A REVIEW OF LIVESTOCK GRAZING AND RANGE MANAGEMENT IN UTAH

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CONTRACT
Setting the Stage for a Livestock
Grazing Policy in Utah

State of Utah Control No. 080720
Utah State University Control No. 080300

February 2009

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INTRODUCTION

The intent of this document is to present an overview of the important dimensions of livestock grazing and range management in Utah and the surrounding intermountain region. A summary on the history of livestock grazing provides a fundamental understanding of past range management practices and the resulting impacts to natural resources. A review of the socioeconomics of livestock grazing is an essential component in recognizing the value of financial development and livelihood for private ranchers and rural ranching communities. Subsequent sections in the document outline the impacts of improper grazing and poor range management practices on natural resources and processes. The topics of animal and plant biodiversity, invasive plants, fire regimes, soil health, and water quality are covered, as a significant amount of research and literature identifies these as the primary constituents influenced by livestock. The effects of livestock grazing on natural resources and processes and research in range science have initiated a transition to more compatible and comprehensive livestock and range management practices. Therefore, each section is accompanied with a series of management strategies that account for multiple aspects of the environment wherein the livestock are grazing. Knowledge and integration of these important dimensions of livestock grazing and range management are imperative in developing holistic plans and sustainable practices.

HISTORY OF LIVESTOCK GRAZING

The beginnings of the livestock industry in the western United States are associated with exploration and colonization. In 1540, the Spanish explorer Francisco Coronado journeyed from Mexico northward into Arizona, New Mexico, and Colorado with a large number of cattle, sheep, hogs, horses, and mules (Sampson, 1952; Stoddart et al., 1975). Subsequent explorations were helpful in extending colonies and livestock into the southwestern and western United States. Missions established in Texas, New Mexico, and Arizona became livestock centers in the early 1700s (Stewart, 1936; Sampson, 1952).

During the next century, livestock markets and production were stimulated by the California Gold Rush (1848–1855) and the Civil War (1861–1865). Large herds were driven to California where they competed with native beef and with stock from Mexico, Arizona, and New Mexico. The demand for meat and animal products during the Civil War brought large cattle shipments from Texas to the Confederate Army (Stewart, 1936; Sampson, 1952). The inflationary period after the Civil War and the completion of the first transcontinental railroad in 1869 initiated a livestock boom which affected much of the western United States. Concurrently, the Mormons (Latter-day Saints) filled the Utah ranges, beginning in 1847, with foundation stock that they drove across the Plains, and with lean cattle and horses obtained by trading with other emigrants (Stewart, 1936; Sampson, 1952). Under the direction of Parley P. Pratt, one of the first apostles and missionaries of the Latter Day Saint movement, they brought with them 358 sheep, 887 cattle, 2,213 oxen, 35 hogs, 124 horses, and 716 chickens (Jacobs, 1984). The development of mining camps in the Great Basin also brought a great demand for wool and mutton (Knapp, 1996).

After the Mormon pioneers arrived in Salt Lake Valley, Brigham Young, President of the Mormon Church, took possession of Antelope Island in the Great Salt Lake for use as a herd ground for livestock. However, Antelope Island became overstocked with grazing animals, so Brigham Young sent large numbers of horses and cattle to new range near the Sevier River. In 1855, a number of families moved a substantial amount of cattle, sheep, and horses to high-quality rangeland in the south end of Rush Valley. By 1875, the range in Rush Valley was extremely depleted, and cows were calving only every other year. Grazing problems in Utah became acute, and it was recognized that principles of good range management had not been learned by the settlers (Jacobs, 1984).

By 1890 the last western open range was fully stocked. It is estimated that there were over 26 million cattle and 20 million sheep in the 17 western states. The resulting competition for forage between cattle and sheep was intense. While cattle and sheep competed for many of the same resources, the impact of sheep on the landscape was greater than that of cattle. Cattle were confined to relatively gentle terrain in sagebrush-bunchgrass ecosystems, whereas sheep could travel into steeper and rougher terrain (Knapp, 1996). As an outcome, range and forest lands were heavily overgrazed and depleted, and stockmen of both factions engaged in bitter struggles over land (Stewart, 1936; Sampson, 1952). High-elevation watersheds on the Wasatch Plateau in central Utah were severely overgrazed, resulting in catastrophic flooding in the adjacent communities of Manti and Ephraim (Prevedel et al., 2005).

As an initial solution to halt overgrazing, the federal government began managing livestock grazing on the established forest reserves. In 1902, Sanpete Valley citizens petitioned the federal government to establish another forest reserve above Manti. Subsequently, President Theodore Roosevelt signed the proclamation creating the Manti Forest Reserve. In 1905, the jurisdiction of the existing forest reserves was transferred from the General Land Office to the Bureau of Forestry in the Department of Agriculture. The agency was shortly renamed the Forest Service, and Gifford Pinchot, the Chief of the Forest Service, imposed grazing fees and established a use-by-permit system (Prevedel and Johnson, 2005). The establishment and expansion of the National Forest system virtually ended the range wars and marked the beginning of scientific range management (Sampson, 1952; Stoddart et al., 1975).

During the 1910s and 1920s, scientific and professional techniques of range management were adopted. Region 4 of the Forest Service established a research station as a model for the implementation of research-validated models. In 1912, the Great Basin Experiment Station was established in Ephraim Canyon on the Manti Forest. Arthur W. Sampson, who is noted for his range and forest research, became the first director, and his research became models for range reconnaissance and carrying-capacity studies. Sampson's work in Utah also provided the justification for deferred and rotation grazing (Alexander, 1987). This practice of technical professionalism and experimentation initiated close cooperation between scientists and range managers and allowed rapid implementation of the research results. Subsequently, it helped Region 4 to develop the capacity to adopt changing techniques and implement effective range management in the Intermountain West. Additionally, long-term records and early studies evaluating the impacts of various levels of grazing at the Great Basin Experimental Station contributed to the advancement of methods in rangeland restoration (Alexander, 1987; Lugo et al., 2006).

In 1933, the Desert Experimental Range in Pine Valley, approximately 40 miles west of Milford, was established. President Herbert Hoover provided the basis for the Desert Experimental Range when he withdrew 87 square miles of land from the public domain as an agricultural range experiment station. The development of the experimental range was prompted by concern for the condition of public rangelands. Expanses of Great Basin rangelands dominated by low-shrubs had nearly become devoid of vegetation (Clary and Holmgren, 1982). In the winter of 1934-35, sheep grazing studies were initiated to study the economic and ecological impacts of grazing at different intensities, seasons, and frequencies (Adams et al., 2004). Early studies concluded that poor range condition was a result of improper grazing practices rather than the cyclical periods of drought. Restoration efforts were attempted; yet cultural improvement practices using planting techniques were not successful. Subsequent studies indicated that range recovery was possible given that higher levels of grazing during the winter months were not permitted and that grazing was not allowed to repeatedly occur on the same area year after year during the late winter-early spring months (Clary and Holmgren, 1982).

The progression of scientific range management was accompanied by additional legislation which sought to regulate grazing on public lands. The Taylor Grazing Act of 1934 created the Grazing Service (presently the Bureau of Land Management) and authorized the establishment of grazing districts on public lands that were considered to be valuable for grazing and raising forage crops. This act also established the permit and leasing system on public lands and defined the requirements for the distribution of funds received from grazing. In 1976, the Federal Land Policy and Management Act was established to limit the length of permits and leases to ten years and to regulate seasonal limits on grazing. In 1978, the Public Rangelands Improvement Act required the Bureau of Land Management and Forest Service to inventory and manage lands in the western states with the commitment to improve the conditions on public rangeland. As a component of this commitment, the grazing fee formula was established to account for cattle density and forage consumption (US GAO, 2005).

SOCIOECONOMICS OF LIVESTOCK GRAZING

Livestock grazing, one of the earliest uses of public land since the western United States was settled, continues to be an important, yet controversial, land use. Livestock grazing is the most widespread economic land use in western North America (Bock et al., 1993). It is allowed on public lands, primarily administered by the Bureau of Land Management and the United States Forest Service, for the purpose of fostering economic development for private ranchers and ranching communities. In the western United States, ranchers depend on vast tracts of both private and federal lands to graze cattle seasonally. Access to forage on federal lands increases the total amount of forage available to livestock, thus enabling greater livestock production for private ranchers (US GAO, 2005).

Although livestock grazing on public rangeland is important to private ranchers and the economies of local communities, the government must manage public rangeland for a variety of multiple use interests (Bastian et al., 1991). Historically, policy with respect to the management of public lands was formulated largely in consideration of its effects upon livestock grazing (Clawson, 1950). However, management and use of public rangeland is becoming more of a controversial issue. Presently, public land managers are faced with allocating rangeland resources among alternative uses, such as domestic livestock production, recreation, wildlife, and watershed health, with an overall goal of maximizing social welfare (Bastian et al., 1991). While some multiple uses of rangeland are complementary, others are largely competitive (Clawson, 1950). Consequently, fervently opposing views exist concerning use of public rangeland (Bastian et al., 1991).

The Federal Grazing System

A system for the management of livestock grazing on the public domain was approved by Congress in 1934 under the Taylor Grazing Act. The general objectives of the Taylor Grazing Act were to stop injury to the public lands by preventing overgrazing and to stabilize the livestock industry dependent upon public rangeland (Foss, 1959). To accomplish these purposes, grazing districts were established on 142 million acres of federal lands that were considered to be valuable for grazing, and the permit and leasing system was instituted (US GAO, 2005; Foss, 1959). Advisory board members of the newly-created Grazing Service in the Department of the Interior agreed upon a uniform grazing fee of five cents per Animal Unit Month (AUM) (Foss, 1959). An AUM is the amount of forage needed to sustain one cow and her calf, one bull, one horse, or five sheep or goats for one month (USDI BLM, 2008). The Secretary of Interior accepted this recommendation and the five cent AUM fee became effective during the 1936 grazing season (Foss, 1959).

Grazing fees on federal lands have consistently been a subject of contention (Holechek and Hawkes, 1993). Since the five cent AUM fee became effective for the 1936 grazing season, several studies have been conducted to determine the validity of a uniform grazing fee on public lands. The Saunderson-Leech Appraisal Study of 1941 evaluated grazing fees on the basis of range appraisals of average forage values in each of the range states. The results of the study indicated that the base values for grazing fees per AUM should have been 8.4 cents for Arizona, 13.8 cents for California, 18.5 cents for Colorado, 16.7 cents for Idaho, 16.7 cents for Montana, 12.4 cents for Nevada, 7.3 cents for New Mexico, 14 cents for Oregon, 12 cents for Utah, and 15.8 cents for Wyoming. The study also recommended that the fees should vary from year to year depending on the price of beef and mutton. In 1944, Clarence L. Forsling, the newly-appointed director of the Grazing Service, revived the fee question and submitted a proposal that detailed a trebled fee schedule that was based on forage value. Both the Saunderson-Leech Appraisal Study and the Forsling Proposal were renounced by the National Advisory Board Council. Grazing fees remained at five cents per AUM when the Grazing Service and the General Land Office were reorganized into the Bureau of Land Management (BLM) in 1946 (Foss, 1959).

