MULTI-SCALE ASSESSMENT OF THE EFFECT OF LIVESTOCK GRAZING ON REMOTELY-SENSED BURN SEVERITY UNDER WILDFIRE CONDITIONS

A Thesis

Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Rangeland Ecology and Management

in the

College of Graduate Studies

University of Idaho

by

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May 2010

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AUTHORIZATION TO SUBMIT THESIS

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ABSTRACT

In North America, the range and quality of sagebrush *(Artemisia* spp.*)* habitat has been declining. Large, severe wildfires burning in hot, dry, and windy conditions, exacerbated by invasive species, are among the greatest contributing factors to this decline. Livestock grazing has been proposed as one management option to reduce fire hazard and decrease burn severity across sagebrush steppe. Remotely-sensed burn severity indices have proven useful tools to assess ecological effects of wildfire on forest vegetation, but little research has been conducted in rangelands. We tested the effectiveness of remotely-sensed burn severity indices including dNDVI, RdNBR, and dNBR in detecting changes in vegetation canopy cover on the Murphy Wildland Fire Complex in southern Idaho, a 263,862 ha fire that burned in 2007. Field data of vegetation cover collected by the Bureau of Land Management in 2006 were compared to field data collected in 2009 and the remotely-sensed burn severity indices of RdNBR (Adjusted R-squared = 42%) and dNBR (Adjusted R-squared = 57%) were the most useful at detecting fire-induced changes in vegetation cover on rangelands. Further, we tested the effect of livestock grazing on burn severity using randomly selected plots in grazed and ungrazed pastures on the Murphy Wildland Fire Complex using ArcGIS overlay analysis across three spatial scales. While burn severity was lower in grasslands than shrub communities ($p<0.10$), there were no differences in burn severity between the three scales ($p>0.10$). However, grazing reduced RdNBR and dNBR burn severity indices ($p<0.10$), and an interaction between vegetation type and grazing status showed that the effect of grazing was different in shrub and grassland vegetation types ($p<0.10$). A paired t-test in shrub and perennial grass vegetation types further investigated the interaction, and livestock grazing was found to reduce burn severity indices of RdNBR and dNBR in shrub communities ($p<0.10$), but not in grasslands ($p>0.10$). While livestock grazing can be used as a tool for land managers to reduce burn severity on shrub steppe rangelands in southern Idaho, livestock grazing is not appropriate to use in all vegetation types and management goals should always be taken into consideration.

ACKNOWLEDGMENTS

This project could not have come together without the help of many people. I am sincerely indebted to Dr. Eva Strand for her guidance and assistance on this project. Thank you Eva, for always being available and offering your support and advice. This project would have been impossible without you. I am also grateful for the help and guidance of Dr. Karen Launchbaugh in getting me started in Rangeland Ecology and Management. Your enthusiasm for the field is infectious. I am also thankful for my other committee members, Dr. Penny Morgan and Chad Hoffman and greatly appreciate their support through this process.

I am further appreciative of the many people with the BLM for helping me gather data on this project. Especially, I would like to thank Patty Courtney, Danelle Nance, and Jesse German for their help on this project. A special thanks to Burt Brackett for taking us on a tour of his allotments on the Murphy Wildland Fire Complex. I am grateful for the help and support of Virginia Harris in our field work data collection efforts. Also big thanks to my mother Rae Morris, other family members and friends for their support through this process. This journey would have been impossible without you!

Finally I would like to thank the David Little Livestock Range Management Endowment for the funding and background support making this project possible.

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INTRODUCTION

In North America, the most prominent semiarid vegetation type is the sagebrush *(Artemisia* spp.*)* steppe (Anderson and Inouye 2001). However, the range and quality of sagebrush communities in the sagebrush steppe have been steadily declining (Knick 1999, Bunting et al. 2002). The sagebrush steppe ecosystem historically covered about 63 million ha in western North America (Miller and Eddleman 2001), but today about 20-30 million ha of sagebrush steppe remains (Wright et al. 2001, Connelly et al. 2004). Large, severe wildfires burning in hot, dry, and windy conditions, exacerbated by invasive species, are among the greatest contributing factors to this decline. (Knick 1999, Anderson and Inouye 2001). Fuel reduction treatments such as livestock grazing and implementation of fuel breaks may help reduce the threat of large severe wildfires, and maintain the function and extent of sagebrush steppe ecosystems (Davison 1996, Weber et al. 2004, Nader et al. 2007).

IMPORTANCE OF SAGEBRUSH STEPPE

The decline of sagebrush communities is of great concern because of its importance as wildlife habitat (Beck and Mitchell 2000, Siegel Thines et al. 2004, Rowland et al. 2006), economic significance as grazing land, recreation value and aesthetic value (Laycock 1979). Reduction of sagebrush steppe has already negatively impacted several wildlife species such as the sage grouse *(Centrocercus urophasianus;* Rowland et al. 2006) and pygmy rabbit *(Brachylagus idahoensis;* Siegel Thines et al. 2004).

Wildlife Habitat

One of the major concerns in the decline of sagebrush steppe ecosystems is the loss of essential wildlife habitat. Sagebrush steppe ecosystems provide crucial habitat for many wildlife species including pygmy rabbits, sage sparrow *(Amphispiza belli)*, sage thrasher *(Oreoscoptes montanus)*, sage grouse and Brewer's sparrow *(Spizella breweri;* Rowland et al. 2006). These species as well as other sagebrush obligate species depend on sagebrush habitats for cover, nesting habitat, protection from predators and food during at least part of the year (Paige and Ritter 1999). For example, the diet of pygmy rabbits is 82-99% sagebrush in the winter months and 10-50% sagebrush in the summer season (Siegel Thines et al. 2004).

In addition to obligate species, a variety of other wildlife species reside in sagebrush steppe habitats as well. Species such as mule deer *(Odocoileus hemionus)*, coyotes *(Canis latrans)*, badgers *(Taxidea taxus)*, bobcats *(Felis rufus)*, mountain lions *(Felis concolor)* and pronghorn (*Antilocapra americana*) use sagebrush steppe habitat throughout the year, while others such as elk *(Cervus elaphus)* and bighorn sheep *(Ovis canadensis)* rely on sagebrush for winter habitat (Rowland et al. 2006, BLM Jarbidge Field Office 2007).

As the historical range of sagebrush steppe ecosystems continues to decline (Knick 1999, Bunting et al. 2002), many sagebrush obligate species have been listed as threatened, endangered or as sensitive species. Pygmy rabbits are endangered in the state of Washington (U.S. Fish and Wildlife Service 2009), and are listed as a sensitive species in Idaho (Idaho Department of Fish and Game 2009). Sage grouse are also listed as a state threatened species in Washington (Washington Department of Fish and Wildlife 2009) and a state sensitive species in Idaho (Idaho Department of Fish and Game 2009).

Grazing Land

In addition to providing key wildlife habitat, sagebrush steppe ecosystems have tremendous economic importance as grazing land (Laycock 1979). About 70% of the existing sagebrush steppe ecosystem is on public lands, including lands managed by the U.S. Forest Service, Bureau of Land Management (BLM), U.S. Fish and Wildlife Service, state agencies, and counties (Wright et al. 2001). These areas are largely protected from urban development, but are subject to resource extraction practices such as grazing (Wright et al. 2001). While sagebrush is not a primary forage species for livestock, understory grasses and forbs in a sagebrush steppe ecosystem provide valuable livestock forage and can be important dormant season forage (Laycock 1979).

Recreational and Aesthetic Value

Recreational opportunities abound on sagebrush steppe ecosystems, and are an important consideration when managing sagebrush steppe (Laycock 1979, Hodgkinson 1989). Hiking, horseback riding, bird watching, photography, and All Terrain Vehicle (ATV) use are among the popular activities on sagebrush steppe ecosystems (Laycock 1979, Box 2006). Diverse plant and animal species draw people to these areas and add to the aesthetic value of the land (Laycock 1979, Harris 1991). Additionally, sagebrush steppe ecosystems are often characterized by large tracts of open space and unique geological sites which add to the beauty of the land and attract many visitors each year (Harris 1991).

FACTORS CONTRIBUTING TO SAGEBRUSH DECLINE

Decline of sagebrush steppe ecosystems in North America has been attributed to habitat fragmentation (Anderson and Inouye 2001), invasion by non-native species, especially annual grasses, (Knick 1999, Miller and Rose 1999, Anderson and Inouye 2001, Brooks et al. 2004), woodland species encroachment (Miller and Rose 1999) and altered fire regimes (Houston 1973, Miller and Rose 1999, Baker 2006). In addition to these factors, the increase of large fires burning in hot, dry, and windy conditions (Knick 1999, Anderson and Inouye 2001) and the invasion of cheatgrass *(Bromus tectorum)* have been some of the greatest influences on the degradation of sagebrush steppe ecosystems in recent decades (Knick 1999, Baker 2006).

Habitat Fragmentation

Much of the sagebrush steppe ecosystem was historically subjected to excessive grazing pressure in the late 1800s and early 1900s which extensively modified these ecosystems (Laycock 1967, Burkhardt 1996, Knick 1999). Historically, tens of thousands of domestic sheep *(Ovis aries),* cattle *(Bos taurus)* and horses *(Equus caballus)* grazed in sagebrush habitats (Burkhardt 1996). Excessive grazing typically influenced sagebrush ecosystems by compacting the soil, decreasing the grass and forb cover, and exposing bare mineral soil which allowed for the invasion of non-native plants such as cheatgrass and medusahead *(Taeniatherum caput-medusae;* Laycock 1967, Burkhardt 1996, Knick 1999).

In addition, many historical range management strategies involved removal of sagebrush as range improvement measures (Little 1992). Sagebrush was removed from rangelands by mechanical control, chemical control, and prescribed fire in order to promote grass beneficial for livestock grazing and some wildlife species (Davison 1996, Nader et al. 2007). As a result, many sagebrush steppe ecosystems were converted to grasslands during the 1900s (Little 1992).

Habitat fragmentation of sagebrush steppe ecosystems is aided by conversion of sagebrush lands to agriculture (Handl and Heilig 1980), human residential occupation (Frandsen 2008) and energy developments (Braun et al. 2002, Knick et al. 2003). Increased technology allows for the expansion of irrigation to many arid sagebrush areas, contributing to the overall loss of sagebrush ecosystems (Handl and Heilig 1980). Additionally, as human populations increase in the western United States, more land is continually being converted from sagebrush steppe to housing and other human developments (Frandsen 2008). Energy development has increased in sagebrush ecosystems in North America, leading to habitat fragmentation compounded by abundant supporting infrastructure including roads, pipelines, and power lines (Braun et al. 2002, Knick et al. 2003). The combination of all these factors yields significant habitat loss and fragmentation in sagebrush steppe ecosystems (Handl and Heilig 1980, Braun et al. 2002, Knick et al. 2003, Frandsen 2008).

Non-Native Species Invasion

Since the early 1900s, cheatgrass and other non-native invasive plant species have been increasing in abundance in sagebrush steppe rangelands across western North America (Knick and Rotenberry 1997, Knick 1999). The increase of non-native invasive species has further led to habitat degradation in sagebrush ecosystems (Knick 1999). Invasive species are able to move into sagebrush ecosystems after disturbances such as fire (Melgoza et al. 1990). Unlike native bunchgrasses, invasive annuals often create continuous fuels in the understory of sagebrush ecosystems. In addition, species such as cheatgrass cure out early in the season, and create flashy fuels (Melgoza et al. 1990) perfect for the spread of fires (Ziska et al. 2005). In what has been called the "grass-fire" cycle, fires readily spread in the continuous fuels, creating ideal conditions for more grass to grow (Vitousek et al. 1996). In a positive feedback, abundant grass in turn fuels more large fires that can burn frequently because the fuels are heavy enough to carry fires in sagebrush habitats (Knick and Rotenberry 1997, Brooks et al. 2004, Ziska et al. 2005).

Woodland Species Encroachment

Another factor which leads to the degradation of sagebrush steppe ecosystems is woodland species encroachment (Miller and Rose 1999). In the last few decades, woody species such as juniper (*Juniperus* spp.) have been expanding into sagebrush habitats, eventually out-competing and eliminating sagebrush (Miller and Rose 1999). Factors leading to woody species expansion include fire suppression, climate change and heavy livestock grazing (Miller and Rose 1999). As woody species continue to expand into rangelands, valuable sagebrush habitat is fragmented and sometimes nearly eliminated from the landscape (Miller and Rose 1999).

Changed Fire Regimes

Currently, there is conflicting information regarding historical fire regimes in sagebrush steppe. In an examination of research by Houston (1973), Young and Evans (1981), Arno and Gruell (1983), Miller and Rose (1999), and others, Baker (2006) estimated that fire rotation in sagebrush steppe varied, with a rotation of 325-450 years in low sagebrush *(Artemisia arbuscula),* 100-240 years in Wyoming big sagebrush *(Artemisia tridentata* ssp. *wyomingensis)* 70-200 years in mountain big sagebrush *(Artemisia tridentata* ssp. *vaseyana),* and 35-100 years in grasslands with sagebrush as a minor component. Wright and Bailey (1982) estimated that fire return intervals in mountain big sagebrush were around 50 years, while fire return intervals in Wyoming big sagebrush were around 100 years. Other research has indicated that low intensity fires were historically more frequent in sagebrush steppe, occurring every 20-30 years (Houston 1973, Knick 1999). When mountain big sagebrush occurred with or near trees where fire scar records could be used to estimate historical fire regimes, fire intervals were estimated to be 30-40 years (Arno and Gruell 1983) and 12-15 years (Miller and Rose 1999). When cheatgrass and other annual grasses are a dominant component of the understory in sagebrush, research has shown that fire frequency increases (Brooks and Pyke 2001).

In the early and mid 1900s, humans greatly altered the natural fire regimes of sagebrush ecosystems by heavy grazing of livestock and suppressing fire. This resulted in long intervals between fires at any given location (Wrobleski and Kauffman 2003). Consequently, woody plant species increased, while grasses and forbs decreased (Burkhardt 1996). After decades of woody fuel accumulation and fire suppression, recent fire activity has been increasing (Knick 1999, Anderson and Inouye 2001). Climate change (Running 2006) and the increase of annual invasive species are other factors contributing to more frequent large-scale wildfires (Knick 1999, Baker 2006). The current warming climate trend has resulted in earlier snow melt and dryer fuels in the spring which can create a longer fire season

(Westerling et al. 2006). The combined effect of annual species such as cheatgrass creating a continuous fuel load in the understory (Melgoza et al. 1990), changing climate (Westerling et al. 2006), and woody fuel accumulation will help to continue the trend of increasingly large, severe fires in the future (Burkhardt 1996).

FIRE EFFECTS ON VEGETATION IN SAGEBRUSH COMMUNITIES

Perennial grasses and shrub species within sagebrush communities respond differently to fire disturbances. Sagebrush is not fire tolerant and it typically does not survive after a burn (White and Currie 1983, Sapsis and Kauffmann 1991, Baker 2006). Additionally, sagebrush generally takes between 30 and 100 years to recover after a burn (Baker 2006). Perennial grasses are often able to respond quickly post-fire. Perennial grass response to wildland fire is often a factor of plant life history traits, such as the location of meristematic tissue (Conrad and Poulton 1966), as well as environmental conditions (Redmann 1978, Daer and Willard 1981, Wright and Bailey 1982, Rhodes 2006, Davies et al. 2007). Different shrub and perennial grass species are characterized by unique responses and ability to recover from fire disturbance.

Perennial Grass Response to Fire

Bluebunch wheatgrass *(Pseudoroegneria spicata)* often responds rapidly after the fire, with recovery evident typically one to three years post-fire (Bunting et al. 1998). Plant morphological traits such as coarse stems and a lack of leafy material often contribute to rapid burning with little heat penetration to the soil (Young 1983, Zamora 1989). Bluebunch wheatgrass is less vulnerable to heat damage because meristematic tissue is often located on buds of horizontal stems below the surface of the soil (Conrad and Poulton 1966). Timing of the burn is another critical factor in determining how bluebunch wheatgrass will respond to the fire, with the greatest amount of damage after fires occurring during the active growing season (McShane and Sauer 1985, Zamora 1989, Sapsis 1990). Other environmental factors that are key in the rapid response of bluebunch wheatgrass after a fire include adequate soil moisture in the growing season following the burn (Robberecht and Defosse 1995).

Several researchers suggest that Idaho fescue *(Festuca idahoensis)* is more susceptible than bluebunch wheatgrass to fire because meristemtatic tissue is located in the root crown, above the surface of the soil (Blaisdell 1953, Conrad and Poulton 1966, Wright et al. 1979). Idaho fescue typically has a greater amount of fine leaves, which may accumulate and burn at the base of the plant producing high enough temperatures to damage or kill the plant (Agee 1996). Root and Haybeck (1972) suggest that the season of burning has less effect on Idaho fescue, with damage occurring throughout all phenological stages. Conversely, other research with controlled fire indicates that while meristematic tissue damage may occur, Idaho fescue responds more rapidly and initiates growth quicker than bluebunch wheatgrass (Robberecht and Defosse 1995, Defosse and Robberecht 1996). Similar to bluebunch wheatgrass, Idaho fescue recovery post-fire is also dependant on availability of moisture in the growing season after the burn (Blaisdell 1953, Wright 1974, Robberecht and Defosse 1995).

Likewise, crested wheatgrass *(Agropyron cristatum)* tends to respond quickly following a fire (Young 1983). Meristematic tissue is often undamaged or only slightly damaged after a fire as a result of deep underground tillers which are insulated from high temperatures during the burn (Bradley et al. 1992). A lack of leafy material at the base of the plant often promotes rapid burning with little heat transfer to the soil (Ralphs and Busby 1979). Season of burn is also important in determining crested wheatgrass recovery post-fire, with most damage occurring during the growing season and favorable responses during the late summer dormant season (Bradley et al. 1992).

