

THE EFFECTS OF PRESCRIBED FIRE ON RIPARIAN GROUNDWATER

by

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ABSTRACT

The hypothesis of this study is that the use of prescribed fire to reduce trees and tree canopy, on a watershed scale, will decrease depth to riparian groundwater, increase riparian plant species diversity, and increase riparian biomass production. To test these hypotheses two watersheds, primarily managed for cattle grazing, located in Fergus (Dry Armells watershed) and Jefferson (Little Whitetail watershed) Counties, Montana, were chosen. Both watersheds were densely dominated with coniferous tree species. Average preburn tree density was 1,276 trees/ha and 350 trees/ha for both the Dry Armells and Little Whitetail sites respectively. Ten riparian drainages were selected for treatment and analysis within the two watersheds. Six of these drainages were burned and four were used as unburned comparisons. Prescribed fires took place in the spring of 2001 at Dry Armells and in the fall of 2005 and spring of 2006 at Little Whitetail. At Dry Armells 18 to 20% of the watershed area was burned. At the Little Whitetail site only 3 to 15% of the watershed area was burned. At Dry Armells, average depth to riparian groundwater significantly decreased from 122 cm below grade to 79 cm below grade after one year, and monitoring continues to show an average annual decrease. Riparian groundwater at Little Whitetail has yet to respond. Plant species diversity did not significantly increase in the post burned riparian areas at either of the sites. Plant species diversity did increase, post burn, in the Dry Armells uplands. However, there was no increase in species diversity, post burn, in the Little Whitetail uplands. In the riparian areas (Dry Armells) that had the strongest response in groundwater, 14% more biomass was produced in 2005 as compared to their unburned counterparts. Benefits of this prescribed fire are enhanced riparian zone process and function, increased biodiversity, and possibly, restoration of perennial streamflow in systems that were previously ephemeral. Management implications based on these benefits could take the form of using fire to increase groundwater and stream discharge, or at the other extreme, using prescribed fire for total ecosystem rehabilitation.

CHAPTER 1

INTRODUCTION

Riparian ecosystems are “the interfaces between terrestrial and aquatic ecosystems. As ecotones, they encompass sharp gradients of environmental factors, ecological processes, and plant communities. Riparian zones are not easily delineated but are composed of mosaics of landforms, communities, and environments within the larger landscape (Gregory 1991).” In the simplest terms they are the catch basins for everything being weathered, eroded, or released (precipitation, soil, detritus, etc.) within a given area (watershed). Riparian ecosystems are highly diverse because as they proceed across the landscape, following rivers and streams of a particular drainage, they gain influence from the surrounding environment and ultimately reflect or resemble all the attributes of the entire watershed. Due to the magnitude of different geologic materials that exist across the landscape it is apparent that no two riparian areas are likely the same and that their biotic diversity must be linked to abiotic diversity. Riparian ecosystems are generally more diverse (species wise) than the environments that craft them, have shallow water tables, are always in a state of flux (primarily due to hydraulic influences) and in the last 35 years or so have been referred to as the single most productive type of habitat, benefiting the greatest number of species (Ames 1997; Hubbard 1977; Patton 1977).

The least obvious component of a riparian ecosystem that enables it to function is, arguably, groundwater. Studies in both Montana (Law et al. 2000) and Arizona (Stromberg et al. 1996) both concluded that riparian communities can persist if depth to

groundwater does not exceed 2 m throughout the year. One prime example of this type of riparian ecosystem is the typical Montana coulee bottom. Flow in this type of drainage is generally limited to the runoff season (late winter to late spring), yet, many facultative/wetland and obligate species inhabit the bottom. If depth to groundwater does drop below 2 m, for whatever reason, the coulee bottom communities therein may lose obligate species, become more characteristic of upland community types, and subsequent flow thereafter becomes classified as “periodic overland flow.” Ultimately, this means loss of riparian area and places more stress on any remaining riparian communities by encroaching livestock and big game.

There are many factors that contribute to groundwater depression that can ultimately lead to the loss of riparian communities. From the largest to smallest scale the possible causes can be change in climate, tectonic uplift, groundwater pumping for potable supply and irrigation purposes, and evapotranspiration. Climate and tectonics are controlled by forces too great to manipulate and groundwater pumping, for the most part, takes place in localized areas. This leaves evapotranspiration as the most widespread process affecting riparian areas. Generally speaking reduction of groundwater occurs when the rate of evapotranspiration exceeds the rate of infiltration. If this imbalance is sustained groundwater recharge can be suppressed (Huxman 2005). This phenomenon may be occurring in many western coniferous forests due to above normal tree densities. Each tree in a forest asserts a certain amount of stress on the water table to maintain growth. As a forest increases in density a threshold may be crossed and the collective amount of stress exerted becomes greater than the rate of recharge or groundwater

recovery. Groundwater is not eliminated by this process (forest would die altogether), however it can become unavailable to shallow rooted plants. When riparian communities lose connectivity with groundwater they begin to unravel (Grover and Musick 1990). One way to reverse this type of riparian degradation is to thin overgrown forests to pre-European settlement levels. The purpose of this paper is to analyze riparian response to groundwater release, following prescribed fire, within two overgrown watersheds located in Montana.

CHAPTER 2

LITERATURE REVIEW

Due to an anthropomorphic based disequilibrium in the natural fire regime of many western forests, watersheds in these forests have overly dense stands of trees (Allen et al. 2002). The natural fire recurrence interval (pre-European settlement) for western ponderosa pine forest ecosystems has been reported to range from 5 to 30 years (Arno 2000). Due to current land agency suppression policy this interval has increased to 100+ years, or 3 natural cycles of fire (Brown et al. 1999). For example, prolonged fire suppression in many southwestern U.S. ponderosa pine forests has allowed tree densities to increase from 50 to 148 trees/ha, to 741 to 2,224 trees/ha in the last 100 years (Paige 2000). This magnitude of change sets the stage for two things: a steady decline in herbaceous and understory diversity due to competition for groundwater and reduced light, and catastrophic wild fire due to unnaturally high fuel loads.

Stream discharge is one of the primary forces that maintain and perpetuate riparian species health and diversity (Bendix and Hupp 2000). Discharge, in turn, results from surface and subsurface flows. Surface flow, when entering a stream, occurs as a sudden surge in discharge immediately following a rain event. Subsurface flow, provided by groundwater, enters a stream much slower and is the primary force in maintaining flow (also referred to as baseflow) until the next precipitation event. Extremely dense forests can affect streamflow in two ways; by foliar interception of precipitation, which

prevents water from reaching the soil, and community transpiration, which reduces groundwater and baseflow.

Interception occurs when precipitation reaching the upper surface of the forest canopy is evaporated back into the atmosphere directly (Ehterington 1975). A study in the Pacific northwest (Rothacher 1963; Spittlehouse 1998) reported, “that nearly all rain from small storms may be intercepted and lost, particularly from the dense canopies of old-growth forests, but the fraction of rain fall lost to evaporation decreases as storms increase in magnitude and intensity. Interception also occurs in the winter. Storck et al. (2002) found that seasonal losses to sublimation of intercepted snow were approximately 100 mm, compared to total winter precipitation of approximately 2 m (water equivalent). This suggests that heavily forested areas with infrequent low intensity rainfall patterns (i.e. arid southwest) can lose substantial amounts of precipitation to interception. This ultimately leads to less water being available for streamflow. Community transpiration can also negatively affect streamflow. For example, a mature loblolly pine (*Pinus taeda* L.) community (approximately 1000 trees/ha) can remove approximately 63,607 l/ha per day (Zahner 1955). Changes in tree density can have a profound effect on groundwater recharge. Thus, as forest stands become dense under fire suppression shallow groundwater may decline and fail to recharge. This dewatering phenomenon can result in rivers and streams becoming ephemeral, a decline of shallow rooted forb and herbaceous species, and a total loss of riparian ecosystems (Stromberg et al. 1996). Further consequences associated with this type of ecosystem degradation are increased erosion, declines in forage productivity, and loss of biodiversity (Grover and Musick 1990).

Numerous studies in the last two decades, (Arthington et al. 1991; Castleberry et al. 1996; Hill et al. 1991; Johnson et al. 1976; Richter et al. 1997; Sparks 1995; Stanford et al. 1996; Toth 1995; Tyus 1990) that focused on human based alterations to the natural flow regime of streams and rivers, have resulted in, “a well grounded scientific perspective on why altering hydrologic variability in rivers is ecologically harmful (Poff et al. 1997).” However, less work has been placed on indirect human caused alterations to the natural flow regime; for example excessive tree density (watershed scale) arising from disrupted fire return intervals.

Removal of trees whether by mechanical means or by fire, has been associated with increased stream discharge (Moody and Martin 2001; Moore and Wonzell 2005; Swank et al. 1989; Ursic 1970). For example, in studies conducted in the Pacific Northwest, low flows were found to be less extreme (stronger base flow component from groundwater) in test streams for up to 16 years after the mechanical removal of conifer tree species (Moore and Wonzell 2005). Further south, in the Sierra Ancha Mountains of Arizona, a low intensity wildfire (low mortality of canopy species) in a mixed ponderosa pine forest produced stream flows following a 15 minute 66 mm/hr storm that overtopped streamflow gages. Peak discharge was estimated to be 7 times greater than any similar storm event recorded during the previous 30 years (Neary et al. 2003). However, the degree of response was highly variable between different watersheds burned during the fire. According to Neary et al. (2003) the magnitude of surface water yield is controlled by several factors such as fire intensity, climate, precipitation, geology, soils, watershed aspect, tree species, and proportion of forest removed (by logging and or fire). As a

result, prediction of surface water yield becomes very difficult. Although groundwater was not directly monitored in the examples above, it can be implied that it did increase. This implication is supported by noted decreases in the extent of low flow periods (stronger baseflow component provided by elevated groundwater) after tree removal (Moore and Wonzell 2005). The linkage between hydrologic conditions and the structure and composition of riparian vegetation suggests that as surface and subsurface flows increase, changes in the riparian community should follow.

Rapid reestablishment of ground cover after tree removal was noted in several experimental watersheds (Brockway and Lewis 1997, Ursic 1970). According to Miller (2000) in post burn environments the reestablishment of groundcover species is dependant on severity of fire. Low intensity fires that only consume some of the surface fuels have little effect on most buried plant parts and can stimulate significant amounts of post fire sprouting (Miller 2000). A study by Brockway and Lewis (1997) in a longleaf pine/wiregrass ecosystem (Georgia) found increases in ground cover as well as species diversity post-burn. The standing biomass of pineland threeawn (*Aristida stricta* Misch.), Curtis' dropseed (*Sporobolus curtissii* Scribn.), bluestem (*Andropogon spp.* L.), and all other grasses and forbs increased significantly following the Georgia prescribed fire.

Lastly, healthy well-managed riparian zones have been shown to mitigate most anthropomorphic and climatic impacts within a watershed (Norris 1993; Naiman and Decamps 1997). They serve as nutrient sinks for non-point source pollution, combat stream-bank erosion, and help filter laminar flow sediments before they enter stream channels. This protects water quality and preserves the overall health of receiving

streams and rivers (Brown and Krygier 1970; Peterson et al. 1992; Norris 1993; Osborne and Kovacic 1993). Improving impaired riparian ecosystems will increase their assimilative capacity. It is also important to manage these floodplains, from the land owner's point of view, to optimize forage and cattle production. If land owners' could efficiently utilize the great amounts of forage in these riparian floodplains without impacting riparian ecosystem buffer function, not only will they increase profits (break cycle of farmer indebtedness), they will also help protect the environment and esthetic attributes of the landscape around them.

Based on a review of pertinent literature the hypothesis of this study are: 1) the reduction of tree density, using low intensity prescribed fire, in ponderosa pine and Douglas fir (*Pseudotsuga menziesii* Franco) dominated watersheds will result in decreased depth to riparian groundwater and, 2) the decreased depth to riparian groundwater will increase species diversity and biomass production in the post burned riparian zones.

CHAPTER 3

MATERIALS AND METHODS

Two sites, both primarily managed for cattle grazing, were chosen for this study (Figure 1). One site is located in the Dry Armells Creek watershed 40 km north of Lewistown, Montana in Fergus County. The second site is located in the Little Whitetail Creek watershed 16 km north of Whitehall, Montana in Jefferson County.

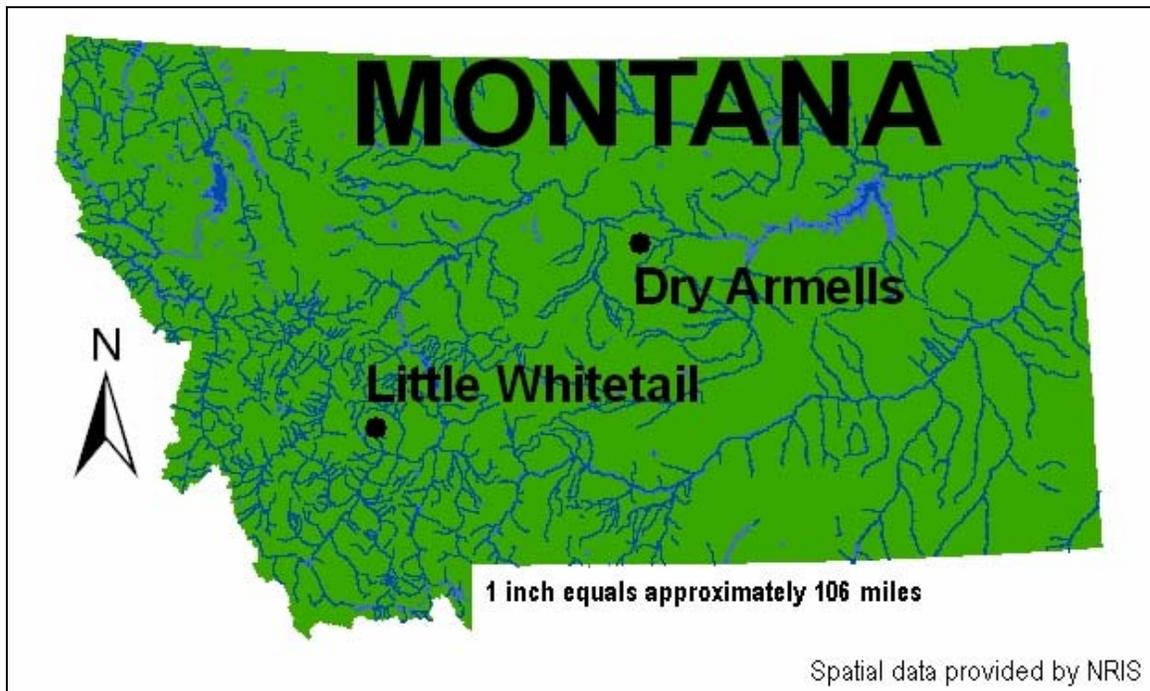


Figure 1. Location of study sites used to monitor the effects of prescribed fire on Ponderosa pine (Dry Armells) and Douglas fir (Little Whitetail) dominated riparian areas.