By 1950, the BLM officials had increased the grazing fees by minute increments, to eight cents and then 12 cents, because the prevailing fees failed to provide enough revenue. In 1952, the director of the BLM, Marion Clawson, instituted another grazing fee study that evaluated the value of forage in different areas according to average livestock prices during the five year period from 1947 to 1951. The recommended grazing rates in the study ranged from 20 cents per AUM to 40 cents per AUM, for an average fee of 28 cents per AUM. In 1954, the National Advisory Board Council agreed to increase fees based on a new fee system accounting for the combined prices of cattle and sheep in the markets of 11 western states. The new fee system became effective in 1955, and a compromise of 15 cents per AUM was adopted, with the understanding that a new formula would become effective in 1957. The grazing fee for 1958, based on livestock prices during 1957, was set at 19 cents per AUM (Foss, 1959).

In 1969, the Bureau of Land Management and the Forest Service established a nationally uniform grazing fee that was to be implemented over a ten-year period. Before the uniform grazing fee was implemented, the Forest Service had 19 different base grazing fee rates for cattle and 17 different base grazing fee rates for sheep on western national forests based on varying forage values within states, national forests, and ranger districts (Bartlett et al., 2002). Prior to the completion of the ten-year period for the nationally uniform grazing fee, the Public Rangelands Improvement Act (PRIA) of 1978 was passed by Congress and a new fee schedule was established for 1979. The federal grazing fee enacted by PRIA, and extended by Executive Order 12548 in 1986, applies to 16 states in the West and Great Plains, including Arizona, California, Colorado, Idaho, Kansas, Montana, Nebraska, New Mexico, Nevada, North Dakota, Oklahoma, Oregon, South Dakota, Utah, Washington, and Wyoming (USDI BLM, 2008; US GAO, 2005).

The uniform federal grazing fee is computed by multiplying a base value of $1.23 per AUM, as determined by a 1966 study, by the sum of three indices. The indices are calculated by the USDA National Agricultural Statistics Service and are based on data collected by agency surveys. The three indices are Forage Value Index (FVI), Beef Cattle Price Index (BCPI), and Prices Paid Index (PPI). The federal grazing fee is adjusted annually based on the three indices, and the indices are affected by drought, wildfire, private land lease rates, beef cattle prices, and the cost of livestock production. Consequently, the fee rises, falls, or remains the same based on market and climatic conditions, with livestock ranchers paying more when conditions are better and less when conditions have declined (USDI BLM, 2008). Under PRIA and Executive Order 12548, increases and decreases in the fee are limited to 25 percent per year, and the fee cannot fall below $1.35 per AUM (US GAO, 2005).

The uniform federal grazing fee on lands administered by the BLM and Forest Service is generally lower than the fees charged by other federal agencies, states, and private entities. For example, grazing fees on private non-irrigated lands in the state of Utah in 2007 were $12.90 per AUM, as compared to the $1.35 per AUM on public lands (Utah Agricultural Statistics Service, 2008). The federal grazing fee is generally low because it was designed to account for livestock industry prices and to support ranchers in the western livestock industry (US GAO, 2005). Others suggest that grazing fees were set low to encourage good stewardship and private investments on public lands.

Despite the low uniform fee, it is well established that public land grazing fees are below the market value of the forage (Torrell and Doll, 1991). Consequently, some economists, researchers, and environmentalists criticize the uniform fee and argue that there should be multiple fees based on forage values determined by site-specific factors, such as forage type, forage quality, forage quantity, location, range condition, and level of range improvement (Bartlett et al., 2002; US GAO, 2005). For instance, nearly four times the amount of land is required per AUM in cold desert rangelands as compared to shortgrass prairie rangelands due to lower precipitation and more rugged terrain. Forage production in cold desert rangelands is between 100 to 400 pounds per acre, whereas forage production in shortgrass prairie rangelands is between 600 to 1,400 pounds per acre (Holechek and Hawkes, 1993).

The argument that public land ranchers pay substantially less to graze livestock on federal lands than do ranchers who lease similar privileges from private landowners contributed to the public policy reform movement known as the Rangeland Reform of '94 (LeFrance and Watts, 1995). Rangeland Reform '94 consisted of three separate reform efforts, including (1) a series of amendments to the regulations and policies that govern grazing on federal land, (2) a proposal to add national standards and guidelines to the regulations in order to establish minimum acceptable ecological conditions for federal rangeland, and (3) a proposal to bring federal grazing fees closer to the fee charged for forage on private land (Nicoll, 2006). The proposed increase in grazing fee would have been based on locational differences, such as the nutritive content of the available forage, the quantity of forage available per unit area, the availability and proximity of livestock water to forage resources, and local market prices for livestock and substitute feed (LaFrance and Watts, 1995).

The proposal to increase public land grazing fees was unquestionably the most controversial element of the reform (Nicoll, 2006). However, economic studies of ranches in the western United States indicate that public grazing fees represent a small portion of total grazing permit operating costs for ranchers, estimating four to five percent (Holechek and Hawkes, 1993). A nominal fee increase over a period of time would not force most western ranchers out of business, but a sudden sizeable increase to the full value of range forage may (Roberts, 1963). The fee formula proposed in the Rangeland Reform '94 attempted to establish a fee structure that would result, over the course of three years, in a fee that more closely represented the fair market value of the forage on federal land (Nicoll, 2006).

The reform generated a great deal of conflict among ranchers, recreationists, environmentalists, and agency bureaucrats. There were strong political economy arguments against the fee increase. The political economy argument to maintain a uniformly low public grazing fee emphasized the importance of ensuring positive profit on low-productivity lands (LaFrance and Watts, 1995). If the proposed fee structure had been approved, it would have increased the grazing fee to $4.28 per AUM as opposed to the $1.86 per AUM charged in 1993 (Nicoll, 2006). Presently, the grazing fee remains at $1.35 per AUM, the lowest attainable value, because beef cattle prices are declining and production costs are rising (USDI BLM, 2008).

Utah Livestock Production and Revenue

Livestock production has continually played an important function in the economic development of rural communities in Utah. The cattle industry has become the dominant sector in Utah agriculture (Godfrey, 2008). The 2008 livestock inventory in Utah consisted of 850,000 cattle and calves (365,000 were beef cattle) and 275,000 sheep. In 2008, the state of Utah ranked 36th in the United States in terms of cattle and calf production; ranked 28th in terms of beef cow production; and ranked 7th in terms of sheep production. Record high beef cow production occurred in 1983 with an inventory of 374,000, whereas record low beef cow production occurred in 1939 with an inventory of 107,000. Record high lamb production occurred in 1930 with an inventory of 1,736,000, whereas record low lamb production occurred in 2007 with an inventory of 235,000 (Utah Agricultural Statistics Service, 2008).

Total cash receipts from livestock and livestock products for 2007 was estimated at 950.8 million dollars, with Beaver County (146.1 million dollars), Sanpete County (125.5 million dollars), Sevier County (102.4 million dollars), Cache County (99.8 million dollars), and Millard County (98.0 million dollars) yielding the highest cash receipts (Utah Agricultural Statistics Service, 2008). Hog production was the primary source of agricultural revenue in Beaver County and turkey production was the primary source of agricultural revenue in Sanpete County. Dairy production was important to the agricultural economies of Cache, Box Elder, Utah, Millard, and Sanpete Counties (Godfrey, 2008). Cash receipts from cattle and calf production for 2007 were estimated at 283.32 million dollars, while cash receipts from sheep production were estimated at 18.3 million dollars. According to the 2007 beef cattle inventory, beef cattle production was highest in Box Elder County (40,000 head), Duchesne County (26,500 head), Sanpete County (25,000 head), Millard County (24,500 head), and Utah County (19,500 head). These five counties combined contributed to 40 percent of the beef cattle production in Utah during 2007 (Utah Agricultural Statistics Service, 2008).

Threats to the Livestock Industry

Livestock grazing in the western United States confronts a highly uncertain future. Current challenges to western ranching include declining profitability, rapid urbanization, escalating concern over endangered species, extensive invasion by non-native plants, and growing environmental opposition to public land grazing (Holechek, 2001). Declining profitability of cattle ranching on both public and private lands, accompanied with increasing production costs, appears to represent the greatest threat to livestock grazing (Holechek and Hawkes, 1993).

Ranchers function under a commodity pricing structure where profitability is variable. Profits are attainable in periods of high prices and demand, but profits are difficult to capture when prices and demand are low. Under commodity pricing systems, prices for commodities, such as cattle and sheep, are cyclical in nature because the level of production eventually overwhelms the level of demand (Field, 2002). Since the formation of the livestock industry in the late 1860s, cattle prices have followed the cyclical nature of commodity pricing systems. There have been four general inflation-deflation cycles. Each cycle is linked with a period of economic inflation caused by a major sociopolitical event, such as war (Holechek et al., 1994).

The first major economic phase occurred during and after the Civil War (1861-1865) and lasted until the depression of 1873. The next major economic expansion was brought about by the technological advances at the turn of the century and World War I. The period from 1914 to 1920 was one of the most favorable for ranchers and farmers in the history of the country, but this inflationary period abruptly ended with the onset of depression in 1930. At the bottom of the depression in 1933, cattle prices had declined 35 percent from the 1929 levels and over 50 percent from the 1920 levels. World War II (1941-1945) brought economic recovery and sustained high cattle prices until the peak of the Korean War in 1951. The inflationary period in the 1970s was influenced by oil shortages, but cattle prices descended from 1979 to 1986 (Holechek et al., 1994).

Since 1996, cattle demand has been growing as a result of increasing human population with rising affluence, increased per capita consumption of beef, and reduced beef production in Europe due to mad cow disease. From 1992 to 2006, domestic beef consumption in the United States grew by 14 percent and beef exports from the United States increased by 85 percent (Holechek and Hawkes, 2007). However, profitability continues to decline, particularly in arid public land ranches, as a result of rising production costs, such as insurance, electricity, taxes, livestock health care, transportation, supplemental feeding costs, and state and private lease grazing fees. Failure of cattle prices to keep up with production cost increases is the primary reason for the decline in profitability. Extended drought in the arid Intermountain West has also been an important factor depressing ranch profitability. It has been postulated that peak global oil production could adversely affect agriculture profitability in the United States (Holechek, 2001; Holechek and Hawkes, 2007; Workman and Evans, 1993).

Agricultural economists have known for decades that livestock production in the western United States does not offer competitive profit or return on investment (Gosnell and Travis, 2005). Traditionally, the rates of return on ranch investments in Utah have been lower than rates of many other investment opportunities (Workman and Evans, 1993; Bartlett et al., 2002). The average western livestock producer has historically yielded about two percent annual return on capital investment (Field, 2002; Holechek et al., 1994).

The lack of profitability has undoubtedly been a critical factor in the decline in the amount of grazing use authorized on public land and the conversion of private western ranches into other uses (Holechek and Hawkes, 2007; USDI BLM, 2008). Authorized grazing use on public land has declined from about 22 million AUMs in 1941 to 12.5 million AUMs authorized in 2008 (USDI BLM, 2008). Estimates indicate that three to four million acres of private rangeland in the United States have been converted to ranchettes and suburban developments (Holechek, 2001; Resnik et al., 2006). Ranchers increasingly view selling land to developers as a viable or even inevitable option to intergenerational inheritance (Gosnell and Travis, 2005). Additionally, improved communication and technology, rising affluence, and demographic shifts have increased the demand for residential real estate and elevated the value of private rangeland property in the Intermountain West (Holechek, 2001). Consequently, ownership of private ranches is marked by a transition from traditional ranchers to non-ranchers who are most interested in the amenities of owning a ranch. This new pattern of land ownership will result in significant changes in private ranch management and the role ranches play in the social, economic, and ecological aspects of western rangelands (Gosnell and Travis, 2005).