Sandberg bluegrass *(Poa secunda)* also is typically undamaged by fire, with rapid recovery following a burn. Wright and Klemmedson (1965) found that Sandberg bluegrass with a basal diameter of 2.5-7.6 cm experienced no size reduction following June, July and August burns. Sandberg bluegrass plants with larger basal diameters conversely were more susceptible to fire damage due to more litter accumulation at the base, and more heat transfer into the soil (Wright and Klemmedson 1965). Additionally, Sandberg bluegrass has been shown to increase during the first year following a fire, and produce more flowering stalks than in unburned areas (Mitchell 1957).

Other factors contributing to the increase of bunchgrasses after a burn include reduced competition for nutrients and water due to the removal of sagebrush (Rhodes 2006), and more available inorganic nitrogen after a burn (Davies et al. 2007). Similarly, Brockway et al. (2002) found that burning during the dormant season on shortgrass prairie resulted in increased cover of graminoids and forbs. Burning in the dormant season also resulted in more phosphorous, potassium, calcium, magnesium, and manganese in the foliar cover of plants than burning during the growing season (Brockway et al. 2002).

Shrub Response to Fire

Wyoming big sagebrush has low tolerance to fire, and is often completely eliminated after a burn (White and Currie 1983, Sapsis and Kauffman 1991, Baker 2006). Big sagebrush also does not resprout readily after a fire (Young and Evans 1978). Young and Evans (1978) found no resprouting sagebrush after fire across three sites in northern Nevada. Lesica et al. (2007) found that Wyoming big sagebrush would take on average, 33 years to recover to pre-fire canopy cover levels. Similarly, a Montana study found that Wyoming big sagebrush had only recovered to 12% of its original canopy cover 18 years after a prescribed burn (Wambolt and Payne 1986). Recovery of sagebrush steppe post-fire is often considered to be related to the proximity of seed sources (Blaisdell 1953). However, other studies indicate that Wyoming big sagebrush did not recover faster in areas closer to a seed source after a burn (Wambolt and Payne 1986, Lesica et al. 2007). Further, timing of the burn is critical in sagebrush ecosystems. Spring burning can result in shorter flame lengths, lower rates of spread, and lower intensity in sagebrush than fall or summer burns (Sapsis and Kauffmann 1991).

Mountain big sagebrush similarly has low tolerance to fire, and often is removed from an ecosystem after a fire (Wambolt et al. 2001). While post-fire recovery may proceed slowly in the first few years after a burn (Harniss and Myrray 1973), especially if burn severity is high (Bunting et al. 1987), it often begins to increase rapidly and return to original density 15 to 20 years after a burn (Bunting et al. 1987). However, other studies indicate that mountain big sagebrush does not fully recover to pre-burn densities for up to 30 years after a fire (Wambolt et al. 2001). Research by Young and Evans (1989) suggests that mountain big sagebrush has a very small amount of seed in the seedbank, as evidenced by low seedling germination (10/ha) in the first growing season following an August wildfire in western Nevada.

Rabbitbrush *(Chrysothamnus* spp.*)* has varied responses to fire depending on the species. The species present in this study include rubber rabbitbrush *(Chrysothamnus nauseosus)* and green rabbitbrush *(Chrysothamnus viscidiflorus)*. Both species of rabbitbrush are typically subject to removal of above ground biomass during a fire (Martin and Dell 1978). However, the removal of vegetation after the fire may enable both green and rubber rabbitbrush to be released from competition and begin to resprout from roots and seeds (McKell and Chilcote 1957). While green rabbitbrush is characterized by vigorous sprouting after a fire (Young and Evans 1974, Ralphs and Busby 1979, Akinsoji 1988), it may not become established as quickly on a site as grass and forbs (Akinsoji 1988). There is evidence that rubber rabbitbrush also sprouts vigorously after fire (Ralphs and Busby 1979). However, other studies found that rubber rabbitbrush does not respond as well to fire with little resprouting in the growing seasons post-fire (Robertson and Cords 1957, Johnson and Strang 1983).

FUEL MANGMENT METHODS IN SAGEBRUSH COMMUNITIES

In order to reduce the size and severity of fires in sagebrush steppe, several treatments to decrease fuel loading have been used including mechanical control, chemical control, hand treatment, prescribed fire, and livestock grazing (Parker 1979, Ueckert et al. 1988 D'Antonio et al. 1998, DiTomaso 2000). Effectiveness of fuel treatments varies with site conditions and specific methods applied (Davison 1996, Nader et al. 2007).

Chemical Control (Herbicides)

Chemical control with herbicides can effectively reduce fuel loads in sagebrush steppe ecosystems (DiTomaso 2000). Herbicides such as tebuthiuron can be used to thin sagebrush in dense stands, and promote a healthy grass understory (McDaniel et al. 2005). In this case, sagebrush fuels can be reduced without completely eliminating it from the ecosystem (McDaniel et al. 2005).

Chemical control of invasive species is one of the most commonly used methods to reduce the abundance of invasive species and reduce fuel loading (DiTomaso 2000). Herbicide application can be accomplished by aircraft, helicopter, vehicle, backpack sprayers and rope wick applicators (DiTomaso 2000). However, there are several drawbacks to using chemical controls to reduce fuel loading, including possible negative effects to the environment (Davison 1996), and the continuous applications that are often needed to achieve desired effects (McDaniel et al. 2005). For example, without continuous application of chemicals, sagebrush was able to respond to pre-treatment cover levels within 20 years of treatment on three of eight sites in New Mexico (McDaniel et al. 2005).

Mechanical Control

Mechanical control using heavy equipment, disking, or plowing is also used to control fuel loading in sagebrush steppe (Parker 1979). While it can be effective at eliminating undesirable vegetation, it is often difficult to operate machinery in uneven terrain (Davison 1996). Additionally, mechanical treatment can have a heavy impact on the environment, and disturb the soil, leaving perfect conditions for non-native species establishment (Parker 1979).

Hand Treatment

Treating sagebrush fuels by hand is an effective alternative to mechanical treatment because there is less impact on the land (Davison 1996, Nader et al. 2007). In addition, hand treatments can be utilized on a variety of terrain and are very selective for undesirable plant species (Davison 1996). D'Antonio and colleagues (1998) found that hand removal of undesirable species was able to promote native species growth within 18 months of treatment. However, it is difficult to cover large areas with hand treatments and it can be very costly to conduct (Davison 1996, Nader et al. 2007).

Prescribed Fire

Prescribed fire is one of the most common tools used to reduce fuels on rangelands (Davison 1996, Nader et al. 2007). Prescribed burns can be very effective at eliminating undesirable plant species such as prickly pear cactus *(Opuntia* spp*;* Ueckert et al. 1988), medusahead (McKell et al. 1962) and yellow starthistle *(Centaurea solstitialis;* DiTomaso et al. 1999). Additionally, prescribed fire can reduce woody plant species such as sagebrush. Sagebrush species are intolerant to fire, and are most often completely eliminated for several years after burns (White and Currie 1983, Sapsis and Kauffmann 1991, Baker 2006). Timing of prescribed burning is critical in sagebrush ecosystems. Spring burning in sagebrush can result in shorter flame lengths, lower rates of spread, and lower fire intensity than fall burns (Sapsis and Kauffmann 1991). Prescribed burning is often very cost effective, but also requires skilled personnel and planning to properly be conducted (Davison 1996, Nader et al.

2007). Other negative consequences of prescribed burning include air pollution from smoke, reduced aesthetic value, and liability issues with fire escape (Nader et al. 2007). Further, annual invasive species such as cheatgrass may increase after a prescribed fire (Young and Evans 1978).

Livestock Grazing

Livestock grazing is one of the most effective tools for reducing fuel loading on sagebrush steppe ecosystems (Taylor 2006, Nader et al. 2007). Historically, heavy grazing negatively affected ecosystems through soil erosion, soil compaction, and altering vegetation composition (Laycock 1967, Burkhardt 1996). However, properly setting livestock grazing numbers on sagebrush rangelands results in lower environmental damage, and additionally reduces fuels (Davison 1996, Taylor 2006). Fuel loads are reduced by livestock grazing by the direct removal of vegetation and trampling of litter (Taylor 2006, Nader et al. 2007).

Unlike prescribed fire, livestock grazing rarely kills sagebrush species as the grass understory is mainly targeted (Laycock 1979). Weber et al. (2004) found that higher stocking rates were more effective at reducing fine fuel loading than lower stocking rates. Models show that livestock grazing can increase the heterogeneity of rangelands, which in turn can decrease the risk of large fires (Kerby et al. 2007). Utilizing diverse species of livestock during different seasons can be useful in targeting specific species of vegetation to remove from an area (Davison 1996). Additionally, livestock grazing is cost effective, is available throughout western North America, and personnel in public agencies are familiar with livestock management (Davison 1996). Negative effects of using livestock grazing to reduce fuel loading include reduced water quality, riparian vegetation damage, soil compaction, disease transmission to wildlife, and spread of non-native invasive species (Taylor 2006, Nader et al. 2007).

BURN SEVERITY

Burn severity is a term frequently used to describe the degree of ecological change due to a fire (Key and Benson 2006, Lentile et al. 2006, Keeley et al. 2009). Unlike fire intensity, which describes energy released along a flaming front, burn severity describes the degree of change in vegetation and soil characteristics as a result of a fire, and the amount of time needed to return to pre-fire vegetation levels or function (Lentile et al. 2006). Burn severity can alternatively be defined as the immediate aboveground and belowground organic material loss due to fire, in which ecosystem responses post-fire are separated from the definition of burn severity (Keeley et al. 2009). Burn severity is often measured remotely using satellite imagery and can also be measured from ecological changes occurring in the field (Key and Benson 2006). The Composite Burn Index (CBI) is a common field measured index of burn severity which focuses on changes in soil and vegetation properties after a fire (Key and Benson 2006).

REMOTELY-SENSED BURN SEVERITY INDICES

Remote sensing techniques can be useful tools to assess burn severity within fire perimeters. While many remote sensing indices and sensors can be used to detect changes in vegetation as a result of wildfire, the most common techniques compare differences in spectral or thermal bands before and after a fire (Lentile et al. 2006). Some of the most commonly used burn severity indices include the Normalized Difference Vegetation Index (NDVI), Normalized Burn Ratio (NBR), delta NBR (dNBR), and the Relative dNBR (RdNBR; Key and Benson 2006, Norton 2006, Miller and Thode 2007).

The traditional index used in quantifying burn severity is the NDVI (Salvador et al. 2000, Diaz-Delgado et al. 2003). Developed by Rouse and colleagues (1973), the NDVI compares the near infrared band (NIR) to the red band (R) associated with leaf tissue (Washington-Allen et al. 2008; equation 1).

$$NDVI = \frac{NIR - R}{NIR + R} \qquad (1)$$

Characteristics of vegetation such as cover, leaf area index, and phytomass have all been postively correlated with NDVI (Sellers 1985). The NDVI can be used to detect burn severity by comparing satellite imagery before and after a burn, and detecting differences in live green vegetation (Norton 2006). Values for the NDVI vary from -1 to +1 (Norton 2006).

A second remote sensing technique to assess burn severity was developed by Lopez Garcia and Caselles (1991). This technique later was termed Normalized Burn Ratio (NBR) by Key and Benson (Key and Benson 1999, Key and Benson 2006, Norton 2006). The NBR (equation 2) was calculated by comparing Landsat band 4 (near-infrared 0.76-0.90 μm) which is reflective of chlorophyll in live vegetation (Lopez Garcia and Caselles 1991, Key and Benson 2006) to Landsat band 7 (middle infrared 2.08-2.35 μm) which is reflective of ash, bare soil, and charred material present after a fire (Jia et al. 2006).

$$NBR = \frac{(band4) - (band7)}{(band4) + (band7)} \qquad (2)$$

On a temporal scale, it is often useful to compare vegetation from before and after the burn. The delta NBR is calculated by subtracting the post-fire NBR from the pre-fire NBR (Key and Benson 1999; equation 3). Data values for NBR range from -1 to + 1, but are usually multiplied by 1000 to convert to the integer format (Key and Benson 2006).

$$dNBR = NBRprefire - NBR\ postfire \tag{3}$$

Since 1999, public land agencies have been using the dNBR index to assess burn severity, and it is the most frequently used index, especially on large fires (Cocke et al. 2005, Key and Benson 2006). Data values for dNBR range from -2 to +2, and like NBR are multiplied by 1000 so that values range from -2000 to +2000 (Key and Benson 2006). Positive dNBR values represent decreased vegetation from pre-fire to post-fire, while negative dNBR values respresent an increase in vegetation (Key and Benson 2006).

Miller and Thode (2007) developed an algorithm that is a relative version of dNBR (equation 4). Relative dNBR is calculated by subtracting the post-fire NBR from the pre-fire relative NBR.

$$RdNBR = \frac{prefireNBR - postfireNBR}{\sqrt{(ABS(prefireNBR/1000))}} \tag{4}$$

Relative dNBR is sensitive to the amount of vegetation killed by the fire in comparison to the pre-fire vegetation cover (Miller and Thode 2007). Relativising the dNBR index thus eliminates correlation to the pre-fire NBR, and is more sensative to low amounts of pre-fire biomass (Miller and Thode 2007). Like dNBR, the RdNBR data values range from -2 to +2, and are often multiplied by 1000 so that values range from -2000 to +2000 (Miller and Thode

2007). Positive RdNBR values represent a decrease in vegetation while negative RdNBR values typically represent an increase in vegetation (Miller and Thode 2007).

POTENTIAL EFFECTS OF LIVESTOCK GRAZING ON BURN SEVERITY

Livestock grazing can reduce fuel loading on rangelands. Madany and West (1983) found that livestock grazing reduced the herbaceous fuel loading while increasing woody species growth in ponderosa pine (*Pinus ponderosa)* and Gambel oak (*Quercus gambelii*) woodlands. Weber et al. (2004) found that livestock grazing can reduce the fine fuel load component of herbaceous material in sagebrush communities, but was most effective at high stocking rates, or in combination with previous wildfires. Additionally, intensive livestock grazing in strips can be used to create fuel breaks (Green et al. 1979). For example, goats have been used in California to control brush and maintain fuel breaks to stop the spread of fire (Green et al. 1979). Conrad and Poulton (1966) suggest that reducing fuel loading around bluebunch wheatgrass and Idaho fescue by utilizing grazing, may decrease the amount of smoldering at the base of plants due to the lack of fuel, thus, decrease fire effects.

Livestock grazing also has the potential of creating a patchwork of grazed and ungrazed areas across a landscape (Kerby et al. 2007). Wildland fires such as the Murphy Wildland Fire Complex which burned over 260,000 ha of sagebrush rangelands in southern Idaho in 2007, typically burn in a mosaic pattern across a landscape (Launchbaugh et al. 2008). Observations of burn severity contrasts along fence lines on the Murphy Wildland Fire Complex indicated that livestock grazing may be influencing the landscape heterogeneity of a wildland fire effects inferred from satellite imagery (Launchbaugh et al. 2008). The burn severity in some fence line contrasts was apparently lower on pastures that were grazed before the fire (Launchbaugh 2008). The decline and fragmentation of sagebrush ecosystems, combined with the expected increase in large fires, have created a need among land managers to better understand this link between livestock grazing and burn severity on rangeland vegetation under wildfire conditions. Although remotely-sensed indices are commonly used for assessing burn severity in both forests and rangelands (e.g. USFS Monitoring Trends in Burn Severity http://www.mtbs.gov/), research establishing clear relationships between remotely-sensed burn severity and post-fire vegetation response in rangelands is still lacking.

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CHAPTER 2: EFFECTS OF LIVESTOCK GRAZING ON REMOTELY-SENSED BURN SEVERITY

INTRODCUTION

In North America, the most prominent semiarid vegetation type is the sagebrush *(Artemisia* spp.*)* steppe (Anderson and Inouye 2001). Sagebrush steppe is vast, covering approximately 20-30 million ha in western North America (Wright et al. 2001, Connelly et al. 2004). These ecosystems are of great concern due to their importance for wildlife (Beck and Mitchell 2000, Siegel Thines et al. 2004, Rowland et al. 2006), economic significance as grazing land, recreation value and aesthetic value (Laycock 1979). Sagebrush steppe ecosystems provide essential habitat for many wildlife species, with sagebrush obligate species dependent on sagebrush habitats during at least part of the year (Paige and Ritter 1999). In addition to providing key wildlife habitat, sagebrush steppe ecosystems have tremendous economic importance as grazing land (Laycock 1979). Lands managed by the U.S. Forest Service, Bureau of Land Management (BLM), U.S. Fish and Wildlife Service, state agencies, and counties comprise about 70% of the existing sagebrush steppe ecosystem. Each of these agencies often includes grazing in their management plans (Wright et al. 2001). While sagebrush is not a primary forage species for livestock, understory grasses and forbs in a sagebrush community are often selected by livestock (Laycock 1979). Recreational opportunities and aesthetic values abound on sagebrush steppe and contribute to the overall importance of this vegetation type in North America (Laycock 1979, Hodgkinson 1989, Harris 1991).

Currently, these ecosystems are of great concern because the range and quality of sagebrush communities in the sagebrush steppe have been steadily declining (Knick 1999, Bunting et al. 2002). The sagebrush steppe ecosystem historically covered about 63 million ha in western North America (Miller and Eddleman 2001), but today only about 20-30 million ha of sagebrush steppe remains (Wright et al. 2001, Connelly et al. 2004). Decline of sagebrush steppe ecosystems in North America has been attributed to habitat fragmentation (Anderson and Inouye 2001), invasion by non-native species, especially annual grasses, (Knick 1999, Miller and Rose 1999, Anderson and Inouye 2001, Brooks et al. 2004), woodland species encroachment (Miller and Rose 1999) and altered fire regimes (Houston 1973, Miller and Rose 1999, Baker 2006). Of these factors, the increase of large fires burning in hot, dry and windy conditions (Knick 1999, Anderson and Inouye 2001) and the invasion of cheatgrass *(Bromus tectorum)* have been among the greatest influences on the degradation of sagebrush steppe ecosystems in recent decades (Knick 1999, Baker 2006).