The sites will be referred to as “Armells” and “Whitehall” henceforth. Both sites were chosen in conjunction with Bureau of Land Management (BLM) plans to reduce hazardous fuel levels. Both sites were treated and monitored under the same protocol,

but due to geologic differences (Table 1) and time since burn they were analyzed separately.

Table 1. Some physical and hydrologic attributes of the research sites.

Study Site	Annual Precipitation	Mean Elevation	Geology	Aspect	Drainage Slope
Dry Armells	350 mm	554-1354 m	marine shale	NNW	< 1 %
Whitehall	280 mm	1585-1770 m	granitic boulder batholith	EES	1 – 3 %

Site Descriptions

Dry Armells

The Dry Armells site is located north of Lewistown, Montana, and represents a ponderosa pine savanna. The 777 ha study area is nested in the Dry Armells Creek watershed and receives approximately 350 mm of precipitation annually (WRCC 2006). The dominate tree species is ponderosa pine (mean age = 82 years). Upland soils represent the Dilt Series soils (Clark 1988). These soils are shallow (0.25 m -0.50 m deep) with low permeability and low water holding capacity. Consequently, runoff and erosion hazard are high. According to Veseth and Montagne (1980) the typical forest understory for this soil type is rocky mountain juniper (*Juniperus scopulorum* Sarg.), bluebunch wheatgrass (*Pseudoroegneria spicata* A. Love), prairie sandreed (*Calamovilfa longifolia* Scribn.), elk sedge (*Carex garberi* Fern) and chokecherry (*Prunus virginiana* L.). The pre-burn inventory (Marlow 2002) of upland conditions revealed areas that contained tree densities (trees > 1.5 m tall) as high as 2,200/ha (average density = 1,276

trees/ha). There was virtually no understory stratification in the forest, but when present consisted of ponderosa pine saplings. Saplings (trees < 1.5 m tall) in some areas were found in excess of 17,000 trees/ha (average density = 8,233 trees/ha). The 3 most abundant groundcover species found in the Armells uplands were western wheatgrass (*Pascopyrum smithii* A. Love), elk sedge and Rocky Mountain juniper (Marlow 2002).

Riparian areas at this site have little or no defined channels and the transitional zone between upland and riparian areas is almost nonexistent. The drainage bottoms are uniform in topography (even surface from bank to bank) and have mild slopes less than 1 %. Ponderosa pine, big sagebrush (*Artemisia tridentata* Nutt.), rabbit brush (*Chrysothamnus spp.* Nutt.), and Rocky Mountain juniper create a sharp delineating edge along the graminoid dominated riparian areas.

The riparian soils are representative of the *Marvan* soil series (Veseth and Montagne 1980). These soils developed in deep clayey alluvium on level to moderately sloping landforms adjacent to shale uplands. The soils are well drained, have very slow permeability, and a high shrink-swell potential. *Marvan* soils have an effective rooting depth of 1.5 m. The potential plant community (Veseth and Montagne 1980) for this soil series is western wheatgrass, green needlegrass (*Nassella viridula* Barkworth), bluebunch wheatgrass and thickspike wheatgrass (*Elymus lanceolatus* Scribn. & J.G. Sm.). However, the 3 most abundant herbaceous species recorded (Marlow 2002) in the Armells riparian areas were redtop bent grass (*Agrostis gigantea* Roth), Baltic rush (*Juncus arcticus* Willd.) and Kentucky bluegrass (*Poa pratensis* L.). No trees or shrubs were found growing within the boundaries of these riparian areas.

Whitehall

The Whitehall site, located in and around the Little Whitetail Creek watershed 16 km north of Whitehall, Montana, is dominated by a Douglas fir/quaking aspen (*Populus tremuloides* Michx.) forest type. The 1,280 ha site receives approximately 280 mm (WRCC 2006) of precipitation per year. The dominate species of the uplands, determined by a pre-burn inventory, were Douglas fir (mean age = 86 years), and Rocky Mountain Juniper (Mean age = 56 years). The pre-burn inventory also revealed tree (trees > 1.5 m tall) densities as high as 700 trees/ha (average density = 366 trees/ha) and sapling (trees < 1.5 m tall) densities, in the same area, as high as 4,400 trees/ha (average density = 2,337 trees/ha).

The soils of this area represent the Boulder Batholith Woodgulch series (Veseth and Montagne 1980). This series consists of deep soils formed in course residuum and colluvium weathered from granitic rock on moderately sloping to steep mountain slopes. These soils are well drained with medium runoff and rapid permeability. The soil profile is sand to loamy sand with 5%-30% angular granitic pebbles. According to Veseth and Montagne (1980) these soils are typically dominated by Douglas fir and ponderosa pine, with an understory of bluebunch wheatgrass, Idaho fescue (*Festuca idahoensis* Elmer), kinnikinnick (*Arctostaphylos uva-ursi* L.), mountain mahogany (*Cercocarpus ledifolius* Nutt.), pine grass (*Calamagrostis rubescens* Buckl.), and western yarrow (*Achillea millefolium* L.). However, our pre-burn survey revealed the dominate species of the uplands to be Douglas fir with an understory of Idaho fescue, Kentucky bluegrass, western wheatgrass and clubmoss (*Huperzia spp.*Bernh).

Riparian areas in this site contained ephemeral streams with channelized flow occurring in early to late spring and in late fall to early winter. The pre-burn inventory found the dominate species to be quaking aspen (mean age =101 years). Tree densities were found as high as 877 trees/ha (average density = 330 trees/ha) and sapling densities as high as 16,172 trees/ha (average density = 7,160 trees/ha). Riparian soils of this area represent the Baxendale series (Veseth and Montagne 1980). This series consists of deep sandy soils formed in residuum and alluvium weathered from granitic rocks on gently to steeply sloping foothills and alluvial fans. These soils are well drained, have medium runoff, and rapid permeability. Typically soils of this series are dominated by rangeland plant species such as Idaho fescue, bluebunch wheatgrass, needle-and-thread (*Hesperostipa comata* Barkworth), rough fescue (*Festuca campestris* Rydb.), Sandburg bluegrass (*Poa secunda* J. Presl), prairie junegrass (*Koeleria macrantha* J.A. Schultes), western wheatgrass, and big sagebrush (Veseth and Montagne 1980). However, our pre-burn survey revealed the dominate species in these riparian areas to be quaking aspen, Rocky mountain juniper, and big sagebrush with an understory of elk sedge, Kentucky bluegrass, and redtop bent grass.

Treatment

Both sites were burned under a hazardous fuel reduction prescription by the BLM. Black-lining (common technique used to prevent fire escape) of the perimeter of each sub-watershed was conducted using hand ignition drip torches, and ATV sprayers. Black-lines were a minimum of 1.4 m wide. Once the black-lines were in place, primary

ignition of each watershed was carried out either by ground crews with drip torches or with a helicopter using a plastic sphere dispenser (PSD) machine. Each sphere contained a mixture of potassium permanganate and ethylene glycol. Choice of method was dictated by the environmental conditions at the time of each burn to ensure the safety of the fire crew and to ensure fulfillment of the burn objectives. The specific target for the Armells and Whitehall sites was to remove 70% of the smaller trees (trees < 1.5 m tall) without removing more than 20% of the larger trees (trees > 1.5 m tall). Each burn was intended to take 3 to 5 days to complete. Prescribed burns were conducted in the spring of 2002 at the Armells site and took 5 days to complete. The prescribed burns at the Whitehall site were conducted in the fall of 2005 and spring of 2006. Each burn at this site took 3 days to complete.

Monitoring Plan

Dry Armells

To monitor the effects of prescribed fire the burn plan was applied so that comparisons could be made between burned and unburned subdrainages. Four drainages were burned and 2 were left unburned to assess any climatic effects (six drainages total). Upland attributes monitored within each of the drainages were species diversity, tree and sapling density, and forest canopy cover. Riparian attributes monitored within each of the drainages were depth to groundwater, species diversity, and biomass production, and utilization by big game and range cattle. Monitoring at this site took place in 2001 (pre-burn), 2002 (immediate post-burn), and 2005 (post-burn). Riparian attributes monitored

were depth to groundwater, biomass production, and utilization. Riparian biomass production and utilization was monitored again in 2006.

Groundwater. To monitor the effects of fire on depth to groundwater, 15 wells were placed in each drainage bottom in a 3 (wide) by 5 (deep) configuration (Figure 2). Well rows (total of 5) were spaced at a distance of 3 times the width of the drainage bottom (Meyers and Swanson 1997). Each well was set to a depth of approximately 2 m and groundwater levels, within each well, were monitored monthly from April to November each year to present. Groundwater was monitored for 3 months prior to treatment (2002).

The slope of the groundwater surface was calculated for three of the six riparian drainages using topographic data. Topographic data was collected in the summer of 2006 using survey grade equipment. To calculate the groundwater surface slope for one month/drainage, the average groundwater elevation for each of the 5 well rows was plotted against the distance between the rows. This provided 4 slope values (WR1 (well row) to WR2 = S1 (slope 1), WR2 to WR3 = S2, WR3 to WR4 = S3, and WR4 to WR5 = S4). Average slope of the groundwater surface for the monitored portion of the drainage was then the average of the 4 slopes.

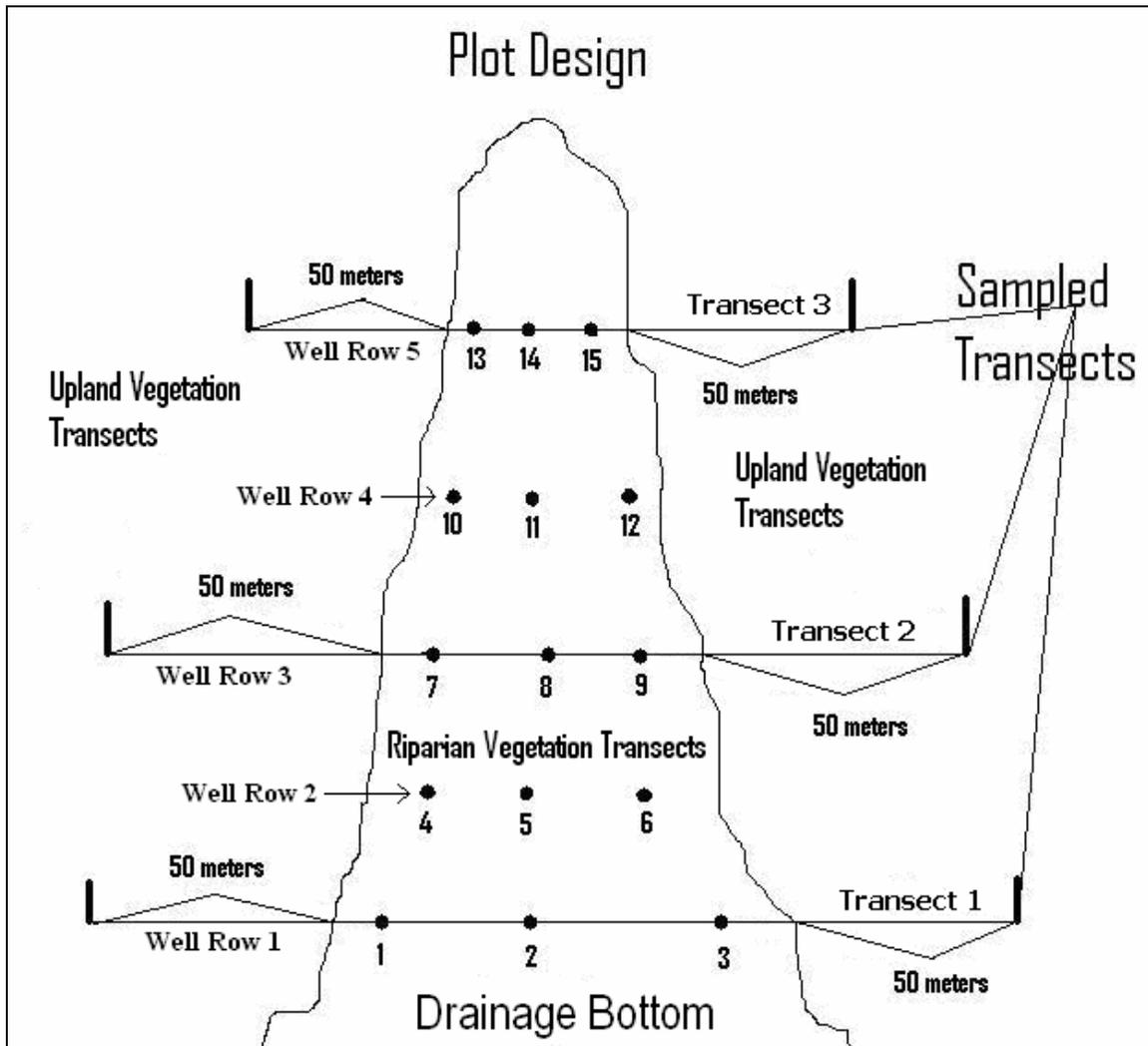


Figure 2. Plot design for each riparian drainage. This figure shows the location of the 15 wells (noted as numbered dots), and the sampled transects for each drainage.