Opportunities for the Livestock Industry

In response to decreasing profitability of livestock production, ranchers and private landowners are creating cost-effective traditional ranches by employing more efficient and sustainable management practices and/or by diversifying into other natural resource industries (Bernardo et al., 1994; Yarbrough et al., 2006). Sustainable management strategies can include rangeland improvements, such as controlling brush or invasive plants on degraded sites, and improving herd management to increase grazing uniformity (Workman and Evans, 1996). Shifting emphasis from capital intensive practices to low risk practices, such as livestock behavioral modification, improved breeding and supplemental feeding programs, and better placement of watering points, can maximize profitability (Holechek and Hawkes, 1993).

Through diversification into other enterprises, private ranchers are finding ways to increase profitability. The majority of private rangeland also has wildlife and recreational value (Clawson, 1950). Therefore, branching into recreation and tourism industries, such as fee hunting, fee fishing, wildlife viewing, nature tours, guest ranching, and dude ranching, can potentially improve revenue (Holechek and Hawkes, 1993; Holechek et al., 1994; Butler, 2002). Niche-marketing of specialty products, such as organic and natural beef and buffalo, landscaping plants, and rodeo stock breeds, has proven to increase profitability (Holechek et al., 1994; Holechek, 2001). Research has indicated that when rangelands are managed for multiple uses, the ecological, sociological, and economic benefits are amplified (Anderson and McCuistion, 2008; Bastian et al., 1991).

Another option to increase profitability of the livestock industry entails the development of partnerships. Partnerships can capture economies of scale that are similar to large ranches (Field, 2002). Generally, profitability has been higher on large ranches as compared to small ranches due to the economies of scale (Holechek and Hawkes, 1993). Many benefits arise from capturing economies of scale, including lower costs, greater access to new technology, the ability to distribute fixed costs over higher levels of production, greater access to markets, and opportunities to secure higher prices (Field, 2002; Yarbrough et al., 2006). For instance, in the beef cattle industry, ranchers with herds larger than 500 head typically have about one-half of the production costs compared to small herds with fewer than 50 head (Field, 2002). Partnerships, as opposed to large-scale consolidations, also have the benefit of sustaining the integrity of rural communities (Yarbrough et al., 2006).

ANIMAL BIODIVERSITY

Livestock grazing is a widespread land use in western North America with approximately seventy percent of the lands in the western United States being utilized by the livestock industry over the course of a year (Fleischner, 1994). Since the rise of the livestock industry, the detrimental effects of poor range management have been documented in detail with much emphasis on the loss of animal biodiversity and the decline in population densities presumed to be due to competition for resources and disruption to habitat (Vavra, 2005; Fleischner, 1994; Chaikina and Ruckstuhl, 2006). However, research in past decades has emphasized that livestock and wildlife can be compatible on the same range and that the biodiversity can be maintained, provided that the management is coordinated with the objectives of the area and the ecology and physiology of the rangeland resources (Anderson and Scherzinger, 1975; Vavra, 2005; Anderson and McCuistion, 2008).

The initial research evaluating compatibilities of livestock and wildlife was focused on big game and wild ungulate species, such as deer and elk (Anderson and Scherzinger, 1975). However, recent research has induced a gradual shift of concern from competition of livestock with big game species to a concern for all wildlife and biodiversity (Bleich et al., 2005; Vavra, 2005). The recognition of impacts to and value of wild ungulate, upland bird, riparian, and threatened and endangered species has provided a foundation for designing comprehensive range management plans. Consequently, many range management plans are being revised to account for the requirements of domestic livestock and multiple wildlife species (Anderson and McCuistion, 2008; Beck and Mitchell, 2000; Bleich et al., 2005; Bock et al., 1993; Fitch and Adams, 1998).

Livestock Interactions with Wild Ungulates

Early research indicated that an overabundance of livestock influences wild ungulate species by causing competition for food resources. Although cattle and wild ungulates often focus on different types of vegetation, diet overlap increases when forage becomes less available in the winter and early spring (Chaikina and Ruckstuhl, 2006; Bastian et al., 1991). Heavy livestock grazing also affects wild ungulate habitat by altering plant biomass, species composition, and structural components, such as vegetation height and cover. Additionally, the physical presence of cattle can cause behavioral changes that make foraging less productive. The combined result of resource competition, modification in rangeland structure, and the presence of livestock can contribute to reduced fat content, reproductive rates, and survival in many wild ungulate species (Chaikina and Ruckstuhl, 2006; Bleich et al., 2005).

Management Strategies

Even though there are several cases that demonstrate the negative impacts of heavy livestock grazing on wild ungulates, there are a considerable number of examples that reveal compatibility between livestock and wild ungulate species (Anderson and Scherzinger, 1975; Chaikina and Ruckstuhl, 2006). In fact, properly managed and specialized livestock grazing systems can maintain or improve habitat for wildlife (Vavra et al., 2007; Bleich et al., 2005). In various ecosystems, grazing is an important ecological process that can increase the chances of survival of some species and enhance community and landscape diversity (West, 1993; Bock et al., 1993).

Seminal research conducted by Anderson and Scherzinger (1975) suggested that specialized livestock grazing systems are capable of manipulating the physiology of forage plants to increase the amount and nutritional quality of winter vegetation for elk. Subsequent research has indicated that moderate amounts of livestock in a deferred or rest-rotational system can improve forage production for deer by increasing forb production through reduced competition from grass. Cattle can create conditions that are beneficial to elk by promoting growth of more nutritious forage plants through the removal of the residual unpalatable vegetation from previous years (Anderson and McCuistion, 2008; McCarthy, 2003). Moderate levels of livestock grazing during the fall have the potential to increase grass and total biomass availability the following spring and allow elk and deer easier access to succulent and nutritious vegetation in the summer (Taylor et al., 2004).

Livestock Interactions with Upland Birds

The research evaluating the effects and compatibilities of livestock grazing on upland birds and migratory land birds is more recent and much more divisive. Species of concern, including sage grouse, have become the center of debate in regard to grazing on western rangelands (Vavra et al., 2007; Bock et al., 1993). Livestock grazing has been identified as one of the factors associated with the widespread decline of sage grouse through the deterioration, loss, and fragmentation of critical sagebrush-grass habitat (Beck and Mitchell, 2000). During the 1970s, large expanses of sagebrush were burned or treated and seeded to increase grass forage for livestock. Evidence suggests that the treatments reduced vital sage grouse habitat. Additionally, research has indicated that heavy grazing can cause the grassy understory in sagebrush steppe ecosystems to be over-utilized, allowing sagebrush species to thicken and eliminating the herbaceous understory required for sage grouse brooding (Anderson and McCuistion, 2008; Bleich et al., 2005). Recent research also indicates that sage grouse brood rearing habitat dominated by dense stands of decadent sagebrush can be manipulated to increase herbaceous components through strategic sheep grazing (Banner, 2008).

Management Strategies

Although sage grouse habitat management has become a contentious issue, research has demonstrated that properly designed and managed grazing plans can maintain and improve sage grouse habitat (Anderson and McCuistion, 2008). Populations of upland birds, including sage grouse, benefit most from a deferred grazing system. The deferment of grazing until mid to late summer, fall, or winter can maintain or enhance sage grouse habitat by increasing the abundance and quality of forb and grass species (Holechek et al., 1982; Anderson and McCuistion, 2008). Forbs are an important dietary constituent for sage grouse; therefore, deferred or intermittent grazing systems should provide improved forage (Vavra, 2005). Deferred grazing systems also reduce disturbance by livestock in the spring during critical periods of sage grouse nesting and brood rearing (Holechek et al., 1982; Anderson and McCuistion, 2008).

Livestock Interactions with Riparian Wildlife

Research examining the consequences of poorly managed livestock grazing on riparian wildlife is comparatively extensive because riparian ecosystems are considered sensitive centers of terrestrial and aquatic biodiversity in arid and semi-arid environments of the western United States (Dobkin et al., 1998; Fleischner, 1994; Bock et al., 1993). Riparian ecosystems are important for wildlife because they provide essential resources, such as breeding, wintering, and migration habitat, that are scarce or absent in the surrounding lands (Hayward et al., 1997). The impact of livestock grazing on wildlife is elevated in riparian zones because cattle are attracted to the water, shade, succulent vegetation, and flatter terrain (Bock et al., 1993; Hayward et al., 1997). The direct impacts of unmanaged continuous livestock grazing in riparian zones include (1) the modification and elimination of streamside vegetation, which changes stream channel and bank structure, (2) the compaction of soil, which increases runoff and decreases water availability to plants and animals, and (3) the alteration of water temperature and chemistry (Fleischner, 1994; Bleich et al., 2005).

Management Strategies

The adverse effects of livestock grazing on riparian wildlife have prompted the design of management approaches that enhance or restore the conditions of these ecosystems (Popolizio et al., 1994). Sustainable range management plans are incorporating strategies that maintain or improve the productivity of vegetation and the integrity of the waterway structure (Bleich et al., 2005). Research suggests that grazing systems that provide rest and deferment can offset the impact of livestock grazing, enhance plant productivity, improve wildlife habitat, stabilize water channels, and improve water quality (Fitch and Adams, 1998).

Some studies indicate that light to moderate late-fall or winter grazing may have relatively little impact on riparian ecosystems because water levels are low, stream banks are dry, and vegetation is dormant; however, grazing during this time of year should be monitored because excessive grazing can remove the protective plant cover that is necessary during the following spring stream-flow periods (Bock et al., 1993). Other research indicates that light to moderate grazing during the spring or early summer may have less of an impact on riparian systems because it can deter livestock away from the wet riparian environment and encourage foraging on the succulent upland vegetation (McCarthy, 2003). This system of grazing also permits recovery of vegetation for use by upland birds during later growing seasons (Anderson and McCuistion, 2008). The restoration of riparian ecosystems has also been accomplished by the placement of temporary or permanent fences along riparian corridors. Although fencing is costly and can be an obstacle to livestock and wildlife, it provides the most rapid recovery of riparian vegetation (Bleich et al., 2005; Fitch and Adams, 1998).

PLANT BIODIVERSITY

Livestock can exert a considerable change on the diversity, composition, structure, and development of native plant communities (Popolizio et al., 1994; Vavra et al., 2007; Orodho et al., 1990). However, the degree of change is highly dependent upon the ecosystem and plant community, the current environmental conditions, and the intensity and timing of grazing (Bock et al., 1993; Milchunas, 2006). Much of the literature indicates that the change has been more drastic and evident in ecosystems where native grazing ungulates were historically scarce or absent (Bock et al., 1993; Hayward et al., 1997; Milchunas, 2006).

The shortgrass steppe ecosystem within the Great Plains is among the most grazing tolerant plant communities in the world because herbivory by native ungulates has played an important role in the ecological and evolutionary history (Milchunas, 2006; Bock et al., 1993). For this reason, the impacts to native plant communities from excessive grazing have been three times less than that of other vegetation communities throughout the world. In contrast, the effects of inappropriate grazing practices and poor livestock management have been more substantial in the Intermountain West because many of the vegetation communities did not evolve with large ungulate species (Bock et al., 1993; Knapp, 1996). Research evaluating the impacts is variable, but the predominant effects include changes in species composition, reductions in individual plant density and species diversity, and modifications in plant succession (Fleischner, 1994).

