare the other pigments for this area.

In the Skutumpah area disturbed sites (burned n=6, and controls n=8) there was no significant difference in the amount of chlorophyll *a* (all three levels summed) at the alpha = 0.05 level (p=0.08) using a t-test. However there was more than twice the amount in the control areas (mean = 0.016 mg chl *a*/g soil, SE = 0.003) as the burned areas (mean = 0.007 mg chl *a*/g soil, SE = 0.004). There was no significant difference in the total amount of pigments between the two treatments (p = 0.23). When examining the presence/ absence of pigments, there was no statistically significant difference for the number of pigments in a plot (p=0.12) due to large standard errors, but on average twice as many pigments occurred in the control plots (mean = 14.0, SE = 2.5) than in the burn plots (mean = 7.2, SE = 3.2). In this area of the Monument, fire also appears to be very damaging to the supplemental pigments cryptobiotic crusts produce.

This analysis of the cryptobiotic crust pigment data is only preliminary and more research is needed to determine the effects of disturbance on the levels of these pigments. Future analyses might include examining the effects of seeding on the pigments and looking at the order in which pigments decrease and increase in a disturbance gradient.

Management Implications

Determining soil characteristics that are strongly correlated to non-native species could be beneficial to land managers when predicting non-native species establishment and dominance within the Monument. For case study 1, it was determined that habitats with greater resource availability and greater diversity were at greatest risk. Habitats containing fertile soils appear most vulnerable to non-native plant species invasions than those with less-fertile soils. Thus, a shift in soil conditions brought about by nutrient inputs could shift the distribution of native and non-native plant species locally. Rare vegetation types (aspen, wet meadow, riparian) appear more vulnerable than common vegetation types, suggesting that management should focus on the protection of these habitats types within the Monument because of their rarity and high biological diversity. Heavily disturbed areas such as roadsides, burned areas, and trampled sites have also been shown to be vulnerable to invasions. Where soil fertility is high, disturbance may greatly enhance the invasion process. In this case, the disturbance would provide available space and nutrients to the non-native invader. Ultimately, management actions may govern the success or failure of non-native species in the Monument.

Of great concern to managers should be the negative effects that fire and especially post-burn mechanical seeding appears to be having on cryptobiotic crusts. Because the Monument's management plan explicitly mandates the protection of these crusts by minimizing negative impacts on their function, health, and distribution (Grand Staircase-Escalante National Monument 2000), rehabilitation efforts following fire need to be carefully considered and their impacts known. Further studies of additional burn and seeded sites need to examine 1) different spatial variations such as steeper slopes and various aspects, 2) alternative rehabilitation techniques such as aerial seeding, and 3) the additional impacts of domestic livestock grazing.

CHAPTER 4

Non-Native Plant Species

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Invasion by non-native plant species has adversely affected many ecosystems within the United States, which has made it one of the most significant issues in natural resource management and conservation biology (Wilcove et al. 1998; Williams and Meffe 1999; Mack et al. 2000). They pose a significant threat to wilderness and other protected lands by their direct and indirect impacts on native species and by their effects on ecosystem processes such as food webs, decomposition cycles, hydrologic cycles, nutrient cycling, and natural disturbance regimes (Ingham et al. 1989; Cole and Landres 1996; Belnap and Phillips 2001). The resulting changes in ecosystem function created by one non-native species may then result in the invasion of more non-native plant species (Parker and Reichard 1998). As these transformations continue, native species could be negatively affected and the stability of the ecosystem would most likely decline (Rosentreter 1994). Therefore, research pertaining to non-native species and their effects on native species and ecosystem function are essential in finding ways to lessen their impact.

Invasion of Different Vegetation Types within the Monument

Uncertainty often arises in regards to the factors that make a vegetation type at high risk to invasion. Ecosystems are thought to be able to combat invasion by non-native species depending on their levels of species richness and stability following disturbance (Simberloff 1986). Biotic resistance has been thought to be correlated with highly disturbed and species-poor communities (Elton 1958). However, this theory has been reputed recently as new research has found a higher risk of invasion in highly diverse vegetation communities with intermediate levels of disturbance, such as tallgrass prairies, wet meadows, and riparian zones (Robinson et al. 1995; Planty-Tabacchi et al. 1996; Wiser et al. 1996; Stohlgren et al. 1999c; Stohlgren et al. 2001).

There are various mechanisms that make species-rich vegetation types more invasible than species-poor vegetation types. Many species-poor habitats have few dominant species that are good competitors. Therefore, non-natives may not be able to establish in these habitats due to limited resources (Parker and Reichard 1998). On the contrary, species-rich vegetation types may have a high availability of nutrients following a disturbance that may be made available to non-native species (Robinson et al. 1995). However, establishment of non-natives is usually dependent on factors related to the disturbance site such as availability within the seed bank (Roberts 1994).

Both natural and anthropogenic disturbance are associated with the invasibility of a vegetation type (Hobbs 1989; Hobbs and Huenneke 1992). Therefore, species-rich habitats may be more invasible due to their frequency of disturbance. Highly diverse vegetation types usually consist of a variety of successional stages. That is, there are patches throughout the site containing different stages of disturbance, succession, and maturation (Hobbs and Huenneke 1992). As sites within the vegetation type become available through disturbance, the establishment of a non-native species may be possible.

Non-Native Species in the Monument

Cheatgrass

One of the greatest ecological threats to the Monument is the spread of non-native invasive plant species, most notably annual grasses (Davidson and Belnap 1999; Belnap et al. 2000). These species may be toxic to livestock and wildlife and have been shown to alter habitats by displacing native plant species throughout their range (Harper et al. 1996). In some areas of the Monument, non-native annual grasses have become locally dominant, representing as much as 85% of plant cover (Belnap et al. 2000).

Most notable is cheatgrass, an invasive grass species native to Eurasia and the Mediterranean that was found to be the most dominant non-native species within the Monument. Cheatgrass has spread rapidly since its introduction to the United States in the late 1800s and early 1900s and now dominates plant communities in disturbed shrub-dominated steppe ecosystems (Upadhaya et al. 1986; Link et al. 1995; Whitson et al. 1996), covering approximately 40 million ha in the Intermountain West by the late 1970s (Mack 1981). Disturbed areas typically invaded by cheatgrass include burned rangelands, cultivated crop areas, roadsides, waste areas, and abandoned fields (Upadhaya et al. 1986; Whitson et al. 1996), but any type of disturbance can potentially give this and other brome grasses a competitive advantage (Harper et al. 1996; Davidson and Belnap 1999).

Pristine shrub-steppe ecosystems are typically dominated by long-lived perennial bunchgrasses and shrubs (Young and Evans 1985; Whisenant 1990). However, shifts in vegetation have occurred because of disturbance and the introduction of non-native annual grasses into this ecosystem. Because of its strong competitive ability, cheatgrass has replaced seedlings of many native perennial grasses and shrubs (Aguirre and Johnson 1991). Cheatgrass's competitive ability stems from a number of factors, including its adaptation to fire (Melgoza et al. 1990), grazing tolerance, and the rapid development of its root structure that allows it to exploit water and nutrients before other species become established (Skipper et al. 1996).

Cheatgrass becomes dry and highly flammable when it senesces, resulting in increased fuel loads (West 1983). The fire return interval of a natural sagebrush grassland ecosystem is between 60 and 100 years. However, some areas that have become invaded by cheatgrass burn on average every three to five years (Whisenant 1990). In addition, cheatgrass seedlings respond positively to fire and are able to take up nutrients released by the disturbance. As more cheatgrass individuals become established, fires become even more frequent and cheatgrass and other annuals increase in dominance (Wright and Klemmedson 1965; Young et al. 1969; West 1983). This positive feedback cycle can result in monocultures of few species instead of the heterogeneous landscape representative of the natural shrub-steppe community (Whisenant 1990; Rosentreter 1994).

Grazing is another common disturbance in the shrub-steppe ecosystem, and cheatgrass is better able to withstand extensive grazing than native perennial grasses. Cheatgrass grazed to just 5-10 cm can still flower and produce viable seed (Pyke 1987). In addition, many grazers prefer native plant species to non-native plant species, so natives tend to experience higher grazing pressures compared to non-natives (Pyke 1987).

Nonetheless, the most important way that cheatgrass competes with other range grasses is through its root system. Because germination occurs in the fall, cheatgrass is able to continue growth of its root system through the winter months (Davidson and Belnap 1999). Most native species in the Monument germinate in spring after cheatgrass has already had time to develop (Young and Evans 1985). The early establishment of its root system allows cheatgrass to acquire water and nutrient resources earlier than other non-established perennials. Because water is a limiting factor for growth in arid and semi-arid environments (Barbour et al. 1987), removal of water from the upper soil profile by cheatgrass can result in plant death or a reduction in reproduction success for native species (Melgoza et al. 1990). Cheatgrass then starts to dominate the area and may even allow for the invasion of other non-native plant species such as knapweed and thistle.

Redstem Storksbill

Redstem storksbill is common worldwide, but was originally found in Eurasia (Whitson et al. 1996). The species was introduced to the United States in the early 1700s and is currently found in all states excluding Florida and Louisiana (Webb et al. 1988). It was the second most common species found within the Monument and occurs in a variety of vegetation types ranging from desert to riparian (Juhren et al. 1956; Kauffman et al. 1983). Within riparian vegetation types, it indicates recent or frequent disturbances (Lisle 1989). The species is also found on other disturbed sites and was found to be the first species to emerge on lands mined for coal in New Mexico (Wagner et al. 1978). Storksbill provides forage for livestock and wildlife (Dittberner and Olson 1983; Whitson et al. 1996) and is resilient to heavy grazing pressure (Humphrey 1950). The plant reproduces sexually and its seeds can remain viable for many years, forming an extensive seed bank. Seeds either fall beneath the parent plant or are dispersed by animals. The seed is capable of burying itself with its coiled style which can expand and contract (Felger 1990).

Russian Thistle

Russian thistle was introduced to the United States in the late 1800s from Russia (Welsh 1993). Since that time, it has become one of the most common weeds in the western United States and was found to be the third most common non-native plant species within the Monument. This species has traits that make it capable of surviving in harsh, arid conditions (Whitson et al. 1996). It can germinate rapidly and establish following sporadic rain events which are commonplace throughout the Monument. It is extremely drought resistant, salt resistant, and is found on nearly all soil types (USU Extension 2002). The species grows best on high, dry land if competition is minimal. Therefore, it does not compete well with other plant species (USU Extension 2002). Once the plant has matured, it will break off as a tumbleweed to be carried by the wind, aiding in seed dispersal. This species has adapted to cultivated dryland agriculture and can be found growing vigorously in dryland grain fields (Welsh 1993; Whitson et al. 1996). It can also be found growing on overgrazed rangelands and disturbed wastelands (Whitson et al. 1996). The species is generally considered a nuisance due to its ability to fill ravines, clog irrigation ditches, and cause other related problems (Welsh 1993). Although this species provides adequate forage for livestock in the early spring, it becomes unpalatable as it matures because of the sharp-pointed leaves it produces (USU Extension 2002).

Tamarisk

Tamarisk (saltcedar) was introduced to the United States from Eurasia in the early 1800s for use as an ornamental (Whitson et al. 1996). Species of this genus were notable for their ability to withstand drought, heat, and diverse soil conditions (Carleton 1914; Di Tomaso 1998). Since the introduction, tamarisk species have escaped cultivation, hybridized, and become naturalized and widespread along riparian areas throughout the western United States (Christensen 1962; Whitson et al. 1996), being especially pervasive throughout Arizona, New Mexico, western Texas, Nevada, and Utah (Westbrooks 1998; Zavaleta 2000). Tamarisk infestations have increased from an estimated 4,000 ha in 1920 to over 500,000 ha by the mid 1960’s (Robinson 1965). The rapid increase in infested land likely stems from increased stream flow regulation following large dam and water diversion construction projects (Everitt 1980; Everitt 1998).

The proliferation and success of the tamarisk invasion may be due, in part, to its’ unique life history characteristics. An individual tamarisk tree can generate up to 600,000 seeds annually, which can be dispersed by wind or water (Robinson 1958). The species also has the ability to vegetatively reproduce by sprouting from branches if adequate conditions exist (Horton 1977). The species is an obligate phreatophyte (needs constant contact with water) and will extend its root system to depths of 50 m to take advantage of water supplies (Blackburn et al. 1982; Pinay et al. 1992). Because tamarisk has a greater tolerance of saline soils, the species will assimilate salts into its roots and excrete them on the surface of the soil. By doing this, high concentrations of salts are accumulated and native species can be negatively affected (Carman and Brotherson 1982; Di Tomaso 1998). Tamarisk is able to withstand a variety of disturbances, making its control difficult (Smith et al. 1998). Thus, methods typically used to control other species (i.e., fire, drought, flooding, cutting) are not successful at controlling tamarisk.

Given the widespread nature and success of tamarisk, it is not surprising that it was found to be the fourth most common species within the Monument. However, a 1979 study in the Escalante River basin (Irvine and West 1979) found that tamarisk populations were confined to within canyon reaches where large boulders provided adequate protection from forceful flooding events or on higher terraces susceptible to only occasional flooding. These patterns will likely increase invasion of these and neighboring sites if flood frequency and stream flow decreases with added impoundments (Irvine and West 1979).

Yellow Salsify

Although the species originated in Eurasia, yellow salsify is a widespread weed throughout temperate North America and is usually found along roadsides and waste sites (Whitson et al. 1996). It was also the fifth most common species found within the Monument. The species is adaptable to many habitats because of its ability to grow in fine, medium, and coarse soils. It is also drought resistant and can grow in areas of low soil fertility. The species is intolerant to fire and unpalatable to livestock (USDA 2003). Yellow salsify is commonly referred to as “goat’s beard” due to the globe of fluffy white pappus bristles attached to the achenes. The pappus bristles allow for wide-ranging wind dispersal, making it one of the most frequent non-native species found across the Monument.

Common Dandelion

Although originally from Eurasia, the common dandelion has become naturalized throughout the United States, occupying all 50 states (United States Department of Agriculture 1971). It is typically found in moist habitats throughout North America, including lawns, meadows, and pastures. The species can spread its seeds considerable distances because of the parachute of bristles that are attached to each seed (Holmgren 1958), allowing for widespread distribution. Common dandelion also has a long-lived seedbank that responds positively to disturbance (Ahlgren 1979). Once the species is established within a disturbed site, it has the potential to gain dominance on that site within two to three years (Bedunah et al. 1988). Because the species is also good forage for livestock and wildlife (Whitson et al. 1996), it is commonly found on overgrazed rangelands (Bergen et al. 1990). Other disturbances such as clearcuts, thinning of forests, and fire have also been known to stimulate the production of this species (Dittberner and Olson 1983).

Kentucky Bluegrass

Although Kentucky bluegrass is generally considered non-native to North America, it has become naturalized throughout many regions of the United States and currently grows in every state and Canadian province (Wheeler and Hill 1957). In parts of the West, the species can occur as an understory dominant in aspen, ponderosa pine, wet meadow, grassland, and riparian vegetation types. For livestock in these areas, the species is highly palatable in early growth stages and provides nutritious forage (Clary 1975; Kauffman et al. 1983; Bowns and Bagley 1986). However, the species is able to witstand heavy grazing and increases rapidly on overgrazed rangelands. Elk, mule deer, and bighorn sheep also use this species as forage (Dittberner and Olson 1983). However, a study within aspen/Kentucky bluegrass community types in Utah found that these areas were rarely used as wildlife habitat because of a lack in plant species diversity (Mueggler and Campbell 1986). Kentucky bluegrass has characteristics that make is highly competitive, including distribution by rhizome expansion, the production of abundant seed, and good seedling recruitment and establishment on disturbed sites. If the species is able to gain dominance, it is persistent and remains a stable component of the system it has invaded (Uchytil 1993).

Case Studies

Case Study 1: *Cheatgrass Biomass Estimates*

The objective of this case study was to determine where cheatgrass biomass was greatest and to determine if disturbance increased its establishment and dominance. It was hypothesized that cheatgrass biomass would be greatest in the more mesic vegetation types because of increased water availability and within the highly disturbed vegetation type because of its ability to tolerate disturbances compared to native plant species (Waters 2003).

Methods

The cheatgrass study included a subset of the plots (N=309) sampled between 1998 and 2001 and included 15 vegetation types (Fig. 4-1). Because productivity is correlated with various climatic factors, specifically water in arid environments (Barbour et al. 1987), an analysis of variance (ANOVA) was performed ($p < 0.05$) to compare climatic data over the four years that the vegetation was sampled (See Appendix D).

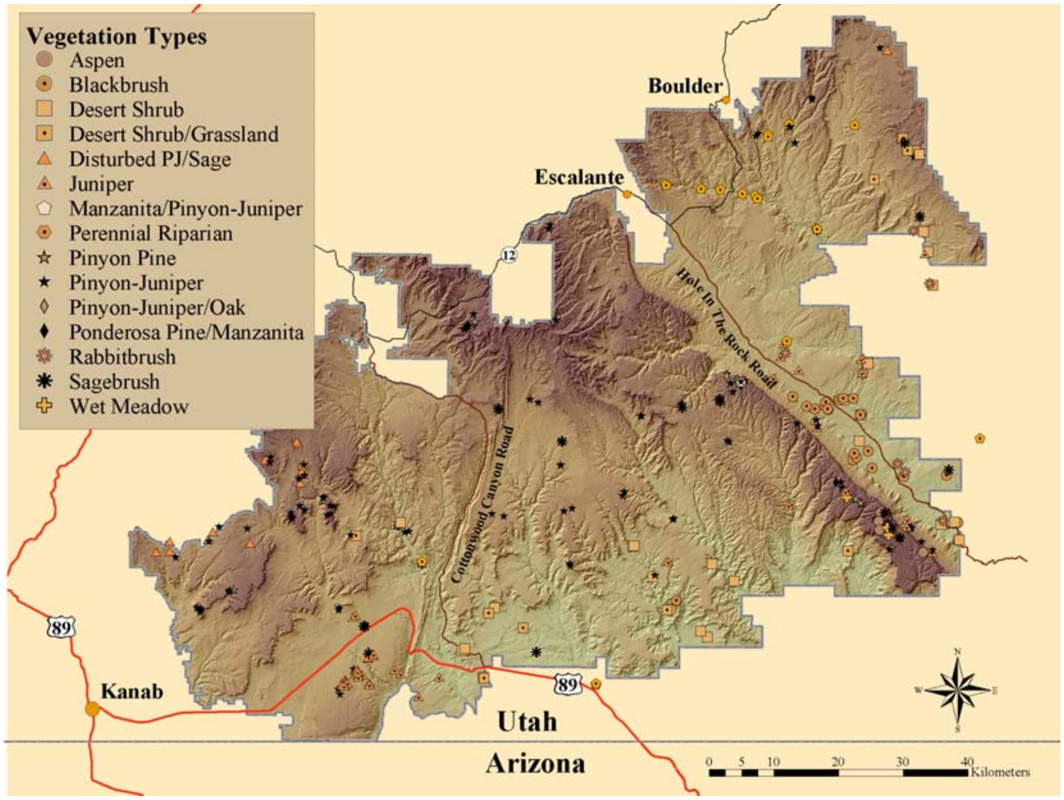

Figure 4-1. Study area for case study 1, including plot locations by vegetation type.

In May, 2002, cheatgrass samples were collected to estimate its aboveground biomass within the 309 plots. Circular 0.25-m^2 plots were placed within two burned areas of the Monument where cheatgrass density was considered highest. Being a relatively dry year, plots were subjectively chosen to ensure finding a variety of cheatgrass densities and heights. For each circular plot (n = 42), we recorded cheatgrass cover and height and clipped aboveground biomass. The biomass clippings were later dried in an oven at 40°C for 48 h and then weighed. Aboveground biomass was estimated as grams per m^2.

To extrapolate biomass estimates to all of the vegetation types, simple linear regressions were plotted for cheatgrass biomass on height, cover, and height multiplied by cover. This non-destructive method of biomass estimation is generally more reliable than other methods, including remote sensing (Tucker 1980). Due to the natural variability in all these parameters, a $\log_{10}$ transformation (x + 1) was taken. Findings of the simple linear regression analysis were then used to extrapolate biomass estimates using the height and cover measurements for all 1-m^2 subplots within the 309 plots using the best correlation equation. To compare cheatgrass biomass among vegetation types, an ANOVA was performed.

It is important to note that the cheatgrass biomass samples were collected during a dry year. It has been shown that non-native annual vegetation is more susceptible to drought than native perennial vegetation (Stewart and Young 1939; Rosentreter 1994). More specifically, perennials have demonstrated twice the biomass production of cheatgrass in a moist year and twelve times the biomass production in a drought year (Stewart and Young 1939). Therefore, biomass estimates are most likely below those of a year with normal precipitation. Although collection of more samples might provide a better estimate for biomass in an average year, the biomass regression captured the entire range of cheatgrass cover and height that was found in the 309 vegetation plots.

Results and Discussion

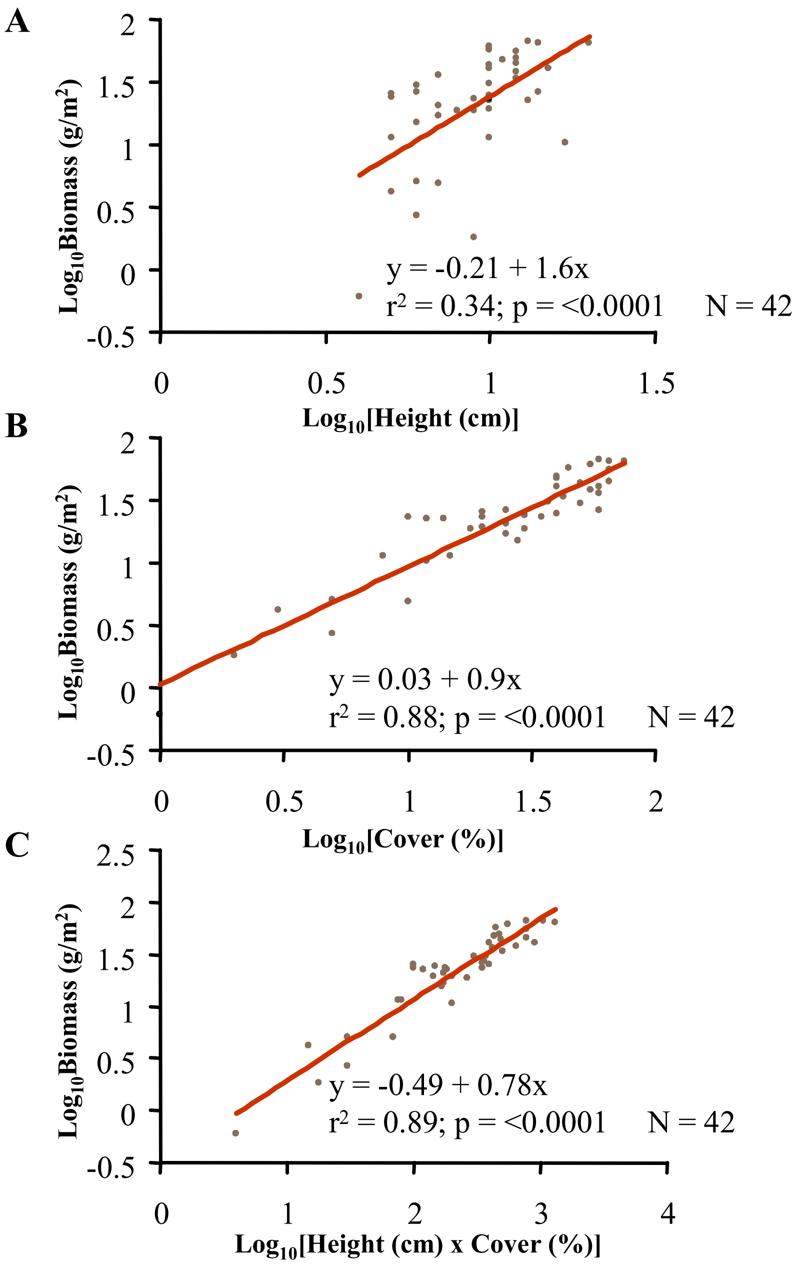

Figure 4-2. Correlation of cheatgrass biomass to height (A), cover (B), and height multiplied by cover (C).

The regression of biomass on height, biomass on cover, and biomass on height by cover for cheatgrass were all highly significant ($p < 0.05$) with R^2 values of 0.34, 0.88, and 0.89, respectively (Fig. 4-2). The strongest correlation equation from the regression of biomass on height by cover (y = -0.49 + 0.78x) was used for the extrapolation of biomass to all subplots within all vegetation types (Table 4-1).

Table 4-1. Average cheatgrass biomass in each of 15 vegetation types.

Vegetation Type	Biomass (g/m^2)
Aspen	19.1
Sagebrush	7.7
Rabbitbrush	7.6
Wet Meadow	7.0
Disturbed Pinyon-Juniper	5.6
Juniper	4.4
Desert Shrub/Grassland	4.2
Perrenial Riparian	3.7
Blackbrush	3.2
Ponderosa Pine/Manzanita	3.1
Desert Shrub	2.8
Pinyon Pine	2.2
Pinyon-Juniper	1.4
Pinyon-Juniper/Oak	1.0
Pinyon-Juniper/Manzanita	0.02

Because cheatgrass has invaded and become dominant in many regions of the Intermountain West, information related to vegetation types most susceptible to invasion by this species is important. Within the Monument, cheatgrass showed high biomass levels in the sagebrush vegetation type and in the more mesic vegetation types.

The sagebrush vegetation type had significantly greater cheatgrass biomass than the other xeric vegetation types. This was contradictory to the hypothesis, which stated that the disturbed pinyon-juniper vegetation type would have the highest biomass for cheatgrass in the xeric habitats. When compared to the pinyon-juniper vegetation type, however, cheatgrass biomass was five times greater in the disturbed type. Cheatgrass is known to dominate plant vegetation types in disturbed shrub-dominated steppe ecosystems (Upadhaya et al. 1986; Link et al. 1995; Whitson et al. 1996), and this finding further supports the idea of natural and anthropogenic disturbance increasing establishment and persistence of non-native species (Fox and Fox 1986; Hobbs 1989; Hobbs and Huenneke 1992). As patches within a vegetation type become available through disturbance, the establishment of a non-native may be possible because of open space and increased nutrient availability (Robinson et al. 1995).

The reason for the lack of a stronger cheatgrass presence within the disturbed pinyon-juniper vegetation type may be a consequence of these plots' successional stage. There have been some contradictions in the research related to cheatgrass persistence following disturbance (Daubenmire 1970; Young and Evans 1973; McLendon and Redente 1991). One study conducted in Colorado has found that cheatgrass will establish within a few years following disturbance, remain dominant for another few years, but will begin to lose its dominance as succession progresses (McLendon and Redente 1991). More information related to the time of last unnatural disturbance for these plots would be needed to determine if this is the reason for the lack of cheatgrass dominance.

It is not unexpected that high levels of cheatgrass biomass were found in the sagebrush vegetation type. Cheatgrass has been shown to favor areas with variations in plant litter and microtopography (Young and Evans 1973). Sagebrush creates such favorable microsites by capturing moisture, providing shade, decreasing wind speed, and recycling nutrients and water at lower depths of the soil profile than other native plant species (Murray 1975; Rosentreter and Jorgenson 1986; Caldwell and Richards 1989; Pierson and Wight 1991). Sagebrush has specifically been shown to minimize soil temperature variability when compared to interspaces, with the temperature range at 1 cm under sagebrush being comparable to that of 10 cm under the interspaces (Pierson and Wight 1991).

As we hypothesized, average cheatgrass biomass was greater in the mesic vegetation types than the xeric vegetation types. In addition, all the mesic vegetation types had a substantial representation of biomass excluding the perennial riparian type (3.7 g/m^2). This may be a result of this species life traits. Cheatgrass has been shown to decrease native species richness through belowground competition, acquiring water and nutrients before these species have had sufficient time to develop (Young and Evans 1985; Davidson and Belnap 1999). Because water is a limiting factor for growth in arid and semi-arid environments (Barbour et al. 1987), removal of water from the upper soil profile by this species can result in plant death or a reduction in reproductive success for native species (Melgoza et al. 1990). Within this vegetation type, there may be enough moisture that this competitive advantage of cheatgrass is no longer advantageous to the species. Bilodeau (2001) found that cheatgrass dominance was dependent on the availability of moisture throughout the summer months. If enough moisture was available at this time, cheatgrass was found to only decrease the biomass of native perennials but not prevent their establishment.

Suprisingly, cheatgrass biomass was significantly greater in the aspen vegetation type when compared to other vegetation types. There has been mention of observations of this species migrating into higher elevation aspen stands (Young et al. 1987), but scientific literature on such an invasion is lacking. However, some studies have shown a high degree of non-native invasion into aspen and other highly diverse vegetation types (Stohlgren et al. 1998a, 1999b, 2002). One such study specifically looked at aspen stands in the Rocky Mountains, and found that 90% of all non-native species recorded in the sampled area were found in aspen stands and that 36% were found only in the aspen vegetation type (Chong et al. 2001). Because aspen communities are important to the maintenance of biodiversity in western landscapes (DeByle and Winokur 1985), the potential impact posed by cheatgrass may become a high management concern for these areas in the near future. Further research should try to identify possible effects that this species is having and could potentially have on aspen stands within the Monument.

Case Study 2: *Habitat Matching in Invading Non-Native Plant Species*

Various studies have sought to quantify the environmental differences that make a vegetation type at high risk to invasion by non-native species when compared to other vegetation types (Robinson et al. 1995; Burke and Grime 1996; Stohlgren et al. 1998b; D'Antonio et al. 1999; Levine and D'Antonio 1999; Rejmanek 1999; Tilman 1999; Lee 2001; Stohlgren 2002). We sought to quantify and describe invasion patterns in a different way than these earlier studies by examining the life history strategies of native and non-native species (*unpublished data*). The potential role of species life history traits in helping define patterns of non-native species invasions has been inconclusive. An existing paradigm suggests that non-native invasive species are more likely to invade areas that have species with dissimilar life traits, thereby sequestering resources that are underutilized (Darwin 1859; Elton 1958; Rejmánek 1989; Vitousek 1990; Chapin et al. 1996; Mack and D'Antonio 1998). Therefore, competitive exclusion would be a controlling force keeping species with similar habitat requirements from establishing in the same areas.

The objective of this study was to quantify patterns of native and non-native species and cover in two large areas (i.e., montane and desert biome) with differing proportions of annual/perennial species to provide insights on broad-scale patterns of invasion by life cycle. We hypothesized that non-native annual and perennial plant species will successfully establish and accumulate foliar cover in sites with high native species richness and cover with similar life cycles and that competitive exclusion will not be a primary driving force. We termed this the habitat matching hypothesis (Stohlgren et al. *in progress*).

Methods

Additional Study Sites

To compare plots within the Monument (desert biome) to that of a montane biome, plots were located within Rocky Mountain National Park and adjacent sites in Colorado (Fig. 4-3). Rocky Mountain National Park is located in north-central Colorado and ranges in elevation from 2,500 to 4,345 m. The Park receives, on average, approximately 50 mm more rain annually (356 mm) than the Monument.

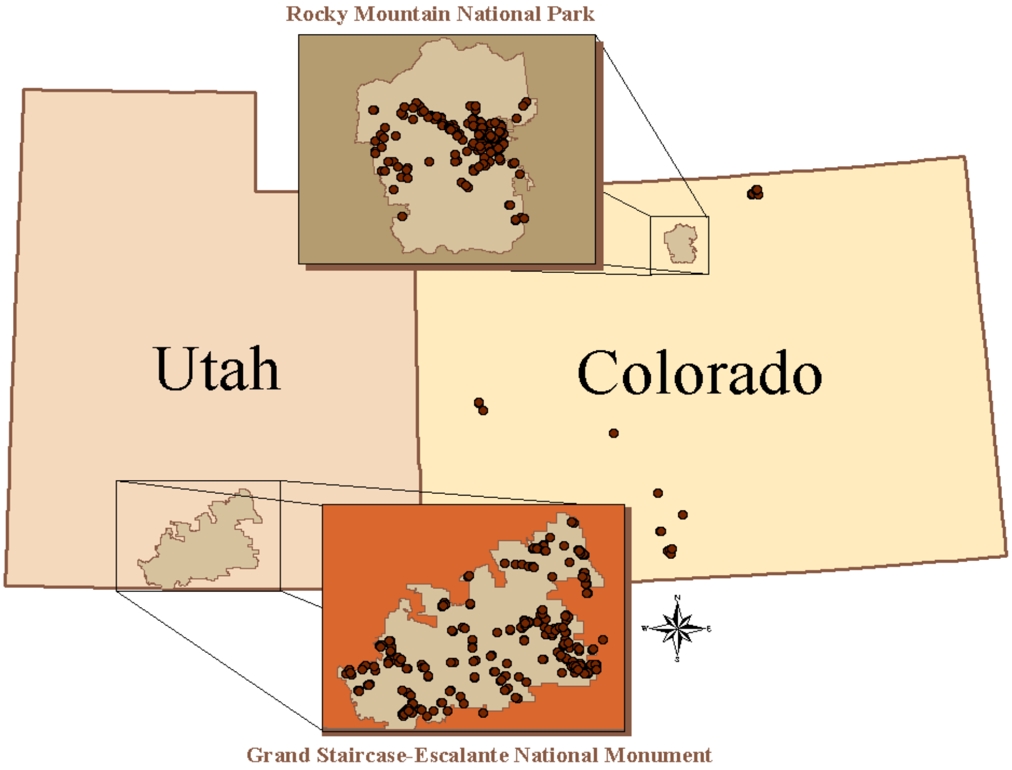

Figure 4-3. Study area for case study 2, including the locations of each plot within Colorado and Utah.

Vegetation sampling in Colorado was conducted between 1994 and 2002 in the same manner as described in the methods section of this report (See Methodology). The Park contained 181 of the total 292 plots in Colorado, with the other 112 plots in mixed conifer forests and mid- to high-elevation shrublands and forests along the Colorado Front Range. Several vegetation types were recognized within this study area, including wet meadow, dry meadow, lodgepole pine, aspen, ponderosa pine, mixed conifer, spruce-fir, forest ecotone combinations thereof, subalpine forests and alpine tundra.

Field Sampling

A total of 593 modified-Whittaker plots were used in this analysis; 301 in the Monument and 292 in Colorado. Plant species in Colorado were identified in the field following Weber and Wittmann (1992). Unidentifiable specimens were carefully collected for later identification or verification at the Colorado State University herbarium. Species origin and life cycle were determined by the USDA PLANTS database (USDA 1998).

Statistical Analyses

All variables were assessed for normality and $\log_{10}$ transformations were used for all statistical analyses (Systat Software Inc. 2001; Table 4-2). The relationships of four independent variables (i.e., number and cover of native annual and perennial species) and each dependent variable (i.e., number or cover of non-native annual or perennial species) were evaluated for Colorado sites, Utah sites, and all sites combined using stepwise forward linear multiple regression. These models included only variables meeting the default $P < 0.15$ criterion **(**Systat Software Inc. 2001).

Table 4-2. Characteristics of vegetation in the plots in Colorado and Utah. Mean values are in bold, standard errors are in parentheses, and maximum values are in italics. T-test P values are on log-transformed data.

Characteristic	Colorado Sites	Utah Sites	t-test P
Number of Plots	292	301	
Number of Native Annuals	**2.9** (0.2) *18*	**3.3** (0.6) *13*	**0.06**
Cover of Native Annuals (%)	**0.8** (0.1) *18.5*	**3.9** (0.4) *42*	**<0.001**
Number of Native Perennials	**26.5** (0.7) *62*	**14.6** (0.4) *36*	**<0.001**
Cover of Native Perennials (%)	**39.3** (1.6) *160*	**24.2** (1.0) *82*	**<0.001**
Number of Non-Native Annuals	**1.9** (0.2) *16*	**1.6** (0.1) *7*	**0.003**
Cover of Non-Native Annuals (%)	**3.2** (0.5) *72*	**2.9** (0.3) *42*	**0.009**
Number of Non-Native Perennials	**1.1** (0.1) *16*	**0.4** (0.05) *5*	**<0.001**
Cover of Non-Native Perennials (%)	**1.2** (0.2) *32*	**1.0** (0.2) *32*	**<0.088**
Ratio of Native Annual to Perennial Species Richness	**0.15** (0.01)	**0.32** (0.03)	**<0.001**
Ratio of Native Annual to Perennial Species Cover	**0.13** (0.07)	**0.57** (0.1)	**<0.001**

Results and Discussion

As the habitat matching hypothesis suggests, it may be possible to predict patterns of non-native plant species by identifying habitats occupied by native plant species with similar life cycles (Table 4-3). From the results of this study, it is apparent that competition from species with similar environmental needs is not affecting non-native species establishment and persistence and that habitat matching by non-native species may prove to be a more important process in invasion ecology.

Quantifying the richness and cover of native annual and perennial species contributed greatly to understanding patterns of non-native plant invasions. Stepwise regression models were able to explain 32% to 81% of the variation in non-native annual species richness and cover for all site data combined (Table 4-3). Within Utah, 95% of the variation in non-native annual species cover could be explained by the very strong positive relationships with native annual species cover. In addition, the richness of native perennial species per plot generally was negatively correlated to the richness and cover of non-native annual species, but generally a positive predictor of non-native perennial species (Table 4-3).

Non-native perennial species richness and cover could not be as easily predicted by native annual and perennial species richness and cover ($R^2 < 0.11$). For the Utah sites, multivariate models to predict the cover of non-native perennial species based on the richness and cover of native annuals and perennials were not significant (Table 4-3). For all sites and only the Colorado sites, multivariate models to predict non-native perennial richness based on cover of native perennials was not significant and models to predict cover of non-native perennials based on native annual richness and native perennial cover were not significant (Table 4-3).

The ability to predict patterns of non-native annual species better than perennial species may be related to the predictability of habitats dominated by annual species (high disturbance, light, and resource availability; Harper 1977; Barbour et al. 1987). Perennials colonize areas with less predictable environmental characteristics and a broader range of habitats from close-canopied forest to open-canopied tundra. In addition, competition for resources might be less for annual species than for perennial species, or the native and non-native species might have different resource use patterns. Thus, it appears that competition from high foliar cover of native annual and perennial species is not inhibiting invasion by non-native plant species from our results (Stohlgren 2002).

Table 4-3. Standardized partial regression coefficients and model statistics for stepwise multiple regressions of four independent variables (number and cover of native annual and perennial species) and each dependent variable (number or cover of non-native annual or perennial species) for all sites combined, Colorado sites, and Utah sites.

All Sites/Dependent Variable	Number of Native Annuals	Cover of Native Annuals	Number of Native Perennials	Cover of Native Perennials	Model R^2, P
Number of Non-Native Annuals	0.33	0.29	-0.25	0.10	0.32, <0.001
Cover of Non-Native Annuals	0.06	0.62	-0.15	0.07	0.46, <0.001
Number of Non-Native Perennials	0.10	0.08	0.26	n.s.	0.07, <0.001
Cover of Non-Native Perennials	n.s.	0.11	0.10	n.s.	0.01, 0.009

Colorado Only/Dependent Variable	Number of Native Annuals	Cover of Native Annuals	Number of Native Perennials	Cover of Native Perennials	Model R^2, P
Number of Non-Native Annuals	0.47	0.17	-0.41	0.12	0.46, <0.001
Cover of Non-Native Annuals	0.35	0.24	-0.60	0.07	0.55, <0.001
Number of Non-Native Perennials	0.18	0.17	0.15	n.s.	0.10, <0.001
Cover of Non-Native Perennials	n.s.	0.32	0.13	n.s.	0.10, <0.001

Utah Only/Dependent Variable	Number of Native Annuals	Cover of Native Annuals	Number of Native Perennials	Cover of Native Perennials	Model R^2, P
Number of Non-Native Annuals	0.12	0.52	-0.10	n.s.	0.33, <0.001
Cover of Non-Native Annuals	-0.17	0.95	n.s.	0.07	0.81, <0.001
Number of Non-Native Perennials	-0.09	0.22	0.14	n.s.	0.05, <0.001
Cover of Non-Native Perennials	n.s.	n.s.	n.s.	n.s.	0.09, <0.001

Case Study 3: *Evaluating Plant Invasions from Both Habitat and Species Perspectives*

Although the characteristics of an invading species are important for a successful invasion, ecologists are now recognizing that the characteristics, dynamics, and history of the site being invaded are just as important (Hobbs and Humphries 1995). Thus, it is crucial to learn more about the interaction of the invader and its target habitat (Lodge 1993).

This case study examines landscape-scale patterns of non-native plant species in a wide array of vegetation types within the Monument to evaluate plant invasions from both a habitat and species approach. The objectives were (1) to identify successfully invaded habitats in a 150,000 ha portion of the Monument; (2) to identify successfully invading species in the study area; (3) to develop generalized models of plant invasions based on habitat characteristics; and (4) to test the general models of plant invasion on individual invasive species. We hypothesized that non-native species locations are not randomly distributed on the landscape and that vegetation types more vulnerable to invasion can be identified based on habitat characteristics measured in the field (Otsuki et al. *in review*).