Historical fire-return intervals of 50 to 100 years have been estimated in Wyoming big sagebrush *(Artemisia tridentata* ssp. *wyomingensis)* communities (Wright and Bailey 1982). Other research however suggests that low intensity fires occurring every 20-30 years in sagebrush were common (Houston 1973, Knick 1999). Sagebrush is not fire tolerant, and it typically does not survive after a burn (White and Currie 1983, Sapsis and Kauffmann 1991, Baker 2006). Additionally, sagebrush usually takes between 30 and 100 years to recover after a burn (Baker 2006).

In the early and mid 1900s, humans greatly altered the natural fire regimes of sagebrush ecosystems by heavily grazing livestock and suppressing fire. This resulted in long intervals between fires at any given location (Wrobleski and Kauffman 2003). Consequently, woody plant species increased, while grasses and forbs decreased (Burkhardt 1996). After decades of woody fuel accumulation and fire suppression, recent fire activity has been increasing (Knick 1999, Anderson and Inouye 2001). Climate change (Running 2006) and the increase of annual invasive species are other factors contributing to more frequent large-scale wildfires (Knick 1999, Baker 2006). The current warming climate trend has resulted in earlier snow melt and dryer fuels in the spring which can also create a longer fire season (Westerling et al. 2006).

In order to reduce the size and severity of fires in sagebrush steppe, several treatments to decrease fuel loading have been used including mechanical control, chemical control, hand treatment, prescribed fire, and livestock grazing (Davison 1996, Nader et al. 2007). Of these factors, livestock grazing has the potential to be one of the most effective methods to reduce fuel loading in sagebrush communities (Davison 1996, Weber et al. 2004, Nader et al. 2007). Livestock grazing can reduce the fine fuel load component of herbaceous material on sagebrush steppe ecosystems (Madany and West 1983, Weber et al. 2004). Grazing also can alter the spatial arrangement of fuels by reducing accumulation at the base of perennial plants, which in turn decreases the amount of smoldering at the base of the plant and reduces fire effects on the vegetation (Conrad and Poulton 1966, Davies et al. 2009). Lack of fine fuels in the understory of sagebrush steppe thus can make it difficult to carry a fire across the landscape (Bunting et al. 1987). Increased vegetation heterogeneity caused by livestock grazing may also be influencing the pattern of burn across the landscape, creating areas of lower burn severity due to fuels reduction (Kerby et al. 2007).

Remote sensing techniques can be useful tools to assess burn severity within fire perimeters. While many remote sensing indices and sensors can be used to detect changes in vegetation as a result of wildfire, the most common techniques compare differences in spectral or thermal bands before and after a fire (Lentile et al. 2006). Some of the most frequently used burn severity indices include the Normalized Difference Vegetation Index (NDVI), Normalized Burn Ratio (NBR), and the Relative NBR (RdNBR; Rouse et al. 1973, Lopez and Caselles 1991, Key and Benson 2006, Norton 2006, Miller and Thode 2007, Washington-Allen et al. 2008). The differences in these indices between pre-fire and post-fire imagery including delta Normalized Difference Vegetation Index (dNDVI), delta Normalized Burn Ratio (dNBR) and Relative delta Normalized Burn Ratio RdNBR are frequently used to determine burn severity and vegetation mortality as a result of fire (Key and Benson 2006, Roy et al. 2006).

While the definition of burn severity is not used consistently in the literature (Lentile et al. 2006, Keeley et al. 2009), burn severity is commonly used to describe the degree of change in vegetation and soil characteristics as a result of a fire, and the amount of time needed to return to pre-fire vegetation levels or function (Lentile et al. 2006). Burn severity can alternatively be defined as the immediate aboveground and belowground organic material loss due to fire, in which ecosystem responses post-fire are separated from the definition of burn severity (Keeley et al. 2009). For the purpose of this research, burn severity is defined as the change in remotely-sensed burn severity indices as a result of wildland fire.

Although remotely-sensed indices are commonly used for assessing burn severity in both forests and rangelands (e.g. USFS Monitoring Trends in Burn Severity http://www.mtbs.gov/), research establishing clear relationships between remotely-sensed burn severity and post-fire vegetation response in rangelands is still lacking. In 2007, the Murphy Wildland Fire Complex burned over 260,000 ha of sagebrush habitat in southern Idaho, providing a unique opportunity to investigate the interaction between livestock grazing and remotely-sensed burn severity. This study hypothesizes that:

- There is a relationship between remotely-sensed burn severity indices and change in vegetation canopy cover measured before and after the Murphy Wildland Fire Complex.
- Livestock grazing prior to the fire reduced burn severity within the Murphy Wildland Fire Complex perimeter.

Specific objectives of this project were to:

1) Gather grazing information within the fire perimeter of the Murphy Wildland Fire Complex regarding the livestock time of use before the fire in 2007.

2) Determine the relationship between remotely-sensed burn severity on the Murphy Wildland Fire Complex by comparing change in field measured vegetation cover from 2006 to 2009 to the dNBR, (Lopez and Caselles 1991, Key and Benson 2006), RdNBR, (Miller and Thode 2007), and the dNDVI (Rouse et al. 1973, Washington-Allen et al. 2008). Pre-fire vegetation data at the species level was collected in 2006 on 494 Ecological Site Inventory

(ESI) points by the BLM. About 160 ESI points were within the Murphy Wildland Fire Complex perimeter. Burn severity immediately after the fire and one year post-fire was measured and this data was used to test hypothesis 1.

3) Determine if remotely-sensed burn severity indices were affected by livestock grazing through a Geographic Information System (GIS) overlay analysis on grazed and ungrazed paired plots in areas currently dominated by shrub and perennial grass cover. Burn severity indices of dNBR, RdNBR and dNDVI measured immediately after the fire and one year post-fire were used in this analysis. This was also done at the plot scale using field collected grazed and ungrazed paired plots, and these data were used to test hypothesis 2.

METHODS

Site Description

Data analysis was conducted on the Murphy Wildland Fire Complex located near the Idaho-Nevada border south of Twin Falls, Idaho. Three wildland fires started on July 16 and 17, 2007 and burned together in this area to create what was later named the Murphy Wildland Fire Complex (Launchbaugh et al. 2008). The Rowland Fire and Elk Mountain Fire were started by lightning strikes near Murphy Hot Springs and Three Creeks, Idaho, while the Scott Creek Fire was ignited by lightening west of Jackpot, Nevada. The Murphy Wildland Fire Complex encompassed 263,862 hectares during the 18 days it burned (NIFC, http://www.nifc.gov/fire_info/nfn.htm) and it was fully contained by August 2, 2007 (Launchbaugh et al. 2008). Weather before and during the fire was hot, dry, and windy, contributing to the large total area burned (Launchbaugh et al. 2008). Relative humidity was 12 to 25 % below the long term average (Launchbaugh et al. 2008). Dry weather, low relative humidity, and high temperatures in the months leading up to the fire led to dry fuel conditions in the area (Launchbaugh et al. 2008). Temperatures exceeding 35˚C, relative humidity less than 10% and a low pressure system with high winds gusting at 54 km/h in the first few days of the Murphy Wildland Fire Complex also created ideal conditions for fire spread (Launchbaugh et al. 2008). By July 24, changes in weather patterns aided firefighting efforts, as daily relative humidity increased to 29 to 32%, and temperatures dropped to below 31˚C. The Murphy Wildland Fire Complex was recognized for uncharacteristically large amounts of complete vegetation removal immediately after the fire (Launchbaugh et al. 2008). However, some of the burn did occur in a mosaic pattern across the landscape (Launchbaugh et al. 2008). Areas within the Jarbidge, Bruneau, and Elko BLM field offices were burned by the Murphy Wildland Fire Complex, along with part of the Humboldt-Toiyabe National Forest. Land managed by the State of Idaho, and private lands were also burned in these fires (Launchbaugh et al. 2008).

The study area was located in a semi-arid climate, with elevations ranging from 1,150 m in the northern region to 2,530 m in the southern region. Average summer temperatures in the northern, lower elevation zone of the Murphy Wildland Fire Complex ranged from 10.8 to 29.6^{o} C over a 45-year period. The annual precipitation average over a 45 year period was 25.4 cm, most of which occurred in the winter, spring and fall. Summer precipitation averages range from 4.1 to 0.05 cm (Western Region Climate Center 2009).

Average summer temperatures in the higher elevation southern region ranged from 4.3 to 27.6˚C over a 47-year period. The annual precipitation average from 1940 to 1987 was 32.8

cm, most of which fell in the spring. Average summer precipitation ranges from 1.0 to 16.8 cm (Western Regional Climate Center 2009).

The area burned by the Murphy Wildland Fire Complex was characterized by diverse plant species associated with sagebrush communities. The majority of the area burned consisted of Wyoming big sagebrush overstory with a native or non-native grass understory. Native grasses include bluebunch wheatgrass (*Pseudoroegneria spicata),* Sandberg bluegrass *(Poa secunda)*, Idaho fescue *(Festuca idahoensis)*, Indian ricegrass *(Achnatherum hymenoides)*, Thurbers needlegrass *(Achnatherum thurberianum)* and needle-and-thread *(Hesperostipa comata;* BLM Jarbidge Field Office 2007a). Non-native planted grasses in the area include crested wheatgrass (*Agropyron cristatum)*, intermediate wheatgrass *(Thinopyrum intermedium)*, and others (Launchbaugh et al. 2008). Several species of sagebrush were also present in the burn area in smaller quantities, including low sagebrush *(Artemisia arbuscula)*, black sagebrush *(Artemisia nova)* and mountain big sagebrush *(Artemisia tridentata* ssp. *vaseyana)*. Annual non-native grasses incuding cheatgrass and medusahead *(Taeniatherum caput-medusae)* were also present in small amounts (less than 5%), mostly in the northern portion of the burn (Launchbaugh et al. 2008). Other vegetation present in the Murphy Wildland Fire Complex include gray rabbitbrush *(Chrysothamnus* spp.*),* antelope bitterbrush *(Purshia tridentata)*, juniper (*Juniperus* spp.), aspen *(Populus tremuloides)*, curl-leaf mountain mahogany *(Cercocarpus ledifolius)*, and a variety of other species (BLM Jarbidge Field Office 2007a).

Wildlife was abundant on the area burned by the Murphy Wildland Fire Complex. Species such as sage sparrow*(Amphispiza belli)*, sage grouse *(Centrocercus urophasianus),* sage thrasher *(Oreoscoptes montanus),* and Brewer's sparrow *(Spizella breweri)* were present in sagebrush areas of the burn, along with other sagebrush-obligate species (Launchbaugh et al. 2008). Numerous large ungulates including mule deer *(Odocoileus hemionus)*, elk *(Cervus elaphus)*, bighorn sheep *(Ovis Canadensis)* and pronghorn *(Antilocapra americana)* also resided in the area of the burn. Predators such as coyotes *(Canis latrans)*, badgers *(Taxidea taxus)*, bobcats *(Felis rufus)* and mountain lions *(Felis concolor)* were present in the area, along with a variety of other wildlife species (BLM Jarbidge Field Office 2007a, Rowland et al. 2006).

According to the Jarbidge Field Office of the BLM, soils have been divided into three main physiographic units: the Snake River Sediments, the Basalt Plains/Plateaus and the Jarbidge Uplands/Foothills (BLM Jarbidge Field Office 2007a). The Snake River Sediments range from 747 to 1,158 m in elevation, and consist of well-drained sandy to silty soil. The Basalt Plains/Plateaus range from 1,128 to 1,707 m in elevation, with silt loam to clay loam soils with rock fragments mixed in. The Jarbidge Uplands/Foothills soils range from 1,701 to 2,225 m in elevation and are the oldest and most well-developed soils. These soils contain loam to clay loam textured soils with rock fragments on the surface and in the soil profile. In addition, all these soils can develop a hardpan layer or cemented layer which is impermeable to water and roots of vegetation (BLM Jarbidge Field Office 2007a). The Murphy Wildland Fire Complex burned through the area managed by the Jarbidge Field Office, and other areas with similar soils.

Available Data

Extensive data was available through the BLM and other sources on the Murphy Wildland Fire Complex. Geospatial data available for the Murphy Wildland Fire Complex included a wide variety of information including fire, vegetation, and land use data. Data available for the fire included daily fire behavior conditions and fire progression dates (BLM Twin Falls District 2007b). Data available for vegetation include classification of pre-fire vegetation community types (Figure 1; BLM Twin Falls District 2007c). Vegetation community classification prior to the Murphy Wildland Fire data was created using field data collected by the BLM from 2002 to 2006 and 2004 National Agriculture Imagery Program (NAIP) imagery, and was tested for accuracy to the 95% confidence interval (BLM Twin Falls District 2007c). Land use data available for the Murphy Wildland Fire Complex area included allotment boundaries, fenceline boundaries, roads, water developments, pipelines, streams and rivers (BLM Idaho State Office 2008). Other information was also available for the Murphy Wildland Fire Complex, including ESI data collected by the BLM. Most this data resides in the Department of Rangeland Ecology at the University of Idaho, and additional data was collected from the BLM Jarbidge Field Office in Twin Falls, Idaho.

Collection of Grazing Data

Additional data was collected on the Murphy Wildland Fire Complex in order to complete the objectives. Data collection was focused on determining if pastures were grazed by livestock before the Murphy Wildland Fire Complex in 2007. This was accomplished by meeting with BLM personnel and ranchers who have allotments in the burned area. Contacts with the BLM Jarbidge Field Office included Patty Courtney and Ken Crane. Other contacts included Kelly Crane, University of Idaho Rangeland Extension Specialist, Jessie German, BLM GIS Specialist, and Bruce Wiley, Earth Resources Observation and Science (EROS) Data Center, Brookings, South Dakota. A meeting with Burt Brackett, a local rancher with grazing allotments within the Murphy Wildland Fire Complex, was conducted in June of 2009.

Information regarding livestock time of use and was also collected from BLM records at the Jarbidge Field Office. BLM records included documentation of which allotments were scheduled to be grazed in 2007 before the fire and field notes. Field notes contained estimates of utilization and determined which of the scheduled allotments were actually being grazed in 2007 before the fire (Figure 2). Grazed allotments were identified as being grazed by cattle *(Bos taurus)*, horses *(Equus caballus)*, or sheep *(Ovis aries)*, before the fire in 2007 in the GIS database (Figure 2). Ungrazed allotments were identified as not being grazed before the fire in 2007 (Figure 2).

Relationship between Remotely-sensed Burn Severity and Change in Vegetation Cover

Pre-fire vegetation data at the species level was collected in 2006 on 494 Ecological Site Inventory (ESI) points by the BLM. About 160 ESI points were within the Murphy Wildland Fire Complex perimeter. We randomly selected a total of 37 of these ESI plots, which were sampled in June of 2009. Plots were paired based on vegetation type (shrub or perennial grass) and grazed status (grazed or ungrazed). Ten plots were located in a perennial grass vegetation type, yielding four grazed and ungrazed pairs and two additional points in grazed areas. Twenty-two plots were located in sagebrush vegetation types, yielding eleven grazed and ungrazed pairs. Five additional plots were sampled in the annual invasive species vegetation type near the northern boundary of the fire. Paired plots were selected on the same ecological sites (soil types and precipitation zone), contained similar vegetation types, and burned on the same day assuming fire weather was comparable (Table 1). Transects were established according to the BLM Sampling Vegetation Attributes Interagency Technical Reference (Interagency Technical Team 1996). At each ESI plot, a 30.5 m transect was set up at a random azimuth, and two other 30.5 m tapes were extended from the center of the plot 120˚ away from one another. If an ESI plot was not used, the selected location was marked with a Global Positioning System (GPS) waypoint. Two photographs were taken at each plot, one close-up view and one general view of the area. Cover was determined by dropping a pin every 1.2 m along the tape for a total of 75 points per plot, and percent cover was estimated as the percent of vegetation "hit" by the pin (Interagency Technical Team 1996). Bare ground (including gravel or stone), annual invasive grass cover, annual forb cover, perennial grass cover, perennial forb cover, and shrub cover were recorded as "hits" of the pin using a data sheet at three layers. The first layer was the species closest to the ground and vegetation layers were listed sequentially above the ground layer (Interagency Technical Team 1996). Canopy cover was calculated for each plot by functional groups including bare ground, annual grass, annual forbs, perennial grass, perennial forbs, and shrubs.

Next, burn severity maps were derived using ArcGIS (ESRI 2008) and LandsatTM 5 images taken about one year before the burn on August 7, 2006, soon after the burn on August 10, 2007, and one year after the burn on August 12, 2008 (Figure 3). The Environment for

Visualizing Images (ENVI) analysis program (RSI 2005) was used to apply radiometric corrections to the images, and to calculate the burn severity indices.

