

**THE EFFECT OF CATTLE GRAZING IN RIPARIAN AREAS
ON WINTER BIODIVERSITY AND ECOLOGY**

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General Abstract

Livestock grazing in riparian habitats alters the vegetation structure, which has a detrimental effect on wildlife. This study examined the effect of cattle grazing in riparian habitats on winter bird biodiversity, small mammal biodiversity, and microclimate. Study sites were ungrazed, moderately grazed, and heavily grazed riparian habitats along the Oldman River, Alberta during winter 2005 and 2006. Bird species richness, individual abundance, and diversity indices were higher in ungrazed habitats than in grazed habitats. Deer mouse population sizes were not different except during spring 2006, when populations were larger in ungrazed sites. Microclimate data were collected in riparian sites and upland sites in winter 2006. Temperatures were higher and wind speeds were slower in riparian sites than in upland sites. Wind speeds were faster in heavily grazed riparian sites than in lightly grazed sites. Faster winds in heavily grazed sites may account for the decreased winter biodiversity in these habitats.

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Chapter 1

General Introduction

Over the last several decades, biological diversity and the conservation of biodiversity have become popular topics in the scientific literature. Biodiversity refers to the variety of species present at a level of biological organization (species richness), the relative abundance of the different species (individual abundance), and the genetic variations present among members of the species (genetic diversity). Biodiversity is important for both its aesthetic value and its practical value. The aesthetic value of biodiversity can be seen in the pleasure that people take from the beauty of nature's diversity, and in the positive effect of contact with natural environments on well-being and quality of life (reviewed in Miller 2005). The practical value of biodiversity is more concrete, as biodiversity is important for maintaining ecosystem services such as clean air, clean water and soil fertility (Schwartz et al. 2000; Hooper et al. 2005), biodiversity appears to be important for maintaining ecosystem stability (Tilman et al. 2006), and many ecosystem goods that have a direct practical value are related to biodiversity (Hooper et al. 2005) – for example: food, lumber, medicines (79% of the 150 most commonly used prescription drugs are derived from various species of plant, fungi, and bacteria [Dobson 1995]).

Measurements of biodiversity have become a common method for classifying habitat health. Biodiversity measurements have also been used to monitor the effects of habitat use on ecosystems in order to ensure sustainability (Franklin 1993). In the

conservation biology field, biodiversity measurements are vital as they give an inventory of the species and individuals that are present, indicate which species are at risk, and indicate which species might be in need of protection. The earth is currently experiencing a biodiversity crisis as species extinctions are occurring at a rate higher than what seems to have been the normal 'background rate' of extinction (Novacek and Cleland 2001; Singh 2002). Considering this biodiversity crisis and the fact that our knowledge of extant species is far from complete (Singh 2002), the study of biological diversity is currently viewed as an important area of study.

Habitat destruction and fragmentation are currently major threats to biodiversity (reviewed in Fahrig 2003). Agriculture is one of the main causes of habitat destruction and degradation in North America, leading to significant biodiversity losses. In general, agricultural land use leads to decreased biodiversity because of the physical changes to the landscape (such as tillage), the inputs of fertilizers and pesticides (McLaughlin and Mineau 1995) and because of the ecosystem simplification that occurs (such as draining wetlands to obtain more land for growing crops, and removing a variety of native plant species in order to grow a small number of crop species [Matson et al. 1997]). Due to these types of ecosystem changes, agricultural land-use often leads to a monoculture ecosystem.

The amount of land used in agriculture has rapidly increased over the last three centuries. Some of the consequences of this expansion include increased erosion, decreased soil fertility, pollution of ground water, river and lake eutrophication, and declines in biodiversity (Matson et al. 1997). Only one percent of Canada's total land area is considered to be prime agricultural land, and this limited amount of land is used so

intensively that some habitat types have been nearly eliminated (McLaughlin and Mineau 1995). The maintenance of biodiversity in areas of intensive agricultural practice is dependent on the conservation of all remaining habitats and a decrease in the agricultural practices which conflict with wildlife.