Vegetation. Permanent vegetation transects were established across each floodplain (3 per drainage) and 50 m into the uplands on both sides of the drainage (6 per site). The 3 transects corresponded to every other row of wells beginning with well row 1 and ending with well row 5. Upland percent cover, by species, was recorded using a Daubenmire (1968) frame at 10 m intervals along each upland transect. However, due to the unequal widths of the riparian areas, frames were placed at an even interval along

each transect to ensure similar area coverage to the upland transects. Species diversity for the upland and riparian areas were summarized with the Shannon diversity index (Magurran 1955). Each drainage received two diversity values; one for the riparian area ($n = 6$) and one for the corresponding upland area ($n = 6$). Forest canopy cover was measured along each transect at 5 m intervals using a Geographic Resource Solutions (GRS) densiometer. Riparian percent forest canopy cover and upland percent forest canopy cover were also summarized separately for each of the drainages.

Biomass Production/Utilization. Big game/cattle exclusionary cages were used to estimate potential herbaceous biomass. Each cage contained 1.5 m² of the riparian community. In 2005 two cages were placed in each of the 6 riparian zones (6 drainages x 2 cages = 12 cages). In 2006 more resources were made available and each riparian zone received an additional cage; totaling 3 per drainage (6 drainages x 3 cages = 18 cages). After 3 months of grazing, in both 2005 and 2006, a 0.25 m² frame was randomly placed within each cage and all grasses/forbs that fell within this frame was clipped, sorted by species, and dried. To monitor utilization a 2nd 0.25 m² frame was clipped (randomly placed) outside each grazing exclusionary cage. The difference in total dry forage (in vs. out) was recorded as percent utilization (inside biomass – outside biomass / inside biomass = % use).

Whitehall

Groundwater, species diversity, and forest canopy closure were monitored at Whitehall using the same protocol used at the Dry Armells site. Riparian biomass

production was not monitored here due to allocation of resources (all grazing cages were at Dry Armells). Utilization also was not monitored here due to BLM mandated pre-burn cattle exclusion and post-burn rest. Another difference between the sites was the number of burned drainages (2 total instead of 4, Little Whitetail canyon “LWT” and Hay canyon). This site also contained 2 unburned drainages (Mule deer canyon, and Pony canyon) to assess climatic affects on vegetation and depth to groundwater.

Statistical Analysis

All data from both the Dry Armells and the Whitehall sites were analyzed by-site due to different burn dates and time-since-burn. Both the Armells and Whitehall data were analyzed using the individual drainage as the experimental unit. Again, Armells contained 6 drainages ($n = 6$) and Whitehall contained 4 ($n = 4$).

Differences in depth to groundwater for burned and unburned drainages at both locations were tested using a 1-tailed Student’s t test ($\alpha = 0.1$). Comparisons were made within treatment and across years. The control units were used to account for changes due to environmental factors.

Differences in species diversity tree density, and canopy closure, due to treatment, for both Armells and Whitehall were tested using a 1-tailed Student’s t test ($\alpha = 0.1$). Comparisons were made within treatment and across years. Control units once again were used to account for changes due to environmental factors.

Differences in riparian biomass production was tested using a 1-tailed Student’s t test ($\alpha = 0.1$). Comparisons were made within year and between treatments.

Analysis was based on the 2005 and 2006 post burn field data. A Pearson's correlation acceptance level of ($r = 0.5$) was used to assess the degree of relationship between the following variables: reduction of forest canopy cover, reduction of trees by location and size class, change in depth to groundwater, utilization, riparian biomass production, and species diversity.

CHAPTER 4

RESULTS

Dry Armells

It is important to note that statistical analysis of the pre-burn data revealed that burn units 1, 2, and lower control had significantly fewer trees, large ($P < 0.001$) and small ($P < 0.001$), per unit area than what existed in burn units 3, 4, and upper control (Figure 3). The post fire survey conducted by the Montana Agricultural Experiment Station (MAES), in 2002, revealed a slight deviation from the fire prescription goals. Originally the plan was to remove 70% of the smaller trees, without removing more than 20% of the larger trees. The survey (Marlow 2002) revealed that both small and large tree numbers were significantly reduced ($P = 0.04$ and $P = 0.004$ respectively) with a 90% reduction in smaller trees (trees < 1.5 m tall) and a 30 % reduction of larger trees (trees > 1.5 m tall) (Figure 4). Post-fire aerial photography analysis revealed that 18 – 22 % of the entire watershed was treated by fire (Table 2). Lastly, the riparian areas, located within the burned units, did not seem to thoroughly burn due to moist conditions at the time of the burn (Figure 5).

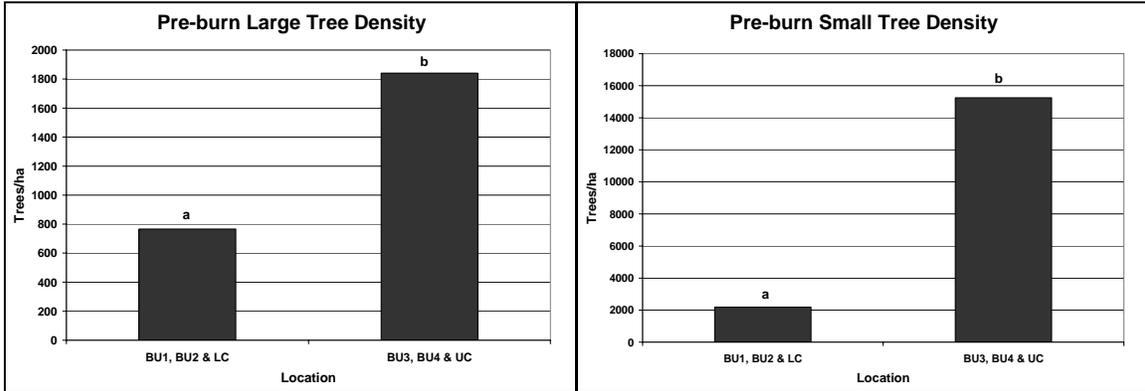


Figure 3. Pre-burn differences in tree density by size class at Dry Armells. Bars with the same letter are not different at $P = 0.10$.

Table 2. Area of watershed treated and percentage of trees removed at Dry Armells and Whitehall study sites.

Site	Area of Watershed Treated	Large trees Removed	Small Trees Removed
Dry Armells	18 – 22 %	Upland = 30 % reduction Riparian = Treeless	Upland = 90 % reduction Riparian = Treeless
Whitehall	3 – 14 %	Upland = 63 % reduction Riparian = 58 % reduction	Upland = 8 % reduction Riparian = 42 % increase

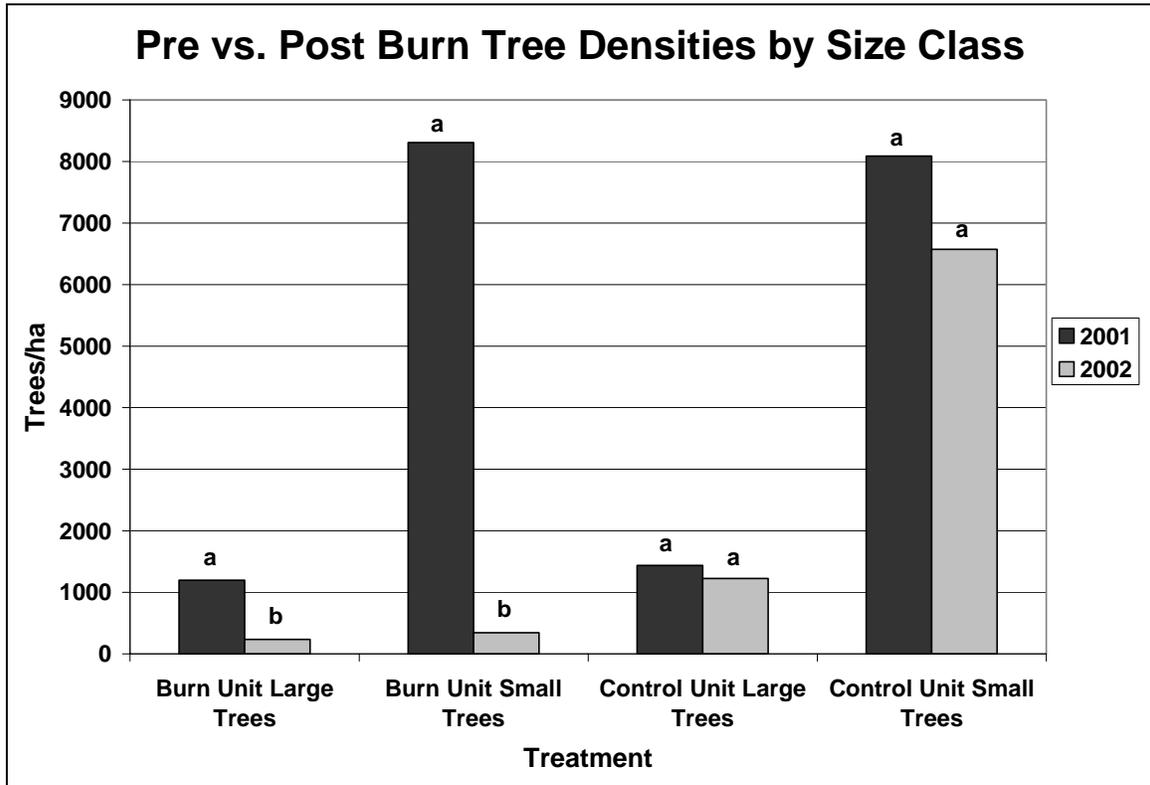


Figure 4. Pre-burn (2001) and immediate post-burn (2002) upland tree densities at the Dry Armells demonstration watershed. Average numbers of both tall and short trees were significantly reduced in the burn units ($n = 4$). There was no difference in small or large trees between 2001 and 2002 in the control units ($n = 2$). Bars with the same letter are not different at $P = 0.10$.



Figure 5. Before and after photos of burn unit 3 (Dry Armells) showing riparian burn characteristics. The top photo was taken in August of 2001 (9 months before the burn) and the photo on the bottom was taken immediately after the burn (June 2002).

Groundwater.

Before the drainages were treated with prescribed fire (April 2002) average depth to groundwater was approximately 122 cm below grade (Figure 6). One year later (2003) average depth to riparian groundwater, in the post burned drainages, ($n = 4$) decreased ($P = 0.065$) from an average depth of 122 cm below grade to 79 cm below grade. Average depth to groundwater remained at this level through 2006.

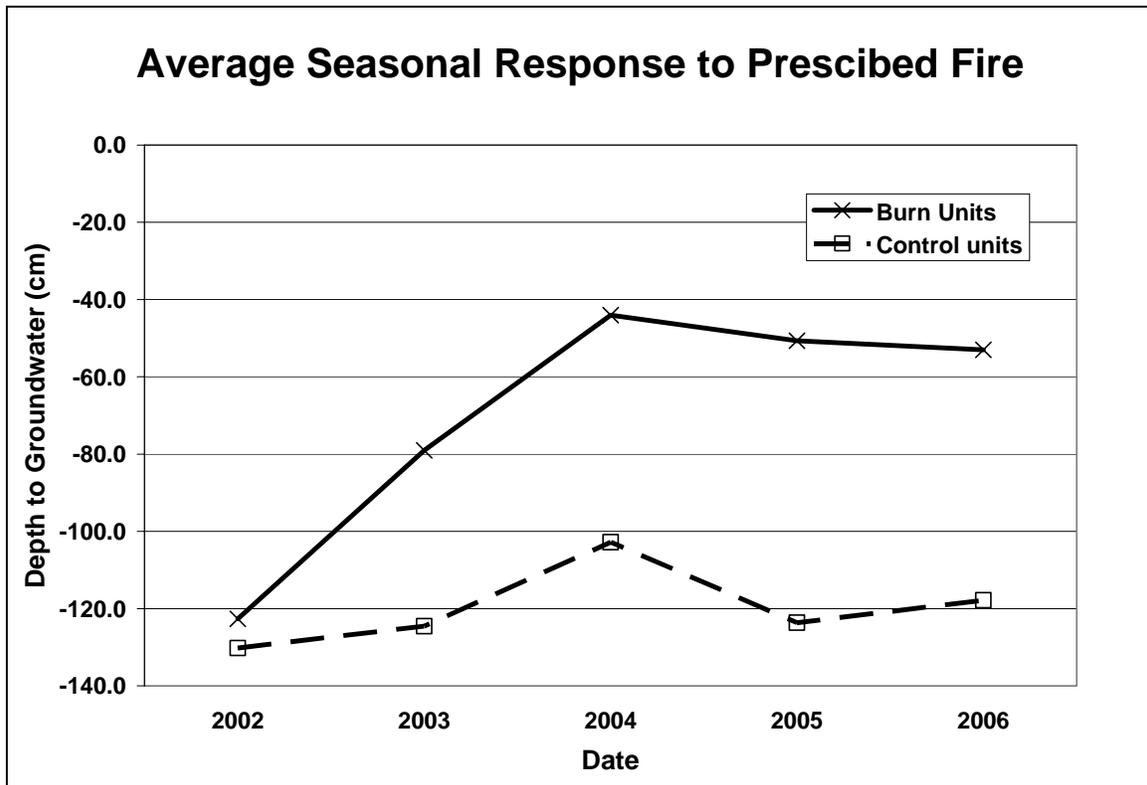


Figure 6. Riparian groundwater response by treatment to the 2002 (April) prescribed fire at Dry Armells. Burn units show an immediate response to fire. The control units remained at approximately 120 cm below grade throughout the entire study period. Burn units ($n = 4$). Control units ($n = 2$).

Placing the burn units into preburn “high tree density” (BU3 & BU4) and “low tree density” (BU1 & BU2) burn unit classes a similar trend was exhibited, however,

magnitude of change was different. One year after treatment (2003) depth to groundwater in the high density units (BU3 & BU4) declined ($P = 0.005$) from an average depth of 122 cm below grade to 36 cm below grade (Figure 7). In 2004 depth to groundwater in the high density units decreased ($P = 0.05$) again to 13 cm below grade. Depth to groundwater in these units remained at this level through 2006. Depth to groundwater in the low density units (BU1 & BU2) did not reflect similar changes until 2004. However, at this time, depth to groundwater in these units ($P = 0.03$) decreased from an average depth of 122 cm below grade to an average depth of 75 cm below grade. Average depth to groundwater for the low density burn units remained at this level through 2006. Throughout the five year period depth to groundwater in the control units did not change.

