Maintaining and establishing culturally important plants after landscape scale disturbance
by Marcus Kirk Denny

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Land Resources and Environmental Sciences
Montana State University

Abstract:
Indigenous plant species have been culturally fundamental to Native Americans for centuries. These indigenous species are becoming recognized as potentially crucial elements to indigenous plant community structure, function, and invasion resistence. Invasions by nonindigenous plant species may displace many indigenous plants. Additionally, long-term herbicide use has potentially negative impacts on indigenous forb and shrub populations, and many are culturally important to Native Americans. Research on herbicide impacts to non-target plant species is limited. The purpose of this research was to quantify indigenous species response to hand removal of an invader, sulfur cinquefoil, and five selective herbicides applied to noninvaded rangeland at three rates each. Completely randomized sulfur cinquefoil treatments were (removed and nonremoved) replicated five times at two sites. Herbicide treatments were replicated three times at each of two sites in a randomized-complete-block design and included: commercial clopyralid + 2,4-D, 2,4-D amine, metsulfuron, picloram, and clopyralid. Species canopy cover was recorded at the four sites in 1999. hi 2000, canopy cover, biomass and density were recorded in each plot. Data were analyzed using ANOVA. Indigenous forb canopy cover and richness depended on hand removal of sulfur cinquefoil. Species canopy cover, density, and biomass depended upon herbicide and rate of application. Forb canopy cover and density decreased when treated with picloram while perennial grass cover and biomass increased following some herbicide treatments. Also, I tested two shapes of 9 m2 seed source islands (rectangle and square) as a mechanism for restoring three cultural species {Artemisia ludoviciana, Echinacea angustifolia, and PendiomeJum esculentum) to a grassland reclaimed after strip mining. Species were seeded in two shapes each replicated three times at two sites in a randomized-complete-block design. Island plant density depended upon species and year following seeding. A. ludoviciana and E. angustifolia established similarly while P. esculentum established at low densities. E. angustifolia produced the highest densities in 1999 thinning in 2000 and 2002. New occurrences across the landscape were not evident until 2002 with E. angustifolia establishing 138 flowered adults throughout the landscape of one site. This research suggests land managers must first consider all species in the understory before applying herbicides. Some herbicides and rates are more applicable for maintaining populations of indigenous forbs helping to increase a communities resistance to reinvasion. Thus, for disturbed communities low in indigenous species diversity, managers may implement low cost seed source islands as a restoration mechanism for desirable species. Invasive species management is critical to sustaining land productivity and land managers must improve their understanding of plant community dynamics and refine management technology in order to meet future land use objectives.

MAINTAINING AND ESTABLISHING CULTURALLY IMPORTANT PLANTS AFTER LANDSCAPE SCALE DISTURBANCE

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Marcus Kirk Denny

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APPROVAL

of a thesis submitted by

Marcus Kirk Denny

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

Dr. Roger L. Sheley Roger L. Sheley April 18, 2003
(Signature) (Date)

Approved for the Department of Land Resources and Environmental Sciences

Dr. Jeffery S. Jacobsen Jeffery S. Jacobsen 4/18/03
(Signature) (Date)

Approved for the College of Graduate Studies

Dr. Bruce R. McLeod Bruce R. McLeod 4-21-03
(Signature) (Date)

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ABSTRACT

Indigenous plant species have been culturally fundamental to Native Americans for centuries. These indigenous species are becoming recognized as potentially crucial elements to indigenous plant community structure, function, and invasion resistence. Invasions by nonindigenous plant species may displace many indigenous plants. Additionally, long-term herbicide use has potentially negative impacts on indigenous forb and shrub populations, and many are culturally important to Native Americans. Research on herbicide impacts to non-target plant species is limited. The purpose of this research was to quantify indigenous species response to hand removal of an invader, sulfur cinquefoil, and five selective herbicides applied to noninvaded rangeland at three rates each. Completely randomized sulfur cinquefoil treatments were (removed and nonremoved) replicated five times at two sites. Herbicide treatments were replicated three times at each of two sites in a randomized-complete-block design and included: commercial clopyralid + 2,4-D, 2,4-D amine, metsulfuron, picloram, and clopyralid. Species canopy cover was recorded at the four sites in 1999. In 2000, canopy cover, biomass and density were recorded in each plot. Data were analyzed using ANOVA. Indigenous forb canopy cover and richness depended on hand removal of sulfur cinquefoil. Species canopy cover, density, and biomass depended upon herbicide and rate of application. Forb canopy cover and density decreased when treated with picloram while perennial grass cover and biomass increased following some herbicide treatments. Also, I tested two shapes of 9 m^2 seed source islands (rectangle and square) as a mechanism for restoring three cultural species (*Artemisia ludoviciana*, *Echinacea angustifolia*, and *Pendiomelum esculentum*) to a grassland reclaimed after strip mining. Species were seeded in two shapes each replicated three times at two sites in a randomized-complete-block design. Island plant density depended upon species and year following seeding. *A. ludoviciana* and *E. angustifolia* established similarly while *P. esculentum* established at low densities. *E. angustifolia* produced the highest densities in 1999 thinning in 2000 and 2002. New occurrences across the landscape were not evident until 2002 with *E. angustifolia* establishing 138 flowered adults throughout the landscape of one site. This research suggests land managers must first consider all species in the understory before applying herbicides. Some herbicides and rates are more applicable for maintaining populations of indigenous forbs helping to increase a communities resistance to reinvasion. Thus, for disturbed communities low in indigenous species diversity, managers may implement low cost seed source islands as a restoration mechanism for desirable species. Invasive species management is critical to sustaining land productivity and land managers must improve their understanding of plant community dynamics and refine management technology in order to meet future land use objectives.

CHAPTER 1

INTRODUCTION

For centuries, tribes of the northern Great Plains relied entirely on the natural surroundings for nourishment, shelter, inner strength and medicine. The prehistoric understanding of the role of native plants and animals was integral to co-existence and survival. In Northern Cheyenne oral teaching, "Every animal and plant has its purpose" (Tallbull 1980). Traditional Cheyenne teachings are passed down from generation to generation as an oral database which preserves lessons of the past for improved decisions today and in the future. As life skills expanded, specialization occurred and tribal members became historians, medicine men and botanists. Man's place in the indigenous circle of life was to work with nature to optimize resource use; in turn, improving life. In the absence of non-indigenous invaders, this life cycle and environment were dynamic, yet sustainable. In the Cheyenne way of knowing, these teachings, even if not written, evolved as a logical process that improved understanding of the world in which plains natives lived. Even a world apart, lifestyles and land management were grounded in what would later be known as ecology.

The nonindigenous anthropogenic invasions of the 1800's had remarkable and near devastating impacts on the indigenous people of the plains and their world. This mass human migration served as a vector for other species' dispersal that played a major role in landscape change (Vitousek 1986). The settling of the Northwest introduced many nonindigenous plant species and today some have dispersed and invaded millions of hectares in the region (Sheley and Petroff 1999). Nonindigenous invasion, by genera such as *Centaurea* spp. and *Euphorbia* spp., is associated with reduction in biomass, density and basal area of indigenous plants (Tyser and Key 1988, Belcher and Wilson 1989, Kedzie-Webb et al. 2001). Many indigenous grasses and forbs may have been displaced on the Northern Cheyenne Reservation with introduction of spotted knapweed (*Centaurea maculosa* Lam.), Russian knapweed (*Centaurea repens* L.), leafy spurge (*Euphorbia esula* L.), whitetop (*Cardaria draba* (L.) Desv.), Dalmation toadflax (*Linaria dalmatica* (L.) Miller), and sulfur cinquefoil (*Potentilla recta* L.). The propagation and dispersal of these species is probable without aggressive preventative measures. The potential for displacement of indigenous grass and forb species, many of which are sacred, by nonindigenous invasive plants represents a significant cultural as well as ecological problem. This problem makes preservation, restoration, and propagation of native plants paramount to sustaining traditional Northern Cheyenne culture and possibly all people inhabiting this region.

Even today, tribal people practice traditional ways which include uses of indigenous plant species for food, ritual, and medicine. In addition, cultural plants may be fundamental components of indigenous plant communities and their functional diversity. Historically, there is scientific literature that supports the theory that more diverse native communities will be less invasible (Elton 1958, Rejmanek 1996, Crawley et al. 1999, Levine and D'Antonio 1999, Pokorny 2002). Conversely, a number of papers have recently challenged this idea by showing greater numbers of nonindigenous plant species in areas with high native diversity (Robinson et al. 1995, Stohlgren et al. 1999).

Maintenance or creation of invasion resistant assemblages may become a fundamental objective for land managers; especially, if preventing establishment of nonindigenous invaders is the goal (Jacobs and Sheley 1999, Pokorny 2002). Thus, understanding the conditions, and processes that allow invasion will be crucial to the development of effective management.

Since the 1950's, herbicidal control of many indigenous forb and shrub species was often implemented to increase livestock forage (Evans et al. 1979). These species were of little interest because they had no obvious agronomic value for grazing livestock. The removal of indigenous groups of plant species resulted in immediate forage release similar to the control of nonindigenous invaders (Hubbard 1975, Chicoine 1984, Roche 1988, Davis 1990, Sheley and Jacobs 1997). Increased forage production provides tangible short-term economic gains; however, invasibility studies are now questioning the value of removing indigenous constituents from the plant community for the sole purpose of increasing forage production (Pokorny 2002).

The quandary for land managers dealing with invaded communities begins when infestations expand beyond the patch scale. Once an invader, such as knapweed or sulfur cinquefoil, becomes established, repeated hand removal or properly timed spot treatment with herbicide can provide favorable control and prevention at the patch scale (Lacey et al. 1997). However, the labor involved often restricts hand removal and spot applications from being economically feasible as the scale of the infestation enlarges to the drainage or watershed scale. Once nonindigenous invaders dominate at the landscape scale, management decisions are more complex when factored with the economics and environmental concerns associated with these weeds and their control. Large scale control is often attempted by cultural controls such as mowing, tillage, biocontrol, broadcasting selective herbicides or an integration of these treatments (Sheley and Petroff 1999). When implementing a large-scale invasive weed control program using herbicides, many land managers will often opt to substitute the labor intensive precision of spot application with the increased application efficiency of broadcast spraying. Large scale treatment is selected based on the thought that it is a more thorough control effort with tangible and immediate benefits. Some studies have shown herbicides have positive influences on the release of grass species combined with drastic decreases in invader biomass, density and seed production during the window of control (Hubbard 1975, Chicoine 1984, Roche 1988, Davis 1990, Sheley and Jacobs 1997). An ancillary outcome of the broadcast method is preventive control of outlying unidentified infestations that are at the patch scale and smaller. Many land managers and owners have devoted the majority of their energy and resources toward controlling specific species with little regard to the existing or resulting plant community. "Because of environmental, ecological, and economical concerns, the appropriateness and effectiveness of rangeland weed management practices are being questioned" (Sheley et al. 1996). Traditional leaders of the Northern Cheyenne are among these concerned about the long-term results of current management practices. Long-term control efforts, focusing on killing weeds, have not predictably restored plant community structure and composition to preinfestation levels nor has leaving infested acres uncontrolled. In both cases, the long-term structure and function of native communities may be at risk (Kedzie-Webb et al.

2001). Either successional trajectory fails to meet contemporary or cultural land use objectives such as sustainable livestock grazing, wildlife habitat, recreation, sustenance and medicinal plant gathering. For a culture grounded in the natural cycle of life, preservation of sacred species is as critical to the Northern Cheyenne homeland and way of life as net profit is to agriculture.

Selective herbicides can be effective tools for controlling the propagation of nonindigenous invaders in the short term, while improving the abundance of coexisting grass species (Davis 1990). One concern of herbicide use is associated with their varying degrees of efficacy on desirable broadleaf species in the community which may be functionally responsible for reducing invasibility and maintaining the ecological function and processes critical to sustainability (Pokorny 2002). Restoration of plant communities to their ecological site potential has become a goal of many land managers. The long-term dynamics after invasion or after nonidigenous species control with herbicides has potentially similar detrimental outcomes with regard to plant community function and structure. The resultant community is generally limited in species composition by the time the target nonindigenous species seed bank is exhausted. Thus, land managers must improve their understanding of plant community dynamics and refine management technology, if they hope to guide succession to a desirable community (Sheley et al. 1996).

This thesis reports on research that was conducted to meet three objectives and test three hypotheses.

I. To characterize the efficacy of five commonly recommended rangeland herbicides on frequently occurring desirable forb species. Specifically, quantify impacts on individual species in the indigenous plant community so that Northern Cheyenne traditionalists can begin to identify potentially impacted cultural species. In addition, I wanted to identify which herbicide and rate combination would be effective on target weeds while maintaining important desirable forbs. I hypothesized that herbicide impact on forbs would depend upon species, herbicide and rate of application.

II. To quantify co-ocurring indigenous species' response to the removal of a nonindigenous invader, sulfur cinquefoil. I hypothesized that repeated hand removal of sulfur cinquefoil would increase indigenous species, diversity, richness, cover, biomass and density with bimonthly removal for two growing seasons.

III. Test densely seeded islands as a mechanism for reduced cost restoration of cultural species on a landscape scale. I hypothesized that densely seeded island populations would provide a source for new colonies and thus may provide a low cost method of introducing desired forbs into a reclaimed plant community. I hypothesized that island shape would affect seedling establishment and dispersal, in that rectangular islands are narrower providing reduced direct sunlight on seedlings reducing heat stress. Additionally, I hypothesized that rectangular islands would have greater dispersal because of longer edge length than square islands of equal area. Many forb species are not available in commercial quantities. Therefore, cost and/or limited availability prohibits their addition to large scale seed mixtures. Thus, lower cost mechanisms for restoration are needed.

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CHAPTER 2

COMMUNITY RESPONSE OF NON-TARGET SPECIES TO HERBICIDE APPLICATION AND REMOVAL OF THE NONINDIGENOUS INVADER *POTENTILLA RECTA* L.

Introduction

For centuries Great Plains natives lived and developed a detailed understanding of the indigenous flora and fauna. In the absence of nonindigenous invaders, these diverse groups of native people interacted with their environment in a manner that was respectful and sustainable, yet dynamic. The settling of this region brought immediate disturbance to the natives' system as well as nonindigenous plant species some of which now inhabit millions of hectares (Sheley and Petroff 1999). Some species of specific concern include spotted knapweed (*Centaurea maculosa* Lam.), Russian knapweed (*Centaurea repens* L.), leafy spurge (*Euphorbia esula* L.), and sulfur cinquefoil (*Potentilla recta* L.). Evidence suggests that these invaders may decrease indigenous species diversity, richness, and biomass (Tyser and Key 1988, Belcher and Wilson 1989, Kedzie-Webb et al. 2001). In addition, invasions may facilitate degradation of ecosystem structure and function (Vitousek 1986, Randall 1996) by altering critical processes such as nutrient cycles, hydrologic cycles, and energy flow (Vitousek and Walker 1986, Lacey et al. 1989).

The displacement of indigenous species is a cultural and ecological concern for plant managers throughout this region. Managing for diverse indigenous plant assemblages not only increases community tolerance of stress events, like drought (Tilman 1996), it may improve resistance to invasion by non-indigenous species (Elton 1958, Rejmanek 1996, Levine and D'Antonio 1999, Pokorny 2002). Herbicides have been used as the primary tool for controlling invaders (Hamaker et al. 1967, Griffith and Lacey 1991). However, there is little evidence that repeated herbicide use alone at landscape scales has consistently displaced invaders nor restored the structure and function of native plant communities. The long-term impact of herbicide use on nontarget native forbs is not well tested, but assumed to be substantial (Harris and Cranston 1979, Cuda et al. 1989). Rice et al. (1997) found minimal herbicide impacts to native forbs over time, after a single application, however retreatment was necessary to control the invader (*Centuaurea maculosa*). Treatments applied frequent enough to truncate new seed production and exhaust the seed bank of nonindigenous species could presumably require several years if not decades of treatment to accomplish (Griffith and Lacey 1991). This process could produce a remnant community low in richness and diversity, which subsequently could be functionally altered and/or made more susceptible to reinvasion by nonindigenous species.

The first objective was to improve understanding of herbicide effects on community dynamics at the species level in order to refine the Tribe's use of technology and advance the development of successional weed management strategies (Sheley et al. 1996). Specifically, this study was implemented to quantify herbicide impacts on species of cultural relevance to the Northern Cheyenne tribe. The study also addresses the objective of identifying potential herbicide and rate combinations that are efficacious on nonindigenous weeds and could have limited impact to indigenous forbs. The second objective of this study was to quantify co-existing indigenous species response to the removal of a nonindigenous invader without herbicide present. The removal would better characterize the impacts of sulfur cinquefoil on indigenous plant communities while improving understanding of the potential reassembly of an invaded community (Lockwood 1997). I hypothesized that native grasslands would exhibit reductions in culturally sensitive forb cover, biomass and density relative to the low, medium and high rates of selective rangeland herbicide. I also hypothesized that removal of an invader, sulfur cinquefoil, would increase indigenous species cover, biomass and density and increase indigenous plant species richness and diversity.