Although livestock grazing appears to cause considerably less disturbance than does cultivation, it too has serious impacts on biodiversity (e.g. Moser and Witmer 2000; Wallis De Vries et al. 2007). Cattle grazing is one of the most common land management practices in western North America, and the ecological costs of this practice are numerous. The ecological costs of livestock grazing often include alterations to soil properties, ecosystem functioning, and plant and wildlife species composition. For example, Dormaar and Willms (1998) found significant changes to fescue grassland soil properties after more than forty years of cattle grazing, such as changes in soil colour, and a 33% to 66% decrease in the amount of moisture held in the soil, with the largest decrease in soil moisture occurring in the spring. Livestock grazing has also been found to disrupt ecosystem functioning such as nutrient cycling (Dormaar and Willms 1998; Bakker et al. 2004). One study reported a 34% decrease in soil carbon, and a 24% increase in soil nitrogen after more than forty years of cattle grazing on fescue grasslands (Dormaar and Willms 1998). Intensive livestock grazing also causes alterations to vegetation communities, such as an increase in the biomass of grasses, and decreases in structural diversity on grazed lands (Scimone et al. 2007). The effect of livestock grazing on the composition of wildlife communities has been well documented, with many studies reporting biomass decreases for a variety of animal species, or the complete loss of a species from a community (for example: Taylor 1986; Hurly et al. 1998; Gonnet

2001; Matlack et al. 2001). One study on the effect of livestock grazing on the species composition of a community reported a 27% decrease in avian species richness, and a 33% decrease in individual abundance of birds on sites grazed by cattle compared to ungrazed sites (Popotnik and Giuliano 2000).

Cattle grazing, which refers to the grazing, browsing, and trampling of vegetation by foraging cattle, is very common on rangelands throughout southern Alberta. A small but vital proportion of this rangeland can be classified as the riparian zone, the habitat located between a watercourse and the upland terrestrial areas. Riparian habitats are ecologically important, as they provide water for plants and wildlife, and are home to a wide diversity of plant species (Jobin et al. 2004). Riparian vegetation is important for the regulation of light and temperature, it regulates the flow of water and nutrients, and it helps promote biodiversity by providing a diverse array of habitats and ecosystem services such as nutrient filtration (Lowrance et al. 1984), and removal of pollutants from water (Gilliam 1994; Hook 2003). Riparian plants also provide food and a variety of habitats for many wildlife species (Fitch and Adams 1998). As a result, riparian ecosystems are among the most biologically rich and complex habitats in arid and semi-arid regions (Fleischner 1994), and nearly 70% of vertebrate species in the region will use a riparian habitat in some way during their life cycle (Naiman et al. 1993). Some estimates say that more than 89% of riparian habitats in North America have been destroyed or damaged over the last two centuries due to logging, intensive agriculture, and land development (Popotnik and Giuliano 2000). Typically, the riparian corridor is the ecosystem-level element that is most sensitive to environmental change. Alterations to the landscape can affect the delivery and routing of sediment, water and woody debris

(Naiman et al. 1993). As a result, in riparian zones and other fluvial ecosystems, habitat preservation and biodiversity conservation require a “whole-ecosystem” perspective that considers the management and preservation of the entire riparian landscape (Naiman et al. 1993).