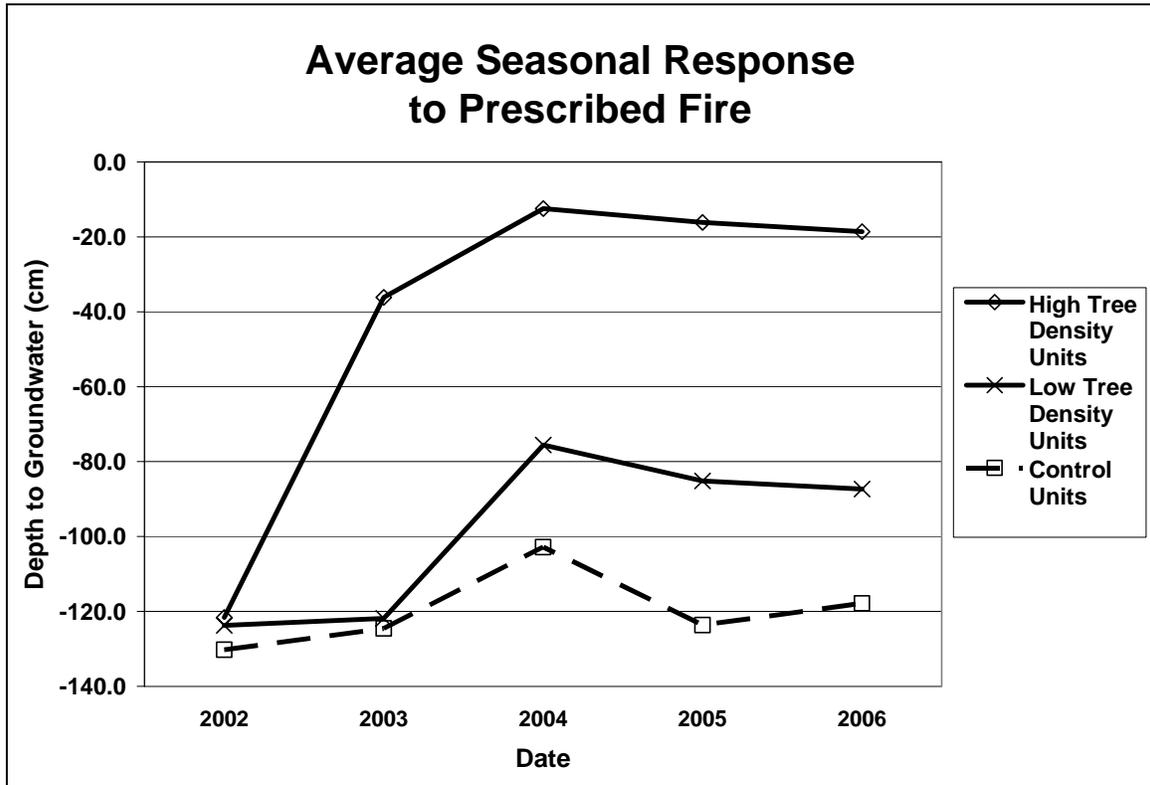


Figure 7. Riparian groundwater response, by tree density class, to the 2002 (April) prescribed fire at Dry Armells. The high density burn units (BU3 & BU4) show a significant and immediate response to fire. The low density burn units (BU1 & BU2) did not respond until the summer of 2003. The control units remained at approximately 120 cm below grade throughout the entire study period.

Groundwater surface slope analysis (Figure 8) revealed that groundwater slope was fairly uniform in BU1, UC, and LC throughout this study (groundwater slope data not available for BU2, BU3, & BU4). Peaks in the graphs indicate a steeper gradient in groundwater slope looking up drainage. Note that the sign of each slope did not change throughout the study.

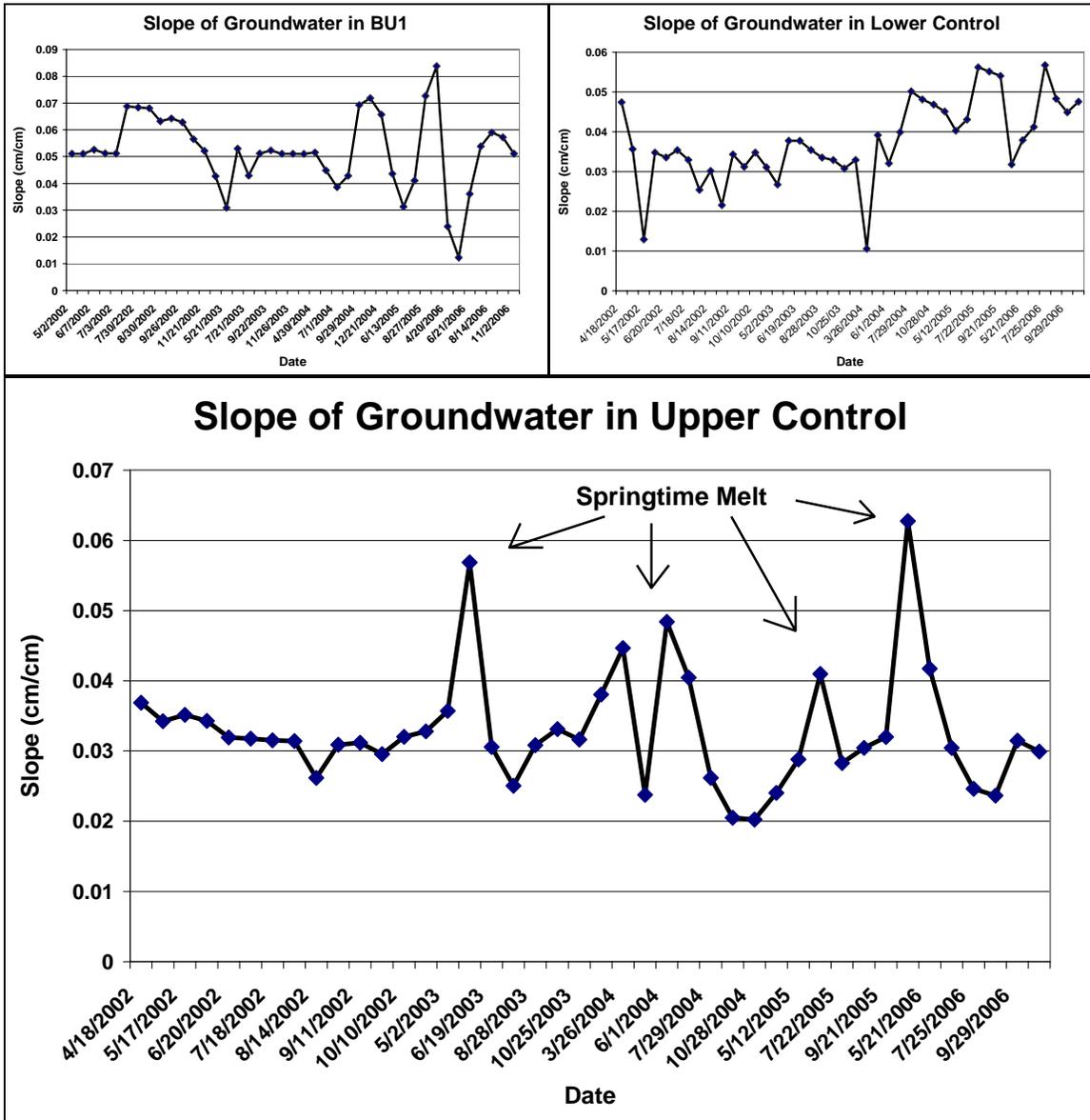


Figure 8. Monthly slope of the groundwater surfaces for BU1, LC, and UC at Dry Armells. Peaks pointed out in lower graph depict a steeper gradient in groundwater looking up drainage.

Vegetation.

Riparian herbaceous species diversity did not exhibit any statistically significant changes following treatment (Figure 9). In 2002 diversity in burn units 1 and 3 slightly increased but units 2 and 4 declined slightly. By 2005 diversity in units 1 and 2 had

increased but declined again in units 3 and 4. However, herbaceous species diversity did increase ($P = 0.08$) in the uplands in 2002, and remained higher ($P = 0.09$) through 2005 (Figure 10). Change in upland diversity between 2002 and 2005 was not significant. Species diversity in the riparian control units increased ($P = 0.004$) from 2001 to 2002 and remained higher ($P = 0.04$) through 2005. Riparian control unit diversity did not change between 2002 and 2005. Lastly, no differences in upland control unit diversity were exhibited throughout the entire study.

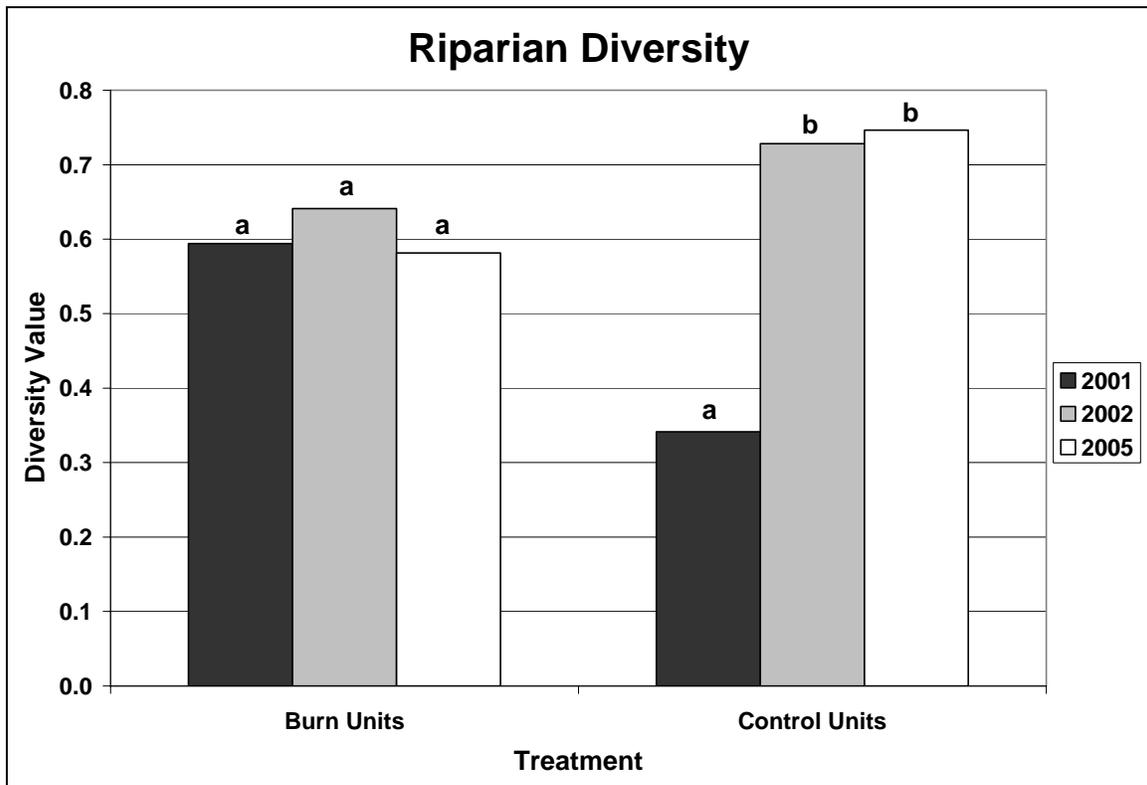


Figure 9. Species diversity trend in riparian areas of the Dry Armells watershed following the 2002 prescribed fire. 2001 = pre-burn conditions. 2002 = immediate post-burn conditions. 2005 = 3 years post-burn conditions. Bars with the same letters are not different at $P = 0.10$.

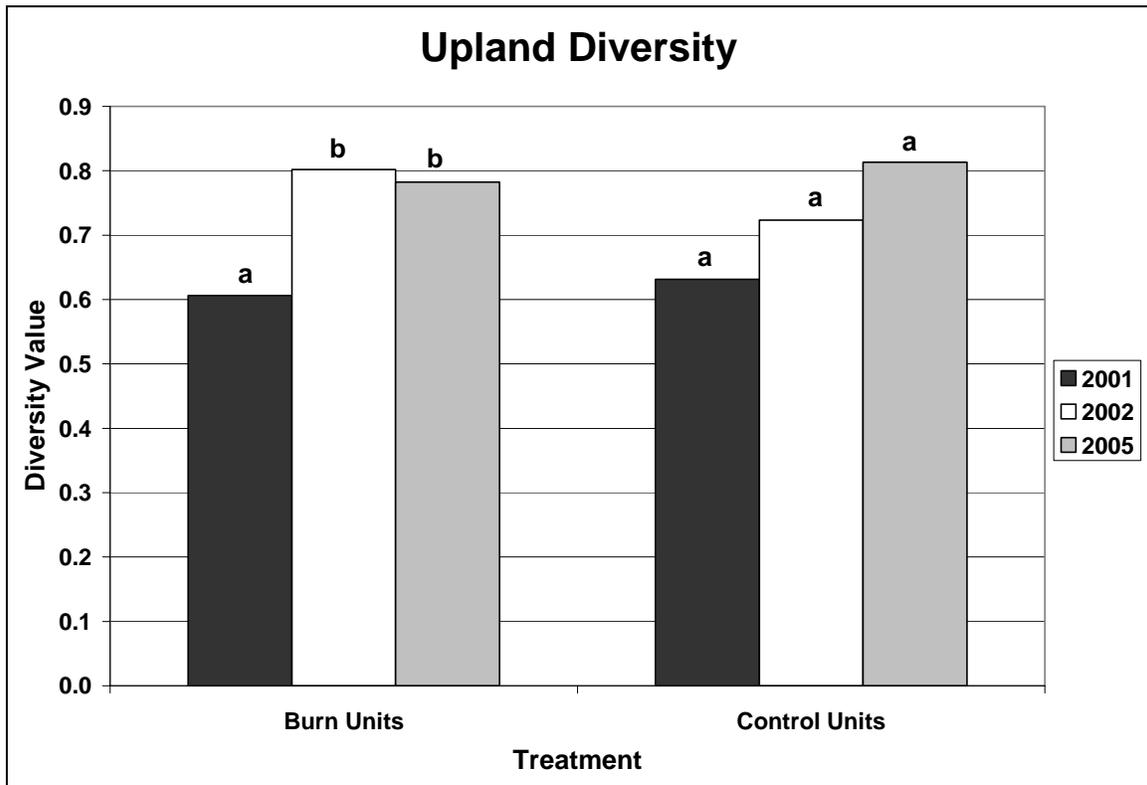


Figure 10. Upland species diversity trend in the Dry Armells watershed following the 2002 prescribed fire. 2001 = pre-burn conditions. 2002 = immediate post-burn conditions. 2005 = 3 years post-burn conditions. Bars with the same letters are not different at $P = 0.10$.