Livestock Influences on Sagebrush Steppe Vegetation

The vegetation communities within sagebrush steppe environments have been dramatically influenced by poor livestock management practices. In many communities, native grass species have been reduced or eliminated, forb cover has decreased, shrub cover has increased, and non-native grasses and forbs have proliferated (Bock et al., 1993). As mentioned in the previous section, large expanses of sagebrush were burned or treated to increase grass forage for livestock. The burned and treated areas were frequently seeded with crested wheatgrass to supplement the forage production. Crested wheatgrass was often exclusively planted to create a monoculture; consequently, plant community structure and diversity changed (Beck and Mitchell, 2000).

The presence of livestock in sagebrush steppe environments has played a role in directly and indirectly modifying plant communities (Urness, 1976). The direct impacts of poorly managed grazing systems are a result of livestock utilizing and trampling plants that are susceptible to physical damage (West, 1993). Livestock can indirectly modify plant community diversity and structure through selective grazing. The more palatable herbaceous species are consumed, and competition with woody vegetation is reduced. Consequently, the propagation of shrubs, such as sagebrush, and the invasion of less palatable, non-native, and poisonous plants is promoted (Beck and Mitchell, 2000; Alexander, 1987). Research has suggested that excessive grazing pressure from livestock during the early 20th Century may be a causative factor in sagebrush dominance (Miller et al., 1994; Stoddart et al., 1975). Unrestricted, selective grazing by livestock, especially during spring months, has contributed to widespread change favoring shrubs that were frequently subordinates prior to European settlement (Dasmann, 1964; Ellison et al., 1951; Hull and Hull, 1974; Leopold, 1950; McConnell and Dalke, 1960; Miller et al., 1994).

Management Strategies

Land managers are implementing grazing strategies that mitigate the effects of excessive livestock grazing by modifying the timing and duration of use. The timing of herbivory can have a significant impact on plant productivity and vigor, especially if livestock are repeatedly present during plant growth and reproductive stages (Vavra et al., 2007). The duration of grazing should be brief to permit photosynthesis and plant recovery. If grazing is properly managed during these critical periods, plants are permitted to build their root systems and increase nutrient storage. Subsequently, plants become more robust, the likelihood of survival increases, and the overall forage production increases (McCarthy, 2003).

The season of use contributes to varying degrees of change in sagebrush steppe vegetation communities. However, regardless of the season of grazing, livestock should be managed to allow optimum growth of forbs, grasses, and sagebrush (Beck and Mitchell, 2000), and the amount of forage removed is not nearly as important as the amount of residue that remains (Holechek et al., 1982; Caldwell, 1984; Briske and Richards, 1995; Reed et al., 1999). If vegetation is actively growing when livestock are present, residual leaf area is required for plants to carry on photosynthesis. Intensive season-long grazing is considered to have the most significant effects on vegetation. Typically, woody vegetation increases, the spread of invasive plant species is promoted, and the vigor and production of herbaceous species is reduced (Beck and Mitchell, 2000). Repetitive spring grazing can also be detrimental to vegetation and may promote the spread of sagebrush and juniper into adjacent communities (Milchunas, 2006). Small-scale experiments have demonstrated that perpetual spring grazing results in decreased plant cover and higher proportion of introduced and annual plants (Seefeldt and McCoy, 2003).

In contrast, moderate levels of fall grazing can increase grass species and total biomass availability the following spring (Taylor et al., 2004). Perennial grasses and forb cover increase and shrub cover decreases (Beck and Mitchell, 2000; Milchunas, 2006). Localized experiments have revealed that fall grazing results in the highest proportion of native perennial grasses and lowest proportion of native annual forbs (Seefeldt and McCoy, 2003). It has also been suggested that the implementation of short-duration grazing seasons in the fall may be an economical way to enhance the diversity of sagebrush steppe vegetation communities. Livestock can aid in minimizing the re-invasion of sagebrush in previously treated areas and can promote structural diversity of herbaceous species within sagebrush stands (Provenza et al., 2003; Vavra et al., 2007).

Livestock Influences on Riparian Vegetation

The impacts of livestock grazing on riparian vegetation communities are well documented and particularly pronounced in arid and semi-arid environments in the Intermountain West (Bock et al., 1993). Grazing practices, prior to the passage of the Taylor Grazing Act in 1934, notably impaired riparian vegetation communities and systems. Although significant effects occurred prior to moderate control over livestock grazing on public lands, recent research indicates that poorly managed livestock remains a key factor in the continued degradation of riparian ecosystems (Belsky et al., 1999). Improper livestock management, which permits long periods of grazing and trampling, can affect riparian vegetation communities through the reduction and elimination of plant cover. The removal of herbaceous plants can affect species composition, species diversity, and production; and the removal of woody vegetation can impact foliage cover, structural height, and stand reproduction (Kauffman and Krueger, 1984). Grazing pressures on woody vegetation can also prevent the establishment of seedlings, resulting in an even-aged nonreproducing vegetation community. Indirect effects of vegetation removal include changes in streambank and channel structure via increased runoff and erosion and alterations in water temperature and chemistry (Fleischner, 1994).

Management Strategies

Livestock grazing can be compatible with riparian systems, provided that the maintenance of ecological functions are included as management objectives and the integrity of the riparian system is kept intact (Lucas et al., 2004). Research has indicated that rest-rotation grazing schemes and specialized grazing schemes in which riparian ecosystems are treated as special use pastures have been the most successful (Kauffman and Krueger, 1984). In addition, as in other vegetation communities, many adverse effects of livestock grazing can be alleviated by manipulating the timing, intensity, and the duration of grazing (Clary and Webster, 1989; Elmore and Kauffman, 1994).

Year-long and summer grazing can inflict the most damage to riparian vegetation, whereas short-term spring, fall, or winter grazing can have less of an impact (Bock et al., 1993). Short-term spring grazing can have several advantages. In many environments, it may be preferred because it can maintain species diversity and streambank structure (Clary and Webster, 1989). Spring grazing has the ability to preserve species richness and diversity because livestock can control the spread of aggressive plant species (Lucas et al., 2004). Species diversity and streambank structure are upheld with spring grazing because it encourages a more even distribution of livestock use between riparian and upland areas. Short-term early-spring grazing also allows riparian plant growth to occur before the dormant period in the fall, given that the livestock are removed before critical growth periods. Therefore, spring grazing has the potential to maintain the vegetative cover that is vital for streambank protection during the following winter and throughout the early spring high streamflow periods (Clary and Webster, 1989; Bock et al., 1993).

Late-fall and winter livestock grazing is an alternative to spring grazing in most areas, but it may not always be the preferred season of use unless utilization levels are carefully monitored (Clary and Webster, 1989). Research has indicated that the impact of fall grazing is variable, with some plant communities being affected and others displaying no discernable effect (Kauffman et al., 1983). Some studies suggest that fall and winter grazing can maintain plant vigor and production because vegetation is dormant (Clary and Webster, 1989). However, this is favorable providing that livestock are managed in such a way that the essential protective plant cover is left for the following winter and spring (Bock et al., 1993). Research has revealed that at the end of a fall grazing season, the residual plant cover is adequate to retain plant vigor and streambank structure (Clary and Webster, 1989).

Another management technique that has been used to restore or prevent further adverse effects to riparian vegetation is through the use of livestock exclosures. Studies suggest that livestock removal can be effective for initiating rapid recovery of riparian willow canopy cover (Holland et al., 2005) and can promote increases in grasses, foliar cover, shrub density and height, and plant diversity (Kauffman et al. 1983; Popolizio et al., 1994). Although exclosures have proved to be beneficial for many riparian vegetation communities, it has been suggested that long-term exclusion may lead to a closed canopy and reduced species diversity because some riparian tree species, such as willows and cottonwoods, may require a moderate amount of disturbance for growth (Lucas et al., 2004; Holland et al., 2005). Therefore, reduced stocking rates after short-term exclusion may be preferred as long as the vegetation community has sufficient canopy cover, adequate soil, and functional riparian processes (Holland et al., 2005). Additionally, moderate grazing of riparian areas may enhance watershed functioning by preventing decadent or stagnant plant communities (Huber et al., 1995).

INVASIVE PLANTS

An increasing threat to rangeland biodiversity and health is the invasion by non-native plant species (Frost and Launchbaugh 2003; Society for Range Management, n.d.). Some of the most prevalent and problematic invasive plants include diffuse knapweed, spotted knapweed, yellow starthistle, leafy spurge, and cheatgrass (DiTomaso, 2000). The vast majority of invasive plants have been introduced from other continents. Cheatgrass, the most widespread and dominant invasive plant in the Intermountain West, was introduced during the mid- to late-1800s by means of imported grain from Eurasia (DiTomaso 2000; Knapp, 1996). The first records of cheatgrass in the Great Basin came from Provo, Utah in 1894; Elko, Nevada in 1905; and Reno, Nevada in 1906 (Knapp, 1996).

The dispersion of non-native plants was originally linked to direct human activity, particularly along railroad lines (Knapp, 1996). However, decades of overgrazing in the Intermountain West during the open range era and poor grazing management practices have facilitated the invasion, establishment, and spread of non-native plant species (Frost and Launchbaugh, 2003; Vavra et al., 2007). Prior to the introduction of non-native plants, Intermountain rangelands were predominantly characterized by perennial bunchgrasses, forbs, and shrubs (Hull and Hull, 1974). However, the proportion of non-native plant species began to increase as the livestock industry expanded and human populations began to flourish. Poorly managed grazing destabilized many native plant communities and encouraged the spread of non-native plants because native perennial grasses do not have high seedling vigor and some do not readily recover from grazing (DiTomaso, 2000). In contrast, invasive winter annual grasses, such as cheatgrass and medusahead, have high seedling vigor, and they outcompete native plants by exploiting valuable resources and completing their life cycle prior to the summer dry period (Frost and Launchbaugh, 2003). The reduced competition from native plants perpetually favors the spread of invasive plants because many are unpalatable, aversive, or toxic to livestock (DiTomaso, 2000).

Livestock can also promote the spread of non-native plants through ground disturbance and the physical dissemination of seeds. Disturbance appears to be an important aspect in the establishment of non-native plant populations because many invasive plants are adapted to soil disturbance, such as that caused from trampling (Vavra et al., 2007). Therefore, high intensities of livestock have been suggested to increase invasibility (Loeser et al., 2001). Livestock can disperse seeds by serving as transportation vectors. Several invasive plant seeds, such as cheatgrass and houndstongue, are dispersed by adhering to the coats of animals; others are dispersed as they pass through digestive tracts (Frost and Launchbaugh, 2003; Fleischner, 1994).

Impacts of Invasive Plants

Invasive plants can have a significant impact on an array of ecological facets. Invasive plants have reduced species richness, plant diversity, and community productivity. Wildlife habitat and forage have been degraded; soil erosion and stream sedimentation has increased; soil moisture and nutrient levels have been depleted; and fire regimes have been altered (Frost and Launchbaugh, 2003; Wallace et al., 2008).

As cheatgrass has become a common component of sagebrush steppe vegetation communities, the nutritional quality of forage has been reduced, the intensity and frequency of fires have changed, and water cycles have been altered. Although many factors are involved, several native animals, such as sage grouse, may have declined as a result of these changes (Society for Range Management, n.d.). Invasive broadleaf species that have deep taproot systems, such as yellow starthistle, have modified surface runoff, stream sediment yields, soil moisture, and soil nutrients (DiTomaso, 2000). Yellow starthistle can extract soil moisture from the entire soil profile and outcompete native shallow- and deep-rooted annual and perennial species (Wallace et al., 2008). Woody plant species, such as salt cedar, have invaded wetland and riparian systems throughout the western United States. Dense populations of salt cedar lower water tables, reduce surface water, alter flood frequency, and reduce the diversity and productivity of the herbaceous understory (Masters and Sheley, 2001). These ecological changes combined suggest that invasive plants can significantly alter ecosystem processes, cause ecosystem instability, displace native plant species that are vital to wildlife and livestock, and reduce the capacity for ecosystems to provide the services required by society (Knapp, 1996; Masters and Sheley, 2001).