Methods

Study sites

Field studies at White Buffalo and Paddy Creek were conducted to quantify impacts of herbicidal control on native plants of the Northern Cheyenne Reservation in southeastern Montana from 1998 to 2000. The first site was established near the head of White Buffalo drainage 16.5 km south of Lame Deer, Montana (45 ^{0} 28' N; 106 ^{0} 35' W). Annual temperatures at White Buffalo range from 37 to - 37° C with an average frost-free period of 108 days (USDA-NRCS 1996). At White Buffalo, mean precipitation is 432 mm at the 1250 m elevation site. Precipitation over the year is bimodal with peaks occuring in spring and autumn. The soil is a Bitton -Twin Creek- Ringling, dry, complex with slopes ranging from 5 to 10% at this site. Bitton and Twin creek loams comprise 65% of the complex. White Buffalo is classified as a *Festuca idahoensis/Agropyron smithii* habitat type similar to that described by Mueggler and Stewart (1980) for western

Montana. Idaho fescue (*Festuca idahoensis* Elmer) dominates the site and had canopy cover of 40.5% in 1999 and 2000. Other infrequently established grasses include Kentucky bluegrass *(Poa pratensis* L.), needle & thread (*Hesperostipa comata* (Trin. & Rupr.) Barkworth), and smooth brome (*Bromus inermis* Leyss.). Predominant forbs include western yarrow (*Achillea millefolium* L.), cudweed sagewort (*Artemisia ludoviciana* Nutt.), many flowered aster (*Aster ericoides* L.) and slimflower scurfpea (*Psoralidium tenuiflorum* (Pursh.) Rydb.). Species richness sampling detected 29 species at White Buffalo in 1999.

The second site was located in the upper reaches of the Paddy Creek drainage 18 km southeast of Busby, Montana (45 ^{0} N 28' ; 106 ^{0} 40' W). This site has 8-15% northeasterly slope and the soil at Paddy Creek is Shambo - Doney - Cabba loams at 40, 30 and 15 %, respectively (USDA-NRCS 1996). Elevation is 1220 m at Paddy Creek which receives an average of 432 mm of bimodal precipitation annually. The frost-free duration averages 112 days. Native species include western wheatgrass (12% canopy cover) associated with western yarrow, and cudweed sagewort. Kentucky bluegrass (19% canopy cover) and black medic (*Medicago lupulina* L.) were the most common non-native species in 1999.

Study sites

Two Northern Cheyenne Reservation study sites, Muddy Creek and Skunk Creek, were established to quantify native plant recovery in the absence of herbicide by hand removing sulfur cinquefoil from a native plant community weekly in 1999 to 2000. The first site was located near the confluence of Muddy Creek and Rosebud Creek, (45 ^{0} 36 '

N;106 ^{0} 44'W). Muddy Creek is located 10 km west of Lame Deer, Montana. Muddy Creek soil is a Yamac loam (USDA-NRCS 1996). Slope ranges from 2 to 8% and this site receives about 305 mm of annual precipitation. Muddy Creek has a frost-free period of 115 to 130 days. At the time the study was initiated, the site was dominated by sulfur cinquefoil with a mean canopy cover of 43% and 170 plants m^{-2} in 1999. Kentucky bluegrass comprised 21% of the total grass canopy cover, which was 26% at Muddy Creek. Other forbs include many flowered aster, scarlet gaura (*Gaura coccinea* Nutt. ex Pursh.) and alyssum (*Alyssum* L.).

The second site, Skunk Creek, was located 16.5 km south of Busby, Montana (45 ^{0} 23'N; 106 ^{0} 58' W). Skunk Creek soil is a Shambo loam with a 2 to 8 % northern slope (USDA-NRCS 1996). Annual precipitation is about 432 mm per year with a frost-free period of 112 days. Annual temperatures range from 37 to -37° C. Sulfur cinquefoil was the dominant species with a mean of 35% canopy cover and mean density of 145 plants m^{-2} for non-removed plots in 1999. Primary grass species include sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.) which had 95 tillers m^{-2} and Kentucky bluegrass which had a mean density of 102 tillers m^{-2}. Other common forbs include many flowered aster and cudweed sagewort.

Experimental design

White Buffalo and Paddy Creek study sites consisted of eighteen treatments (5 herbicides, 3 rates and a non-treated control) which were applied to 2-m x 10-m plots on June 24, 1998. Plots were sampled in late June of the following year when cool season species were seeded and most forbs were flowering. The treatments were replicated three times arranged in a randomized-complete-block design at each site due to suspected environmental gradients caused by slope. Treatments consisted of three rates each of: a commercial formulation of 2,4-D+clopyralid respectively (a. 0.28 kg ai/ha + 0.0532 kg ai/ha, b. 0.56 kg ai/ha +0.1064 kg ai/ha, c. 0.84 kg ai/ha + 0.1596 kg ai/ha), 2,4-D amine (a. 0.532 kg ai/ha, b. 1.064 kg ai/ha, c. 1.596 kg ai/ha), metsulfuron (a. 0.0042kg ai/ha, b. 0.021 kg ai/ha, c. 0.032 kg ai/ha), picloram (a. 0.14 kg ai/ha, b. 0.28 kg ai/ha, c. 0.56 kg ai/ha) and clopyralid (a. 0.05025 kg ai/ha, b. 0.21 kg ai/ha c. 0.42kg ai/ha). Plots were hand sprayed in 1998 using a CO_2 backpack spray unit applying 130 l/ha total spray volume. Conditions were overcast, heavy dew, wind < 5 mph with a temperature of 18.3^0 C.

Muddy Creek and Skunk Creek treatments (sulfur cinquefoil removed and non-removed) were replicated five times in a completely randomized design at each site in $1m^2$ plots. Due to plot size an environmental gradient was not suspected thus the experiment was completely randomized. Sulfur cinquefoil removal treatments were accomplished by hand pulling 100% of the sulfur cinquefoil twice during June, July, and August for two growing seasons.

Sampling

Sampling began at White Buffalo, Paddy Creek, Muddy Creek and Skunk Creek in 1999. Canopy cover was estimated on a per species basis for all species present in three randomly placed 20 x 50-cm frames within each plot at White Buffalo and Paddy Creek. Canopy cover, at Muddy Creek and Skunk Creek, was collected by the same procedure with a single randomly placed frame in each plot. Canopy cover was the only data collected within each plot in 1999. In 2000, canopy cover, density and biomass data were collected 24 months after initial treatment at both sites. Grass density, at White Buffalo and Paddy Creek, was recorded by counting all tillers of each species present within three randomly placed 20 x 50-cm frames. Grass density, at Muddy Creek and Skunk Creek, was counted from a single randomly placed 20 x 50-cm frame in each plot. Simultaneously, forb density was collected by counting the number of individuals, ramets and genets combined for each species within the frame. Biomass, at all four locations, was collected by clipping and sorting individual species from a randomly placed 0.44-m^2 hoop. Samples were air dried at 60 ° C for 48 hours and weighed.

Data analysis

White Buffalo and Paddy Creek data were initially combined and analyzed as a split plot for species richness. Analysis of variance (ANOVA) detected White Buffalo and Paddy Creek differed in species richness significantly which prompted site specific analysis. Treatment effects on diversity, richness, canopy cover, biomass and density were analyzed. ANOVA was used on the average of the three subsamples to test the hypothesis that native forb species respond variably to different rates of rangeland herbicides (SAS 1990).

First, at White Buffalo and Paddy Creek, ANOVA for a split-plot was used to determine effects of herbicide, rate, herbicide*rate, year, year*herbicide, year*rate, and year*herbicide*rate effect on canopy cover, diversity (Shannon-Weiner $H' = -\sum P_i \bullet \log (P_i)$), and species richness where year was the split plot in the ANOVA. Herbicide, rate and herbicide*rate were the whole-plots. Block*herbicide*rate was used as the whole plot error term. Year and all interactions with year were the split-plots. The model error was used as the split plot error term.

Second, at White Buffalo and Paddy Creek, ANOVA for a randomized-complete-block was used to determine herbicide, rate and herbicide*rate effects on biomass and density for grasses and forbs. These dependent variables were analyzed first as groups and then individually by species. Block*herb*rate was used as the error term for these analyses. When treatment or interaction were significant at the $P \leq 0.05$ level then mean separations were achieved using Fisher's protected Least Significant Difference (LSD) (Peterson 1985). With the exception of western yarrow cover at White Buffalo which is presented at the $P \leq 0.06$ level to illustrate similar trend between sites.

For Muddy Creek and Skunk Creek, sites were combined and ANOVA for a completely randomized design, split for year effects on canopy cover, was used to determine effects of site, removal treatment, year and year*removal on diversity (Shannon-Weiner) and canopy cover for grasses and forbs separately. Sulfur cinquefoil was not included in the forb group. T-tests were conducted to detect differences in richness, density and biomass in the perennial grass group and again for the indigenous forb group. Richness between grasses and forbs was not tested.

Results

Perennial grass canopy cover

Muddy Creek & Skunk Creek. Perennial grass canopy cover effects were not detected following the hand removal of sulfur cinquefoil (Table 2.1).

Table 2.1. P-values for hand removing sulfur cinquefoil. P-values for cover were considered significant at the $P \leq 0.05$ level.

Source	df	P. Grass Cover	Forb Cover
site	1	0.31	0.47
removal	1	0.08	<0.02
year	1	0.06	0.61
Year x removal	1	0.11	0.07

White Buffalo. The impact of herbicide on total perennial grass canopy cover, which included indigenous and nonindigenous grasses, was dependent on the interaction of herbicide and rate of application at White Buffalo (Table 2.2). Total perennial grass canopy cover in the untreated control plots was 51%. Clopyralid applied at 0.42 kg ai/ha, clopyralid at 0.0532 kg ai/ha plus 0.28 kg ai/ha of 2,4-D, and metsulfuron at 0.0042 kg ai/ha increased total perennial grass canopy cover to 63, 61, and 63%, respectively (Fig. 2.1). All other treatments produced cover similar to that of the control. No treatment effects were detected for Idaho fescue cover.

Table 2.2. P-values for perennial grass cover at White Buffalo and Paddy Creek. P-values indicate significant differences at the P≤0.05 level.

Site		White Buffalo			Paddy Creek	
Source	df	Idaho fescue	All perennial grass	western wheatgrass	Kentucky bluegrass	All perennial grass
Block	2	0.01	<0.01	0.01	0.01	0.32
Herb	4	0.31	0.93	0.91	0.02	0.03
Rate	2	0.66	0.14	0.67	0.14	0.13
Herb*rate	8	0.19	<0.01	0.10	0.89	0.38
Year	1	0.32	0.06	0.08	<0.01	<0.01
Year*herb	4	0.60	0.71	0.18	0.94	0.64
Year*rate	2	057	0.65	0.95	0.32	0.56
Year*herb*rate	8	0.61	0.85	0.28	0.70	0.34

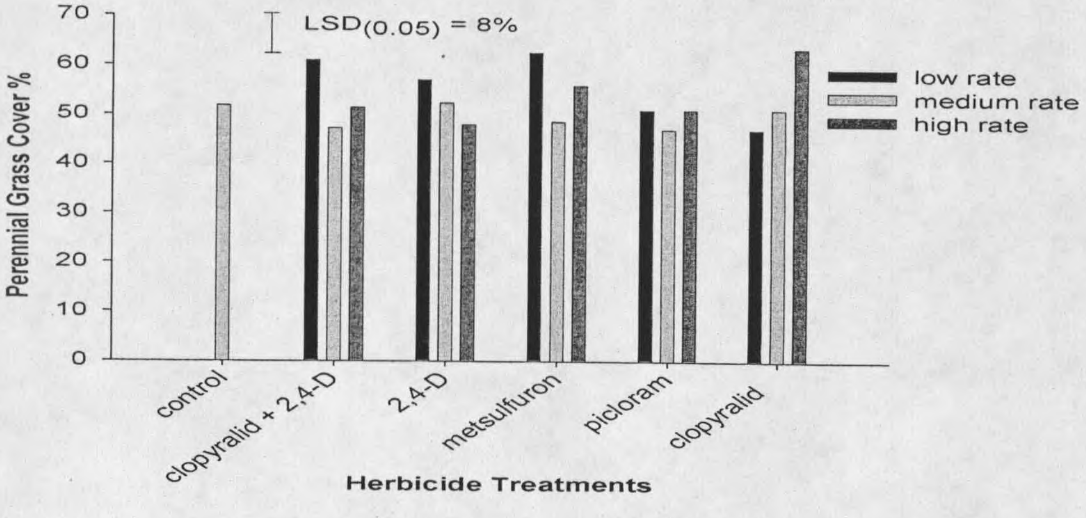

Figure 2.1. White Buffalo. 1999-2000 perennial grass cover response to herbicides applied in 1998. Bar represents the Least Significant Difference.

Paddy Creek. Total perennial grass and Kentucky bluegrass canopy cover differed between years and depended upon herbicide main effects (Table 2.2). In 1999, mean perennial grass canopy cover was 38% and increased to 50% two years after treatment. Picloram treatment resulted in 54% perennial grass canopy cover, which was 13% higher than that of the control (Fig. 2.2). In 1999, Kentucky bluegrass had 23% canopy cover which increased to 33% in 2000. Picloram increased Kentucky bluegrass canopy cover from 23% to 38% both years.

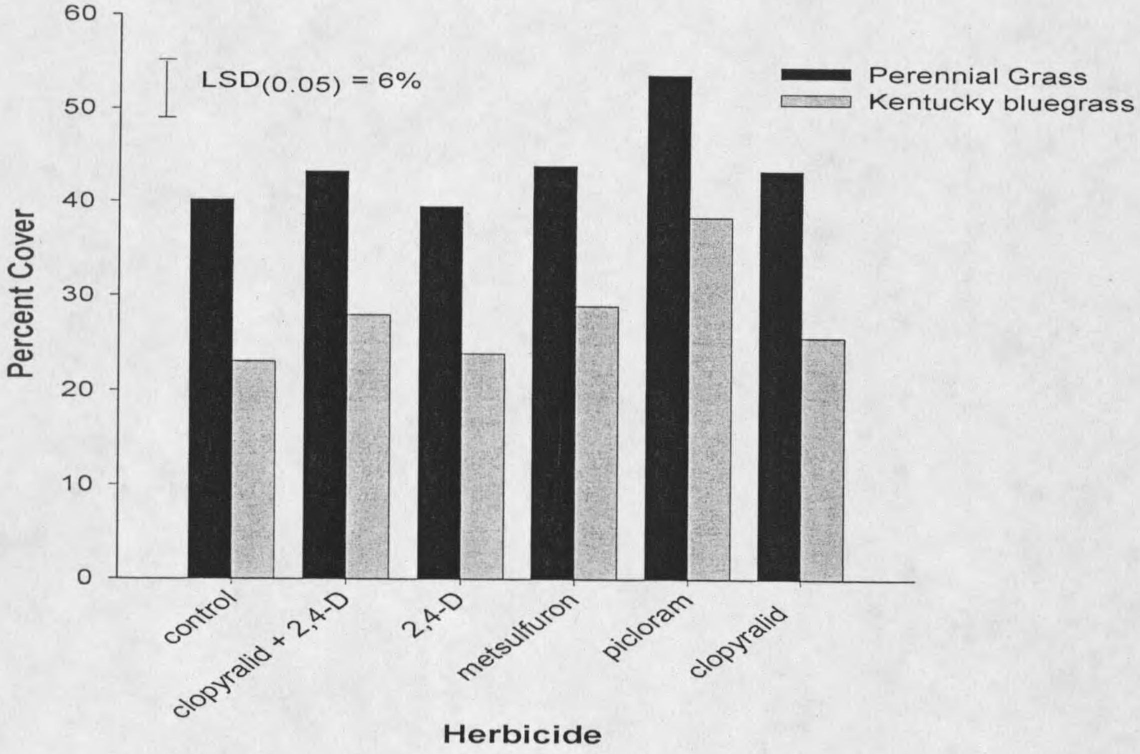

Figure 2.2. Paddy Creek. Herbicide main effect on perennial grass cover compared with herbicide main effect on Kentucky bluegrass cover. Bar represents the Least Significant Difference.

Perennial grass biomass

Muddy Creek & Skunk Creek. Hand removal of sulfur cinquefoil did not produce significantly detectable differences in perennial grass biomass.