Forest canopy cover was decreased ($P = 0.009$) following treatment in 2002 (Figure 11) and remained lower ($P = 0.08$) in 2005 as compared to 2001. However, in 2005 forest canopy was found to have increased ($P = 0.099$) from its post-burn (2002) level. No differences in forest canopy cover were exhibited in the control units throughout this study.

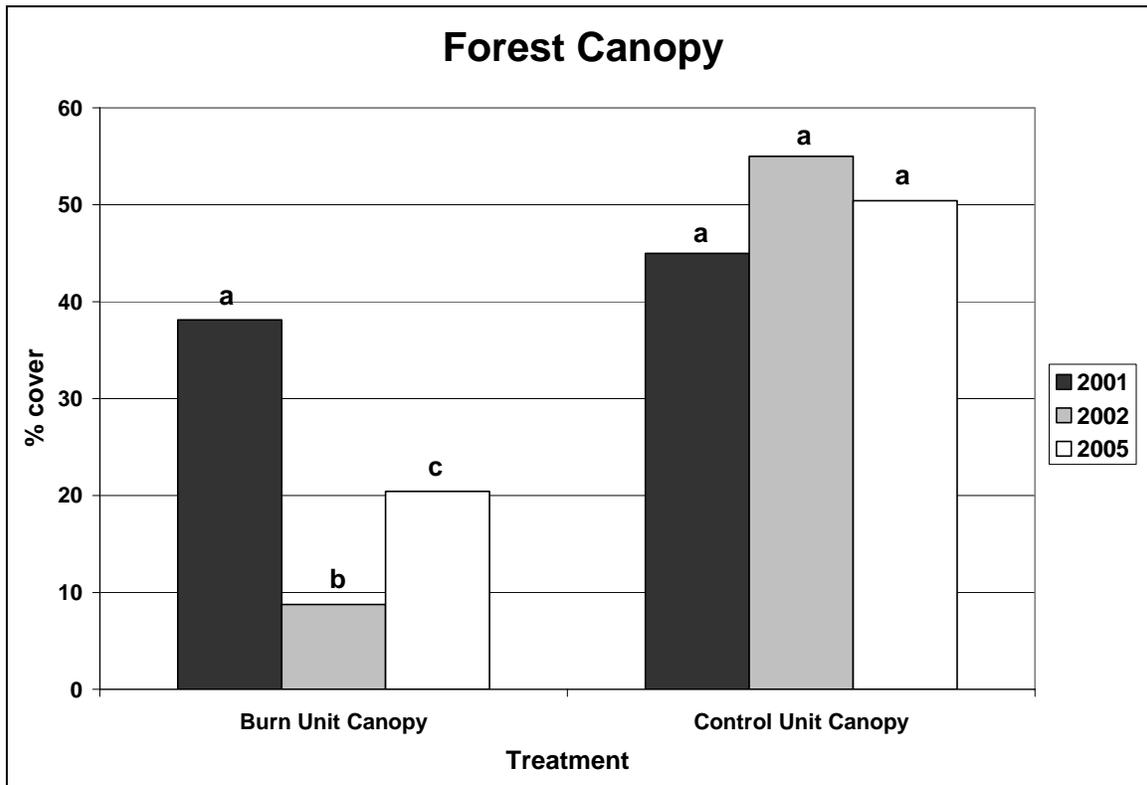


Figure 11. Forest canopy cover trend in the Dry Armells watershed following the 2002 prescribed fire. 2001 = pre-burn conditions. 2002 = immediate post-burn conditions. 2005 = 3 years post-burn conditions. Bars with the same letters are not different at $P = 0.10$.

Biomass Production/Utilization

Not having any forage data prior to burning we compared forage production, based on dry weight, between the burned and non-burned drainages in both 2005 and in 2006. In 2005, there was no difference in riparian biomass production between the burn units and control units (Figure 12). There was also no difference detected in 2006.

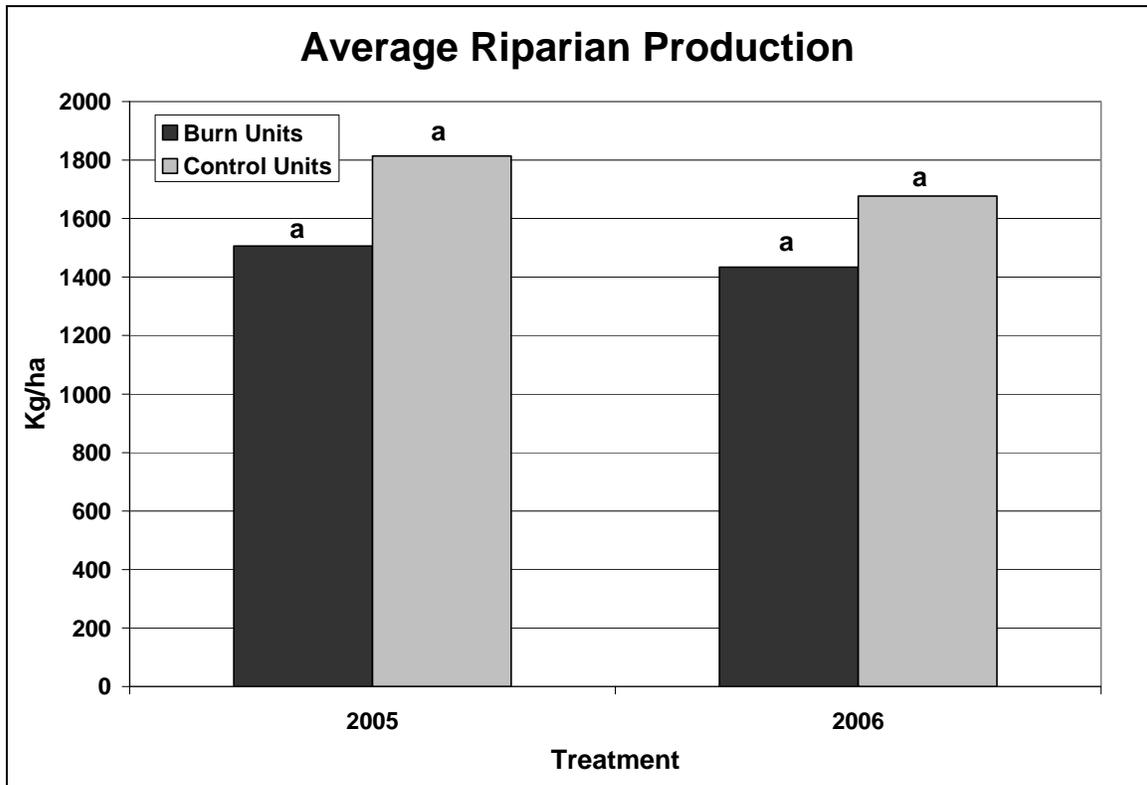


Figure 12. Potential production (weights obtained from within cages) within the Dry Armells riparian areas. There was no difference in forage production between the burn units and control units for both 2005 and 2006.

Graminoid cover (Figure 13) increased ($P = 0.03$) in the uplands in 2002 and remained higher ($P = 0.008$) through 2005. No differences in graminoid cover were exhibited in the upland control units at any time during this study. However, graminoid cover did significantly ($P = 0.06$) increase in the riparian control units in 2002 (immediate post-burn).

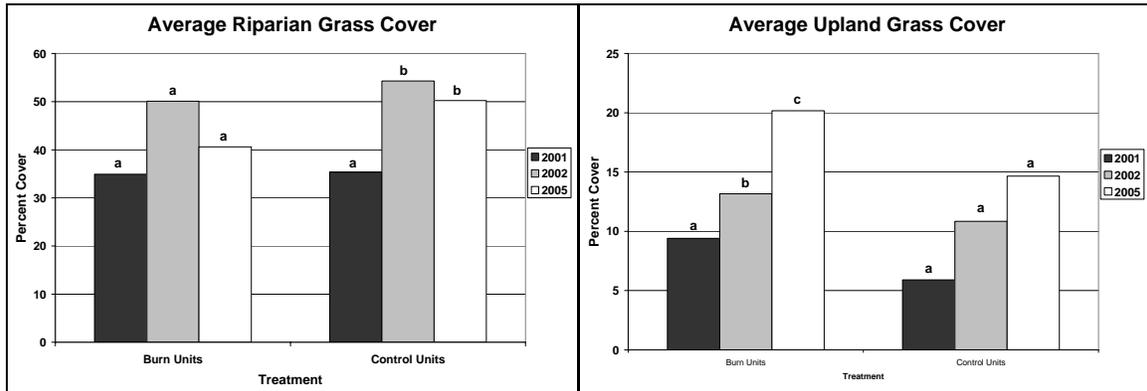


Figure 13. Dry Armells riparian and upland graminoid species cover trend following prescribed fire (2002). 2002 = immediate post-burn conditions. 2005 = 3 years post-burn conditions. Bars with the same letters are not different at $P = 0.10$.

No differences in biomass utilization were found between 2005 and 2006 for either the burn units or the control units (Figure 14). In both years burn units 3 and 4 exhibited little to zero utilization. In the units that were utilized (BU1 & BU2) no differences were found in either 2005 or 2006 when compared to the upper and lower control units.

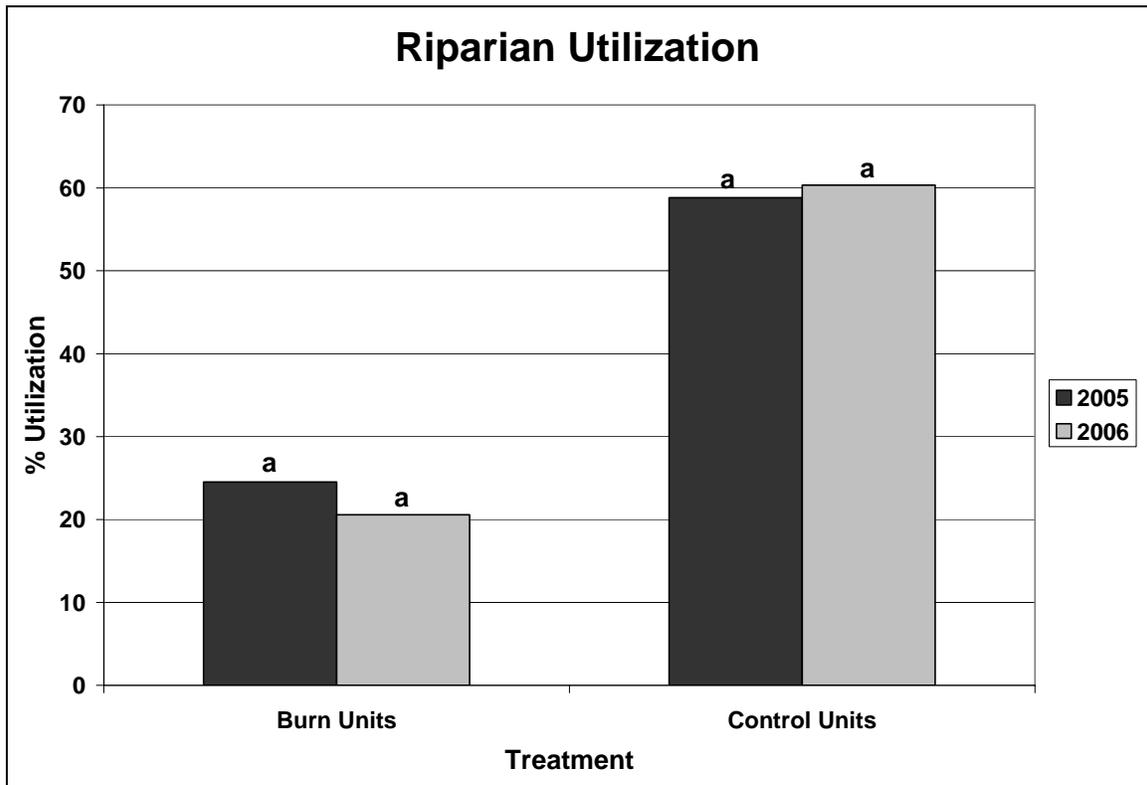


Figure 14. Riparian area biomass utilization at Dry Armells (2005 & 2006). No differences were found in utilization for the burn units or the control units for both 2005 and 2006.

Making Connections at Armells

In 2002 change in riparian diversity was related to change in groundwater, reduction of forest canopy, large tree removal, and small tree removal (Figure 15). Also in 2002 upland diversity was related to reduction of forest canopy, large tree removal, and small tree removal. Lastly, in 2002 change in groundwater was not related to large or small tree removal. In 2005 riparian diversity was related to change in groundwater, reduction of forest canopy, large tree removal, and small tree removal. Also in 2005 groundwater was related to reduction of large trees as well as reduction of small trees. In 2005 upland diversity was not related to reduction of forest canopy, large tree removal,

and small tree removal. Lastly there was no correlation exhibited between groundwater and biomass production, or between canopy reduction and biomass production for both the 2005 and 2006 correlations.