The invasion of non-native plant species not only produces various ecological modifications, but also results in substantial socioeconomic impacts, particularly to the livestock industry and land management agencies responsible for fire suppression. Invasive plant species cause more economic loss on rangeland than all other pests combined. Invasive plants reduce the carrying capacity for livestock by lowering the forage yield. Consequently, the costs of managing and producing livestock increase (DiTomaso, 2000).

Research has demonstrated that leafy spurge and knapweed species can reduce grazing capacity by more than half. However, some rangelands have deteriorated to the point that desirable species are either not present or in such low abundances that plant community recovery is slow or will not occur without revegetation efforts (Masters and Sheley, 2001). Although cheatgrass is used to some degree as livestock forage, in some years it only provides ten percent of the productivity of the perennial species it replaced. Cheatgrass can be a nutritious and palatable forage crop during the growing season, but it is often an unreliable source because of its dependency on annual precipitation, and awned cheatgrass seed can pose severe health problems to livestock after it has matured (Knapp, 1996).

Invasive plant species, specifically cheatgrass, have altered the fire regimes of many environments in the western United States, and consequently imposed an economic burden on management agencies faced with fire suppression. Prior to the invasion of cheatgrass in sagebrush steppe ecosystems, the fire return interval was approximately 60 to 110 years; however, cheatgrass has changed the fire frequency to 3 to 5 years (Pimentel et al., 2005). Cheatgrass fires are common because the amount of fine fuel that accumulates is greater than what occurs in sagebrush-bunchgrass ecosystems (Knapp, 2005). The increased fire frequency does not permit establishment by native annuals and perennials, and therefore, native plants are diminishing and monocultures of cheatgrass are dominating (Knapp, 2005; Pimentel et al., 2005). The cost of wildfire suppression on public land is rising with the federal fire bureaucracy spending hundreds of millions of dollars annually on resource losses, suppression costs, pre-suppression costs, fire management, and rehabilitation programs (Dombeck et al., 2004; Knapp, 1996).

Invasive Plant Control Techniques

Attempts to manage and eradicate invasive plant species have been made utilizing various control methods. Historically, mechanical and chemical control techniques were the predominant invasive plant management methods; however, biological and cultural control techniques have been implemented and integrated with other practices.

Mechanical control techniques include hand-pulling, hoeing, mowing, tilling, chaining, and bulldozing. Hand-pulling and hoeing are effective in controlling small infestations of shallow-rooted weeds in loose, moist soils (DiTomaso, 2000). Mowing is commonly used to control invasive range annuals and some perennials; however, the success of mowing is highly dependent on timing. Annuals and some perennials can be suppressed and controlled if mowing occurs before viable seeds form. If not properly timed, mowing can promote the spread of invasive plants by encouraging the spread of seeds and stimulating the production of new stems from vegetative buds below the cut surface (DiTomaso, 2000; Masters and Sheley, 2001). Tilling practices can control annual species, but rarely provide control of perennial species. In fact, perennial or biennial species, such as spotted knapweed and perennial pepperweed, often spread as a result of tilling (DiTomaso, 2000). More expensive mechanical control techniques, such as chaining and bulldozing, are effective in controlling invasive shrub and tree species. Although these methods require gentler terrain and are becoming increasingly expensive, they are effective in controlling shrubs and trees that do not readily resprout from root systems (DiTomaso, 2000; Masters and Sheley, 2001).

Chemical control techniques include the application of herbicides, such as 2,4-D, glyphosate, picloram, and teburthiuron. Herbicides are the primary method of invasive plant control in most rangeland systems (DiTomaso, 2000; Masters and Sheley, 2001). However, most herbicides do not provide adequate control without several successive annual applications (Knipe, 1983) and they seldom provide long-term control (DiTomaso, 2000). Timing of herbicide application is also essential to effective control because it is highly dependent on the species and the herbicide being applied. Additionally, herbicides that are effective in controlling invasive plants are often toxic to native herbaceous plants and have the potential to contaminate surface and ground water (DiTomaso, 2000; Masters and Sheley, 2001).

Biological control includes the planned use of living organisms to reduce the reproductive capacity, density, and effect of invasive plant species (Masters and Sheley, 2001). The primary goal of biological control techniques is to exert environmental stress on invasive plants by reestablishing interactions with natural enemies (DiTomaso, 2000; Masters and Sheley, 2001). Although there have been many attempts to control invasive plants on rangelands, the success has been variable and limited. Biological control has been moderately effective in controlling leafy spurge and salt cedar. However, important factors that have contributed to the limited success of biological control are often attributed to a high level of genetic diversity in the target species and opportunistic predation and parasitism by the biological control insect or agent (Masters and Sheley, 2001).

Cultural control techniques include prescribed burning, reseeding or revegetation efforts, the modification of grazing management plans, and the implementation of prescription or targeted grazing (Masters and Sheley, 2001). Prescribed burning is often used for long-term suppression of woody species in sagebrush and juniper ecosystems and can stimulate native annual and perennial grass growth (DiTomaso, 2000). Seeding and other revegetation efforts are often alternatives for managing invasive plants in areas that lack desirable species. Revegetation with competitive grasses and forbs may suppress non-native plants, enhance plant community resistance to further invasion, and improve forage production and quality (Masters and Sheley, 2001).

Recent cultural control techniques have focused on the modification of grazing management plans and the implementation of prescription grazing. Properly managed livestock can minimize the spread of invasive plants on rangelands (Wallace et al., 2008; DiTomaso, 2000). Moderate grazing levels can minimize the impact to native plants; intensive grazing can counteract the dietary preferences of cattle, resulting in equal impacts to all forage species including invasive plants; and multispecies grazing can distribute the impact of livestock more uniformly among desirable and undesirable species (DiTomaso, 2000). Adjusting the timing of grazing to coincide with the susceptible life-cycle phases of invasive plants can also have substantial control impacts (DiTomaso, 2000).

Targeted or prescription grazing is the application of livestock grazing at a specified season, intensity, and frequency to achieve specific vegetation management goals, such as the control of invasive plants (Wallace et al., 2008). Successful prescription grazing should cause significant damage to the target plant, limit damage to native vegetation, be consistent with livestock production goals, and be integrated with other control methods. Prescription grazing also entails the modification of livestock grazing behavior (Frost and Launchbaugh, 2003). The species of livestock suited for control of invasive plants depends on the species of concern and the production setting. Research has evaluated the effectiveness of cattle, sheep, and goats in targeted grazing.

Cattle have large rumens that are well adapted to ferment fibrous material (Frost and Launchbaugh, 2003). Although cattle can manage fibrous herbaceous vegetation, such as dormant grasses, they appear to offer the least potential for control of invasive plants. However, experiments have been conducted in the early spring to determine if cattle can inflict physical damage to leafy spurge (Brock, 1988). More recent research has suggested that supplementing the diet of cattle with protein may enhance the tolerance of cattle for invasive plants high in chemical compounds (Provenza et al., 2003).

Sheep are considered an excellent species to accomplish control of herbaceous plants. The physical characteristics of sheep allow them to selectively graze and tolerate substantial fiber content (Frost and Launchbaugh, 2003). Sheep have been used to control several invasive rangeland plants, including leafy spurge, tall larkspur, and tansy ragwort (Brock, 1988; Frost and Launchbaugh, 2003). Goats are the most well-known domestic grazer that functions as a plant control agent (Brock, 1988). Goats are classified as browsers, and their physical characteristics allow them to select individual leaves or chew entire branches. Although they can be very selective herbivores, goats are reputed to utilize a wider range of vegetation than other livestock species (Knipe, 1983). They also have a large liver mass relative to cattle or sheep, and can therefore process plants that contain secondary chemical compounds, such as tannins or terpenes. Goats are generally more effective than cattle or sheep in controlling leafy spurge because of its high content of secondary compounds (Frost and Launchbaugh, 2003). Both sheep and goats have proved to provide better control of leafy spurge than using picloram (Sheley et al., 2004).

Integrated Management Strategies

The implementation of one control method is rarely effective in achieving the desired results for curtailing the spread of invasive plants. Successful long-term and cost-effective management programs should integrate a variety of mechanical, chemical, biological, and cultural techniques (DiTomaso, 2000). Integrated management involves the deliberate selection, combination, and implementation of effective invasive plant management strategies with due consideration of economic, ecological, and sociological consequences (Sheley et al., 2004).

Although integrated management emerged as a viable concept in the 1970s, the practice has not been systematically implemented until recently because effective integrated management plans and programs require a thorough understanding of the ecology and biology of invasive plants and the invaded plant community (Masters and Sheley, 2001). Presently, there are several examples of integrated strategies used to manage invasive plants and improve rangeland communities. Much attention has been focused on the integration of targeted or prescription grazing with other control methods, as the incorporation of grazing management is an essential component in successfully addressing invasive plant problems (Frost and Launchbaugh, 2003).

Research has demonstrated that targeted grazing on yellow starthistle by sheep and cattle can enhance the effectiveness of biological control (insect herbivores) by reducing root biomass, limiting seed production, and opening up the plant canopy. These conditions combined can make invasive plants more susceptible to damage (Wallace et al., 2008). Spring application of 2,4-D has been effective in removing adult spotted knapweed plants, but repeated sheep grazing after the chemical control is required to limit seedlings and juvenile plants. If accomplished, the density, cover, and biomass of spotted knapweed decreases, and grasses are allowed to reoccupy the sites (Sheley et al., 2004). Leafy spurge has been controlled using a variety of integrated management techniques, including a sheep grazing/biological control and goat grazing/chemical control (DiTomaso, 2000).

FIRE REGIMES

Prior to the late 1800s, fire played an important role in the health of many ecosystems by recycling nutrients, improving soil productivity, and by maintaining biodiversity, community composition, habitat structure, and watershed condition (DiTomaso, 2000; Dombeck et al., 2004; Miller and Heyerdahl, 2008). However, historic livestock grazing practices and fire suppression policies have modified the frequency, intensity, severity, and seasonality of fire in Intermountain West ecosystems by altering fire fuel loads and arrangements and by promoting the invasion of non-native plants (Bock et al., 1993; Davison, 1996). Shrub steppe and upland forest ecosystems in the Intermountain West have experienced widespread change.

Effects on Shrub Steppe Ecosystems

Fire is a natural and essential component in native sagebrush steppe and semi-desert shrubland ecosystems. The frequency of fires in native vegetation communities is variable and depends on sagebrush, shrub, or woodland species, geographic location, climatic variables, and soil properties. In many semi-desert shrubland communities, the structure, characteristics, and lack of a continuous fuel source do not readily promote the spread of fire; therefore, the fire return interval typically ranges from 60 to 110 years (Pimental et al., 2005). Sagebrush steppe and mountain brush plant communities occurring at higher elevations and latitudes in the Great Basin desert have a shorter mean fire return interval of 30 to 100 years because the shrub cover is denser and the shrub architecture is more flammable (Brooks and Pyke, 2001). Native communities of mountain big sagebrush have a mean fire return interval of 12 to 25 years (Miller and Tausch, 2001).