White Buffalo. Herbicides interacted with rate to determine perennial grass biomass (Table 2.3). Total grass biomass in the untreated control plots was 779 kg/ha (Fig. 2.3). Clopyralid applied at 0.16 kg ai/ha plus 2,4-D at 0.84 kg ai/ha increased perennial grass biomass to 1701 kg/ha. Two,4-D applied at 0.532 kg ai/ha yielded 1573 kg/ha, while metsulfuron at 0.02 kg ai/ha produced 1768 kg/ha. When picloram was applied at 0.56 kg ai/ha, perennial grass yielded about 2.5 times more biomass (1954 kg/ha). Other herbicide by rate combinations did not significantly alter biomass from that of the control. Biomass of individual species were analyzed and no significant differences were detected.

Table 2.3. P-Values for herbicide and rate main effects with herbicide x rate interactions on perennial grass biomass. P-values considered significant at the $P \leq 0.05$.

Site		White Buffalo		Paddy Creek		
Source	df	Festuca idahoensis	perennial grass	western wheatgrass	Kentucky bluegrass	perennial grass
Block	2	0.03	0.60	0.02	0.15	0.24
Herb	4	0.68	0.66	0.62	0.07	0.09
Rate	2	0.43	0.26	0.79	0.21	0.27
Herb*rate	8	0.22	0.03	0.38	0.03	0.03

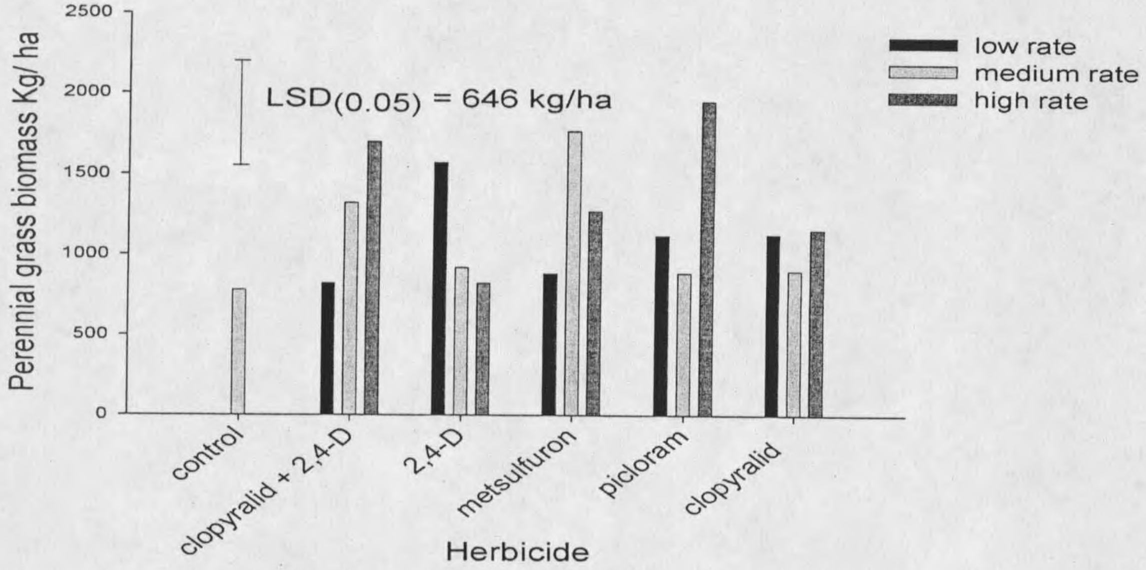

Figure 2.3. White Buffalo. Herbicide by rate interaction effects on perennial grass biomass. Bar represents the Least Significant Difference.

Paddy Creek. Herbicides interacted with rate to affect perennial grass and Kentucky bluegrass biomass (Table 2.3). Clopyralid at 0.1064 kg ai/ha plus 2,4-D at 0.56 kg ai/ha increased perennial grass biomass to 1512 kg/ha over that of the control (726 kg/ha) (Fig. 2.4). Picloram at 0.56 kg ai/ha yielded nearly 2 times as much perennial grass (1442 kg/ha) as that of the control. At Paddy Creek, the control produced 187 kg/ha of Kentucky bluegrass. Picloram applied at 0.56 kg ai/ha increased Kentucky bluegrass biomass to 498 kg/ha (Fig. 2.5). Clopyralid applied at 0.1064 kg ai/ha plus 2,4D at 0.56 kg ai/ha increased Kentucky bluegrass to 521 kg/ha, which was similar to that of 2,4-D applied at 1.596 kg ai/ha alone.

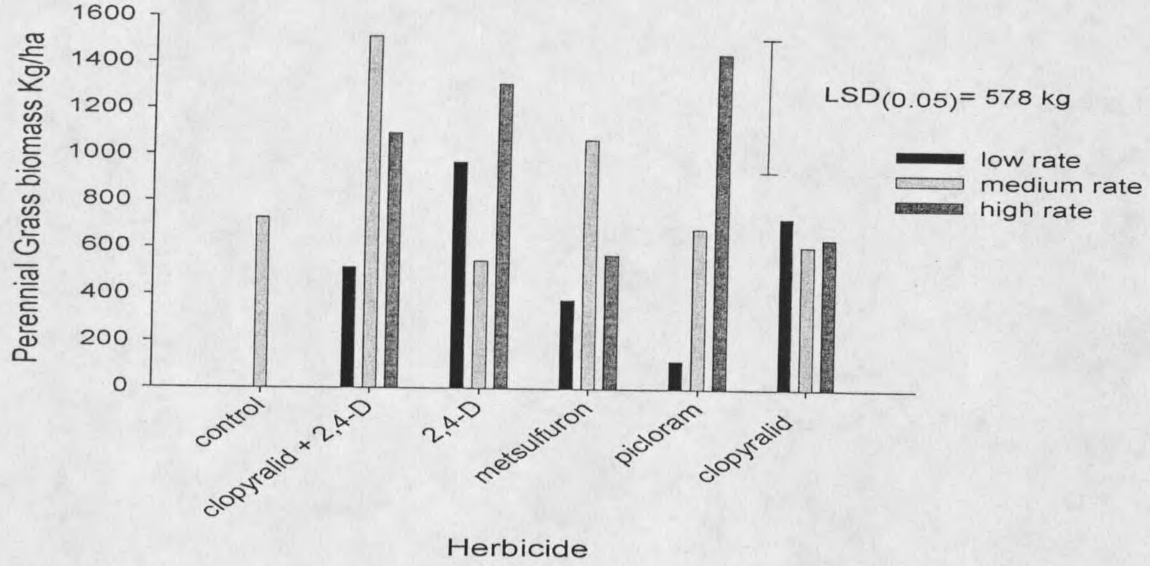

Figure 2.4. Paddy Creek. Herbicide interaction with rate effect on perennial grass biomass. Bar represents the Least Significant Difference.

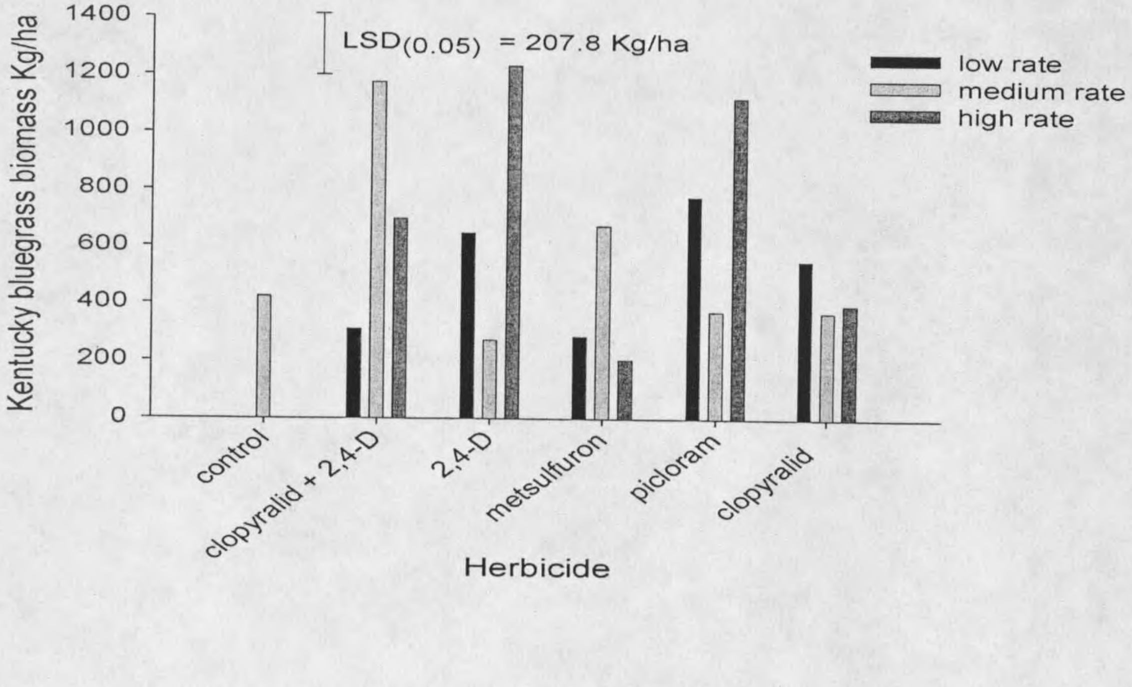

Figure 2.5. Paddy Creek. 2000. Herbicide interaction with rate effect on Kentucky bluegrass biomass. Bar signifies the Least Significant Difference.

Perennial grass density.

Treatment effects on perennial grass density were not detected by hand removal (Pr > [t] = .19). Herbicides did not affect grass density for any of the sites (Table 2.4).

Table 2.4. P-values for herbicide and rate main effects with herbicide x rate interactions on perennial grass density at both sites. P-values considered significant at the P ≤0.05 level.

Site		White Buffalo		Paddy Creek		
Source	df	Idaho fescue	perennial grass	western wheatgrass	Kentucky bluegrass	perennial grass
Block	2	0.30	0.94	0.04	0.03	0.45
Herb	4	0.27	0.77	0.9	0.89	0.80
Rate	2	0.45	0.69	0.31	0.71	0.95
Herb*rate	8	0.60	0.62	0.29	0.6	0.41

Forb canopy cover

Muddy Creek and Skunk Creek. Non-target indigenous forb canopy cover depended upon the main effect of pulling sulfur cinquefoil (Table 2.1). Total forb canopy cover was about 5% in the unpulled control and increased to around 12% in the removed plots.

White Buffalo. Total forb canopy cover was dependent upon year after treatment and herbicide main effect at White Buffalo (Table 2.5). Total forb cover was around 28% in 1999, the first year after treatment, and decreased to 24% in 2000. Total forb cover ranged from 31% following 2,4-D treatments to around 19% after picloram (Fig. 2.6a). The control produced 28% total forb cover. Picloram decreased forb cover to 19%. Two,4-D alone produced 31% total forb cover and clopyralid plus 2,4-D produced 30%. Both treatments produced more forb cover than picloram but were similar to the control. Metsulfuron and clopyralid alone produced similar forb cover as the control and others.

Table 2.5. P-values for herbicide and rate effects at White Buffalo for forb canopy cover in 1999 and 2000. P-values considered significant at P≤ 0.05.

Source	df	western yarrow	cudweed sagewort	many flowered aster	slim flower scurfpea	All forbs
Block	2	0.00	0.00	0.38	0.28	0.13
Herb	4	0.06	0.34	0.32	0.59	0.03
Rate	2	0.58	0.98	0.15	0.21	0.5
Herb*Rate	8	0.73	0.35	0.77	0.21	0.22
Year	1	0.23	0.24		0.00	0.02
Year*herb	4	0.15	0.36		0.74	0.25
Year*rate	2	0.02	0.16		0.17	0.06
Year*Herb*Rate	8	0.69	0.60		0.19	0.52

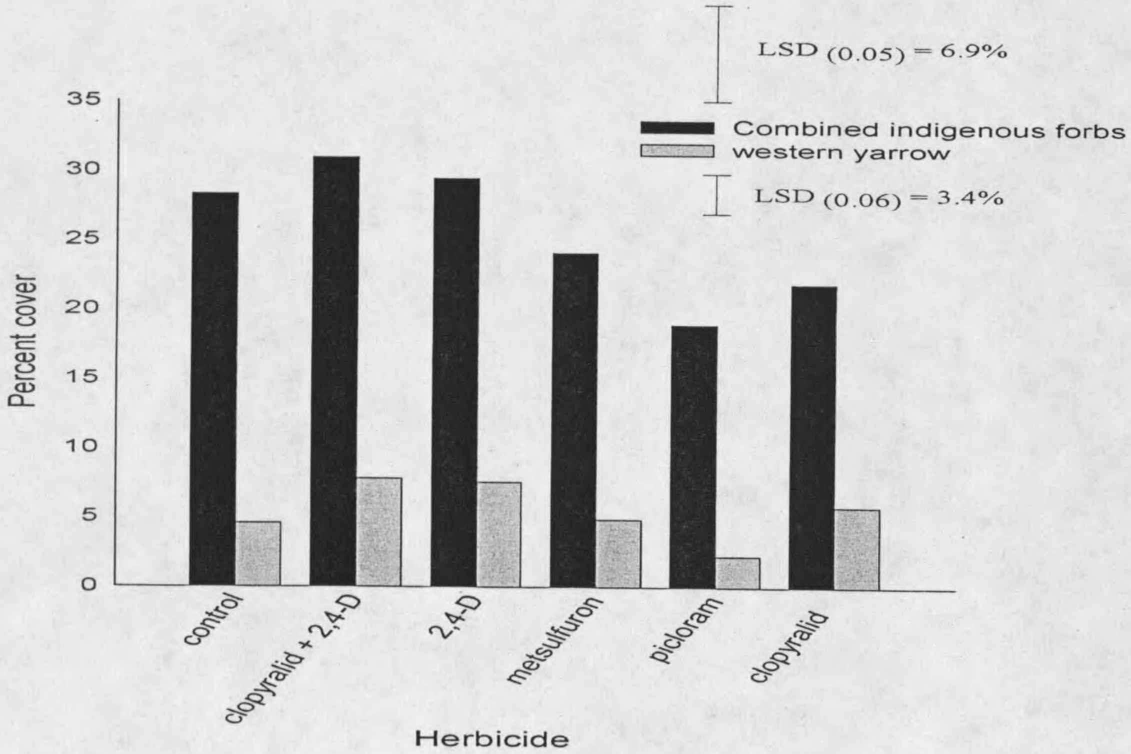

Figure 2.6a. White Buffalo. Herbicide main effect on combined forb canopy cover and western yarrow canopy cover trend. Bar signifies the Least Significant Difference.

Since yarrow is a cultural species, it is presented. Yarrow canopy cover was not significantly affected by herbicide main effect but did exhibit an interesting trend response to herbicide at White Buffalo (Table 2.5). Yarrow treated with 2,4-D alone or clopyralid plus 2,4-D produced about 8% cover, nearly 3.5 times more cover than those receiving picloram. All herbicide treatments produced similar cover to the control (Fig. 2.6a). Species specific forb canopy cover analysis detected that western yarrow cover was dependent upon the interaction between year following treatment and the rate of herbicide applied (Table 2.5). In 1999, highest rate herbicide treatments reduced yarrow cover to around 2% versus that of the control at 5% (Fig. 2.6). Two years after treatment, yarrow cover increased to 7% in plots receiving the highest herbicide rates over that of the control.

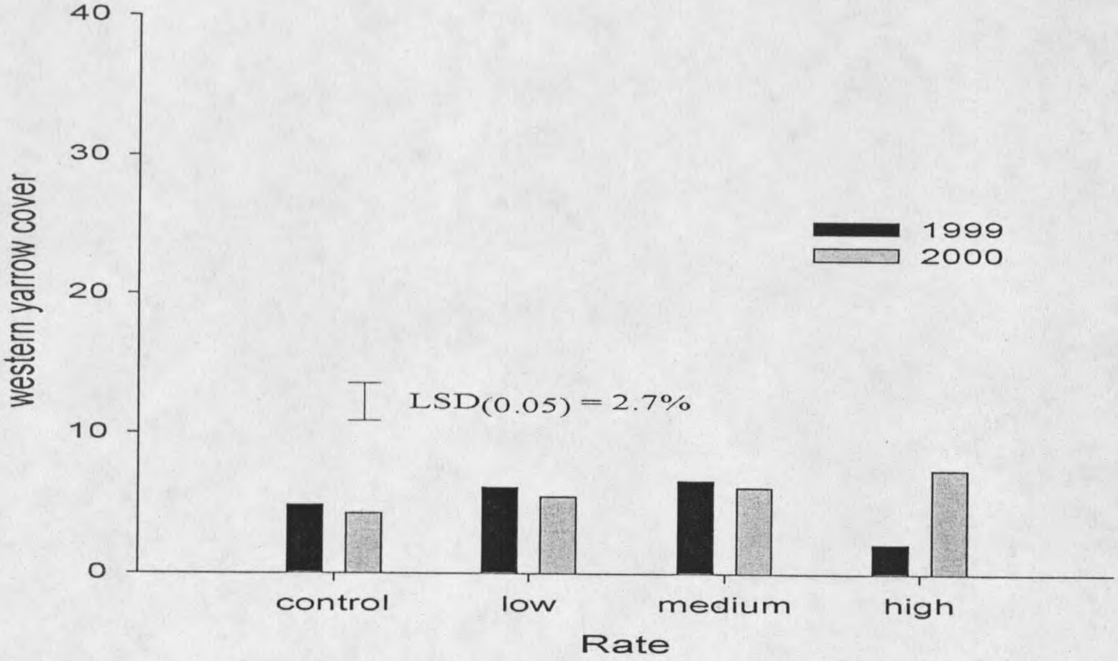

Figure 2.6. White Buffalo. Year interaction with rate effect on western yarrow canopy cover. Bar signifies the Least Significant Difference.