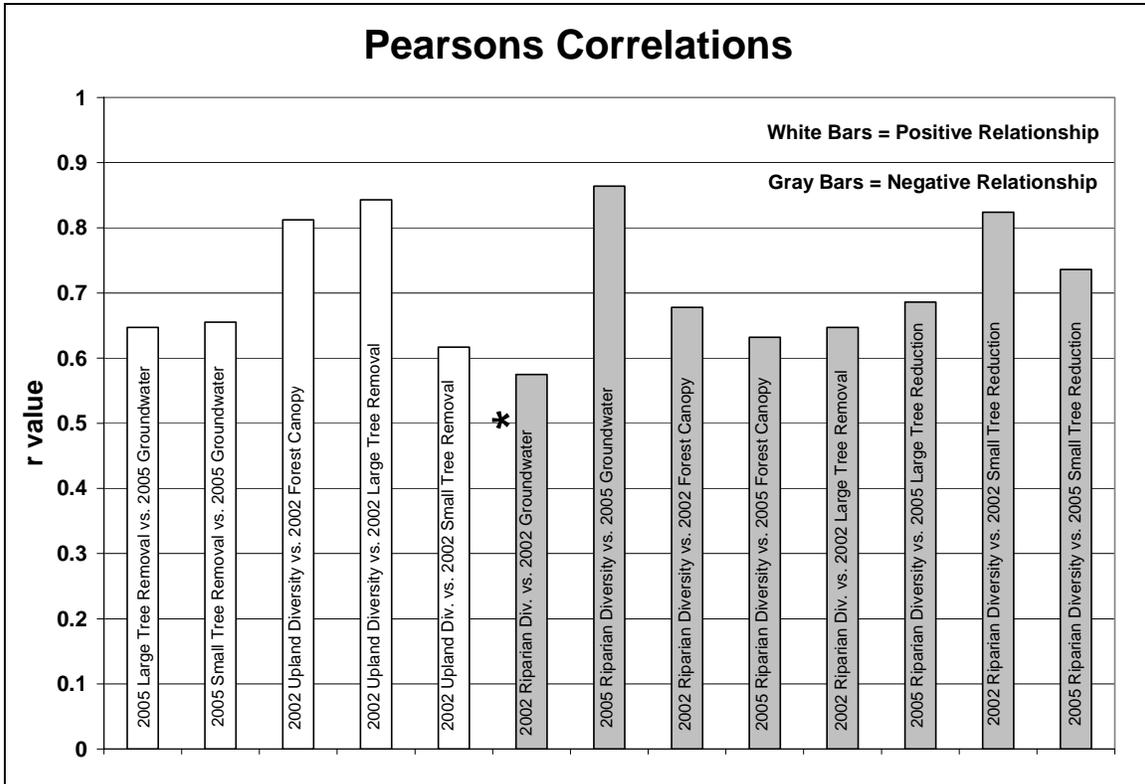


Figure 15. Results of correlations made at Dry Armells. Each variable was based on percent change from the 2001 baseline data. For example; 2002 riparian diversity vs. 2002 groundwater * = the percent change in riparian diversity vs. the percent change in riparian groundwater from 2001 to 2002.

Whitehall

The post fire survey conducted by Montana State University (Bozeman) in 2006 revealed a departure from the fire prescription goals. The plan, just like the one set at Dry Armells, was to remove 70% of the smaller trees, without removing more than 20%

of the larger trees (in both the upland and riparian areas). The survey revealed no change in small upland tree (trees < 1.5 m tall) density (Figure 16). There was also no change detected in small riparian tree density. The survey also revealed that there was no change in large tree (trees > 1.5 m tall) density in either the uplands or in the riparian areas. Lastly, approximately 3% of the Little Whitetail canyon and 14% of Hay canyon were treated with fire (Table 2).

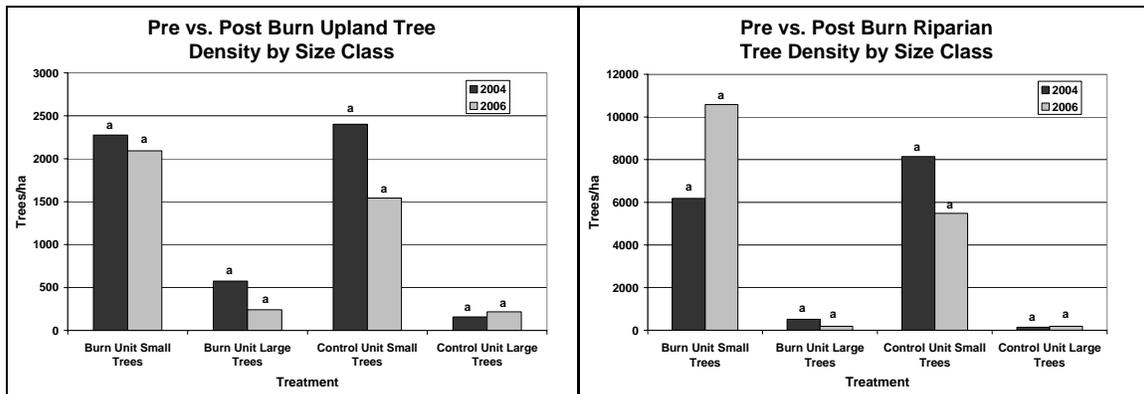


Figure 16. Pre-burn (2004) and immediate post burn (2006) upland and riparian tree densities at the Whitehall experimental watershed. 2004 = pre-burn conditions. 2006 = immediate post-burn conditions. Bars with the same letters are not different at $P = 0.10$.

Groundwater

Average depth to groundwater for the two treated drainages (Hay and Little Whitetail canyons) was approximately 67 cm below grade for 2003, 2004, 2005 (Figure 17). Average depth to groundwater for the two control drainages was approximately 75 cm below grade for the same time period. No statistical differences in depth to groundwater for either the treated or control units were found during any time of this study.

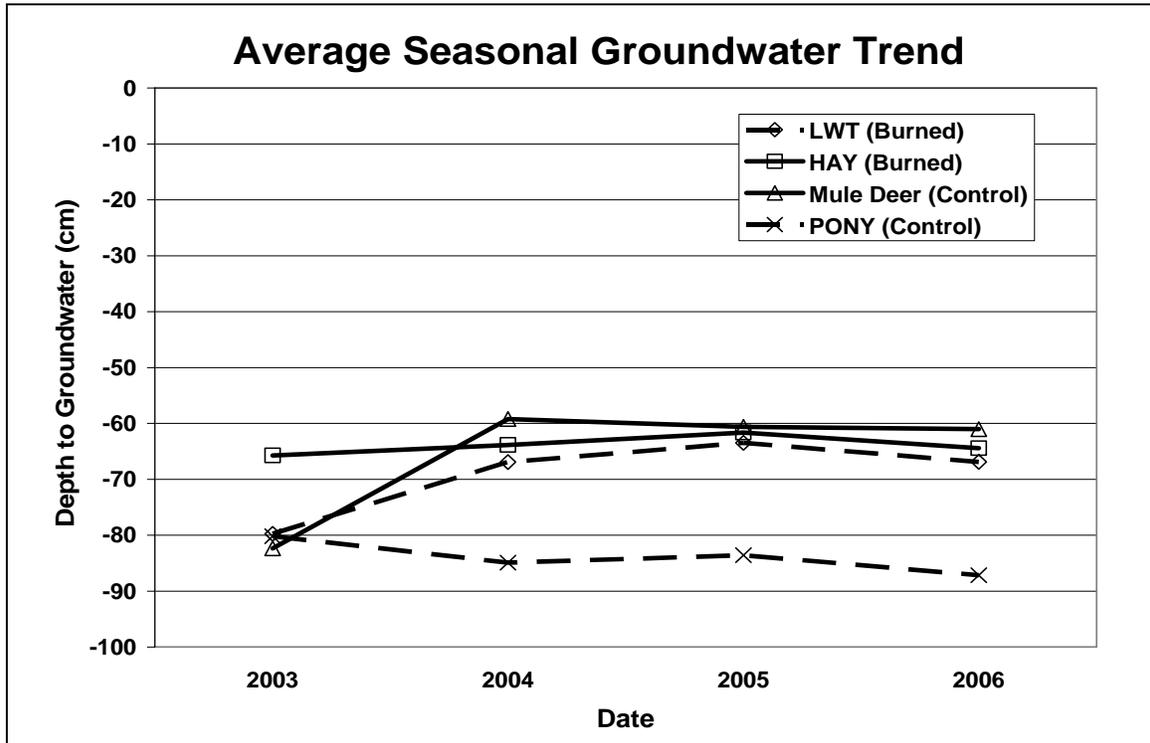


Figure 17. Riparian groundwater response to the 2005 (September) and 2006 (April) prescribed fires at Whitehall. No differences in depth to groundwater have been exhibited in any of the drainages since the fire.

Vegetation

Herbaceous species diversity did not change, post fire, in the uplands or in the riparian areas in the treated drainages. There was also no change in upland or riparian diversity in the control units (Figure 18).

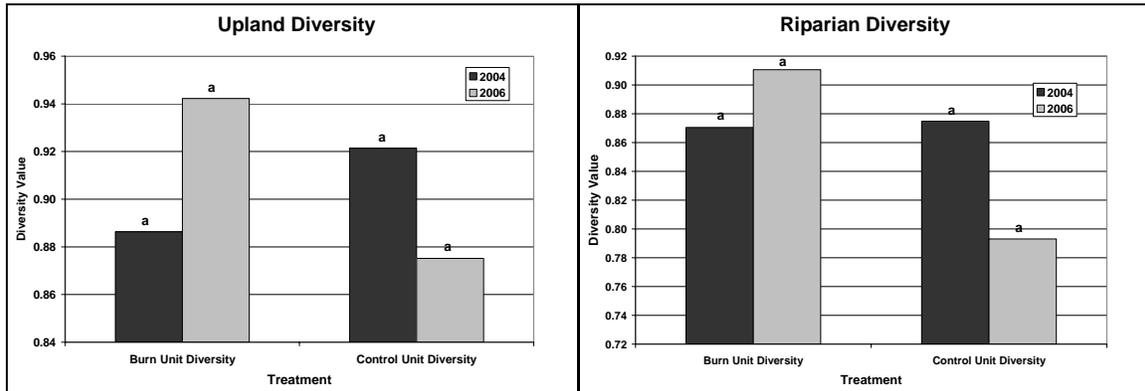


Figure 18. Upland and riparian species diversity response to prescribed fire at Whitehall. 2004 = pre-burn conditions. 2006 = post-burn conditions. Bars with the same letters are not different at $P = 0.10$.

Forest canopy cover in the uplands did not change post burn in the uplands (Figure 19). There was also no change in riparian forest canopy cover. No differences in forest canopy cover were exhibited in the control units throughout this study.

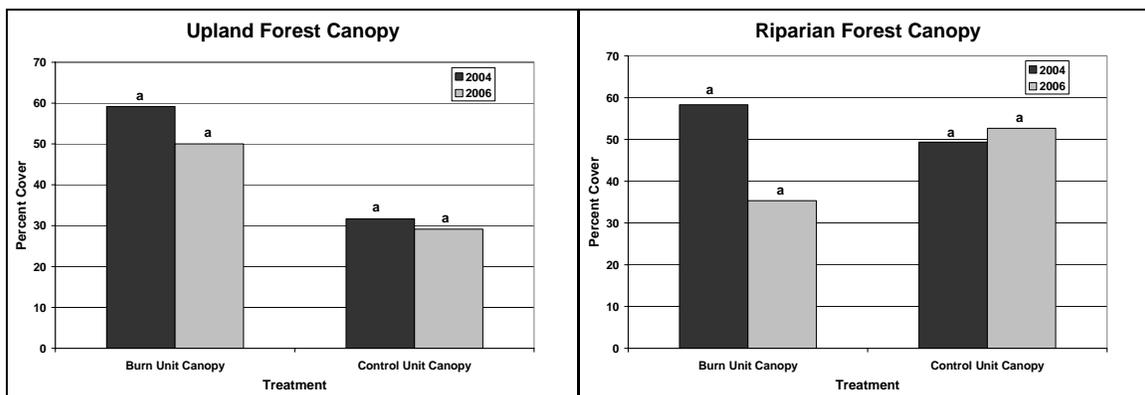


Figure 19. Upland and riparian tree canopy response to prescribed fire at Whitehall. 2004 = pre-burn conditions. 2006 = post burn conditions. Bars with the same letters are not different at $P = 0.10$.

Making Connections at Whitehall

Fire did not significantly change any of the measurable parameters (depth to ground water, tree density, canopy cover, species diversity) at this site. Therefore, correlations based on change, due to fire, were not calculated.

CHAPTER 5

DISCUSSION

Dry Armells

As a result of treatment overall stand density was reduced and the anticipated decrease in depth to groundwater was achieved. Aided by the relationship found between depth to groundwater and tree removal ($r \approx 0.65$ for both large and small trees) it is likely that the reduction in stand density decreased the demand the trees were placing on the shallow groundwater table. This reduced demand for water is our best explanation for the recorded increases in groundwater levels after the fire. It is possible that the marginal response in depth to groundwater in burn units 1 and 2 was due to a lower (pre-burn) tree density than what existed in burn units 3 and 4 (Figure 3). Thus, there may be a threshold density, for this ecosystem and soil type, that needs to be exceeded before prescribed fire will generate the response noted in burn units 3 and 4.

Groundwater surface slope was uniform (slope did not change sign) in Upper Control, BU1, and Lower Control throughout the entire time of this study (Figure 8). This analysis checked for two possible confounding variables that may have affected our depth to groundwater measurements. For example, if slope had gone negative at any time it would have suggested groundwater drawn down somewhere up-valley (i.e. turning on an irrigation pump in the uplands). A negative slope could have also suggested that groundwater was being recharged from a filling stock pond. This was a concern because 2 of our drainages (BU4 & Upper Control) had stock ponds located at the downstream

end. Although we were not able to analyze BU4's groundwater slope we are assuming that the groundwater in this drainages acted similarly to the groundwater in Lower Control. Lastly, BU2, BU3, and BU4 were not analyzed due to lack of time and resources.

Contrary to anticipated results overall species diversity did not increase in the riparian burn units due to decreased depth to groundwater or to opening of the forest canopy, but it did increase in the riparian control units. This increase is likely due to the greater amounts of disturbance from cattle grazing in the unburned riparian areas in both 2005 and 2006. Note that riparian utilization in the burn units was much less than in the controls. The reason the burn units did not increase in species diversity post-burn may be due to the lack of disturbance typical to riparian areas intersected by an active channel. Absence of a defined channel eliminates the creation of exposed soil or sediment deposits that could be occupied by new species. If channels do form and sustained flow is created from the increased groundwater levels, surface disturbance may be sufficient to stimulate diversity increases. It is also possible that prescribed fire did not increase species diversity in these riparian areas because a thorough burn of the riparian areas was not achieved (again little disturbance). In the thoroughly burned upland areas, species diversity did significantly increase. So it is reasonable to assume, especially in this fire dependant ecosystem, that had fire carried through the riparian zones there would have been an increase in riparian diversity (Romme 1982).