Developed by Rouse and colleagues (1973), the NDVI was calculated by comparing the near infrared band (NIR) to the red band (R) associated with leaf tissue (Washington-Allen et al. 2008; equation 1).

$$NDVI = \frac{NIR-R}{NIR+R} \qquad (1)$$

The NBR (equation 2) was calculated by comparing Landsat band 4 (near-infrared 0.76-0.90 µm) which is reflective of chlorophyll in live vegetation (Lopez Garcia and Caselles 1991, Key and Benson 2006) to Landsat band 7 (middle infrared 2.08-2.35 µm) which is reflective of ash, bare soil, and charred material present after a fire (Jia et al. 2006).

$$NBR = \frac{(band4)-(band7)}{(band4)+(band7)} \qquad (2)$$

The delta NBR is calculated by subtracting the post-fire NBR from the pre-fire NBR (Key and Benson 1999; equation 3).

$$dNBR = NBRprefire - NBR\ postfire \qquad (3)$$

Miller and Thode (2007) developed an algorithm that is a relative version of dNBR (equation 4). Relative dNBR is calculated by subtracting the post-fire NBR from the pre-fire relative NBR.

$$RdNBR = \frac{prefireNBR-postfireNBR}{\sqrt{(ABS(prefireNBR}/1000))} \qquad (4)$$

Two sets of burn severity maps were created using LandsatTM 5 images. First, burn severity immediately after the burn was calculated using dNBR, RdNBR, and dNDVI burn severity indices (Rouse et al. 1973, Lopez Garcia and Caselles 1991, Key and Benson 2006, Miller and Thode 2007, Washington-Allen et al. 2008). Imagery from August 7, 2006 was compared to imagery from August 10, 2007 for this analysis. Second, burn severity one year post-fire was also calcualted on the Murphy Wildland Fire Complex using the dNBR, RdNBR and dNDVI burn severity indices (Rouse et al. 1973, Lopez Garcia and Caselles 1991, Key and Benson 2006, Miller and Thode 2007, Washington-Allen et al. 2008). Imagery from August 7, 2006 was compared to imagery from August 12, 2008 for this analysis. Images were selected so that vegetation was in similar phenological stages, and cloud cover was less than 10%.

The vegetation response to fire was determined by calculating the difference in canopy cover of annual forbs, perennial grass, shrubs, annual grass, and perennial forbs from the ESI data collected in 2006 before the fire, and in the 2009 field season as part of this study. Percent canopy cover of each individual functional group was used in this analysis.

To test if there was a relationship between remotely-sensed burn severity and change in vegetation cover, a forward selection stepwise regression analysis was conducted using a cutoff value of $\alpha = 0.10$ for rejecting or accepting the predictor variables. A total of 25 plots encompassing both grass and shrub cover were used in this analysis. Dependant variables were the burn severity indices dNBR immediately after the burn, one year post-fire dNBR, RdNBR immediately after the burn, one year post-fire RdNBR, dNDVI immediately after the burn and one year post-fire dNDVI. Predictor variables included the change in vegetation from 2006 to 2009 in perennial grass cover, perennial forb cover, annual grass cover, annual forb cover and shrub cover. Predictor variables were correlated with the burn severity values to determine if burn severity on the Murphy Wildland Fire Complex could be explained by the change in vegetation cover.

Livestock Grazing Effects on Remotely-sensed Burn Severity

To test if livestock grazing reduced burn severity at the plot scale, a paired t-test was used to determine if there was a difference in the change in vegetation cover between paired grazed and ungrazed plots (Ott and Longnecker 2001). The change in annual forb cover, perennial grass cover and shrub cover were compared on grazed and ungrazed plots. Seven paired plots were used in this analysis in shrub vegetation, while four paired plots were analyzed in grassland cover types.

To test if livestock grazing reduced burn severity at the landscape scale, paired points within grazed and ungrazed pastures were identified and an overlay analysis of the datasets in ArcGIS was conducted. First, paired sites within the Murphy Wildland Fire Complex perimeter were selected. Paired plots were randomly selected using ArcGIS in grazed and ungrazed allotments. Paired plots were located on the same ecological sites (soil type and precipitation zone) within the Murphy Wildland Fire Complex, and same vegetation type including Wyoming sagebrush, mountain big sagebrush or rabbitbrush *(Chrysothamnus* spp.*)* with an understory of Idaho fescue, bluebunch wheatgrass, crested wheatgrass or Sandberg bluegrass. Grazed and ungrazed paired plots were selected in areas that burned on the same day to reduce errors associated with different burning conditions (Table 1).

Paired plots were also stratified according to vegetation type and grazed status (Table 1) resulting in four plot types: grazed shrub, grazed perennial grass, ungrazed shrub and ungrazed perennial grass (Figure 1). These were replicated by generating random locations using ArcGIS within the Murphy Wildland Fire Complex perimeter. Paired points were located at least 200 m apart to avoid autocorrelation. Autocorrelation analysis was conducted in ArcInfo (ESRI 2008), and correlation was less than 10% at 200 m. Sixty paired points (120 total) within each vegetation type/grazed status were used in the analysis, each encompassing one pixel size. Data on specific utilization levels throughout pastures before the Murphy Wildland Fire Complex did not exist. However, to more completely represent the effect of grazing, two buffers were created around each of the 60 random paired points using ArcGIS, and burn severity was averaged within each buffer zone to produce an average value for analysis. Spatial analyst tools (ESRI 2008), were used in ArcGIS to calculate the mean and standard deviation burn severity for each burn severity index. The first circular buffer zone was 64 m, encompassing 9 pixels. The second circular buffer zone was 106 m encompassing 25 pixels.

A three-way analysis of variance (ANOVA) statistical test was used to compare burn severity across spatial scales (pixel, 64 m buffer, and 106 m buffer), vegetation type (grassland or shrub habitat), grazed status (grazed or ungrazed) and their interaction (Ott and Longnecker 2001). The dNDVI, RdNBR, and dNBR immediately after the fire and one year post-fire indices were used in this analysis. If interaction terms were significant, a paired t-test was then used to further investigate the interaction between factors (Ott and Longnecker 2001).

Further, the variability of burn severity around each randomly selected point was also measured. Spatial analyst tools were used in ArcGIS to calculate the mean and standard deviation burn severity for each index used. RdNBR immediately after the burn, RdNBR one year post-fire, dNBR immediately after the burn and dNBR one year post-fire were the burn severity indices used for this analysis. The coefficient of variation was calculated by dividing the standard deviation by the mean (Ott and Longnecker 2001). The coefficient of variation was then compared between grazed and ungrazed plots using a one-way ANOVA test (Ott and Longnecker 2001) to determine if there were significant differences in variability between grazed and ungrazed plots in shrub communities or grasslands.

All statistical tests were conducted using the MINITAB15 Software (Minitab 2010). Anderson-Darling (Anderson and Darling 1952) statistical tests for normality were conducted and variances were tested for equality to determine if assumptions for paired t-tests were met. Residuals were also examined for normality using a probability plot for three-way ANOVA tests and regression analysis. Power was analyzed for the sample size of 60 random plots in shrub communities and grassland vegetation.

RESULTS

Remotely-sensed burn severity indices were explained by a change in vegetation cover on the Murphy Wildland Fire Complex (Table 2). At the plot scale, plots within grazed and ungrazed pastures were not different (Table 3, Table 4) in change in annual forb, perennial grass, or shrub cover. Conversely, at the landscape scale, there was a difference in burn severity between grassland and shrub vegetation (Table 5), with mean burn severity in grasslands being lower than mean burn severity in shrub communities considering the dNDVI, RdNBR and dNBR indices one year post-fire and dNBR immediately after the burn index (Table 5). However, there was no difference in burn severity between the three spatial scales (Table 5). Further, burn severity was lower on grazed pastures than ungrazed pastures with the RdNBR one year post-fire index and the dNBR immediately after the burn index (Table 5). There was an interaction between vegetation type and grazed status considering the RdNBR one year post-fire, RdNBR immediately after the burn, and dNBR one year post-fire indices (Table 5) indicating that the effect of grazing depended on the vegetation type. However, there was no interaction between scale and vegetation type or scale and grazing (Table 5). To further investigate the interaction between grazed status and vegetation type, a paired t-test was conducted. Results of the paired t-test show that points within grazed pastures had lower burn severity values in the shrub vegetation type (Table 6, Figure 4) using the RdNBR one year post-fire, RdNBR immediately after the burn, and dNBR one year post-fire burn severity indices. Conversely, there was not a significant difference (Table 7, Figure 5) in burn severity on points within grazed and ungrazed pastures in the grassland vegetation type.

Results: Relationship between Remotely-sensed Burn Severity and Change in Vegetation Cover

Results of the forward selection stepwise regression analysis show that there was a relationship between remotely-sensed burn severity indices and change in vegetation cover from 2006 to 2009 (Table 2). Change in shrub cover and annual forb cover were correlated with all burn severity indices (dNBR, RdNBR and dNDVI immediately after the burn and one year post-fire). Change in perennial grass cover was a significant predictor of the RdNBR immediately after the burn index and the dNBR immediately after the burn index (Table 2). Change in vegetation cover best predicted the RdNBR and dNBR burn severity indices immediately after the burn (Adjusted R-squared = 42% and 57% respectively).

Results: Livestock Grazing Effects on Change in Vegetation Cover at the Plot Scale

At the plot scale, paired t-test results show that there was no difference in the change in annual forb cover, shrub cover, or perennial grass cover from 2006 to 2009 between grazed and ungrazed plots in shrub habitat (Table 3). Similarly, there was no difference in the change in annual forb cover, shrub cover, or perennial grass cover between grazed and ungrazed plots in grasslands (Table 4). Grazing thus was not influencing the vegetation response of annual forbs, shrubs, or perennial grasses after the fire at the plot scale.

Results: Livestock Grazing Effects on Remotely-sensed Burn Severity at the Landscape Scale

At the landscape scale, mean burn severity was lower in grasslands than shrub communities when examined with the dNDVI index immediately after the burn and one year post-fire, RdNBR index one year post-fire, and the dNBR index immediately after the burn and one year post-fire (Table 5). Additionally, mean burn severity was lower in grazed pastures than ungrazed pastures when examined with the RdNBR one year post-fire and the dNBR immediately after the burn indices (Table 5). There was also an interaction between grazed status and vegetation type when examined with the RdNBR index immediately after the burn and one year post-fire, and the dNBR index one year post-fire (Table 5), indicating that grazing may be effecting burn severity differently depending on the vegetation type. However, there was no difference in burn severity between the three spatial scales (30 m pixel, 64 m buffer, and 106 m buffer), and there were no interactions between spatial scale, vegetation type, and grazed status (Table 5). Residual probability plots showed that most distributions were normal. Extreme outliers were removed from non-normal plots so that residuals met the assumptions of normality for the three-way ANOVA.

Results: Livestock Grazing Effects on Burn Severity in the Interaction between Grazed Status and Vegetation Type

Livestock Grazing Effects in Shrub Vegetation Type

Due to the fact that there were no differences in burn severity between spatial scales, the interaction between grazed status and vegetation type was investigated at the pixel (30 m) scale only. In shrub vegetation type on the Murphy Wildland Fire Complex, burn severity was lower in grazed pastures than ungrazed pastures when examined with the RdNBR one year post-fire, RdNBR immediately after the burn, and dNBR one year post-fire indices (Table 6, Figure 4).

Livestock Grazing Effects in Grassland Vegetation Type

In grassland vegetation on the Murphy Wildland Fire Complex, there was no difference in burn severity between grazed and ungrazed pastures when examined with all the burn severity indices (dNDVI, RdNBR and dNBR immediately after the burn and one year post-fire; Table 7, Figure 5). Samples which did not meet the assumptions of normality when tested with the Anderson-Darling normality test (Anderson and Darling 1952) for all paired t-tests at the landscape scale were analyzed with the two-sample Wilcoxon signed-rank nonparametric test (Wilcoxon 1945).

Results: Variability in Burn Severity

Results show that there were no differences in the coefficient of variance between grazed and ungrazed areas on either shrub or grassland vegetation types (Table 8). Grazing did not reduce nor increase the variability in burn severity across randomly selected plots within shrub communities or grasslands in the Murphy Wildland Fire Complex (Table 8).

DISCUSSION

There have been few studies using remote sensing to assess burn severity in rangeland vegetation. Burn severity within grazed and ungrazed pastures in sagebrush steppe ecosystems has not been analyzed using remote sensing. This study used three different remotely-sensed burn severity indices to detect differences in burn severity on grassland and sagebrush vegetation in southern Idaho. In addition, this study used remote sensing indices to detect differences in burn severity on grazed and ungrazed pastures within grassland and shrub vegetation types. Burn severity in grasslands was lower than burn severity in shrub communities, and burn severity on grazed pastures was lower than burn severity in ungrazed pastures within shrub communities.

Burn Severity Indices

While all burn severity indices were correlated with a change in vegetation cover on the Murphy Wildland Fire Complex, the RdNBR index may have more practical applications in sagebrush steppe ecosystems. The RdNBR index was most consistently able to differentiate burn severity between grassland and shrub communities, as well as grazed and ungrazed pastures. While dNBR did also detect changes, it was not as useful at differentiating between grazed and ungrazed pastures in shrub communities. This may due to the fact that the dNBR index is correlated with the amount of pre-fire biomass (Miller and Thode 2007). Miller and Thode (2007) found that use of dNBR alone in heterogeneous landscapes in the Sierra Nevada Mountains may not accurately represent high burn severity classes, and that the RdNBR was particularly useful in areas with low pre-fire biomass. Norton (2006) similarly found that RdNBR was the most accurate burn severity index (73% overall accuracy) for characterizing burn severities on rangelands in Idaho. In her study, Norton (2006) defined burn severity as the completeness of above-ground vegetation removal during the burn. The dNBR index may have more practical applications in forested ecosystems, or communities with homogenous vegetation (Miller and Thode 2007).

The dNDVI index was not useful in detecting changes in burn severity on grazed and ungrazed pastures on the Murphy Wildland Fire Complex. The NDVI index is particularly useful in detecting the photosynthetic capabilities of healthy green vegetation (Sellars 1985). The satellite imagery for this study was taken in August, when most understory grasses and forbs in the area of the Murphy Fire are senescent. The NDVI index may be more practical to use when vegetation is in earlier phenological stages on rangelands. However, research with varied levels of cotton canopy cover indicates that the greenness index (NDVI) is not practical to use when vegetation biomass is low (Huete et al. 1987). Similarly, Norton (2006) found that modified NDVI burn severity indices were very poor at detecting changes in vegetation cover after prescribed burns on sagebrush steppe rangelands in southern Idaho.

Livestock Grazing Effects on Change in Vegetation Cover at the Plot Scale

Grazing was not a factor in the vegetation response post-fire under wildfire conditions. Change in annual forb cover, shrub cover, and perennial grass cover from before the wildfire in 2006 and after the burn in 2009 were not different between grazed and ungrazed plots. Bates et al. (2009) showed a similar response after prescribed fire treatments in Wyoming big sagebrush steppe in eastern Oregon. Herbaceous vegetation cover, bare ground and surface litter did not vary between grazed and ungrazed plots treated with prescribed fire (Bates et al. 2009).

Grassland and Shrub Differences in Burn Severity on the Landscape Scale

Lower remotely-sensed burn severities on grasslands as compared to shrub communities, especially one year post-fire, reflects the rapid recovery of many grass communities within one year of a fire. The rapid response during the growing season after the fire may be due to abundant available moisture, season of burn, seeding rehabilitation efforts, meristematic tissue survival and a potential increase in available nitrogen (Blaisdell 1953, Conrad and Poulton 1966, Bradley et al. 1992, Robberecht and Deffose 1995, Rhodes 2006). Perennial bunchgrass cover on sagebrush steppe rangelands often increases post-fire (Robberecht and Defosse 1995, Bates et al. 2009, Davies et al. 2009). The 2008 and 2009 spring growing seasons in the region surrounding Murphy Wildland Fire Complex experienced above average moisture which likely contributed to healthy bunchgrass response in this area (Western Regional Climate Center 2009). Redmann (1978), Robberecht and Defosse (1995) and Bates et al. (2009) all concluded that available water during the regrowth period of bunchgrasses was a key contributing factor to their healthy response after a fire. Further, the Murphy Wildland Fire Complex occurred from July 16 to August 2, when bunchgrasses are typically in summer dormancy (Daer and Willard 1981). Season of burn has the potential to negatively affect perennial bunchgrass response to fire (Wright and Bailey 1982). Burns that occur while cool season bunchgrasses are dormant are less damaging than burns that occur during the active growing season (Wright and Klemmedson 1965, Wright and Bailey 1982). Field observations on the Murphy Wildland Fire Complex indicate that in areas where perennial grasses did not respond vigorously after the fire, annual forbs were abundant. A strong annual forb response post-fire may also be contributing to lower burn severity in grasslands than shrub communities, as remote sensing in our study was unable to detect differences between annual forbs and perennial grasses.

The inability of Wyoming big sagebrush and mountain big sagebrush to survive fire, as well as the slow recovery time of shrubs relative to perennial grasses, were likely strong contributing factors to higher remotely-sensed burn severity in shrub communities than grasslands on the Murphy Wildland Fire Complex. Wyoming big sagebrush, mountain big sagebrush, and rabbitbrush typically lose all above ground biomass immediately following a wildfire (Martin and Dell 1978, White and Currie 1983, Sapsis and Kauffman 1991, Baker 2006). While remnant sagebrush was present on the Murphy Wildland Fire Compex in some areas characterized by lower severity, much of the above ground biomass of sagebrush was lost in the fire. Wyoming big sagebrush often takes over 30 years to return to pre-fire canopy cover (Lesica et al. 2007), while mountain big sagebrush typically needs 15-20 years to recover following a fire (Bunting et al. 1987), and may even require up to 30 years (Wambolt et al. 2001). Although rabbitbrush is often classified as a vigorous sprouter after fire (Young and Evans 1974, Ralphs and Busby 1979, Akinsoji 1988), it may not respond as quickly as grasses and forbs (Akinsoji 1988), and may also take several growing seasons after the fire to become successfully established (Robertson and Cords 1957, Johnson and Strang 1983). Thus, satellite imagery taken one year after the fire would most likely reflect a greatly reduced shrub canopy cover on the Murphy Wildland Fire Complex, while grasses likely were already beginning to become established from undamaged meristematic tissue and seeding rehabilitation efforts.