Paddy Creek. Year following treatment and herbicide main effects influenced forb canopy cover at Paddy Creek (Table 2.6). In 1999, total forb cover was 50%, but decreased to 35% in 2000 (Fig. 2.7). Picloram reduced forb cover from 45% in the control to 27%. All other herbicides produced similar cover to the control.

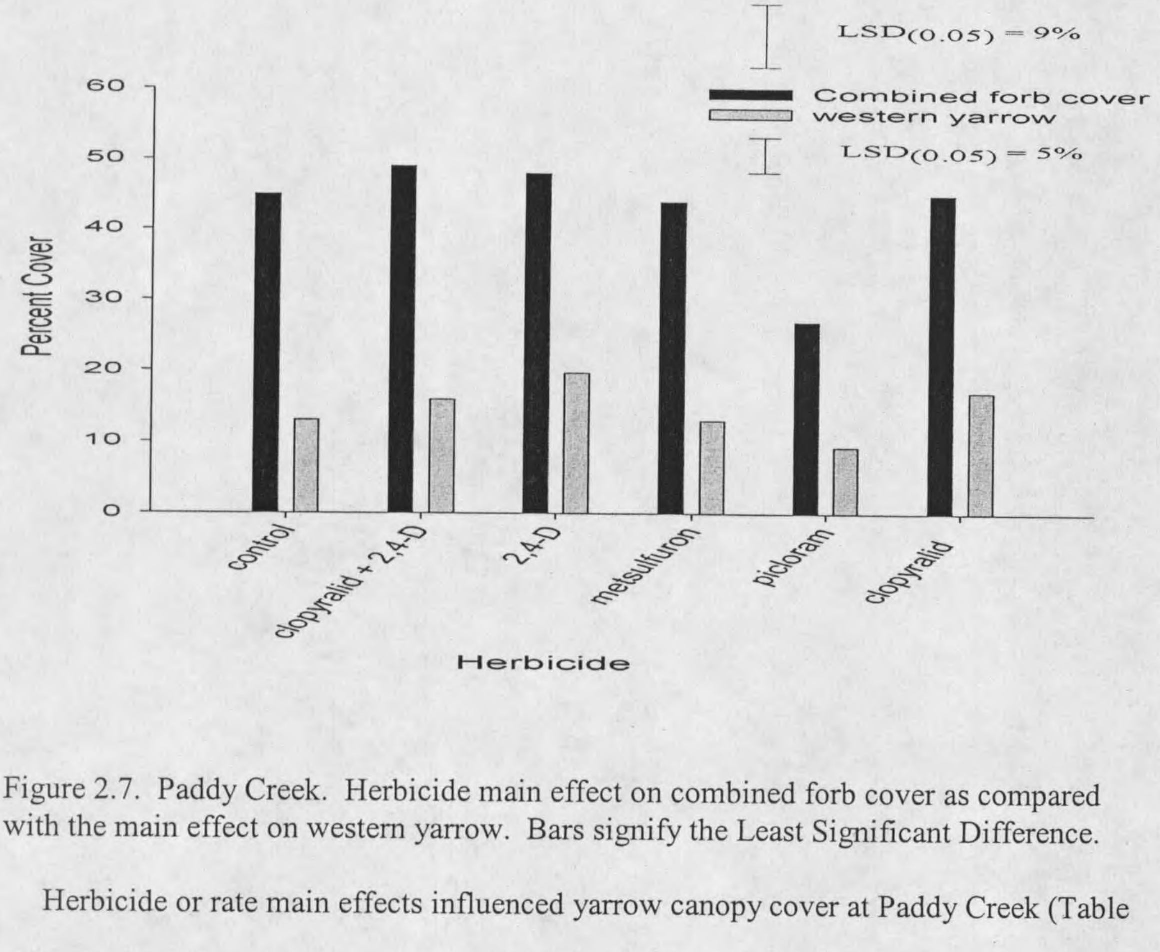

Figure 2.7. Paddy Creek. Herbicide main effect on combined forb cover as compared with the main effect on western yarrow. Bars signify the Least Significant Difference.

Herbicide or rate main effects influenced yarrow canopy cover at Paddy Creek (Table 2.6). Two,4-D increased yarrow cover from 13% in the untreated control to 20%. Picloram reduced yarrow to 9% which was below clopyralid plus 2,4-D (16%), 2,4-D alone (20%) or clopyralid alone (17%, Fig. 2.7). Low and moderate rates of herbicide increased yarrow cover from 13% in the control to 18%. High rates decreased yarrow cover to 10%. High rates produced yarrow cover similar to the control.

Table 2.6. Paddy Creek. P-values for forb canopy cover. P-values considered significant at the $P \leq 0.05$ level.

Source	df	American vetch	western yarrow	cudweed sagewort	black medic	death camas	many flowered aster	Total
Rep	2	0.38	0.06	0.11	0.36	0.02	0.5	0.96
Herb	4	0.02	0.01	0.92	0.01	0.39	0.25	0.01
Rate	2	0.44	0.01	0.75	0.98	0.53	0.7	0.12
Herb*Rate	8	0.04	0.07	0.11	0.21	0.17	0.94	0.39
Year	1	0.01	0.74	0.01	0.01	0.01		0.01
Year*herb	4	0.03	0.84	0.94	0.01	0.65		0.14
Year*rate	2	0.6	0.91	0.61	0.57	0.74		0.56
Year*Herb*Rate	8	0.63	0.46	0.07	0.61	0.35		0.15

Species specific cover analysis detected year following treatment interacted with herbicide to determine vetch canopy cover at Paddy Creek (Table 2.6). In 1999, vetch covered about 3% in the control, clopyralid plus 2,4-D, 2,4-D alone or metsulfuron produced similar vetch cover. Picloram or clopyralid alone decreased vetch cover to trace amounts. In 2000, vetch cover was reduced to a trace throughout the site and differences between herbicides and the control were not detected. Herbicide interacted with rate to affect vetch cover (Table 2.6). When treated with 1.064 kg ai/ha 2,4-D or metsulfuron at 0.0042 kg ai/ha vetch cover increased to 4% over that of the control at 2% (Fig. 2.8). Picloram at 0.56 kg ai/ha reduced vetch cover to a trace (<1%). Vetch response to other herbicide by rate combinations was not detected.

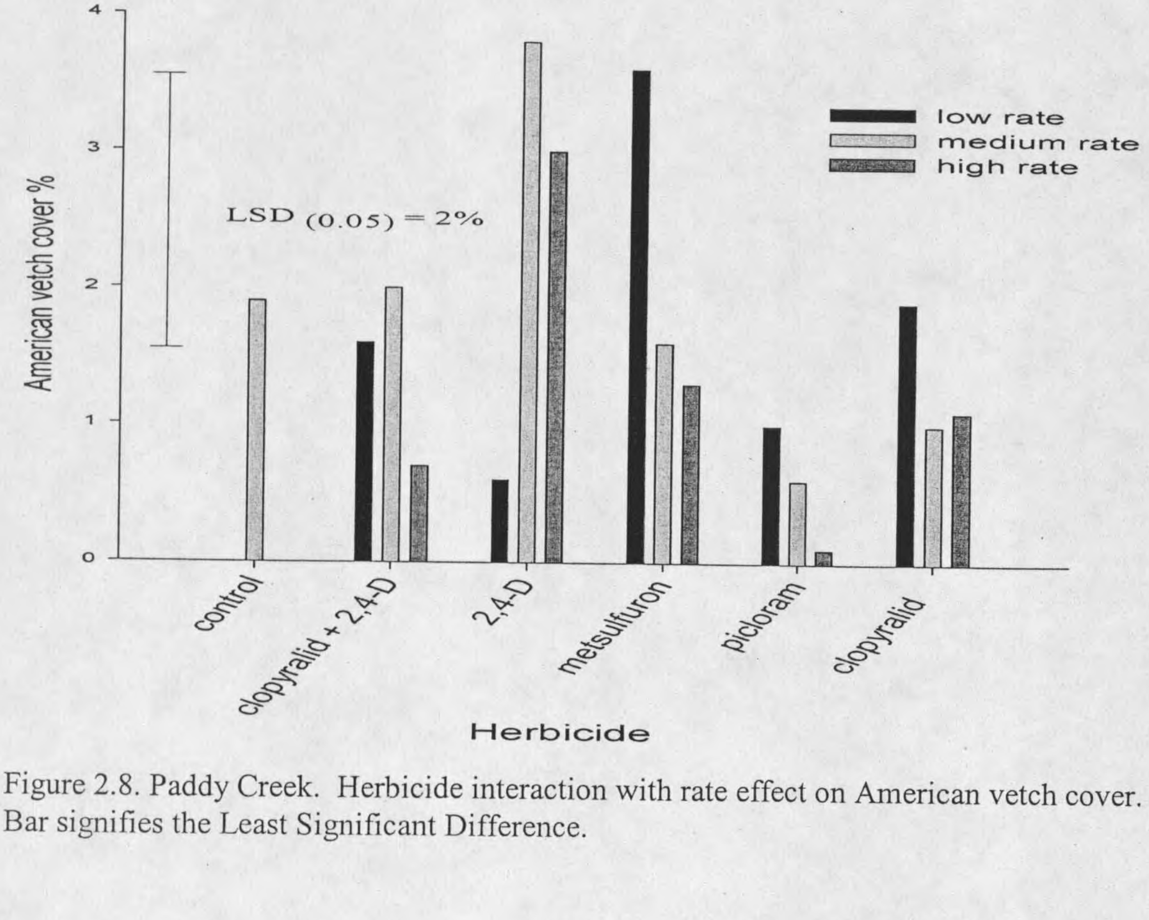

Figure 2.8. Paddy Creek. Herbicide interaction with rate effect on American vetch cover. Bar signifies the Least Significant Difference.

Year interacted with herbicide to determine black medic canopy cover (Table 2.6). In 1999, 2,4-D decreased medic cover from 28% in the control to 20% (Fig. 2.9). Picloram reduced medic cover by 74% one year after treatment suppressing medic cover to about 7%. All other herbicide produced similar black medic cover as the control. In 2000, medic cover was 11.8% in the control. None of the herbicide treatments significantly differed from the control or one another in 2000.

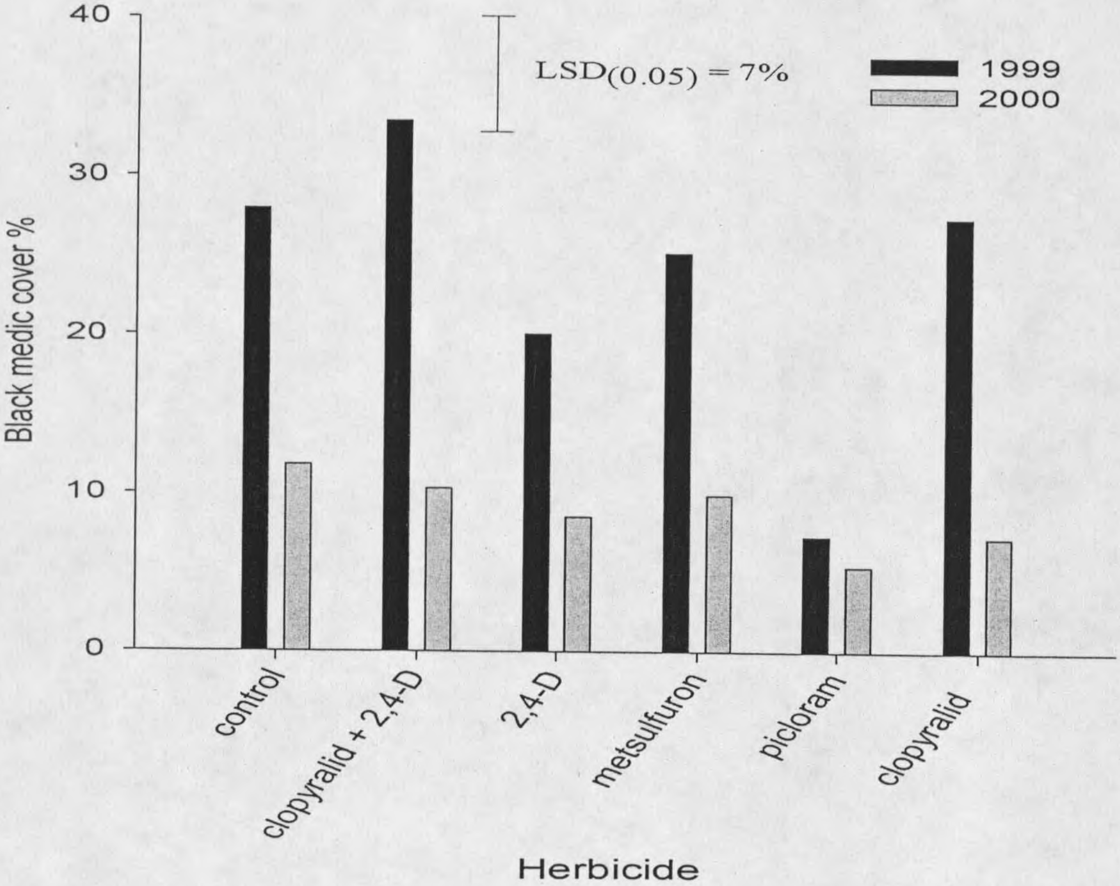

Figure 2.9. Paddy Creek. Herbicide interaction with year after initial treatment effect on black medic canopy cover. Bar signifies the Least Significant Difference.

Forb biomass

Muddy Creek and Skunk Creek. Shifts in forb biomass production were not detected.

White Buffalo. Forb biomass was not affected at White Buffalo. Species specific analysis did not detect any effects on forb biomass.

Paddy Creek. Total forb biomass was not affected at Paddy Creek. Individual species analysis detected that rate affected cudweed sagewort biomass (Table 2.7). Low herbicide rates increased cudweed sagewort biomass from 49.95 kg/ha in the control to 137.9 kg/ha (Fig. 2.10). Other rate effects were not detected.

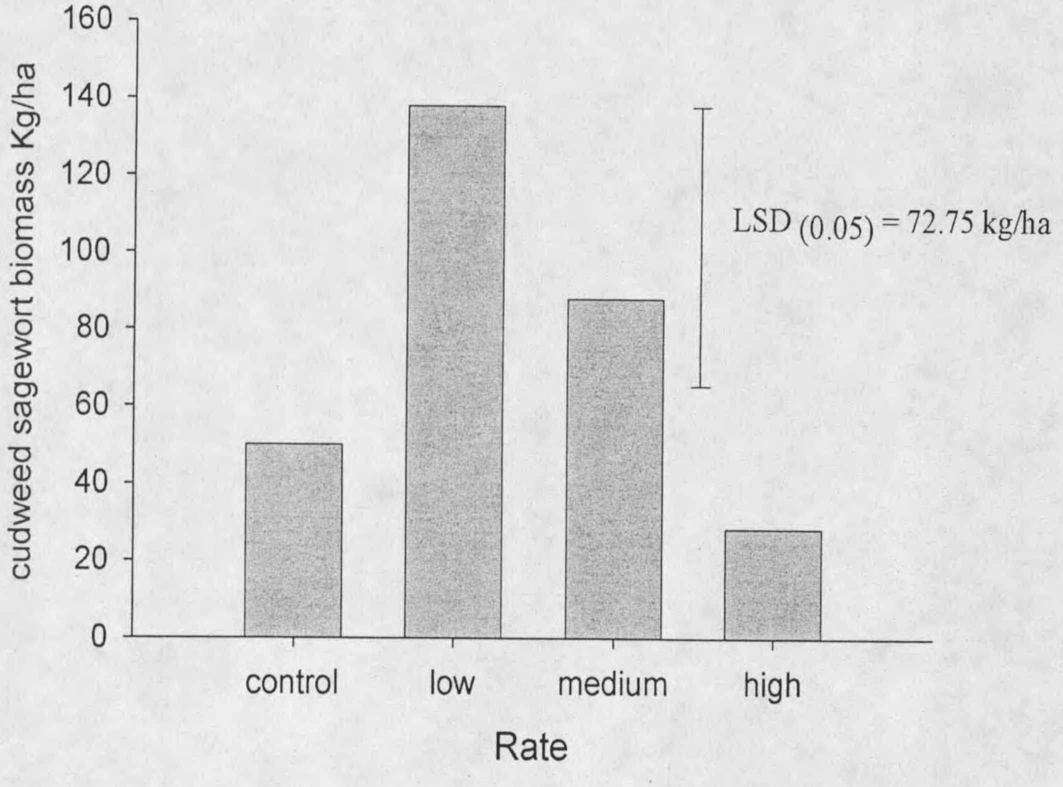

Figure 2.10. Paddy Creek 2000. Cudweed sagewort biomass response to herbicide rate. Bar signifies the Least Significant Difference.