Also contrary to anticipated results riparian biomass production was not greater in the burn units ($n = 4$) as compared to the controls ($n = 2$) in either 2005 or 2006.

However, biomass production was greater in burn units 3 and 4 in 2005 as compared to the control units. In 2006 burn units 3 and 4 produced the same amount of biomass as compared to the control units. However, when burn units 1 and 2 were averaged with burn units 3 and 4 the overall burn unit production (for both 2005 and 2006) dropped below the levels exhibited by the control units. There was no correlation exhibited between groundwater and biomass production, or between canopy reduction and biomass production for both the 2005 and 2006 correlations. Further investigation is needed to explain this result.

Although production (kg/ha) did not increase due to fire in the riparian areas, graminoid species cover did increase significantly in the uplands (Figure 13). This indicates that production may have increased across the entire watershed, but without having any grazing cages in the uplands, we are not able to substantiate this claim. Graminoid species cover also increased in the riparian control units in 2002. This difference may be due to sampling in 2002 taking place in July/August as compared to September/October when the 2001 baseline was collected. Sampling in 2005 and 2002 took place during the same months.

Whitehall

Due to the fact we were not able to perform our post fire survey until several weeks past treatment, our post burn riparian small tree density values at Whitehall were elevated or higher than the pre-burn values. This was a result of immediate quaking aspen response following the fire.

Very little change has occurred in average depth to groundwater for all the drainages to date since they were installed in 2003. Unlike the pattern measured at Dry Armells, no appreciable differences have been recorded in depth to groundwater since the burn. Further monitoring will allow judgments pertaining to the effectiveness of prescribed fire to increase depth to riparian groundwater in a Douglas fir/aspen forest type.

The lack of significant change in species diversity (riparian and upland) following the burn may be a product of inadequate sample size (4 drainages vs. 6 at Dry Armells). Nonetheless the increase in species diversity in the burned drainages is in sharp contrast to the recorded decrease in species diversity in both the uplands and the riparian areas of the unburned drainages (Figure 18).

CHAPTER 6

CONCLUSIONS

From the trend observed at Armells we expect to see the groundwater at Whitehall to begin to rise in one to two years. The lack of groundwater response following the burn may be due to a smaller percentage of trees removed at Whitehall as compared to Armells. Another complicating factor is the rapid proliferation of saplings in the treated Whitehall riparian areas. Unlike Armells, the riparian areas at Whitehall were dominated by aspen. So, any water potentially released by the removal of large coniferous trees in the uplands may be utilized just as rapidly by the newly regenerated aspen stand. Lastly, the amount of area treated by fire at Whitehall was below 15% of the entire watershed while 18- 22 % of the watershed was burned at Dry Armells. Stednick (1996) points out that 15 % or more of a watershed must burn to generate a hydrologic response.

Upland species diversity at both Dry Armells and Whitehall responded to prescribed fire positively. However, riparian species diversity at each of these two sites responded much differently. At Armells riparian diversity decreased as a result of treatment but increased at Whitehall (note that both changes stated here were not significant). This may be due to a riparian disturbance regime difference between the two sites. There are no channels at the Dry Armells site, however, channels and ephemeral stream flow does exist at Whitehall. Periodic hydrologic disturbance at Whitehall provides safe sites (bare areas along stream channels) for species recruitment

and regeneration. After the fire, existing species that survived and those arising from the seed bank may not have had to compete as strongly with each other (more open areas) resulting in a slight increase in species diversity. This was not the case at Armells. Beyond the fact that fire was not thoroughly carried through riparian areas at Armells, bare areas in the riparian zones were minimal (no disturbance from fire or flood scouring) which resulted in no change in species diversity. Some new or increased disturbance regime may be needed at Dry Armells for riparian species to react positively.

Grazing cages used at Armells in 2005 and 2006 will be used in the upcoming years at the Whitehall site. Unlike Dry Armells grazing cages will be placed in the riparian and upland areas at Whitehall. The additional cages will allow quantification of the effects of fire on upland biomass production. At Dry Armells there was no difference in riparian biomass production due to elevated groundwater. This result may have been due to inadequate sample size ($n = 6$). Having fewer drainages ($n = 4$) at Whitehall as compared to Dry Armells ($n = 6$) may create further difficulty analyzing the effects of prescribed fire on biomass production. Inadequate sample size may have also been the reason no differences were detected in tree removal at Whitehall. For example; despite removing 58 % and 63% of the large riparian and upland trees at Whitehall no statistical differences were detected (pre-burn vs. post-burn). However, large sample sizes required for traditional statistics are difficult to obtain in landscape level studies due to elevation, soil, geologic and resource constraints. For example, to achieve a large enough sample size researchers may be forced to make comparisons across many landscapes with differing geology and precipitation. Sample size is no longer a problem; however, results

then become confounded due to sampling time (due to large sampling area) and environmental variation. Without a new statistical model for landscape level studies sample size requirements will continue to plague researchers.

CHAPTER 7

IMPLICATIONS

The reduction of coniferous tree species, using prescribed fire, on a watershed scale has been shown to increase riparian groundwater levels at Dry Armells. Burn units 3 and 4 now have year long standing water. Units 1 and 2 have the same in the spring through mid-summer. This was not the case at the time prescribed fire was introduced to the system. Decreases in depth to groundwater and surface water of this magnitude can only be beneficial to the surrounding and down slope ecosystems of the Missouri River Breaks. These areas can now supply water to wild as well as domestic animals, act as sediment catch basins, serve as nutrient sinks for non-point source pollution (animal wastes), and provide a slow and steady release of groundwater down slope to other plant communities. Moreover, as a result of the fire, upland species diversity, and graminoid cover increased. Although, there was minimal riparian vegetation response, it is not unreasonable to suggest that prescribed fire has had a significant positive affect on riparian process and function due to the noted increases in groundwater. Furthermore, the use of prescribed fire may lend itself to other types of ecosystem rehabilitation due to noted increases in favorable upland attributes.

While studies have looked at the effects of tree removal on water release (stream discharge), they generally address coastal regions (non-rangelands) or areas that experience more than 38 cm of annual precipitation. Hibbert (1983) and Clary (1975) argue that 38 cm is the minimum for hydrologic response to fire or mechanical thinning

treatments. During the course of this study (2001 to 2006) Dry Armells only received an average of 31.6 cm of precipitation annually. Although perennial flow was not stimulated at this site, riparian groundwater important to riparian process and function and season-long standing water important to the cattle industry and wildlife was increased. This suggests that further research, focusing on groundwater response to tree and shrub removal in areas with minimal rainfall (< 38 cm annually), is needed. Lastly, differences between Dry Armells and Whitehall, in terms of groundwater response, may aid in the identification of thresholds, such as watershed size, percent of watershed treated, pre-burn tree density and precipitation that needs to be met or surpassed to stimulate hydrologic response in future studies.

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APPENDICES

APPENDIX A

PRECIPITATION DATA

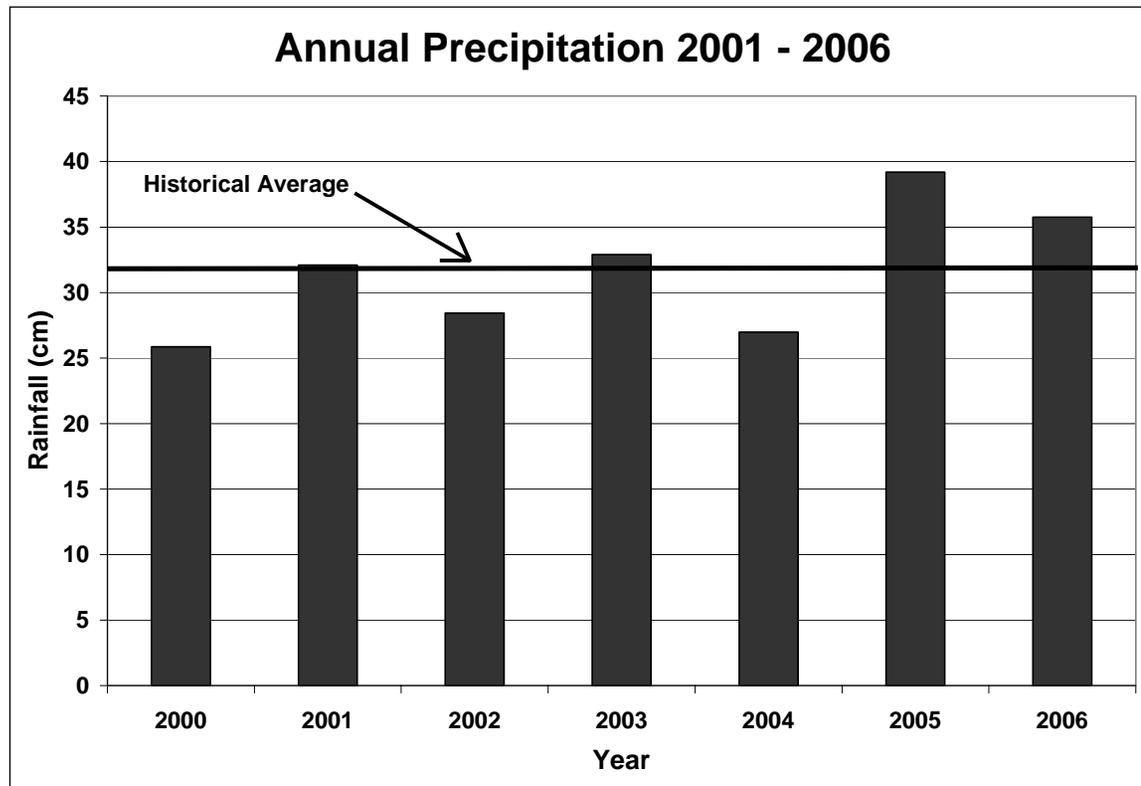


Figure 20. Annual precipitation and historical average at Dry Armells.

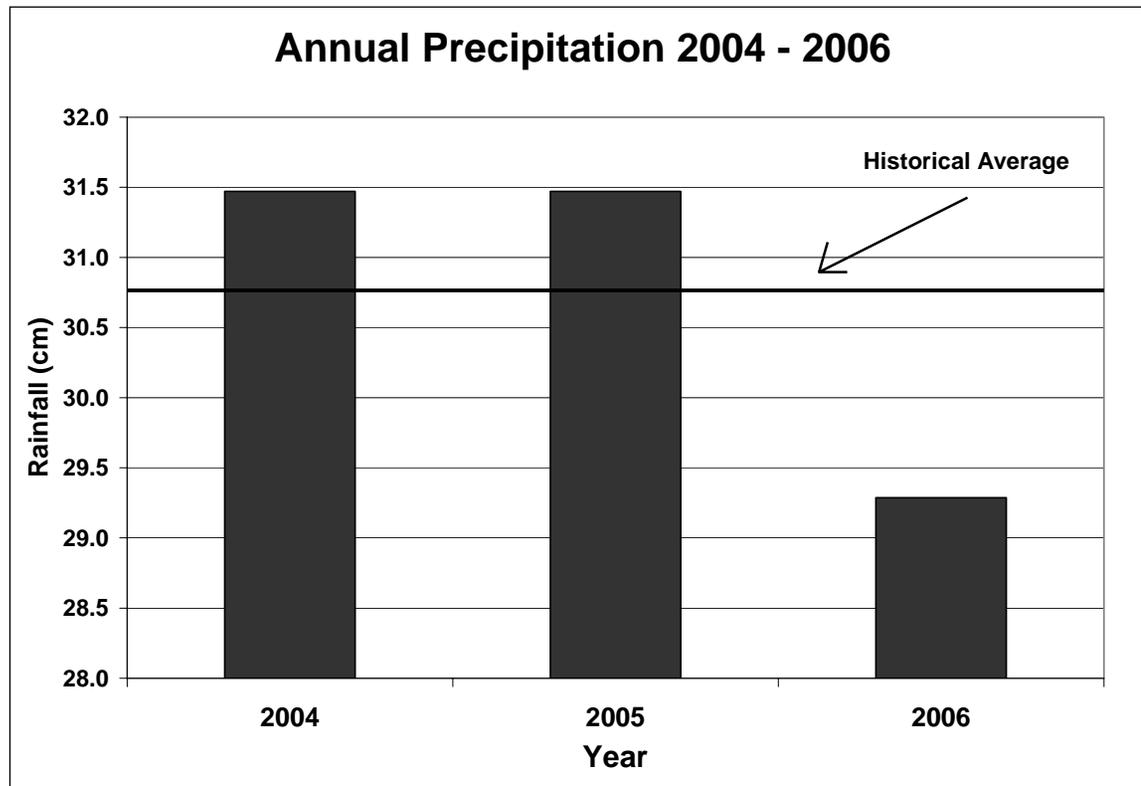


Figure 21. Annual precipitation and historical average at Little Whitetail.