Livestock Grazing Effects on Remotely-sensed Burn Severity at the Landscape Scale

Livestock grazing reduced burn severity in pastures dominated by shrubs, but did not reduce burn severity in grasslands on the Murphy Wildland Fire Complex. Livestock grazing in southeastern Idaho can reduce fuel loading on rangelands, especially under high stocking rates (Weber et al. 2004). Lack of fine fuel in the understory of Wyoming big sagebrush can make it difficult for fire to carry across the landscape (Bunting et al. 1987). Similarly, Beardall and Sylvester (1976) found sagebrush steppe in Nevada was hard to burn with fuel loading under 680 kg/ha. Livestock grazing in shrub communities on the Murphy Wildland Fire Complex likely reduced fuel loading in the interspaces between shrubs. Less fuel loading between shrubs has the ability to influence fire behavior, resulting in lower burn severity on grazed pastures. However, as sagebrush canopy cover increases, less fuel loading in the understory is needed to carry a fire from crown to crown in sagebrush dominated communities (Bunting et al. 1987). This would suggest that grazing in areas with lower shrub canopy cover would be more effective at reducing burn severity than grazing in areas with high shrub canopy cover.

Conversely, grazing did not reduce burn severity in grassland areas within the Murphy Wildland Fire Complex immediately after the fire or one year post-fire. Davies et al. (2009) found that perennial grass cover increased on grazed areas after a prescribed burn, and annual forb cover increased on ungrazed areas after a prescribed fire on the Northern Great Basin Experimental Range in Oregon. Lack of grazing can allow fuels to accumulate at the base of perennial grasses, enabling fire to slowly smolder at the plant crown (Conrad and Poulton 1966, Davies et al. 2009). This may result in more tissue damage to perennial grasses in ungrazed areas, which can permit annual forbs to increase post-fire (Davies et al. 2009). However, remotely-sensed imagery in this research was not able to detect differences between annual forbs and perennial grasses. The lack of difference in burn severity between grazed and ungrazed pastures one year post-fire thus is most likely due to strong vegetative response of annual forbs or perennial grasses on both grazed and ungrazed pastures, and the inability of remote sensing to detect differences between these two functional groups. Immediately post-fire, grazing was not affecting burn severity in grasslands mostly likely due to the complete aboveground biomass loss on both grazed and ungrazed pastures.

Variability in Burn Severity

The burn severity coefficient of variance analysis showed no difference between grazed and ungrazed pastures considering all burn severity indices in both shrub communities and grasslands on the Murphy Wildland Fire Complex. While we do not know how grazing influences the variability in burn severity on rangelands, some studies have shown how grazing and fire can influence heterogeneity in vegetation across a landscape. Harrison et al. (2003) showed that on California grasslands, grazing on non-serpentine soils typically created more homogeneous vegetation, with less variability in plant species. However, fire in California grasslands was likely to increase the heterogeneity of plant communities (Harrison et al. 2003). Grazing is capable of increasing the heterogeneity of plant communities (Fuhlendorf and Engle 2001, Kerby et al. 2007). Using FARSITE models, Kerby et al. (2007) was able to show that grazing likely contributed to fires burning in a mosaic patchwork with higher and lower impacts on vegetation across a landscape (Kerby et al. 2007). While our research focuses more on the variability in burn severity on rangelands due to livestock grazing, it appears to contradict these studies focusing on differences in landscape heterogeneity due to fire and livestock grazing.

CONCLUSIONS

Under wildfire conditions, there is a lack of research examining the use of remotely-sensed burn severity indices to detect differences on rangeland ecosystems. Our research showed that remotely-sensed burn severity indices (especially RdNBR indices) are useful at detecting differences between grazed and ungrazed pastures on rangelands. On the plot scale, grazed and ungrazed plots responded similarly to fire, with no significant changes in vegetation cover two years post-fire. On the landscape scale, grasslands within the Murphy Wildland Fire Complex perimeter were distinguished by overall lower remotely-sensed burn severities than shrub vegetation types especially one year post-fire. This is likely attributable to the rapid response of grassland vegetation post-fire when compared to shrub vegetation. Grazing during the spring and summer before the Murphy Wildland Fire Complex reduced remotely-sensed burn severity in shrub communities, but not in grasslands. Grazing may have reduced fuel loading in the interspaces between shrubs, altering fire behavior and yielding lower burn severity. In grasslands, however, strong vegetative recovery of annual forbs and perennial grasses on both grazed and ungrazed pastures were likely contributing factors in the lack of differences in burn severity due to grazing. Based on these conclusions, remotely-sensed burn severity indices can be used to measure differences between grazed and ungrazed pastures on sagebrush rangelands, but only considering certain burn severity indices and vegetation types.

MANAGEMENT IMPLICATIONS

As the threat of large wildfires fanned by hot, dry, windy conditions and invasive species increases in the western United States (Knick 1999, Anderson and Inouye 2001, Westerling et al. 2006), land managers increasingly need a suite of innovative tools to manage wildland fire and fuels. Remote sensing can be a useful tool for detecting burn severity on rangelands, and livestock grazing can be an effective method for reducing burn severity under certain conditions.

Strong correlations between remotely-sensed burn severity and change in vegetation cover on the Murphy Wildland Fire Complex show that remote sensing (especially RdNBR indices) are useful in determining burn severity on sagebrush rangelands. Remote sensing is practical and follows repeatable procedures. Landsat data is available free for public use, and allows for multiple year analysis. Further, remote sensing reduces field data collection costs. While remote sensing, image processing and radiometric corrections can be technical, there are resources available outlining repeatable steps to conduct these procedures (Jensen 2007). Remotely-sensed burn severity indices are especially useful at depicting changes in burn severity one or more growing seasons post-fire (Roy et al. 2006).

Another available resource to help reduce burn severity on rangelands is livestock grazing. Livestock grazing that occurred immediately before the Murphy Wildland Fire Complex reduced the RdNBR burn severity in shrub vegetation types. Reduction of burn severity in shrub communities of southern Idaho has the potential to benefit natural ecosystem processes, including recruitment of species into burned areas and providing wildlife habitat. Under wildfire conditions, sagebrush obligate species such as the sage grouse (Rowland et al. 2006) will potentially benefit by reduced burn severity due to livestock grazing. When burn severity is reduced due to livestock grazing, there is greater potential for unburned islands of sagebrush to remain after a fire which can provide essential habitat value for sagebrush obligate species (Rowland et al. 2006). Unburned islands also can enhance natural recruitment of species into burned areas, by providing locally adapted seeds for reestablishment of species post-fire (Blaisdell 1953).

While burn severity was overall lower in grassland than shrub vegetation types one year post-fire, livestock grazing did not reduce most remotely-sensed burn severity indices on grasslands within the Murphy Wildland Fire Complex. Remotely-sensed burn severity indices may have a more practical use in shrub communities than grasslands, due to the quick recovery time of perennial grasses after a fire and recruitment of annual forbs. Remote sensing is likely detecting a strong vegetative response (annual forbs and perennial grasses) post-fire, on both grazed and ungrazed grasslands, resulting in no differences in burn severity due to grazing.

While livestock grazing did reduce burn severity in sagebrush communities on the Murphy Wildland Fire Complex, results must be taken into context of this specific study. Retrogressive research is limited because we are restricted to using data that was collected before the fire. This study was not able to directly ascertain levels of utilization in field plots, random points, or buffer zones. We are limited to the knowledge that pastures were grazed before the fire in 2007. To encompass the effects of grazing within grazed pastures, a large number of random points were created in addition to averaging burn severity across 64 m and 106 m buffers. Further limitations include that plot scale change in vegetation cover data was calculated on grazed and ungrazed plots using only seven paired plots in shrub vegetation types, and four paired plots in grasslands. A larger sample size would make this analysis stronger, but this study was constrained by availability of data collected before the fire by the BLM for comparison and accessibility of remotely located plots. Future research could be directed at collecting more ground reference data for this analysis.

In addition, further research is needed using remotely-sensed burn severity indices combined with field reconnaissance on rangelands to firmly establish relationships between livestock grazing and burn severity. Future studies could examine the interactions between fire and grazing under variable weather patterns such as drought or above average moisture. Additionally, the effects of grazing on burn severity could be studied under more moderate fire conditions than observed in the Murphy Wildland Fire Complex.

On the Murphy Wildland Fire Complex, livestock grazing was able to reduce burn severity on shrub vegetation types, especially when examined with RdNBR burn severity indices, yet caution should always be exercised when using livestock grazing as a management tool to reduce fire hazard. Projects should always be taken into context of current management objectives, desired stocking rates, utilization goals, and site specific characteristics.

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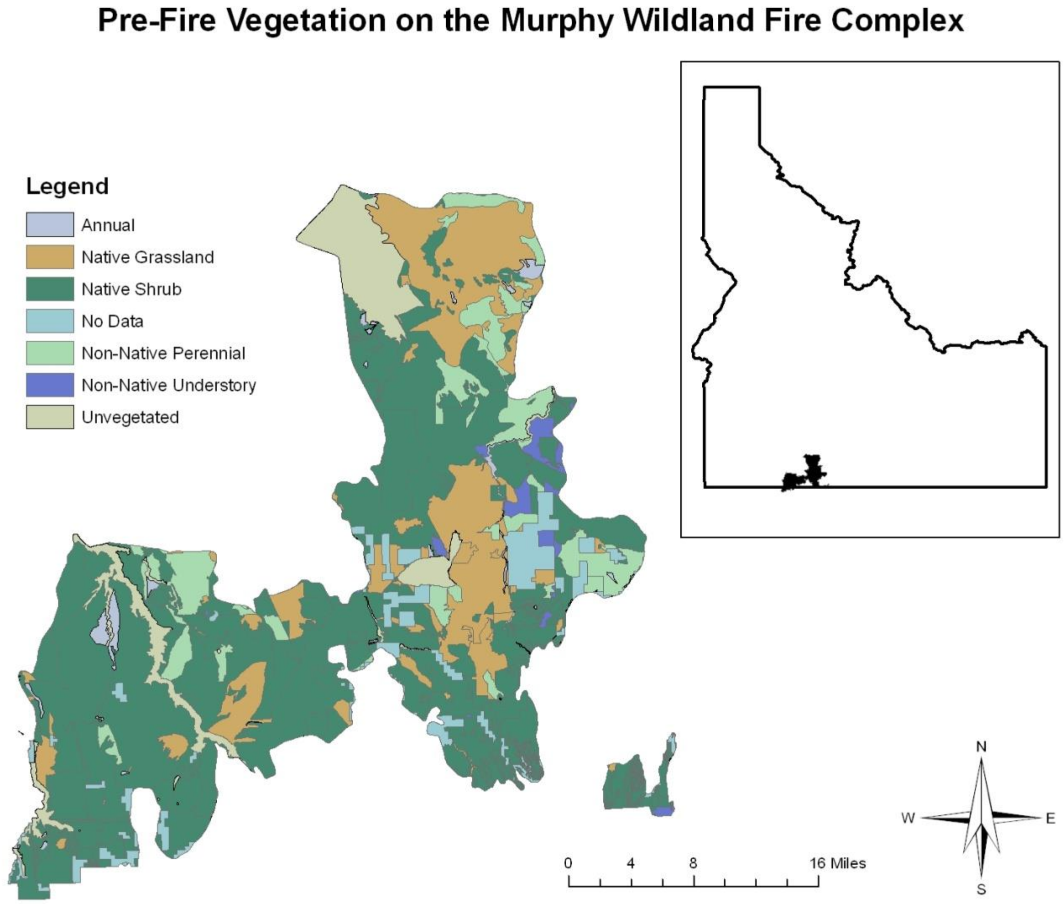

Figure 1. Map of pre-fire vegetation by functional groups on the Murphy Wildland Fire Complex. The location of the Murphy Wildland Fire Complex is depicted in the state of Idaho. Vegetation community classification prior to the Murphy Wildland Fire data was created using field data collected by the BLM from 2002 to 2006 and 2004 National Agriculture Imagery Program (NAIP) imagery, and was tested for accuracy to the 95 percent confidence interval.

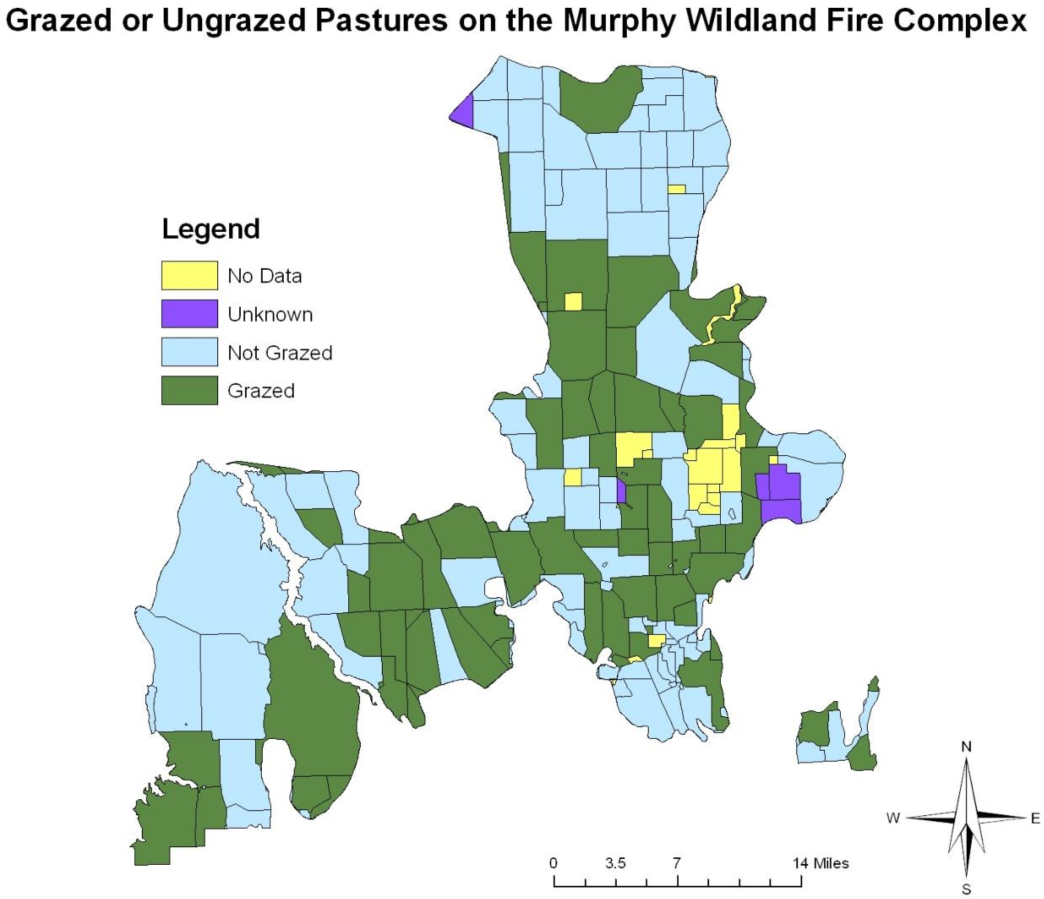

Figure 2. Grazed status of pastures within the Murphy Wildland Fire Complex. Pastures were characterized as grazed if they were grazed in 2007 before the burn. Pasture grazing data was obtained by comparing BLM scheduled livestock use in 2007 to field notes of actual livestock use in 2007 to determine if a pasture had been grazed before the fire.

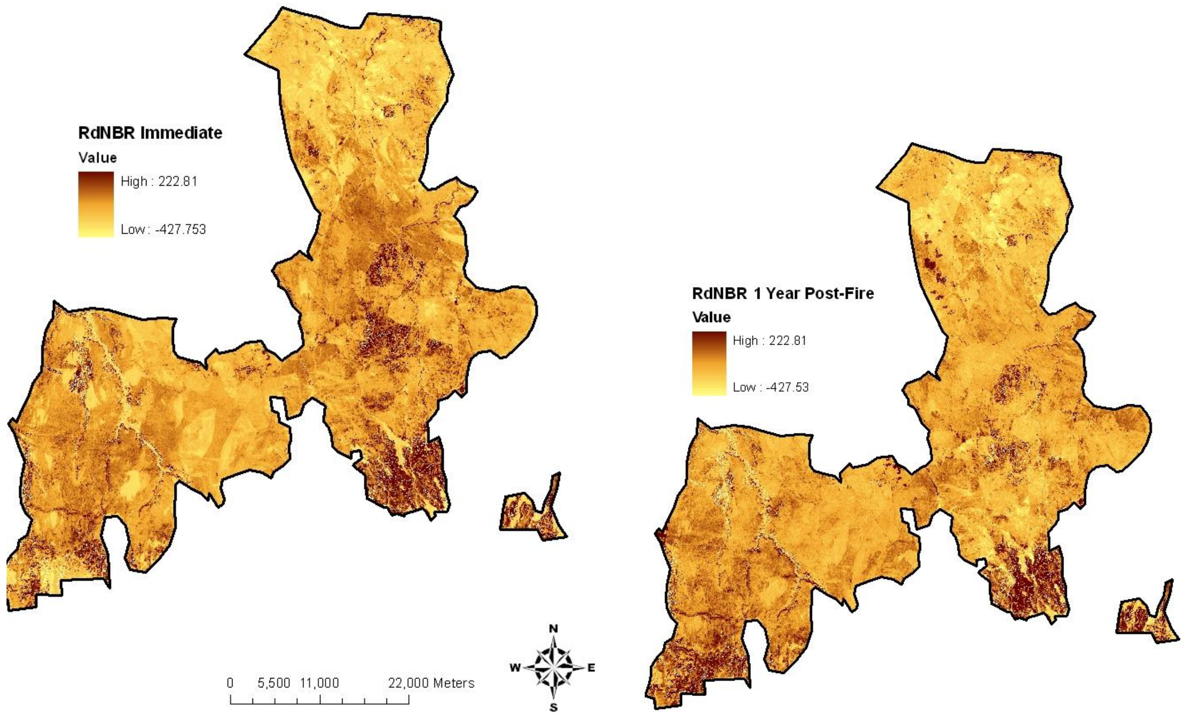

Figure 3. Burn severity maps of the RdNBR burn severity index on the Murphy Wildland Fire Complex calculated immediately after the burn and one year post-fire.