Table 2.7. P-values for forb density response to herbicide and rate main effects and their interaction. P-values considered significant at the P ≤ 0.05 level. Cudweed sagewort P-values are results for main effect and interaction on biomass.

Site		White Buffalo			Paddy Creek			
Source	df	western yarrow	many flowered aster	total forb density	American vetch	western yarrow	cudweed sagewort biomass	total forb density
Rep	2	0.18	0.55	0.56	0.02	0.16	0.81	0.06
Herb	4	0.03	0.57	0.06	0.04	0.32	0.88	0.35
Rate	2	0.46	0.03	0.86	0.79	0.05	0.05	0.11
Herb*Rate	8	0.55	0.69	0.90	0.89	0.04	0.19	0.27

Forb density

Muddy Creek and Skunk Creek. Forb density changes were not detected when sulfur cinquefoil was removed.

White Buffalo. Herbicide did not significantly affect forb density at White Buffalo, but did exhibit a notable trend (Table 2.7). In 2000, total forb density was 167 plants m^{-2}. Metsulfuron, picloram or clopyralid reduced total forb density to 104, 98 and 134 plants m^{-2}, respectively (Fig. 2.11). Clopyralid plus 2,4-D or 2,4D alone produced similar forb density to the control.

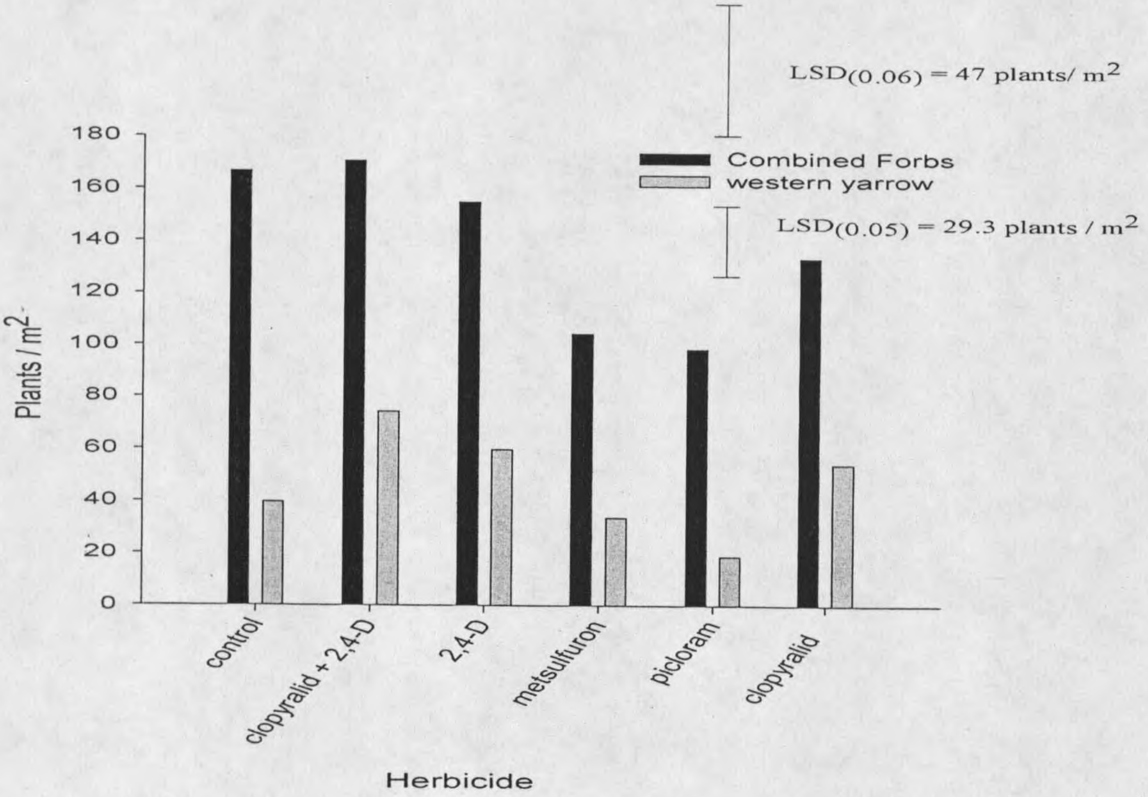

Figure 2.11. White Buffalo 2000. Herbicide main effect on combined forb density compared with density response to western yarrow. Bars signify the Least Significant Difference.

Species specific forb density analysis detected a herbicide main effect on yarrow density (Table 2.7). In 2000, yarrow density increased from 39.8 plants m^{-2} in the control to 74.3 plants m^{-2} after 2,4-D was applied (Fig. 2.11). Picloram decreased yarrow density to 18.9 plants m^{-2} which was lower than 2,4-D (74.3 plants m^{-2}), clopyralid plus 2,4-D (59.8 plants m^{-2}) or clopyralid alone (54.3 plants m^{-2}). Picloram produced similar yarrow density to that of the control.

Many flowered aster density depended on rate effect at White Buffalo (Table 2.7). The many flowered aster control contained 83.6 plants m^{-2} (Fig. 2.12). The lowest applied herbicide rate thinned many flowered aster density to 56.5 plants m^{-2}. Middle and high rates reduced aster density to 28.3 and 23.1 plants m^{-2}, respectively.

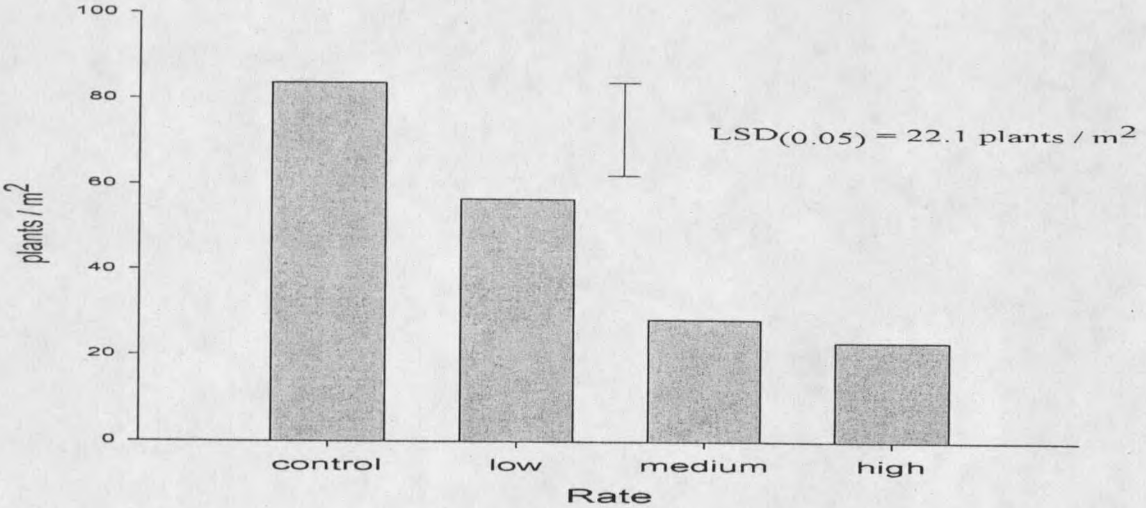

Figure 2.12. White Buffalo2000. Response of many flowered aster to herbicide rate. Bar signifies Least Significant Difference.

Paddy Creek. In 2000, forb density was not affected at Paddy Creek (Table 2.7). Species specific forb density analysis detected a herbicide main effect on vetch density while herbicide interacted with rate to influence yarrow density. Vetch increased from 5.1 plants m^{-2} to 32.2 plants m^{-2} after being treated with clopyralid (Fig. 2.13). Herbicide interacted with rate to determine yarrow density (Table 2.7). Yarrow measured 103.9 plants m^{-2} in the control (Fig. 2.14). Yarrow density was similar to the control for all treatments, but density differed among herbicide by rate combinations. In 2000, 2,4-D at 1.064 kg ai/ha, metsulfuron at 0.0042 kg ai/ha, picloram at 0.14 kg ai/ha or clopyralid at 0.21 kg ai/ha produced 207.7, 189, 153 and 139 yarrow plants m^{-2}. Picloram at 0.56 kg ai/ha decreased yarrow to 9.7 plants m^{-2}.

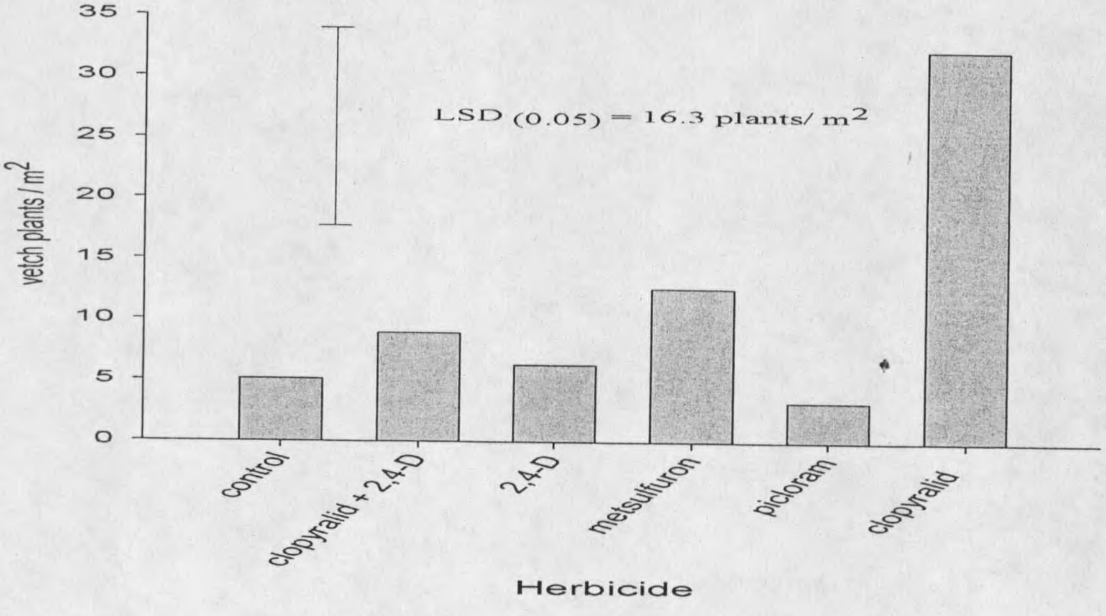

Figure 2.13. Paddy Creek 2000. Herbicide main effect on American vetch density. Bar signifies the Least Significant Difference.

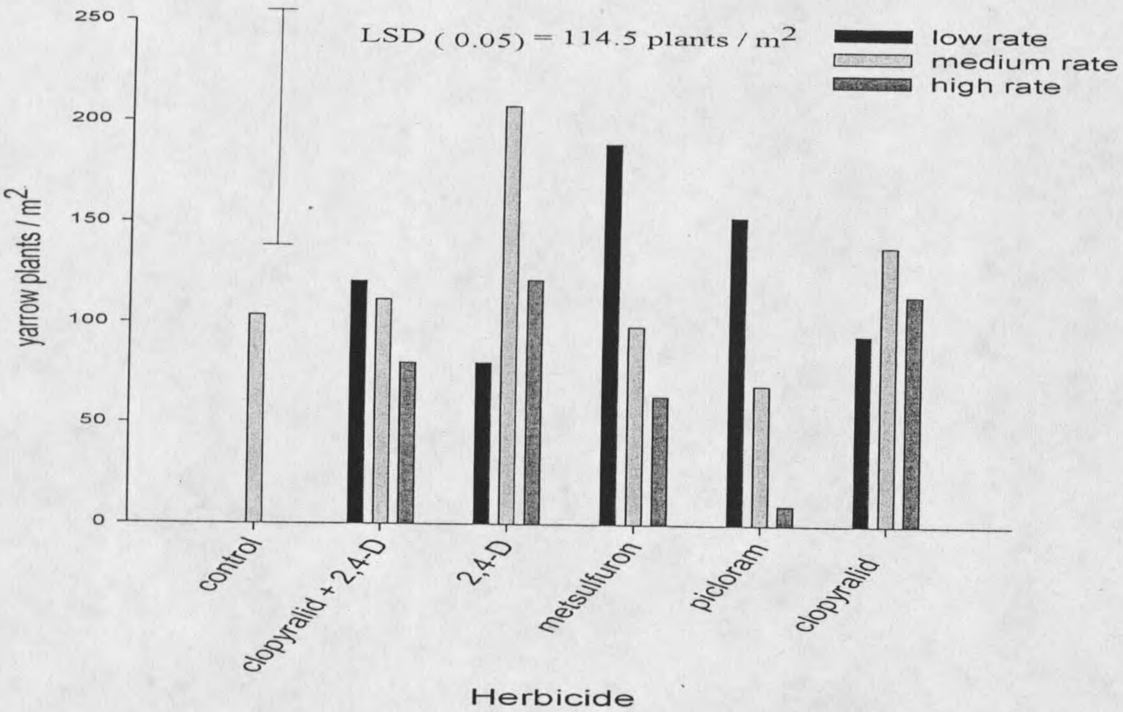

Figure 2.14. Paddy Creek 2000. Herbicide interaction with rate effect on western yarrow. Bar signifies the Least Significant Difference.

Species diversity

Shannon-Weiner Diversity was not affected by hand removal. Shannon-Weiner Diversity was not affected by herbicide treatment at any site.

Species richness

Muddy Creek and Skunk Creek. Total richness depended on the main effect of removing sulfur cinquefoil from the community (Table 2.8). Hand removal increased co-occurring species richness from about 2.85 species to about 4.05 in the removed plots. The increase occurred predominantly in the indigenous forb group. Forb richness increased from 1.25 to 2.4 species in 2000.

Table 2.8. Muddy Creek and Skunk Creek combined. T-test of hand removing sulfur cinquefoil on species richness. Results considered significant at the Pr ≥ [t] 0.05 level.

Variable	df	Pr > [t]
Richness	38	0.0105
Grass richness	38	0.8773
Forb richness	38	0.0073

Paddy Creek. Richness was different between year 1 and year 2 at Paddy Creek. Herbicide and rate interacted altering richness among herbicides by rates. No treatments differed from controls. Richness ranged from 7.7 species with picloram at .56 kg ai/ha to 11.2 species in the metsulfuron at .032 kg ai/ha treatment. Control richness was 9.17 species.

Discussion

Low species richness and relative abundance of coexisting species is typical in the presence of a highly efficient non-indigenous invader like sulfur cinquefoil (Rice 1993, Randall 1996). Based on temporal and spatial similarities among forbs and sulfur cinquefoil, I hypothesized that hand removal of this invader would increase species richness and diversity. Hand removing sulfur cinquefoil increased the total richness of the sites and total indigenous forbs were significantly higher in richness and cover. These findings are similar to those of Tyser and Key (1988) for spotted knapweed and Belcher and Wilson (1989) for leafy spurge. However, species diversity weighted by abundance (Shannon-Weiner index) was unaffected by removing sulfur cinquefoil. Hand removing sulfur cinquefoil did not increase grass richness, density or production after two seasons. I speculate that forbs may have benefitted more from the increased resource availability resulting from invader removal than grasses because forbs may occupy niches similar to that of sulfur cinquefoil (Sheley et al. 1996, Pokorny 2002).

Forbs are essential components of plant communities because they provide structure and function central to ecosystem sustainability. Pokorny (2002) emphasized the importance of considering indigenous forbs when making invasive plant management decisions because of their abundance. Forbs may also be critical because they preempt resource use by weedy species and provide a barrier to invasion (Pokorny 2002). In this study, species richness and diversity were not affected by any herbicide treatment, similar to that found by Rice et al. (1997). However, picloram reduced forb cover below that of the control at both sites regardless of rate. In addition, picloram, metsulfuron and clopyralid reduced total forb density at White Buffalo. Loss of important forbs may open niches and create safe sites for nonindigenous invasion (Levine and D'Antonio 1999, Dukes 2001, Pokorny 2002).