APPENDIX B

SITE MAPS

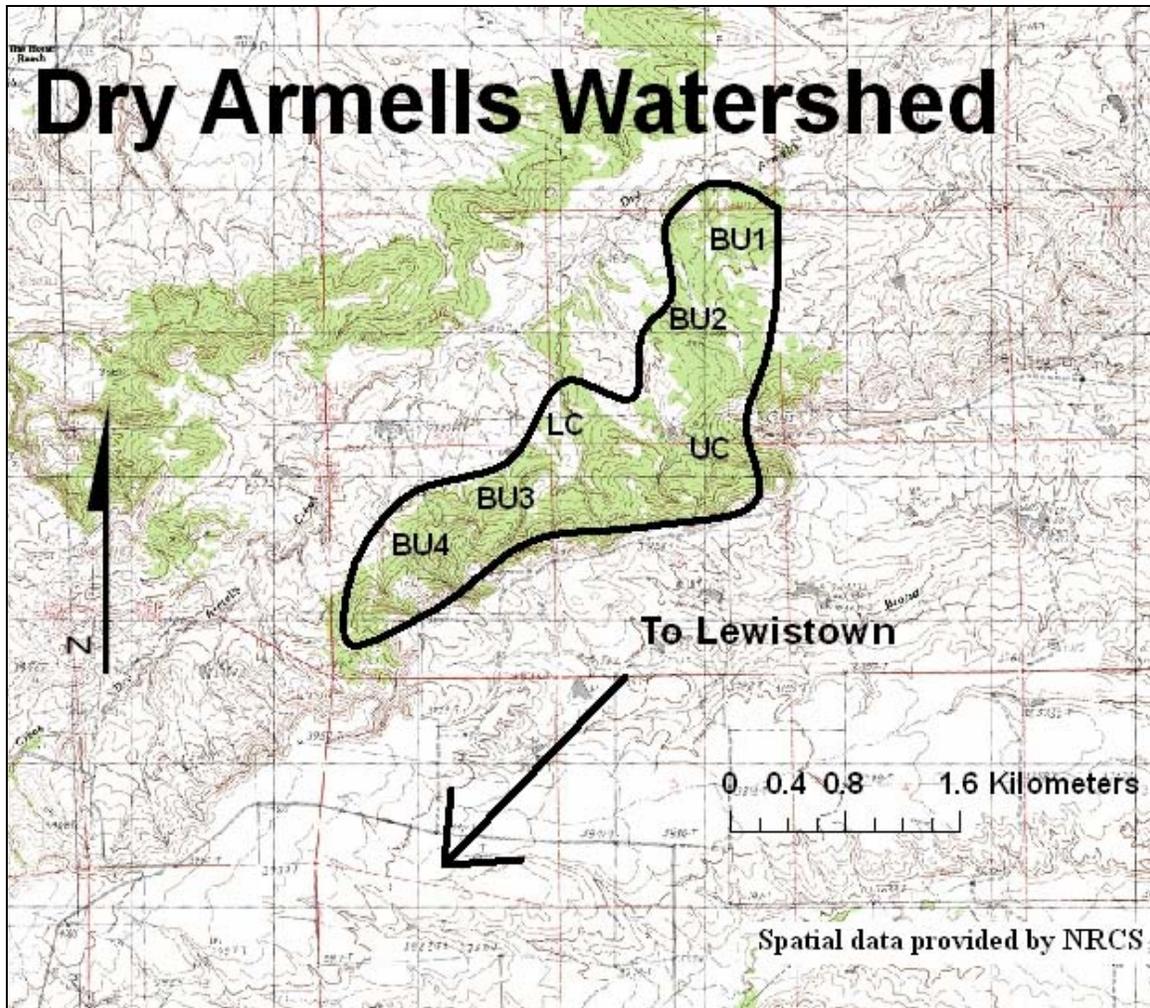


Figure 22. Map of the Dry Armells watershed (Montana). BU = Burn Unit. UC = Upper Control. LC = Lower Control.

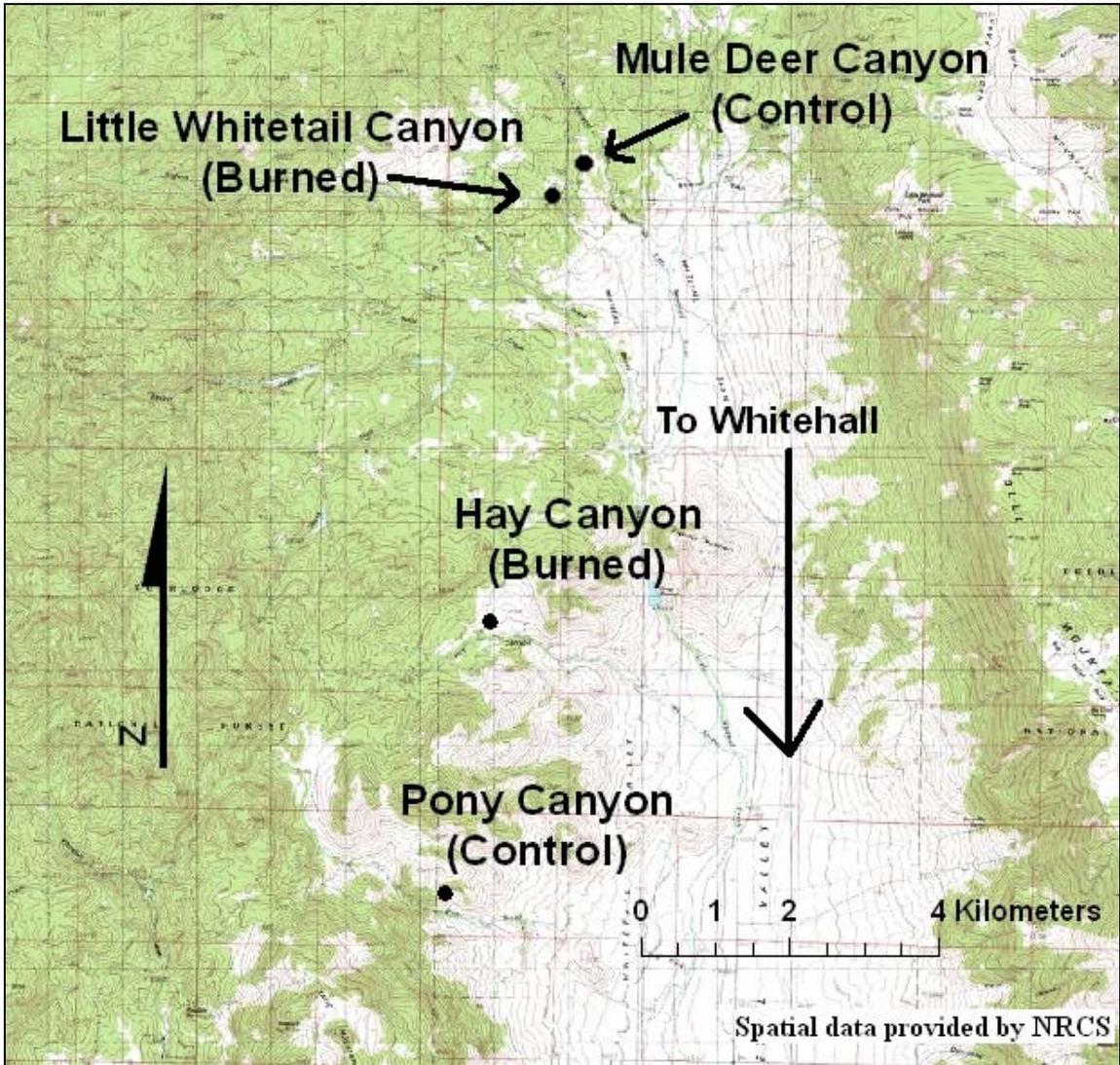


Figure 23. Overview map of Whitehall watersheds (Montana).

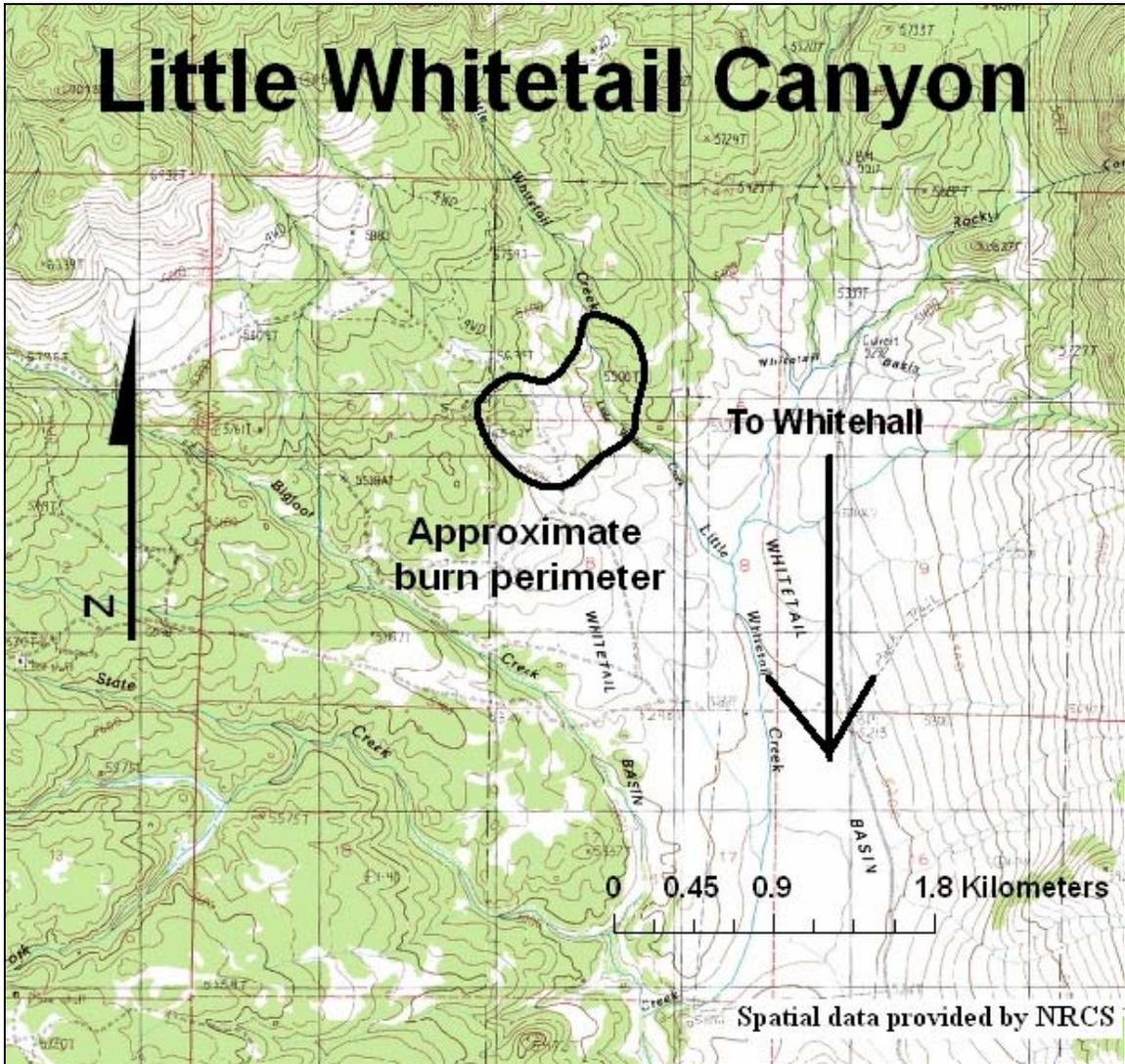


Figure 24. Map of Little Whitetail canyon (Montana) with approximate burn perimeter.

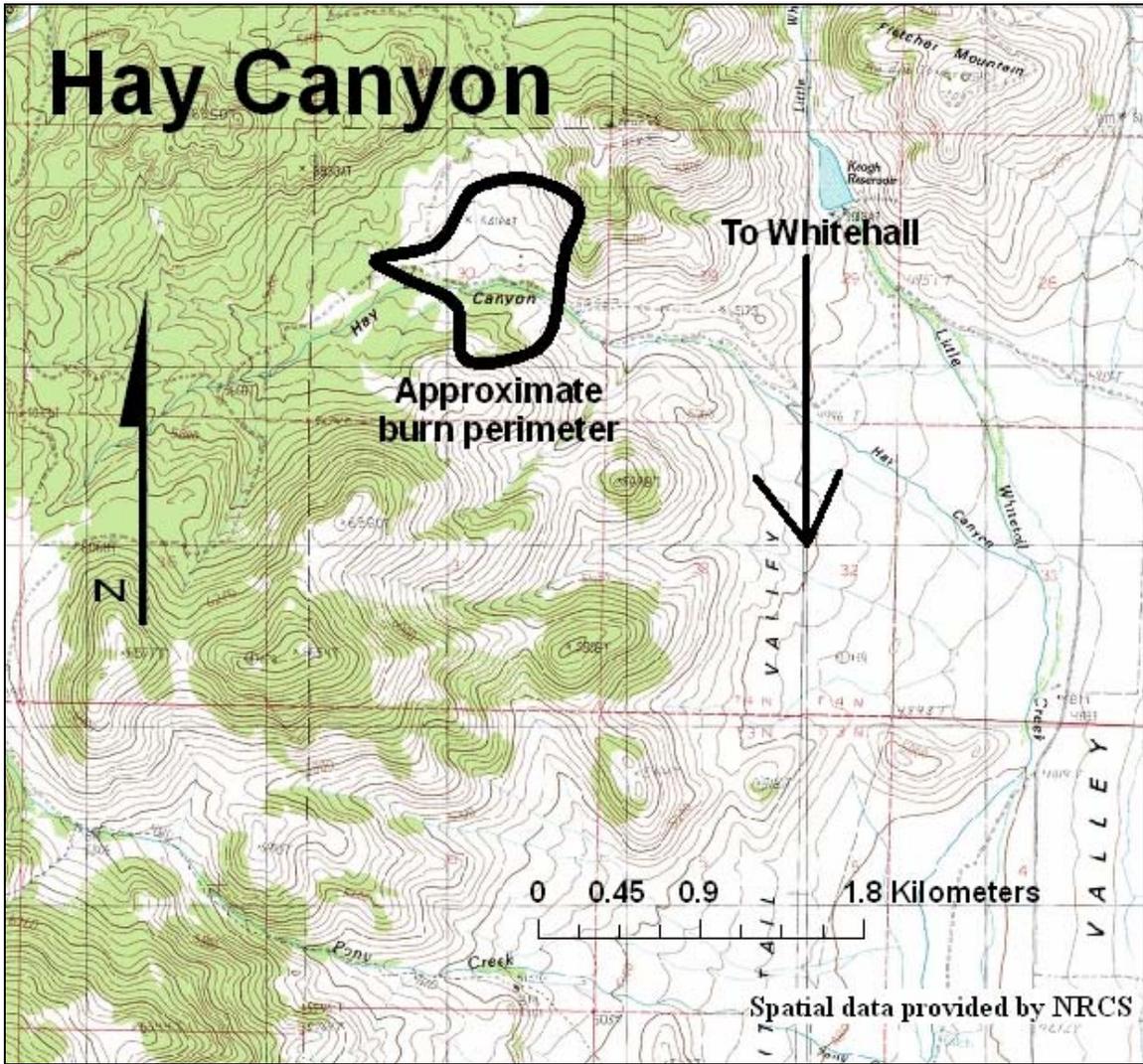


Figure 25. Map of Hay canyon (Montana) with approximate burn perimeter.