Methods

This study included 142 plots within 14 different vegetation types sampled during 1998 and 1999 in the eastern portion of the Monument (Fig. 4-4). This region contains most of the Escalante Canyon region and the northeast corner of the Kaiparowits Plateau. Each vegetation type was assigned a moisture index value, ranging from one to ten for xeric vegetation types and 11 to 14 for mesic vegetation types. Vegetation sampling, soil sampling, and soil analyses were conducted as described in the methods chapter of this report (See Methodology).

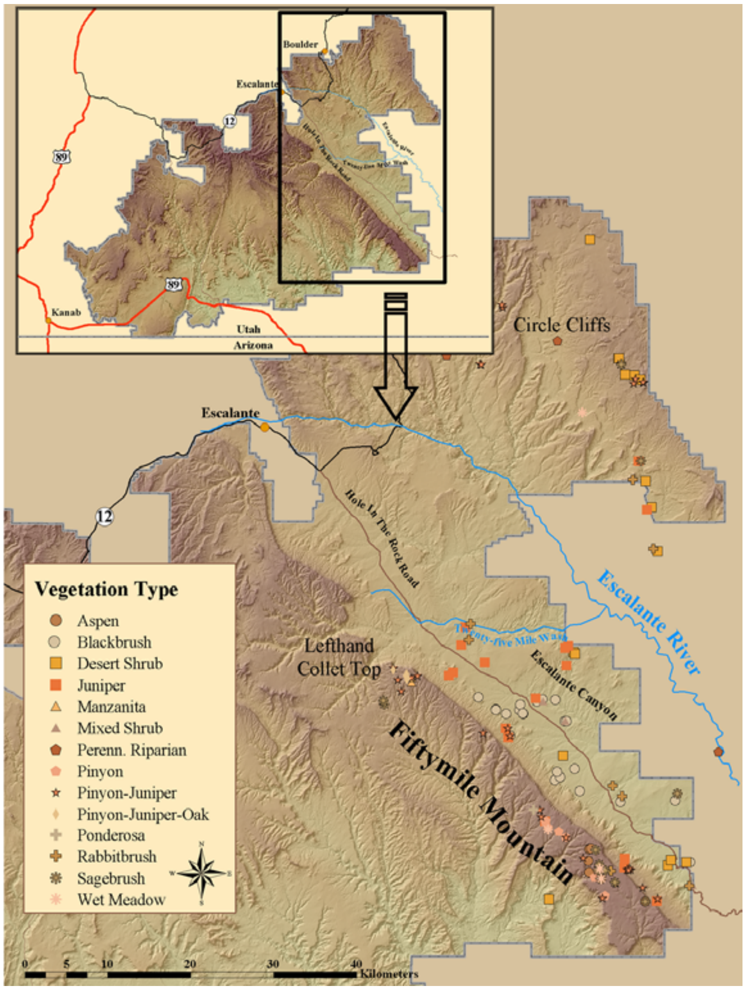

Figure 4-4. Study area for case study 3, including plot locations by vegetation type.

Statistical Analyses

All statistical analyses were performed using SYSTAT and an α value of 0.05 to determine significance. Variables were tested for normality and transformed using $\log_{10}(x+1)$ or square root transformations when needed. To examine general trends in the dataset, mean non-native species richness and mean non-native foliar cover were calculated for each vegetation type, and an analysis of variance (ANOVA) was conducted to compare these values.

Plot Invasive Index and Species Invasion Index

Two indices were developed to aid in analysis. Plot Invasive Index (PII) values were used to determine the most heavily invaded vegetation types, and the Species Invasion Index (SII) values were used to determine the most successful invading species. The PII is calculated by multiplying the total non-native species richness per plot by the total foliar cover for those species per plot (Lee 2001). The SII value is calculated by multiplying cumulative foliar cover, frequency in the plots, and the number of vegetation types invaded (Lee 2001). Thus, those species considered "invasive" are characterized by establishment in many plots and vegetation types and high foliar cover (i.e., high SII value).

Multiple Regression and Regression Tree Analyses

To learn more about the relationship between all the environmental variables measured in the field and non-native species richness and cover, stepwise-forward multiple regression and regression tree analyses were performed. Multiple regression allows the simultaneous testing and modeling of multiple independent variables. Stepwise-forward multiple regression does this by first choosing the independent variable which explains the most variation in the dependent variable, then choosing a second variable, and then another until there are no more independent variables that significantly explain the residual variation (Systat Software Inc. 1999).

Regression tree analysis works by assembling all subplots into one cluster and then separating them by attribute into a hierarchical binary tree. Therefore, the terminus of each branch of the tree will represent a cluster of plots that are more similar to each other than to a member of the twin cluster (Systat Software Inc. 1999). This type of analysis is able to identify key independent variables from the list of variables identified in multiple regression analysis (Lee 2001). Therefore, regression tree analysis will help demonstrate the relationship between certain environmental characteristics and non-native invasive patterns at landscape-scales using fewer variables than multiple regression.

Cluster Analysis

Data on cumulative foliar cover, plot frequency, number of vegetation types invaded, and the maximum percent cover were used in K-mean cluster analysis to group non-native species into four distinct groups. K-mean cluster analysis was used because it will produce a set number of different clusters of greatest possible distinction by maximizing between-cluster variation and minimizing within-cluster variation (Systat Software Inc. 1999). All non-native species could be classified into four groups following this analysis: (1) dominant generalists (i.e., high frequency and cover); (2) subdominant generalists (i.e., high frequency and low cover); (3) specialists (i.e., low frequency and high cover); and (4) rare or transient species (i.e., low frequency and low cover). An ANOVA was performed to test the differences in means of each data category among the four groups. Tukey's HSD method was used to compare cluster means if the ANOVA indicated statistical significance ($p < 0.05$).

Selected Invasive Species

Seven non-native invasive species (cheatgrass, storksbill, Kentucky bluegrass, Russian thistle, tamarisk, common dandelion, yellow salsify) were selected for more intensive study because they were found in more than ten plots. The seven species that were selected had high SII scores and represented three of the four groups classified in the cluster analysis: one dominant generalist, four subdominant generalists, and two specialists. The stepwise-forward multiple regression models originally used to predict non-native species richness and cover in the previous analysis were used to predict patterns of the seven non-native species. These models had a low explanatory power. Therefore, we focused on logistic regression models to better explain patterns for each of the seven invasive species.

Logistic Regression

Stepwise forward logistic regression was used to better describe patterns of invasion by identifying significant predictor variables for each of the seven species. Logistic regression creates a model used to study the association between a binary response and a set of independent variables (Systat Software Inc. 1999). Therefore, each plot was given a binary value for each of the seven species to indicate presence (value = 1) or absence (value = 0). The primary output used from this analysis consisted of McFadden's Rho-squared values and a prediction success table. The McFadden Rho-squared value is a transformation of the likelihood statistic intended to mimic an R-squared value. Therefore, a higher value corresponds to more significant results (Systat Software Inc. 1999). However, Rho-squared values tend to be much lower than R-squared values, and values ranging between 0.20 and 0.40 are considered satisfactory (Hensher and Johnson 1981). The prediction success table summarizes the classificatory power of the model, and includes additional analytical results on the ability of the model to successfully predict occurrence and the gain that this model shows over a purely random model (Systat Software Inc. 1999). The results from the logistic regression models were compared to the results from the multiple regression analyses to examine whether the same predictors were identified as significant.

A complete logistic regression was then run four times for each of the selected species using significant variables identified in the previous analyses: (1) multiple regression model for non-native species richness; (2) multiple regression model for non-native species cover; (3) regression tree model for non-native species richness; and (4) regression tree model for non-native species cover. The results from the logistic analyses were then compared with the results from the cluster analyses to see if we could generalize the relationships between habitat characteristics and non-native species patterns (clusters).

Results and Discussion

Mesic Habitats Most Vulnerable

Native and non-native plant species richness and cover varied greatly by vegetation type (Table 4-4). The sites most heavily invaded usually occurred within the more mesic vegetation types, containing more than three non-native species per plot on average and greater non-native species cover values. The pinyon pine vegetation type (moisture index = 7) was an exception because of an increase in non-native species as a result of some plots being located in washes. Three of the eight plots were located in these lowland areas and averaged 6.0 ± 0.9 non-native species per plot. In contrast, the other five plots were located in upland areas and averaged 3.0 ± 0.3 non-native species per plot.

Table 4-4. Summary characteristics of 14 different vegetation types within this study. Mean values of native and non-native species richness and cover were calculated for each plot. The total non-native species richness within the same vegetation type is shown as cumulative non-native species richness. Standard errors are shown in parentheses. Analysis of variance (ANOVA) was used to compare the differences in means within the same columns.

Moisture Index/ Vegetation Type	N	Cumulative Non-Native Species	Native Species Richness	Non-Native Species Richness	Native Cover (%)	Non-Native Cover (%)
1: Desert Shrub	16	7	28 (1.5)	2 (0.2)	26 (3.9)	2.0 (0.6)
2: Blackbrush	22	3	22 (1.3)	1 (0.1)	30 (2.3)	3 (1.0)
3: Mixed Shrub	5	3	26 (1.8)	1 (0.3)	30 (2.3)	3 (1.0)
4: Sagebrush	12	5	27 (1.9)	1 (0.3)	33 (4.1)	5 (1.7)
5: Juniper	17	9	29 (1.6)	2 (0.5)	24 (3.7)	4 (1.7)
6: Pinyon-Juniper	30	8	27 (1.1)	1 (1.6)	19 (2.4)	1 (0.5)
7: Pinyon Pine	8	12	33 (1.7)	4 (0.8)	33 (7.8)	4 (1.3)
8: Pinyon-Juniper/ Oak	5	4	39 (3.5)	1 (0.6)	20 (4.7)	<1 (0.3)
9: Ponderosa Pine	2	1	37 (6.0)	1 (0)	48 (16.8)	3 (1.5)
10: Manzanita	2	0	32 (3.0)	0 (0)	40 (3.6)	0 (0)
11: Rabbitbrush	8	12	26 (1.5)	3 (0.9)	23 (4.6)	8 (3.6)
12: Aspen	6	15	35 (3.7)	6 (1.4)	44 (11)	11 (3.4)
13: Wet Meadow	4	19	32 (4.2)	8 (1.2)	36 (10)	24 (8.0)
14: Perennial Riparian	5	18	35 (4.3)	10 (1.1)	47 (4.8)	12 (2.6)
All Types	**142**	**34**	**28 (0.6)**	**2 (0.2)**	**27 (1.6)**	**4 (0.6)**
ANOVA	F-ratio		4.74	20.16	3.16	7.04
	P<		0.001	0.001	0.001	0.001

The 23 plots located in the mesic vegetation types contained 31 of 34 non-native species found within the study area, and 14 of these species were only found within these vegetation types. The 119 plots located in the xeric vegetation types contained 20 of 34 non-native species. Only two non-native species were confined to xeric plots, while 17 species occurred in both mesic and xeric vegetation types.

Few plots had multiple non-native species with high foliar cover within them. Therefore, the Plot Invasive Index (PII) had a skewed distribution. Sixteen plots (11%) appeared more heavily invaded than the others, with PII scores greater than 500. Most of the invaded plots (90%) from the PII list were located in mesic vegetation types (Table 4-4). There was only one plot not within a mesic site, and it was located in the ecotone between juniper (*Juniperus* sp.; moisture index = 5) and rabbitbrush vegetation types (*Chrysothamnus* sp.; moisture index = 11) and was heavily invaded by cheatgrass. Generally, high PII values in xeric vegetation types were a result of high cheatgrass cover.

Mesic vegetation types (especially aspen and perennial riparian) are rare on the Monument landscape and contain unique plant assemblages. These results show that they are more heavily invaded than more common vegetation types such as blackbrush and pinyon-juniper. When examining the 15 most invaded plots, five were located within perennial riparian vegetation types, three in the wet meadow vegetation type, and two in the aspen vegetation type. All these sites are high in moisture, native species richness, and soil fertility. Other studies have also found that highly diverse habitats are supported by more available resources and may be more susceptible to invasion (Hobbs and Huenneke 1992; Planty-Tabacchi et al. 1996; Lee 2001; Stohlgren et al. 1998b; 1999b; 1999c; 2001).

Predicting Habitat Vulnerability

Using stepwise forward multiple regression, seven habitat variables (moisture index, elevation, percent soil phosphorus, native species richness, maximum crust development class, percent bare ground, and percent rock) were identified as significant predictors of non-native species richness (Table 4-5), explaining 60% of the variance. For non-native species cover, these same habitat characteristics (excluding native species richness) explained 46% of the variation (Table 4-5). In general, non-native species richness and cover were greatest in more mesic vegetation types, and in areas with higher soil phosphorus, making these results comparable to those found in the previous analysis.

Non-native species invasions were also more successful in plots with less developed microbiotic soil crusts and low crust cover, with the exception of cheatgrass. Although it was not directly measured in this study, low crust cover could be related to anthropogenic disturbances such as trampling by livestock and off-road vehicle use. Disturbance is known to be correlated with non-native plant species presence. Crust recovery from trampling can also take decades (Belnap 1999), allowing opportunity for the establishment of non-native species.

The regression tree analyses provided another way to view the relationships between environmental variables and non-native species invasion. The regression tree for non-native species richness found that soil moisture index and phosphorus gave a proportional reduction in error (PRE) value of 0.57 (Fig. 4-5). PRE values (similar to R^2 values) were calculated to determine the amount of variation explained by the independent variables in the model (Hansen et al. 1996). Therefore, over half of the variation in non-native species richness was explained using only two habitat characteristics. Similar to the regression model, mesic vegetation types related more strongly to non-native species richness than xeric vegetation types. Within the xeric vegetation types, high soil phosphorus was a good indicator of non-native species richness (Fig. 4-5).

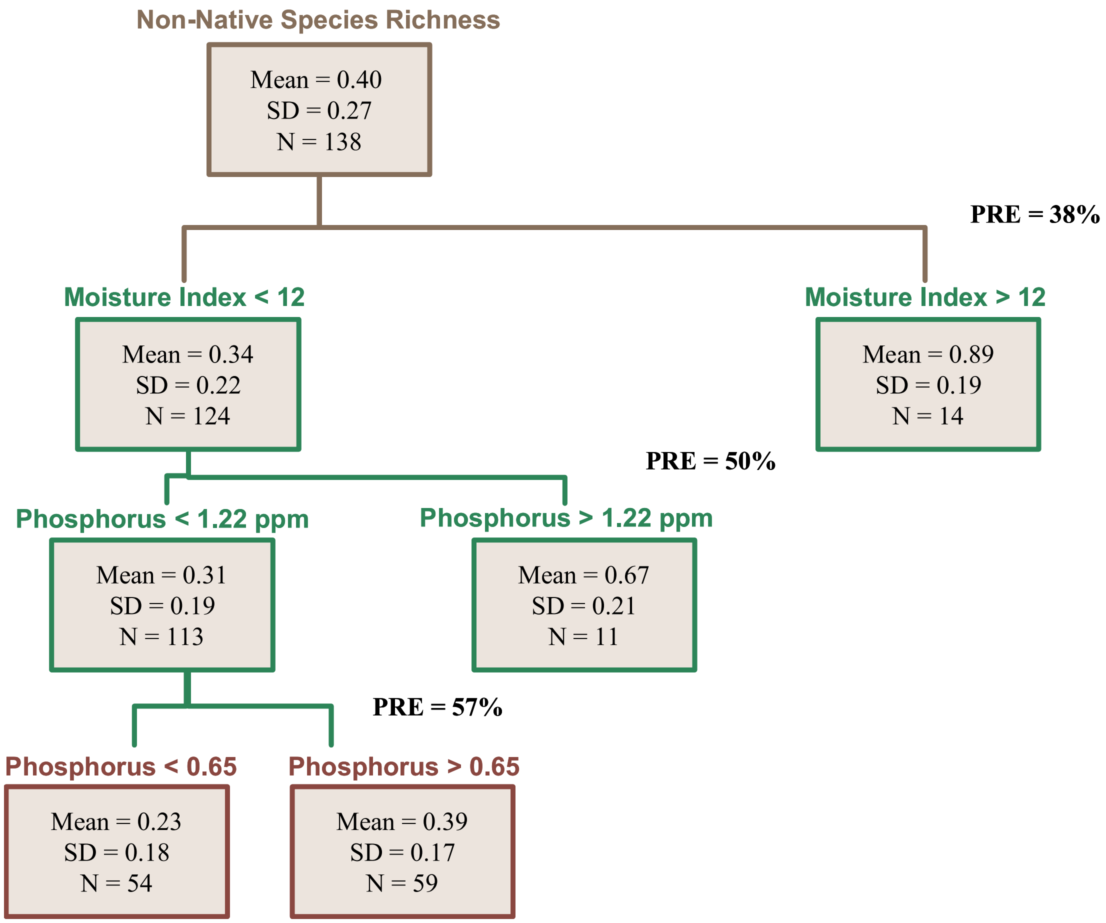

Figure 4-5. Regression tree analysis with $\log_{10}$ non-native species richness as the dependent variable. Moist sites are most heavily invaded, while drier sites with less soil phosphorus are less heavily invaded.

The non-native cover regression tree identified moisture index, bare ground, and soil phosphorus as independent predictors, explaining 45% of the variation in non-native cover (PRE = 0.45; Fig. 4-6). Non-native cover was also greatest in the more mesic vegetation types. Within the xeric vegetation types, plots with less bare ground and higher phosphorus had greater non-native species cover (Fig. 4-6).

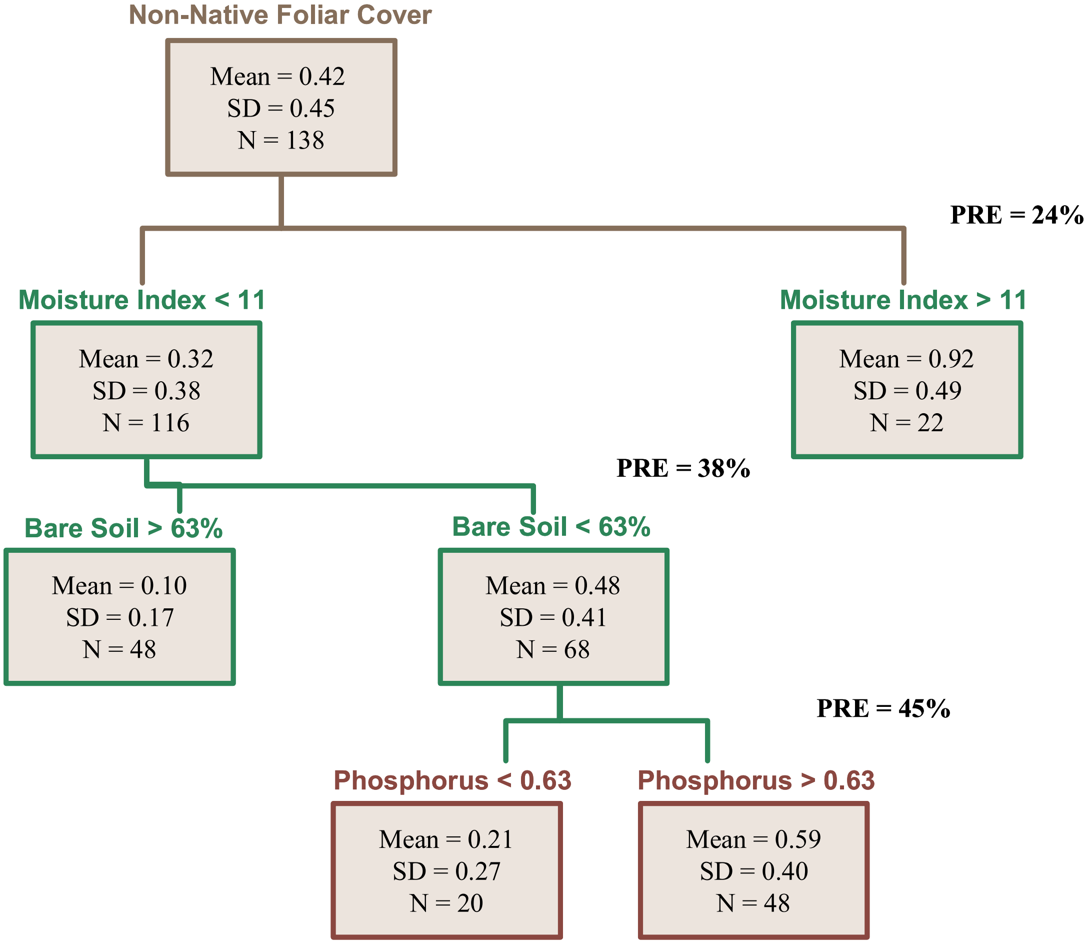

Figure 4-6. Regression tree analysis with $\log_{10}$ non-native foliar cover as the dependent variable. Moist sites are most heavily invaded, while drier sites with less bare ground and less soil phosphorus are less heavily invaded.

In this case study, the moisture index, soil phosphorus, and elevation were the most important indicators of non-native plant invasions, explaining 50% of the variation. Other factors such as native species richness and topographic position have been found in other studies to be strong predictors of non-native species richness and cover (Robinson et al. 1995; Stohlgren et al. 1997; 1998b; 1999c; Chong et al. 2001a; 2001b). In this case study, non-native plants were more successfully established in mesic vegetation types with higher soil phosphorus at lower elevations.

Table 4-5. Multiple linear regression results for explaining non-native species richness and cover from habitat characteristics. Most values were transformed prior to analysis.

Dependent Variable/ Habitat Characteristics	Coefficient	Standardized Coefficient	t	P-value
Non-Native Species Richness (Model $R^2 = 0.60$, $F = 30.8$, df = 7 and 130, $P < 0.001$)				
Moisture Index	0.024	0.318	5.18	0.001
Elevation (m)	-0.000	-0.359	-4.99	0.001
% Soil P	0.308	0.396	5.27	0.001
Native Species Richness	0.006	0.154	2.57	0.011
Maximum Crust Dev. Class	-0.021	-0.161	-2.72	0.007
% Bare Ground	-0.005	-0.306	-4.82	0.001
% Rock	-0.006	-0.285	-4.74	0.001
Non-Native Species Cover (Model $R^2 = 0.46$, $F = 20.4$, df = 6 and 131, $P < 0.001$)				
Moisture Index	0.023	0.180	2.62	0.010
Elevation (m)	-0.000	-0.196	-2.41	0.017
% Soil P	0.341	0.262	2.99	0.003
Maximum Crust Dev. Class	-0.049	-0.228	-3.32	0.001
% Bare Ground	-0.010	-0.384	-5.21	0.001
% Rock	-0.009	-0.268	-3.82	0.001

Plant Invasions from a Species Perspective

Although it is important to determine the vegetation types most vulnerable to invasion, it is also important to collect species-specific information that is needed for targeted control efforts that often rely on species-specific biological controls or less specific chemical controls.

There were a total of 519 species identified in the study area, with 32 of these (6.6%) being non-native. Cheatgrass was the most dominant non-native species, with 65% of the total non-native species cover on the landscape. Big sagebrush (*Artemisia tridentata*) and blackbrush (*Coleogyne ramosissima*) were native species having similar cover values, with 3.0 ± 0.6% and 3.0 ± 0.8%, respectively.

Cheatgrass and the native broom snakeweed (*Gutierrezia sarothrae*) occurred in 80% of the plots, making them the most frequently encountered species. Generally, species that were frequently encountered had low foliar cover. For example, yellow salsify (*Tragopogon dubius*) was found in 22 plots but averaged less than 1% cover. Meadow fescue (*Festuca pratensis*), in contrast, had the highest mean foliar cover (5% per plot) but was only encountered in two plots.

Testing the General Models of Plant Invasion for Individual Non-Native Species

From the cluster analysis, cheatgrass was singled out as the only dominant generalist from the non-native species list (Table 4-6). This annual grass is widespread across the Monument, occupying 114 plots in 13 vegetation types. Eight species were classified as subdominant generalists, and these species had an average plot frequency less than 8%. The specialist group had plot and habitat frequencies similar to the subdominant generalists, but they had relatively high mean cumulative cover and maximum foliar cover values. The 19 species classified as transient had low values in all categories.

Table 4-6. Mean plot frequency, habitat frequency, cumulative cover, and maximum foliar cover for the four invasive species groups. Standard errors are in parentheses. Means within each column were compared separately by invasive species group with on-way ANOVA and the F-ratio and p-value are listed. Values followed by a different letter within a column are significantly different.

Invasive Species Group	Number of Species	Plot Frequency (%)	Habitat Frequency (%)	Mean Cumulative Cover (%)	Maximum Foliar Cover/ Plot (%)
Dominant Generalists	1	80a	93a	379a	94a
Subdominant Generalists	8	8b (1.7)	38b (6.3)	10c (2.5)	13c (3.0)
Specialist	6	7b (1.2)	32b (4.8)	25b (6.9)	60b (5.4)
Transient	19	2c (0.3)	12c (1.8)	3c (0.8)	5c (1.6)
P value		0.0001	0.0001	0.0001	0.0001
F-ratio		229.4	23.2	681.1	89.5

The Species Invasive Index (SII), which determines successful invasion of a species, had a highly skewed distribution with only seven species having values much greater than 0. These were the seven species selected for the logistic regression analysis. The logistic regression analyses for all seven focal species gave Rho-squared values within a satisfactory range (0.34-0.62; Hensher and Johnson 1981; Table 4-7). In addition, the total correct rate for each of the models was satisfactory, ranging from 80 to 93% (Table 4-7).

Table 4-7. Summary of logistic regression analyses for seven focal non-native species in the Monument. All models were significant at $P < 0.001$.

Species	Predictor Variable (P)	Rho^2	Total % Correct
Cheatgrass (Dominant Generalist)	Soil P (0.001), UTM-N (0.065), Elevation (0.061). Bare Ground (0.006), Crust Cover (0.010), Herbaceous Species Richness (0.067)	0.34	80%
Storksbill (Subdominant Generalist)	Elevation (0.001), Soil P (0.002), UTM-N (0.001), UTM-E (0.004)	0.48	89%
Kentucky Bluegrass (Specialist)	Topographic Position (0.001), Moisture Index (0.002), Low Cover Richness (0.001), Succulent Species Richness (0.004)	0.60	93%
Russian Thistle (Subdominant Generalist)	Elevation (0.016), UTM-N (0.052), UTM-E (0.005), % Crust Cover (0.066)	0.42	90%

Table 4-7. Summary of logistic regression analyses for seven focal non-native species in the Monument. All models were significant at $P < 0.001$.

Species	Predictor Variable (P)	Rho^2	Total % Correct
Tamarisk (Specialist)	Moisture Index (0.004), UTM-N (0.07), Elevation (0.002), % Sand (0.029)	0.61	93%
Common Dandelion (Subdominant Generalist)	Soil P (0.001), Moisture Index (0.001), Native Species Richness (0.091), Bare Ground (0.157)	0.61	92%
Yellow Salsify (Subdominant Generalist)	Soil P (0.001), Elevation (0.016), Native Species Richness (0.146)	0.62	89%

Storksbill (Subdominant Generalist)

Four variables (elevation, soil phosphorus, UTM-N, UTM-E) were identified by the logistic model for predicting the presence of storksbill. The model explained 48% of the variation in species presence and was correct 89% of the time (Table 4-7). Elevation and UTM coordinates had strong negative correlations with storksbill distribution, suggesting the presence of this species in lower elevation plots in the southwestern portion of the study area. However, a positive relationship existed between this species and phosphorus.

Russian Thistle (Subdominant Generalist)

Four variables (elevation, UTM-N, UTM-E, percent crust cover) were identified as predictors of Russian thistle. This model provided a 45% gain over the random model and generated 90% correct results (Table 4-7). The model also explained 42% of the variation in Russian thistle presence. Because elevation and UTM coordinates were significant indicators of Russian thistle, this suggests that this species has a greater occurrence in the lower elevation, northwestern portions of the study area. Percent crust cover was only significant when considering the interaction with the other three variables (Table 4-7).

Storksbill and Russian thistle shared general common traits, but both had unique invasion patterns. Both species are annual forbs often associated with disturbed sites and were classified as co-dominant generalists in this study. The logistic models for these species showed that the location variables UTM-E, UTM-N, and elevation could explain 35% of the variation for the two species. However, the distributions of both species did not overlap. Storksbill occupied the northwestern portion of the study area while Russian thistle occupied the southeastern portion.

Kentucky Bluegrass (Specialist)

Four variables (topographic position, moisture index, transient species richness, succulent species richness) explained 60% of the variation in Kentucky bluegrass distribution (Table 4-7). This model indicates that Kentucky bluegrass is invading lowland wash sites that contain less succulent and more transient species. This model demonstrated a 54% gain over a purely random model for species presence and was correct 92% of the time.

Tamarisk (Specialist)

Four variables (moisture index, UTM-N, elevation, percent sand) were identified by the logistic model as predictors of tamarisk (Table 4-7). All these variables, except for UTM-N, were significant when examined independently. Therefore, salt cedar successfully invaded moist, lower elevation sites with less sandy soil with a prediction success of 93% (Table 4-7).

Common Dandelion (Subdominant Generalist)

Four variables (soil phosphorus, moisture index, native species richness, and bare ground) were identified by the logistic model as predictors of common dandelion (Table 4-7). All four variables were strongly, positively correlated with the species. The model explained 61% of the variation in common dandelion presence and was correct 92% of the time (Table 4-7).

Yellow Salsify (Subdominant Generalist)

Yellow salsify was the second most frequently encountered non-native species in the Monument. Soil phosphorus, elevation, and native species richness were found to be predictors of this species and explained 62% of the variation in species presence. Similar to common dandelion, the model indicated weak positive effects of native species richness interacting with the other two significant variables (i.e., soil phosphorus, elevation). The model produced correct results 89% of the time, and produced the highest McFadden's Rho-squared value of 0.62 with the fewest number of variables among the seven focal species (Table 4-7).

Common dandelion and yellow salsify, two subdominant generalists, demonstrated similar habitat preferences in this study. The presence of both species showed strong positive relationships to sites with higher soil phosphorus, which made it the single most significant variable in any of the models. When including native species richness in these models, 51% of the variation in common dandelion could be explained and 49% of the variation in yellow salsify could be explained. These species will rarely dominate a landscape, but they do have traits that allow them to establish in a wide range of environmental conditions. Both species have prolific reproduction, many adaptive biotypes, and effective seed dispersal mechanisms (Holm 1997; Clements et al. 1998). Based on total non-native species richness, logistic models found that these two widespread species had highly predictable distributions.

Cheatgrass (Dominant Generalist)

Six variables were identified by the logistic regression analysis for predicting the presence of cheatgrass and were able to describe 34% of the variation in cheatgrass presence. Soil phosphorus and percent of total cryptobiotic crust cover had strong positive direct effects, and percent bare ground had negative direct effects on cheatgrass distribution. Location, elevation, and herbaceous species richness were only significant when considering the interaction with other identified variables, suggesting that these variables may only have indirect effects on the distribution of this species. Based on the results of prediction success indices, this model produced a gain of 6% over a purely random model for responses (cheatgrass present) and 28% over a purely random model for reference cases (absent), producing a total correct rate of 80%.

Cheatgrass was widespread throughout the Monument, occurring in 93% of the plots, making it the most difficult species to predict. The other six focal non-native species of concern were confined to fewer habitats, so predictions of species occurrence based on logistic regression tended to be more accurate than the predictions for the generalist cheatgrass. Therefore, the results from this case study demonstrate the individualistic nature of plant species invasion patterns in the Monument. Some species had a higher predictability than others. Specifically, non-native species that invade more diverse and fertile habitats are identified more easily in general models than non-native species that occupy drier, upland sites with various soil types.

Overall, few habitat characteristics were needed to predict the presence of the seven focal species. More importantly, the field data used to evaluate the invasion of specific vegetation types can also be used to predict non-native species that are highly invasive. Therefore, a habitat and species approach may be applicable to a majority of landscapes and can be used to target non-native species control efforts. Since patterns of invasion are expected to change over time, these approaches may be strengthened by continuing to inventory and monitor.

Case Study 4: *Early Detection and Mapping Techniques of Tamarisk in Hackberry Creek*

Eradication of tamarisk has become a high priority for many government agencies, resource managers, policy makers, and private landowners. However, controlling tamarisk infestations and reducing new invasions has largely been unsuccessful. Recently, a number of new and effective control strategies, including mechanical, chemical and biological treatments, have been developed and are rapidly being implemented in the field (Duncan and McDaniel 1998; Stomberg 1998; Sher et al. 2002; DeLoach et al. 2003; McDaniel and Taylor 2003). Despite these recent advancements in control methods, accurate and cost-efficient methods for early detection, stand inventories, and mapping remain poor and undeveloped.

Given the potential for rapid and complete invasion of tamarisk and the high costs associated with large-scale eradication efforts (McDaniel and Taylor 2003), many control strategies and management plans focus on prevention and early detection of new infestations. For this strategy to be successful, resource managers must rely on detailed spatial data of tamarisk distributions at multiple scales, which is unavailable in most cases. Large monocultures of tamarisk can easily be surveyed by aircraft or satellites, but methods for detecting new infestations, isolated individuals, germinating seedlings, and new sprouts remain undeveloped (Everett 1996). Likewise, resource managers require standardized methods for measuring species abundance and attributes, or for monitoring the performance of control or restoration efforts. Field measurements often lack consistency and detail, reducing its value for mapping initiatives, statistical analyses, and spatial modeling.

Mapping of tamarisk has proven to be quite challenging, especially when stands are small and dispersed, intermixed with native tree species, or found in narrow canyons that obscure aerial views. Techniques in mapping riparian habitats and invasive species have largely focused on different applications of remote sensing and aerial surveys (Everett 1990; Everett 1996; Rowlinson 1999; Hirano 2003; Lachowski 2003). Although these techniques demonstrate varying degrees of success, they often fail to distinguish the species of interest from other vegetation, are only applicable at course-scales, cannot detect small plants and early invasions, and tend to be too costly for management budgets.

In this case study, we critique a new scale-dependent plot that is designed to meet many of the previously mentioned needs associated with tamarisk invasion and control. The scale-dependent plot was field tested in a recently invaded site that had tamarisk present at varying densities and age classes. Cottonwood (*Populus fremontii*), willow (*Salix* spp.), hackberry (*Celtis occidentalis*), box elder (*Acer negundo*), and Russian olive (*Elaeagnus angustifolia*) were also present and often intermixed with tamarisk. The scale-dependent plot was designed for early detection of infestation and compatibility with detailed mapping, spatial analyses with Geographic Information Systems (GIS), and monitoring initiatives.

Methods

Study Area

Our study area was located in the lower stretch of Hackberry Canyon within the Monument (Fig. 4-7). Hackberry Creek, a small intermittent stream that flows through the canyon, begins approximately 16 km north of the mouth of the canyon. Its watershed covers approximately 370 km^2, receiving its water from runoff of seasonal precipitation and three springs situated along the canyon. Hydrological monitoring at the confluence of Cottonwood Creek between August 1998 and June 2001 indicate that water flow does not persist all year, and usually runs dry between June and August for varying periods of time (J. Vanderbilt, *pers. comm.*).

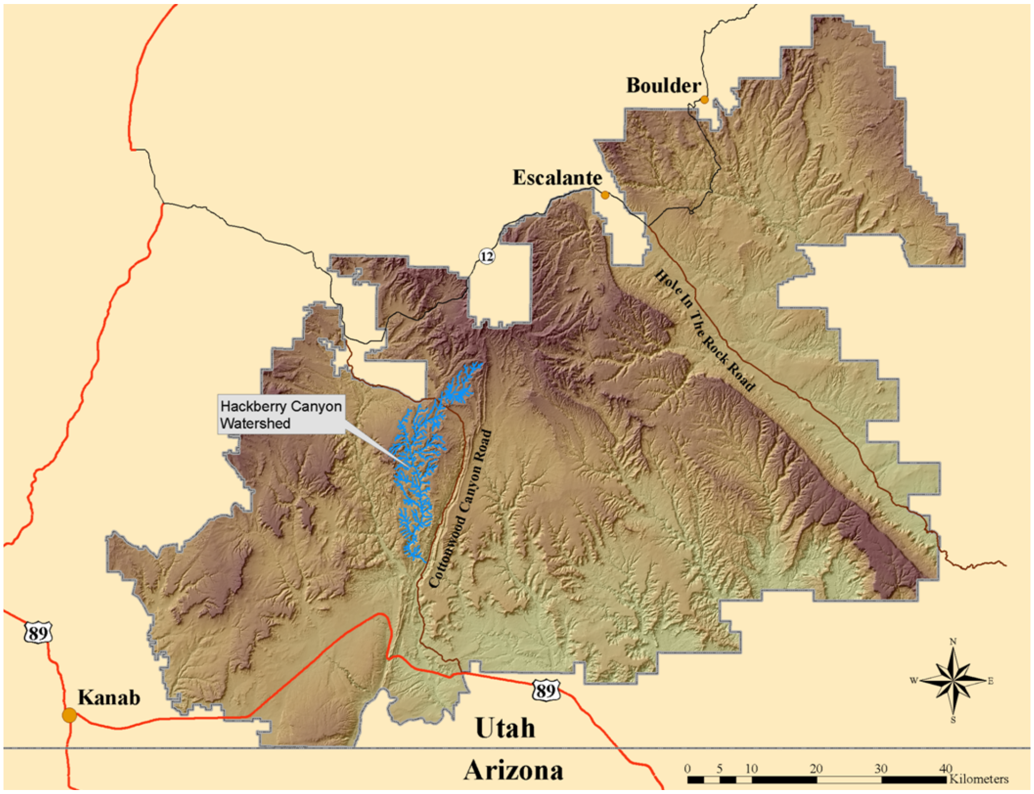

Figure 4-7. Map of Hackberry Canyon.

Field Measurements

Field measurements were taken in May and September 2003. We categorized the landscape within the Hackberry corridor into three easily distinguishable vegetation zones: wash, riparian, and upland. The wash was defined by running water, moist soils, lack of terrestrial vegetation, and physical indicators of annual flooding (e.g. water marks, gravel). The riparian was characterized as high banks, sand bars, and the establishment of riparian vegetation (e.g. cottonwood and willow). This zone was not necessarily immune to flooding, but the vegetation structure suggests that it was an occasional occurrence and not an annual event. The upland zone was occupied by more xeric vegetation such as sagebrush (*Artemisia* spp.) and juniper (*Juniperus osteosperma*), and biological soil crusts were often present. Beginning at the confluence of Hackberry Creek and Cottonwood Creek, we measured the widths and elevations of the wash and riparian zones and boundaries at every 0.5 km interval. These measurements were later used to validate vegetation zones delineated from GIS mapping and spatial analyses of a digital elevation model (DEM).

Field measurements of tamarisk were collected using the new scale-dependent plot that is adjustable to the dimensions of an individual tree or stand of interest (See Appendix I). For each plot, we recorded the length of each side (X_{length} and Y_{length}), the azimuth of one side ($Y_{azimuth}$), the coordinates of one corner using a global positioning system (GPS) receiver, and which of the three vegetation zones was occupied. In a few cases, GPS coordinates were not obtained for plots due to extreme topographic features. For these situations, we recorded the azimuth and estimated the distance to the nearest plot with obtained coordinates and later converted to UTM coordinates using a custom program in Visual Basic for Applications (VBA). Individual trees were measured for height and the percent basal cover of the plot. For plots established around more than one tree, we estimated total percent basal cover, average height, maximum height, and number of individuals.

Mapping

Mapping field data and risk modeling were conducted using ArcGIS 8.1© software. We used a 10 m DEM and field measurements to calculate the riparian and upland zones on a GIS map. The wash, on average, was too narrow within the study area to be represented by a GIS layer and was incorporated with the riparian zone. After deriving a raster layer of the Hackberry Creek watershed from the DEM, we reclassified cell values to either 1 (representing the wash) or 0 (all other cells). New cell values were multiplied by the DEM to determine elevation of the wash throughout the study area. This new layer provided a foundation for estimating the riparian area, using distance allocation analysis in ArcGIS© Spatial Analyst and, width and elevation data collected from the 0.5 km intervals. Specifically, the riparian layer created in the GIS display cells that are within 150 m distance of the wash and have elevation values between the wash and the riparian/upland interface.

Prior to mapping tamarisk distributions, we calculated the UTM coordinates for the corners of each plot using the length of each side (X_{length} and Y_{length}) and GPS coordinates from the field using a VBA macro written in MS Excel©. Based on the four UTM coordinates at each corner of the plot and the recorded azimuth, we generated a polygon layer representing the locations of all the plots by age-class (Fig. 4-8). A model of tamarisk abundance within the study area was developed to spatially locate different levels of current and predicted risk of invasion. The risk model was developed by creating a VBA macro, in ArcMap©, that calculated the approximate spatial coordinates of every tamarisk recorded within each scale-dependent plot. Based on the estimated number of individuals recorded in the field, the risk model randomly generates a point representing each tamarisk within the boundaries of the sample plot. The distribution of points within each polygon follows the Poisson distribution of complete spatial randomness (Green 1966). Generated tamarisk points were then assigned its corresponding plot number and age-class. The risk model uses an assigned search radius to find total number of points within that area. The total number of points was then divided by the area of the radius to calculate the level of risk for each 10 x 10 m pixel. For our test of the risk model, we used a search radius of 100 m as an estimate of the potential distance of spread by seed dispersal and other mechanisms of tamarisk establishment.