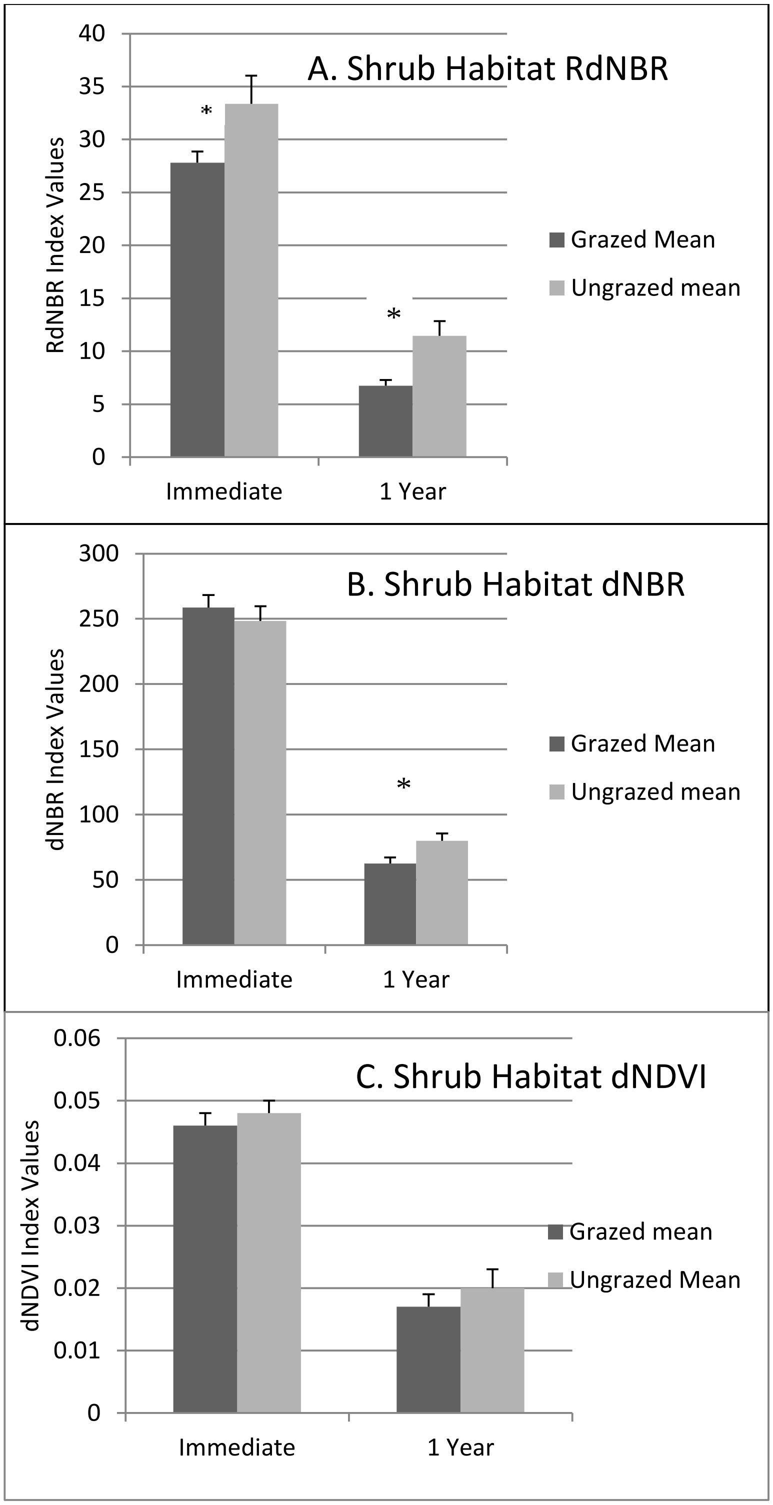

Figure 4. Mean burn severity values in paired grazed and ungrazed plots in shrub habitat on the Murphy Wildland Fire Complex (A. RdNBR burn severity, B. dNBR burn severity and C. dNDVI burn severity) with Standard Error bars. Sixty grazed plots and sixty ungrazed plots were used in this analysis. Burn severity was averaged across one pixel (30 m). Burn severity was calculated immediately after the fire and one year post-fire for all indices. Significant differences are indicated by an *.

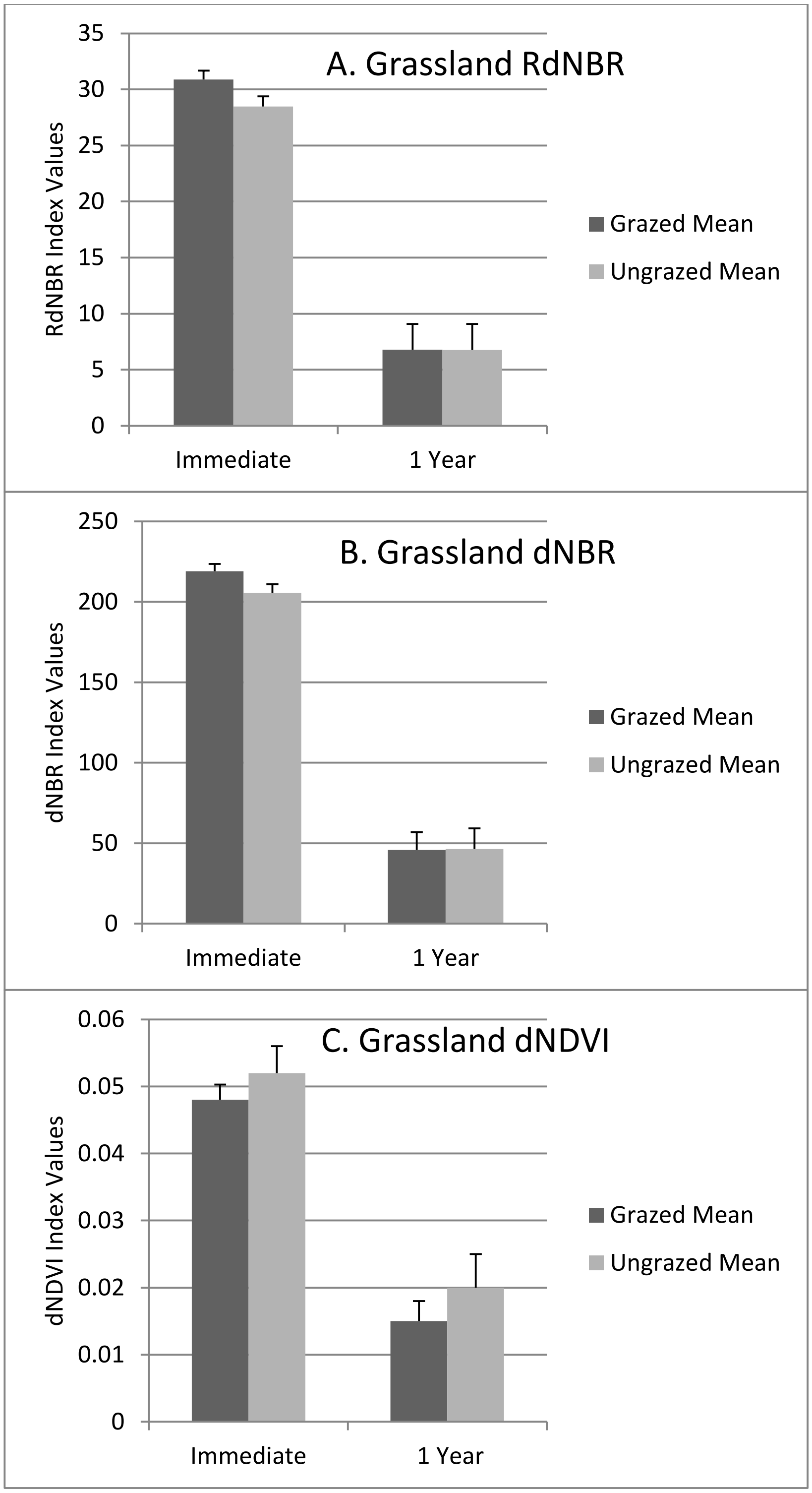

Figure 5. Mean burn severity values in paired grazed and ungrazed plots in grasslands on the Murphy Wildland Fire Complex (A. RdNBR burn severity, B. dNBR burn severity and C. dNDVI burn severity) with Standard Error bars. Sixty grazed plots and sixty ungrazed plots were used in this analysis. Burn severity was averaged across a 64 m buffer. Burn severity was calculated immediately after the fire and one year post-fire for all indices. Significant differences are indicated by an *.

Table 1. Characteristics for selecting paired sites on the Murphy Wildland Fire Complex, including grazed status and vegetation type. Each of the paired plots were burned the same day, and were on the same ecological sites (soil type and precipitation zone).

Number of Plots	Vegetation Type	Grazing Status
60	Shrub	Grazed
60	Shrub	Ungrazed
60	Perennial Grass	Grazed
60	Perennial Grass	Ungrazed

Table 2. Forward selection stepwise regression analysis results for change in vegetation cover and burn severity indices on the Murphy Wildland Fire Complex in southern Idaho. An $\alpha = 0.10$ cutoff was used to include or not include the change in different vegetation types as predictors of burn severity. Predictors include shrub, annual forb, perennial grass, annual grass and perennial forb functional groups; annual grass and perennial forb predictors were not significant predictors. Change in vegetation cover was calculated by subtracting vegetation canopy cover in 2009 from canopy cover in 2006. Dependant variables were burn severity indices. Twenty-five plots were used in this analysis.

	dNDVI		RdNBR		dNBR	
Predictor Variables in Model (p-value)	Immediate	1 Year	Immediate	1 Year	Immediate	1 Year
Shrub Change	0.074	0.002	<0.001	0.002	<0.001	0.001
Annual Forb Change	0.03	0.003	0.001	0.003	<0.001	0.002
Perennial Grass Change	>0.10	>0.10	0.076	>0.10	0.009	>0.10
Adjusted R-Squared %	13	33	42	33	57	38
Number of Steps	2	2	3	2	3	2

Table 3. Paired t-test results of change in vegetation cover on paired grazed and ungrazed plots in shrub vegetation types on the Murphy Wildland Fire Complex. Change in vegetation cover was calculated by subtracting canopy cover in 2009 from canopy cover on the plot in 2006. Negative numbers show an increase in vegetation cover after the burn. Seven paired plots (total of 14 plots) were used in this analysis.

	Total Vegetation Cover Change	Perennial Grass Cover Change	Shrub Cover Change
Grazed Mean	2.96	-1.86	16.44
Ungrazed Mean	0.36	-4.09	18.20
Grazed SE Mean	5.23	4.28	4.45
Ungrazed SE Mean	7.14	6.08	2.87
P-value	0.746	0.757	0.623

Table 4. Paired t-test results of change in vegetation cover on paired grazed and ungrazed plots in grassland cover types on the Murphy Wildland Fire Complex. Change in vegetation cover was calculated by subtracting canopy cover in 2009 from canopy cover on the plot in 2006. Negative numbers show an increase in vegetation cover after the burn. Four paired plots (total of 8 plots) were used in this analysis.

	Total Vegetation Cover Change	Perennial Grass Cover Change	Shrub Cover Change
Grazed Mean	4.87	-1.40	0.85
Ungrazed mean	6.68	-3.00	0.83
Grazed SE Mean	4.95	4.18	0.42
Ungrazed SE Mean	7.68	7.11	0.80
P-value	0.767	0.852	0.977

Table 5. Results of a three-way ANOVA comparing burn severity between scale (pixel, 64 m buffer, 106 m buffer), vegetation type (grassland, shrub habitat), grazed status (grazed, ungrazed) and their interaction. Degrees of freedom, p-value, mean square (MS) and mean square error (MSE) are reported. Sixty random grazed and sixty random ungrazed points were used in each vegetation type, resulting in 120 points in each vegetation type and 240 total points. Significant differences are indicated by an *, α=0.10.

		dNDVI				RdNBR				dNBR			
		Immediate		1 Year		Immediate		1 Year		Immediate		1 Year	
Factors	df	p	MS	p	MS	p	MS	p	MS	p	MS	p	MS
Scale	2	0.734	0.000	0.974	0.000	0.182	353.2	0.725	9.66	0.588	3471	0.989	13
Vegetation Type	1	0.027*	0.002	0.011*	0.002	0.833	9.2	<0.001*	690.94	<0.001*	334526	<0.001*	124213
Grazed Status	1	0.776	0.000	0.583	0.000	0.635	46.6	0.040*	127.79	0.003*	57710	0.298	1323
Scale:VegType	2	0.992	0.000	0.602	0.000	0.881	26.2	0.922	2.44	0.977	150	0.770	319
Scale:Grazed Status	2	0.877	0.000	0.674	0.000	0.954	9.7	0.892	3.44	0.973	182	0.883	152
VegType:Grazed	1	0.483	0.000	0.914	0.000	<0.001*	2939.9	<0.001*	795.57	0.851	231	<0.001*	13158
MSE	678		0.000		0.000		206.5		30.05		6541		1221

Table 6. Results of landscape scale paired t-test of burn severity between paired grazed and ungrazed points in shrub vegetation on the Murphy Wildland Fire Complex. Sixty random points were used in this analysis, and burn severity was averaged within 1 pixel (30 m). Significant differences between grazed and ungrazed points in shrub vegetation are indicated by an * α=0.10.

	dNDVI		RdNBR		dNBR	
	Immediate	1 Year	Immediate	1 Year	Immediate	1 Year
p-value	0.755	0.392	0.046*	0.003*	0.491	0.035*
Grazed mean	0.046	0.017	27.80	6.760	258.6	62.52
Ungrazed Mean	0.048	0.020	33.37	11.46	248.5	80.00
Grazed SE Mean	0.002	0.002	1.06	0.52	9.60	4.64
Ungrazed SE Mean	0.003	0.002	2.67	1.40	11.1	5.73
Power (1-β)	0.06	0.13	0.77	0.89	0.14	0.76

Table 7. Results of landscape scale paired t-test of burn severity between paired grazed and ungrazed points in grassland vegetation on the Murphy Wildland Fire Complex. Sixty random points were used in this analysis and burn severity was averaged within 1 pixel (30 m). Significant differences between grazed and ungrazed points in grassland vegetation are indicated by an * α=0.10.

	dNDVI		RdNBR		dNBR	
	Immediate	1 Year	Immediate	1 Year	Immediate	1 Year
p-value	0.478	0.293	0.435	0.981	0.358	0.929
Grazed mean	0.048	0.015	30.88	6.775	219.1	45.92
Ungrazed Mean	0.052	0.020	28.48	6.747	205.6	46.51
Grazed SE Mean	0.003	0.002	2.31	0.792	11.0	4.49
Ungrazed SE Mean	0.005	0.004	2.33	0.887	12.7	5.30
Power (1-β)	0.11	0.18	0.25	0.13	0.13	0.06

Table 8. Results from one way ANOVA using coefficient of variance statistic comparing grazed and ungrazed areas to burn severity indices in grassland and shrub habitat. The coefficient of variance was averaged within randomly selected plots in the Murphy Wildland Fire Complex. Degrees of freedom, p-value, mean square (MS) and mean square error (MSE) are reported. A sample size of N = 60 for each vegetation type was used, α=0.10.

		RdNBR				dNBR			
		Immediate		1 Year		Immediate		1 Year	
Factors	df	p	MS	p	MS	p	MS	p	MS
Grass	1	0.124	0.168	0.503	39.80	0.395	0.041	0.319	17.42
Shrub	1	0.175	0.467	0.655	0.022	0.285	0.299	0.463	0.112
MSE Grass	118		0.070		88.30		0.056		9.80
MSE Shrub	118		0.251		0.109		0.299		0.112

APPENDIX A: Plot Scale Analysis and Field Data

Table 9. Grazed status, vegetation type and burn severity classification by plot on the Murphy Wildland Fire Complex.