Due to the high availability of food, water and shelter in riparian habitats, livestock tend to congregate in these areas (Zuo and Miller-Goodman 2004). As a result, the ecological costs of grazing are magnified in riparian sites (Fleischner 1994). Some of the well-documented ecological costs of grazing are biodiversity losses and decreases in population density. For example, Hurly et al. (1998) found that cattle grazing in a riparian area resulted in an approximately 50% decrease in the number of individual breeding birds present. Also, a riparian study by Taylor (1986) found a decrease in bird biodiversity of approximately 59% after cattle grazing. Similarly, a study of riparian small mammals found a 25% decrease in small mammal biodiversity in grazed habitats (Giuliano and Homyack 2004). Other studies of riparian cattle grazing have reported damage to fish populations and a decrease in population densities of amphibians and reptiles (reviewed in Ohmart 1996). The impacts of cattle grazing on riparian ecology are largely due to physical disturbance such as trampling and soil compaction by cattle, and the alteration of plant communities by the act of grazing and browsing. One southern Alberta study reported that with increasing grazing intensity, the number of native shrub species decreased by approximately 85%, the percentage of ground covered by shrubs at 1 meter above ground decreased by more than 95%, and the number of exotic grass and forb species present increased by more than 100% (Hurly et al. 1998).

Although several studies have examined the effects of riparian cattle grazing on wildlife biodiversity, few studies have attempted to identify the biological mechanisms causing the changes to biodiversity. Holgate (2003) examined two possible mechanisms for the biodiversity differences of breeding birds in grazed and ungrazed riparian habitats. In one of these studies, the territory sizes of breeding birds were compared in riparian areas with varied grazing histories. In another study, the rates of food provisioning by adult house wrens to young were examined in the same riparian sites. Territory sizes were significantly smaller in ungrazed sites than in heavily grazed sites (Holgate 2003), and house wrens provided a higher number of food items per hour, and a higher number of food items per nestling in ungrazed riparian sites than in heavily grazed riparian sites (Holgate 2003). Taken together, these two studies provide support for the hypothesis that grazing influences food supply, food provisioning rates, and territory sizes for breeding birds (Holgate 2003).

Numerous studies have examined the effect of grazing on the biodiversity of various wildlife groups in riparian habitats during spring, summer and fall. For example, during the summer months, a decrease in shrub cover due to cattle grazing is thought to reduce the number of nest sites available for breeding birds and increase predation on nests (Hurly et al. 1998). Sedgwick and Knopf (1987) found that moderate grazing in the late fall had no impact on the biodiversity of migratory birds. Alternatively, Saunders and Hurly (2000) found that even moderate grazing in riparian habitats had a negative impact on the biodiversity of migratory birds. Compared to ungrazed sites, a 26.3% decrease in individual abundance and a 14.4% decrease in species richness was recorded in moderately grazed sites, and a 58.8% decrease in individual abundance and a 28.8%

decrease in species richness was recorded in heavily grazed sites (Saunders and Hurly 2000). Although the spring, summer and fall seasons are reasonably well studied, no information is available on the impact of livestock grazing on winter biodiversity in riparian areas. During the winter season in southern Alberta, defined as the non-growing season and typically lasting from November until late March, wildlife must contend with both low temperatures and high wind speeds. In more northern climates, snow provides insulation for burrowing small mammals throughout most of the winter season. However, in southern Alberta snow rarely stays on the ground for more than a few days, making the winter season an energetically stressful time of year for these and other animal species.

Some winter studies have occurred on upland grazed lands. Bock and Bock (1999) found that over a two-year study, the abundance of ground-foraging birds in winter was 1.7 to 2.7 times higher on ungrazed upland sites than on an upland site that experienced rotational cattle grazing. In another study, Bock et al. (1984) found that winter bird densities did not vary between a grazed upland site and an adjacent livestock enclosure. In winter months, riparian sites in southern Alberta are important habitats for several resident bird species that remain in the area throughout the winter months. Non-migratory small and large mammal species also continue to use riparian zones throughout the winter season, and relatively little is known about winter season responses of these species to cattle grazing in riparian habitats.