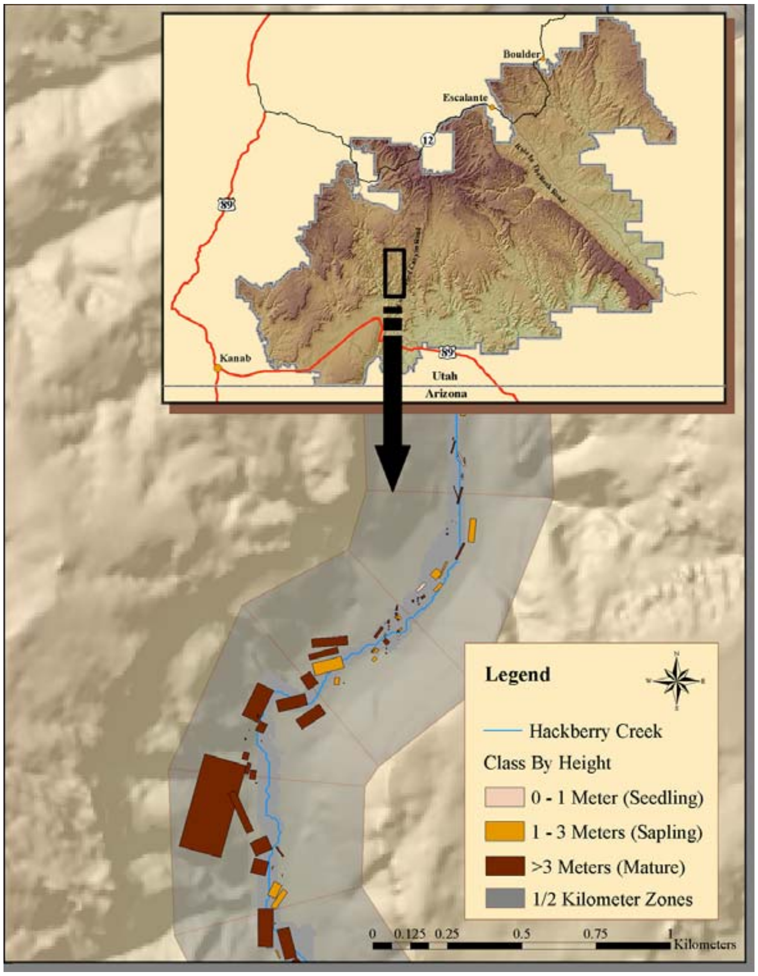

Figure 4-8. Tamarisk locations within Hackberry Creek by age class.

Results

Tamarisk in Hackberry Canyon was widely dispersed throughout our study area, and densities among the different age-classes were highest in the central portion of Hackberry Canyon. Nearly all the tamarisk was found in the riparian zone with none occurring in the upland. The few tamarisk trees found in the wash were mostly new recruits and established individuals appeared to be more susceptible to seasonal flooding events. Most tamarisk stands in the upper and lower stretches of the study area were relatively small with a patchy distribution. The lower section of Hackberry Creek had flowing water at the time of sampling, but was contained in steep narrow canyons with a narrow riparian zone. The upper section of Hackberry Creek lacked water flow at the time of sampling and was characterized by a more xeric landscape. Tamarisk, cottonwood, and willow tapered off rapidly around the 16 km mark, which was also the furthest extent of the riparian zone. Beyond this point, only one additional tamarisk tree was found near the 18 km mark. Based on our field data, we estimate the total basal cover of tamarisk to be 3.6 ha in our study area. Of the total cover, seedlings and sprouts occupied 212 m^2 of the total infested area; saplings occupied 4,437 m^2; and, mature trees occupied 31,827 m^2.

The average elevation difference between the wash and the riparian/upland interface was 1.2 m from the lower end of Hackberry Creek to the 16 km mark. Using the riparian layer generated from the DEM, two meters were added to the elevation values of the wash to represent the potential habitat that tamarisk can occupy. The estimated potential habitat along the Hackberry corridor is 71.3 ha. Of that area, tamarisk of all age-classes occupies 5%. Seedlings and sprouts occupy 0.03 % of the potential habitat; saplings occupy approximately 0.62%; and, mature trees 4.4%.

By distinguishing tamarisk densities by age-classes and spatially analyzing their locations, we could identify the approximate location where the invasion begun, and see patterns of spread following its establishment to the time of our sampling (Fig. 4-8). The risk model allows us to predict and display areas most susceptible to future infestations based on the 100 m search radius (Fig. 4-9).

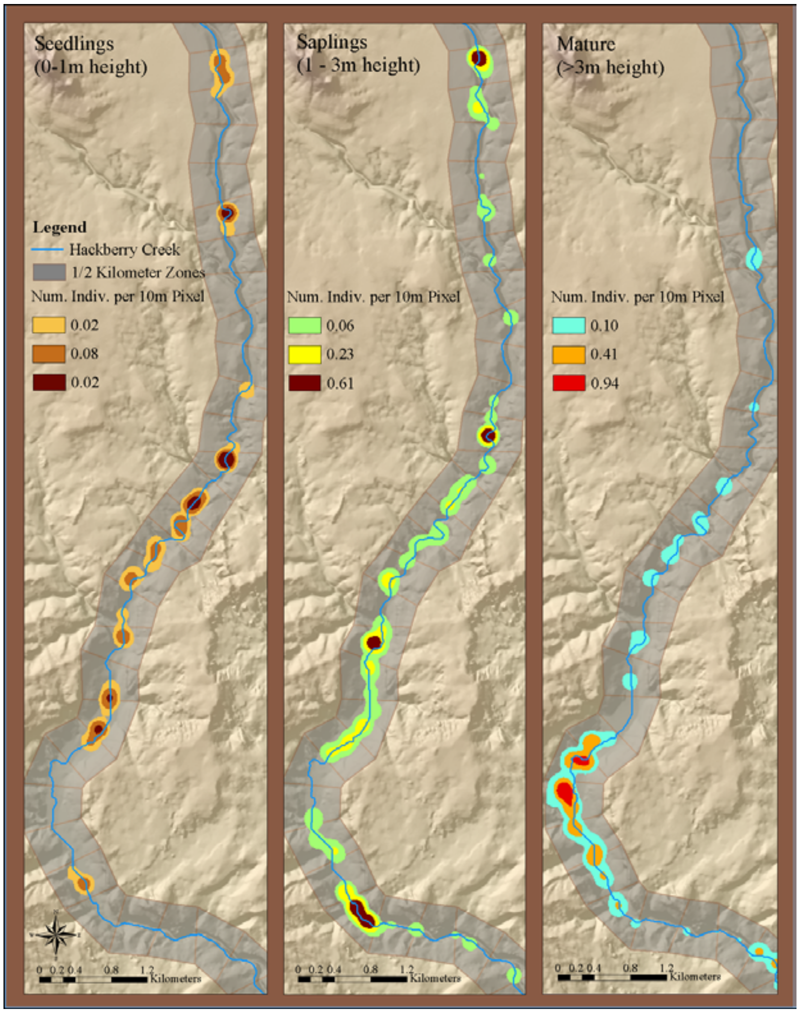

Figure 4-9. Potential future spread of tamarisk infestation in Hackberry Canyon based on scale-dependent plots and our risk model. Each map illustrates infestation risk associated with each of the three age-classes: seeding and sprouts, saplings, and mature trees.

Discussion

Review of the Scale-Dependent Plot

Time and cost spent in data acquisition is always of concern for resource managers. Too often, accuracy and precision of data collection are compromised by constraints in budgets and personnel. Current mapping methodologies of tamarisk can be very expensive, provide only limited data on physical characteristics, and are often incapable of detecting seedlings, small sprouts, and individuals growing in close proximity to other species. Utilization of the scale-dependent plot proved to be an effective tool for gathering highly detailed data on tamarisk distribution and physical characteristics of new infestations, while maintaining cost-efficiency. Field data were collected by two field technicians over the course of ten days. High canyon walls restricted access of Hackberry Creek to its confluence with Cottonwood Creek. These travel limitations added at least two days to our time spent in the field, which could be substantially reduced if better access to the study site were available.

The largest constraint when utilizing the scale-dependent plot is with large stands of tamarisk that cover (area > 0.5 ha). The first problem we encountered with large stands was that the sides of the scale-dependent plots were often too long to measure with a tapeline and had to be paced out for its length. Although the error in pacing out distances may be insignificant for most purposes, dense vegetation and abrupt topographic changes can facilitate considerably more when trying to establish the shape of the scale-dependent plot. Secondly, large stands were not easily encompassed in a square or rectangular shaped plot, which hindered our ability in estimating the percent basal cover from the ground. These large plots often encompassed vast areas that were free of tamarisk, creating additional difficulties estimating basal cover when mixed with other species. If measuring a single tree with the scale-dependent plot greater than 1 m^2, the basal cover of an individual will consistently hover around 80% of the plot area. However, as stands became larger and irregular in shape, our cover estimates were often less than 50%. Furthermore, measuring large stands reduced our ability to distinguish age-classes (average height) and to determine accurate estimates of densities. For these cases, we used the same scale-dependent plot on a particular tamarisk stand and collected separate data (percent cover and number of individuals) for each of our age-classes. On a map, three separate plots were displayed at the same location with identical areas, but had corresponding data for the three different age-classes. This approach may not always yield easier field estimates, but it did allow us to separate data collection into age-classes rather than averaging all heights.

The data collected in the field, however, were extremely detailed and may have many useful applications to both researchers and resource managers in understanding patterns of invasion, predicting the spread of future infestations, detecting increases or decreases in distribution, and monitoring the effectiveness of mitigation treatments. Furthermore, the data collected is easily encompassed into a GIS, allowing accurate mapping, various spatial analyses, and predictive modeling. We believe the risk model developed for our field data produced reasonably accurate results. This model can be enhanced, however, by incorporating hydrological data and with a better understanding of reproductive mechanisms of tamarisk (e.g. potential distance of seed dispersal). For large established tamarisk stands that cover an area greater than 0.5 ha, the scale-dependent plot is not recommended. Large-scale mapping projects and research studies would be better served developing remote sensing capabilities and predictive modeling.

Management Implications

As stated previously, non-native plant species may pose the most significant threat to the ecological integrity of the Monument. Although there are no guaranteed solutions for controlling these species, the case studies presented here provide a starting point for targeting and monitoring control efforts.

Cheatgrass was found to be the most dominant non-native plant species within the Monument. Knowledge of which vegetation types are susceptible to cheatgrass invasion will help guide and direct future management efforts of this species. As a generalist invader, cheatgrass has the potential to alter a large portion of the Monument by dominating in some areas and by potentially displacing native species. This species was represented in all of the vegetation types sampled and showed a substantial foliar cover in some, such as the aspen vegetation type. Cheatgrass could potentially gain dominance in sagebrush habitats, reducing native species richness and possibly displacing sagebrush by altering the fire regime (Wright 1985; Belnap et al. 2000). The possible displacement of sagebrush by cheatgrass could be ecologically detrimental to this vegetation type. It could reduce the amount of soil moisture available throughout the year, reduce establishment of grass and forb seedlings that use the shade provided by this species, reduce nutrient cycling and therefore productivity, and reduce available habitat for many native animal species (Rosentreter 1994). Further research should be conducted to examine potential impacts that this non-native plant could be having on hotspots of native biodiversity.

It was demonstrated through case study 2 that non-native annual plant species richness and cover could be predicted without complex multi-scale environmental data that can be costly and time consuming to collect. Other studies found in this report relied on expensive soils data (Stohlgren et al. 2001; Bashkin et al. 2003) and climate data (Stohlgren et al. 2001), but these data did not explain nearly as much variation in invasion success as the species' life cycles used here. Therefore, a plant's life history may be an important predictor of successful invasion by non-native species because this strategy integrates environmental variables (Stohlgren et al. 1998b; D'Antonio et al. 1999; Bashkin et al. 2003). Predictive models should take advantage of the important role of life cycle in the successful invasion of non-native plant species.

Soil analyses and other measurements of environmental variability may cost approximately $50/plot. From these results, there is the potential for the use of inexpensive field measurements on the richness and cover of native annual and perennial species to be used as surrogates to other environmental variables in predicting patterns of plant invasions is promising.

To address important and general aspects in the management and control of invasive species, case study 3 tested different approaches for evaluating species-environment relationships at the landscape scale. Using the habitat approach, land managers can set priorities for future surveys and monitoring. Our research shows that mesic vegetation types within the Monument are most vulnerable. Control efforts within the more heavily invaded vegetation types (Table 4-4), which are also high in native species richness and soil moisture, will require care to reduce impacts to native species (Chong et al. 2001b) and water resources. Using the species approach, it was found that many non-native species are locally rare transients (19 out of 34 species). Therefore, only a few non-native species will become invasive and have a significant effect on natural resources. Thus, new research and analytical tools are needed to identify which species can cause and will cause significant impacts (Hiebert 1997).

It is also important to note that with a relatively modest sampling intensity, the general predictive models used in case study 3 were able to explain 60% and 46% of the variation in non-native species richness and cover, respectively. In addition, logistic regression models for the seven focal species provided useful tools to characterize habitat preferences for each individual species. On average, these models were able to predict 89% of invaded sites.

Therefore, from the findings of these case studies, it has been shown that some of the most notable invasive plant species within the Monument are highly predictable. Using information on species life history traits and environmental characteristics, managers should have a firm basis to initiate control efforts

Second only to cheatgrass, tamarisk is perhaps the greatest non-native threat that Monument managers will have to face in the immediate future. Although limited to riparian corridors, washes, and springs, tamarisk has been quite successful in displacing native riparian vegetation (e.g. cottonwood, willow) and forming dense monocultures that are having devastating affects on ecosystem processes by altering soil chemistry, hydrology, and wildlife habitat. These affects are most noticeable along the Escalante River, Cottonwood Creek, and Johnson Wash watersheds. In fact, we found very few waterways and washes that were absent of tamarisk restricting our ability to conduct and site comparisons. Calf Creek appears to have been fairly resistant to tamarisk invasion in the past compared to other areas; however, tamarisk infestation has occurred below the falls, and the presence of young sprouts and seedlings suggests that invasion is imminent.

It is suggested that tamarisk mitigation and restoration projects be initiated before the invasion grows further. Mitigation plans should include the participation of surrounding public and private lands, and begin upstream working down. Additional strategies should target new infestations and the prevention of invasion to areas that are still tamarisk free. All of these strategies should also include Russian olive, which can often be found intermixed with tamarisk and also poses a high risk to riparian zones, washes, and springs.

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CHAPTER 5

Disturbance

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Disturbance is a natural process that affects nearly all natural ecosystems. However, altered natural disturbance regimes have changed vegetation and landscape-scale patterns of diversity. Therefore, it is important to document past and current human land use and disturbance in and adjacent to the Monument. Monitoring deviations from the range of natural conditions will provide insight to managers seeking to restore these conditions.

History of Disturbance within the Monument

In 1872, Almon H. Thompson was the first European-American explorer to survey the Monument region, making it the last area to be mapped within the continental United States (Chesher 2000). Although the Monument has remained free of large-scale land development since this time, it has not been isolated from other anthropogenic disturbances. Settlements began in the area as early as the mid 1860s, and the town of Escalante was established in 1875 (Woolsey 1964). These settlements remained largely adjacent to the current Monument's boundary because of its harsh environment and limited water supplies (Grahame and Sisk 2002). As houses and other permanent structures were being constructed, timber was needed. Therefore, logging began in river valleys and high plateaus during this settlement period and continued until the 1940s when many people began leaving the area due to a declining economy (Woolsey 1964).

The lack of water and land suitable for cultivation made it necessary to find more profitable ways to use the land (Grahame and Sisk 2002). Livestock grazing became the primary choice because of the large expanses of suitable rangeland. By 1904, livestock grazing was extensive within the Escalante region with numbers as high as 75,000 sheep and 12,500 cattle on the Aquarius Plateau alone (Woolsey 1964). Livestock permits would not decline until the late 1920s when range conditions began to deteriorate and the United States began entering into the Great Depression (Woolsey 1964; Table 5-1).

Table 5-1. Grazing permits for Escalante, Utah (Woolsey 1964).

Year	Cattle and Horses	Sheep
1922	8,550	23,200
1930	5,550	21,150
1940	5,353	15,600
1950	5,073	5,896
1960	4,807	1,400
1963	2,452	none

In order to improve the deteriorating rangelands, restrictions were placed on cattle grazing by the Taylor Grazing Act of 1934 and reseeding projects begun to help revegetate the area. Between 1959 and 1964, 700 ac had been plowed and seeded, and 3,000 ac had been broadcast seeded within the Escalante region. The Bureau of Land Management was also involved in various range improvements after 1934. Projects have included dividing rangelands among permitees, fencing permit boundaries, and building watering ponds and roads (Woolsey 1964). Since the establishment of the Monument, certain areas have had grazing restricted, but cattle can still be found within its boundaries.

Mining also proved to be a profitable use of the land within the Monument prior to its establishment. Prospectors discovered uranium near the Circle Cliffs in 1898, and numerous mines harvested uranium from this area throughout the 1950s and 1960s. Oil was also found in the Upper Valley Field, located in the northwest corner of the Monument, where 25 million barrels of oil have been removed (Chesher 2000). Large quantities of coal (62 billion tons) have also been found within the Straight Cliffs Formation of the Kaiparowits Plateau (Grahame and Sisk 2002). Although several companies have requested permits to mine this area, large-scale mining has been restricted within the Monument's boundaries.

Case Studies

Case Study 1: *Comparison of a Near-Relict Site and a Grazed Site in a Pinyon-Juniper Vegetation Type*

One way to monitor change is through the use of relict sites. These areas have been left undisturbed and can serve as research controls for comparison to areas affected by anthropogenic disturbance. However, few areas within the United States have been isolated from such disturbances, making relict sites difficult to find. Several studies have used comparisons between disturbed and undisturbed sites with similar geology, slope, aspect, and vegetation type to determine the effects of grazing. Many of these studies focused on vegetation differences (Schmutz et al. 1967; Madany and West 1983; Madany and West

1984), while others looked at impacts to cryptobiotic crust cover (Anderson et al. 1982a; Jeffries and Klopatek 1987; Beymer and Klopatek 1992;Belnap and Gillette 1998). Our objectives were to examine native and non-native species cover and richness, cryptobiotic crust cover, and soil factors on a grazed and near-relict site using large, multi-scale plots (Guenther et al. 2004).

Methods

Study Areas

No Man's Mesa was grazed during 1927 and 1928 by goats (Mason et al. 1967), but cattle have never grazed the site. Although not a true relict site, the Mesa was chosen to be adequate as a control for comparison to a nearby grazed site. Located 40 km northeast of Kanab, Utah, the Mesa is 7 km by 2 km, covers an area of 715 ha, and ranges in elevation from 2,072 to 2,200 m (Fig. 5-1). The dominant vegetation type is pinyon-juniper. Two soil types exist, a sandy upland (479 ha) and sandy shallow breaks (236 ha), but only the area occupied by sandy upland soil was sampled.

The other site, Deer Spring Point, is located 2 km southwest of No Man's Mesa and has similar geology, slope, aspect, vegetation, and elevation (2,011 to 2,228 m). This site has been summer grazed by domestic livestock since the late 1800s.

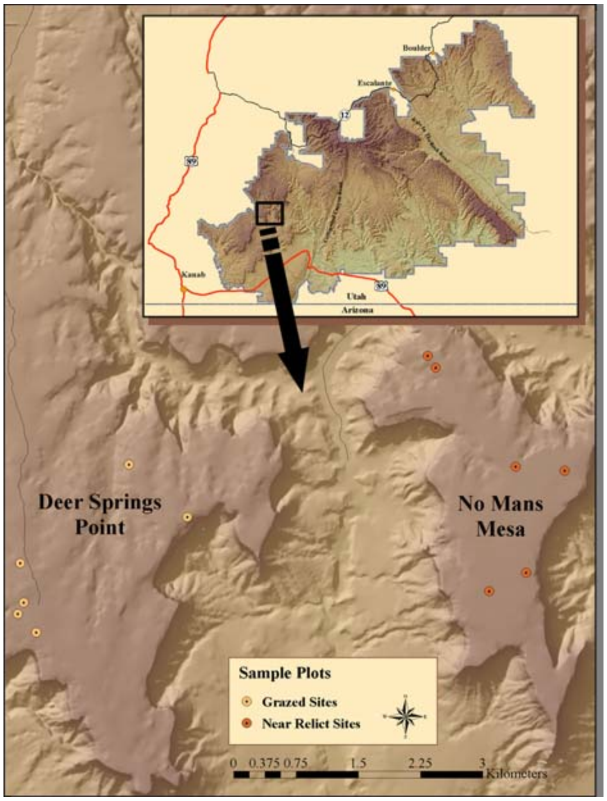

Figure 5-1. Study area for case study 1, including plot locations.

Field Sampling

During June 2001, six modified-Whittaker plots were placed in each treatment type using a systematic random sampling design. Two plots were placed into the northern, central, and southern portions of No Man's Mesa, excluding the sand shallow breaks area. Sites on Deer Spring Point were chosen which corresponded to similar geology type, soil type, vegetation type, slope, and aspect. In addition to data recorded for each plot as noted in the methods chapter of this report (See Methodology), the number of cow pats (>5 cm) in a 1-ha area including and surrounding each plot were counted as an index of recent cattle usage.

Statistical Analyses

All data were checked for normality and skewed data were transformed using a $\log_{10}(x+1)$ transformation prior to analysis. The percent difference in native species richness was calculated using the following formula:

$$\%\ difference = \frac{site\ 1\ species\ richness - site\ 2\ species\ richness}{site\ 1\ species\ richness}\ x\ 100$$

Results and Discussion

Native Species

Mean native species richness at the subplot scale (1 m^2) was not significantly different between the two sites. Deer Spring Point had 7.1 $\pm$ 0.45 species, and No Man's Mesa had 6.5 $\pm$ 0.3 native species. Mean native species richness at the plot scale (1,000 m^2) was greater for No Man's Mesa (44.2 $\pm$ 5.0) than for Deer Spring Point (42.2 $\pm$ 8.0), and this difference was also not significant. This trend differed at the landscape scale (6,000 m^2). Native species richness was significantly greater on No Man's Mesa (93 species) than on Deer Spring Point (75 species; Fig. 5-2). However, when compared to No Man's Mesa, Deer Spring Point had greater total and native vegetation cover (Table 5-2).

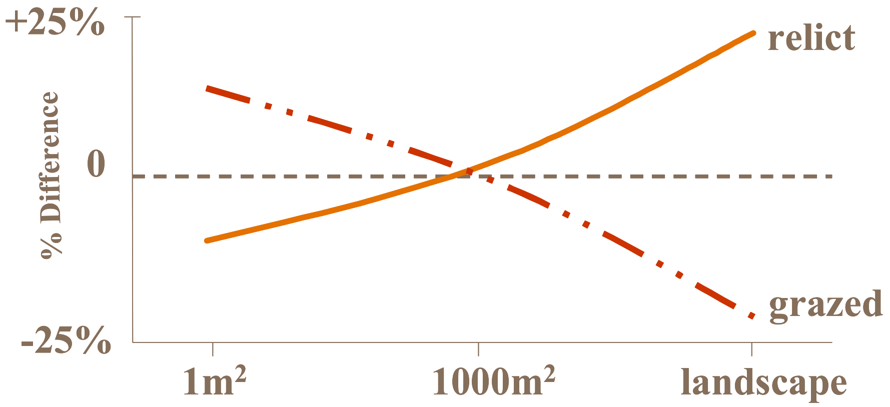

Figure 5-2. Percent difference in mean native species richness for No Man's Mesa and Deer Spring Point at sixty 1-m^2 subplots per site, six 1,000-m^2 plots per site, and the sum of six plots at each site (landscape).

Mean native species richness at the plot scale on both the grazed and near-relict sites was substantially high when compared to other vegetation types within the Monument, far exceeding those of other pinyon-juniper plots (27.1 ± 0.73). Suprisingly, these values also exceeded those of species-rich, mesic vegetation types such as aspen (32.3 ± 3.1) and wet meadow (30.2 ± 4.9; Stohlgren et al. 2001b). We used extrapolations from species accumulation curves to determine the possible number of species per ha for each site. This analysis suggested a possible 109.8 species/ha on No Man's Mesa and 88.6 species/ha on Deer Spring Point. However, xeric mesa tops seem unlikely to support so many species (Harner and Harper 1976) even though a few endemics were encountered. Southwestern beardtongue (*Penstemon laevis*; a regional endemic) and Paria breadroot (*Pediomellum pariense*; a local endemic) were found on No Man's Mesa. These species in addition to Newberry's twinpod (*Physaria newberryi*; a regional endemic) occurred on Deer Spring Point. Such a high species number may be occurring because of the geology type of these areas and their isolation.

Table 5-2. Mean, standard error values, and p-values for two sample t-tests for cover of abiotic variables, vegetation, cryptobiotic crusts, annual and perennial plants, plant life form, and plant species origin (n=60 1-m^2 plots per site) on No Man's Mesa and Deer Spring Point.

Variable	No Man's Mesa	Deer Spring Point	P value
Bare Ground	7.1 (1.4)	13.9 (2.2)	0.010
Litter	30.0 (4.2)	45.8 (4.8)	0.053
Standing Duff	2.7 (0.4)	0.25 (0.1)	0.001
Woody Debris	3.8 (1.0)	6.2 (1.2)	0.004
Total Cryptobiotic Crust	46.3 (4.2)	30.1 (4.0)	0.020
Young Cryptobiotic Crust	36.4 (3.7)	26.6 (3.5)	0.056
Well-Developed Cryptobiotic Crust	9.8 (2.2)	3.5 (1.1)	0.012
Total Vegetation	30.9 (4.0)	44.4 (4.7)	0.081
Annual Plants	0.6 (0.1)	1.4 (0.1)	0.001
Perennial Plants	30.3 (4.0)	43.0 (4.7)	0.140
Forb	2.9 (0.3)	2.4 (0.2)	0.200
Grass	3.5 (0.5)	2.6 (0.4)	0.142
Subshrub	3.1 (0.5)	1.8 (0.4)	0.033
Shrub	3.8 (0.9)	18.9 (3.6)	0.001
Tree	17.6 (4.0)	18.6 (4.1)	0.730
Non-Native Plants	0 (0)	0.02 (0.01)	none
Native Plants	30.1 (3.9)	43.6 (4.6)	0.085

Vegetation Differences

No Man's Mesa and Deer Spring Point are both dominated by perennial species, but there were significantly more annual species found on the grazed site (1.4% ± 0.1%) than on the relict site (0.6% ± 0.1%). No significant differences were found between forb and tree cover, but there was significantly more subshrub cover on No Man's Mesa (3.1% ± 0.5%) than Deer Spring Point (1.8% ± 0.4%; Table 5-2). Deer Spring Point and No Man's Mesa did not differ significantly in grass cover, but there were differences in grass composition (Table 5-2). No Man's Mesa had twice as much cover of muttongrass (*Poa fendleriana*) and needle-and-thread grass (*Stipa comata*); muttongrass being the most dominant species (Jameson et al. 1962; Beymer and Klopatek 1992; Rowlands and Brian 2001).

A major site difference was in shrub cover and richness. Deer Spring Point had significantly greater shrub cover (18.9% ± 3.6%) than No Man's Mesa (3.8% ± 0.9%) with bitterbrush (7.3%), manzanita (4.4%), and big sage (3.7%) being the most dominant species on the grazed site. No Man's Mesa, however, had a greater diversity of shrubs, including six species not found on Deer Spring Point (Mason et al. 1967). The high shrub cover on the grazed site is likely a result of reduced competition from perennial grasses that are removed as a result of grazing preference (Schmutz et al. 1967; Madany and West 1983). Also, the presence of shrubs most likely results in a greater vegetation cover and litter cover on this site compared to No Man's Mesa.

Non-Native Species

Although we hypothesized finding significantly more non-natives on Deer Spring Point as a result of greater disturbance (Stohlgren et al. 2001b), this was not the case. Non-native species were not abundant on either site (Table 5-2). Cheatgrass was the only non-native species found on No Man's Mesa, and only a few individuals were encountered off-plot. Four non-native species were found on Deer Spring Point, including flixweed (*Descurainia sophia*), cheatgrass, crested wheatgrass (*Agropyron cristatum*), and lambsquarter (*Chenopodium album*). Although not highly invaded, it should be noted that these sites have the potential for larger-scale invasion if conditions are favorable for the spread of these already established species.

Cryptobiotic Soil Crusts

Soil texture and nutrients were not found to be significantly different between the two sites (Table 5-3). More litter, bare ground, and woody debris cover occurred on Deer Spring Point, and the site had 27% less young cryptobiotic crust cover and 60% less well-developed cryptobiotic crust cover than No Man's Mesa. These results may be due to cattle trampling on Deer Spring Point, a disturbance which increases soil erosion and reduces the nitrogen-fixing benefits of the crusts (Belnap 1996). Such disturbances have also been shown to facilitate invasion by non-native plant species. Through the removal of vegetation and cryptobiotic crust cover, non-native species are provided the resources and space necessary to establish and become problematic (Stohlgren et al. 2001a). This may also be why more non-native species were found on Deer Spring Point. The loss of older, well-developed cryptobiotic crust cover at this site compared to the near-relict site is also a concern. These crusts take many years to develop, and even a short grazing rotation among sites may reduce their overall cover by not allowing enough developmental recovery time (Anderson et al. 1982b; Beymer and Klopatek 1992).

Table 5-3. Comparison of No Man's Mesa and Deer Spring Point site characteristics. T-tests for soil comparisons had p-values of 0.60, 0.81, and 0.58 for carbon, nitrogen, and phosphorous, respectively.

Characteristic	No Man's Mesa	Deer Spring Point
Geological Substrate	Navajo Sandstone capped by Carmel Fm	Navajo Sandstone capped by Carmel Fm
Elevation Range	2072-2200 m	2011-2228 m
Slope Range	0-5%	0-5%
Aspect Range	North-Northwest	North-Northwest
Plant Vegetation Type	Pinyon-Juniper	Pinyon-Juniper
Soil Texture	Loamy Sand	Loamy Sand
Soil Organic C (%)	0.543	0.657
Soil N (%)	0.047	0.051
Soil P (mg/kg)	2.88	2.45
Sampling Period	20-27 June 2001	23-29 June 2001
Average Pat Count	0	294 ± 143

Case Study 2: Skutumpah Road Disturbed Areas

A unique area is along the Skutumpah Road area in the northwest region of the Monument. Portions of the area have been burned, seeded, chained, and/or treated with Tebuthiuron herbicide. We established 11 paired modified-Whittaker plots (22 total) in an attempt to measure the effects of these different treatments on the plant community and cryptobiotic crusts at the request of the Monument's range and science staff. Since sample sizes are small and the comparisons were not always even, the results of these studies are very preliminary. However, we would like to discuss some interesting patterns that emerged.

It was hypothesized that (1) disturbed plots would have more non-native plant species richness and cover than control plots; (2) disturbed plots would have less cryptobiotic crust cover than the control plots; and (3) disturbed plots (especially burned plots) would have higher levels of nutrients than the control plots (*unpublished data*).

Methods

All plots were sampled in May and June 2000. Field sampling, soil sampling, and soil analyses were conducted as previously described in the methods chapter (See Methodology).

Study Area

Ford Pasture

Ford Pasture contains historical greater sage-grouse (*Centrocerus urophasianus*) lekking grounds. This species has disappeared from most of its historic habitat because of significant reductions in range and population. There is interest as to whether the pinyon-juniper vegetation type is encroaching into sagebrush flats that are suitable habitat for the sage-grouse. Various management practices were applied to this area in the early 1980's that included burning, seeding, chaining, and plowing. We completed three plots in Ford Pasture, two in a treated area (burned and seeded) and one in a control area (Table 5-4).

Swallow Park

This site was chosen because of its high level of ecological diversity in a relatively small area. The site was aerially seeded with non-native species. The reestablishment of cryptobiotic crusts was of interest in this area because it has a history of being chained and/or seeded more than 30 years ago. Therefore, this site might provide insight into the regeneration of cryptobiotic crusts following such disturbances and the effectiveness of aerial seeding. We completed six plots in the area, four in chained and seeded areas and two in control areas (Table 5-4).

Crawford Bench

This site burned approximately 50 years ago and the area was allowed to revegetate naturally. Thus, data could be obtained on natural succession processes of vegetation and cryptobiotic soil crusts. Four plots were completed on Crawford Bench, two each in burn and control areas (Table 5-4).

Calf Pasture

This area was aerially seeded in 1983 to improve livestock forage. As far as we know, there was no other disturbances to the site although chaining was recommended. One plot was completed in the treated area and one in the control area (Table 5-4).

Skutumpah

This area was prescribed burned in 1997. Because some of the burned area was seeded and some area left unseeded, it was a good site to study the efects of post-fire management practices. Six plots were completed here, one plot in the burned but unseeded area, two plots in the burned and seeded area, and three control plots (Table 5-4).

Pine Point

Various seeding and vegetation control methods were used in this area. In 1971, approximately 3,350 ac of this site were chained and drill seeded. Later in 1986, approximately 400 ac of this site burned. Three hundred of these acres were seeded aerially while the remaining 100 ac were drill seeded. Spike (Tebuthiuron) was also used at this site in 1990 to control pinyon-juniper encroachment. One plot was completed in the treatment area and one in the control area (Table 5-4).

Table 5-4. Summary of study areas and number of modified-Whittaker plots completed in each area.

Study Area	Description	Control Plots	Treatment Plots
Ford Pasture	Historic sage grouse strutting ground, burned and seeded in early 1980's	1	2
Swallow Park	High diversity area, burned, chained, and seeded over 30 years ago	3	3
Skutumpah	Burned in 1997, seeded and unseeded	3	3
Crawford Bench	Burned 50 years ago, no other treatment		
Calf Pasture	Burned, chained, and aerial seeded in 1983	1	1
Pine Point	Burned, chained, seeded, spiked with Tebuthiuron over past 10 to 30 years	1	1

Statistical Analyses

No extensive statistical analyses were performed on this data set due to the small sample size and variable control and treatment sites. For the variables measured, means and standard errors were calculated.

Results and Discussion

Ford Pasture

There were many non-native plant species found in Ford Pasture, but this was mostly due to non-native species used in the seed mixtures (Fig. 5-3). However, there was very little cheatgrass in the treatment plots and an unusually high amount of cheatgrass in the control plot (Fig. 5-4). This may or may not be representative of the entire area since time only allowed us to sample one control plot. There were considerably more highly developed cryptobiotic crusts in the control than the treatment areas (Fig. 5-5).

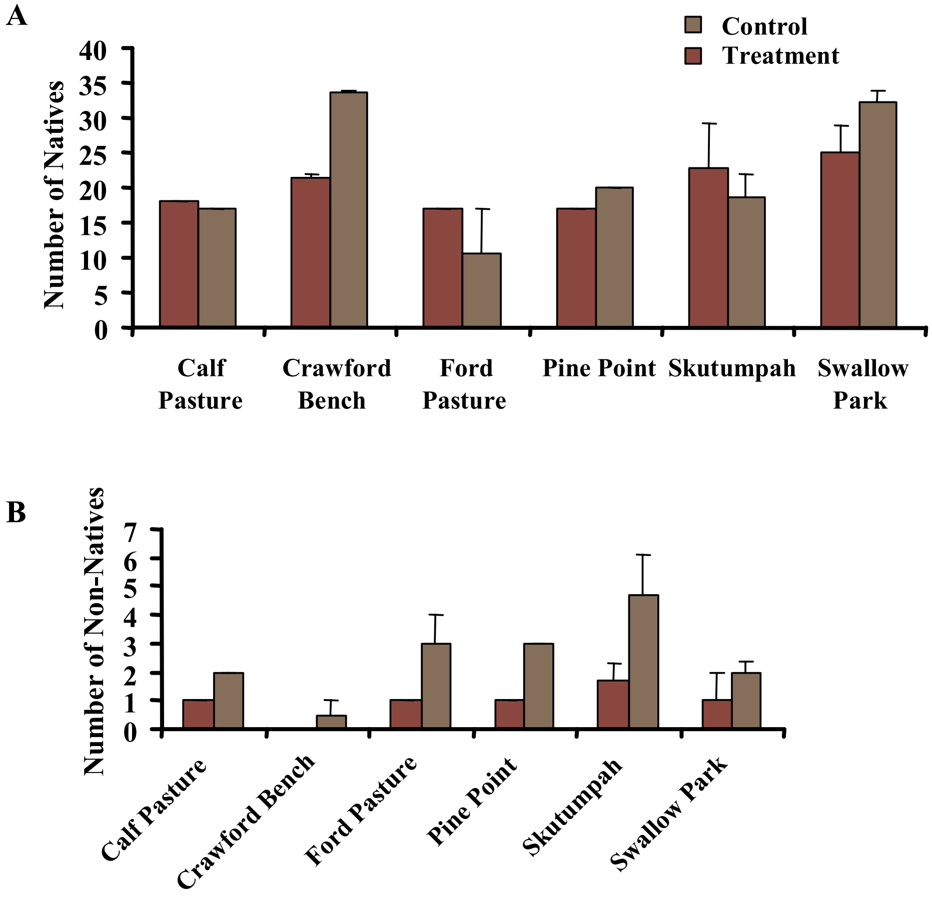

Figure 5-3. Native species (A) and non-native species richness (B) for all sites in the Skutumpah Road study area.

Swallow Park

We found high native species richness in both control and treatment plots in Swallow Park, but also high non-native species numbers and cover in the treatment plots due to non-native seeded species (Fig. 5-3). Cheatgrass was low overall, and in fact even lower in seeded areas (Fig. 5-4). Virtually no well-developed crusts were found in this area but younger crusts were in equal abundance in treatment and control areas (Fig. 5-5).

Crawford Bench

This area is of particular interest because there was only one occurrence of cheatgrass out of 40 1-m^2 subplots (Fig. 5-4). Otherwise, no other non-native species were found in these plots (Fig. 5-3). This is surprising considering the high soil fertility values for these plots (Fig. 5-6). Crawford Bench has high species richness and is dominated by big sage (*Artemisia tridentata*), black sage (*Artemisia nova*), snowberry (*Symphoricarpos oreophilus* var. *utahensis*), Gambel's oak (*Quercus gambelii*), and manzanita (*Arctostaphylos pungens;* Fig. 5-3).

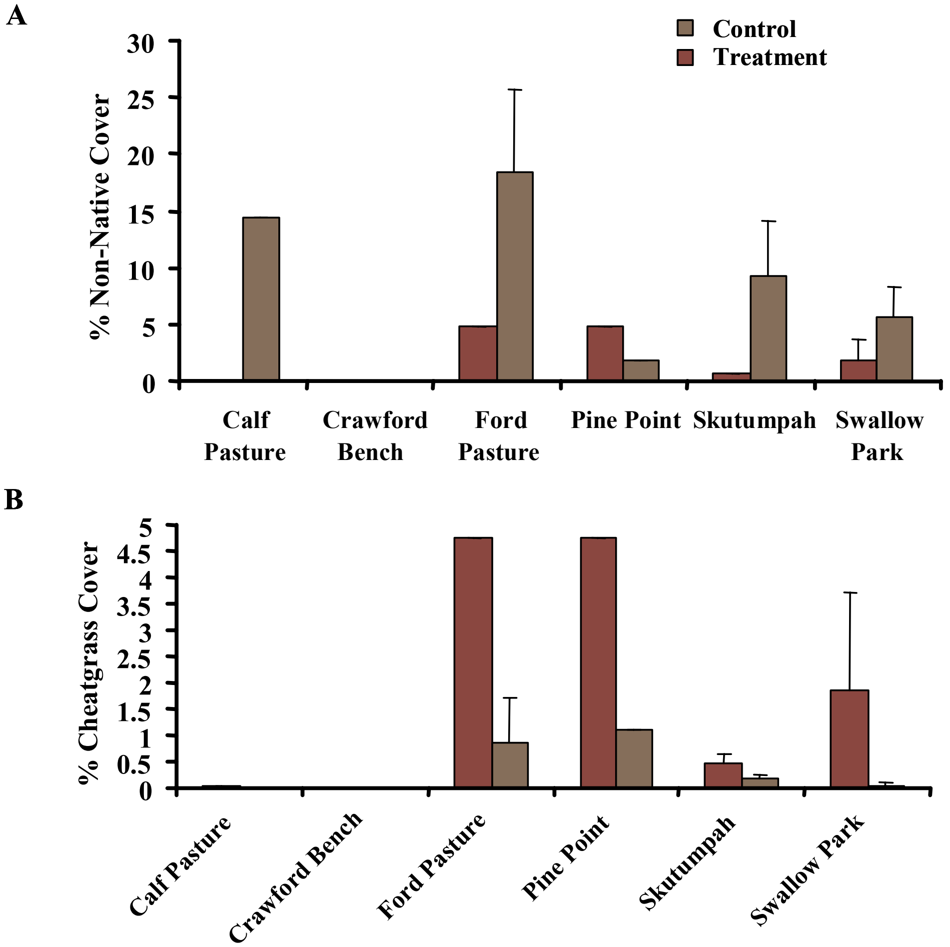

Figure 5-4. Cover of non-natives (A) and cheatgrass (B) for all sites within the Skutumpah study area.