Plot Number	Grazed Status	Vegetation Type	Mean dNBR Immediate	Mean dNBR 1 Year	Mean RdNBR Immediate	Mean RdNBR 1 Year	Mean dNDVI Immediate	Mean dNDVI 1Year
1	Grazed	Shrub	95.931	32.494	10.977	3.796	0.014	-0.011
2	Ungrazed	Shrub	214.752	115.657	28.167	15.298	0.043	0.011
3	Grazed	Shrub	159.441	39.400	18.936	4.711	0.040	0.005
4	Ungrazed	Shrub	249.590	88.914	30.353	10.830	0.042	0.018
5	Ungrazed	Shrub	314.414	122.400	29.225	11.420	0.051	-0.003
6	Grazed	Shrub	66.981	28.429	7.623	3.100	0.014	-0.007
7	Grazed	Grass	23.512	80.952	3.024	10.316	0.035	0.046
8	Ungrazed	Grass	308.547	57.048	33.508	6.186	0.065	0.043
735	Ungrazed	Shrub	46.373	-57.891	4.296	-5.285	0.012	0.013
805	Ungrazed	Shrub	128.400	-39.420	12.685	-3.896	0.011	0.005
821	Ungrazed	Shrub	63.857	32.024	7.325	3.892	0.034	0.048
845	Grazed	Shrub	254.400	95.048	32.092	11.989	0.075	0.011
849	Grazed	Shrub	60.980	16.050	5.939	1.612	0.019	0.005
903	Ungrazed	Shrub	358.769	55.126	39.682	6.141	0.044	0.048
911	Grazed	Shrub	328.985	143.274	42.490	18.504	-0.009	-0.065
937	Grazed	Grass	224.710	18.138	23.959	1.891	0.073	0.028
938	Ungrazed	Grass	253.932	-10.850	32.264	-1.380	0.102	0.047
945	Grazed	Shrub	334.831	71.802	37.044	7.944	0.040	0.063
947	Grazed	Grass	162.482	48.817	23.592	7.222	0.036	0.021
949	Ungrazed	Shrub	394.749	49.179	39.955	5.060	0.050	0.067
950	Ungrazed	Grass	233.739	42.795	28.299	5.319	0.065	0.022
953	Ungrazed	Shrub	246.816	64.589	25.863	6.786	0.036	0.005
954	Ungrazed	Shrub	180.403	72.032	18.617	7.436	0.030	0.012

957	Ungrazed	Shrub	201.117	65.031	23.155	7.567	0.029	0.008
958	Grazed	Shrub	171.649	55.250	17.519	5.667	0.034	0.012
963	Ungrazed	Shrub	289.162	87.134	31.291	9.562	0.036	0.020
975	Ungrazed	Grass	269.762	43.571	29.029	4.804	0.040	-0.002
984	Grazed	Shrub	397.764	107.227	48.744	13.158	0.056	0.045
994	Grazed	Shrub	438.523	102.511	44.756	10.454	0.050	0.010
1004	Ungrazed	Grass	280.522	45.193	36.200	5.859	0.044	0.004
1006	Grazed	Grass	205.818	16.602	28.414	2.248	0.037	0.006
1074	Ungrazed	Shrub	385.216	131.240	65.940	22.447	0.120	0.051
1153	Grazed	Shrub	20.244	-95.687	1.449	-6.822	0.030	0.018
1158	Grazed	Grass	231.207	37.172	34.878	5.618	0.043	-0.002
1169	Grazed	Grass	264.972	58.060	51.081	11.370	0.063	0.029
1222	Grazed	Grass	212.085	45.117	23.490	4.973	0.034	0.004

Table 10. Percent canopy cover of vegetation by plot collected on the Murphy Wildland Fire Complex in June 2009 according to the BLM Sampling Vegetation Attributes Interagency Technical Reference (Interagency Technical Team 1996).

Plot Number	Canopy Cover Bare Ground 09	Canopy Cover Perennial Grass 09	Canopy Cover Perennial Forb 09	Canopy Cover Annual Grass 09	Canopy Cover Annual Forb 09	Canopy Cover Shrub 09	Canopy Cover Total 09
1	24	40	8	0	0	28	0
2	56	36	0	0	0	8	76
3	31	32	25	0	3	9	44
4	41	43	8	1	7	0	69
5	53	32	4	3	8	0	59
6	47	27	1	0	0	25	47
7	41	47	0	11	1	0	53
8	55	12	1	17	15	0	59
735	36	31	4	7	20	3	45

805	48	23	1	0	28	0	65
821	47	39	0	3	12	0	52
845	42	43	4	0	9	3	54
849	36	36	5	0	0	22	59
903	42	18	8	0	32	0	63
911	29	61	11	0	0	0	58
937	29	67	3	1	0	0	72
938	21	56	22	0	0	1	71
945	73	15	0	5	7	0	79
947	43	52	5	0	0	0	27
949	97	0	0	0	3	0	57
950	47	35	0	6	12	0	3
953	40	42	6	5	6	0	53
954	53	46	0	0	1	0	59
957	47	38	15	0	1	0	47
958	35	49	12	0	1	4	54
963	44	42	8	1	4	0	66
975	34	60	4	1	0	1	55
984	49	37	7	0	7	0	66
994	49	25	13	0	12	1	51
1004	21	78	1	0	0	0	51
1006	36	45	20	0	0	0	79
1074	39	49	11	0	0	1	65
1153	48	33	0	11	8	0	61
1158	41	50	8	1	0	0	52
1169	32	32	4	28	3	0	59
1222	15	77	8	0	0	0	67

Table 11. Percent canopy cover data collected at ESI points by the BLM in 2006, during the season preceeding the Murphy Wildland Fire Complex.

Plot Number	Canopy Cover Bare Ground 06	Canopy Cover Perennial Grass 06	Canopy Cover Perennial Forb 06	Canopy Cover Annual Grass 06	Canopy Cover Annual Forb 06	Canopy Cover Shrub 06	Canopy Cover Total 06
735	45.3	24.7	1.3	10	2.7	16	54.7
805	47.7	24.8	0.7	0	0	26.8	52.3
821	58.1	26.7	0.7	8	0	6.7	42.1
845	42.7	24.7	0.7	0	0	32	57.4
849	51.3	28	0.7	0	0	20	48.7
903	45	25	1	0	0	29	55
911	27.9	60.7	4	0	0	7.4	72.1
937	34	54.8	0.7	3.3	5.3	2	66.1
938	27.4	60	0.7	3.3	5.4	3.3	72.7
945	46	30	0.7	0	0	23.3	54
947	38	60	1.3	0	0.7	0	62
949	55.8	22.8	0.7	0.7	2	18.1	44.3
950	24.1	47.3	0	14	12.6	2	75.9
953	57.2	24	0.7	0	1.3	16.7	42.7
954	54.7	33.4	0	0	2	10	45.4
957	53.9	36	0	0	1.3	8.7	46
958	43.6	39.6	4.1	0	0.7	12.1	56.5
963	54.8	16.7	0	1.3	0.7	26.6	45.3
975	17.3	52	0.7	23.3	6.7	0	82.7
984	38	36	0.7	0	0	25.3	62
994	41	34	0	0	0	25	59
1004	27.5	57.7	0.7	6.7	7.3	0	72.4
1006	58.1	31.9	2.8	2	2	3.3	42

1074	39.3	37.5	3.3	0	0.7	19.3	60.8
1153	0	0	0	0	0	0	0
1158	39.9	50.6	3.4	1.3	4	0.7	60
1169	14.7	30.0	0.0	6.7	8	0.7	85.4
1222	0	0	0	0	0	0	0

Table 12. Change in field collected vegetation percent canopy cover from 2006 to 2009 on the Murphy Wildland Fire Complex.

Plot Number	Change from 06-09 Bare Ground	Change from 06-09 Perennial Grass	Change from 06-09 Perennial Forb	Change from 06-09 Annual Grass	Change from 06-09 Annual Forb	Change from 06-09 Shrub	Change from 06-09 Total Vegetatio n
735	9.3	-6.3	-2.7	3	-17.3	13	-10.3
805	-0.3	1.8	-0.3	0	-28	26.8	0.3
821	11.1	-12.3	0.7	5	-12	6.7	-11.9
845	0.7	-18.3	-3.3	0	-9	29	-1.6
849	15.3	-8	-4.3	0	0	-2	-14.3
903	3	7	-7	0	-32	29	-3
911	-1.1	-0.3	-7	0	0	7.4	0.1
937	5	-12.2	-2.3	2.3	5.3	2	-4.9
938	6.4	4	-21.3	3.3	5.4	2.3	-6.3
945	-27	15	0.7	-5	-7	23.3	27
947	-5	8	-3.7	0	0.7	0	5
949	-41.2	22.8	0.7	0.7	-1	18.1	41.3
950	-22.9	12.3	0	8	0.6	2	22.9
953	17.2	-18	-5.3	-5	-4.7	16.7	-16.3
954	1.7	-12.6	0	0	1	10	-1.6

957	6.9	-2	-15	0	0.3	8.7	-8
958	8.6	-9.4	-7.9	0	-0.3	8.1	-9.5
963	10.8	-25.3	-8	0.3	-3.3	26.6	-9.7
975	-16.7	-8	-3.3	22.3	6.7	-1	16.7
984	-11	-1	-6.3	0	-7	25.3	11
994	-8	9	-13	0	-12	24	8
1004	6.5	-20.3	-0.3	6.7	7.3	0	-6.6
1006	22.1	-13.1	-17.2	2	2	3.3	-23
1074	0.3	-11.5	-7.7	0	0.7	18.3	-0.2
1158	-1.1	0.6	-4.6	0.3	4	0.7	1

Table 13. Paired grazed and ungrazed pairs used in the paired t-test analysis comparing a change in vegetation cover on points in grazed and ungrazed plots.

Vegetation Type	Point in Grazed Plot	Ungrazed Plot
Shrub Steppe	849	954
Shrub Steppe	958	957
Shrub Steppe	845	953
Shrub Steppe	984	949
Shrub Steppe	945	903
Shrub Steppe	911	1074
Shrub Steppe	994	963
Grassland	937	938
Grassland	1158	1004
Grassland	1169	975
Grassland	947	950

APPENDIX B: Landscape Level Analysis Data (Randomly Selected Plot Data)

Table 14. dNDVI burn severity indices calculated immediately after the fire and one year post-fire for randomly selected grazed and ungrazed pairs in shrub steppe on the Murphy Wildland Fire Complex.

Vegetation Type	Pairs Grazed	Pairs Ungrazed	dNDVI 1 Y ear Grazed	dNDVI 1Year Ungrazed	dNDVI Immediate Grazed	dNDVI Immediate Ungrazed
Shrub Steppe	188	13	-0.012	0.042	0.037	0.098
Shrub Steppe	203	130	0.028	0.020	0.071	0.045
Shrub Steppe	313	307	0.041	-0.013	0.085	0.075
Shrub Steppe	574	771	0.000	0.027	0.078	0.133
Shrub Steppe	31	549	0.050	-0.013	0.062	0.053
Shrub Steppe	105	592	0.019	0.009	0.057	0.067
Shrub Steppe	167	729	0.035	0.045	0.064	0.083
Shrub Steppe	321	1	0.015	-0.007	0.064	0.039
Shrub Steppe	335	50	0.002	0.000	0.039	0.040
Shrub Steppe	440	58	0.016	-0.005	0.046	0.029
Shrub Steppe	544	63	0.000	0.009	0.053	0.051
Shrub Steppe	650	83	0.019	0.009	0.038	0.038
Shrub Steppe	664	87	-0.002	0.024	0.043	0.097
Shrub Steppe	680	104	0.026	0.016	0.040	0.023
Shrub Steppe	752	112	-0.003	0.002	0.042	0.010
Shrub Steppe	769	115	0.036	0.047	0.049	0.078
Shrub Steppe	65	12	0.020	0.020	0.062	0.055
Shrub Steppe	67	16	0.018	0.014	0.049	0.049
Shrub Steppe	90	28	0.030	0.027	0.042	0.050
Shrub Steppe	93	54	0.021	0.008	0.045	0.030
Shrub Steppe	119	85	0.019	0.047	0.049	0.036

Shrub Steppe	146	88	0.018	0.019	0.019	0.046
Shrub Steppe	152	89	0.059	0.003	0.102	0.031
Shrub Steppe	153	99	0.007	0.035	0.046	0.050
Shrub Steppe	156	107	0.017	0.044	0.058	0.056
Shrub Steppe	187	111	0.054	0.037	0.048	0.049
Shrub Steppe	262	116	0.008	0.034	0.045	0.063
Shrub Steppe	291	117	-0.006	-0.001	0.048	0.046
Shrub Steppe	292	125	0.042	0.035	0.048	0.027
Shrub Steppe	311	135	0.041	0.029	0.063	0.056
Shrub Steppe	316	145	0.001	0.005	0.044	0.034
Shrub Steppe	317	175	0.025	0.026	0.040	0.066
Shrub Steppe	342	193	0.038	0.027	0.048	0.059
Shrub Steppe	354	194	0.024	0.014	0.046	0.058
Shrub Steppe	364	195	0.050	0.027	0.096	0.040
Shrub Steppe	410	198	0.030	0.028	0.055	0.066
Shrub Steppe	35	27	0.014	0.046	0.021	0.090
Shrub Steppe	64	101	0.014	0.023	0.031	0.031
Shrub Steppe	84	114	0.016	0.008	0.063	0.028
Shrub Steppe	235	258	0.003	0.026	0.049	0.017
Shrub Steppe	446	310	0.006	0.031	0.061	0.039
Shrub Steppe	469	332	0.021	0.028	0.039	0.023
Shrub Steppe	717	390	-0.004	0.030	0.022	0.056
Shrub Steppe	736	413	0.045	0.029	0.057	0.048
Shrub Steppe	739	422	-0.008	0.029	0.026	0.055
Shrub Steppe	14	17	0.015	0.036	0.047	0.045
Shrub Steppe	21	36	-0.027	0.025	0.027	0.036
Shrub Steppe	82	72	0.011	0.002	0.026	0.045

Shrub Steppe	56	134	-0.002	0.026	0.035	0.024
Shrub Steppe	59	172	0.006	-0.002	0.022	0.034
Shrub Steppe	61	206	-0.012	0.021	0.001	0.058
Shrub Steppe	66	211	0.010	0.005	0.033	0.008
Shrub Steppe	140	234	0.015	0.033	0.019	0.057
Shrub Steppe	151	318	-0.012	0.004	0.032	0.033
Shrub Steppe	162	344	0.005	0.000	0.027	0.046
Shrub Steppe	200	398	0.015	0.041	0.019	0.049
Shrub Steppe	224	457	0.024	0.010	0.046	0.027
Shrub Steppe	232	460	0.030	0.027	0.053	0.005
Shrub Steppe	274	472	0.039	0.021	0.058	0.043
Shrub Steppe	279	510	0.013	-0.008	0.048	0.028

Table 15. RdNBR burn severity indices calculated immediately after the fire and one year post-fire for randomly selected grazed and ungrazed pairs in shrub steppe on the Murphy Wildland Fire Complex.

Vegetation Type	Pairs Grazed	Pairs Ungrazed	RdNBR 1 Year Grazed	RdNBR 1Year Ungrazed	RdNBR Immediate Grazed	RdNBR Immediate Ungrazed
Shrub Steppe	188	13	4.176867485	38.917	25.709	74.295
Shrub Steppe	203	130	13.63250542	10.670	36.754	30.143
Shrub Steppe	313	307	5.231208801	0.000	26.745	0.000
Shrub Steppe	574	771	7.036287308	75.283	36.434	145.343
Shrub Steppe	31	549	5.034587383	10.739	28.296	44.430
Shrub Steppe	105	592	2.720922232	25.498	35.559	66.174
Shrub Steppe	167	729	2.399983644	3.875	26.976	33.270
Shrub Steppe	321	1	12.3874712	22.236	28.686	83.789

Shrub Steppe	335	50	0.161321238	13.569	18.642	29.910
Shrub Steppe	440	58	6.322587967	4.803	25.280	22.235
Shrub Steppe	544	63	1.225060582	12.891	20.771	40.098
Shrub Steppe	650	83	5.262804031	14.912	32.164	33.257
Shrub Steppe	664	87	8.347249031	15.014	33.036	43.579
Shrub Steppe	680	104	9.523566246	12.368	33.021	31.085
Shrub Steppe	752	112	4.918403625	7.425	27.810	24.084
Shrub Steppe	769	115	7.961026669	19.177	26.346	47.246
Shrub Steppe	65	12	4.782261372	12.034	19.573	39.424
Shrub Steppe	67	16	8.556615829	14.892	33.878	39.800
Shrub Steppe	90	28	9.999685287	4.843	29.566	18.749
Shrub Steppe	93	54	1.352822661	7.485	37.674	26.420
Shrub Steppe	119	85	7.687530994	5.305	36.137	34.700
Shrub Steppe	146	88	11.39353085	7.788	35.692	29.647
Shrub Steppe	152	89	9.080849648	7.117	30.640	26.529
Shrub Steppe	153	99	2.792351007	10.303	36.090	31.460
Shrub Steppe	156	107	9.872787476	3.450	23.475	30.251
Shrub Steppe	187	111	4.09992981	2.026	30.348	29.481
Shrub Steppe	262	116	8.420536041	8.913	49.660	33.389
Shrub Steppe	291	117	0.251643986	18.936	31.007	36.069
Shrub Steppe	292	125	4.490848541	6.744	34.552	38.392
Shrub Steppe	311	135	14.79875755	15.481	40.624	39.873
Shrub Steppe	316	145	12.55416012	7.173	23.508	28.375
Shrub Steppe	317	175	5.34923315	25.226	28.097	38.041
Shrub Steppe	342	193	2.55237627	10.479	9.254	30.697
Shrub Steppe	354	194	10.19147873	18.596	37.781	35.426
Shrub Steppe	364	195	11.59232616	10.699	29.912	34.702
Shrub Steppe	410	198	1.70984292	9.694	25.268	28.218
Shrub Steppe	35	27	8.780562401	8.983	32.947	46.662

Shrub Steppe	64	101	1.187488079	18.329	20.237	32.553
Shrub Steppe	84	114	8.238661766	5.155	22.497	26.459
Shrub Steppe	235	258	12.85592079	3.975	23.860	40.206
Shrub Steppe	446	310	1.02829206	3.368	24.736	32.032
Shrub Steppe	469	332	6.998293877	8.993	26.182	39.062
Shrub Steppe	717	390	5.476175785	13.282	20.302	33.825
Shrub Steppe	736	413	6.041908741	12.122	34.586	26.945
Shrub Steppe	739	422	11.56770897	6.565	37.871	22.049
Shrub Steppe	14	17	9.094739914	13.569	22.331	37.516
Shrub Steppe	21	36	2.033175945	18.192	28.239	34.414
Shrub Steppe	82	72	5.697331429	6.952	25.549	29.162
Shrub Steppe	56	134	2.678179741	1.393	22.213	5.015
Shrub Steppe	59	172	9.043879509	6.963	22.482	21.363
Shrub Steppe	61	206	3.075581074	8.433	8.106	31.542
Shrub Steppe	66	211	12.47415829	4.453	29.945	9.455
Shrub Steppe	140	234	3.958716631	12.752	6.972	29.898
Shrub Steppe	151	318	10.06293583	5.626	29.470	16.376
Shrub Steppe	162	344	1.336058855	2.604	7.654	15.367
Shrub Steppe	200	398	7.448441029	9.985	23.987	24.553
Shrub Steppe	224	457	7.018699169	5.090	24.184	6.563
Shrub Steppe	232	460	16.41714287	0.692	41.376	-2.735
Shrub Steppe	274	472	10.23069763	4.008	28.092	16.290
Shrub Steppe	279	510	4.924	11.495	19.481	28.914

Table 16. dNBR burn severity indices calculated immediately after the fire and one year post-fire for randomly selected grazed and ungrazed pairs in shrub steppe on the Murphy Wildland Fire Complex.