The limited impact of 2,4-D is consistent with Jacobs and Sheley's (1999) findings that 2,4-D can be applied at a phenological stage which will minimally affect total forb production. Since 2,4-D is somewhat effective in controlling sulfur cinquefoil and spotted knapweed by applying the herbicide later in the growing season, controlling these invaders while maintaining competitive forbs may reduce re-invasion. Increased niche occupation is believed to be a critical element necessary for native communities to enhance their resource capture and preempt acquisition by an invader (Pyke and Archer 1991, Tilman 1996, Jacobs and Sheley 1999).

From a cultural and medicinal perspective, certain individual species are also important when considering the influence of herbicides on plant community composition. Western yarrow and cudweed sagewort were affected by various herbicides. High rates of any herbicide decreased cover and, in many cases, density of yarrow below that of the low or medium rate of herbicides at White Buffalo. By the second year, this species' cover fully recovered. This species was also sensitive to high rates of herbicides in a study conducted in western Montana (Rice and Toney 1996). Interestingly, low herbicide rates increased cudweed sagewort biomass at Paddy Creek.

In this rangeland herbicide study, I hypothesized that species would respond independently to different herbicides and their rate of application with graminoid species capitalizing most on available site resources (Pechanec et al. 1965, Rice et al. 1997, Jacobs and Sheley 1999). As expected, the perennial grass group increased in cover and biomass at both White Buffalo and Paddy Creek over the two years. The herbicides that increased grass cover and biomass were different depending on site. My results suggest that the grass species response depended upon site and herbicide treatments, therefore, managers must consider all species in a community before selecting a herbicide (Sheley et al. 1996, Jacobs and Sheley 1999).

Herbicides can be very effective in increasing grass production (Dahl et al. 1989). Perennial grass cover and biomass increased following herbicide application. In many cases, herbicides more than doubled perennial grass production two years after herbicide treatment. This shift toward grasses may have occurred, in part, because of the increase in resources available after forb control. Most earlier studies focused on controlling indigenous forbs for increased grass production (Blaisdell and Mueggler 1956, Laycock and Phillips 1968, Tabler 1968, Schumaker and Hanson 1977, Miller et al. 1980). Increasing grass production by controlling both indigenous and nonindigenous species is currently a major focus of invasive plant management (Sheley and Petroff 1999). I believe that weed management must focus on restoring the function and structure of the plant community, rather than simply controlling weeds for grass production (Sheley et al 1996).

Improvement in range condition is typically a desirable outcome for range managers (Dyksterhuis 1949). In order to improve condition, the desirable decreaser species must recover and substantially increase their relative abundance. On my sites, improvement in range condition did not occur because many of the climax decreasers no longer are present. In fact, the use of herbicides, picloram in particular, allowed for increased abundance of Kentucky bluegrass and shifted the community composition further away from the potential natural community at Paddy Creek (Meuggler and Stewart 1980).

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CHAPTER 3

SEED SOURCE ISLANDS AS A MECHANISM FOR LANDSCAPE SCALE RESTORATION OF CULTURAL SPECIES

Introduction

For centuries, Northern Cheyenne people coexisted on the Great Plains relying entirely on their surroundings for food, shelter and medicine. Indigenous life was interconnected with the flora, fauna and natural cycles that perpetuated the tribe's existence. As Tall Bull (1980), a Cheyenne elder, described "For the earth is our mother and the plants and animals our relatives and friends." The innate understanding of indigenous plants and their role in the environment is best encapsulated in the traditional philosophy that "Every plant has its purpose." The indigenous people of the Great Plains recognized and accepted the dynamic fluctuations in populations and processes as integral occurrences necessary for the earth to sustain life for all species. These dynamic and diverse systems were sustainable in the absence of invasive non-indigenous species. However, as non-indigenous settlers dispersed and established in the west in the 1800's; so did other species of non-indigenous invasive plants many of which now inhabit millions of hectares throughout the Northwest (Sheley and Petroff 1999).

Non-indigenous invaders such as spotted knapweed (*Centaurea maculosa* Lam.), Russian knapweed (*Centaurea repens* L.), leafy spurge (*Euphorbia esula* L.), and sulfur cinquefoil (*Potentilla recta* L.) are some of the most efficient invaders inhabiting the Northern Cheyenne Reservation. Evidence suggests that these invaders reduce indigenous species cover, density, biomass and basal area over time (Tyser and Key 1988, Belcher and Wilson 1989, Kedzie-Webb et al. 2001). The introduction of non-indigenous invasive species has profoundly altered the structure and function of regional and global ecosystems (Vitousek 1986). Many displaced indigenous species have dietary, medicinal and/or ceremonial significance to the tribe. The loss of indigenous species is culturally and ecologically unacceptable to most tribes.

Unfortunately, the tribal landbase is already hosting substantial amounts of nonindigenous invaders. Management of these invasive species ranges from no management to intensive long-term herbicide control. To date, none of the management actions, including long-term herbicide use, have favorably preserved or restored cultural plant populations. Long-term impacts of herbicides on non-target or cultural species is dependent upon the type, rate and frequency of application (Chapter 2, Rice and Toney 1996). Tribal monitoring data suggest infested areas of the reservation, whether treated or not, have decreased in indigenous species richness and relative abundance compared to noninfested lands. Potentially, plant communities frequently treated with selective herbicides could shift into graminoid exclusive assemblies because of the exhaustion of the nonindigenous and indigenous forb species seed bank. Removal of distinct groups of indigenous forbs has potentially significant impacts on a plant community's ability to function and resist reinvasion (Pokorny 2002).

The reestablishment of desirable forbs and/or culturally significant shrubs has cultural importance, but may be essential for restoring community function and structure as well as reducing future invasibility; however many of these species are in limited supply, unavailable commercially, and/or available at high cost. This in turn makes landscape scale revegetation cost prohibitive. The development of low cost methods of reestablishing desired species into a community at landscape scales will be required to rebuild desirable indigenous species richness and abundance.

In order to implement this reduced cost patch style restoration, one must consider the complex fundamental factors which affect plant dispersal or invasion and how these factors influence the seed dispersal loop (Wang and Smith 2002). Critical elements such as size, shape and productivity of the seed source, distance to and availability of safe sites, combine with temporal environmental conditions and secondary dispersal vectors to determine seedling recruitment, survival, and ultimately the development of dispersed populations (Hobbs 1986).

The objective of this study was to quantify the establishment of islands or patches of culturally significant forb species in grasslands that were previously reclaimed following strip mining of coal. None of the culturally important species were part of the reclaimed community nor were nonindigenous invasive weeds. An additional objective was to test the impact of island shape on the invasion potential of selected indigenous species as part of a restoration effort. I hypothesized that desired species may more effectively establish and propagate from densely seeded rectangular islands over that of equal area square islands. Rectangular islands are narrower, reducing direct sunlight stress on seedlings and have greater edge length which increases opportunities to contact vectors. I hypothesized new occurrences of cultural species would occur in greatest abundance close to the source and decrease over a spatial gradient away from the source. New occurrences would establish throughout the landscape by utilizing natural primary and secondary dispersal processes over time.

Methods

Study sites

Field study sites were established in 1998 on the Western Energy Company reclaimed coal mine site. The study area was in reclaimed area "E" 2 km south of Colstrip, Montana (46 °30'N, 106° 25'W). The soils and plant communities present at both sites were anthropogenically altered by strip mining from 1980 to 1983. Topsoil was stockpiled from a previous disturbance in 1978 and redistributed onto sites one and two in 1990 and 1993, respectively. Subsoils consist of a deep homogenized overburden from a substrate that was brought to the surface and eventually backfilled and recontoured following coal removal. A 15 to 30-cm layer of stockpiled sandy loam was replaced as topsoil. Precipitation at Colstrip is bimodal and ranges from 305 mm to 380 mm with the majority received as snow and rain in April through June and again in September through November. The sites have a frost free period of 90 to 110 days.

Site one, the basin, is predominantly level (slope 0-2%) and is at an elevation of 945m. In 1990, the basin was reclaimed with one of two seed mixes depending on the site. Uplands were seeded with a blend including the following species and their proportion based on 100%: western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Love 19.8%), bluebunch wheatgrass (*Pseudorogenaria spicata* (Pursh.) A. Love 20.4%), green needlegrass (*Stipa viridula* (Trin.) Barkworth 21.9%), sideoats grama (*Bouteloua curtipendula* (Michx.) Torr. 2.1%), slender wheatgrass (*Elymus trachycaulus* (Link)

Gould ex Shinners l. 15.2%), Indian ricegrass (*Acnatherum hymenoides* (R. & S.) Barkworth 10.4%) and thickspike wheatgrass (*Elymus macrourus* (Turcz.) Tzvelev 10.2%).

Lowlands were seeded with equal parts: slender wheatgrass, western wheatgrass, Canada bluegrass (*Poa compressa* L.), Kentucky bluegrass (*Poa pratensis* L.) and streambank wheatgrass (*Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould). In 1991, a supplemental mix was incorporated on site which consisted of: blue grama (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths 43.6%), Canby bluegrass(*Poa secunda* J. Presl. 46.6%), purple prairie clover (*Dalea lasiathera* Gray. 3.0%), western yarrow (*Achillea millefolium* L. .9%), prairie coneflower (*Ratibida columnifera* (Nutt.) Woot. & Standl. 3.9%), Scarlet globemallow (*Sphaeralcea coccinea* (Nutt.) Rydb. 1.0%) and Lewis flax (*Linum lewisii* Pursh.1.0%). Species noted present in addition to seeded species include western ragweed (*Ambrosia psilostachya* DC) and yellow sweetclover (*Melilotus officinalis* L.).

Site two, the northeast slope, was located 1 km east of the basin and 3 km south of Colstrip, Mt. Physical and climatological data is similar at both sites. The site was recontoured with a 20° slope and was revegetated in 1993 with the Western Energy conifer mix consisting of: western wheatgrass (5%), bluebunch wheatgrass (9%), prairie sandreed (*Calamovilfa longifolia* (Hook.) Scribn. 15%), sand bluestem (*Andropogon hallii* Hack.18%), little bluestem (*Schizachyrium scoparium* (Michx.) Nash 22%), sideoats grama (24%), Indian ricegrass (5%) and prairie coneflower (2%).

Table 3.1. Livestock grazing history for site one, the basin, at the Western Energy Co. reclamation area "E."

Year	Animal Units	Grazing duration	Estimated Use
1998	86 yearling heifers (.8 AU) 4 bulls (1.2 AU)	May 28 through July 30	48%
1999	None	rested	0%
2000	84 yearling heifers 2 bulls	June 19 through July 31	44%
2001	84 yearling heifers 3 bulls	June 15 through July 31	64%
2002	None	rested	0%

Site one, the basin, was grazed in a rest rotation system (Table 3.1). Site two, northeast slope, was not grazed by domestic livestock at any time from seeding to present. The sites possess limited populations of indigenous forb species associated with grasslands in the region and were essentially devoid of the three species selected for seed source islands in this study. Neither site has record of nonindigenous invasive forbs or herbicide treatment. Wildlife observed near the sites include mule deer (*Odocoileus hemionus*), pronghorn antelope (*Antilocapra americana*) and various migratory birds.

Experimental design

The study was designed as a randomized-complete-block design with 3 replications at each of two sites. Treatments (3 species x 2 shapes) were established near the center of 64-m x 64-m plots. In the fall of 1998, the plot seed beds were tilled into either a 1-m x 9-m rectangle or a 3-m x 3-m square. One of three indigenous forb species were sown: purple coneflower (*Echinacea angustifolia* DC.) planted at 2,133 seeds/m^2, cudweed sagewort (*Artemisia ludoviciana* Nutt.) planted at 161 seeds/m^2 and breadroot scurfpea

(*Pendiomelum esculentum* (Pursh). Rydb.) planted at 161 seeds/m^2. Species were broadcast and seeds incorporated by raking in February of the following winter. All islands were covered completely with 2.5 cm thick permeable cellulose matting to improve moisture retention and reduce wind and water erosion on the islands. Matting was removed from all islands in July of 1999. Neither site was grazed during the establishment year (1999).

Sampling

Sampling began in 1999 by collecting establishment data. Data was collected using a randomly placed 20-cm by 50-cm frame to gather three subsamples per tilled plot. Seedling density for planted species was counted in each frame and the mean was used to estimate whole island density. Plots were sampled once per year in 1999 (when matts were removed), once in 2000 (mid June) when forbs were flowering, and again in 2002 at peak standing crop (September) to determine species demographics within the plots. In 2002, the islands were precisely mapped and the plots were sampled for new populations by conducting walking transects spaced 25 m apart across the extent of each study site. In 2002, highly visible flowered adults served as phenological markers for detecting new populations. Each flowered adult plant was documented with a Trimble Pro XR Global Positioning System (GPS) which is capable of logging data that is sub-centimeter accurate when post processed. Fifteen positions were logged for each flowered adult to ensure centimeter level accuracy. Populations that exceeded 1 m^2 were documented as species polygons by walking around the patch while the GPS unit logged one position per second.

Data analysis

Seedling density data were analyzed using ANOVA procedures for a split-plot-in-time to test the hypothesis that cultural species may establish better in rectangular tilled plots. (PROC GLM in SAS 1990). Year was considered the split-plot with species and shape as the whole-plots. Block*year*species*shape was the split-plot error term and block*species*shape was the whole-plot error term. When treatment or interaction were significant at the $P \leq 0.05$ level then mean separations were achieved using Fisher's protected Least Significant Difference (LSD) (Peterson 1985).

Spatial arrangements of raw data were downloaded from the GPS unit and post corrected in Pathfinder Office 2.51. Corrected data was then analyzed by utilizing ArcView 3.2a. The seed source islands, species point, and area themes were mapped. A customized ArcView 3.2a spatial analyst script was used to compute the distance between new species occurrences and their nearest seed source island. These occurrence data were then categorized into 10-meter bins originating from the source island to illustrate the occurrence gradient.

Results

Initial establishment and population trend

Seedling density depended upon site, species and year after seeding for the Colstrip experiment (Table 3.2). Breadroot scurfpea was the least successful to establish of the three species. Breadroot scurfpea did not establish in the basin plots and had only trace numbers of seedlings on the northeast slope. The number of breadroot scurfpea individuals slightly increased on the northeast slope over time from 2 plants/9m^2 (0.18%

of seed) in 1999 to about 12 plants/ 9m^2 (0.8%) in 2002.

Table 3.2. P-values for site, shape, species and year effect on seedling establishment density for cultural species seed source islands at Colstrip, Montana.

Source	df	P-value
rep	2	0.9428
site	1	0.0003
species	2	<0.0001
site*species	2	0.0132
shape	1	0.9722
site*shape	1	0.6359
species*shape	2	0.585
site*species*shape	2	0.0937
year	2	<0.0001
site*year	2	0.0015
species*year	4	0.0001
site*species*year	4	0.0355
shape*year	2	0.9281
site*shape*year	2	0.8034
species*shape*year	4	0.5182
site*species*shape*year	4	0.1361

Cudweed sagewort density depended upon site and year (Table 3.2). In the basin, cudweed sagewort established 165 plants/ 9m^2 (11.4% of seed) in 1999. Only about 25 plants/ 9m^2 (1.7%) survived into 2000 and 2002 (Fig. 3.1). On the northeast slope, cudweed sagewort established 748/ 9m^2 (51.6%) in 1999. Natural events reduced the populations to 58 plants/ 9m^2 (or 3.9%) by July of 2000. In 2002, the cudweed sagewort populations had rebounded to about 122 plants/ 9m^2.