Calf Pasture

These plots had almost no cheatgrass cover and most of the non-native cover is due to the non-native seed mixtures (Fig. 5-4). The control plot had considerably more well-developed crusts than the treated plot (Fig. 5-5). When comparing our data to anecdotal descriptions of the vegetation in 1983 before the seedings, it appears that the vegetation is very similar with the addition of crested wheatgrass (*Agropyron cristatum*) and intermediate wheatgrass (*Elymus hispidus*).

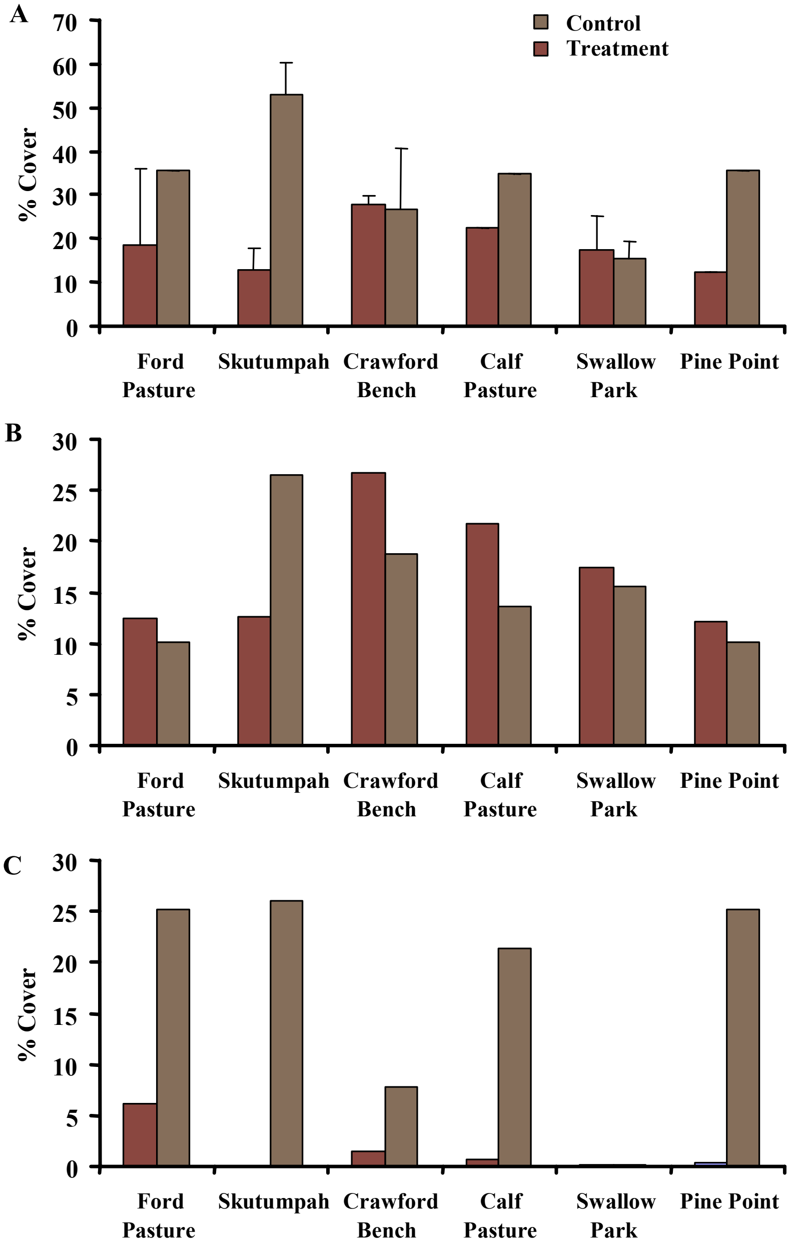

Figure 5-5. Total cryptobiotic soil crust cover (A), cover of young cryptobiotic crusts (levels 1-4; B), and cover of well-developed cryptobiotic crusts (levels 6-20; C) for all sites within the Skutumpah study area.

Skutumpah

The treated plots had a high number of non-native species, mostly due to the non-native seed mixtures, but again, very low cheatgrass levels in both the treated and control plots (Fig. 5-4). There was considerably more cryptobiotic cover in the control than treatment areas including more cover of well-developed cryptobiotic crusts (Fig. 5-5).

Pine Point

Again in the Pine Point treatment area there were more non-native species present due in large part to the non-native seed mixture (Fig. 5-3). The control plot for Pine Point was the same one used for Ford Pasture wherein it had unusually high levels of cheatgrass (Fig. 5-4). Again, this may or may not be representative of the entire area since time only allowed us to record one control plot. The treatment plot did have low levels of cheatgrass. There were considerably more well-developed cryptobiotic crusts in the control than in the treatment plots (Fig. 5-5).

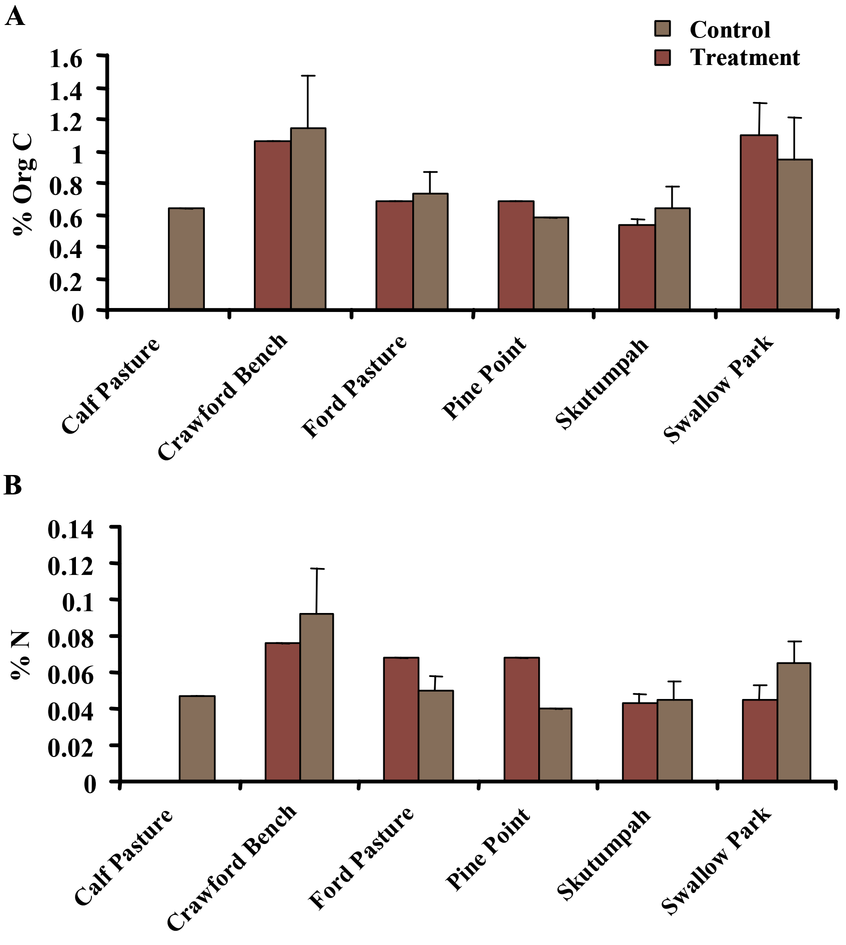

Figure 5-6. Percent organic carbon (A) and nitrogen (B) for all sites within the Skutumpah study area.

Skutumpah Terrace (all plots)

It can be said that in general, there is relatively low cheatgrass cover (0.6%), regardless of treatment, in this area compared to the Monument average (2.0%; Fig. 5-4)). Higher non-native species numbers in the plots are generally explained by non-native species used in the seed mixtures (Fig. 5-3). The treatments, however, seem to cause lower occurrences of well-developed cryptobiotic crusts (Fig. 5-5).

Case Study 3: *Vegetation Response to Fire and Post-Burn Seeding Treatments in Juniper Woodlands*

This case study sought to determine which environmental variables influence the success rate of restoration efforts following fire and post-fire treatments (Evangelista et al. 2004). The primary objective was to provide scientific data for resource managers in the Monument to help guide future restoration efforts while complying with mandates outlined by the Monument's Management Plan (Grand Staircase-Escalante National Monument 1999). This includes the protection of native plant species and cryptobiotic soil crusts.

Methods

Study Area

The study area included Buckskin Mountain, which is located in the south-central portion of the Monument within the Mollie's Nipple Grazing Allotment. Elevations range between 1,645 m to 1,830 m, and the topography is relatively flat (slopes <3%). Soils are of limestone parent materials, and surface textures are gravelly sandy loam with a mixture of gravelly sandy loam and clay loam sub-surfaces (Chapman 1996; Chapman 1997). Although the study area is open to grazing during winter months, grazing is believed to be minimal because of the site's distance from available water sources (Chapman 1996).

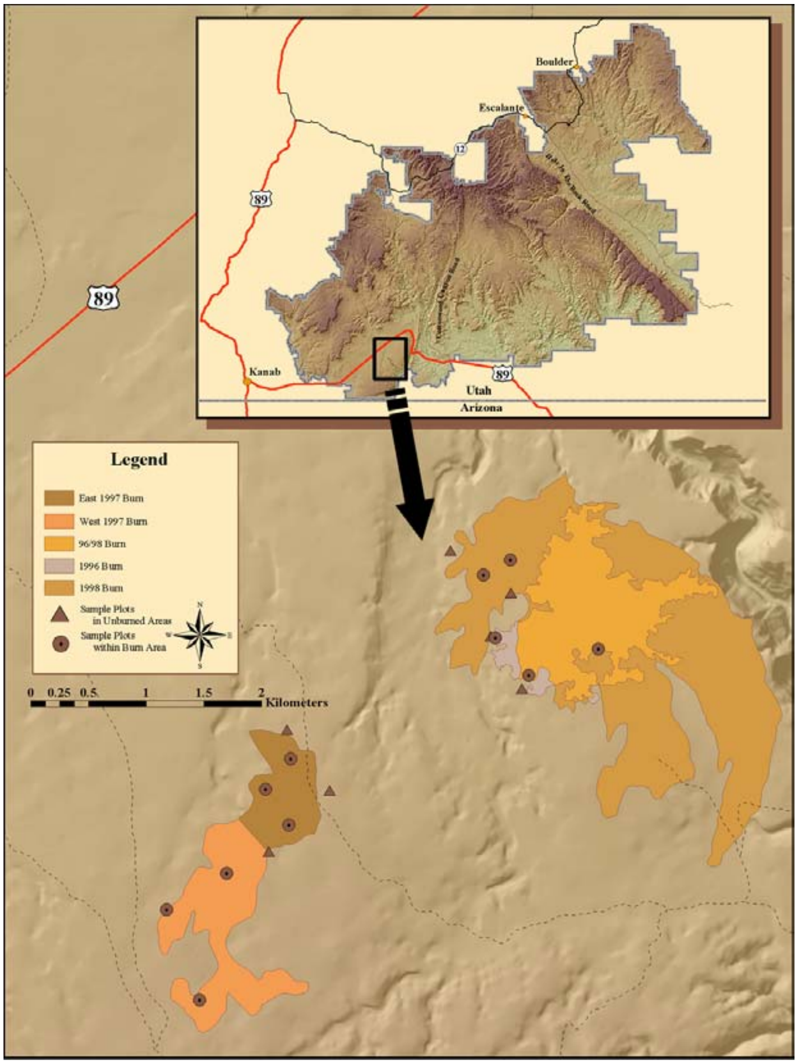

Figure 5-7. Map of Buckskin Mountain study area, including plot locations.

Three naturally ignited burns occurred in the study area between 1996 and 1998 (Fig. 5-7). The first fire occurred on July 14, 1996 and burned approximately 140 ha. The next spring, following the burn, the site was seeded with a native and non-native species seed mixture (Table 5-5). The second fire occurred on July 20, 1997 and also burned approximately 140 ha. Two different native seed mixtures were applied in October of that same year, one to the east side of the burn site and the other to the west side (Table 5-5). For both the 1996 and 1997 burn sites, seeds were planted using a rangeland drill pulled by a bulldozer (Chapman 1996; Chapman 1997). The third fire occurred on August 1, 1998, burning approximately 120 ha and overlapping a portion of the 1996 burn site. This site received no post-burn treatment.

Table 5-5. Seed mixes of each burn applied by a mechanized drill-seeder. Non-native species are highlighted in bold print and species captured in our modified-Whittaker plots are noted with

Seed Mix for 1996 Burn	Seed Mix for 1997-East burn	Seed Mix for 1997-West burn
cliff-rose (*Purshia mexicana*)*	thickspike wheatgrass (*Elymus lanceolatus*)	thickspike wheatgrass (*Elymus lanceolatus*)
four-wing salbush (*Atriplex canescens*)*	Indian ricegrass (*Stipa hymenoides*)*	Indian ricegrass (*Stipa hymenoides*)*
Wyoming big sagebrush (*Artemesia tridentata*)	needle and thread (*Stipa comata*)*	blue bunch wheatgrass (*Elymus spicatus*)*
kochia (*Kochia scoparia*)	blue bunch wheatgrass (*Elymus spicatus*)*	Lewis flax (*Linum perenne* spp. *lewisii*)*
small burnet (*Sanguisorba minor*)*	Lewis flax (*Linum perenne* spp. *lewisii*)*	four-wing saltbush (*Atriplex canescens*)*
yellow sweetclover (*Melilotus officinalis*)*	Palmer penstemon (*Penstemon palmeri*)*	Wyoming big sagebrush (*Artemesia tridentata*)*
crested wheatgrass (*Agropyron cristatum*)*	cliff-rose (*Purshia mexicana*)*	antelope bitterbrush (*Purshia tridentata*)
	four-wing saltbush (*Atriplex canescens*)	
	Wyoming big sagebrush (*Artemesia tridentata*)*	
	antelope bitterbrush (*Purshia tridentata*)	

Field Sampling

During April and May 2000, three modified-Whittaker plots were randomly established within each burned site, burned/seeded site, and adjacent unburned sites. Only two unburned control sites (6 plots) were selected because of logistical constraints. These areas were adjacent to the overlap of the 1996 and 1998 burns and the 1997 burn sites. Field sampling and soil analyses were conducted as described in the methods chapter of this report (See Methodology).

Statistical Analyses

Statistical analyses were conducted using SYSTAT software (Systat Software Inc. 2001), and a p-value < 0.05 to determine significance. All data were checked for normality, and $\log_{10}(x+1)$ transformations were conducted if needed. To see general trends in the data set, the mean number and cover of vegetation by species, the cover of biological soil crusts, and the total vegetative cover for each of the 1-m^2 subplots were calculated. To compare environmental variables among burned and associated unburned sites for all subplots, two-way analyses of variance (ANOVA) were used with Tukey's means comparison test. Discriminant analyses were then used to see if unburned sites were significantly different from burned or burned and seeded sites. Regression trees were also generated to identify significant independent variables that influence non-native species richness and cover within the burned versus unburned sites. Six variables were used in this analysis, including (1) number of native species; (2) cover of native species; (3) disturbance class (1=burned; 2=unburned); (4) total cover of soil crusts; (5) soil nitrogen; and (6) soil phosphorus. Proportion in reduced error (PRE) values were determined to describe the amount of variation explained by the independent variables used in each regression tree (Hansen et al. 1996).

Results and Discussion

1996 Burn Site

A total of 36 species (26 native, 9 non-native) were encountered within this study site. Of this total, two native species and three non-native species had been seeded during post-burn treatments. Although more native species were found on the site, non-native species were twice as frequent as native species, and non-native species cover was five times greater than native species cover on average within the subplots (Table 5-6). The non-native species, cheatgrass, had the highest average cover within the burned sites (8.6%), and occupied 42% of the total vegetation cover within the subplots. Another non-native, crested wheatgrass (*Agropyron cristatum*; seeded), had the second highest average cover (3.9%) and 22.0% of total vegetation cover within the burned site. Cryptobiotic soil crusts only averaged 1.9% cover within the burned subplots.

Unburned control plots had 30 total species, 28 native and two non-native. Within the subplots, native species richness was three times higher than non-native species richness, and native species cover was almost 25 times higher than non-native species cover (Table 5-6). Juniper was the most frequently encountered species, averaging 19.8% cover per subplot and occupying 66.0% of the total vegetation cover. The second most encountered species was big sagebrush, with an average 5.6% cover per subplot and 18.0% of total vegetation cover. Although cheatgrass had the third highest cover value in the unburned plots, it only averaged 1.2% cover per subplot and 14.0% of total vegetation cover. Cryptobiotic soil crusts were more abundant in the unburned plots, averaging 28.4%.

From these results, species richness was significantly greater in the burned plots than the unburned plots at the 1996 study site. However, 25.0% of the total number of species were non-native in the burned plots while only 6.0% of the total number of species were non-native in the unburned plots. Non-native species cover was also significantly greater on the burned site than the unburned site and also greater than any other site in this study.

Table 5-6. Mean number of vegetation species and percent cover, and biological soil crusts of 1-m^2 subplots within burned and unburned sites. Standard errors are in parentheses and significant differences are noted by (*).

Indices	Total Species	Non-Native Species	Native Species	Total Cover	Non-Native Cover	Native Cover	Crusts Cover
1996 Burn/Native and Non-Native Seed	4.4 (0.3)	3.0 (0.2)	1.4 (0.2)	17.4 (1.4)	14.8 (1.5)	2.7 (1.0)	1.9 (0.6)
96/98 Unburned Sites	3.7 (0.3)	0.8 (0.08)	2.9 (0.3)	29.6 (5.9)	1.2 (0.4)	28.5 (5.8)	28.4 (5.3)
P-value	< 0.1	<.001*	<.001*	0.05*	<.001*	<.001*	<.001*
1997 West Burn/ Native Seed	3.2 (0.2)	1.6 (0.1)	1.6 (0.2)	25.6 (2.5)	12.2 (1.5)	13.4 (2.5)	0.6 (0.1)
1997 East Burn/ Native Seed	4.1 (0.3)	1.1 (0.07)	3.0 (0.3)	27.5 (2.5)	4.7 (0.7)	22.8 (2.6)	3.2 (0.9)
P-value	0.009*	<.001*	<.001*	0.59	<.001*	0.01*	<.001*
1997 East Burn/ Native Seed	4.1 (0.3)	1.1 (0.07)	3.0 (0.3)	27.5 (2.5)	4.7 (0.7)	22.8 (2.6)	3.2 (0.9)
1997 Unburned Sites	4.0 (0.2)	1.1 (0.08)	2.9 (0.9)	36.9 (4.2)	20.4 (3.3)	16.5 (2.4)	1.3 (0.4)
P-value	0.004*	0.001*	<.001*	0.02*	0.03*	0.04*	0.1
1997 West Burn/ Native Seed	3.2 (0.2)	1.6 (0.1)	1.6 (0.2)	25.6 (2.5)	12.2 (1.5)	13.4 (2.5)	0.6 (0.1)
1997 Unburned Sites	4.0 (0.2)	1.1 (0.08)	2.9 (0.9)	36.9 (4.2)	20.4 (3.3)	16.5 (2.4)	1.3 (0.4)
P-value	0.004*	0.001*	<.001*	0.02*	0.03*	0.04*	0.1
1998 Burn/Natural Regeneration	2.9 (0.2)	1.5 (0.1)	1.4 (0.2)	12.7 (1.1)	7.5 (0.8)	5.2 (1.1)	10.4 (2.8)
96/98 Unburned Sites	3.7 (0.3)	0.8 (0.08)	2.9 (0.3)	29.6 (5.9)	1.2 (0.4)	28.5 (5.8)	28.4 (5.3)
P-value	0.07	<.001*	<.001*	0.01	<.001*	<.001*	0.02*

1997 Burn Site

This study site was divided into East and West sites as a result of different seed mixtures being used on different portions of the burned area. Thirty total species were encountered in the eastern portion of this study site (i.e., East site), including 26 native species and four non-native species. Seven of the native species that were encountered were present in the seed mixture used in the post-burn treatment. Within the subplots, native species richness and cover was greater than non-native species richness and cover (Table 5-6). Cheatgrass was encountered most frequently, having 9.1% cover on average and occupying 42.0% of the total vegetation cover. The native species, smallflower globemallow (*Sphaeralcea parvifolia*), was the second most frequently encountered species, averaging 7.1% cover per subplot and occupying 26.0% of the total vegetation cover. Cryptobiotic soil crusts covered 3.2% of each subplot on average in the burned site, but only covered 1.3% in the unburned site.

Within the western portion of the 1997 burn site (i.e., West site), 29 species were encountered. Of the 24 native species that were recorded, five had been seeded during the post-burn treatment. Within the subplots, 3.2 species (50% native, 50% non-native) were encountered on average (Table 5-6). Native and non-native species also had an equal distribution of cover. Smallflower globemallow had the greatest average cover within the subplots (13.1%) and a 51.0% total vegetation cover. Broom snakeweed (*Gutierrezia sarothrae*) had the second highest cover value, occupying 6.2% of a subplot on average and 24.0% of the total vegetation cover. Cheatgrass had the next highest cover, averaging 4.5% per subplot and having a total vegetation cover of 21.0%. Cryptobiotic soil crusts averaged only 0.6% cover per subplot in this burned site.

The unburned sample plots had 31 total species, 28 native species and three non-native species. Within the subplots, native species richness was greater than non-native species richness, but non-native species cover was slightly higher than that for native species. Cheatgrass had the highest average cover within these subplots (19.6%) and occupied 51.0% of the total vegetation cover. Big sagebrush had the next highest cover (10.8%) and occupied 26.0% of the total vegetation cover.

No significant differences in total species richness and cover existed between the East site and the unburned site at the subplot scale. However, this burned site had significantly less non-native species cover than the unburned site. The West site had significantly fewer total species and native species, and significantly more non-native species than the unburned plots. Total cryptobiotic soil crust cover was not significantly different between the East site or the West site and the unburned control site.

Although the post-burn seed mixtures used for both portions (East and West) of the 1997 burn site were similar, average total species was significantly higher on the East site. The East site also had higher native species richness and cover and less non-native species richness and cover than the West site. Although cryptobiotic soil crust cover was significantly less in the western portion of the study area, it was also substantially low in the unburned plots.

Role of Site Factors in Post-Fire Recovery

The results of the 1997 burn demonstrate that site factors (e.g., soil nutrient content, native plant cover) are as important or may be more important than post-burn seeding in regards to the success of restoration treatments. The East and West sites of the 1997 burn received similar post-burn seeding treatments of native species (Table 5-5), but their responses to these treatments differed significantly (Table 5-6). The East site had the most successful seeding treatment and represented a more natural array of vegetation. In contrast, the West site had an unsuccessful seeding treatment and remained high in non-native species richness. Surprisingly, the 1997 unburned sites had more non-native species cover than both the burned sites. The unburned sites also had lower cryptobiotic soil crust cover than the East site, which was mechanically seeded.

Therefore, despite similar physical properties among these sites, there were a number of other variables that played significant roles in determining the successional trend following fire and the success of post-burn rehabilitation. This may be a result of such environmental variables also affecting the distributions of native and non-native plant species.

However, negative effects (e.g., increased non-native species, decreased native species, decreased crypto soil cover) can still be strongly associated with burning and restoration activities (Table 5-7).

Table 5-7. Variable means in 1-m^2 subplots within unburned plots and burned and/or seeded plots. Standard errors are in parentheses and P-values of each t-test are noted with (*) when significantly different.

Variables	Unburned	Burned or Burned and Seeded	P-value
n	60	120	
Number of Native Species	2.9 (0.2)	1.8 (0.1)	*0.001**
Number of Non-Native Species	1.0 (0.1)	1.8 (0.1)	*0.001*
% Native Cover	22.5 (3.2)	11.0 (1.2)	*0.001**
% Non-Native Cover	10.8 (2.1)	9.8 (0.7)	*0.65*
% Cheatgrass Cover	10.4 (2.0)	7.3 (0.6)	*0.14*
Sum of Crust	15.0 (3.2)	4.1 (0.8)	*0.001*
% Soil C	1.1 (0.1)	1.2 (0.1)	*0.18*
Soil P (mg per kg soil)	17.6 (1.2)	17.3 (0.9)	*0.83*
% Non-Native Cover of Total Cover	32.7 (4.3)	55.0 (3.0)	*0.001**

1998 Burn Site

A total of 30 species were encountered at this site, with an average of 2.9 species per subplot. Five of the 30 species encountered were non-native, but an equal number of native and non-native species were encountered within each subplot on average. However, average cover was greater for non-native species compared to native species. Once again, cheatgrass had the highest average percent cover (7.0%) and a high total vegetation cover (55.0%). Smallflower globemallow had the next highest average cover of 2.9% and a total vegetation cover of 22.0%. Cryptobiotic soil crust cover was 10.4% on average.

The burned plots had significantly more non-native species and significantly less native species than the unburned control plots. Similarly, non-native species cover was significantly higher and native species cover and cryptobiotic soil cover significantly lower in the burned plots compared to the control plots.

Comparing Burned and Unburned Sites

Several differences existed between the burned and unburned plots in this case study. In general, native species richness, native species cover, and cryptobiotic crust cover were highest on unburned plots compared to burned plots (Table 5-7). The regression tree for non-native species richness (combined for all sites) identified five independent variables that were significant (Fig. 5-8), and they explained 63.0% of the variation in the model. Native species cover was the most influential variable and was negatively correlated with non-native species richness. Fire or fire and seeding treatments, soil carbon, and the average number of native species present were other variables that can be used to predict non-native species richness. Thus, non-native species richness should be highest in areas that have been burned and contain low native species cover, high native species richness, and high soil carbon.

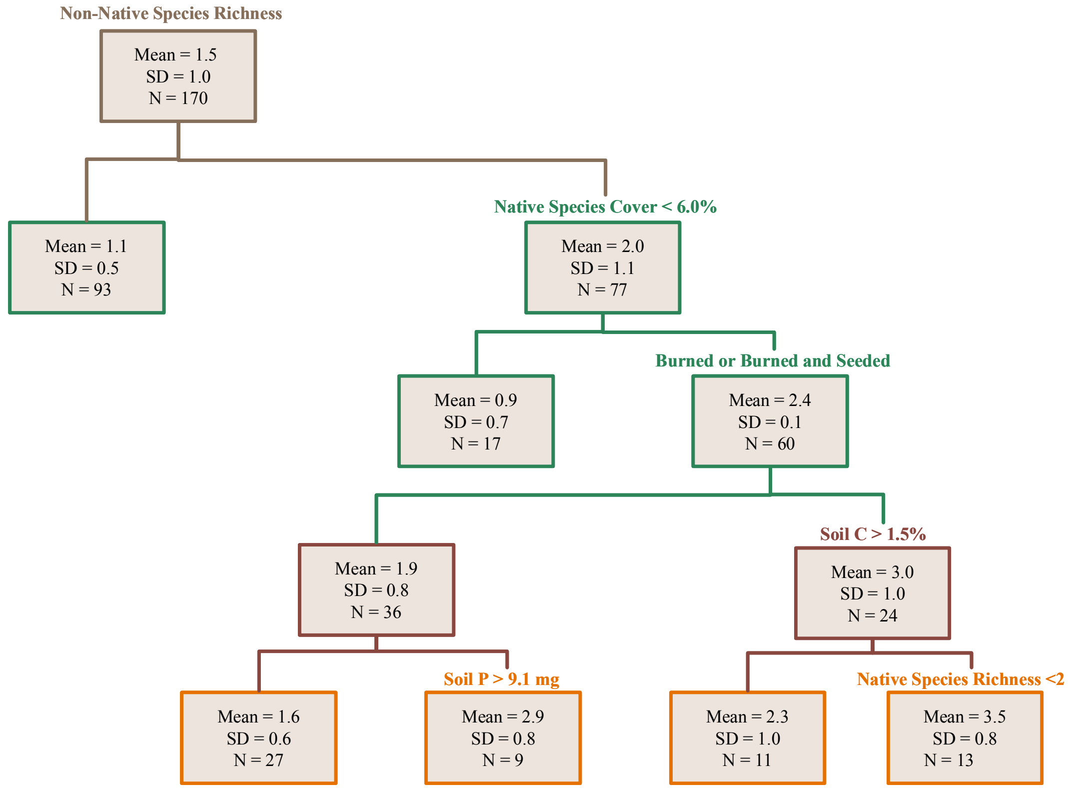

Figure 5-8. Regression tree analysis for predicting the mean number of non-native species in 1-m^2 subplots of the burned and unburned sites. The number of species and percent cover are expressed as means, standard deviations are in parentheses, and PRE = 0.63.

The regression tree for non-native species cover found three variables to be significant, explaining 67.0% of the variation in the model (Fig. 5-9). It is suggested by this model that low native species richness facilitates higher non-native species cover. In addition, non-native species will proliferate in areas of high soil C and may out compete native species for soil P.

Based on soil C, cryptobiotic soil crust cover, soil crust by developmental stage, richness and cover of non-natives, and richness and cover of natives, it was shown by discriminant analysis that burned plots could be distinguished from unburned plots 83.0% of the time. These results justified using regression tree analysis on these two groups.

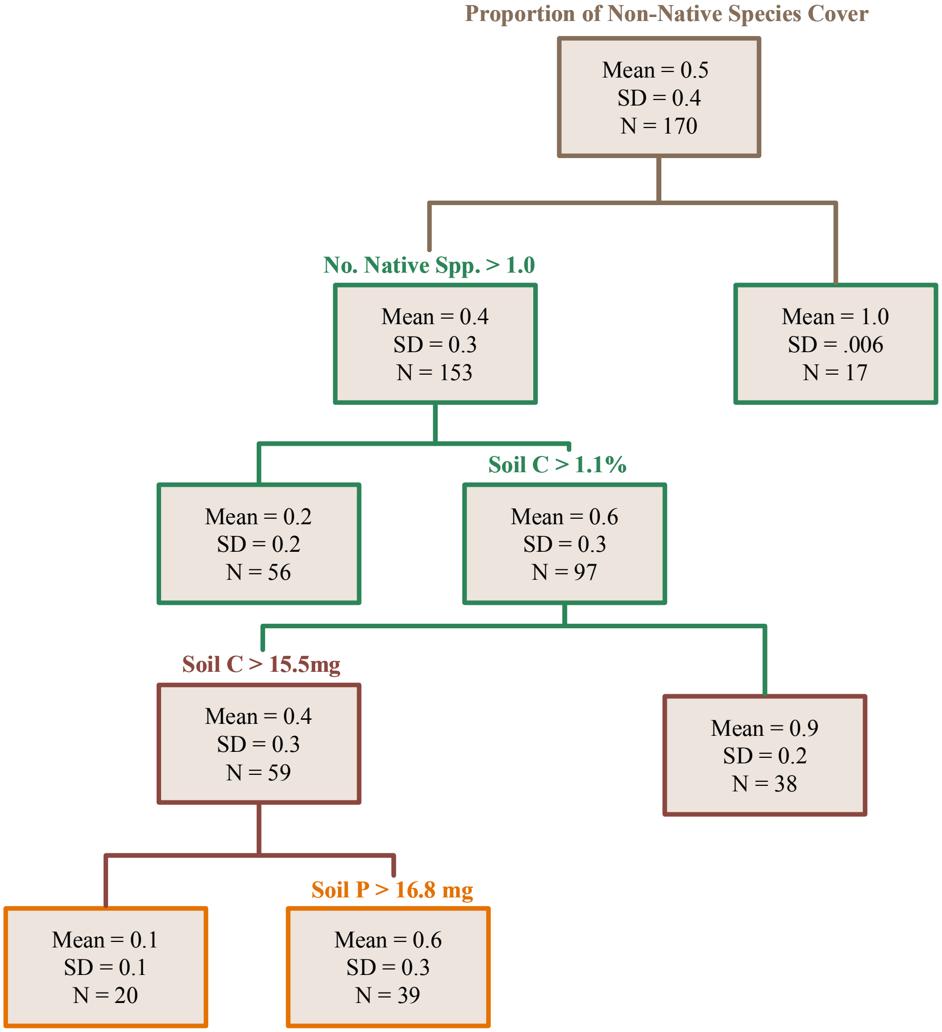

Figure 5-9. Regression tree analysis for predicting the mean percent cover of non-native species in 1-m^2 subplots of the burned and unburned sites. The number of species and percent cover are expressed as means, standard deviations are in parentheses, and PRE = 0.67.

A regression tree for non-native species richness was generated for all the burned plots combined (Fig. 5-10). The model was able to describe 62.0% of the variation in non-native species richness and found that non-native species richness is greatest in areas of low native species cover, high soil C, and high native species richness. A regression tree for non-native species richness was also generated for all the unburned plots combined, and recognized three significant independent variables. They were able to explain 44.0% of the variation in non-native species richness. Cryptobiotic soil crust cover was the most significant variable tested, and was negatively correlated with non-native species richness.

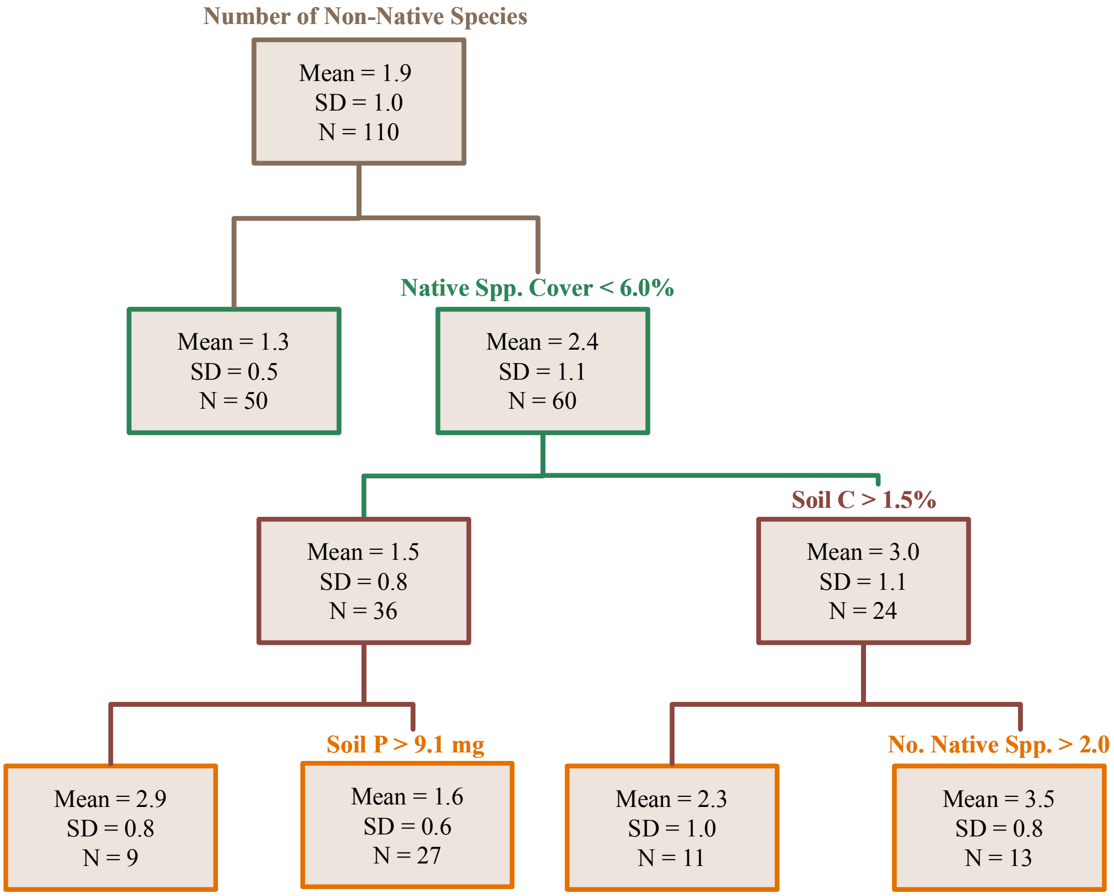

Figure 5-10. Regression tree analysis for predicting the mean number of non-native species in the 1-m^2 subplots of the burned or burned and seeded sites. The number of species and percent cover are expressed as means, standard deviations are in parentheses, and PRE = 0.62.

The regression tree models for non-native species cover for the burned and unburned sites were both weak. They only predicted 32.0% and 29.0% of the variation in non-native cover, respectively. Native species cover and soil carbon were the most significant variables tested for the burned sites while cryptobiotic crust cover was the most significant variable for the unburned sites.

Site Factors and Non-Native Plant Invasion

When comparing burned and unburned sites, some surprising findings emerged. Although cryptobiotic crust cover, native species richness, and native species cover were greater for the unburned sites as would be expected, non-native species cover was also significantly higher on the unburned sites than the burned sites. Each site has complex factors that may have been associated with these results, but such factors were not examined by this case study. For example, climatic variability, additional nutrient content of soils, livestock grazing, and other management activities are likely to have influenced each site in different ways. Post-burn seeding treatments have been shown to attract livestock, which might accelerate trampling of cryptobiotic soil crusts and grazing of native species.

Cheatgrass was the non-native species of greatest concern among all the sites. It had the greatest vegetation cover on four of the study sites and had the third highest cover on the other two study sites. It is probable that the burn sites are providing ideal conditions for germination of non-native species, especially cheatgrass. Once these plants have established within these sites, they are capable of facilitating local seed production and invasion into adjacent unburned areas. However, no insight can be provided on how cheatgrass might affect future successional trends due to the short duration of this study and the short interval between the burns examined. Close examination of the 1998 burn site reveals a pattern that provides evidence that cheatgrass can increase fire frequency and the spatial extent of a fire. These conditions are more favorable for cheatgrass to become dominant. Under natural conditions, the 1998 fire would most likely not have spread across the 1996 burn site, which resulted in a two-year fire return interval. Because cheatgrass has become a successful dominant species in the surrounding unburned woodlands, we can expect a positive feedback loop between its widespread presence and reoccurring fire unless a means of controlling this non-native species is identified.

Need for Experimentation

This case study was an observational survey following a series of events. To isolate the effects of fire, restoration, and grazing on the use of post-treatment effects, controlled experiments will be needed. With the variability expressed in Table 5-6, it is likely that several replicate plots will be required at multiple study areas. There should also be a minimal distance established between burned and unburned plots so that the effects of source-sink dynamics and propagule pressure can be reduced.

Even with these limitations in our study, some obvious differences between burned and unburned sites can be noted and results from restoration efforts could be seen (Table 5-7). Restoration efforts can have both positive and negative effects to treated areas depending on site factors and seeding treatments. As found in the 1997 East burn site, seeding with native species when conditions are favorable can have positive effects on native species richness and cover after fire. As expected, seeding with non-native species was not beneficial to native species richness and cover. As seen in the 1996 burn site, non-native species cover was more than ten times greater than the unburned sites. In the interst of preserving soil crusts, natural regeneration may be preferable to drill seeding; however, the presence of local non-native seed sources leaves burned areas vulnerable to invasion (Table 5-6).

Therefore, controlled experiments should target restoration techniques that are not disturbing to cryptobotic soil crusts and that promote native species richness and cover. As non-native plant species continue to proliferate, the protection of these crusts and native species will be more difficult. While it will be no small feat to slow invasion by non-native species, results of the 1997 East burn have shown that rehabilitation efforts can be worthwhile.

Management Implications

The Monument has a challenge to manage its lands for multiple uses. Therefore, Monument managers, have to develop ways to protect natural and cultural resources while providing for recreation, grazing, mineral exploration, and natural fire regimes on the landscape (Grand Staircase-Escalante National Monument 2000). Managers should be most concerned with landscape-scale processes and the modified-Whittaker design aids in identifying different processes at this and smaller scales. Although the grazed site in case study 1 had greater species richness than the near-relict site at the subplot scale (1 m^2), species richness was significantly different at the landscape scale (6,000 m^2). Thus, grazing does not appear to be directly influencing non-native plant invasions in this study area. However, grazing has the potential to indirectly affect future invasions, possibly resulting in a large-scale invasion.

Because greater shrub and litter cover were found within the grazed site, fuel loads may be great enough to increase fire frequency and promote the spread of already established non-native plant species. Results from data collected near Buckskin Mountain have indicated that fire can increase the cover of cheatgrass and other non-native species if a seed source is present (Evangelista et al. 2002b). This species was found on both sites. Because this species proliferates following fire (D'Antonio and Vitousek 1992), subsequent fires could result in the spread of this species throughout these sites.

Case study 2 found that mechanical seeding and chaining can be detrimental to highly developed cryptobiotic crusts and should be minimized as management techniques. From the findings of these disturbed sites, it is also recommended that non-native seeds no longer be used in reseeding areas and aerial seeding be considered over mechanical.