Any biodiversity differences that exist between grazed and ungrazed riparian sites during the winter may be an indirect result of the habitat modifications made by grazing cattle. At an ungrazed site, the well-developed shrub structure might act to shelter the

site from the wind, while the reduced shrub structure in a heavily grazed site would not provide shelter from the wind, so wind speeds in heavily grazed sites might be faster. These hypothesized microclimate differences between grazing categories would result in different energetic demands for animals living in these sites, as less energy would be required to live in a less windy site, and more energy would be required to live in a windier site. Microclimate differences between sites with different grazing intensities may act as a mechanism for biodiversity differences, as the energetic demands of living in a windy habitat may be too high for an individual to survive, so individuals of that species might be absent from habitats experiencing a higher intensity of livestock grazing.

The purpose of this study was to compare winter biodiversity of birds and small mammals in heavily grazed, moderately grazed and ungrazed riparian habitats, and to examine microclimates in these riparian habitats to determine if variation in microclimate is associated with variation in avian and mammalian biodiversity in winter. Over two winter field seasons, I carried surveys of birds and small mammals in 12 riparian sites to test the hypothesis that winter bird and mammal biodiversity differ among sites that vary in cattle grazing intensity. I carried out this research in twelve riparian habitats (four in each of heavily grazed, moderately grazed and ungrazed habitats) along a 50 km stretch of the Oldman River between Fort Macleod and Lethbridge in southern Alberta (see Appendix 1 for a map of the study area). I collected weather data at two upland locations, and microclimate data in twelve riparian sites to compare microclimates between the upland sites and the riparian habitats, and to test the hypothesis that winter microclimates vary among grazing categories.

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Chapter 2

Effect of Riparian Cattle Grazing on Winter Bird Biodiversity

Abstract

Livestock grazing is a widespread agricultural practice in western North America and is common in southern Alberta. Cattle spend a disproportionate amount of time in riparian habitats, which leads to changes to the vegetation structure of the riparian habitat as grazing intensity increases. To determine the effect of cattle grazing in riparian habitats on winter bird biodiversity, I performed bird surveys in ungrazed, moderately grazed, and heavily grazed riparian habitats along the Oldman River in southern Alberta during winter 2005 and 2006. Bird species richness and individual abundance were higher in ungrazed sites than in heavily grazed sites (in 2006, ungrazed sites had an average of 3.4 bird species and 19.2 individual birds, moderately grazed sites had an average of 3.1 bird species and 13.1 individual birds, and heavily grazed sites had an average of 2.1 bird species and 7.1 individual birds). Shannon-Wiener Diversity Indices, which combine information on both species richness and population sizes, were higher in ungrazed habitats than in grazed habitats in 2005 and 2006. Differences in bird species richness, individual abundance and the Shannon-Wiener Diversity Index values followed the hypothesized pattern of higher bird biodiversity in ungrazed riparian habitats versus grazed riparian habitats.

Introduction

Agricultural practices such as wetland draining, tillage, herbicide application, pesticide application, and livestock grazing have negative effects on the biodiversity of wild plants, animals and microorganisms (Mineau et al. 2005; Tschardt et al. 2005). When land is used to grow crops, the native vegetation is removed from the habitat to clear the land, and when land is used for livestock grazing, the grazing animals damage the native vegetation. Because of physical changes to the land and the removal of food sources due to agriculture, many wildlife species are driven out of the habitat once agricultural development occurs (McLaughlin and Mineau 1995). Although only about 1% of Canada's total land area can be considered prime agricultural land, a far greater portion of the land is used intensively in agriculture (McLaughlin and Mineau 1995), leading to significant impacts on biodiversity across the country.

Livestock grazing is a widespread agricultural practice across western North America, and is a very common land management practice in Alberta. Approximately 2% of Alberta rangelands can be classified as riparian zones, the transitional habitats between a watercourse and the surrounding upland terrestrial habitat (Adams and Fitch 1998). Over the last two centuries, 89% of riparian habitats have been destroyed due to agricultural intensification (Popotnik and Giuliano 2000), but riparian areas are vitally important to a wide variety of wildlife species and need to be protected if these species are to be preserved.