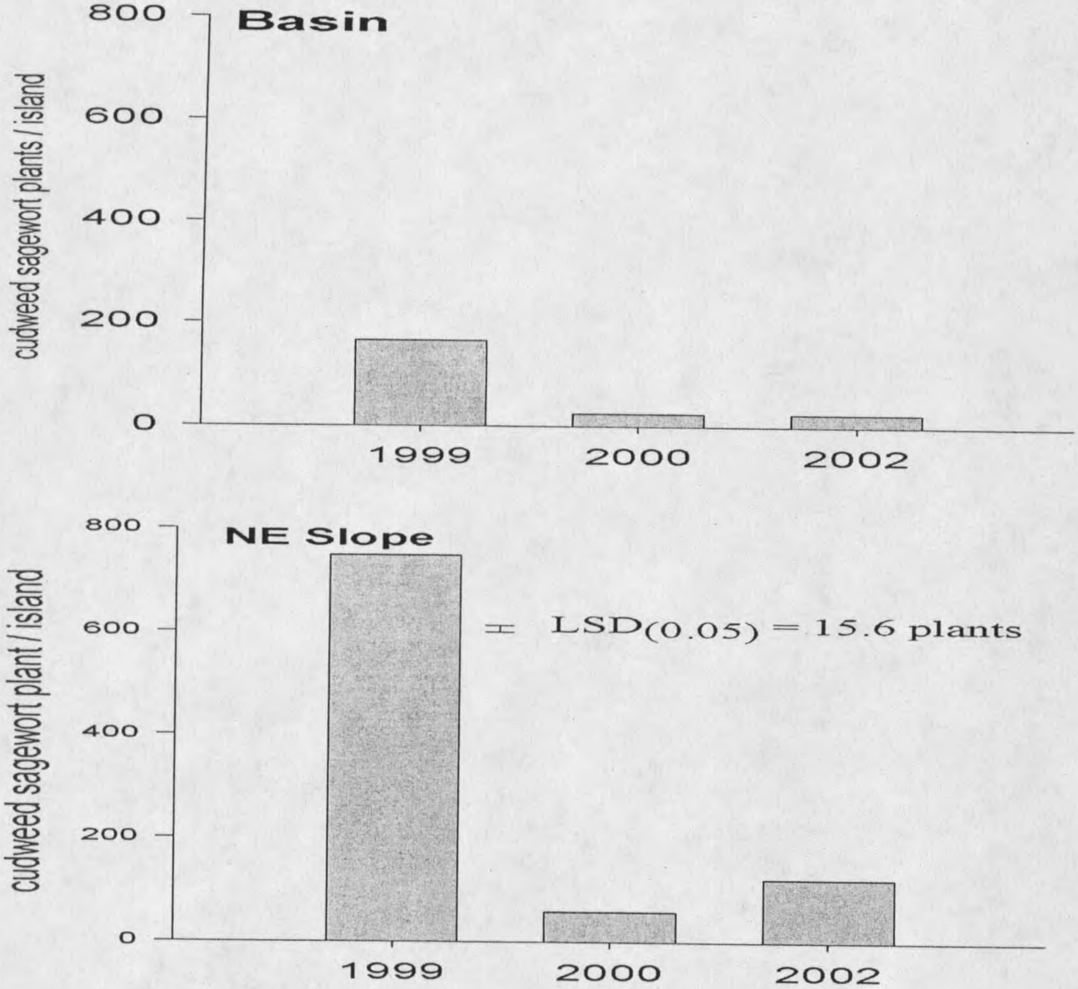

Figure 3.1. Cudweed sagewort island density for each year sampled. Bar signifies the Least Significant Difference.

Purple coneflower and cudweed sagewort were initially seeded at different rates. Their germination percent and establishment rates were similar when adjusted for the initial difference in seeding rate (Table 3.2). Purple coneflower established a mean of 2,120 plants/ 9m^2 (11% of seed) in 1999 (Fig. 3.2). Natural occurring conditions thinned the populations to 183 plants/ 9m^2 (1%) and 115 plants/ 9m^2 (< 1%) in 2000 and 2002, respectively. On the northeast slope, purple coneflower established densely with a mean of 5,595 plants/ $9m^2$ (29% of seed) in 1999. These populations also decreased by natural mortality to 595 plants/ $9m^2$ (3.1%) and 422 plants/ $9m^2$ (2.2%) in 2000 and 2002, respectively.

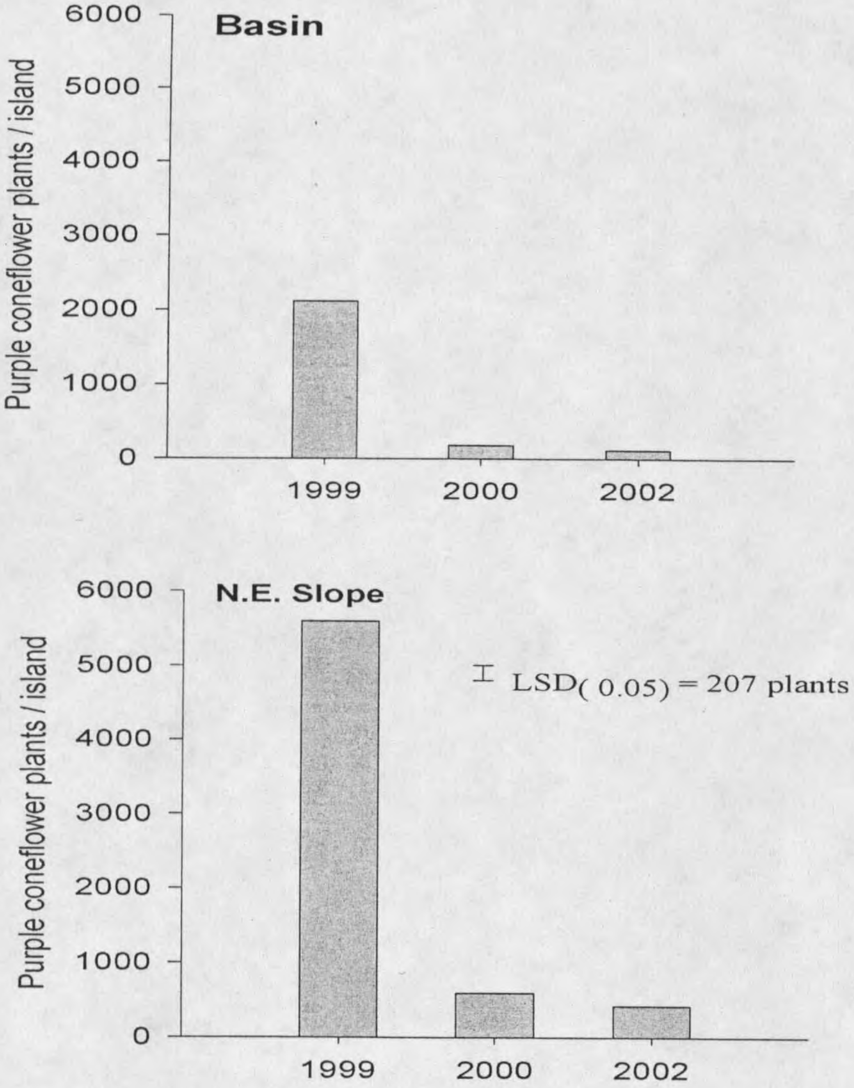

Figure 3.2. Purple coneflower island density for each year sampled. Bar signifies the least Significant Difference.

Landscape scale occurrence and recruitment

Cudweed sagewort did not flower in 1999 or 2000. Consequently, seeds were not available for primary dispersal and seedlings were not detected within a 10-m radius around the islands at any observation time. By 2002, cudweed sagewort dispersed to 3 new locations in the basin all of which were nearest to square seed source islands (Fig. 3.3). On the northeast slope, cudweed sagewort was detected in 8 new locations within the project area (Fig. 3.4). New populations at both sites were not abundant enough to distinguish any dispersal pattern.

None of the purple coneflower matured to a reproductive state in the islands at either site. Consequently, seeds were not produced within the islands in 1999 and 2000. No seedlings were detected within a 10-m radius around any island for the same period. New recruitment was not evident until 2002. In 2002, 138 flowering purple coneflower adults occurred throughout the basin (Fig. 3.5). Each adult was associated with a 1 to 15 juveniles in close proximity. Juveniles were not inventoried.

The adult inventory was coded to the nearest island and the measurements were organized into increments of ten meters from the nearest island. Populations associated with rectangular islands appeared to establish closer to the source than square islands. Populations associated with rectangular islands occurred more frequently between 10 and 40 meters (Fig. 3.7). Populations associated with the square islands occurred more frequently in the 40 to 70-m range in the basin (Fig. 3.7).

Dispersal patterns were not evident on the northeast slope. Only 8 purple coneflower adults were located on site, six of which were correlated to rectangular islands (Fig. 3.6).

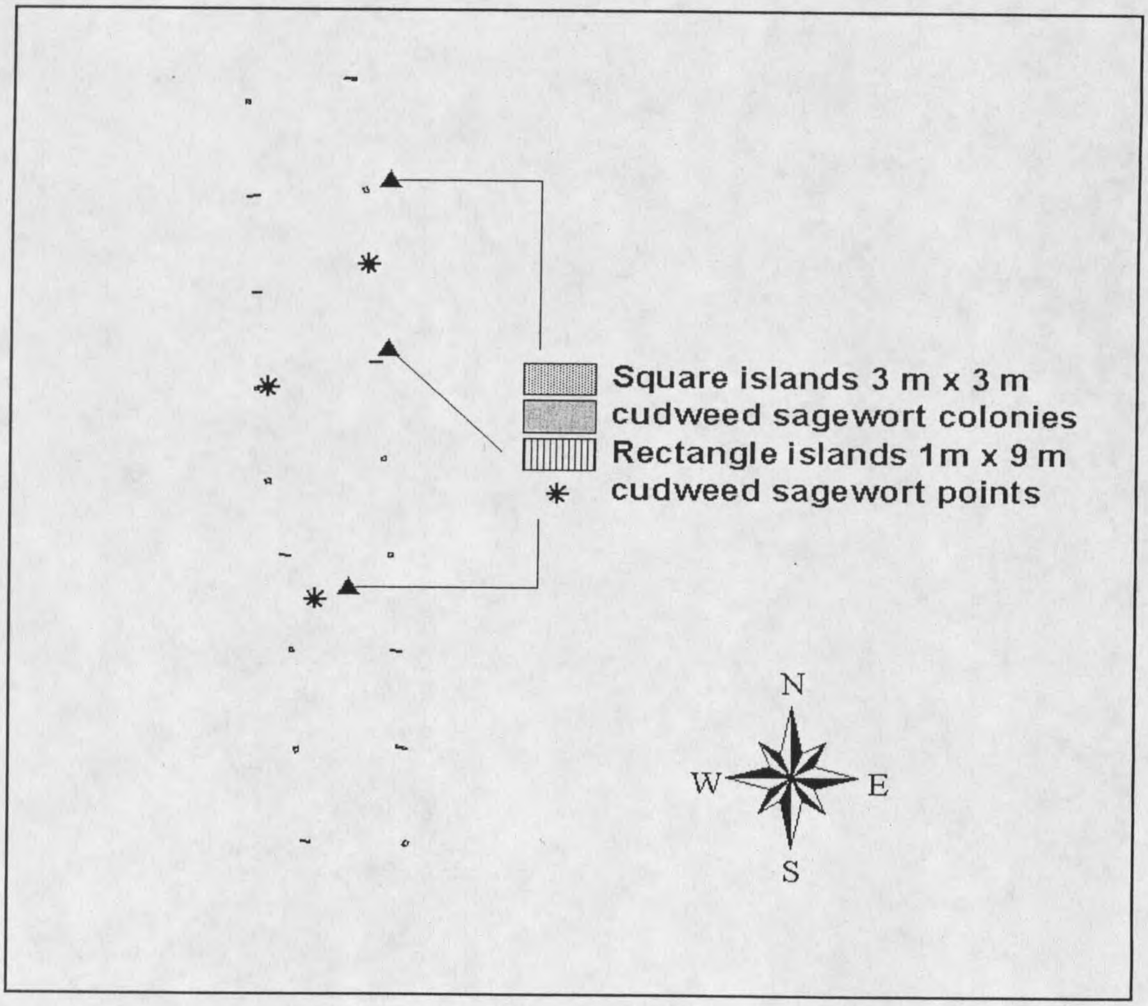

Figure 3.3. The Basin. Spatial illustration of cudweed sagewort occurrence in 2002.

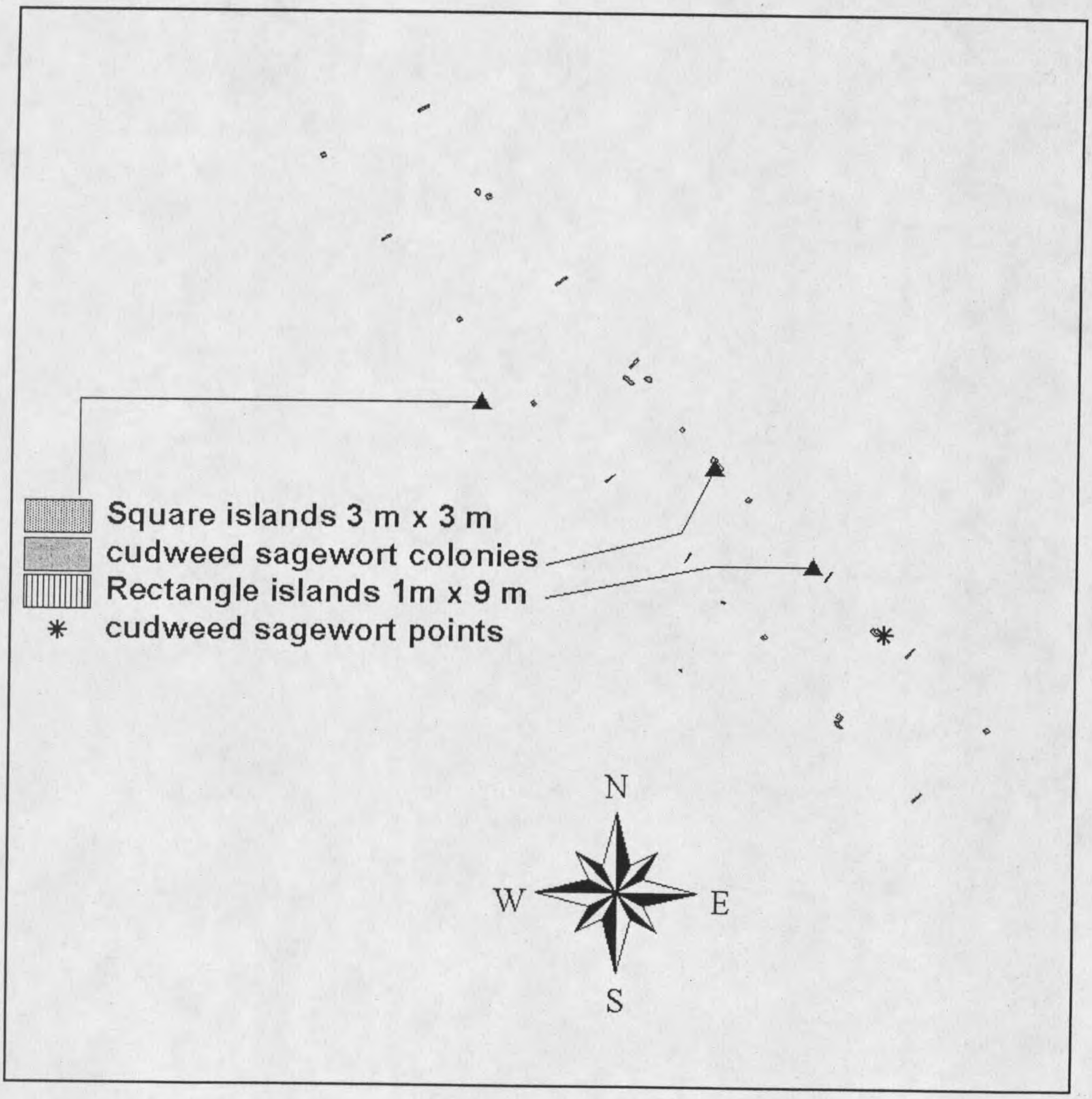

Figure 3.4. The N.E. slope. Spatial illustration of cudweed sagewort occurrence in 2002.