Vegetation Type	Pairs Grazed	Pairs Ungrazed	dNBR 1Year Grazed	dNBR 1Year Ungrazed	dNBR Immediate Grazed	dNBR Immediate Ungrazed
Shrub Steppe	188	13	30.531	190.641	187.919	363.947
Shrub Steppe	203	130	114.754	84.345	309.384	238.284
Shrub Steppe	313	307	48.429	38.755	247.595	310.305
Shrub Steppe	574	771	54.774	211.237	283.621	407.819
Shrub Steppe	31	549	53.067	48.184	298.255	199.354
Shrub Steppe	105	592	26.699	67.902	348.922	176.219
Shrub Steppe	167	729	21.504	39.408	241.710	338.372
Shrub Steppe	321	1	132.546	60.968	306.945	229.740
Shrub Steppe	335	50	1.676	103.310	193.650	227.716
Shrub Steppe	440	58	54.414	39.213	217.569	181.541
Shrub Steppe	544	63	10.681	88.252	181.099	274.516
Shrub Steppe	650	83	42.407	79.702	259.176	177.756
Shrub Steppe	664	87	84.448	137.577	334.224	399.314
Shrub Steppe	680	104	98.825	106.980	342.651	268.888
Shrub Steppe	752	112	48.381	54.957	273.557	178.268
Shrub Steppe	769	115	78.118	134.481	258.519	331.307
Shrub Steppe	65	12	46.199	98.990	189.087	324.292
Shrub Steppe	67	16	78.869	107.231	312.261	286.577
Shrub Steppe	90	28	99.992	42.475	295.651	164.434
Shrub Steppe	93	54	12.469	69.797	347.258	246.350
Shrub Steppe	119	85	65.611	42.085	308.414	275.268
Shrub Steppe	146	88	89.758	64.450	281.183	245.348
Shrub Steppe	152	89	94.328	53.414	318.276	199.115
Shrub Steppe	153	99	27.725	94.404	358.328	288.263

Shrub Steppe	156	107	98.396	35.713	233.964	313.111
Shrub Steppe	187	111	39.090	21.213	289.341	308.729
Shrub Steppe	262	116	72.738	86.686	428.977	324.720
Shrub Steppe	291	117	2.492	163.794	306.996	311.997
Shrub Steppe	292	125	45.057	62.433	346.664	355.423
Shrub Steppe	311	135	123.317	109.854	338.514	282.942
Shrub Steppe	316	145	108.593	65.727	203.342	259.999
Shrub Steppe	317	175	40.499	113.188	212.724	170.686
Shrub Steppe	342	193	24.178	93.894	87.663	275.045
Shrub Steppe	354	194	99.117	158.291	367.437	301.548
Shrub Steppe	364	195	108.275	86.084	279.382	279.218
Shrub Steppe	410	198	17.658	85.616	260.951	249.223
Shrub Steppe	35	27	74.741	37.299	280.447	193.741
Shrub Steppe	64	101	10.335	135.670	176.131	240.955
Shrub Steppe	84	114	80.842	46.323	220.749	237.746
Shrub Steppe	235	258	116.040	36.284	215.368	367.006
Shrub Steppe	446	310	8.569	26.715	206.121	254.104
Shrub Steppe	469	332	63.168	73.427	236.320	318.923
Shrub Steppe	717	390	54.759	103.225	203.007	262.876
Shrub Steppe	736	413	57.604	113.035	329.747	251.251
Shrub Steppe	739	422	101.450	60.778	332.131	204.119
Shrub Steppe	14	17	73.178	119.839	179.676	331.349
Shrub Steppe	21	36	21.120	175.922	293.334	332.792
Shrub Steppe	82	72	67.140	74.382	301.082	312.038
Shrub Steppe	56	134	28.427	13.381	235.781	48.182
Shrub Steppe	59	172	79.876	64.329	198.561	197.372
Shrub Steppe	61	206	35.124	88.883	92.575	332.460
Shrub Steppe	66	211	107.754	43.527	258.675	92.417
Shrub Steppe	140	234	39.835	121.581	70.154	285.056

Shrub Steppe	151	318	104.422	54.401	305.808	158.365
Shrub Steppe	162	344	14.176	25.496	81.204	150.470
Shrub Steppe	200	398	75.018	54.962	241.590	135.150
Shrub Steppe	224	457	58.084	42.607	200.141	54.930
Shrub Steppe	232	460	137.119	6.551	345.578	-25.912
Shrub Steppe	274	472	98.746	34.495	271.140	140.196
Shrub Steppe	279	510	47.963	105.669	189.761	265.787

Table 17. dNDVI burn severity indices calculated immediately after the fire and one year post-fire for randomly selected grazed and ungrazed pairs in grassland cover types on the Murphy Wildland Fire Complex.

Vegetation Type	Pairs Grazed	Pairs Ungrazed	dNDVI 1Year Grazed	dNDVI 1Year Ungrazed	dNDVI Immediate Grazed	dNDVI Immediate Ungrazed
Grassland	3	8	0.016	0.019	0.078	0.052
Grassland	25	120	-0.010	0.030	0.034	0.071
Grassland	40	131	0.028	0.037	0.085	0.077
Grassland	57	143	-0.001	0.046	0.027	0.050
Grassland	103	164	0.031	-0.002	0.043	0.078
Grassland	123	209	0.000	-0.001	0.060	0.015
Grassland	171	214	-0.002	0.013	0.042	0.068
Grassland	212	230	-0.003	0.025	0.037	0.089
Grassland	247	261	0.024	0.033	0.078	0.053
Grassland	265	326	-0.029	0.034	0.024	0.082
Grassland	282	330	0.044	0.008	0.036	0.032
Grassland	327	353	0.033	0.000	0.066	0.042
Grassland	374	397	0.016	0.000	0.060	0.020
Grassland	404	424	0.008	-0.034	0.054	0.058
Grassland	416	465	0.037	0.031	0.083	0.088

Grassland	492	478	-0.005	0.016	0.036	0.021
Grassland	527	548	0.005	-0.011	0.047	0.034
Grassland	594	559	-0.003	-0.012	0.046	0.065
Grassland	636	595	0.015	0.024	0.070	0.038
Grassland	698	627	0.024	-0.042	0.051	0.065
Grassland	699	631	0.035	0.015	0.074	0.039
Grassland	728	637	0.016	0.041	0.038	0.075
Grassland	770	653	0.012	0.015	0.041	0.074
Grassland	783	655	-0.017	0.011	0.050	0.052
Grassland	160	277	0.010	0.048	0.026	0.054
Grassland	516	599	0.043	-0.005	0.079	0.055
Grassland	647	620	-0.031	0.025	0.038	0.056
Grassland	10	163	0.040	-0.027	0.025	0.055
Grassland	22	204	0.030	0.191	0.041	0.255
Grassland	24	379	0.036	0.025	0.039	0.072
Grassland	51	382	0.023	0.028	0.077	0.059
Grassland	53	407	0.049	0.044	0.056	0.082
Grassland	69	453	-0.002	0.048	0.077	0.081
Grassland	141	454	0.047	0.023	0.091	0.043
Grassland	148	480	0.019	-0.019	0.063	0.020
Grassland	539	575	0.041	0.027	0.082	0.021
Grassland	562	589	0.031	0.012	0.067	0.074
Grassland	576	692	0.020	-0.011	0.056	0.023
Grassland	602	789	0.041	0.027	0.053	0.051
Grassland	515	6	-0.025	0.023	0.038	0.050
Grassland	523	666	-0.004	-0.002	0.032	0.019
Grassland	244	521	-0.008	-0.011	0.012	0.023
Grassland	218	38	0.012	0.020	0.027	0.043
Grassland	341	98	-0.020	-0.002	0.083	0.048

Grassland	96	49	0.024	0.031	0.035	0.072
Grassland	369	158	-0.001	0.031	0.014	0.054
Grassland	501	185	-0.007	0.028	0.020	0.053
Grassland	537	189	0.020	0.030	0.048	0.041
Grassland	180	81	0.005	0.011	0.023	0.016
Grassland	301	176	0.007	0.007	0.037	0.066
Grassland	165	75	0.013	-0.002	0.005	0.014
Grassland	210	166	0.008	0.024	0.038	0.028
Grassland	239	340	-0.009	0.006	0.064	0.018
Grassland	593	426	0.026	0.015	0.024	0.011
Grassland	5	7	0.040	0.021	0.040	0.010
Grassland	254	9	0.032	0.051	0.036	0.008
Grassland	329	34	0.031	0.038	0.028	0.055
Grassland	359	41	0.037	0.042	0.094	0.033
Grassland	370	55	0.014	0.034	0.035	0.056
Grassland	399	73	0.012	0.044	0.018	0.048

Table 18. RdNBR burn severity indices calculated immediately after the fire and one year post-fire for randomly selected grazed and ungrazed pairs in grassland cover types on the Murphy Wildland Fire Complex.

Vegetation Type	Pairs Grazed	Pairs Ungrazed	RdNBR 1Year Grazed	RdNBR 1Year Ungrazed	RdNBR Immediate Grazed	RdNBR Immediate Ungrazed
Grassland	3	8	15.621	-1.852	68.904	16.249
Grassland	25	120	2.330	15.661	23.514	45.874
Grassland	40	131	8.813	-1.324	38.382	25.609
Grassland	57	143	3.411	7.071	21.253	39.755
Grassland	103	164	7.788	22.145	24.619	79.669
Grassland	123	209	8.330	3.984	72.546	21.744

Grassland	171	214	5.370	4.935	39.049	23.639
Grassland	212	230	19.542	5.678	75.008	38.804
Grassland	247	261	13.365	10.448	50.986	27.083
Grassland	265	326	10.414	0.976	41.669	21.353
Grassland	282	330	8.829	8.733	32.059	47.111
Grassland	327	353	4.172	8.437	28.918	36.236
Grassland	374	397	10.216	7.456	45.164	25.916
Grassland	404	424	6.576	7.237	33.886	72.599
Grassland	416	465	7.253	0.403	29.952	19.099
Grassland	492	478	10.232	9.621	32.538	30.419
Grassland	527	548	15.555	24.694	52.897	58.438
Grassland	594	559	5.196	3.830	27.824	27.452
Grassland	636	595	11.111	4.045	49.531	30.026
Grassland	698	627	12.839	-3.639	49.733	29.150
Grassland	699	631	31.198	10.913	83.840	30.107
Grassland	728	637	10.737	2.204	35.250	19.736
Grassland	770	653	3.837	17.798	34.918	80.040
Grassland	783	655	9.336	1.962	43.122	19.408
Grassland	160	277	9.831	6.842	41.851	23.427
Grassland	516	599	14.944	-0.931	51.849	20.106
Grassland	647	620	-5.123	6.818	20.612	18.681
Grassland	10	163	6.503	2.993	1.347	58.591
Grassland	22	204	6.274	10.307	20.006	37.073
Grassland	24	379	3.354	5.843	17.013	24.992
Grassland	51	382	15.683	10.486	52.952	23.713
Grassland	53	407	10.912	11.313	31.784	32.482
Grassland	69	453	6.624	36.644	39.628	73.890
Grassland	141	454	14.145	4.147	48.249	18.303
Grassland	148	480	3.667	5.787	19.739	32.647

Grassland	539	575	2.969	-0.896	20.097	8.071
Grassland	562	589	6.606	9.986	22.906	45.811
Grassland	576	692	0.858	11.277	31.677	38.564
Grassland	602	789	3.556	0.227	20.461	13.313
Grassland	515	6	1.287	10.395	27.253	28.383
Grassland	523	666	4.923	4.980	30.461	27.859
Grassland	244	521	1.033	1.947	6.416	4.149
Grassland	218	38	8.033	6.849	17.733	23.123
Grassland	341	98	6.920	12.940	31.881	28.905
Grassland	96	49	5.372	8.467	21.442	23.211
Grassland	369	158	0.115	10.185	16.840	22.249
Grassland	501	185	-0.130	1.054	11.135	14.729
Grassland	537	189	2.116	9.088	22.019	9.256
Grassland	180	81	8.803	6.367	22.694	16.265
Grassland	301	176	5.947	8.653	10.758	22.664
Grassland	165	75	2.047	7.625	13.007	42.371
Grassland	210	166	2.296	10.977	17.044	28.729
Grassland	239	340	0.815	4.060	25.539	11.743
Grassland	593	426	3.837	7.266	14.392	23.328
Grassland	5	7	-1.729	-2.656	7.419	7.002
Grassland	254	9	-0.109	2.978	8.353	-13.277
Grassland	329	34	-0.609	-0.097	8.819	11.940
Grassland	359	41	14.562	-1.049	42.556	5.536
Grassland	370	55	0.535	1.961	18.770	22.747
Grassland	399	73	-2.455	0.588	2.264	12.497

Table 19. dNBR burn severity indices calculated immediately after the fire and one year post-fire for randomly selected grazed and ungrazed pairs in grassland cover types on the Murphy Wildland Fire Complex.

Vegetation Type	Pairs Grazed	Pairs Ungrazed	dNBR 1Year Grazed	dNBR 1Year Ungrazed	dNBR Immediate Grazed	dNBR Immediate Ungrazed
Grassland	3	8	72.050	-18.048	317.812	158.344
Grassland	25	120	20.575	95.306	207.675	279.166
Grassland	40	131	60.810	-11.187	264.850	216.424
Grassland	57	143	27.448	71.426	171.003	401.565
Grassland	103	164	69.195	89.657	218.737	322.549
Grassland	123	209	38.421	33.916	334.613	185.091
Grassland	171	214	35.153	35.801	255.622	171.484
Grassland	212	230	106.763	40.006	409.791	273.400
Grassland	247	261	80.737	89.582	308.003	232.226
Grassland	265	326	63.849	8.199	255.476	179.391
Grassland	282	330	72.976	56.367	264.983	304.084
Grassland	327	353	35.849	66.927	248.456	287.452
Grassland	374	397	65.044	38.759	287.551	134.723
Grassland	404	424	59.358	34.112	305.863	342.216
Grassland	416	465	61.102	3.536	252.311	167.501
Grassland	492	478	89.737	51.424	285.364	162.587
Grassland	527	548	91.336	131.055	310.608	310.142
Grassland	594	559	37.136	32.139	198.880	230.362
Grassland	636	595	65.700	37.185	292.868	276.010
Grassland	698	627	70.142	-34.177	271.704	273.797
Grassland	699	631	114.681	57.918	308.185	159.783
Grassland	728	637	82.346	19.213	270.344	172.076
Grassland	770	653	24.595	79.859	223.832	359.130
Grassland	783	655	60.681	18.201	280.293	180.074

Grassland	160	277	73.273	62.087	311.925	212.587
Grassland	516	599	97.132	-8.711	337.013	188.066
Grassland	647	620	-42.345	44.953	170.370	123.172
Grassland	10	163	57.350	14.544	11.875	284.749
Grassland	22	204	53.172	157.961	169.537	568.167
Grassland	24	379	27.720	45.345	140.615	193.970
Grassland	51	382	92.091	81.875	310.929	185.156
Grassland	53	407	103.320	95.047	300.934	272.891
Grassland	69	453	45.710	127.188	273.443	256.465
Grassland	141	454	85.448	37.337	291.469	164.799
Grassland	148	480	33.672	41.669	181.264	235.073
Grassland	539	575	24.241	-9.958	164.080	89.727
Grassland	562	589	50.016	67.320	173.424	308.840
Grassland	576	692	8.872	77.205	327.530	264.016
Grassland	602	789	27.431	2.082	157.849	122.257
Grassland	515	6	11.198	104.345	237.193	284.910
Grassland	523	666	38.039	43.016	235.351	240.655
Grassland	244	521	10.956	16.878	68.047	35.962
Grassland	218	38	76.054	46.940	167.901	158.483
Grassland	341	98	59.332	114.283	273.365	255.290
Grassland	96	49	48.697	69.130	194.383	189.507
Grassland	369	158	1.041	95.133	152.265	207.807
Grassland	501	185	-1.261	9.513	108.292	132.875
Grassland	537	189	19.872	81.283	206.821	82.780
Grassland	180	81	67.025	49.485	172.783	126.401
Grassland	301	176	58.500	81.369	105.830	213.125
Grassland	165	75	20.174	39.637	128.194	220.268
Grassland	210	166	20.960	75.237	155.582	196.905
Grassland	239	340	7.999	32.964	250.601	95.349

Grassland	593	426	30.954	62.534	116.107	200.763
Grassland	5	7	-16.806	-23.571	72.132	62.145
Grassland	254	9	-1.120	20.444	86.154	-91.138
Grassland	329	34	-5.802	-0.896	84.082	110.149
Grassland	359	41	85.509	-11.560	249.886	60.996
Grassland	370	55	5.461	15.383	191.680	178.433
Grassland	399	73	-24.147	5.922	22.271	125.967