In many sagebrush steppe and semi-desert shrubland ecosystems, the behavior and characteristics of fire have been modified as a result of fuel reductions by livestock grazing and human-induced fire suppression (Brooks and Pyke, 2001). Large concentrations of livestock have significantly reduced the cover of native grasses and forbs, and consequently facilitated the establishment of invasive plant species. Invasive plants, such as cheatgrass, have increased the fire return interval and inhibited the germination and propagation of native annuals and perennials (Pimentel et al., 2005). Cheatgrass provides a dense and continuous fuel source that extends the seasonality and increases the frequency of fires (USGS, 2002). Consequently, it often converts arid low-elevation sagebrush-bunchgrass communities into annual dominated grasslands (Davison, 1996). The change in natural fire regime and conversion to non-native annual grasses has had inadvertent impacts on wildlife species (USGS, 2002).

Historic overgrazing and fire suppression has also encouraged the expansion of woodlands into areas previously occupied by sagebrush and semi-desert shrubs. Historically, fire played an integral role in maintaining sagebrush-steppe communities by limiting conifer encroachment, but the rapid increase in domestic livestock and reductions in fire frequency created ideal conditions for the establishment of woodland seedlings, such as juniper and pinyon pine (Bock et al., 1993; Madany and West, 1983). Woodland species began increasing into low and mountain big sagebrush communities during the late-1800s when grazing by livestock reduced the fine fuels required for low-intensity fires and decreased the competition provided by native herbaceous species (Miller and Rose, 1999). The extent of mountain big sagebrush has been significantly reduced by recent woodland expansion because the fire return interval has increased to greater than 100 years in some regions (Miller and Tausch, 2001).

Effects on Upland Forest Ecosystems

Historically, fire influenced the structure, composition, and dynamics of semi-arid western, interior forests (Zimmerman and Neuenschwander, 1984; Belsky and Blumenthal, 1997). Western forest tree densities, particularly in juniper woodlands and ponderosa pine forests, were maintained by two natural phenomena – low-intensity surface fires and competitive exclusion of tree saplings by dense understory grasses (Belsky and Blumenthal, 1997). Modifications in western forest ecology have occurred as a result of postsettlement land-use change and management, heavy grazing by sheep and cattle, reduced return intervals for low-intensity ground fires that served to thin dense stands of younger trees, and favorable climate years for tree reproduction around the turn of the nineteenth century (Borman, 2005; Belsky and Blumenthal, 1997; Miller and Tausch, 2001).

Juniper and pinyon-juniper woodlands have experienced pronounced change in both the distribution and density across the Intermountain West. Prior to Euro-American settlement, juniper and pinyon pine species were primarily confined to rocky ridges or surfaces where sparse vegetation limited fire. Woodlands were characteristically open, sparse, and savannah-like from frequent low-intensity fires (Miller et al., 1995; Madany and West, 1983). However, juniper and pinyon-juniper woodlands throughout the Great Basin began to expand during late 1800s and early 1900s. The expansion coincides with the introduction and increase of livestock, and the subsequent reduction in herbaceous species that served as fine fuel loads. Additionally, between 1850 and 1916, winters became milder and precipitation was greater than the long-term average. The wetter, milder conditions promoted vigorous growth in conifers (Miller and Rose, 1999; Miller and Tausch, 2001). The expansion of juniper and pinyon-juniper woodlands has been most dramatic in areas with deeper well-drained soils; consequently, sagebrush steppe, semi-desert shrubland, grassland, aspen, and riparian plant communities are being invaded and displaced (Miller et al., 1995; Miller and Tausch, 2001).

In Rocky Mountain forests, the most extensive and heavily impacted communities have been those dominated by ponderosa pine and aspen. Historically, ponderosa pine forests were characterized by open stands of trees with a lush herbaceous understory of perennials and varying densities of shrubs. Frequent, non-catastrophic fire was an important determinant in maintaining plant community structure and composition, particularly in dry southwestern ponderosa pine forests (Bock et al., 1993; Schoennagel et al., 2004). The historic mean fire return interval of ponderosa pine forests varied from four to 36 years (Schoennagel et al., 2004); however, research indicates that mean fire return interval in intermediate- and high-elevation ponderosa pine forests was longer, ranging from 30 to greater than 40 years (USDI NPS, n.d.). Aspen, a disturbance-dependent species, has also declined over much of its former range due to fire suppression and conifer encroachment (Bartos and Campbell, 1998).

Although, higher elevation ponderosa pine forests typically have longer fire return intervals and higher intensity fires than ponderosa pine forests in the southwest (USDI NPS, n.d.), livestock grazing and fire suppression policies have promoted widespread stands of dense fire-sensitive and disease-susceptible trees (Belsky and Blumenthal, 1997). Intensive grazing by sheep and cattle was the primary agent in reducing the herbaceous vegetation and modifying vegetation structure (Madany and West, 1983; Touchan et al., 1995). The shift in vegetation structure encouraged the proliferation of trees, reduced flammability, and decreased fire frequency (Madany and West, 1983). Extensive fire prevention efforts from 1930 through 1960 intensified the effects (Borman, 2005). Presently, the dense stands of trees can provide fuels at intermediate heights that can carry fire up into continuous canopy fuels, thus increasing the probability of large, catastrophic, and stand-replacing fires (Schoennagel et al., 2004; Baker and Ehle, 2001).

Management Strategies

Historic grazing and fire suppression efforts have interrupted the natural fire cycle in many intermountain rangeland environments (Weber et al., 2001). The frequency, intensity, severity, and seasonality of fire have been altered. Vegetation and wildlife communities have been modified, rangeland productivity has decreased, fuel loads have reached unprecedented levels, fire-tolerant non-native plants have proliferated, and catastrophic fires have become common (Bock et al., 1993; Davison, 1996; Weber et al., 2001). Consequently, federal and state agencies are beginning to focus on management strategies that reduce fuel build-up and the risk of fire. The two primary methods being implemented and evaluated are prescribed burning and targeted grazing.

Prescribed burning is the controlled application of fire to wildland fuels to attain planned resource management goals (Johnson, 1984). Research conducted in the Rocky Mountains confirmed the widespread use of fire by native people to manipulate and improve vegetation communities (Kay, 2007). When prescribed burning policies are founded on ecological principles, prescribed burning can reduce wildfire risk and severity, control invasive plants, suppress woody species, improve forage and rangeland productivity, and enhance wildlife habitat and native plant communities (Brooks and Pyke, 2001; DiTomaso, 2000; Dombeck et al., 2004; Madany and West, 1983; Yoder et al., 2003). Prescribed burning can mitigate fire severity through the reduction of tree and shrub density and accumulated fuels (Pollet and Omi, 2002; Madany and West, 1983).

Prescribed burning is an ecologically sound way to improve wildlife habitat. Land management plans that integrate prescribed burning can enhance the habitat of game species and plants and/or animals of concern. It can open areas for increased movement, reduce ground litter, control brush encroachment, increase nutritional value, and diversify plant species (Anderson and McCuistion, 2008). Fire removes litter and dead standing herbage of low nutritional value and increases forage production. Consequently, herbivores can more efficiently select nutritious plant material (Bleich et al., 2005, Madany and West, 1983; Anderson and McCuistion, 2008).

Prescribed burning can be used to control invasive plants. However, the decision to use fire as a management tool must evaluate interrelationships between fire and invasive plants because fire may promote rapid recovery of invasive species and/or the establishment of other fire-tolerant invasive plants (Brooks and Pyke, 2001; DiTomaso, 2000). Information on the physiology, anatomy, life history, and seed dispersal and longevity of invasive plants is integral to the decision (Brooks and Pyke, 2001). The timing of prescribed burning is critical for success. In general, prescribed burns should be conducted following seed dispersal by invasive plants and senescence of native grasses and forbs (DiTomaso, 2000). December is the preferred month to avoid damage to native forbs in sagebrush steppe environments (Anderson and McCuistion, 2008). Additionally, rehabilitation work, such as seeding with mixes of native species, is often required after prescribed burns (Beck and Mitchell, 2000). Immediate revegetation with desirable and competitive plant species is a sustainable long-term method for suppressing invasive plants, while providing high forage production on rangeland (DiTomaso, 2000).

Although prescribed burning is gaining favor in many areas, it has some drawbacks. Prescribed burning is an inherently risky resource management tool because there is a threat that the fire may escape and spread, people may be injured, and equipment may be lost. Therefore, prescribed burning can impose unintended costs (Yoder et al., 2003; Johnson, 1984). Also, the smoke and pollution produced by prescribed burns may violate regulations, such as the Clean Air Act, and may impact surrounding communities (Davison, 1996). Because of air quality concerns and the need for correct fire-weather conditions, there is usually a narrow period of time in which prescribed burning can be conducted (Nader et al., 2007).

Alternatively, targeted grazing can achieve many of the desirable outcomes related to fuel reductions without some of the problems inherent in prescribed burning (Davison, 1996). Targeted grazing is the controlled grazing of livestock to accomplish specific vegetation management objectives. Unlike conventional grazing management, livestock are used as a tool for improving land health by performing weed control, reducing wildland fire, and aiding in restoration projects (Launchbaugh and Walker, 2006). Targeted grazing has the potential to be an ecologically and economically sustainable management tool for reduction of fuel loads (Nader et al., 2007). The time of grazing and type of livestock used can be selected to minimize environmental damage and increase cost effectiveness (Davison, 1996).

Research has indicated that livestock can reduce fuel loads by removing and consuming vegetation and by incorporating fine fuels into the soil via trampling (Nader et al., 2007). Invasive annual grasses, such as cheatgrass and medusahead, dominate vast areas in the Great Basin and form dense carpets of flammable material (Taylor, 2006). Livestock can be used to manage invasive annual grasses and the associated fuel loads in the sagebrush-bunchgrass communities to control fire frequency and intensity (Davison, 1996; Weber et al., 2001). Targeted sheep grazing during early seasons has proven to substantially reduce the fuel loads from invasive annual grasses (Taylor, 2006). Additionally, targeted grazing reduces fuel loads in a more selective fashion and avoids the potential sterilizing effects of extremely intense fires (Weber et al., 2001). Targeted grazing has also shown to be effective in creating firebreaks. Firebreaks, strips of land where vegetation has been reduced or removed, can slow or stop the spread of fire. Firebreaks can be created by temporarily confining high densities of livestock to a strip of land (Taylor, 2006).

Both prescribed burning and targeted grazing have demonstrated to be effective in reducing fuel loads and invasive plants; however, one of the best ways to address rangeland problems is to integrate livestock grazing with prescribed fire, chemical, and/or mechanical treatments (Taylor, 2006). When balanced with ecological objectives, proper livestock grazing can be one of the most effective and least expensive methods of fuel management. If coupled with prescription burning, targeted grazing can reduce the occurrence and the impact of catastrophic fires (Brown, 2002).

SOIL HEALTH

Soil is a dynamic resource that supports plants and numerous species of living organisms. Soil has biological, physical, and chemical properties, some of which are modified with livestock grazing management practices (USDA NRCS, 2001). Livestock grazing can have profound effects on soil structure and function.

Most studies have shown that intensive livestock grazing increases soil compaction, reduces water infiltration and holding capacity, decreases soil organic matter, increases soil erosion and loss, and modifies nutrient cycles (Belsky and Blumenthal, 1997; Bock et al., 1993; Fleischner, 1994; Ingram et al., 2008; Kauffman and Krueger, 1984; Milchunas, 2006; Orodho et al., 1999; Stephenson and Veigel, 1987). Disturbances to surface soils can influence arid-land ecosystem productivity and fertility by altering the soil physical and chemical properties (Neff et al., 2005). Livestock grazing can also have a considerable effect on biological soil crusts and the associated functions in cold-desert regions of the Colorado Plateau (Belnap, 1996; Brooks and Pyke, 2001; Fleischner, 1994; Neff et al., 2005).