Case study 3 has restated what other studies have found in relation to post-burn succession and non-native plant invasions (D'Antonio and Vitousek 1992; Stohlgren et al. 2001). The complexity of natural systems can affect recovery from fire, successional trends, and rehabilitation efforts. Three recommendations for Monument managers can be taken from this study. Because mechanized seeding disturbs fragile cryptobiotic crusts and may promote non-native plant invasions, alternate rehabilitation techniques should be considered (e.g., aerial seeding, natural revegetation). When using seeding as a post-fire restoration treatments, only native species should be used whenever possible. Also, long-term monitoring of these sites and predictive models may help guide future management actions and increase our knowledge of successional changes in juniper vegetation types following some of these disturbances. In the absence of detailed monitoring and consideration of multiple site factors, managers will have difficulty prescribing successful post-fire treatments.

CHAPTER 6

Modeling

Nathaniel W. Alley, Mohammed A. Kalkhan, and Thomas J. Stohlgren

Only a small fraction of the Monument can be affordably sampled; therefore, spatial predictive models are important tools that can be used to extrapolate findings to the larger un-sampled landscape. The models can assist in planning, teaching and training, and public outreach. We identified hotspots of native and non-native plant species diversity, cryptobiotic crust cover, and soil characteristics using two different techniques as described in the following case studies.

Case Studies

Case Study 1: *Iterative Model Development for Natural Resource Managers*

This study sought to quantify patterns of native plant species cover, cryptobiotic crust cover, and cheatgrass cover relative to environmental variables using field data on vegetation, soils, and cryptobiotic soil crusts to provide land managers scientific data from which they may base their conservation efforts. We also took a correlative approach to examine variables under which non-native plants have successfully established in the Monument. The specific objectives of this study were (1) to use a comprehensive geographic information system (GIS) to evaluate patterns of cheatgrass relative to native species cover and cryptobiotic soil crust cover; (2) to create predictive spatial maps showing the estimated percent cover of native plant species, cryptobiotic soil crusts, and cheatgrass; and (3) to use statistical analyses to quantify these patterns and determine which variables are correlated to these patterns (Alley et al. 2003 *in press*).

Methods

This study included a subset of the plots (N = 367) sampled, but included all 19 vegetation types. Field sampling, soil sampling, and soil analyses methods were identical to those described in the methods chapter (See Methodology). A GIS database was created for this study using coverages that consisted of several independent variables composed of data collected in the field. These variables are considered to influence variability in plant species richness and the cover of non-native plant species. The coverages of the GIS database included a 10 m Digital Elevation Model (DEM; Department of the Interior; USGS) used to create a 10 m grid of slope and aspect (ARC/INFO; ESRI 1997) and a point coverage based on the UTM location of each plot. This point coverage was generated in ArcView by joining to an MS Access database using a Structured Query Language (SQL) connection.

Elevation, slope, and aspect for each individual plot was then derived based on the DEM coverage and associated grids. Plot data were spatially joined to coverages of geologic parent material and soil type to determine these values for each of the plots. The information derived from these analyses was exported to an Access database to conduct all statistical analyses (Systat Software Inc. 2001).

Statistical Analyses

Based on the field data from the ten 1-m^2 subplots within each plot, trend surface models were developed for each of the cover types of interest (i.e., the percent native plant species cover, percent total cryptobiotic crust cover, and percent cheatgrass cover). This analysis produced predictive maps that estimate the percent cover for each cover type using an Inverse Distance Weighting (IDW) method. This method predicts values for all unmeasured locations, assuming that each measured point has a local influence on unmeasured locations (i.e., predicted values) that diminishes with distance. Regression trees were then developed to further examine the variables that influence hotspots for each cover type of interest.

To perform regression tree analysis, all variables were first assessed for normality and transformed using $\log_{10}(x + 1)$ transformations when needed. Regression trees for native plant species, total cryptobiotic crust, and cheatgrass cover were developed from subplot data based on eight independent variables (i.e., percent total cryptobiotic crust cover, percent native plant species cover, percent cheatgrass cover, elevation, percent sand, percent clay, percent inorganic carbon, percent nitrogen, and phosphorus (ppm)). The dependent variable was removed from the list of independent variables when creating each tree. Proportion of reduction in error (PRE) values (similar to R^2 values) were calculated to determine the amount of variation explained by the independent variables in the model (Hansen et al. 1996).

Results and Discussion

General Statistics

There was much variation in native and non-native plant species cover and cryptobiotic crust cover among the 19 vegetation types identified in this study. Average native species cover ranged from 15.4 ± 1.3% to 42.6 ± 2.9%, and average cryptobiotic crust cover ranged from 11.9 ± 2.6% to 65.4 ± 2.1%. Average non-native species cover ranged from 0.1 ± 0.0% to 27.5 ± 5.0%, and average cheatgrass cover ranged from 0.0 ± 0.0% to 7.4 ± 2.1% (Table 6-1).

Table 6-1. Basic statistics for native, non-native, cheatgrass, and total cryptobiotic crust cover used in the regression tree analysis. Minimum and maximum values for each variable are in bold and standard errors are shown in parentheses.

Vegetation Type	N	Native Cover (%)	Non-Native Cover (%)	Cheatgrass Cover (%)	N (Crypto)	Total Crypto Cover (%)
Desert Shrub	290	16.3 (1.0)	2.0 (0.3)	1.5 (0.3)	290	57.3 (1.7)
Blackbrush	270	29.2 (1.4)	2.5 (0.4)	2.3 (0.4)	270	50.6 (1.8)
Desert Shrub/ Grassland	160	21.5 (1.6)	2.2 (0.3)	1.7 (0.3)	160	48.1 (2.4)
Sagebrush	310	24.6 (1.3)	3.1 (0.4)	2.2 (0.4)	310	55.9 (1.7)
Juniper	220	16.6 (1.6)	0.4 (0.1)	0.4 (0.1)	290	**65.4 (2.1)**
Juniper/Sage	170	24.7 (1.8)	7.4 (0.9)	6.6 (0.9)	170	36.6 (2.2)
Disturbed Pinyon-Juniper/Sage	280	**15.4 (1.3)**	12.6 (0.9)	7.1 (0.8)	280	38.9 (1.5)
Pinyon-Juniper/ Sage	210	31.2 (2.1)	0.9 (0.2)	0.9 (0.2)	210	54.4 (2.1)
Pinyon-Juniper	**880**	24.3 (1.1)	1.4 (0.2)	0.9 (0.1)	**880**	45.2 (1.1)
Pinyon-Juniper/ Oak	180	**42.6 (2.9)**	0.5 (0.1)	0.3 (0.1)	180	38.5 (2.4)
Pinyon-Juniper/ Manzanita	60	35.8 (3.8)	**0.1 (0.0)**	0.1 (0.0)	60	39.1 (4.6)
Pinyon Pine	40	35.3 (6.2)	1.3 (0.4)	1.3 (0.4)	40	40.1 (6.0)
Mountain Shrub	110	34.6 (2.9)	1.4 (0.5)	1.4 (0.5)	110	39.3 (2.8)
Ponderosa Pine/ Manzanita	70	37.8 (4.5)	0.8 (0.5)	0.8 (0.5)	70	29.3 (3.8)
Rabbitbrush	90	29.7 (3.1)	4.5 (1.5)	4.5 (1.5)	90	46.1 (3.1)
Aspen	60	57.6 (4.4)	**7.4 (2.1)**	**7.4 (2.1)**	60	**11.9 (2.6)**
Wet Meadow	30	37.0 (5.2)	2.0 (0.8)	2.0 (0.8)	30	16.4 (5.6)
Spring	**20**	29.9 (7.5)	**0.0 (0.0)**	**0.0 (0.0)**	**20**	30.3 (7.0)
Perennial Riparian	220	35.3 (2.6)	1.6 (0.4)	1.6 (0.4)	220	36.4 (2.4)

There was also a lot of variation that existed among vegetation types for soil characteristics. Percent clay of soils, which greatly affects water-holding capacity among vegetation types, ranged from 8.5 ± 0.4% to 21.4 ± 1.2%. Soil nutrients (N, P, and inorganic C) ranged from 0.0 ± 0.0% to 0.9 ± 0.0% for percent nitrogen, 2.2 ± 0.1 to 24.4 ± 0.8 ppm of phosphorus, and 0.1 ± 0.0% to 1.0 ± 0.0% for percent inorganic carbon (Table 6-2).

Table 6-2. Basic statistics for soil data used in the regression tree analysis. Minimum and maximum values for each variable are in bold and standard errors are shown in parentheses.

Vegetation Type	N (Soil)	% Sand	% Clay	% Inorganic Carbon	% N	P (ppm)
Desert Shrub	280	68.0 (1.5)	**21.4 (1.2)**	0.9 (0.1)	**0.0 (0.0)**	4.7 (0.2)
Blackbrush	240	80.1 (0.1)	13.9 (0.5)	0.8 (0.0)	**0.0 (0.0)**	4.8 (0.2)
Desert Shrub/ Grassland	150	69.4 (1.3)	18.7 (1.0)	0.7 (0.1)	**0.0 (0.0)**	8.9 (0.6)
Sagebrush	310	69.0 (1.1)	16.0 (0.5)	0.7 (0.0)	0.1 (0.0)	8.5 (0.3)
Juniper	220	73.2 (1.3)	15.8 (0.7)	0.9 (0.1)	**0.9 (0.0)**	3.5 (0.1)
Juniper/Sage	150	62.3 (1.2)	20.5 (0.7)	0.7 (0.1)	0.5 (0.1)	13.3 (0.7)
Disturbed Pinyon-Juniper/Sage	280	**62.0 (1.0)**	20.1 (0.4)	0.4 (0.0)	0.7 (0.0)	13.0 (0.5)
Pinyon-Juniper/ Sage	210	78.5 (1.0)	11.5 (0.6)	0.4 (0.1)	0.2 (0.0)	6.5 (0.2)
Pinyon-Juniper	**870**	69.5 (0.6)	19.0 (0.3)	**1.0 (0.0)**	0.1 (0.0)	6.0 (0.2)
Pinyon-Juniper/ Oak	170	75.2 (1.4)	12.9 (0.5)	0.4 (0.1)	0.4 (0.1)	9.3 (0.5)
Pinyon-Juniper/ Manzanita	60	79.3 (2.6)	11.2 (1.4)	**0.1 (0.0)**	0.1 (0.0)	**2.2 (0.1)**
Pinyon Pine	40	79.4 (3.5)	14.6 (1.7)	0.2 (0.1)	**0.0 (0.0)**	6.3 (0.7)
Mountain Shrub	110	71.8 (1.3)	17.6 (0.7)	0.7 (0.1)	0.1 (0.0)	4.8 (0.3)
Ponderosa Pine/ Manzanita	70	85.9 (1.1)	**8.5 (0.4)**	0.2 (0.0)	**0.0 (0.0)**	5.9 (0.6)
Rabbitbrush	90	84.5 (0.6)	9.8 (0.2)	0.7 (0.1)	**0.0 (0.0)**	6.9 (0.6)
Aspen	60	80.0 (0.6)	13.3 (0.3)	**0.1 (0.0)**	0.1 (0.0)	**24.4 (0.8)**
Wet Meadow	30	83.4 (0.0)	9.7 (0.3)	0.2 (0.0)	0.1 (0.0)	17.3 (0.6)
Spring	**20**	62.0 (3.6)	18.9 (1.9)	**1.0 (0.0)**	**0.0 (0.0)**	2.9 (0.5)
Perennial Riparian	190	**86.4 (0.4)**	8.6 (0.3)	0.7 (0.0)	0.2 (0.0)	4.2 (0.2)

Predictive Spatial Modeling

Creation of predictive spatial maps was a simple way to identify possible hot spots of occurrence for each of the cover types studied here (Fig. 6-1). By comparing each map that was generated, general trends were recognized that could potentially benefit resource managers.

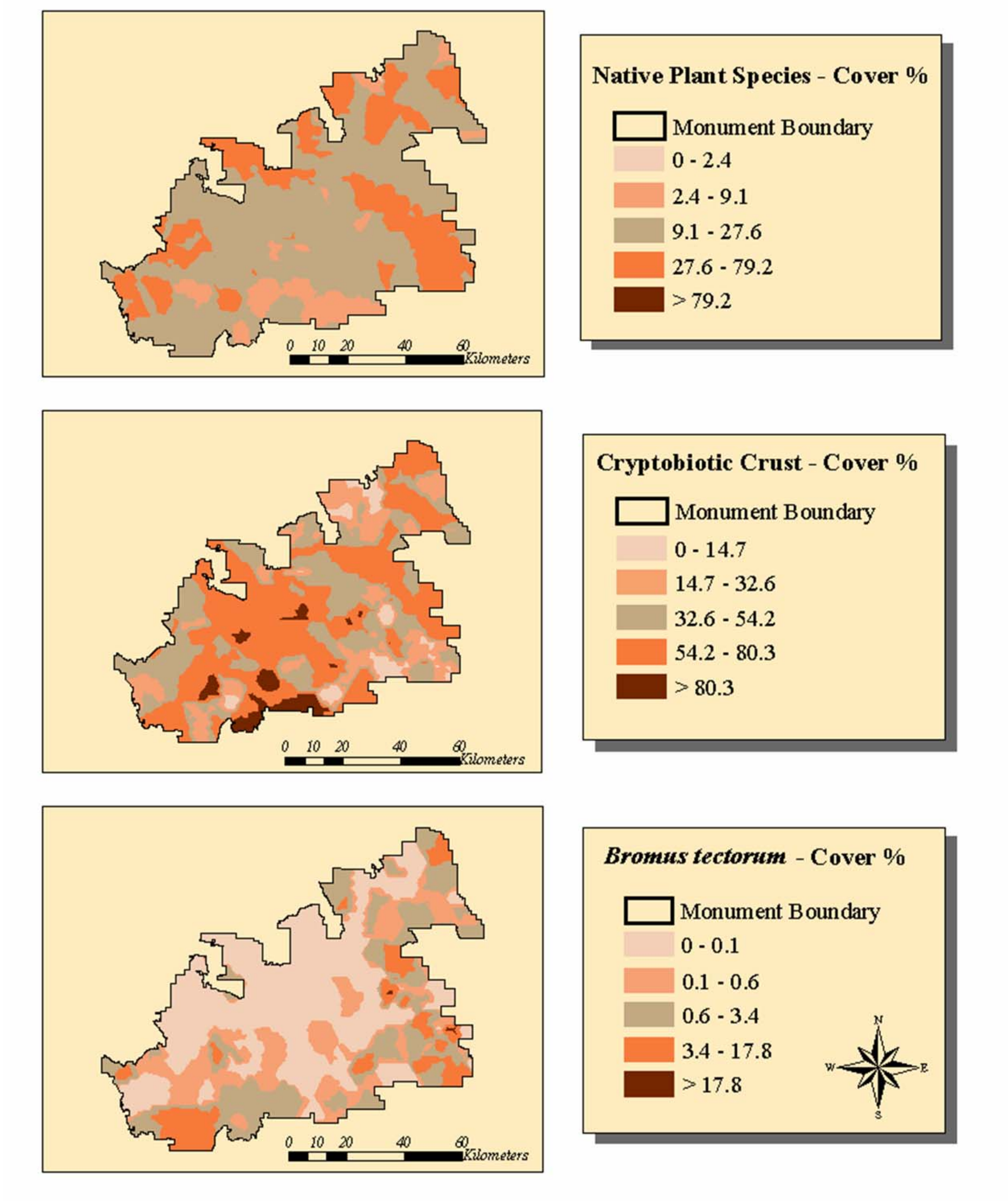

Figure 6-1. Predictive maps with inverse distance weighting.

From the predictive spatial models, it was found that native cover was relatively evenly distributed throughout the Monument with values ranging from 9.1 to 27.6%. However, in the eastern portion of the Monument along Fifty-Mile Mountain, native cover values were greater than the average ranging from 27.6 to 79.2% (Fig. 6-1). In contrast, southern portions of the Monument had areas of low native cover (< 9.1%) that corresponded with areas of high cryptobiotic crust cover (> 79.2%; Fig. 6-1). Looking across the entire landscape, high cryptobiotic crust cover (> 52.4%) was associated with areas containing less native species cover (< 27.6%) and low cheatgrass cover (< 3.4%). Thus, by comparing the maps, it was found that areas containing higher percent cover of native species tend to have less cryptobiotic soil crust cover and areas of high cryptobiotic crust cover tend to have less native plant cover.

Relative cover (i.e., cover of a particular species as a percentage of total plant cover) was generally below 1% but often fell below 0.1%. High cheatgrass cover was limited and patchily distributed in the southern and eastern portions of the Monument (Fig. 6-1). However, it was present in 74% of the randomly distributed sample plots at low concentrations (usually < 0.1% cover). Thus, cheatgrass frequency was relatively high across the Monument. A high frequency and widespread distribution demonstrate that a large seed source is present. If so, fire may result in increased invasion. Management should be particularly sensitive to areas in the southern and eastern portions of the Monument where the percent cover of cheatgrass is very high (0.6 to > 17.8%).

An example of how disturbance can affect native, cryptobiotic crust, and cheatgrass cover is exemplified at Buckskin Mountain. Buckskin Mountain is located in the south-central portion of the Monument. Three naturally ignited fires occurred in this area during 1996, 1997, and 1998, and the area was given post-fire treatments. The predictive map for cryptobiotic crust cover shows a significant decline in crust cover in this area (< 14.7%) when compared to surrounding areas (> 54%). In addition, the area also shows greater cheatgrass cover, ranging between 3.4 to 17.8% (Fig. 6-1). Another case study within this report found that burned sites had significantly lower native species cover (11.0%) and soil crust cover (4.1%) than adjacent unburned sites (Evangelista et al. 2004b). Most of the burned plots in this study also had significantly higher non-native species richness and cover and lower native species richness than nearby control sites (See Soils and Cryptobiotic Crusts).

Regression Trees

Regression trees were used to further identify the variables that were associated with trends demonstrated by the predictive spatial modeling analysis. The regression tree using native plant species cover as the dependent variable identified total cryptobiotic cover as the only significant independent variable (Fig. 6-2), which accounted for 19% of the variation in native species cover. The mean native cover for all tested plots was 26.4%, but the highest percent native species cover (48.8%) occurred when total cryptobiotic cover was less than 8.5%. When total cryptobiotic cover ranged between 8.5 and 69%, mean native species cover fell to 27.4%. When total cryptobiotic cover was greater than 69%, mean native species cover was only 11.6% (Fig. 6-2). This further supports the results from the predictive spatial modeling that suggested areas with greater native plant species cover have less cryptobiotic crust cover.

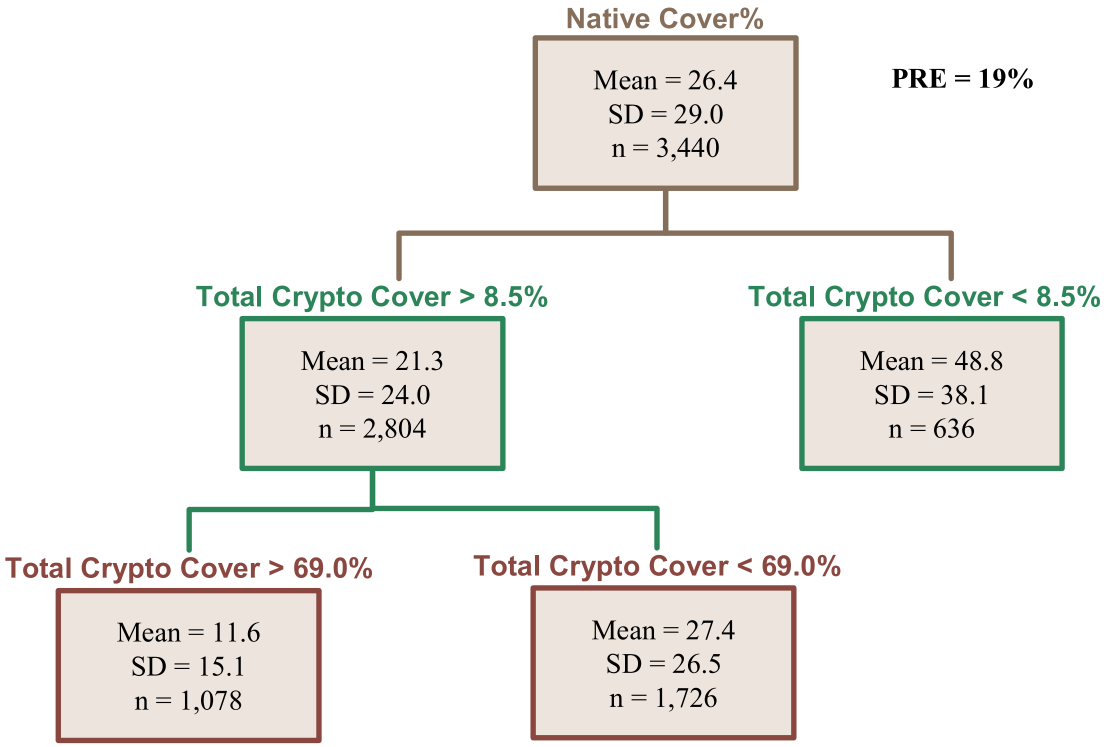

Figure 6-2. Regression tree for native plant species cover.

The regression tree that used total cryptobiotic crust cover as a dependent variable identified native cover and soil nitrogen as independent variables (Fig. 6-3), which accounted for 21% of the variation in total cryptobiotic crust cover. The mean total cryptobiotic crust cover for all subplots was 45.7%. The first split of the regression tree found that average total cryptobiotic cover was above this average when native cover was less than 26.6%. The final split of the regression tree suggests that total cryptobiotic cover increased even more if nitrogen was below 0.08%.

Therefore, the correlation between native species and cryptobiotic crust cover was also found in the regression tree for total cryptobiotic crust cover, which additionally introduces the presence of soil nitrogen as an independent variable (Fig. 6-3). Specifically, crust cover is shown to be higher when soil nitrogen levels are < 0.08% and native cover is < 26.5%. A study has suggested that cryptobiotic soil crusts are a dominant source of nitrogen for semi-desert ecosystems (Belnap 1996). This data also shows that crusts may be playing an important role in areas of low nitrogen availability by filling a niche presented by low native plant species cover.

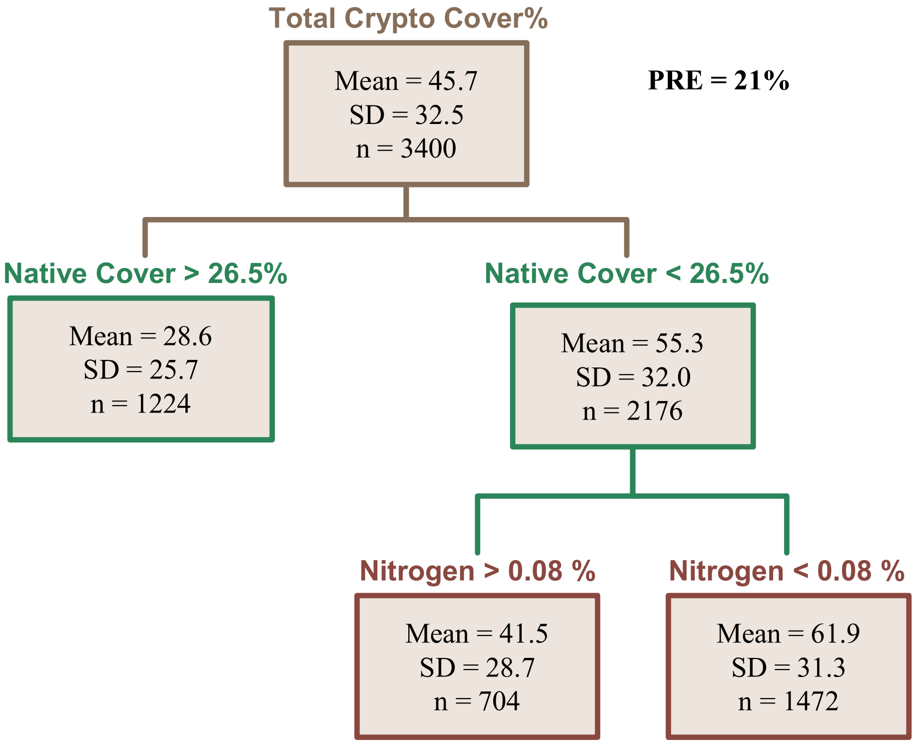

Figure 6-3. Regression tree for total cryptobiotic crust cover.

The regression tree that used cheatgrass cover as a dependent variable identified phosphorus and nitrogen as independent variables (Fig. 6-4), which accounted for 23% of the variation in cheatgrass cover. The mean cover of cheatgrass for all the subplots was 2.2%. The first split of the regression tree occurred when phosphorus levels were greater than 48 ppm. At this point, cheatgrass cover was much higher than the average (41.4%), but this situation occurred in only 0.003% of the subplots. When phosphorus levels fell below 48 ppm, cover averaged 2.0%. Between 18 and 48 ppm, cheatgrass had a mean cover value of 8%. The final split of the regression tree occurred when nitrogen was below 1.4%. At this point, average cheatgrass cover was estimated at 5.1%. When nitrogen values rose above 1.4%, the average cover of cheatgrass was 25.1%, much higher than the average.

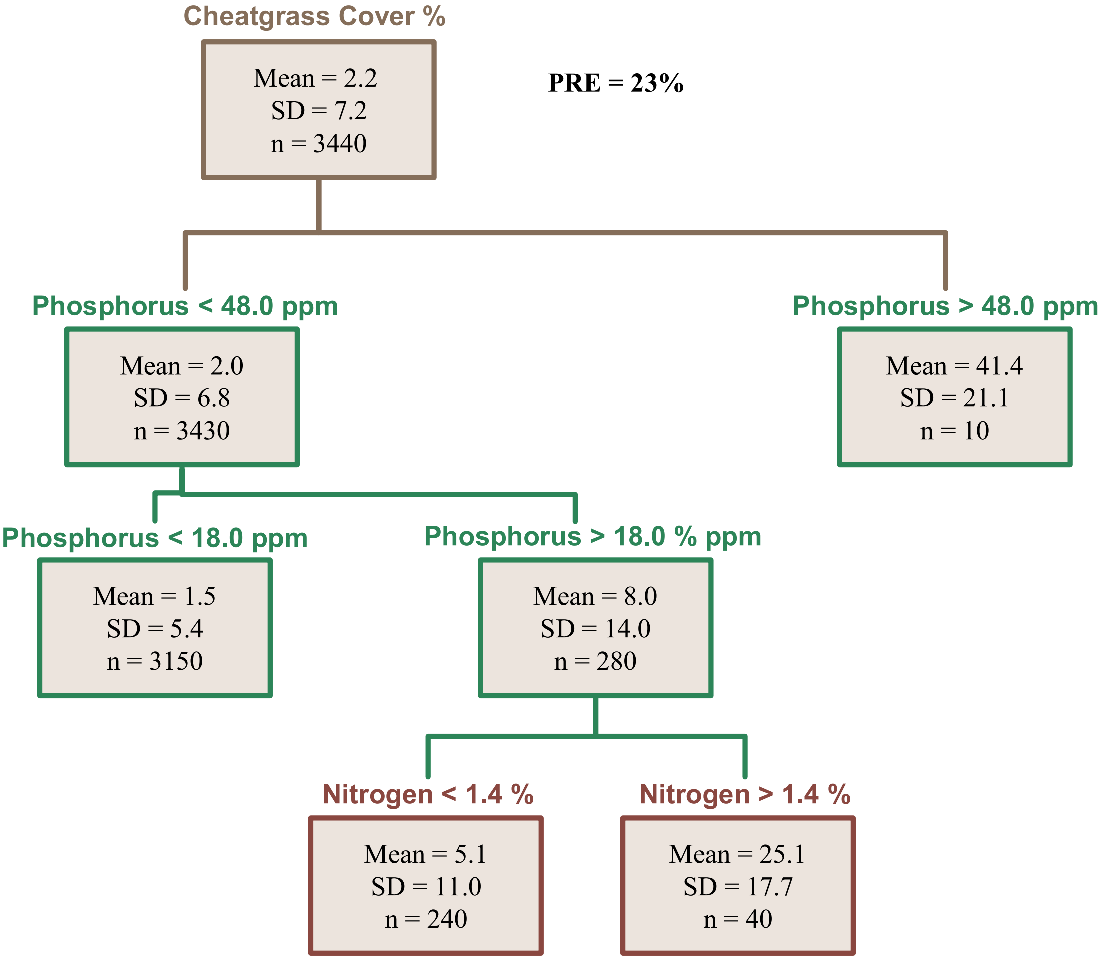

Figure 6-4. Regression tree for cheatgrass cover.

One study has suggested that soil phosphorus may prove to be a greater indicator of non-native species establishment and success (Bashkin et al. 2003). Other observations have also shown that areas high in nitrogen and phosphorus produce areas of high cover of cheatgrass. These areas (primarily disturbed by fire or more mesic sites) make up only a small portion of plots (n = 40), but they have the potential to be hot spots and source populations for future invasions.

Case Study 2: *Geospatial Statistical Modeling and Mapping*

Assessing the distribution, presence, and pattern of native and non-native plant species and correlating these variables to topographic and edaphic factors are top research priorities for many land management agencies (Kalkhan and Stohlgren 2000). Integration of geospatial information, field data, and spatial statistics are commonly used in vegetation studies and can increase the accuracy of landscape structure analysis at different spatial scales (Kalkhan et al. 2004). However, the integration of several types of data when modeling the spatial dynamics of an individual population poses a new challenge. There are two aspects to the problem: first, the integration of data from different sources must be at a fine enough resolution for the questions posed; and, second, the spatial dynamics of an individual population must be modeled (Reich et al. 1999).

Ecological variables (e.g., vegetation, soil, and hydrologic conditions) and environmental characteristics measured in the field are important elements for modeling and mapping the small-scale variability within landscapes. Gown et al. (1994) stated that many spatial datasets (i.e., remotely sensed data) provide reliable information for macro-scale ecological monitoring, but they fall short in providing the precision required by more refined ecosystem resource models. Spatial statistics provide a means to develop spatial models that can be used to correlate coarse scale geographical data (i.e., remotely sensed imagery, topographic variables) with field measurements of biotic variables. If a satellite image is geographically referenced to a base-map, one can overlay the location of field plots on the image to obtain reflectance values associated with each of the field plots. Then, if the field data are spatially correlated with reflectance from the remotely sensed image, it is possible to develop a model describing this spatial continuity (Cliff and Ord 1981).

Kalkhan et al. (2000; 2001; 2004) developed geospatial-statistical models and maps for describing fine and coarse scale spatial variability to forecast plant diversity, vegetation-soil characteristics, cryptobiotic crust cover, non-native species, and fire fuel loading parameters within several Rocky Mountain regions. All models were developed and modified based on Reich et al. (1999), in which they described a model based on a process using stepwise regression, trend surface analysis of geographical variables (e.g., elevation, slope, aspect, and landform), and measures of local taxa to evaluate coarse-scale spatial variability. The spatial statistical analyses in this study are similar and were accomplished using S-plus (MathSoft Inc. 2000), as defined here:

$$\Phi_0 = \sum_{i+}^{p} \sum_{j \leq p}^{p} \beta_{ij} \, x_{10}^{i} \, x_{20}^{j} + \sum_{K=1}^{q} \gamma_k \, y_{k0} + \eta_0$$

where β_{ij} are the regression coefficients associated with the trend surface component of the model, γ_k is the regression coefficient associated with the q auxiliary variables, y_{k0}, is available as a coverage in the GIS database, and η_0 is the error term, which may or may not be spatially correlated with its neighbors (Kallas 1997; Metzger 1997).

The geospatial statistical models based on coarse to fine scale variability may be used in place of specific information on the location of every individual within the population (Reich and Kalkhan 1999). It would be prohibitively expensive or impossible to collect information on every individual within a population. Thus, to develop the trend surface, spatial field data were collected on the plants animals along with other environmental variables thought to influence their presence. This information was then used to model the spatial interaction of individual species (i.e., native vs. non-native plants) with themselves and other species and their environment using trend surface analysis and stepwise regression which is based on the ordinary least squares (OLS) or other statistical estimates. The objectives of this study were to:

1.) Forecast and understand the landscape structure and pattern/distribution of plant diversity, soil parameters, and cryptobiotic crust cover at a landscape scale based on geospatial statistical modeling at coarse and fine spatial scales. Trend surface analyses and binary regression classification trees with other geostatistical approaches were used to develop these models and calculate spatial statistics for vegetation, soil, and cryptobiotic crust cover, with the use of a multi-phase sampling design (i.e., double sampling) and an unbiased multi-scale nested sampling design on the ground. The modeling-mapping approach used 1) field data, including vegetation-soil characteristics and cryptobiotic crust cover 2) remote sensing data (e.g., Landsat- 7 ETM+), and 3) GIS ancillary datasets.
2.) Predict hot spots of diversity, the distribution and patterns of plant species richness, and potential threats from non-native species at multiple-scales within the study area.
3.) Develop a new tool to assist in efforts to better manage rangeland and natural resources, and determine long term monitoring of landscape condition within the Monument. Improve the capabilities of resource managers to create, restore, and maintain habitat for indigenous species in riparian and connected ecosystems (e.g., upland, streams, and river) with the spatial models.

Methods

Geospatial Analysis

For this study, we used the same approach as Kalkhan and Stohlgren (2000), Kalkhan et al. (2000; 2001; 2004), and Brown and Kalkhan (2004) on using the cross-correlation statistic to test the null hypothesis of no spatial cross-correlation among all pairwise combinations of vegetation variables and topographic characteristics (Table 1). In calculating the cross correlation-statistic (I_{YZ}), the inverse distance between sample plots was used as a weighting factor to give more weight to values in the closest plots and less to those in plots that are farther away. The null hypothesis of no spatial cross-correlation was rejected when the *P*-value associated with the test statistic was less than 0.05. *Moran's I* (Moran 1948; Mantel 1967), which is a special case of the cross-correlation statistic I_{YZ} (Czaplewski and Reich 1993), was used to calculate the spatial auto-correlation associated with each of the variables used in this study (Table I). Cliff and Ord (1981) showed that I_{YZ} ranges from –1 to +1, although it can exceed these limits with certain types of spatial matrices. Data distributions that were strongly skewed were transformed prior to analysis. Aspect data were transformed using the absolute value from due south (180^o; high solar radiation, and represent continues data; Kalkhan and Stohlgren 2000; Kalkhan et al. 2000; 2001; 2004)

Table 6-3. List of vegetation, biotic (soil and cryptobiotic crust cover), and geospatial information (remote sensing and GIS data) used to develop geospatial statistical models and maps at the Monument.

Vegetation Characteristics	Biotic Parameters	Geospatial Information
Total Plant Species Native Plant Species Non-Native Plant Species Total Plant Cover (%) Native Plant Cover (%) Non-Native Plant Cover (%)	Cryptobiotic Crust Cover (%) Soil Variables: nitrogen, potassium, magnesium, phosphorus, sodium, organic carbon, inorganic carbon, soil pH, carbon-nitrogen ratio	TM band1, band2, band3, band4, band5, *band6 (L and H), band7, band8, Tassel Cap (band 1,2,3,4,5, and 6); elevation, slope, absolute aspect, and landform

*Band 6 (L and H): L is the log gain and H is the high gain in thermal band

**Soil parameter units were measured in %, except phosphorus in ppm.

Modeling Development

For the purpose of developing the spatial models and producing a spatial map, the following steps were needed to understand how the variables of interest contributed to the model development. The steps are listed as follows:

Test for spatial auto-correlation using *Moran's I*, cross-correlation statistic using *Bi-Moran's I*, and *Pearson* linear correlation (*r*) between the variables of interest (i.e., elevation, slope, aspect, remotely sensed data, the presence of exotic-native species richness, cryptobiotic crust cover and soil parameters).

1.) Use a stepwise regression to screen the independent variables with their interaction of each variable.
2.) Compute Ordinary Least Squares (OLS) for the coarse-scale variability.
3.) Examine the residuals to test for spatial autocorrelation.
4.) Utilize the semi-variogram to evaluate fine-scale variability, and select the model (variogram type gaussian, exponential, spherical) with the lowest Akaike's Information Corrected Criteria (AICC) and smallest variance (Akaike, 1997).
5.) Apply kriging of the residuals to interpolate values for fine-scale variability based on the semi-variogram model with the lowest AICC value.
6.) Combine the OLS and regression binary classification trees (RT) for the final surfagrid (spatial model – map).

Spatial Information (Remote Sensing – GIS) Variables

Prior to developing the spatial statistics model and mapping, topographic data such as elevation, slope, aspect, and landform, were derived from Digital Elevation Model (DEM) data using GRID in ARCINFO or ARCVIEW GIS. Landsat ETM+ (Landsat 7) data were used and converted to GRID-GIS layers and were integrated with spatial statistical modeling to produce thematic spatial maps of vegetation, soil, and cryptobiotic crust cover characteristics. For image processing and analysis, ERDAS-IMAGINE (version 8.6) and GRID-ARCINFO (version 8.3) were used for remote sensing data and were integrated with GIS data layers for the purpose of spatial modeling and thematic mapping products. S-plus (MathSoft Inc. 2000) was used for data analysis for developing predictive spatial models.

Data used in modeling included nine bands of Landsat TM Data, six bands of transformed tasseled cap indices (using ERDAS-IMAGINE 8.6), topographic derived data (elevation, slope, aspect, landform; ARCINFO version 8.3**,** ESRI 2000), and vegetation data (total number of plant species, number of native plant species, number of non-native plant species; percent cover for total, native, and non-native species; soil and cryptobiotic crust cover characteristics). All spatial information from remotely sensed data and GIS layers were converted to a grid and resampled to 10 m pixel sizes using ARCINFO (ESRI 2000, version 8.3)(ESRI 2000). A program was written in AML (*ARC MACRO LANGUAGE,* ESRI 2000) and used to extract the 367 geospatial (digital numbers from remote sensing imagery and GIS layers) data points with respect to their UTM-X and Y coordinates within the study area. All data were then used for the development of the spatial models using S-plus software (MathSoft Inc. 2000; Table 6-3). It is important to mention that the use of a tassel cap transformation adds unique information to geospatial statistical modeling. Tassel caps are derived from Landsat TM data using ERDAS-IMAGINE software. They are used to measure brightness (i.e., soil characteristics) and greenness (i.e., biomass and vegetation characteristics). Adding this information contributes to more accurate interpretation of the landscape characteristics in reas such as the Monument.

Results and Discussion

To develop the predictive geospatial statistical models and maps, we used 367 data points based on the modified-Whittaker nested plot design (Stohlgren et al. 1995) we extracted values from Landsat TM data (nine bands, and six bands of tasseled cap transformation indices), topographic data (elevation, slope, absolute aspect, landform), vegetation and soil characteristics, and cryptobiotic crust cover at these points (Table 6-4).

Table 6-4. Estimated multiple R^2 values (%) for vegetation, cryptobiotic crust cover, and soil parameters based on 367 plots using Ordinary Least Squares (OLS) with a binary regression classification tree (RT).

Predicted Variables	Multiple R^2 (%) using OLS	Multiple R^2 (%) using OLS + RT
Total Plant Species	11	18
Total Plant Cover (%)	24	67
Non-Native Plants	10	61
Non-Native Plant Cover (%)	3	51
Native Plants	16	25
Native Plant Cover (%)	26	68
Cheatgrass Cover (%)	6	34
Probability Cheatgrass Cover (%)	10	25
Cryptobiotic Crust Cover (%)	8	62

Table 6-4. (Continued) Estimated multiple R^2 values (%) for vegetation, cryptobiotic crust cover, and soil parameters based on 367 plots using Ordinary Least Squares (OLS) with a binary regression classification tree (RT).

Predicted Variables	Multiple R^2 (%) using OLS	Multiple R^2 (%) using OLS + RT
Phosphorus (ppm)	14	47
Inorganic Carbon (%)	27	49
Organic Carbon (%)	6	36
Nitrogen (%)	13	64
Organi Carbon-Nitrogen Ratio	6	39
Sodium (ppm)	14	39
Magnesium (ppm)	8	48
Potassium (ppm)	10	58
Soil pH	21	68
Sand (%)	26	42
Silt (%)	30	67
Clay(%)	14	32

Spatial Relationships

The preliminary results for our field data using Moran's *I* (Moran 1948; Mantel 1967) and the bivariate cross correlation-statistic "I_{YZ}" (Czaplewski and Reich 1993; Bonham et al. 1995) to test for spatial auto-correlation and cross-correlation with residuals suggested that, at large-scales, the probabilities of presence and absence of non-native plant species and their percent cover were spatially independent throughout the study site. That is, the spatial relationships were not statistically significant. Native species richness was not independent (Kalkhan and Stohlgren 2000). However, these results may be different for individual plant species (Kalkhan et al. 2000; Kalkhan et al. 2001; Kalkhan et al. 2004). In general, coarse-scale patterns of species distributions were controlled by topographic factors such as elevation, aspect, and slope with complex spatial patterns. This may explain why negative spatial autocorrelation and cross correlation resulted when large-scale plots were used (Kalkhan et al. 2000). These results may have been different if individual native or non-naive plant species had been used in the analysis (Kalkhan et al. 2000; 2001; 2004).