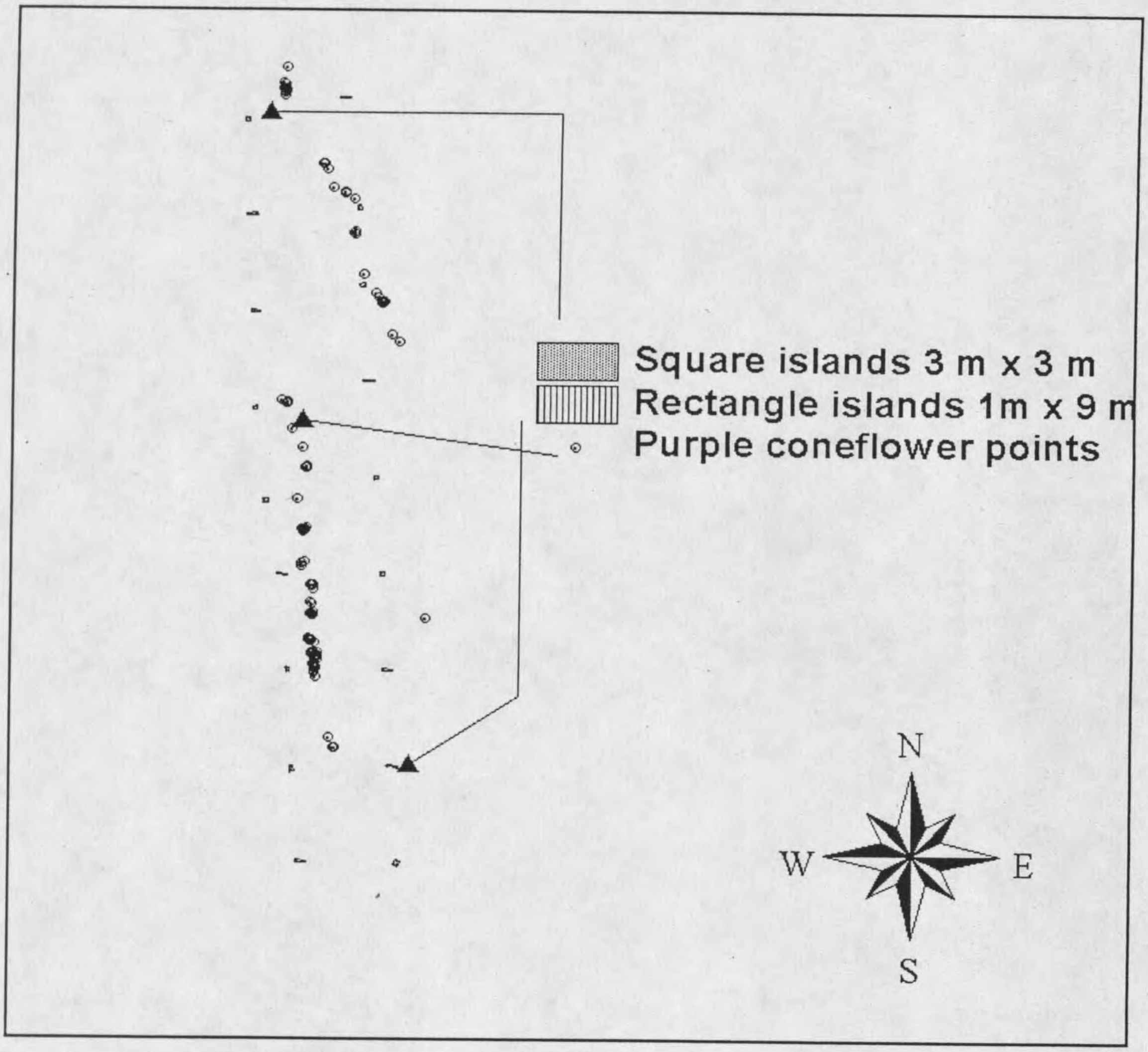

Figure 3.5. The Basin. Spatial illustration of purple coneflower occurrence in 2002.

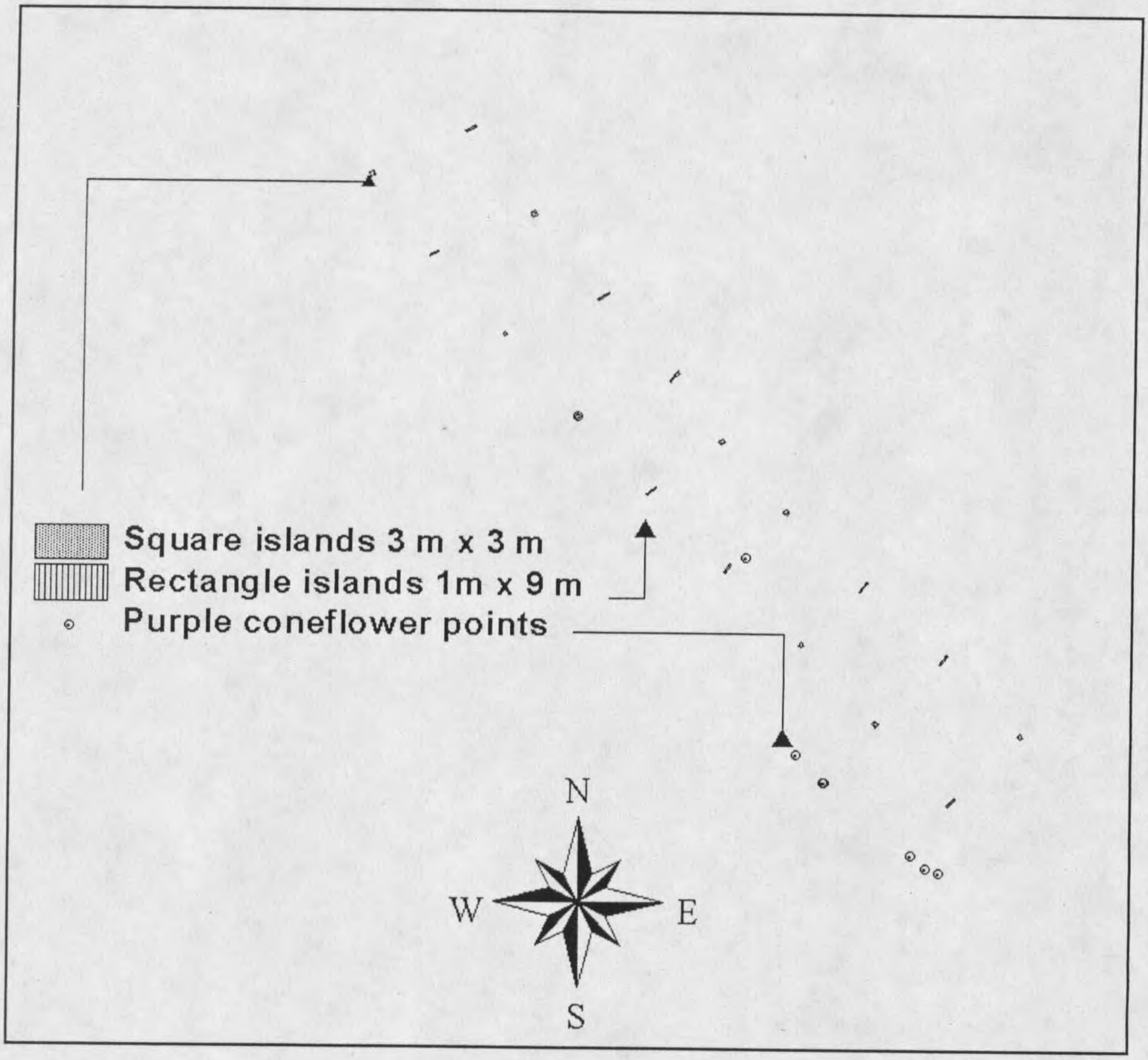

Figure 3.6. The N.E. Slope. Spatial illustration of purple coneflower occurrence in 2002.

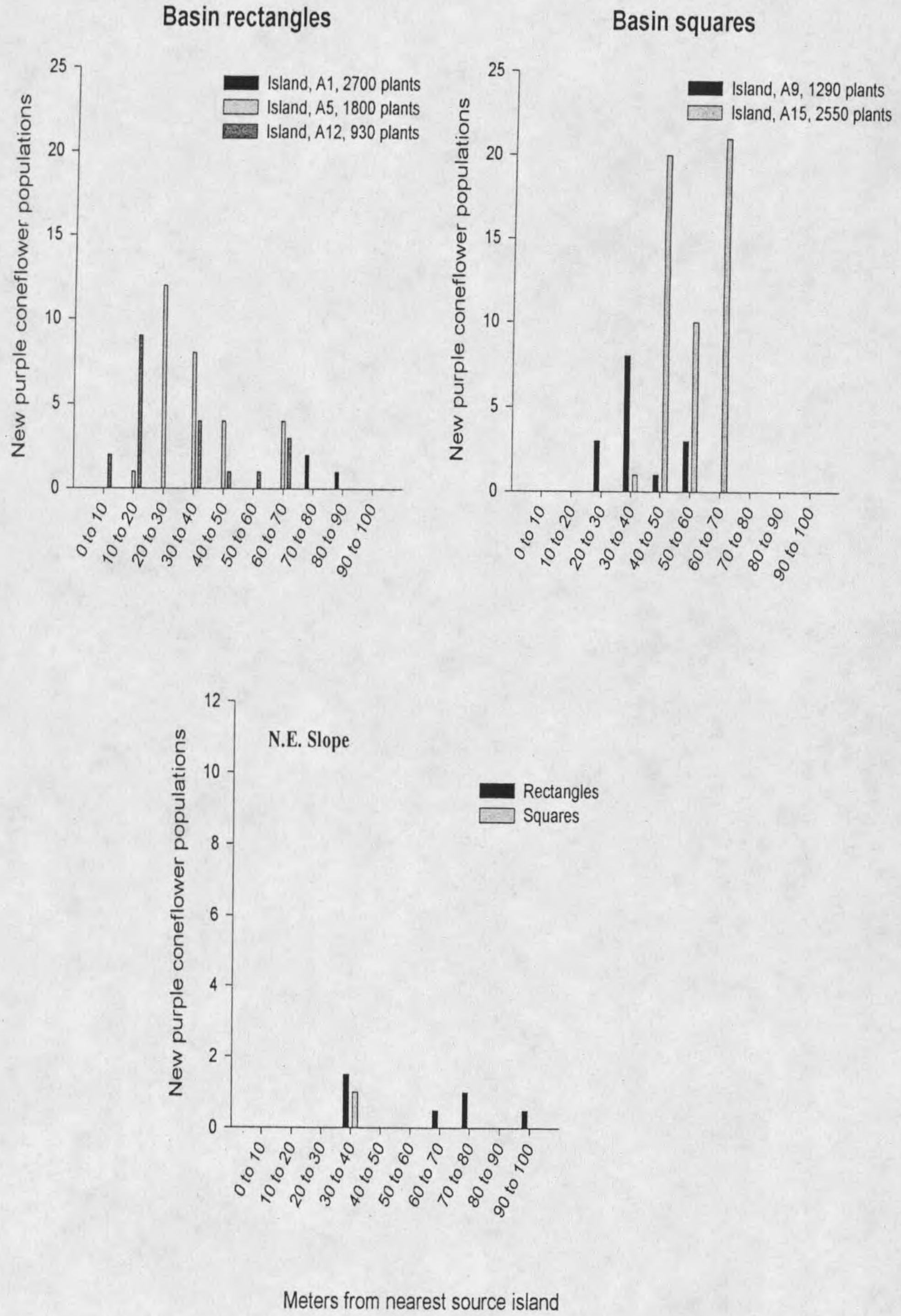

Figure 3.7. Occurrence of purple coneflower populations from the nearest seed source island by shape in the basin compared to new populations by shape on the N.E. slope. Legend defines island identity and initial plant density in 1999 for the basin.

Discussion

Anthropogenically created monocot-dominated rangeland may ultimately be the outcome of long-term high intensity medicinal plant harvest, the long-term use of herbicide for treatment of invasive species, or indigenous forb control for increased forage production (Hurlburt 1999, Rice and Toney 1996, Harris and Cranston 1979). In either case, human influence can change the resultant structure and function of the plant community significantly from the model or reference community (Rejmanek 1996, Vitousek 1986). The inherent dynamics of communities combined with physical disturbance can make restoration to pre-disturbance richness and diversity difficult, if not impossible (Lockwood 1997). The goal of land managers and reclamation professionals should be to use the best methods available to assemble communities to achieve sustainable structure and function in order to meet long-term land use objectives (Sheley et al. 1996). Since non-indigenous species are a formidable factor influencing the sustainability of plant communities, improving community resistence to future invasion is key in slowing or stopping the invasion by non-indigenous invaders (Pokorny 2002).

In many cases, reclaimed mine areas epitomize anthropogenic plant community composition following drastic perturbation. The grass component is often well-established, dominated by cool and warm season species. Because of environmental patterns within Montana, cool season species tend to dominate (Munshower 1994). Indigenous forbs are limited to the low number of species available at time of seeding. Natural forb regeneration is generally limited because of the extended period of topsoil storage,. Since diverse functional group representation was shown to reduce invasibility to species like spotted knapweed (Pokorny 2002), introduction of indigenous functional species during reclamation may reduce future invasibility of the community by improving the community's structure and function. It is also very important to reestablish cultural species across the landscape to preserve tribal traditions.

I hypothesized that the establishment of densely seeded islands across the landscape could be utilized as a low cost mechanism for reestablishing species at this scale. My objective was to substitute time and natural mechanisms of primary and secondary dispersal in place of high cost and labor associated with traditional seeding methods. The establishment of cudweed sagewort, breadroot scurfpea and purple coneflower varied between sites and among species. After adjusting for initial seeding rate differences, dormant broadcast seeding after tillage established cudweed sagewort and purple coneflower similarly in the first growing season. Purple coneflower produced more seedlings because it was seeded at a higher rate. Conditions at site two, the northeast slope, were apparently most conducive to improved germination and emergence. This leads me to speculate that the relatively wetter and cooler conditions on the northeast slope improved seeding success. These findings are similar to those of Wartidiningsih et al. (1994) who observed that purple coneflower requires cold stratification and establishment under controlled conditions to achieve higher than natural levels of germination. Direct seeding produces non-uniform populations.

Mid-summer conditions during establishment were typically extreme, reaching daytime highs in excess of 100° F (37.8 ° C), and plots received only trace amounts of precipitation during the summer months. Late-season observation indicated that all seedlings were senescent and possibly lost to drought stress. In 2000, early sampling verified that the island populations had thinned, but low densities survived. I believe use of well-adapted indigenous forb species, such as cudweed sagewort and purple coneflower, were crucial to survival and persistence of these field planted populations. *Echinacea* and *Artemisia* species are winter hardy [-13 to -40° F (-25 to -40° C)] and drought resistant which provides for species resilience (McGregor 1968, Chapman and Auge 1994).

Seed production and possibly island shape were hypothesized to be a critical element in the dispersal, establishment, and long-term occurrence of culturally important species. None of the test species flowered in any islands during the first three years and shape did not influence establishment. These findings are consistent with Thomas' (1998) observations in which he characterized purple coneflower as rarely flourescing prior to the third year. Further sampling did not identify occurrence of cultural test species around the islands. At this point, I believed that more time would be necessary for the plants to reach the productivity and seed output necessary for initiating new populations.

Not until late-summer of 2002, the optimal time to view flourescing adults (McGregor 1968, Hobbs 1994), did inventory reveal signs of successful flowering and potential dispersal. At this time colonization was found throughout the basin while the northeast slope had only 8 new adults. Interestingly, dispersed populations exhibited similar phenology to the island populations. I had expected that the plants would establish along a dispersal gradient with the greatest quantity of seedlings being close to their respective island and exhibiting traits of recent emergents. However, the majority of purple coneflower occurred most often between 10 m and 70 m and of sufficient maturity to flower. Hurlburt (1999) found that purple coneflower recruitment was not always strongly correlated to source populations exhibiting high flower counts. Purple coneflower may be adapted to utilize some forms of secondary dispersal of hard seed.

After plotting the spatial arrangement, I speculated that some site variability or secondary form of dispersal was responsible for the establishment of the new populations. Island shape may have had some effects on dispersal distance. It seemed that new populations occurred relatively closer to rectangular islands. Aspect, species composition, residual biomass and management could also be responsible for substantial differences in colonization.

Purple coneflower is believed by some to be sensitive to grazing (Eddy 1990, Weaver and Hansen 1941) due in part to predation of seed heads (Lesica 1995, Hurlburt 1999). Intensive spring grazing by livestock is thought to be partially responsible for the scarcity of purple coneflower in the Flint Hills of Kansas (Eddy 1990). Conversely, my findings indicate that grazing intensively for short periods of time may have aided long-range dispersal of hard seed (Table 3.1). It is interesting that the slope, which produced higher initial establishment and received no livestock grazing, has substantially less establishment across the landscape.

The role of herbivores in species dispersal is largely dependent upon herbivore size, management, and the scale at which they are studied or managed (Olff and Ritchie 1998). Even though herbivores may predate seed heads and can be managed to overuse desirable species, they also have potentially positive influences with regard to consumption of other competitors, especially cool season grasses. Reducing competition may be compounded with the digging and trampling which create safe sites ultimately improving dispersal success of community propagules (Grubb 1977, Fischer et al. 1996).

The art and science of reclaiming, rehabilitating, restoring or assembling post disturbance communities is becoming more important nationally. For the Northern Cheyenne, the connection to our dynamic, yet sustainable indigenous environment is simply fundamental to the cycle of life. Unfortunately, non-indigenous invaders are well-established and dispersing throughout our homeland. These species not only threaten the structure and function of our system (Mack and D'Antonio 1998) they also threaten to out-compete and displace species critical for Cheyenne sustenance, medicine and ceremony (Tyser and Key 1988). These infested areas are a high priority for control and eradication, but most methods threaten the desirable species as well as the target species (Pokorny 2002). New strategies are needed that cost-effectively increase diversity and invasion resistance. Maintenance or addition of functional forbs appears to be fundamental to keeping communities and the species that rely on them vigorous (Pokorny 2002). Island seeding may have potential to provide a low input method combining time, dense seed supply and natural dispersal mechanisms to increase species frequency across disturbed landscapes. This method may be most applicable for landscapes with topography and/or a location that limit the use of conventional rehabilitation methods.

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