Livestock Influences on Soil Health

Research indicates that high concentrations of livestock alter soil structure. Soil structure is essential for soil health and productivity, as it controls the movement of air, water, roots, and soil organisms into and through the soil (Roberson, 1996). Livestock grazing modifies the soil structure primarily by compaction. Soil compaction increases with grazing intensity and is typically more pronounced in mesic environments, such as those found in riparian and wetland ecosystems, where the soil moisture content is high (Hamza and Anderson, 2005; Kauffman and Krueger, 1984; Milchunas, 2006). Soil compaction has important implications in terms of plant and soil organism productivity, infiltration rates, runoff, and erosion (Orodho et al., 1999). Compaction reduces water and air movement into and through the soil, restricts plant growth by decreasing water availability and the amount of soil pore space, reduces soil biota, such as earthworms, reduces soil nutrient and water-holding capacities, decreases soil stability, and increases surface water runoff and soil erosion (Fleischner, 1994; Kauffman and Krueger, 1984; Milchunas, 2006; Roberson, 1996; Southorn and Cattle, 2004).

The infiltration rate, the rate at which water enters the soil, is directly affected by the degree of soil compaction. Heavy grazing has been shown to decrease infiltration rates and water-holding capacities in a range of soil types and geographic areas throughout the western United States (Roberson, 1996). Research has demonstrated that there are greater infiltration rates in exclosures than in moderately or heavily grazed areas (Milchunas, 2006). Decreased infiltration rates are amplified by reduced vegetative cover and plant litter. Livestock often consume and trample large amounts of plant material, and consequently increase the proportion of bare ground. Reductions in plant cover may have adverse consequences, particularly in forested ecosystem, because standing vegetation and plant litter are critical in slowing surface water runoff, promoting water infiltration, and protecting the soil from the erosive forces of precipitation (Belsky and Blumenthal, 1997).

Decreased infiltration rates and water-holding capacities typically intensify surface water runoff. Increased surface water runoff promotes plant cover reductions, sediment production, and soil erosion (Milchunas, 2006). Soil erosion has considerable effects on soil productivity and ecosystem function because the majority of nutrients, organic matter, microorganisms, soil fauna, and roots are concentrated in the top soil horizon (Roberson, 1996). Consequently, soil erosion can result in decreased on-site productivity and increased susceptibility of downstream flooding (Warren et al., 1986; Fleischner, 1994). Accelerated soil erosion in riparian, wetland, or meadow environments also has substantial impacts on water quality, channel structure, water table depth, and hydrologic regimes (Roberson, 1996; Wheeler et al., 2002).

In undisturbed environments, plant cover and litter layers maintain ideal soil temperatures for plant growth. However, the removal of vegetation either through livestock grazing or accelerated erosion can increase the proportion of bare ground (Belsky and Blumenthal, 1997; Kauffman and Krueger, 1984). Exposed soil surfaces increase the amount of incoming solar radiation, and consequently, rise soil temperatures and increase evapotranspiration (Kauffman and Krueger, 1984). Increases in soil temperature can alter seasonal plant growth (Kauffman et al., 1983), and in some cases plant community composition can change to warm-season plant species (Ingram et al., 2008). Increases in evapotranspiration in the surface soil can decrease soil moisture. Lower soil moisture can reduce plant productivity and increase water stress during periods of drought (Belsky and Blumenthal, 1997).

Livestock grazing can impact soil chemistry and fertility by modifying nutrient cycles within the plant-soil system (Schuman et al., 1999; Roberson, 1996). Livestock, through herbivory, digestion, and excretion, can increase decomposition rates and alter the amount, distribution, and availability of nutrients stored within the soil (Roberson, 1996). Carbon and nitrogen inputs into the soil can be reduced if plant productivity decreases. Soil organic matter, a primary source of nutrients, can be lost through erosion. Rates of plant decomposition can be accelerated with increasing soil temperatures (Milchunas, 2006; Belsky and Blumenthal, 1997; Shariff et al., 1994; Schuman et al., 1999). The magnitude and direction of change in soil nutrient cycles is dependent on the grazing management system (i.e. grazing intensity, frequency, and duration), as well as the soil physical and chemical properties and the geographic location (Manley et al., 1995; Stephenson and Veigel, 1987).

The responses of soil nutrients to grazing are variable. Some studies indicate higher levels of carbon and nitrogen in the surface soil in grazing pastures, while others report substantial depletions in soil carbon and nitrogen (Manley et al., 1995; Neff et al., 2005). These findings demonstrate the complex interactions between grazing and soil nutrients within different ecosystems (Manley et al., 1995). One consistent finding is that grazing appears to affect soils in arid and semi-arid systems differently than temperate or mesic environments where grazing by large native ungulates contributed to plant productivity and nutrient cycling (Neff et al., 2005).

The soils in arid and semi-arid environments are often protected by biological soil crusts composed of lichens, mosses, and cyanobacteria. Biological soil crusts are prevalent in cold-desert regions of the Colorado Plateau, and they play a major role in soil stabilization, nitrogen fixation, water infiltration, water holding capacity, and plant germination (Belnap, 1996; Brooks and Pyke, 2001; Fleischner, 1994; Neff et al., 2005). Surface disturbance by livestock, people, and recreational vehicles reduces nitrogen fixation activity (Belnap, 1996). Since nitrogen is a limiting factor in desert environments, impacts to biological soil crust can affect ecosystem productivity and fertility (Belnap, 1996; Fleischner, 1994; Neff et al., 2005).

Management Strategies

Proper management of plant communities is the best strategy for maintaining rangeland soil health (USDA NRCS, 2001). In well-managed pastures, the impacts to soils are likely to be moderated by vigorous ground cover (Southorn and Cattle, 2004). Increasing vegetative cover and plant productivity can reduce the detrimental impacts of soil compaction by improving soil organic matter. Promoting the growth of a mix of species with different rooting depths and patterns can increase infiltration rates and reduce the probability of accelerated erosion (USDA NRCS, 2001). Research has demonstrated that soils high in organic matter, with high soil aggregate stability and containing a large quantity and density of roots, can resist compaction pressures and recover more quickly from impacts associated with livestock grazing (Southorn and Cattle, 2004). Plant and soil health can also be promoted by minimizing soil surface disturbance, especially in arid areas, and by decreasing the extent of soil compaction through the avoidance of high-intensity grazing when soils are wet (USDA NRCS, 2001).

Grazing management practices should encourage sufficient residual vegetation on both upland and riparian sites to maintain appropriate water infiltration and to protect soils from water and wind erosion (USDI BLM, 2007). The amount of residual vegetation is more important than the forage utilized, as the condition of most ranges will deteriorate when little residual vegetation remains (Holechek et al., 1982). Management practices in any grazing system should leave sufficient vegetation for the protection of the site and maintenance of plant vigor. Residual plant cover, a measure of the herbaceous vegetation remaining after grazing, has been used to gauge the impacts of grazing (Clary and Leininger, 2000). Clary and Webster (1989) recommend a four- to six-inch stubble height at the end of the grazing season to preserve plant vigor and protect soils in healthy riparian areas. Stubble heights greater than six inches are appropriate for unhealthy riparian systems, easily eroded streambanks, and critical fisheries (Clary and Webster, 1989). However, some researchers and managers have questioned and cautioned against the premise of stubble height as a management objective and indicator of long-term rangeland condition (Laycock, 1998; UOI Stubble Height Review Team, 2004).

The development of objectives relative to biological soil crusts is an important part of rangeland management and soil health (USDA NRCS, 2001). Not only do biological soil crusts play a vital role is soil stabilization, water infiltration, and nutrient cycling, recovery time can be extensive. Regeneration is posited to range from decades to centuries, depending on the ecosystem, species composition, and soil type (Neff et al., 2005; Brooks and Pyke, 2001). Disturbance to biological soil crusts can be minimized by deferring grazing and recreational use during periods of susceptibility. Biological soil crusts on sandy soils are less susceptible to disturbance when the soils are wet or moist, while the biological soil crusts on clayey soils are less susceptible to damage when the soils are dry (USDA NRCS, 2001).

Appropriate grazing practices can be effective in protecting soil health (Roberson, 1996). Promoting periodic rest or deferment from grazing during critical growth phases, supporting adequate periods for recovery and regrowth, and providing opportunities for seed dissemination and seedling establishment can individually or collectively enhance the plant-soil system in rangelands (USDI BLM, 2007). There is increasing evidence that rotational grazing systems support greater persistence and productivity of perennial pastures with beneficial changes to botanical composition and soil quality. The rest period associated with rotational grazing appears to be an important component of recovery (Southorn and Cattle, 2004). Additionally, adaptive management techniques that alternate the season of use, reduce the duration of grazing, and limit the utilization can be effective in controlling soil compaction, surface water runoff, and soil erosion (Roberson, 1996).

Innovative management techniques can amend traditional livestock grazing patterns and significantly improve the sustainability of rangelands in arid ecosystems. The uniformity of grazing can be increased and sensitive rangeland habitats can be protected by changing attributes of the pasture or by modifying animal behavior (Bailey, 2004). Water developments, salt ground relocation, and fencing have been used to successfully improve livestock grazing distribution and rangeland health. Development of off-stream water sources combined with strategic salt placement can be an economically beneficial approach to decrease grazing pressure in riparian zones. Fencing, a direct method of altering livestock grazing patterns, can protect sensitive sites and prevent congregations of livestock in localized areas (Bailey, 2004; Roberson, 1996).

In severely degraded rangelands, many researchers indicate that complete rest is the most effective and rapid method to repair grazing damage to soils, particularly in riparian ecosystems (Roberson, 1996). Livestock exclusion has consistently resulted in the most dramatic and rapid rates of recovery (Belsky and Blumenthal, 1997; Clary and Webster, 1989; Kauffman et al., 1983; Kauffman and Krueger, 1984; Milchunas, 2006; Roberson, 1996). Recovery of a montane riparian zone to pre-disturbance soil physical properties has been shown to occur within one year after a heavy grazing event (Wheeler et al., 2002). Other research indicates that more than one year of protection from grazing is necessary for significant soil response, and that recovery can continue for up to four years (Stephenson and Veigel, 1987).

Rangeland management strategies that contribute to improved plant and soil health can also provide opportunities for soil carbon sequestration. Rangeland ecosystems contain 10 to 30 percent of the soil organic carbon on earth (Kimble et al., 2001). Changes in rangeland soil carbon storage can occur in response to a wide range of management and environmental factors, including grazing, fertilization, fire, and soil erosion. The majority of soil carbon in rangeland ecosystems is concentrated near the soil surface where it is susceptible to loss or redistribution by water and wind (Schuman et al., 2002). Accordingly, sustainable management practices that encourage healthy and productive rangelands can promote carbon sequestration (Ingram et al., 2008).

WATER QUALITY

Recent National Water Quality Inventory reports indicate that nonpoint source pollution from agricultural practices, including livestock grazing, is the leading source of water quality impacts to rivers and lakes and a major contributor to ground water contamination (US EPA, 2007). Nonpoint source pollution is water pollution caused by diffuse sources with no discernable distinct point of source, and it is intensified by surface water runoff events (USDA NRCS, 2006). Under certain conditions, livestock can directly and indirectly degrade water quality by enhancing sedimentation, increasing nutrients, such as nitrogen and phosphorus, promoting the spread of enteric pathogens, and rising water temperature (Campbell and Allen-Diaz, 1997; Doran and Linn, 1979; George and Clawson, 1993; Hubbard et al., 2004).