Models using ordinary least square (OLS) were developed for coarse-scale variability of the total number of plants (both native and non-native species) and percent plant cover (total, native, and non-native). The trend surface models identified using stepwise multiple regressions had R^2 values ranged from 3% to 30% and all variables were significant at $\alpha < 0.05$ (Table 2). In addition, fine-scale variability models were used to examine the spatial continuity of variability and were developed using regression binary classification trees (RT). Model parameters were estimated using weighted least squares (Cressie 1985). The residuals were also analyzed for spatial autocorrelation and cross-correlation (Czaplewski and Reich 1993; Kalkhan et al. 2000; 2001; 2004) with the geographical variables (e.g., elevation, slope, other). Inverse distance weighting was used to define the spatial weights matrix. The regression binary classification trees (RT) models were cross-validated to assess the variability in the prediction errors. The cross-validation included deleting ten observations from the data set and predicting the deleted observations using the remaining observations (Kalkhan et al. 2004). This process was repeated for all observations in the data set.

The final models (trend surface plus the regression classification trees using residuals from OLS) had R^2 values ranging from 10% to 68% (Table 6-4). In addition, the accuracies of the regression binary classification trees models were assessed using the relative mean squared error suggested by Hevesi et al. (1992). The predicted geospatial statistical models and maps proved to be a unique approach to forecast the landscape within the Monument and will assist resource management teams to make better decisions on how to manage the landscape and to monitor areas vulnerable to invasive species, to protect hot spots of diversity, gauge wildfire hazards, and to make ecological-environmental problem assessments.

However, in some cases, these models resulted in low R^2 values. Potential sources of these lower R^2 values may have been the drought conditions during the fifth year of the field season, or other ecological factors. The lack of a good vegetation classification map for the Monument precluded their use in improving the geospatial statistical models and maps. This map could have been integrated as a new variable to improve the predictive models. Another factor was the orientation of modified-Whittaker plot designs on the ground, which made it hard to maximize the spectral information derived from remote sensing imagery. The plot layout is 20 m x 50 m, where the Landsat TM is a square in dimension and based on pixel resolution. Bands 1, 2, 3, 4, 5, and 7 have pixel resolutions of 30m x 30m, while band 6 (Low and High gain) has a pixel resolution of 60m x 60m, and band 8 (Panchromatic Mode) has a 15 m x 15 m resolution. Last, some of the plot locations were clustered, which interfered with our ability to capture coarse-fine scale variability of the landscape within the Monument. Avoiding clustering of plot locations may better approximate the heterogeneity of the landscape, and might improve the predictive spatial models and maps, contributing to higher R^2 values.

Examples of predictive spatial statistical maps which were based on the trend surface (OLS) and RT models of total exotic species richness, non-native plant species cover (%), cryptobiotic crust cover (%), and soil parameters (i.e., magnesium) within the Monument are shown (Fig. 6-5; Fig. 6-6; Fig. 6-7; Fig. 6-8; Fig. 6-9; Fig. 6-10). Figure 6-10 shows examples of the standard errors associated with predicting native plant species richness (map of uncertainty). The figure shows that standard error values increased with distance from the sample points, as would be expected. The standard error values indicated significant utility of the map of non-native plant species richness for directing future management activities.

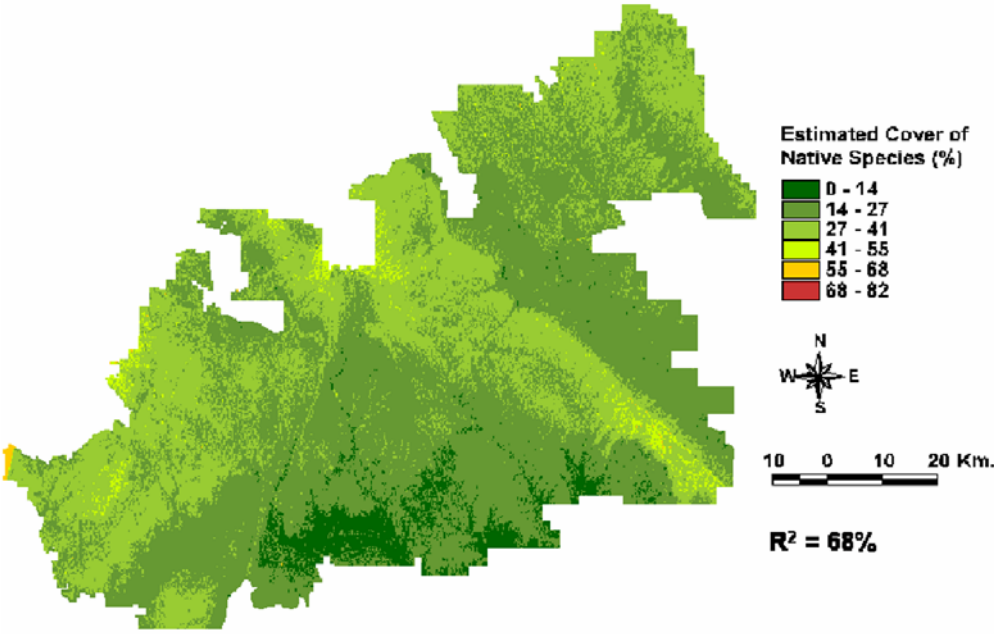

Figure 6-5. Spatial predictive map of native species cover (%).

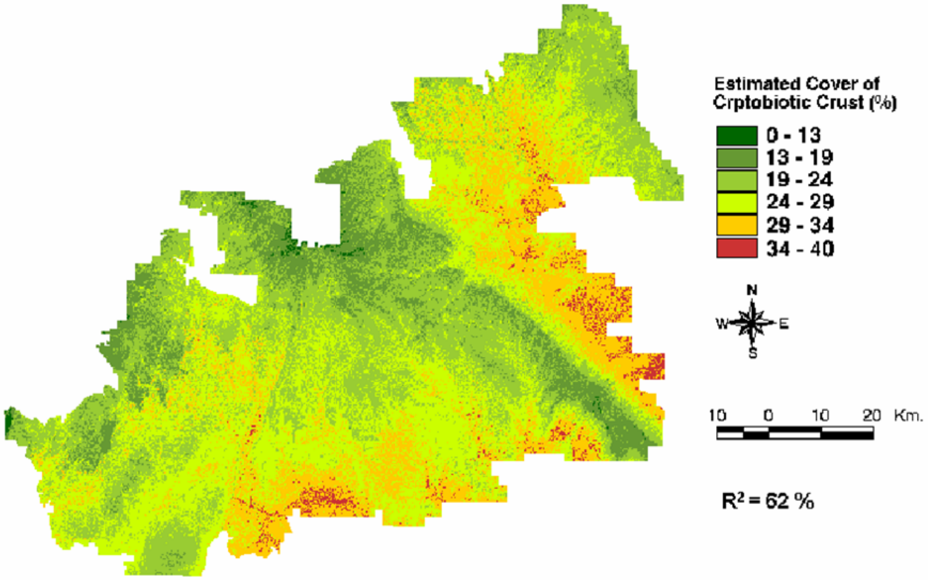

Figure 6-6. Spatial predictive mapof cryptobiotic soil crust cover (%).

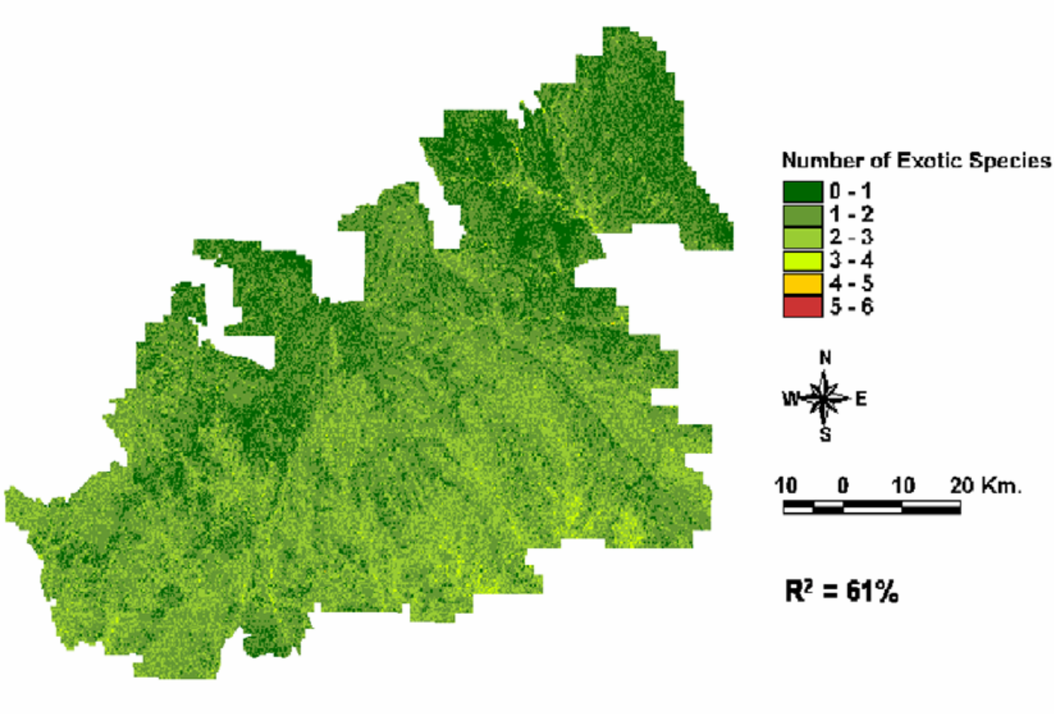

Figure 6-7. Spatial predictive map of non-native species richness.

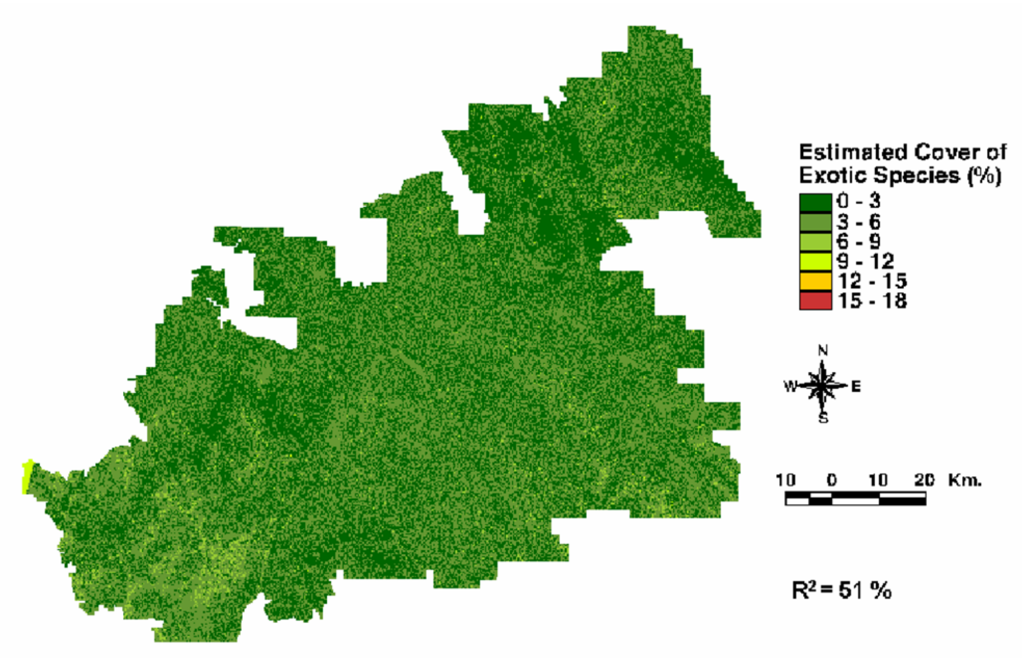

Figure 6-8. Spatial predictive map of non-native species cover (%).

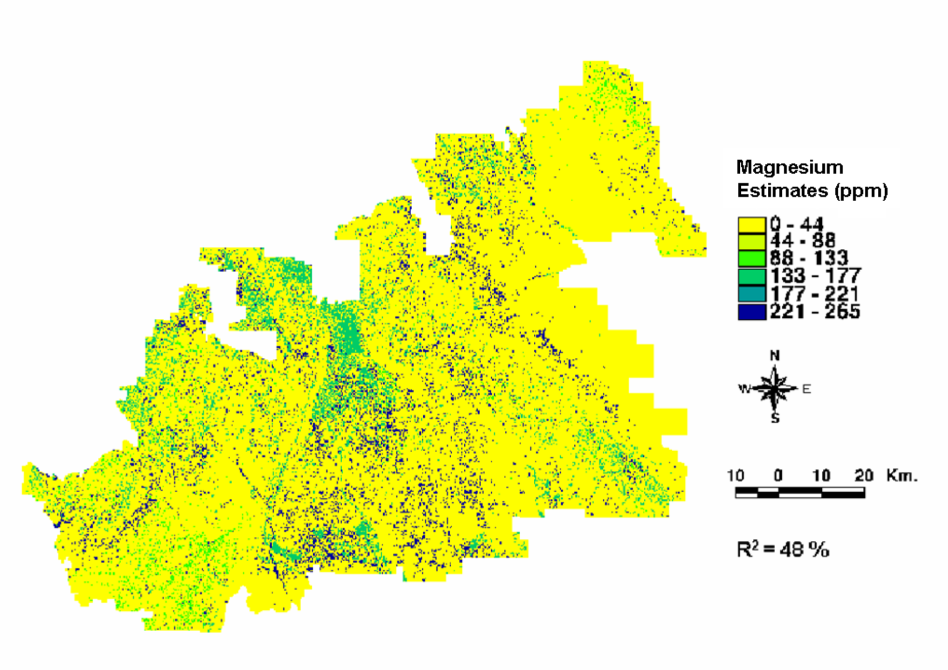

Figure 6-9. Spatial predictive map of magnesium in the soil (ppm).

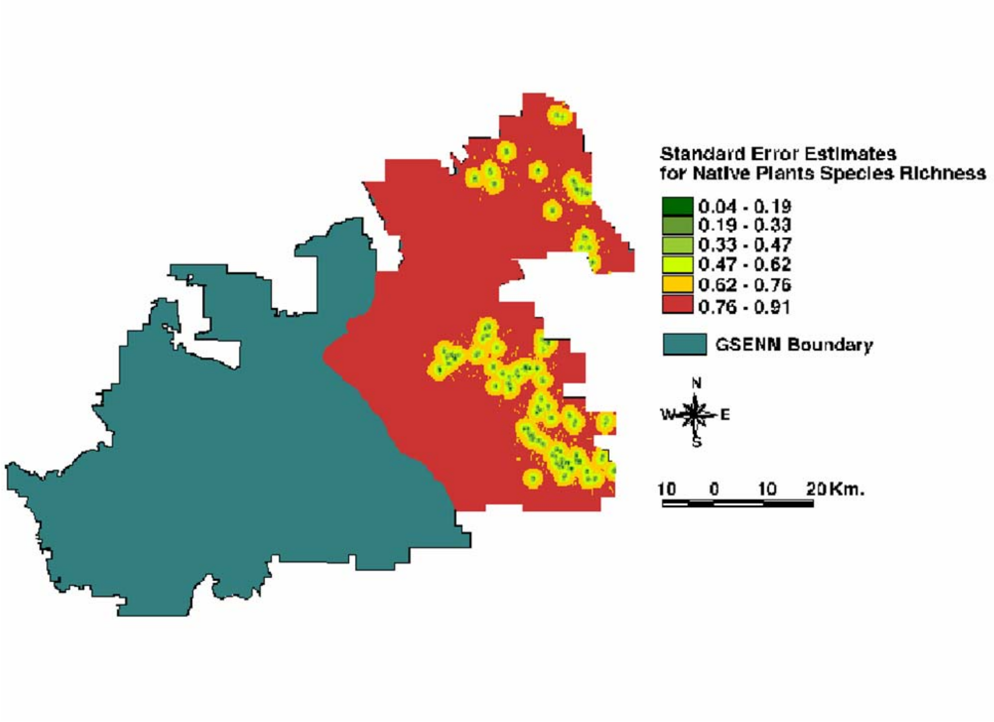

Figure 6-10. Spatial predictive map of standard error estimates for native plants species richness.

This technique of geospatial modeling and mapping provides a unique way to describe landscape-scale plant diversity, predict hot spots of diversity, and wildfire patterns, and may contribute to better management decisions. Also, using these new tools of spatial mapping is an important new step in the efforts to quantify the full range of ecological complexity among plant diversity lands, which can be integrated with previous landscape-ecological research within the Monument. Adding more sampling points and examining ecological relationships (e.g., between vegetation, cryptobiotic crust cover, soil, and other biotic parameters) may help to improve predictive spatial statistical models and their accuracy (i.e., error reductions).

Management Implications

Modeling can be an important tool in identifying special areas of interest for managers with limited resources. Identifying the independent variables that influence a particular cover type can help concentrate efforts focused on hindering non-native species invasions or protecting native and rare vegetation types.

Differences existed between the two case studies presented here. The methods of the first case study are intended to provide a rapid assessment of a cover type of interest (e.g., native cover, non-native cover, cheatgrass cover). However, the predictive spatial maps using IDW do not incorporate independent variables. These maps are strictly based on the value of a particular cell and its distance to other cells with values greater than zero. The addition of regression trees to this type of analysis shows that certain independent variables influence the distribution of a particular cover type. This step is critical for validation of the predictive spatial maps. Several, more intensive, methods do exist (e.g., kriging, cokriging). An example of such a method is presented in case study two. Although this type of method improves accuracy, it is computationally intensive, costly, and requires extensive expertise in remote sensing.

CHAPTER 7

Future Inventory, Monitoring, and Research Needs

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Project Summary

We have just scratched the surface of the complex ecological ice berg known as the Grand Staircase – Escalante National Monument. The patterns of native and non-native plant diversity, soils, and soil crusts are very complex due to the myriad of variation caused by combinations in slope, aspect, geology, soil, and vegetation types. We relied on the real strength of unbiased, multi-scale vegetation and soil surveys combined with the early development of a new breed of spatial predictive models to quantify the plant diversity and soil crust patterns. Since only a small fraction of the total area can be affordably sampled, the predictive spatial models were an important tool to extrapolate findings to the much larger unsampled landscape. We identified hotspots of native and non-native plant diversity and non-overlapping areas of endemic richness, unique species assemblages, and native species rich areas in the Monument. We surveyed, modeled, and mapped the relatively high cryptobiotic crust cover in the southern and southeastern part of the Monument and the hotspots of non-native species richness in the major drainages. The models will help direct management efforts to these sensitive and vulnerable areas. Our surveys have set the stage for cost-efficient monitoring. As shown in this report, the models can be powerful tools in planning, teaching and training, and public outreach.

Caveats

The generalizations and recommendations that follow should be viewed with some caution and a few caveats should be considered in our studies. We did the majority of our field sampling in May and June, consequently we missed recording some of the later summer monsoon plant species at their peak. Hanging gardens and very rare habitats were also not sampled and their contribution to species richness is not included in the models. The species-area curve for the Monument is tapering off at the 550 plant species that we recorded, indicating that we captured the most species for the effort involved in random surveys. However, we found that species rarity is very common in the Monument with 62% of the flora estimated to be uncommon and patchily distributed. More specific, non-random, targeted surveys will be needed to capture the less frequently occurring flora.

We found the Monument to be extremely variable making it difficult to generalize or characterize. Selecting a subset of plots for long-term monitoring will be difficult since there is no such thing as a “typical” area of the Monument. Even a six-year study is a snapshot in time. The climate, land use patterns, and non-native plant invasions are constantly changing the landscape. The addition of fine resolution data on livestock grazing (which are generally unavailable), detailed soils maps (which were not available until relatively recently), and additional remote sensing data (e.g., MODIS time-series data, LIDAR) would help to improve the spatial models presented here. Small-scale and large-scale disturbances undoubtedly have an effect on native and non-native plant species distributions and cryptobiotic crust cover, and quantifying these disturbance histories in a spatial form also would improve the models.

Consistent Patterns and Future Resource Management Needs

Over the past six years, some consistent patterns emerged from our studies that highlight future research and management needs:

1.) Native and non-native plant species thrive in rare, mesic habitats that are high in soil fertility (especially phosphorous), moisture, and foliar cover, occurring in relatively lower elevations. Unique habitats (areas with rare native plant species or those occurring in low frequencies) are also plagued with high frequencies of non-native species. Careful monitoring of rare habitats and targeted early detection, rapid response, and restoration activities are needed to contain non-native species in habitats high in native species richness and near sensitive water resources.

2.) Highly disturbed habitats such as post-burn areas have exceedingly high levels of plant invasions related to the destruction of soil crusts and local displacement of native species by non-native species. Although fire can increase soil fertility with the release of phosphorous and nitrogen, the availability of seed sources allow cheatgrass to readily establish before native species. Increased fire frequencies and post-fire mechanical restoration efforts further reduce crust regeneration, which can further facilitate non-native plant species invasion. We recommend that fire restoration efforts use native seed mixtures with aerial seeding to minimize further crusts disturbance and non-native infestation. Preventing un-naturally frequent wildfires in the Monument may be very important in containing harmful invasive plant species.

3.) More common xeric habitats with low fertility and vegetation structures that are high in endemic species richness have considerably lower non-native species and cover. A subset of these areas should be monitored to track key endemic species populations.

4.) We found that with many plant species, life history can be an important predictor of successful invasion because it integrates specific environmental variables. Non-native annual species favor habitats high in native annual species richness, and non-native perennial species favor sites high in native perennials. Most non-native species are not a widespread threat and only a relatively few species are considered highly invasive. We identified seven non-native species of concern to resource managers that each will require aggressive, systematic strategies for containment and control.

5.) The high frequency of occurrence of non-native species in the Monument is a great cause for concern. In addition, the number and cover of non-native plant species is probably increasing in the Monument, but this will require standardized, long-term monitoring. Two invasive plant species, in particular, may significantly transform ecological processes across the landscape. Cheatgrass is, by far, the most invasive species in the Monument in terms of frequency and cover. Perhaps the most dramatic impact of this prevalent weed is its ability to alter fire regimes in the Monument, increasing both the frequency and size of fires. Considered a generalist, cheatgrass is found in most habitats in the Monument on most soil and vegetation types. Future fires and the destruction of soil crusts may lead to great increases in cheatgrass cover in wet years. Likewise, tamarisk (salt cedar) likely will increase along stream banks in favorable climate years. Our surveys conducted in Hackberry Canyon, where tamarisk is far from reaching its invasive potential, show vast potential for tamarisk spreading upstream at a faster rate and steadily downstream. Early detection of tamarisk seedlings and saplings is the only cost-efficient way to contain tamarisk. Though biological control, manual and chemical control, and site restoration is possible for large areas, it will be very costly to protect native plant species assemblages in the future. Careful planning is needed.

6.) Cryptobiotic crusts play an integral role in the Colorado Plateau ecosystem. Their ability to function is crucial to the success of these native systems and is of major concern for resource managers balancing various land uses in the Monument. Sites with high cryptobiotic crust cover tend to have lower non-native species richness and cover. Our plots and models show the highest densities of cryptobiotic crusts in the xeric southern regions of the Monument. Areas with well-developed crusts may need further protection.

7.) Livestock grazing may have subtle long-term effects in the Monument. Our preliminary study at No Man's Mesa and Deer Spring Point inferred that long-term grazing may homogenize the composition of native species on large scales by reducing sensitive locally rare species while increasing more common species. Additional research is needed at broader scales, which may require additional areas closed temporarily to grazing to more carefully assess the potential increase in woody shrub and litter cover in grazed sites, potential displacement of locally rare or unique species, and future changes in fuel loading and fire frequency. Cheatgrass establishment following a wet spring has the potential to facilitate a negative feedback loop that can result in greater non-native infestations and further increased fire frequencies. Plots placed along the Escalante River will provide invaluable baseline data for future monitoring of a riparian system that has had livestock recently removed or limited. Only multi-scale plots and baseline information will allow patterns such as homogenization of the landscape to be accurately quantified.

Future Research Needs

There are countless avenues for continued research in the Grand Staircase – Escalante National Monument. Disturbance plays an extremely large role in the destruction of cryptobiotic crusts and the spread of non-native species. Consequently, more studies on post-burn seeding techniques with differing rates and ratios of native species seed and alternative application techniques less-destructive to the cryptobiotic crusts such as aerial seeding should be explored. Spatial variation in the landscape plays a large role in the response of native and non-native species and cryptobiotic crusts to disturbance; therefore, more disturbance studies in other geology and soil types of the Monument are needed. Relict sites that have been protected from grazing are extremely rare, but more exploration for such sites would be worthy of the time required to find them. Establishment of large, long-term exclosures would also provide informative comparison data.

The urgent question of how best to deal with the non-native species in the Monument remains to be answered. The challenge of controlling non-native species in rare, mesic, native species rich habitats needs to be carefully researched. Control methods that eliminate the non-native species with the least amount of disturbance (non-target effects) and that allows native species a competitive advantage should be explored and carefully documented. Research evaluating different control methods in different environments would be extremely valuable. The direct impact of cheatgrass on specific native plant species is also unknown. Studying a matrix of species associations would indicate which species cheatgrass affects the most. Continued research on cost-efficient survey techniques for non-native species is essential. A science-based "early detection and rapid response" pilot study is needed to track invasive plant species that are currently found in low frequencies in the Monument such as Bermudagrass (*Cynodon dactylon*) and field bindweed (*Convovulus arvensis*) that are on the Utah noxious weed list. Curly dock (*Rumex crispus*), rabbitsfoot grass (*Polypogon monspeliensis*), and poison hemlock (*Conium maculatum*) are presently uncommon species that prefer moist habitats. Because the mesic habitats are the most vulnerable to invasion, early attention should be paid to these species. Monitoring for other detrimental invasive organisms will also be important. Diseases such as the West Nile virus will undoubtedly have an impact on the ecology of the Monument, and sudden oak death, Argentine ants, chronic wasting disease and many other highly invasive species may be on the way. Quick assessments for new non-native species are key to keeping a future invasion at bay. Further analysis into the pathways of invasion, barriers to invasion, and effects of invasions are important to quantify to control non-native species.

The protection of cryptobiotic crusts may also require additional research. We are concerned about the rapid rate of crust depletion in burned areas, and the extremely slow rate of recovery. Cumulative effects of long-term crust disturbance and recovery over large areas should be better quantified.

A more detailed, fine-resolution vegetation map is needed for the Monument, and could be combined with species distribution maps for key native and non-native species (plants and crusts). Our data sets can be effectively used for "ground truthing" such maps, and the developed models would help to further improve and calibrate the maps. The Monument staff could develop many of these modeling capabilities, like the iterative models discussed in this report, which would allow continual refining of the already established models to aid in management decisions. Individual species models will be key to hone in on more specific management needs.

While we only sampled <0.0043% of the Monument's land area, we were able to detect consistent ecological patterns that held true throughout most of the Monument. Though surveys like ours are time intensive, the collection of baseline information with large multi-scale plots combined with smaller plots and observations in a nested-intensity design may prove extremely cost-efficient. For example, observing the homogenization of plant species composition at large spatial scales was a surprise in our near-relict versus grazed site study. Large baseline studies and continued monitoring are needed to quantify cumulative effects.

Cumulative effects research is new to ecology yet it is vital for managers to know these cumulative effects for accurate environmental impact statements and to comply with federal regulations. Our plots were marked with benchmarks for long-term monitoring, and it is our hope that these plots will not only aid in answering questions about the complexity of the Monument ecology now, but also to detect important future changes. We hope we provided a "legacy" data base for Monument staff and future generations.

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Appendices

Appendix A

The following are a list of products from the project for the years 1997 through April, 2004.

PUBLICATIONS

Accepted/Published Papers

Alley, N.W., T.J. Stohlgren, P.H. Evangelista, and D.A. Guenther. 2004. **Iterative model development for natural resource managers: a case example in Grand Staircase-Escalante National Monument.** *Accepted by Geographic Information Sciences, 2004.*

Stohlgren, T.J., Belnap, J., Chong, G.W., and Reich, R. 1997. **A plan to assess native and exotic plant diversity and cryptobiotic crusts in the Grand Staircase-Escalante National Monument.** Grand Staircase-Escalante National Monument Science Symposium Proceedings at Southern Utah University. Nov. 4-5, 1997.

Stohlgren, T.J., M.W. Kaye, A.D. McCrumb, Y. Otsuki, B. Pfister, and C.A. Villa. 2000. **Using new video mapping technology in landscape ecology.** BioScience 50:529-536.

Stohlgren, T.J., Y. Otsuki, C.A. Villa, M. Lee, and J. Belnap. 2001. **Patterns of plant invasions: a case example in native species hotspots and rare habitats.** Biological Invasions 3:37-50.

Bashkin, M., T. Stohlgren, Y. Otsuki, M. Lee, P. Evangelista, and J. Belnap. 2003. **Soil characteristics and exotic plant species invasions in the Grand Staircase-Escalante National Monument, Utah.** Applied Soil Ecology 22:67-77.

Waters, M.A. 2003. **The need for a multivariate approach to understand patterns of species richness and invasion: A case study in Grand Staircase – Escalante National Monument, Utah.** M.S. Thesis. Colorado State University, Fort Collins, CO.

Guenther, D.A., T.J. Stohlgren, and P. Evangelista. 2004. **A comparison of a near-relict site and a grazed site in a pinyon-juniper community in the Grand Staircase – Escalante National Monument, Utah.** Pages 153-162 *in* van Riper III, C., and K. L. Cole (Eds) The Colorado Plateau: cultural, biological, and physical research. The University of Arizona Press. Tucson, AZ.

Evangelista, P., D. Guenther, T.J. Stohlgren, and S. Stewart. 2004. **Fire effects on cryptobiotic soils crusts in the Grand Staircase – Escalante National Monument, Utah.** Pages 121-128 *in* van Riper III, C., and K. L. Cole (Eds) The Colorado Plateau: cultural, biological, and physical research. The University of Arizona Press. Tucson, AZ.

Evangelista, P., T.J. Stohlgren, D.A. Guenther, and S. Stewart. **Vegetation response to fire and post-burn seeding treatments in juniper woodlands of the Grand Staircase-Escalante National Monument, Utah**. *Accepted by Western North American Naturalist, to be published July 2004.*

Papers in Review

T. J. Stohlgren, D. A. Guenther, P. H. Evangelista, and N. Alley. Patterns of plant species richness, rarity, endemism, and uniqueness in an arid landscape. *Submitted to Ecological Applications, December 2003.*

Otsuki, Y., T.J. Stohlgren, D.A. Guenther, and C. Villa. **Evaluating plant invasions from both habitat and species perspectives.** *Submitted to Western North American Naturalist, February 2004.*

Stohlgren, T.J., C. Crosier, G. Chong, D. Guenther, and P. Evangelista. **Native annual and perennial species characterize the vulnerability of sites to non-native plant invasions.** *To be submitted February 2004.*

Papers in Progress

Waters, A., Stohlgren, T.J., Guenther, D.A., and Evangelista, P.H. **Natural variation in diversity and invasion patterns.**

Evangelista, P.H., Guenther, D.A., Stohlgren, T.J., and J. Belnap. **Use of visual assessments of cryptobiotic crusts to evaluate productivity.**

Evangelista, P.H., T.J. Stohlgren, N.W. Alley, and J. Graham. **Early Detection and Mapping Techniques for Tamarisk in Hackberry Canyon, Utah.**

PRESENTATIONS AND OUTREACH

<u>Outreach:</u> Denver Museum of Natural History – Jane Goodall Outreach Program – March 2003
“Landscape scale assessment of native and exotic plant diversity in the Grand Staircase-Escalante National Monument”, “Modeling soil characteristics and plant diversity in the Grand Staircase-Escalante National Monument, Utah”, and “Exploring the Monument” educational posters, as well as hands-on plant demonstrations with the dissecting scope

<u>Outreach:</u> Colorado Mountain College – Guest Speakers – November 2002
presented the oral presentations “A landscape scale assessment of the Grand Staircase-Escalante National Monument” and “The contribution of rare habitats and endemic species to overall plant diversity in the Grand Staircase-Escalante National Monument” and discussed current issues in the Monument

<u>Presentations at the Ecological Society of America 87th Annual Meeting, Tucson, Arizona, August 2002</u>

Schnase, J.L., T.J. Stohlgren, R.M. Reich, M.A. Kalkhan, and J.A. Smith. **Next-generation spatial modeling for ecological forecasting.** *Oral presentation.*

Guenther, D.A., T.J. Stohlgren, P.H. Evangelista, and M.A. Kalkhan. **The contribution of rare habitats and endemic plant species to overall plant diversity in the Grand Staircase-Escalante National Monument, Utah.** *Oral presentation.*

Alley, N.W., R.O. Coleman, T.J. Stohlgren, P.H. Evangelista, and D.A. Guenther. **Integrating various data layers and multi-media in a comprehensive format for land managers: A case study at Grand Staircase – Escalante National Monument, Utah.** *Poster presentation.*

<u>Presentation:</u> Tom Stohlgren, Invited Speaker at the 40th Goddard Memorial Symposium: Partnering with NASA – The Wave of the Future. March 20-21, 2002, Greenbelt, MD. Title: Ecological Forecasting of Invasive Species – High Performance Computing Needs.

<u>Presentation:</u> Tom Stohlgren, Invited Speaker at the Biology and Mapping meeting at EROS Data Center (April 30-May 2, 2002): Won Best Talk Award.

<u>Presentations at the Conference of Research on the Colorado Plateau, Flagstaff, Arizona, November 2001</u>

Guenther, D.A., P.E. Evangelista, and T.J. Stohlgren. **Invasive and native plant species on relict sites compared to grazed landscapes in the Grand Staircase – Escalante National Monument, Utah.** *Oral presentation.*

Kalkhan, M.A., T.J. Stohlgren, P.E. Evangelista, and D.A. Guenther. **Using multi-scale sampling, spatial information, and spatial statistics to predict vegetation, cryptobiotic, and soil characteristics: a case study at Grand Staircase-Escalante National Monument, Utah.** *Oral presentation.*

Evangelista, P., D. Guenther, and T. Stohlgren. **Effects of disturbances on cryptobiotic crusts in Grand Staircase – Escalante National Monument, Utah.** *Poster presentation.*

<u>Presentations at the Ecological Society of America 86th Annual Meeting, Madison, Wisconsin, August 2001</u>

Otsuki, Y., T. J. Stohlgren, D. Guenther, and C. Villa. **Evaluating plant invasions from both habitat and species perspectives.** Oral presentation.

Evangelista, P., D. Guenther and T. Stohlgren. **Disturbance effects on cryptobiotic crusts in the Grand Staircase-Escalante National Monument, Utah.** *Oral presentation.*

Kalkhan, M., T. Stohlgren, P. Evangelista and D. Guenther. **Predictive spatial models of vegetation, cryptobiotic, and soil characteristics: A landscape-scale assessment using spatial information and spatial statistics.** *Oral presentation.*

Presentation at the Society of American Foresters Conference, Washington, D.C.

Evangelista, P.H., Y. Otsuki, and T.J. Stohlgren. Effects of fire and post-burn treatments on native and exotic vascular plant richness and cryptobiotic soils in the Grand Staircase – Escalante National Monument, Utah. *Poster presentation.*

Presentations at the Bureau of Land Management, Grand Staircase - Escalante National Monument Project Meeting, Kanab, Utah, October 30, 2000

Participants included Grand Staircase Escalante Monument staff and other area researchers.

Stohlgren, T.J., M. A. Kalkhan, P.E. Evangelista, M. Bashkin, D.A. Guenther. **An overview of objectives, goals, management implications, and future directions for the landscape-scale assessment of plant diversity at Grand Staircase-Escalante National Monument.**

Evangelista, P.E., T.J. Stohlgren, Y.Otsuki, and D. Guenther. **Effects of fire and post-seeding treatments on native and exotic vascular plant richness and cryptobiotic soils in the Buckskin Gulch Burn area of Grand Staircase-Escalante National Monument.**

Bashkin, M., T.J. Stohlgren, Y. Otsuki, M. Lee, P. Evangelista, J. Belnap. **Soil characteristics and exotic plant species invasions in the Grand Staircase-Escalante National Monument, Utah.**

Kalkhan, M.A., T.J. Stohlgren, and G.W. Chong. **Landscape-scale assessment of plant diversity: Integration of spatial information and spatial statistics.**

Guenther, D.A., T.J. Stohlgren, P.E. Evangelista, M. Hart, R. Beam. **Preliminary results for plant, soil, and cryptobiotic crust data collected on burned, seeded, and/or chained areas in the Skutumpah Area of Grand Staircase-Escalante National Monument.**

Guenther, D.A., T.J. Stohlgren, P.E. Evangelista, S. Stewart. **Preliminary results for baseline plant, soil, and cryptobiotic crust data collected on the Escalante River of Grand Staircase-Escalante National Monument.**

Presentations at the Ecological Society of America 85th Annual Meeting, Snowbird, Utah, August 2000

Y. Otsuki, T. Stohlgren, M. Lee, J. Belnap, C. Villa. **Exotic plant invasions in GSENM.** *Oral presentation.*

M. Khalkan, T. Stohlgren, G. Chong. **Landscape-scale assessment of plant diversity: Integration of spatial information and spatial statistics**. *Oral presentation.*

D. McCrumb, M. Coughenour, D. Ojima, T. Stohlgren, B. Woodmansee. **Video imagery: A communication tool for ecological studies**. *Poster presentation.*

P. Evangelista, Y. Otsuki, T. Stohlgren. **Effects of fire and post-burn seeding treatments on native and exotic vascular plant richness and cryptobiotic soils in GSENM, Utah**. *Poster presentation*

Presentation: 9/13/1999: Tom Stohlgren presented the results from the 1998 data analyses to the GSENM Planning Committee in Cedar City, Utah.

Presentation: 3/5/1999; Tom Stohlgren presented "New approaches to assess native and exotic plant diversity at landscape scales" at the Wyoming Native Plant Society meeting.

Presentation: 3/5/1999; Tom Stohlgren presented "New approaches to assess native and

exotic plant diversity at landscape scales" at the Utah Native Plant Society meeting.

OTHER PRODUCTS

Spatial Predictive Models

For Plants: non-native species richness, non-native species cover, native species richness, native species cover, total plant species richness, total plant species cover, cryptobiotic crust cover, cheatgrass cover

For Soils: clay, sand, silt, clay, magnesium, potassium, sodium, nitrogen, organic carbon, inorganic carbon, organic carbon to nitrogen ratio

Other Layers: band 1, 2, 3, 4, 5, 6 (low gain), 6 (high gain), 7, 8; tasel cap 1, 2, 3, 4, 5, 6

Pick and Click

Pick and click software that includes spatially referenced pictures of each plot and plot summaries.

Herbarium

A working herbarium that includes 692 mounted plants.

Database

The complete Access database that includes all plot data.

Posters/Photographs

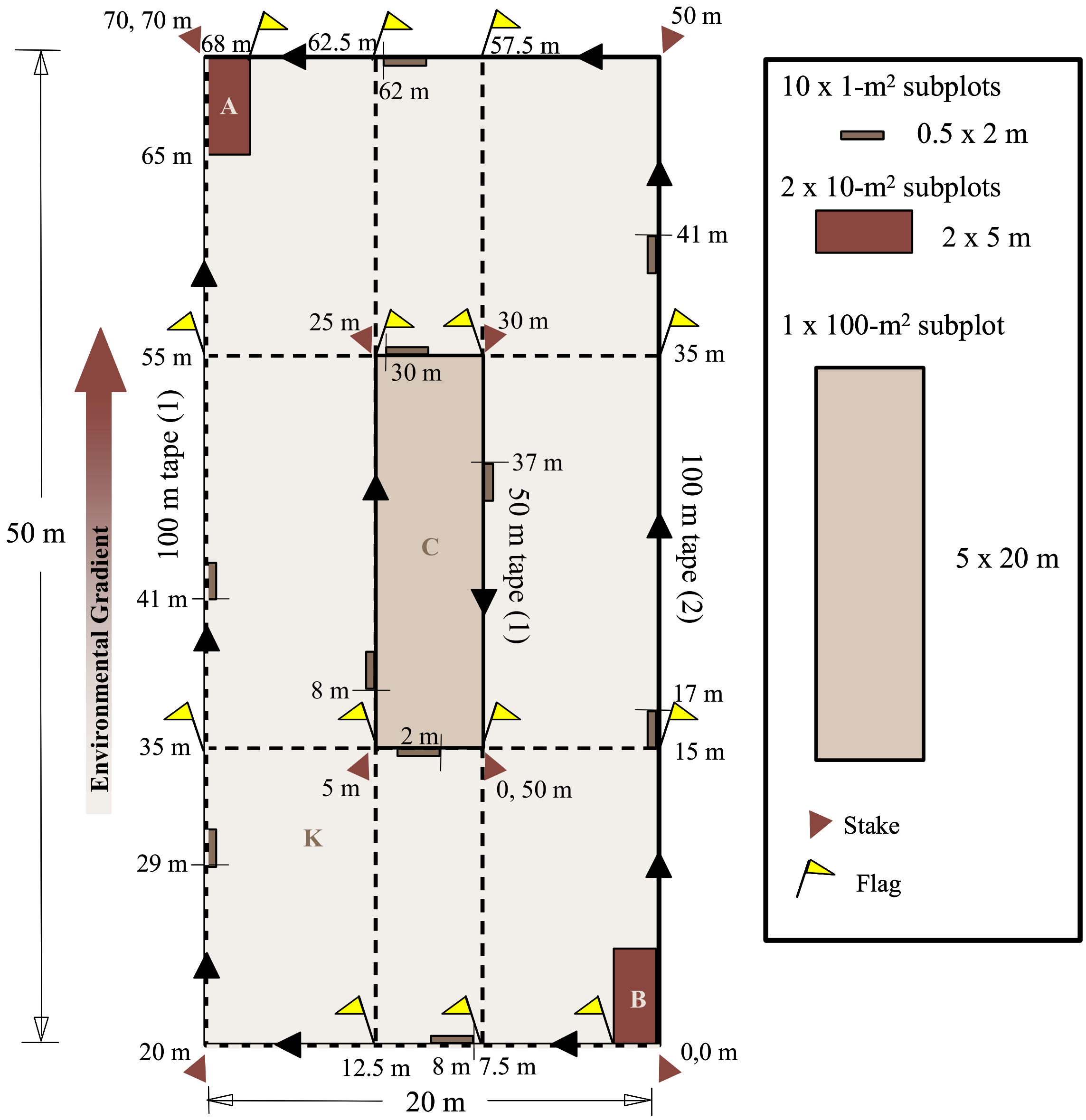
A CD that includes posters used for presentations associated with the project and all photographs taken during the study period.

Website

A website was created for the project and can be found at http://www.nrel.colostate.edu/projects/escalante/index.html.

Appendix B

The following diagram and text (1-9) that follows gives step-by-step instructions for setting up the modified-Whittaker plot.



1.) Look around the area and determine the environmental gradient in order to position the long side (50 m) of the 1000 m^2 plot (K) parallel to the gradient. (The goal is to cover the most variation possible.)

2.) Using two 100 m tapes (1 and 2), lay out the main plot (K) 20 x 50 m rectangle. Anchor tape ends at the starting point (right bottom corner, label 0,0). Using a compass to shoot a 90^o angle, run tapes out following the arrows. Anchor the 20 m and 50 m corners.

3.) Flag 7.5, 12.5, 35, and 55 m marks of tape (1), and 15, 35, 57.5, and 62.5 m marks of tape (2).

4.) Using a 50 m tape (3) and the flags marked at 7.5 m and 35 m of tape (1) and 15 m of tape (2), determine and anchor the starging point (0, 50 m) of subplot C (inner rectangle). Moving clockwise, lay out this subplot.

5.) Take a magnetic azimuth (bearing) of tape (1) and tape (2) at the (0, 0) corner. Write it down on your field notebook with the plot name, site description and UTM coordinates from the GPS unit. (Or use the site description sheet.)

6.) Starting from the (0,0) point of K, walk to the 8 m mark of tape (1) and place the 0.5 x 2 m subplot frame along the inside boundary of the K plot. Record all the species present within the subplot. Measure their average heights and determine (estimate) their percent cover. Flag unknown species as you encounter them, and make their labels (numbers) even if you aren't collecting them right away.

7.) When finished at this location, pick up the subplot frame and moving clockwise, repeat step 6 until all ten 1-m^2 subplots are completed. There are six subplots around the inside of K plot, and four subplots around the outside of the C subplot.

8.) Using flags and the subplot frame, set up subplot A and B. (Subplot A and B can be sampled in sequence with the 1-m^2 subplots). Record the species present in the A, B, and C subplots.

9.) Walk through the entire K (1,000-m^2) plot and record any species new to the plot.

10.) Label and collect unknowns.

Other tips can be found at the following web sites: http://www.nrel.colostate.edu/projects/stohlgren/stohlgrensamplingmwnotes.html; http://www.nrel.colostate.edu/projects/fhm/equipment/SamplingFrame/VegSamplingFrameInstructions.htm; http://www.nrel.colostate.edu/projects/fhm/equipment/PlantCollecting/PlantCollectingRig.htm.

Appendix C

In 1998, a handful of plots had their names changed after returning from the field to aid in analysis. Therefore, these plots have slightly different names on their benchmarks in the field than what was used in the database. The affected plots are listed here to aid in relocating the plots in the future.

Plot Name: Database	Plot Name: Benchmark	UTM E	UTM N	Comments
wash22b	pine22b	486816	4130614	
dry103a	dry3a	481094	4133273	already had a dry3a-see next row
dry03a	dry3a	488030	4128180	
wash103b	wash3b	481033	4133031	
wet134a	wet34a	501598	4142175	

Appendix D

Summary climate data from 1998-2002 for five weather stations surrounding the Monument (NCDC 2003). Data for 2003 is not yet available, but can be accessed at http://www4.ncdc.noaa.gov/cgi-win/wwcgi.dll?WWDI~getstate~USA at a later date.

Table Appendices-1. Weather data from Bryce Canyon National Park Headquarters, Utah (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
January 1998	-4.0	3.2	-11.2	9.4	-20.0	24.4	330.2
February 1998	-5.7	-0.1	-11.3	3.3	-21.1	72.1	787.4
March 1998	-1.9	5.9	-9.7	16.7	-20.0	43.2	381.0
April 1998	1.3	8.2	-5.4	19.4	-10.0	26.9	304.8
May 1998	7.2	15.8	-1.4	21.1	-5.0	9.9	0.0
June 1998	11.1	20.4	1.7	31.1	-2.2	40.4	63.5
July 1998	17.7	26.7	8.7	32.2	5.6	81.5	0.0
August 1998	16.5	25.6	7.4	29.4	2.2	20.6	0.0
September 1998	11.4	19.2	3.7	23.9	-1.7	135.9	0.0
October 1998	4.5	12.1	-3.2	21.1	-7.2	84.1	165.1
November 1998	-0.7	6.9	-8.2	12.2	-16.1	40.6	452.1
December 1998	-4.5	2.9	-11.9	11.7	-23.9	15.2	254.0
1998 Average	**4.4**	**12.2**	**-3.4**	**19.3**	**-10.0**	**49.6**	**228.2**
January 1999	-1.9	4.4	-8.2	10.6	-14.4	6.6	139.7
February 1999	-2.3	4.9	-9.5	9.4	-19.4	17.3	139.7
March 1999	2.4	10.2	-5.3	15.0	-9.4	3.0	38.1
April 1999	0.8	7.4	-5.7	20.6	-13.9	60.2	548.6
May 1999	6.5	15.2	-2.3	21.1	-6.1	11.7	33.0
June 1999	12.9	22.2	3.5	27.8	-1.7	30.0	50.8
July 1999	16.8	24.9	8.7	30.6	5.0	41.4	0.0
August 1999	14.8	22.8	6.9	25.6	3.9	99.8	0.0
September 1999	10.8	19.1	2.6	25.6	-5.6	25.9	0.0
October 1999	7.4	18.0	-3.2	22.8	-7.8	0.0	0.0
November 1999	1.8	10.9	-7.3	17.2	-18.9	1.8	76.2
December 1999	-4.3	3.6	-12.2	8.9	-18.3	1.5	114.3
1999 Average	**5.5**	**13.6**	**-2.7**	**19.6**	**-8.9**	**24.9**	**95.0**
January 2000	-2.8	3.7	-9.3	11.7	-20.6	14.2	177.8
February 2000	-2.4	3.7	-8.4	9.4	-17.8	77.2	599.4
March 2000	-0.8	6.5	-8.2	14.4	-12.8	60.7	635.0
April 2000	6.2	15.2	-2.7	23.9	-6.7	2.8	25.4
May 2000	11.4	20.5	2.3	28.9	-9.4	6.1	0.0
June 2000	NA	NA	NA	NA	NA	NA	NA

Table Appendices-1. (Continued) Weather data from Bryce Canyon National Park Headquarters, Utah (NCDC 2003).

Month	Monthly Temperature (°C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
July 2000	17.9	27.3	8.4	32.2	2.2	46.5	0.0
August 2000	17.4	26.2	8.6	32.2	3.9	62.0	0.0
September 2000	12.5	22.7	2.3	28.9	-6.1	15.2	0.0
October 2000	5.8	12.3	-0.7	23.9	-7.2	161.5	393.7
November 2000	-5.2	1.3	-11.8	7.2	-21.1	9.9	203.2
December 2000	-1.8	4.2	-7.9	8.3	-13.9	11.9	127.0
2000 Average	**5.3**	**13.1**	**-2.5**	**20.1**	**-9.9**	**42.6**	**196.5**
January 2001	-5.4	0.0	-10.8	10.0	-17.8	77.7	947.4
February 2001	-4.1	1.6	-9.7	11.1	-21.1	40.1	419.1
March 2001	0.4	6.0	-5.3	12.2	-13.9	41.1	373.4
April 2001	3.2	10.2	-3.7	19.4	-12.8	52.8	612.1
May 2001	10.6	19.4	1.7	26.7	-5.6	2.8	7.6
June 2001	14.9	24.0	5.7	28.3	-1.1	10.7	0.0
July 2001	17.1	25.0	9.1	29.4	4.4	38.9	0.0
August 2001	16.4	24.7	8.1	28.9	5.0	28.2	0.0
September 2001	13.1	22.4	3.7	26.1	-3.3	3.3	0.0
October 2001	8.6	16.9	0.2	22.8	-6.1	3.0	0.0
November 2001	0.6	6.9	-5.7	13.9	-17.8	23.4	198.1
December 2001	-6.4	-0.6	-12.3	4.4	-19.4	11.2	172.7
2001 Average	**5.7**	**13.1**	**-1.6**	**19.4**	**-9.1**	**27.8**	**227.5**
January 2002	-5.5	1.4	-12.4	10.0	-21.1	12.2	198.1
February 2002	-4.3	3.6	-12.3	12.8	-22.2	1.5	25.4
March 2002	-1.7	5.8	-9.1	15.0	-19.4	14.0	241.3
April 2002	6.0	14.4	-2.5	19.4	-10.6	3.3	0.0
May 2002	9.0	18.2	-0.2	29.4	-8.3	0.5	0.0
June 2002	16.2	26.4	5.9	30.0	-2.8	1.5	0.0
July 2002	19.7	28.3	11.0	36.7	6.7	14.7	0.0
August 2002	17.2	26.8	7.6	31.7	3.3	2.3	0.0
September 2002	11.6	19.2	3.9	26.1	-1.7	100.6	0.0
October 2002	4.6	12.1	-3.1	18.3	-5.6	40.4	292.1
November 2002	-0.6	7.0	-8.2	12.2	-13.9	2.8	0.0
December 2002	-5.4	0.8	-11.6	6.1	-19.4	18.0	190.5
2002 Average	**5.6**	**13.7**	**-2.6**	**20.6**	**-9.6**	**17.7**	**79.0**

Table Appendices-2. Weather data from Escalante, Utah (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
January 1998	1.6	8.8	-5.6	15.0	-12.2	2.0	12.7
February 1998	1.1	6.2	-4.0	10.0	-9.4	40.6	228.6
March 1998	5.7	13.7	-2.3	24.4	-8.9	14.0	50.8
April 1998	8.2	16.1	0.3	26.7	-5.0	21.6	114.3
May 1998	13.7	23.2	4.2	27.8	0.0	14.0	0.0
June 1998	17.7	28.3	7.2	37.2	3.9	9.7	0.0
July 1998	24.0	34.2	13.8	39.4	11.1	15.0	0.0
August 1998	23.1	32.9	13.2	37.2	8.9	33.8	0.0
September 1998	17.8	26.5	9.1	31.1	3.9	98.3	0.0
October 1998	10.5	18.7	2.3	26.7	-1.7	52.1	25.4
November 1998	4.7	11.9	-2.6	17.2	-8.9	10.7	76.2
December 1998	0.4	7.9	-7.1	16.1	-15.0	0.0	0.0
1998 Average	**10.7**	**19.0**	**2.4**	**15.7**	**-2.8**	**26.0**	**42.3**
January 1999	3.2	10.6	-4.2	15.6	-11.1	1.5	0.0
February 1999	3.6	11.9	-4.7	18.3	-11.1	9.1	50.8
March 1999	7.8	17.0	-1.4	21.1	-6.7	0.0	0.0
April 1999	7.3	14.9	-0.3	26.7	-7.8	45.2	444.5
May 1999	13.9	23.7	4.2	29.4	-1.7	10.9	0.0
June 1999	19.2	29.6	8.8	36.1	3.3	24.6	0.0
July 1999	22.9	31.9	13.8	37.2	11.1	63.2	0.0
August 1999	20.8	29.6	12.1	32.2	7.8	93.0	0.0
September 1999	17.1	26.4	7.8	32.8	-1.1	27.2	0.0
October 1999	13.0	23.9	2.1	30.0	-5.0	0.0	0.0
November 1999	6.3	16.1	-3.4	21.7	-12.2	0.0	0.0
December 1999	0.7	8.4	-7.1	13.9	-13.3	2.5	33.0
1999 Average	**11.3**	**20.3**	**2.3**	**26.3**	**-4.0**	**23.1**	**44.0**
January 2000	2.3	9.7	-5.0	17.2	-13.9	2.5	0.0
February 2000	4.1	10.9	-2.8	15.6	-8.3	22.6	50.8
March 2000	6.1	13.8	-1.8	22.8	-6.7	23.6	190.5
April 2000	12.8	23.0	2.7	31.1	-2.8	1.3	0.0
May 2000	17.7	28.2	7.2	35.6	-2.8	2.3	0.0
June 2000	21.8	32.5	11.2	35.6	6.7	9.1	0.0
July 2000	24.1	35.3	12.9	38.9	7.2	1.5	0.0
August 2000	23.6	32.9	14.3	38.9	8.3	67.6	0.0
September 2000	18.6	28.7	8.5	34.4	-0.6	11.4	0.0

Table Appendices-2. Weather data from Escalante, Utah (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
October 2000	11.2	18.8	3.6	31.1	-1.7	86.1	0.0
November 2000	1.3	8.6	-6.0	14.4	-12.8	1.8	0.0
December 2000	1.3	9.2	-6.5	14.4	-11.7	2.8	38.1
2000 Average	**12.1**	**21.0**	**3.2**	**27.5**	**-3.2**	**19.4**	**23.3**
January 2001	-1.9	5.5	-9.3	11.1	-17.8	47.0	495.3
February 2001	1.3	7.7	-5.1	13.3	-15.0	32.8	228.6
March 2001	7.2	15.0	-0.7	22.2	-8.3	27.2	0.0
April 2001	10.5	18.8	2.2	28.3	-5.6	6.6	0.0
May 2001	17.2	27.6	6.9	33.3	-1.7	16.5	0.0
June 2001	21.4	32.6	10.2	37.8	3.9	14.2	0.0
July 2001	22.9	32.8	13.1	38.3	7.2	28.7	0.0
August 2001	22.5	32.3	12.8	36.1	6.7	52.6	0.0
September 2001	19.7	30.3	9.0	33.9	2.8	4.6	0.0
October 2001	13.7	23.4	3.9	29.4	-1.7	4.3	0.0
November 2001	5.4	13.2	-2.5	20.6	-13.9	10.7	63.5
December 2001	NA	NA	NA	NA	NA	NA	NA
2001 Average	**12.7**	**21.7**	**3.7**	**27.7**	**-3.9**	**22.3**	**71.6**
January 2002	-0.8	6.5	-8.2	12.2	-15.6	0.8	20.3
February 2002	1.5	10.7	-7.7	18.3	-20.0	0.8	12.7
March 2002	5.1	14.2	-4.2	24.4	-12.8	0.5	0.0
April 2002	12.9	22.6	3.2	27.2	-3.3	4.1	0.0
May 2002	15.7	26.6	4.7	36.7	-1.7	0.0	0.0
June 2002	22.4	33.8	10.9	38.3	2.2	1.0	0.0
July 2002	25.8	36.2	15.3	40.0	11.1	12.2	0.0
August 2002	23.3	34.2	12.3	40.0	7.8	7.4	0.0
September 2002	18.3	27.6	8.9	34.4	2.8	41.9	0.0
October 2002	10.2	19.3	1.1	25.6	-2.8	37.1	0.0
November 2002	4.6	12.3	-3.2	16.7	-8.9	4.1	0.0
December 2002	-0.2	6.3	-6.7	12.8	-12.8	11.9	20.3
2002 Average	**11.5**	**20.9**	**2.2**	**27.2**	**-4.5**	**10.1**	**4.4**

Table Appendices-3. Weather data from Kanab, Utah (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
January 1998	2.9	9.9	-4.1	15.6	-10.0	41.1	91.4
February 1998	2.7	7.8	-2.4	12.2	-10.0	97.5	330.2
March 1998	6.8	14.6	-0.9	23.3	-7.8	75.7	152.4
April 1998	8.7	16.3	1.1	25.6	-5.6	48.3	342.9
May 1998	13.2	22.7	3.7	27.2	-2.2	13.0	0.0
June 1998	17.9	28.2	7.6	36.1	2.8	11.4	0.0
July 1998	23.9	33.1	14.7	38.9	10.0	45.5	0.0
August 1998	23.9	32.3	15.4	35.6	8.3	15.5	0.0
September 1998	18.8	26.7	10.9	31.7	4.4	140.2	0.0
October 1998	11.8	19.8	3.7	26.7	-3.9	54.6	0.0
November 1998	6.2	13.8	-1.4	19.4	-8.3	70.6	177.8
December 1998	2.6	10.0	-4.8	18.3	-14.4	1.0	12.7
1998 Average	**11.6**	**19.6**	**3.6**	**25.9**	**-3.1**	**51.2**	**92.3**
January 1999	4.1	12.3	-4.3	17.2	-8.9	15.0	111.8
February 1999	5.0	13.1	-3.1	21.1	-8.3	25.1	25.4
March 1999	8.4	18.3	-1.4	22.8	-6.7	1.8	0.0
April 1999	8.8	16.6	1.1	26.7	-8.9	53.8	228.6
May 1999	14.8	24.1	5.4	28.9	-1.7	6.4	0.0
June 1999	19.8	29.9	9.8	36.1	2.8	21.3	0.0
July 1999	23.0	30.9	15.1	37.2	11.7	107.4	0.0
August 1999	22.9	31.2	14.6	33.9	8.9	38.1	0.0
September 1999	19.1	28.1	10.1	32.8	3.9	35.6	0.0
October 1999	14.4	25.6	3.2	30.0	-2.8	0.5	0.0
November 1999	9.4	18.6	0.2	25.0	-10.0	0.8	0.0
December 1999	2.3	10.6	-5.9	16.1	-10.0	3.0	76.2
1999 Average	**12.7**	**21.6**	**3.7**	**27.3**	**-2.5**	**25.7**	**36.8**
January 2000	4.1	11.3	-3.1	18.3	-13.9	24.6	63.5
February 2000	6.1	12.5	-0.3	18.3	-7.8	65.5	114.3
March 2000	7.4	15.2	-0.3	22.8	-4.4	58.2	177.8
April 2000	13.9	23.0	3.8	30.0	-1.7	10.4	0.0
May 2000	18.3	28.3	8.3	35.0	-1.7	2.0	0.0
June 2000	22.2	32.1	12.4	35.6	6.7	13.5	0.0
July 2000	23.6	33.8	13.3	38.3	6.7	15.7	0.0
August 2000	23.6	31.8	15.3	37.2	9.4	54.6	0.0
September 2000	20.1	29.5	10.6	33.9	2.2	8.9	0.0

Table Appendices-3. (Continued) Weather data from Kanab, Utah (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
October 2000	13.1	20.4	5.8	30.0	-1.1	102.6	0.0
November 2000	3.2	10.1	-3.7	16.7	-10.6	7.9	139.7
December 2000	3.7	11.4	-3.9	16.1	-8.3	15.0	50.8
2000 Average	**13.3**	**21.7**	**4.9**	**27.7**	**-2.0**	**31.6**	**45.5**
January 2001	1.4	7.2	-4.3	16.7	-12.2	56.1	571.5
February 2001	3.9	10.3	-2.5	18.3	-13.3	60.5	457.2
March 2001	8.4	16.1	0.7	22.2	-4.4	46.7	121.9
April 2001	11.0	19.4	2.5	28.3	-4.4	37.8	228.6
May 2001	18.2	28.6	7.9	34.4	-3.3	13.0	0.0
June 2001	21.4	32.3	10.6	36.7	3.9	3.0	0.0
July 2001	23.2	32.6	13.8	38.3	7.8	30.2	0.0
August 2001	23.6	33.3	13.9	36.7	9.4	37.1	0.0
September 2001	20.3	30.6	9.9	32.8	6.7	16.3	0.0
October 2001	14.7	24.3	5.1	30.0	-1.1	6.9	0.0
November 2001	7.1	14.3	-0.1	21.1	-11.1	17.8	152.4
December 2001	0.1	6.2	-6.0	12.2	-17.8	49.0	360.7
2001 Average	**12.8**	**21.3**	**4.3**	**27.3**	**-3.3**	**31.2**	**157.7**
January 2002	1.8	9.0	-5.5	17.8	-15.0	2.3	50.8
February 2002	4.2	13.4	-5.2	20.6	-10.0	2.3	50.8
March 2002	6.8	15.9	-2.3	24.4	-11.7	8.4	139.7
April 2002	13.3	23.2	3.4	27.8	-3.9	7.4	0.0
May 2002	15.6	26.3	4.8	36.1	-2.2	3.0	0.0
June 2002	21.5	33.0	10.0	35.6	3.9	0.0	0.0
July 2002	25.9	35.8	16.1	40.0	11.7	8.1	0.0
August 2002	22.9	33.8	12.1	38.3	7.2	5.8	0.0
September 2002	18.7	27.4	10.0	33.9	2.8	65.0	0.0
October 2002	12.2	20.4	.0	27.8	-2.8	50.3	0.0
November 2002	6.9	14.3	-0.4	21.7	-5.6	73.9	0.0
December 2002	1.4	7.7	-5.0	14.4	-11.7	35.3	238.8
2002 Average	**12.6**	**21.7**	**3.5**	**40.0**	**-15.0**	**21.8**	**40.0**

Table Appendices-4. Weather data from Panguitch, Utah (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
January 1998	-1.2	6.7	-9.0	12.2	-18.9	6.4	88.9
February 1998	-1.1	4.7	-6.8	8.9	-18.3	17.0	63.5
March 1998	2.5	11.7	-6.7	22.8	-18.3	15.7	0.0
April 1998	5.3	13.9	-3.3	23.9	-8.9	8.9	0.0
May 1998	11.1	21.3	0.8	26.7	-4.4	2.0	0.0
June 1998	14.8	26.2	3.3	35.6	-1.7	18.0	0.0
July 1998	21.0	32.1	10.0	38.3	5.6	45.0	0.0
August 1998	20.2	31.3	8.9	3.9	3.3	20.3	0.0
September 1998	15.1	24.9	5.4	30.0	-1.1	92.7	0.0
October 1998	7.6	17.0	-1.9	25.0	-6.7	70.4	101.6
November 1998	2.4	10.8	-6.1	16.7	-17.8	7.9	38.1
December 1998	-3.2	6.4	-12.8	13.3	-27.2	1.8	38.1
1998 Average	**7.9**	**17.3**	**-1.5**	**23.9**	**-9.5**	**25.5**	**27.5**
January 1999	4.4	9.9	-1.1	14.4	-3.3	2.8	0.0
February 1999	6.0	12.5	-0.5	20.6	-6.1	3.6	0.0
March 1999	11.3	18.6	4.2	23.9	0.6	2.3	0.0
April 1999	11.1	17.8	4.3	27.2	-2.2	20.6	0.0
May 1999	18.3	25.7	11.0	31.7	3.3	4.3	0.0
June 1999	24.6	31.8	17.3	38.3	10.0	5.1	0.0
July 1999	27.3	34.1	20.6	40.0	18.3	26.7	0.0
August 1999	25.4	32.3	18.6	35.6	15.6	40.4	0.0
September 1999	21.4	28.1	14.6	32.8	6.1	12.2	0.0
October 1999	16.3	24.1	8.5	31.7	3.3	0.0	0.0
November 1999	8.8	15.6	1.9	23.3	-5.0	0.0	0.0
December 1999	2.5	8.0	-3.0	15.0	-7.2	3.6	0.0
1999 Average	**14.8**	**21.5**	**8.0**	**27.9**	**2.8**	**10.1**	**0.0**
January 2000	4.4	9.8	-0.9	15.6	-7.2	9.1	0.0
February 2000	6.2	11.7	0.7	17.8	-2.2	11.7	0.0
March 2000	8.8	14.7	2.9	22.2	-2.2	19.3	25.4
April 2000	16.9	24.3	9.5	32.8	2.8	1.0	0.0
May 2000	22.2	29.7	14.6	38.9	6.7	2.8	0.0
June 2000	26.4	33.9	19.0	38.3	14.4	1.0	0.0
July 2000	28.9	36.6	21.3	39.4	16.1	0.8	0.0
August 2000	27.9	34.7	21.2	38.9	15.0	9.7	0.0
September 2000	23.1	30.2	15.9	36.1	5.0	2.0	0.0

Table Appendices-4. (Continued) Weather data from Panguitch, Utah (NCDC 2003).

Month	Monthly Temperature (^oC)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
October 2000	15.2	20.8	9.6	32.8	2.8	79.0	0.0
November 2000	4.2	8.8	-0.4	13.9	-3.9	4.3	0.0
December 2000	3.3	8.1	-1.3	11.7	-5.6	6.1	0.0
2000 Average	**15.6**	**21.9**	**9.3**	**28.2**	**3.5**	**12.2**	**2.1**
January 2001	1.6	5.7	-2.5	9.4	-5.6	22.4	109.2
February 2001	4.2	9.1	-0.7	15.0	-8.9	18.8	50.8
March 2001	10.1	15.9	4.2	22.8	-0.6	16.8	0.0
April 2001	14.0	20.3	7.7	28.9	1.1	6.1	0.0
May 2001	21.4	28.9	13.9	36.1	3.9	3.0	0.0
June 2001	25.9	33.9	18.0	37.8	9.4	16.8	0.0
July 2001	27.7	34.7	20.7	38.9	17.2	21.1	0.0
August 2001	26.9	34.1	19.8	37.8	15.0	22.9	0.0
September 2001	24.1	31.2	16.8	35.0	11.7	5.1	0.0
October 2001	17.2	24.0	10.3	30.6	5.6	0.5	0.0
November 2001	9.2	14.8	3.7	22.2	-5.0	3.6	0.0
December 2001	1.3	5.9	-3.3	12.2	-8.9	7.1	0.0
2001 Average	**15.3**	**21.6**	**9.0**	**27.2**	**2.9**	**12.0**	**13.3**
January 2002	-3.4	4.4	-11.3	11.1	-22.8	2.0	2.5
February 2002	-1.2	9.3	-11.8	17.8	-21.7	0.5	0.0
March 2002	1.9	12.2	-8.3	22.8	-21.1	8.9	0.0
April 2002	9.7	20.1	-0.7	26.7	-6.7	0.3	0.0
May 2002	12.4	24.4	0.5	35.0	-6.7	1.5	0.0
June 2002	18.8	31.7	5.8	36.1	-1.7	2.3	0.0
July 2002	22.3	33.6	11.0	38.9	6.1	17.0	0.0
August 2002	19.3	31.7	6.9	35.6	2.2	6.9	0.0
September 2002	14.4	24.6	4.3	31.7	-1.7	71.4	0.0
October 2002	6.4	16.4	-3.7	22.2	-7.2	31.8	0.0
November 2002	1.6	10.9	-7.8	15.6	-13.3	0.0	2.5
December 2002	-3.9	4.4	-12.2	11.1	-20.6	5.1	53.3
2002 Average	**8.2**	**18.6**	**-2.3**	**25.4**	**-9.6**	**12.3**	**4.9**

Table Appendices-5. Weather data from Page, Arizona (NCDC 2003).

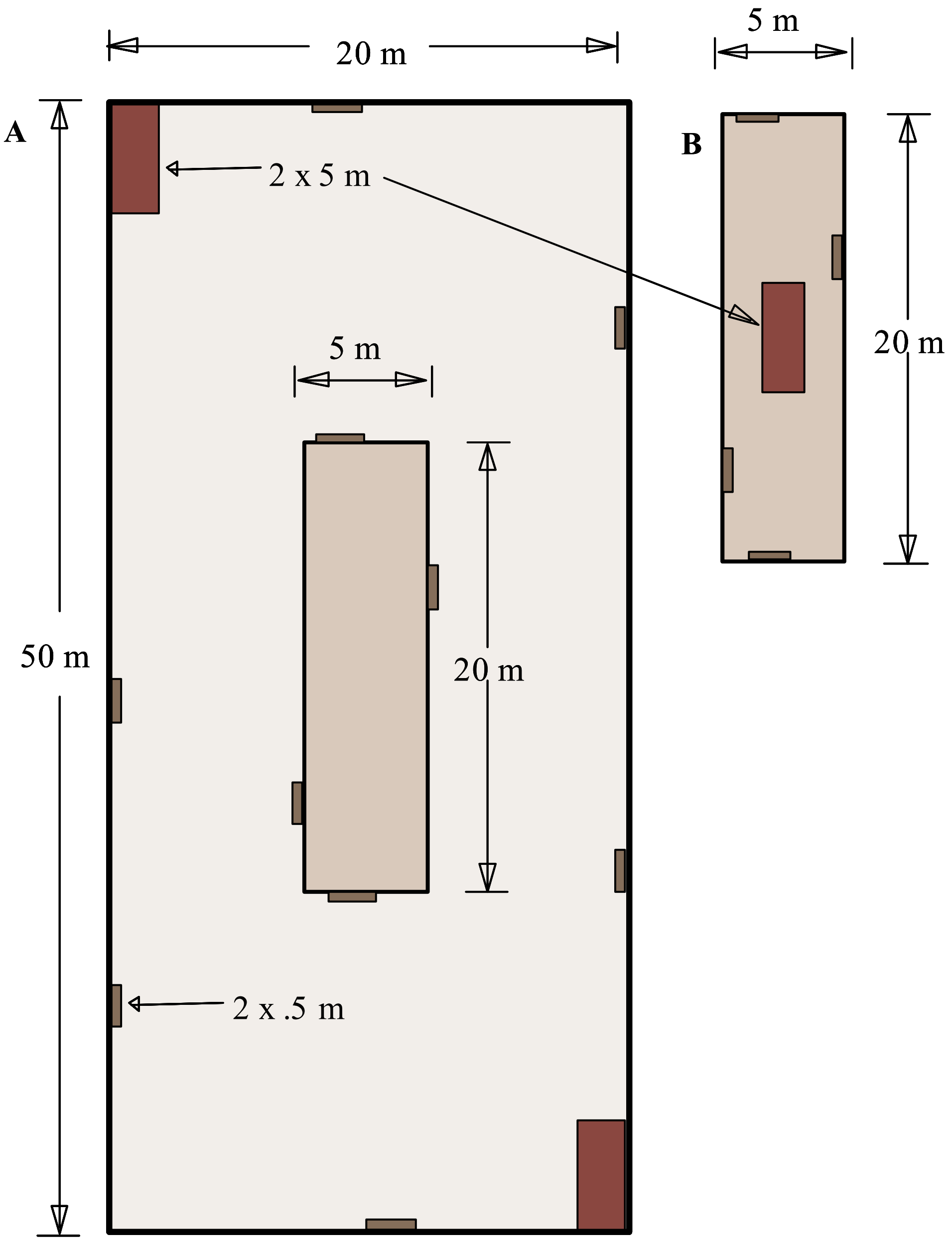
Month	Monthly Temperature (°C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
January 1998	4.1	8.9	-0.8	13.9	-4.4	5.3	0.0
February 1998	4.5	9.1	-0.1	16.1	-5.0	7.9	0.0
March 1998	8.6	14.6	2.5	27.2	-5.0	17.0	33.0
April 1998	11.3	17.4	5.3	29.4	0.0	16.8	0.0
May 1998	18.2	25.2	11.3	31.1	5.0	8.4	0.0
June 1998	23.1	30.6	15.6	38.3	9.4	2.3	0.0
July 1998	28.9	36.3	21.5	40.6	18.3	2.5	0.0
August 1998	27.5	34.6	20.4	37.2	16.1	10.2	0.0
September 1998	21.9	27.7	16.0	33.3	11.1	62.7	0.0
October 1998	13.7	19.6	7.8	27.2	2.2	75.4	0.0
November 1998	7.3	12.6	2.0	20.0	-2.8	12.4	0.0
December 1998	2.0	6.8	-2.8	14.4	-8.3	14.5	58.4
1998 Average	**14.2**	**20.3**	**8.2**	**27.4**	**3.1**	**19.6**	**7.6**
January 1999	4.4	9.9	-1.1	14.4	-3.3	2.8	0.0
February 1999	6.0	12.5	-0.5	20.6	-6.1	3.6	0.0
March 1999	11.3	18.6	4.2	23.9	0.6	2.3	0.0
April 1999	11.1	17.8	4.3	27.2	-2.2	20.6	0.0
May 1999	18.3	25.7	11.0	31.7	3.3	4.3	0.0
June 1999	24.6	31.8	17.3	38.3	10.0	5.1	0.0
July 1999	27.3	34.1	20.6	40.0	18.3	26.7	0.0
August 1999	25.4	32.3	18.6	35.6	15.6	40.4	0.0
September 1999	21.4	28.1	14.6	32.8	6.1	12.2	0.0
October 1999	16.3	24.1	8.5	31.7	3.3	0.0	0.0
November 1999	8.8	15.6	1.9	23.3	-5.0	0.0	0.0
December 1999	2.5	8.0	-3.0	15.0	-7.2	3.6	0.0
1999 Average	**14.8**	**21.5**	**8.0**	**27.9**	**2.8**	**10.1**	**0.0**
January 2000	4.4	9.8	-0.9	15.6	-7.2	9.1	0.0
February 2000	6.2	11.7	0.7	17.8	-2.2	11.7	0.0
March 2000	8.8	14.7	2.9	22.2	-2.2	19.3	25.4
April 2000	16.9	24.3	9.5	32.8	2.8	1.0	0.0
May 2000	22.2	29.7	14.6	38.9	6.7	2.8	0.0
June 2000	26.4	33.9	19.0	38.3	14.4	1.0	0.0
July 2000	28.9	36.6	21.3	39.4	16.1	0.8	0.0
August 2000	27.9	34.7	21.2	38.9	15.0	9.7	0.0
September 2000	23.1	30.2	15.9	36.1	5.0	2.0	0.0

Table Appendices-5. (Continued) Weather data from Page, Arizona (NCDC 2003).

Month	Monthly Temperature (^{o}C)					Monthly Precipitation (mm)	
	Mean monthly temperature	Mean Max Temperature	Mean Min Temperature	Highest Temperature	Lowest Temperature	Precipitation	Snowfall
October 2000	15.2	20.8	9.6	32.8	2.8	79.0	0.0
November 2000	4.2	8.8	-0.4	13.9	-3.9	4.3	0.0
December 2000	3.3	8.1	-1.3	11.7	-5.6	6.1	0.0
2000 Average	**15.6**	**21.9**	**9.3**	**28.2**	**3.5**	**12.2**	**2.1**
January 2001	1.6	5.7	-2.5	9.4	-5.6	22.4	109.2
February 2001	4.2	9.1	-0.7	15.0	-8.9	18.8	50.8
March 2001	10.1	15.9	4.2	22.8	-0.6	16.8	0.0
April 2001	14.0	20.3	7.7	28.9	1.1	6.1	0.0
May 2001	21.4	28.9	13.9	36.1	3.9	3.0	0.0
June 2001	25.9	33.9	18.0	37.8	9.4	16.8	0.0
July 2001	27.7	34.7	20.7	38.9	17.2	21.1	0.0
August 2001	26.9	34.1	19.8	37.8	15.0	22.9	0.0
September 2001	24.1	31.2	16.8	35.0	11.7	5.1	0.0
October 2001	17.2	24.0	10.3	30.6	5.6	0.5	0.0
November 2001	9.2	14.8	3.7	22.2	-5.0	3.6	0.0
December 2001	1.3	5.9	-3.3	12.2	-8.9	7.1	0.0
2001 Average	**15.3**	**21.6**	**9.0**	**27.2**	**2.9**	**12.0**	**13.3**
January 2002	1.7	6.7	-3.3	11.7	-9.4	7.9	63.5
February 2002	4.0	10.2	-2.2	16.7	-7.8	0.0	0.0
March 2002	8.7	16.3	1.1	25.0	-6.7	1.0	0.0
April 2002	17.4	24.8	9.9	30.0	2.8	0.0	0.0
May 2002	20.3	28.4	12.2	39.4	6.1	0.0	0.0
June 2002	27.6	35.9	19.3	39.4	12.8	0.8	0.0
July 2002	30.3	37.4	23.1	41.1	19.4	8.1	0.0
August 2002	28.3	35.8	20.7	40.0	17.2	7.4	0.0
September 2002	22.2	28.8	15.6	35.0	9.4	35.6	0.0
October 2002	14.1	20.3	7.9	27.2	2.8	26.4	0.0
November 2002	7.3	12.7	1.8	18.9	-2.2	6.4	0.0
December 2002	2.3	6.4	-1.8	12.8	-11.7	8.9	0.0
2002 Average	**15.3**	**22.0**	**8.7**	**28.1**	**2.7**	**8.5**	**5.3**

Appendix E

The layout of the (A) modified-Whittaker plot and the (B) 100-m^2 multiple-scale intensive plot (Barnett et al. 2003) used within Hackberry Canyon. These smaller plots were used when modified-Whittaker plots were too large to fit into the study area.



Appendix F

Descriptions of each of the vegetation types used throughout this technical report. It is important to note that these community designations are arbitrary, with one community often meshing into another in one large gradient. Therefore, the designations should be used judiciously.

Vegetation Type	Description
Desert Shrub	Dominated by a mixture of desert shrubs including *Ephedra, Gutierrezia sarothrae, Atriplex confertifolia, Ceratoides lanata, Chrysothamnus* sp. and a small amount of *Coleogyne ramosissima,* soils are very high in inorganic carbon and sodium.
Blackbrush	Dominated by *Coleogyne ramosissima* with few other shrub species present, soils are very low in nitrogen.
Desert Shrub/Grassland	Dominated by the mixture of desert shrubs from above as well as grasses such as *Stipa hymenoides, Hilaria jamesii, Agropyron smithii, Bouteloua gracilis, Bromus tectorum,* or *Stipa comata,* soils are very low in organic carbon.
Sagebrush	Dominated by *Artemisia tridentata* with no pinyon or juniper in the immediate vicinity
Rabbitbrush	Dominated by *Chrysothamnus nauseosus* but can have other mixed shrubs present, soils are low in nitrogen
Juniper/Sage	Dominated by both *Juniperous osteosperma* and *Artemisia tridentata*
Juniper	Overstory dominated purely by *Juniperus osteosperma* (with no Pinyon in the immediate vicinity and little or no *Artemisia tridentata*), soils are high in inorganic carbon
Pinyon-Juniper/Sage	Dominated by *Juniperous osteosperma, Pinus edulis*, and *Artemisia tridentata*
Disturbed Pinyon-Juniper/ Sage	Areas dominated by *Juniperous osteosperma, Pinus edulis,* and *Artemisia tridentata* that have been disturbed in the past by fire, drill seeding, chaining, herbicides (for Juniper) or some combination thereof, soils are very high in nitrogen.
Pinyon-Juniper	Overstory dominated by *Juniperous osteosperma* and *Pinus edulis* with little or no *Artemisia tridentata,* soils are high in inorganic carbon
Pinyon-Juniper/Manzanita	Overstory dominated by Pinyon-Juniper with an *Arctostaphylos patula* dominated understory
Pinyon-Juniper/Oak	Overstory dominated by Pinyon-Juniper and co-dominated by *Quercus gambelli* or other *Quercus* spp. (with 2 to 44% *Quercus* cover)
Pinyon Pine	Overstory dominated purely by *Pinus edulis* (no Juniper in the immediate vicinity), soils are low in nitrogen
Ponderosa Pine/Manzanita	Overstory dominated by *Pinus ponderosa* and most plots have an *Arctostaphylos patula* or mountain shrub dominated understory
Mountain Shrub	Overstory dominated by *Cercocarpus montanus, Purshia mexicana, Amelanchier* spp., and *Symphoricarpos* spp. with possibly *Juniperus osteosperma* or *Pinus edulis* present but not dominant
Aspen	Overstory dominanted by *Populus tremuloides*. Soils are high in nitrogen, phosphorous, and organic carbon and low in inorganic carbon.
Wet Meadow	Water saturated meadow dominated by *Juncus* spp., *Carex* spp., and *Poa* spp. Soils very high in phosphorous, organic carbon, and calcium.
Perennial Riparian	Communities along permanent running water sources, dominated by *Populus fremontii, Tamarix* spp., *Salix* spp., *Equisetum* spp., and *Elaeagnus angustifolia,* soils are low in organic carbon and nitrogen but high in potassium.
Spring	Vegetation surrounding a spring, usually dominated by *Scirpus* and *Juncus* spp. Due to size of MW plot, sometimes upland vegetation is included as well. Soils are high in phosphorous.

* Dominance was determined based on photographs and the top 5 dominant species by cover in the plot.