The degree of water quality degradation is exacerbated by riparian habitat alteration and destruction. It is estimated that 80 percent of stream and riparian ecosystems in the arid western United States have been affected by livestock grazing (Agouridis et al., 2005; Belsky et al., 1999). Riparian vegetation is a major constituent in the maintenance of water quality because it buffers water bodies from incoming sediment and other potential pollutants (Kauffman et al., 1983; Fitch and Adams, 1998). Therefore, when livestock modify the physical condition of riparian ecosystems, water quality can be adversely affected (Campbell and Allen-Diaz, 1997).

Livestock Influences on Water Quality

Soil erosion and sedimentation are the primary contributors to lowered water quality from rangeland. Pasture and rangeland generally become a source of sedimentation when livestock remove a large percentage of the vegetative cover for an extended period of time. The removal of plant cover exposes the soil surface to the erosive actions of water and wind. Eroded soils subsequently become sources of sediment, creating the potential for water degradation (George and Clawson, 1993). Instream trampling and loss of streambank stability from soil compaction can accelerate streambank erosion and sedimentation (Belsky et al., 1999; George and Clawson, 1993).

Excessive amounts of dissolved or suspended sediment can have a major impact in altering stream ecosystems by modifying the turbidity of water (Kauffman and Krueger, 1984). Turbidity is a measure of water cloudiness caused by suspended sediment. Waters with extremely high turbidity reduce light penetration into the water and reduce photosynthesis of aquatic vegetation (USDA NRCS, 2006). Excessive sediment can also cover fish spawning habitat, reduce foraging success by aquatic organisms, disrupt fish migration, distress respiratory systems of invertebrates, and alter food webs. Increased sediment loads may reduce reservoir storage capacity and increase the costs for filtration of domestic water supplies (Belsky et al., 1999).

Nutrients from livestock manure and urine are often cited as a major factor of water pollution (Doran and Linn, 1979). Excessive loss of nutrients through surface runoff and soil leaching are principal causes of surface and ground water degradation (Hooda et al., 2000). Nutrients from livestock can negatively affect water quality when the number of grazing animals per land area exceeds the fertility needs of the vegetation (Hubbard et al., 2004). Nutrient problems are usually most critical in riparian areas where livestock congregate for water and forage. Nutrient loading can also be problematic during rainy seasons and periods of runoff (George and Clawson, 1993). Nutrients that are not absorbed onto sediment are more likely to be transported in overland flow when soil moisture is high or when soils are frozen (Mosley et al., 1999).

Nitrogen, in the form of nitrate, and phosphorus, in the form of phosphate, are the principal nutrients of concern (Hubbard et al., 2004; Mosley et al., 1999). Both nitrogen and phosphorus are essential to plant and animal productivity; however, excessive amounts of either can over-stimulate aquatic plant growth (Mosley et al., 1999). If the resulting plant growth is moderate, it may provide a food base for the aquatic community; however, if the plant growth is excessive, the nutrients can stimulate algal blooms (Belsky et al., 1999). Algal blooms are rapid growths of algae in and on a body of water as a result of high nutrient concentrations (USDA NRCS, 2006). Subsequent decomposition of the algae increases the demand for oxygen, resulting in low dissolved oxygen levels and eutrophication (Belsky et al., 1999). Eutrophic bodies of water with extremely low dissolved oxygen levels can endanger and reduce the survival of local populations of fish and aquatic life (USDA NRCS, 2006; Belsky et al., 1999; Hubbard et al., 2004).

Water quality in many rivers and lakes has been impaired by the presence of high levels of pathogens (Hubbard et al., 2004). Livestock waste can carry a variety of bacterial and protozoan pathogens (Hooda et al., 2000). Research indicates that livestock grazing activities increase the bacterial counts of water in grazed watersheds and in streams adjacent to grazed pastures (Buckhouse and Gifford, 1976; Hooda et al., 2000). Contamination of water by fecal matter and pathogens has traditionally been assessed using counts of selected bacterial indicators, such as coliform bacteria (Doran and Linn, 1979; Hooda et al., 2000). Coliform bacteria are a group of bacteria inhabiting the intestines of animals (USDA NRCS, 2006). Although coliform bacteria are not considered to be pathogenic, they indicate the existence of fecal contamination and the potential existence of bacterial and protozoan pathogens (Belsky et al., 1999; George and Clawson, 1993).

Escherichia coli, Salmonella, and Leptospira are common bacterial pathogens; and Giardia and Cryptosporidium parvum are common protozoan pathogens (Belsky et al., 1999; Buckhouse and Gifford, 1976; Campbell and Allen-Diaz, 1997; Hooda et al., 2000). Water bodies used for sources of drinking water or for recreational activities can be contaminated when surface runoff or leaching occurs due to rainfall. Consequently, elevated concentrations of pathogens pose a potential health hazard for people who use the water for drinking, bathing, and recreating (Hubbard et al., 2004). Current municipal water treatment practices can remove most bacterial pathogens; however, protozoan pathogens are more resistant to water treatment chemicals and processes. Although Giardia is often removed with chlorine, it is now the leading waterborne parasitic disease in the United States (Hooda et al., 2000). Cryptosporidium parvum is believed to be one of the most concerns to health because it is resistant to levels of chlorine routinely used in water treatment plants (Mawdsley et al., 1995).

Thermal pollution, or increased water temperature, can be an indirect effect of livestock grazing. Water temperatures can rise as a consequence of reductions in stream shade due to heavy or continuous grazing of streamside vegetation (George and Clawson, 1993). Overgrazing of streambanks can result in drier, hotter ambient conditions and warmer water temperatures (Belsky et al., 1999). Increases in water temperature have been shown to have drastic effects on fisheries, particularly cold-water fish habitat, and aquatic insect populations (Belsky et al., 1999; Kauffman and Krueger, 1984; Campbell and Allen-Diaz, 1997). Changes in average temperature or daily fluctuations can create entirely new aquatic ecosystems

(Kauffman and Krueger, 1984). Temperature changes may also be important in terms of allowing the environment to become more conducive to pathogen growth (Campbell and Allen-Diaz, 1997).

Management Strategies

Controlling nonpoint source pollution from livestock grazing is a necessary step to improving the water quality of streams, lakes, and reservoirs. Enhanced water quality will most likely result from alterations in grazing patterns (Agouridis et al., 2005). Grazing patterns can be changed to reduce the impact by modifying the timing, duration, frequency, and intensity of use. Since biological and physical processes, such as plant nutrient uptake, water infiltration, surface water runoff, and streambank stability are all affected by the season of the year and weather conditions, determining the timing or season of grazing is important for manipulating livestock patterns (Mosley et al., 1999; Bailey, 2004).

Upland forage quality is typically higher during the spring and early summer, and therefore, forage utilization by livestock may be more uniform. Conversely, upland forage quality is lower during the late summer when plants are mature and soils are dry. Consequently, livestock often spend a disproportionate amount of time in riparian areas where the quality and volume of forage is higher (Bailey, 2004; Clary and Webster, 1989). Timing of grazing is also important in relation to seasonal precipitation and runoff events. Precipitation and runoff events can increase nutrient transport from rangelands to water bodies, particularly when soil moisture is high (Mosley et al., 1999).

Grazing frequency and stock density are important variables in controlling or mitigating water pollution. The frequency of grazing refers to how often a plant is utilized by livestock, and the stock density, or animal concentration, is simply the number of animals per unit area at a specific time. Some studies indicate that stock density is the most important variable affecting rangeland condition, as even light to moderate stock densities over a longer period of time may promote selective grazing in riparian areas and limit grazing uniformity (Heady and Child, 1975; Holechek et al., 1998; Stoddart et al., 1975). Continuous periods of grazing can ultimately stress the plant-soil-water resource by increasing sediment production, overland erosion, nutrient enrichment, and pathogen concentration. In contrast, light to moderate stock densities for brief periods of time can usually be sustained on healthy rangelands as long as grazing does not occur during critical biological or physical periods (Mosley et al., 1999).

Several management practices exist to control or mitigate nonpoint source pollution. Research suggests that several cultural and structural control management practices will need to be implemented for successful water quality improvements. Cultural control management practices, such as off-stream watering sources, are designed to minimize pollutant input into waterways, while structural control management practices, such as riparian buffer strips, are designed to modify the transport of pollutants to waterways (Agouridis et al., 2005).

Water access is an essential component of livestock management. The development of off-stream watering systems is a management practice that can decrease grazing pressure in riparian ecosystems, reduce streambank erosion, and minimize the direct deposition of livestock waste (Bailey, 2004). Studies have revealed that alternate water sources can improve water quality by reducing streambank erosion, total suspended solids, total nitrogen, total phosphorus, sediment-bound phosphorus, and fecal bacteria. Although off-stream water sources can improve water quality, research has suggested that the exclusive use of alternate water sources is not sufficient to maintain water quality improvements (Agouridis et al., 2005). Additionally, the development of off-stream watering systems at frequent intervals may not be cost effective in arid rangelands (Bailey et al., 2004).

Exclusion fencing is a method of improving water quality by limiting livestock access into riparian zones (Bailey, 2004). In areas where water quality, streambanks, and riparian vegetation are degraded, livestock can be excluded by installing high tensile fences, solar-powered electric fences, and woven fences (Hoorman and McCutcheon, 2005). Fencing of stream channels can provide a buffer zone between waterways and livestock, thereby reducing the health hazard associated with waterborne enteric pathogens (Buckhouse and Gifford, 1976). Exclusion fencing can promote accelerated restoration of riparian vegetation. Riparian recovery can stabilize stream channels, improve water quality by buffering the stream from incoming sediment and pollutants, and enhance habitat for wildlife and aquatic organisms (Fitch and Adams, 1998). Although exclusion fencing may be one of the most effective management practices, there are significant economic constraints (Agouridis et al., 2005).

Riparian buffer strips are another effective tool for reducing water pollution. Strategically placed buffer strips in rangelands can effectively mitigate the movement of sediment, nutrients, and livestock waste. Buffer strips can control soil erosion by wind and water; improve soil quality; enhance water quality by removing sediment, nutrients, pathogens, and other pollutants; increase water infiltration; and maintain or reduce streamwater temperatures. Riparian buffer strips can also improve aquatic and terrestrial life habitat by conserving and enhancing plant and habitat biodiversity (USDA NRCS, 2006).

The effectiveness of riparian buffer strips as a management practice can be increased through the use of both deep and shallow rooted vegetation. Riparian buffers primarily consisting of trees and shrubs are highly effective at stabilizing streams and are moderately effective at filtering dissolved nutrients; while riparian buffers primarily consisting of grasses are effective at filtering sediment and moderately effective in filtering dissolved nutrients (Agouridis et al., 2005). In forest ecosystems, the Natural Resources Conservation Service national specifications for riparian buffer systems recommend a three-zone buffer strip. The first zone is a narrow band of trees adjacent to the stream channel, the second zone is a forest management zone where maximum biomass production is stressed, and the third zone is a grass buffer strip used to provide control of sediment and overland flow. In drier regions of the western United States where tree growth is limited, buffers of grasses and shrubs have been advocated (Hubbard et al., 2004).

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