** Immediate vicinity refers to the Modified-Whittaker plot and a buffer of about 20 m around it.

Appendix G

Soil cores were taken at two depths, 0-5 cm and 0-15 cm at the Monument in the summer of 2001 to evaluate whether sampling at a shallower soil depth would affect our soil chemistry results. Sites included the Buckskin Mountain area, No Man's Mesa, Deer Spring Point, and Timber Mountain. The dominant vegetation types on these sites were Pinyon, Juniper, and Sage. No significant differences were found in texture or nutrients when comparing the two soil depths. Averages are reported below with the standard errors in parentheses.

Soil Depth	Sand	Silt	Clay	N	Org C	P	Ca	Mg	Na	K
0-5 cm	78.7 (3.6)	10.1 (2.3)	11.3 (1.4)	0.071 (0.01)	0.729 (0.11)	8.19 (1.8)	664.6 (56.4)	134.3 (30.8)	32.5 (3.2)	83.6 (16.4)
0-15 cm	77.0 (3.8)	9.4 (2.1)	13.7 (1.8)	0.070 (0.01)	0.941 (0.20)	5.79 (1.4)	727.5 (59.1)	149.7 (32.2)	33.1 (3.4)	88.9 (17.6)
23	23	23	23	24	23	24	24	24	24	24
p-value	0.75	0.82	0.29	0.94	0.37	0.3	0.44	0.73	0.91	0.83

Appendix H

A list of the 552 plant species found during our sampling effort.

NRCS Code	Scientific Name	Common Name	Origin
abfr2	*Abronia fragrans*	snowball sand verbena	Native
abna	*Abronia nana*	dwarf sand verbena	Native
acsp	*Acamptopappus sphaerocephalus*	rayless goldenhead	Native
acgl	*Acer glabrum*	rocky mountain maple	Native
acgr3	*Acer grandidentatum*	bigtooth maple	Native
acne2	*Acer negundo*	boxelder	Native
acmil	*Achillea millefolium ssp. lanulosa*	native yarrow	Native
agau2	*Agoseris aurantiaca*	orange agoseris	Native
aggl	*Agoseris glauca*	pale agoseris	Native
agcr	*Agropyron cristatum*	crested wheatgrass	Introduced
agst2	*Agrostis stolonifera*	creeping bentgrass	Native
alac4	*Allium acuminatum*	tapertip onion	Native
alne	*Allium nevadense*	nevada onion	Native
amal	*Amaranthus albus*	prostrate pigweed	Native
ambl	*Amaranthus blitoides*	mat amaranth	Introduced
amac2	*Ambrosia acanthicarpa*	flatspine burr ragweed	Native
amut	*Amelanchier utahensis*	utah serviceberry	Native
anbr4	*Androstephium breviflorum*	pink funnellily	Native
anmi3	*Antennaria microphylla*	littleleaf pussytoes	Native
apca	*Apocynum cannabinum*	indianhemp	Native
apme	*Apocynum medium*		Native
aquil	*Aquilegia*	columbine	Native
arde	*Arabis demissa*	nodding rockcress	Native
ardr	*Arabis drummondii*	drummond's rockcress	Native
arhop3	*Arabis holboellii hornem. var. pinetorum*	holboell`s rockcress	Native
armi	*Arabis microphylla*	littleleaf rockcress	Native
arpe2	*Arabis perennans*	perennial rockcress	Native
arpu2	*Arabis pulchra*	beauty rockcress	Native
ardi3	*Arceuthobium divaricatum*	pinyon dwarf mistletoe	Native
arpa6	*Arctostaphylos patula*	greenleaf manzanita	Native
arpu5	*Arctostaphylos pungens*	pointleaf manzanita	Native
arfe3	*Arenaria fendleri*	fendler's sandwort	Native
armu	*Argemone munita*	flatbud pricklypoppy	Native
arpu9	*Aristida purpurea*	purple threeawn	Native
arbi2	*Artemisia biennis*	biennial wormwood	Native and Introduced
arbi3	*Artemisia bigelovii*	bigelow's sagebrush	Native
arca12	*Artemisia campestris*	field sagewort	Native

arca13	*Artemisia cana*	silver sagebrush	Native
ardr4	*Artemisia dracunculus*	wormwood	Native
arfi2	*Artemisia filifolia*	sand sagebrush	Native
arfr4	*Artemisia frigida*	fringed sagewort	Native
arlu	*Artemisia ludoviciana*	louisiana sagewort	Native
arno4	*Artemisia nova*	black sagebrush	Native
arpy2	*Artemisia pygmaea*	pygmy sagebrush	Native
arsp5	*Artemisia spinescens*	bud sagebrush	Native
artr2	*Artemisia tridentata*	big sagebrush	Native
ascr	*Asclepias cryptoceras*	pallid milkweed	Native
asla	*Asclepias labriformis*	utah milkweed	Native
asma10	*Asclepias macrosperma*	largeseed milkweed	Native
assp	*Asclepias speciosa*	showy milkweed	Native
assu2	*Asclepias subverticillata*	whorled milkweed	Native
asch2	*Aster chilensis*	pacific aster	Native
asam5	*Astragalus amphioxys*	crescent milkvetch	Native
asam6	*Astragalus ampullarius*	gumbo milkvetch	Native
asarp	*Astragalus argophyllus var. panguicensis*	silverleaf milkvetch	Native
asbi2	*Astragalus bisulcatus*	twogrooved milkvetch	Native
asca9	*Astragalus calycosus*	matted poison milkvetch	Native
asce	*Astragalus ceramicus*	painted milkvetch	Native
asci2	*Astragalus cibarius*	silky milkvetch	Native
asco12	*Astragalus convallarius*	timber milkvetch	Native
asde3	*Astragalus desperatus*	rimrock milkvetch	Native
asha2	*Astragalus hallii*	hall's milkvetch	Native
asle8	*Astragalus lentiginosus*	specklepod milkvetch	Native
aslo3	*Astragalus lonchocarpus*	rushy milkvetch	Native
asma5	*Astragalus malacoides*	kaiparowits milkvetch	Native
asmo4	*Astragalus moencoppensis*	moenkopi milkvetch	Native
asmot	*Astragalus mollissimus var. thompsoniae*		Native
asmu3	*Astragalus musiniensis*	ferron's milkvetch	Native
asnen	*Astragalus newberryi var. newberryi*		Native
asnum2	*Astragalus nuttallianus var. micranthiformis*	turkeypeas	Native
asprp2	*Astragalus praelongus var. praelongus*		Native
asse7	*Astragalus sesquiflorus*	sandstone milkvetch	Native
assu6	*Astragalus subcinereus*	silver's milkvetch	Native
asut	*Astragalus utahensis*	utah milkvetch	Native
aszi	*Astragalus zionis*	zion milkvetch	Native
atca2	*Atriplex canescens*	fourwing saltbush	Native
atco	*Atriplex confertifolia*	shadscale saltbush	Native
atco4	*Atriplex corrugata*	mat saltbush	Native

atpo2	*Atriplex powellii*	powell's saltweed	Native
atsa	*Atriplex saccaria*	sack saltbush	Native
basa	*Baccharis salicina*	great plains falsewillow	Native
basa3	*Balsamorhiza sagittata*	arrowleaf balsamroot	Native
beoc2	*Betula occidentalis*	water birch	Native
bosp	*Boerhavia spicata*	creeping spiderling	Native
bocu	*Bouteloua curtipendula*	sideoats grama	Native
boer4	*Bouteloua eriopoda*	black grama	Native
bogr2	*Bouteloua gracilis*	blue grama	Native
brca3	*Brickellia californica*	california brickellbush	Native
brmi	*Brickellia microphylla*	littleleaf brickellbush	Native
brob	*Brickellia oblongifolia*	mojave brickellbush	Native
brin2	*Bromus inermis*	smooth brome	Introduced
brru2	*Bromus rubens*	red brome	Introduced
brte	*Bromus tectorum*	cheatgrass	Introduced
caau8	*Calochortus aureus*	golden mariposa lily	Native
cafl	*Calochortus flexuosus*	winding mariposa lily	Native
canu3	*Calochortus nuttallii*	sego lily	Native
cala38	*Calylophus lavandulifolius*	lavenderleaf sundrops	Native
caboc	*Camissonia boothii ssp. condensata*		Native
caex10	*Camissonia exilis*	cottonwood springs suncup	Native
camu13	*Camissonia multijuga*	froststem suncup	Native
casc18	*Camissonia scapoidea*	barestem eveningprimrose	Native
caro2	*Campanula rotundifolia*	bluebell bellflower	Native
caau3	*Carex aurea*	golden sedge	Native
cado2	*Carex douglasii*	douglas' sedge	Native
cala30	*Carex lanuginosa*	woolly sedge	Native
cane2	*Carex nebrascensis*	nebraska sedge	Native
capr5	*Carex praegracilis*	clustered field sedge	Native
caro5	*Carex rossii*	ross' sedge	Native
caro6	*Carex rostrata*	beaked sedge	Native
casi2	*Carex simulata*	analogue sedge	Native
cast40	*Carex stenophylla*	narrowleaf sedge	Native
cach7	*Castilleja chromosa*	desert paintbrush	Native
caex6	*Castilleja exilis*	marsh paintbrush	Native
cali4	*Castilleja linariifolia*	wyoming indian paintbrush	Native
casc19	*Castilleja scabrida*	rough indian paintbrush	Native
caaq3	*Catabrosa aquatica*	water whorlgrass	Native
cacr11	*Caulanthus crassicaulis*	thickstem wild cabbage	Native
cein7	*Cercocarpus intricatus*	littleleaf mountain mahogany	Native
cemo2	*Cercocarpus montanus*	true mountain mahogany	Native
chdo	*Chaenactis douglasii*	douglas' dustymaiden	Native

chma	*Chaenactis macrantha*	bighead dustymaiden	Native
chst	*Chaenactis stevioides*	steve's dustymaiden	Native
cher2	*Chaetopappa ericoides*	rose heath	Native
chal7	*Chenopodium album*	lambsquarters	Introduced
chfr3	*Chenopodium fremontii*	fremont's goosefoot	Native
chgl3	*Chenopodium glaucum*	oakleaf goosefoot	Introduced
chle4	*Chenopodium leptophyllum*	narrowleaf goosefoot	Native
chte2	*Chorispora tenella*	crossflower	Introduced
chth5	*Chorizanthe thurberi*	red triangles	Native
chde2	*Chrysothamnus depressus*	longflower rabbitbrush	Native
chgr6	*Chrysothamnus greenei*	greene's rabbitbrush	Native
chli3	*Chrysothamnus linifolius*	spearleaf rabbitbrush	Native
chna2	*Chrysothamnus nauseosus*	rubber rabbitbrush	Native
chpa13	*Chrysothamnus parryi*	parry rabbitrush	Native
chva2	*Chrysothamnus vaseyi*	vasey's rabbitbrush	Native
chvi8	*Chrysothamnus viscidiflorus*	green rabbitbrush	Native
ciar3	*Cirsium arizonicum*	arizona thistle	Native
CINE	*Cirsium neomexicanum*	new mexico thistle	Native
civu	*Cirsium vulgare*	bull thistle	Introduced
ciwh	*Cirsium wheeleri*	wheeler's thistle	Native
clli2	*Clematis ligusticifolia*	western white clematis	Native
cllu2	*Cleome lutea*	yellow spiderflower	Native
clpa3	*Cleomella palmeriana*	rocky mountain stickweed	Native
cora	*Coleogyne ramosissima*	blackbrush	Native
coump	*Comandra umbellata ssp. pallida*	bastard toadflax	Native
coma2	*Conium maculatum*	poison hemlock	Introduced
coar4	*Convolvulus arvensis*	field bindweed	Introduced
coca5	*Conyza canadensis*	canadian horseweed	Native
copa9	*Cordylanthus parviflorus*	purple bird's-beak	Native
cowr2	*Cordylanthus wrightii*	wright's bird's beak	Native
coam8	*Corispermum americanum*	american bugseed	Native
covi9	*Coryphantha vivipara*	vivaparous coryphantha	Native
crac2	*Crepis acuminata*	longleaf hawksbeard	Native
crin4	*Crepis intermedia*	limestone hawksbeard	Native
crte4	*Croton texensis*	texas croton	Native
crba4	*Cryptantha bakeri*	baker's catseye	Native
crca13	*Cryptantha capitata*	capitate cryptantha	Native
crci3	*Cryptantha cinerea*	james' catseye	Native
crci2	*Cryptantha circumscissa*	cushion catseye	Native
crco12	*Cryptantha confertiflora*	basin yellow catseye	Native
crcr3	*Cryptantha crassisepala*	thicksepal catseye	Native
crde	*Cryptantha decipiens*	gravelbar cryptantha	Native

crfe3	*Cryptantha fendleri*	sanddune catseye	Native
crfl5	*Cryptantha flava*	brenda's yellow catseye	Native
crfl6	*Cryptantha flavoculata*	roughseed catseye	Native
crfu	*Cryptantha fulvocanescens*	tawny catseye	Native
crgr3	*Cryptantha gracilis*	narrowstem catseye	Native
crhu2	*Cryptantha humilis*	roundspike catseye	Native
crmi	*Cryptantha micrantha*	redroot cryptantha	Native
crne2	*Cryptantha nevadensis*	nevada cryptantha	Native
crpt	*Cryptantha pterocarya*	wingnut catseye	Native
cyac	*Cymopterus acaulis*	plains springparsley	Native
cymu2	*Cymopterus multinervatus*	purplenerve springparsley	Native
cyne	*Cymopterus newberryi*	sweetroot springparsley	Native
cypu	*Cymopterus purpurascens*	widewing springparsley	Native
cypu2	*Cymopterus purpureus*	purple springparsley	Native
cyda	*Cynodon dactylon*	bermudagrass	Introduced
dase3	*Dalea searlsiae*	searls' prairie clover	Native
dawr2	*Datura wrightii*	sacred thornapple	Native
dean	*Delphinium andersonii*	anderson's larkspur	Native
denu2	*Delphinium nuttallianum*	nuttal's larkspur	Native
depi	*Descurainia pinnata*	western tansymustard	Native
deso2	*Descurainia sophia*	herb sophia	Introduced
disp	*Distichlis spicata*	inland saltgrass	Native
diwi6	*Dithyrea wislizeni*	touristplant	Native
drcu	*Draba cuneifolia*	wedgeleaf whitlowgrass	Native
drpa2	*Dracocephalum parviflorum*	american dragonhead	Native
ecen	*Echinocereus engelmannii*	engelmann's hedgehog cactus	Native
ectr	*Echinocereus triglochidiatus*	kingcup cactus	Native
elan	*Elaeagnus angustifolia*	russian olive	Introduced
elpa3	*Eleocharis palustris*	common spikerush	Native
elro2	*Eleocharis rostellata*	beaked spikerush	Native
elca4	*Elymus canadensis*	canada wildrye	Native
elel5	*Elymus elymoides*	bottlebrush squirreltail	Native
elhi6	*Elymus hispidus*	intermediate wheatgrass	Introduced
ella3	*Elymus lanceolatus*	streambank wheatgrass	Native
elsm3	*Elymus smithii*	western wheatgrass	Native
elsp3	*Elymus spicatus*	bluebunch wheatgrass	Native
eltr7	*Elymus trachycaulus*	slender wheatgrass	Native
eltr3	*Elymus triticoides*	beardless, creeping wildrye	Native
epto	*Ephedra torreyana*	torrey's jointfir	Native
epvi	*Ephedra viridis*	mormon tea	Native
epan2	*Epilobium angustifolium*	fireweed	Native
epci	*Epilobium ciliatum*	hairy willowherb	Native

epgi	*Epipactis gigantea*	giant helleborine	Native
eqar	*Equisetum arvense*	field horsetail	Native
eqhy	*Equisetum hyemale*	scouringrush horsetail	Native
eqla	*Equisetum laevigatum*	smooth horsetail	Native
erdi2	*Eriastrum diffusum*	miniature woolstar	Native
ersp3	*Eriastrum sparsiflorum*	great basin woollystar	Native
erab3	*Erigeron abajoensis*	abajo fleabane	Native
erbe2	*Erigeron bellidiastrum*	western daisy fleabane	Native
erdi4	*Erigeron divergens*	spreading fleabane	Native
erea	*Erigeron eatonii*	eaton's fleabane	Native
erfl	*Erigeron flagellaris*	trailing fleabane	Native
erpuc	*Erigeron pumilus ssp. concinnoides*	low fleabane	Native
erre7	*Erigeron religiosus*	clear creek fleabane	Native
erut	*Erigeron utahensis*	utah fleabane	Native
eral4	*Eriogonum alatum*	winged buckwheat	Native
erce2	*Eriogonum cernuum*	nodding buckwheat	Native
erco14	*Eriogonum corymbosum*	crispleaf buckwheat	Native
erda	*Eriogonum darrovii*	darrow's buckwheat	Native
erfl12	*Eriogonum flexum*	bent buckwheat	Native
ergo	*Eriogonum gordonii*	gordon's buckwheat	Native
erin4	*Eriogonum inflatum*	native american pipeweed	Native
erle9	*Eriogonum leptocladon*	sand buckwheat	Native
ermi4	*Eriogonum microthecum*	slender buckwheat	Native
ernu4	*Eriogonum nummulare*	money buckwheat	Native
erov	*Eriogonum ovalifolium*	cushion buckwheat	Native
erpa11	*Eriogonum palmerianum*	palmer's buckwheat	Native
erpa13	*Eriogonum panguicense*	panguitch buckwheat	Native
erra3	*Eriogonum racemosum*	redroot buckwheat	Native
ersa14	*Eriogonum salsuginosum*	salty buckwheat	Native
ershl	*Eriogonum shockleyi var. longilobum*	shockley`s buckwheat	Native
ersu5	*Eriogonum subreniforme*	kidneyshape buckwheat	Native
erum	*Eriogonum umbellatum*	sulphur wildbuckwheat	Native
erwe	*Eriogonum wetherillii*	wetherill's buckwheat	Native
erpu8	*Erioneuron pulchellum*	low woollygrass	Native
erci6	*Erodium cicutarium*	redstem stork's bill	Introduced
eras2	*Erysimum asperum*	western wallflower	Native
eubr	*Euphorbia brachycera*	horned spurge	Native
eufe2	*Euphorbia fendleri*	fendler spurge	Native
eugl3	*Euphorbia glyptosperma*	ridgeseed spurge	Native
eupa6	*Euphorbia parryi*	dune spurge	Native
evnu	*Evolvulus nuttallianus*	shaggy dwarf morningglory	Native
fapa	*Fallugia paradoxa*	apacheplume	Native

fear3	*Festuca arundinacea*	tall fescue	Introduced
fepr	*Festuca pratensis*	meadow fescue	Introduced
fran2	*Fraxinus anomala*	singleleaf ash	Native
frat	*Fritillaria atropurpurea*	spotted missionbells	Native
gapa	*Gaillardia parryi*	parry's blanketflower	Native
gapi	*Gaillardia pinnatifida*	red dome blanketflower	Native
galiu	*Galium*	bedstraw	Native
gara	*Gayophytum racemosum*	blackfoot groundsmoke	Native
gara2	*Gayophytum ramosissimum*	pinyon groundsmoke	Native
genti	*Gentiana*	gentian	Native
gicof	*Gilia congesta var. frutescens*		Native
gihu	*Gilia hutchinsifolia*	desert pale gilia	Native
giin2	*Gilia inconspicua*	shy gilia	Native
gile3	*Gilia leptomeria*	sand gilia	Native
gilo2	*Gilia longiflora*	flaxflowered ipomopsis	Native
gipo4	*Gilia polycladon*	spreading gilia	Native
gisu	*Gilia subnuda*	coral gilia	Native
glst	*Glyceria striata*	fowl mannagrass	Native
glle3	*Glycyrrhiza lepidota*	american licorice	Native
gnaph	*Gnaphalium*	cudweed	Native and Introduced
grsp	*Grayia spinosa*	spiny hopsage	Native
grsq	*Grindelia squarrosa*	curlycup gumweed	Native
gumi	*Gutierrezia microcephala*	threadleaf snakeweed	Native
gusa2	*Gutierrezia sarothrae*	broom snakeweed	Native
hasp5	*Habenaria sparsiflora*	watson bog orchid	Native
haac	*Haplopappus acaulis*	goldenweed	Native
haar2	*Haplopappus armerioides*	thrifty goldenweed	Native
hadr2	*Haplopappus drummondii*	drummond`s goldenbush	Native
hasp3	*Haplopappus spinulosus*	spiny goldenweed	Native
hedr	*Hedeoma drummondii*	drummond's falsepennyroyal	Native
hebo	*Hedysarum boreale*	northern sweetvetch	Native
hemi2	*Helianthella microcephala*	purpledisk helianthella	Native
heun	*Helianthella uniflora*	oneflower helianthella	Native
hean3	*Helianthus annuus*	common sunflower	Native
hean4	*Helianthus anomalus*	western sunflower	Native
hepe	*Helianthus petiolaris*	prairie sunflower	Native
hevi4	*Heterotheca villosa*	hairy goldenaster	Native
hija	*Hilaria jamesii*	galleta grass	Native
hoju	*Hordeum jubatum*	foxtail barley	Native
hupr	*Hutchinsia procumbens*	prostrate hutchinsia	Native
hyfi	*Hymenopappus filifolius*	fineleaf hymenopappus	Native

hyac4	*Hymenoxys acaulis*	stemless woolybase	Native
hyco2	*Hymenoxys cooperi*	cooper's rubberweed	Native
hyri	*Hymenoxys richardsonii*	pingue hymenoxys	Native
ipag	*Ipomopsis aggregata*	skyrocket gilia	Native
irmi	*Iris missouriensis*	rocky mountain iris	Native
jute	*Jjuncus tenuis*	poverty rush	Native
juar2	*Juncus arcticus*	arctic rush	Native
jubu	*Juncus bufonius*	toad rush	Native
juen	*Juncus ensifolius*	swordleaf rush	Native
julo	*Juncus longistylis*	longstyle rush	Native
juto	*Juncus torreyi*	torrey's rush	Native
juos	*Juniperus osteosperma*	utah juniper	Native
jusc2	*Juniperus scopulorum*	rocky mountain juniper	Native
kopr80	*Kochia prostrata*	prostrate summercypress	Introduced
kosc	*Kochia scoparia*	common kochia	Introduced
koma	*Koeleria macrantha*	prairie junegrass	Native
krla2	*Krascheninnikovia lanata*	winterfat	Native
lase	*Lactuca serriola*	prickly lettuce	Introduced
lase3	*Langloisia setosissima*	great basin langloisia	Native
laoc3	*Lappula occidentalis*	flatspine stickseed	Native
lala3	*Lathyrus lanszwertii*	thickleaf peavine	Native
lede	*Lepidium densiflorum*	common pepperweed	Native
lela	*Lepidium lasiocarpum*	shaggyfruit pepperweed	Native
lemo2	*Lepidium montanum*	mountain pepperweed	Native
lepe2	*Lepidium perfoliatum*	clasping pepperweed	Introduced
lepu	*Leptodactylon pungens*	granite pricklygilia	Native
lewa	*Leptodactylon watsonii*	watson's pricklygilia	Native
lear4	*Lesquerella arizonica*	arizona bladderpod	Native
lein3	*Lesquerella intermedia*	mid bladderpod	Native
lelu	*Lesquerella ludoviciana*	foothill bladderpod	Native
lere3	*Lesquerella rectipes*	straight bladderpod	Native
lemo4	*Leucocrinum montanum*	common starlily	Native
lipo	*Ligusticum porteri*	porter's licoriceroot	Native
liar3	*Linum aristatum*	bristle flax	Native
lipe2	*Linum perenne*	blue flax	Native
liin2	*Lithospermum incisum*	narrowleaf gromwell	Native
lofo	*Lomatium foeniculaceum*	desert biscuitroot	Native
lone	*Lomatium nevadense*	nevada biscuitroot	Native
lopa	*Lomatium parryi*	utah desertparsley	Native
loin5	*Lonicera involucrata*	twinberry honeysuckle	Native
lout3	*Lotus utahensis*	utah bird's-foot trefoil	Native
luar3	*Lupinus argenteus*	silvery lupine	Native

lubr2	*Lupinus brevicaulis*	shortstem lupine	Native
luca	*Lupinus caudatus*	tailcup lupine	Native
lupu	*Lupinus pusillus*	rusty lupine	Native
luse4	*Lupinus sericeus*	silky lupine	Native
lyan	*Lycium andersonii*	water jacket	Native
lypa	*Lycium pallidum*	pale wolfberry	Native
lyph	*Lycurus phleoides*	common wolfstail	Native
lygr	*Lygodesmia grandiflora*	largeflower skeletonplant	Native
maca2	*Machaeranthera canescens*	hoary aster	Native
magr2	*Machaeranthera grindelioides*	rayless aster	Native
mata2	*Machaeranthera tanacetifolia*	tanseyleaf aster	Native
mafr3	*Mahonia fremontii*	fremont's mahonia	Native
mare11	*Mahonia repens*	oregongrape	Native
magl3	*Malacothrix glabrata*	smooth desertdandelion	Native
maso	*Malacothrix sonchoides*	sowthistle desertdandelion	Native
maaf	*Malcolmia africana*	african mustard	Introduced
mane	*Malva neglecta*	common mallow	Introduced
mavu	*Marrubium vulgare*	horehound	Introduced
melu	*Medicago lupulina*	black medick	Introduced
mesa	*Medicago sativa*	alfalfa	Introduced
meof	*Melilotus officinalis*	yellow sweetclover	Introduced
mear4	*Mentha arvensis*	wild mint	Native
meal6	*Mentzelia albicaulis*	whitestem blazingstar	Native
mema5	*Mentzelia marginata*	colorado blazingstar	Native
memu3	*Mentzelia multiflora*	manyflowered mentzelia	Native
migr	*Microsteris gracilis*	slender phlox	Native
mial5	*Mirabilis alipes*	winged four o'clock	Native
mili3	*Mirabilis linearis*	narrowleaf four o'clock	Native
mimu	*Mirabilis multiflora*	colorado four o'clock	Native
miox	*Mirabilis oxybaphoides*	smooth spreading four o'clock	Native
mosq	*Monroa squarrosa*	false buffalograss	Native
muan	*Muhlenbergia andina*	foxtail muhly	Native
muas	*Muhlenbergia asperifolia*	alkali muhly	Native
mupo2	*Muhlenbergia porteri*	bush muhly	Native
mupu2	*Muhlenbergia pungens*	sandhill muhly	Native
nare	*Nama retrorsum*	betatakin fiddleleaf	Native
niat	*Nicotiana attenuata*	coyote tobacco	Native
oeal	*Oenothera albicaulis*	whitest eveningprimrose	Native
oeca10	*Oenothera caespitosa*	tufted evening-primrose	Native
oeel	*Oenothera elata*	hooker's eveningprimrose	Native
oeho2	*Oenothera howardii*	howard's eveningprimrose	Native
oelo	*Oenothera longissima*	longstem eveningprimrose	Native

oepa	*Oenothera pallida*	pale eveningprimrose	Native
opba2	*Opuntia basilaris*	beavertail pricklypear	Native
oper	*Opuntia erinacea*	grizzlybear pricklypear	Native
oppo	*Opuntia polyacantha*	plains pricklypear	Native
opwh	*Opuntia whipplei*	whipple cholla	Native
orfa	*Orobanche fasciculata*	clustered broomrape	Native
orlu	*Orobanche ludoviciana*	louisiana broomrape	Native
oxor2	*Oxytropis oreophila*	mountain oxytrope	Native
pece	*Pedicularis centranthera*	dwarf lousewort	Native
pepa41	*Pediomelum pariense*	paria river indian breadroot	Native
pegl	*Pellaea glabella*	dwarf cliffbrake	Native
peam	*Penstemon ambiguus*	gilia beardtongue	Native
peanv	*Penstemon angustifolius var. venosus*		Native
peat4	*Penstemon atwoodii*	kaiparowits beardtongue	Native
peba2	*Penstemon barbatus*	beardlip penstemon	Native
peca4	*Penstemon caespitosus*	mat penstemon	Native
peco5	*Penstemon comarrhenus*	dusty beardtongue	Native
peea	*Penstemon eatonii*	eaton's penstemon	Native
pela15	*Penstemon laevis*	southwestern beardtongue	Native
pele7	*Penstemon leiophyllus*	smoothleaf beardtongue	Native
peli2	*Penstemon linarioides*	toadflax penstemon	Native
peop	*Penstemon ophianthus*	coiled anther penstemon	Native
pepa6	*penstemon pachyphyllus*	thickleaf beardtongue	Native
pepa8	*penstemon palmeri*	palmer's penstemon	Native
pero10	*Penstemon rostriflorus*	bridge penstemon	Native
pery	*Penstemon rydbergii*	rydberg's penstemon	Native
peth2	*Penstemon thompsoniae*	thompson's beardtongue	Native
peut	*Penstemon utahensis*	utah penstemon	Native
pera4	*Peraphyllum ramosissimum*	squaw apple	Native
pepu7	*Petradoria pumila*	grassy rockgoldenrod	Native
phcr	*Phacelia crenulata*	cleftleaf wildheliotrope	Native
phde	*Phacelia demissa*	intermountain scorpionweed	Native
phhe2	*Phacelia heterophylla*	varileaf phacelia	Native
phin	*Phacelia integrifolia*	gypsum scorpionweed	Native
phiv	*Phacelia ivesiana*	ives' phacelia	Native
phma10	*Phacelia mammillarensis*	nipple beach phacelia	Native
phau3	*Phlox austromontana*	desert phlox	Native
phho	*Phlox hoodii*	spiny phlox	Native
phlo2	*Phlox longifolia*	longleaf phlox	Native
phju	*Phoradendron juniperinum*	juniper mistletoe	Native
phau7	*Phragmites australis*	common reed	Native
phac4	*Physaria acutifolia*	sharpleaf twinpod	Native

phch2	*Physaria chambersii*	chambers' twinpod	Native
phne5	*Physaria newberryi*	newberry's twinpod	Native
pied	*Pinus edulis*	twoneedle pinyon	Native
pifl2	*Pinus flexilis*	limber pine	Native
pipo	*Pinus ponderosa*	ponderosa pine	Native
plma2	*Plantago major*	common plantain	Native
plpa2	*Plantago patagonica*	woolly plantain	Native
pofe	*Poa fendleriana*	muttongrass	Native
pogl	*Poa glauca*	glaucous bluegrass	Native
popr	*Poa pratensis*	kentucky bluegrass	Introduced
pose	*Poa secunda*	sandberg bluegrass	Native
poin3	*Poliomintha incana*	hoary rosemarymint	Native
posu2	*Polygala subspinosa*	spiny polygala	Native
poav	*Polygonum aviculare*	prostrate knotweed	Introduced
podo4	*Polygonum douglasii*	douglas' knotweed	Native
pora3	*Polygonum ramosissimum*	bushy knotweed	Native
pomo5	*Polypogon monspeliensis*	annual rabbitsfoot grass	Introduced
poan3	*Populus angustifolia*	narrowleaf cottonwood	Native
pofr2	*Populus fremontii*	fremont's cottonwood	Native
potr5	*Populus tremuloides*	quaking aspen	Native
poac5	*Populus x acuminata*	lanceleaf cottonwood	Native
poan5	*Potentilla anserina*	silverweed cinquefoil	Native
pogr9	*Potentilla gracilis*	northwest cinquefoil	Native
prex	*Prenanthella exigua*	brightwhite	Native
prvi	*Prunus virginiana*	common chokecherry	Native
psme	*Pseudotsuga menziesii*	douglas fir	Native
pssp	*Psilostrophe sparsiflora*	greenstem paperflower	Native
psju2	*Psoralidium junceum*	rush lemonweed	Native
psla3	*Psoralidium lanceolatum*	lemon scurfpea	Native
psfr	*Psorothamnus fremontii*	fremont's dalea	Native
pudi	*Puccinellia distans*	weeping alkaligrass	Native
punu2	*Puccinellia nuttalliana*	nuttall's alkaligrass	Native
pume	*Purshia mexicana*	cliffrose	Native
putr2	*Purshia tridentata*	antelope bitterbrush	Native
quea	*Quercus eastwoodiae*	eastwood's oak	Native
quga	*Quercus gambelii*	gambel's oak	Native
quha3	*Quercus havardii*	havard's oak	Native
qutu2	*Quercus turbinella*	shrub live oak	Native
racy	*Ranunculus cymbalaria*	alkali buttercup	Native
rasc3	*Ranunculus sceleratus*	celeryleaf buttercup	Native
rate	*Ranunculus testiculatus*	curveseed butterwort	Introduced
rhar4	*Rhus aromatica*	fragrant sumac	Native

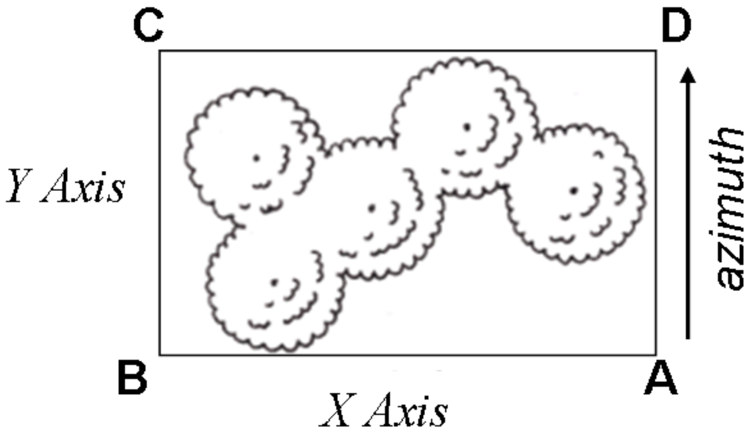
rhgl	*Rhus glabra*	smooth sumac	Native
ribes	*Ribes*	currant	Native
rowo	*Rosa woodsii*	woods' rose	Native
rucr	*Rumex crispus*	curly dock	Introduced
ruhy	*Rumex hymenosepalus*	canaigre dock	Native
sabe2	*Salix bebbiana*	bebb willow	Native
saex	*Salix exigua*	sandbar willow	Native
saluc	*Salix lucida ssp. caudata*		Native
saib	*Salsola iberica*	russian thistle	Introduced
sapa8	*Salsola paulsenii*	barbwire russian thistle	Introduced
sami3	*Sanguisorba minor*	small burnet	Introduced
save4	*Sarcobatus vermiculatus*	greasewood	Native
scar	*Schismus arabicus*	arabian schismus	Introduced
scli	*Schoenocrambe linifolia*	flaxleaf plainsmustard	Native
scac	*Scirpus acutus*	hardstem bulrush	Native
scpu3	*Scirpus pungens*	common threesquare bulrush	Native
scva	*Scirpus validus*	softstem bulrush	Native
scwh	*Sclerocactus whipplei*	whipples fishhook	Native
sela	*Sedum lanceolatum*	spearleaf stonecrop	Native
sedol	*Senecio douglasii var. longilobus*	threadleaf ragwort	Native
sehy2	*Senecio hydrophilus*	water groundsel	Native
semu3	*Senecio multilobatus*	lobeleaf groundsel	Native
sesp3	*Senecio spartioides*	broom groundsel	Native
sest3	*Senecio streptanthifolius*	cleftleaf groundsel	Native
shar	*Shepherdia argentea*	silver buffaloberry	Native
shro	*Shepherdia rotundifolia*	roundleaf buffaloberry	Native
sial2	*Sisymbrium altissimum*	tall tumblemustard	Introduced
side4	*Sisyrinchium demissum*	dwarf blueeyed grass	Native
siid	*Sisyrinchium idahoense*	idaho blueeyed grass	Native
smst	*Smilacina stellata*	stellate smilacinia	Native
sotr	*Solanum triflorum*	cutleaf nightshade	Native
sosp5	*Solidago sparsiflora*	fewflowered goldenrod	Native
soas	*Sonchus asper*	spiny sowthistle	Introduced
spgr	*Spartina gracilis*	alkali cordgrass	Native
spco	*Sphaeralcea coccinea*	scarlet globemallow	Native
spgr2	*Sphaeralcea grossulariifolia*	gooseberryleaf globemallow	Native
sppa2	*Sphaeralcea parvifolia*	smallflower globemallow	Native
spai	*Sporobolus airoides*	alkali sacaton	Native
spco4	*Sporobolus contractus*	spike dropseed	Native
spcr	*Sporobolus cryptandrus*	sand dropseed	Native
spfl2	*Sporobolus flexuosus*	mesa dropseed	Native
stpi	*Stanleya pinnata*	desert princesplume	Native

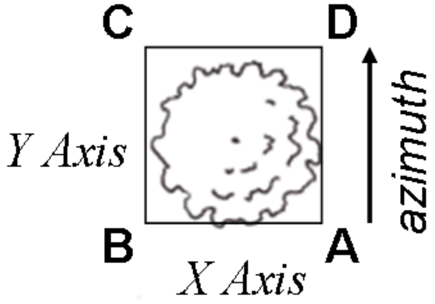
stja3	*Stellaria jamesiana*	tuber starwort	Native
stex	*Stephanomeria exigua*	small wirelettuce	Native
stpa3	*Stephanomeria parryi*	parry's wirelettuce	Native
stsp6	*Stephanomeria spinosa*	thorn skeletonweed	Native
sttet	*Stephanomeria tenuifolia var. tenuifolia*		Native
star2	*Stipa arida*	morman needlegrass	Native
stco4	*Stipa comata*	needle and thread grass	Native
stco5	*Stipa coronata*	crested needlegrass	Native
sthy6	*Stipa hymenoides*	indian ricegrass	Native
stle4	*Stipa lettermanii*	letterman needlegrass	Native
stsp3	*Stipa speciosa*	desert needlegrass	Native
stlo4	*Streptanthella longirostris*	longbeak streptanthella	Native
stco6	*Streptanthus cordatus*	heartleaf twistflower	Native
suto	*Suaeda torreyana*	bush seepweed	Native
swal3	*Swertia albomarginata*	white-margined frasera	Native
sylo	*Symphoricarpos longiflorus*	desert snowberry	Native
syoru	*Symphoricarpos oreophilus var. utahensis*	utah snowberry	Native
tabr	*Talinum brevifolium*	pygmy fameflower	Native
tara	*Tamarix ramosissima*	saltcedar	Introduced
taof	*Taraxacum officinale*	common dandelion	Introduced
teca2	*Tetradymia canescens*	spineless horsebrush	Native
thali2	*Thalictrum*	meadowrue	Native
thsu	*Thelesperma subnudum*	sand fringedpod	Native
thin	*Thelypodium integrifolium*	entireleaved thelypody	Native
thmo6	*Thermopsis montana*	mountain goldenbanner	Native
thmo5	*Thlaspi montanum*	alpine pennycress	Native
toan	*Townsendia annua*	annual townsend daisy	Native
toin	*Townsendia incana*	hoary townsendia	Native
tomi2	*Townsendia minima*	least townsendia	Native
troc	*Tradescantia occidentalis*	prairie spiderwort	Native
trdu	*Tragopogon dubius*	yellow salsify	Introduced
trre3	*Trifolium repens*	white clover	Introduced
trmi6	*Tripterocalyx micranthus*	smallflower sandverbena	Native
tydo	*Typha domingensis*	southern cattail	Native
tyla	*Typha latifolia*	broadleaf cattail	Native
vast3	*Vanclevea stylosa*	pillar false gumweed	Native
vebr	*Verbena bracteata*	bigbract verbena	Native
vean2	*Veronica anagallis-aquatica*	water speedwell	Native
viam	*Vicia americana*	american vetch	Native
vigui	*Viguiera*	goldeneye	Native
viloa	*Viguiera longifolia var. annua*	longleaf false goldeneye	Native
viso2	*Viguiera soliceps*	tropical false goldeneye	Native

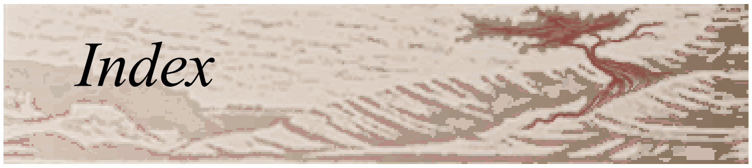
vine	*Viola nephrophylla*	bog violet	Native
vuoc	*Vulpia octoflora*	sixweeks fescue	Native
wysc	*Wyethia scabra*	badlands wyethia	Native
xast	*Xanthium strumarium*	rough cockleburr	Native
xyco3	*Xylorhiza confertifolia*	henrieville woodyaster	Native
xycr2	*Xylorhiza cronquistii*	cronquist's woodyaster	Native
xytoi	*Xylorhiza tortifolia var. imberbis*		Native
yuan2	*Yucca angustissima*	narrowleaf yucca	Native
yuba	*Yucca baccata*	banana yucca	Native
yuba2	*Yucca baileyi*	navajo yucca	Native
yuha	*Yucca harrimaniae*	spanish bayonet	Native
yuka2	*Yucca kanabensis*	Kanab yucca	Native
zipa2	*Zigadenus paniculatus*	foothill deathcamas	Native

Appendix I

Field measurements of tamarisk were collected using the new scale-dependent plot that is adjustable to the dimensions of an individual tree or stand of interest. The area of each scale-dependent plot varies in size, having different lengths and widths depending on the size and basal cover of the tree or stand being measured. When establishing a scale-dependent plot, the size and area of the frame is minimized to encompass no more then the area occupied by specie(s) interest in its entirety.







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