Wildlife species are attracted to riparian habitats because these areas offer more food, shelter and water than the surrounding upland habitats. Livestock are attracted to riparian habitats for the same reasons (Fleischner 1994), and spend a disproportionate

amount of time in these habitats compared to the surrounding upland areas (Zuo and Miller-Goodman 2004). Livestock grazing physically alters the riparian habitat by trampling plants and by removing plants by grazing and browsing. The shrub structure is reduced by the grazing livestock, decreasing the amount of diverse habitat available for many species of birds (Naiman et al. 1993). For these reasons, a decline in bird biodiversity, including both species richness and individual abundance, is often seen in riparian areas grazed by cattle.

Numerous studies have used bird surveys as a means of determining the health of a grazed riparian habitat. These studies have almost exclusively taken place in the spring and summer months, when birds are breeding and easy to survey due to their increased abundance and tendency to sing. The majority of these studies have found that cattle grazing in riparian areas has a negative effect on breeding bird biodiversity. For example, Taylor (1986) found that bird species richness in a grazed riparian habitat was approximately 59% lower than bird species richness in an ungrazed riparian habitat, Popotnik and Giuliano (2000) found that bird species richness was 1.6 times greater on ungrazed than grazed riparian sites, and Hurly et al. (1998) found that bird species richness was 1.4 times higher on ungrazed than grazed riparian areas. Cattle grazing in riparian areas also has an effect on the individual abundance of birds counted in spring and summer surveys. For example, Popotnik and Giuliano (2000) found 1.5 times more birds living on ungrazed riparian sites than on grazed riparian sites, and Hurly et al. (1998) found twice as many individual birds on ungrazed riparian sites than on grazed riparian sites.

Changes to bird biodiversity on land grazed by cattle may be the result of changes to the vegetation structure by the grazing animals (Knopf et al. 1988). Grazing by cattle removes plants, especially grasses (Clary and Medin 1990), which leads to increases in soil temperatures and the evaporation of soil moisture (Fleischner 1994). Cattle grazing has a larger impact on plant biomass than wildlife grazing. For example, Kauffman et al. (1983) found that grazing by cattle resulted in a 44% to 73% reduction in plant biomass in riparian meadows, while riparian meadows that were grazed only by wildlife experienced only a 1 to 2% reduction in plant biomass. The movement of cattle also leads to soil compaction and trampling of plants (Fleischner 1994). Cattle tend to rub on dead branches of shrubs, destroying foraging and habitat sites for some bird species (Knopf et al. 1988). Several studies have shown that cattle grazing in riparian habitats leads to a decrease in the number of shrubs present. For example, Schulz and Leininger (1990) found that cattle grazing in riparian habitats lead to a 81% decrease in shrub cover compared to ungrazed riparian habitats, and Popolizio et al. (1994) found a 71 to 78% decrease in shrub cover in grazed riparian habitats compared to habitats that had been protected from livestock grazing for at least 25 years. The percentage of ground covered by shrubs in a riparian habitat is particularly important to shrub nesting and shrub foraging bird species, therefore these bird species are the most likely to be affected by cattle grazing (Sedgwick and Knopf 1987).

Little is known about how birds use riparian habitats during the winter months. A survey of the relevant literature has found no studies of the effect of grazing on winter bird biodiversity in riparian habitats, and very few studies on the effect of grazing on winter birds living in upland habitats. Bock et al. (1984) studied the effect of livestock

enclosure in a grassland site in both summer and winter and found that the grazed site supported more birds in summer, but the number of birds did not differ between the grazed site and the cattle enclosure site during the winter months. Alternatively, another study found that the individual abundance of ground-foraging seed-eating winter birds was 1.7 to 2.7 times higher on ungrazed grassland sites than on grazed grassland sites, while the abundance of non-ground foraging winter birds did not differ between grazed and ungrazed grassland sites (Bock and Bock 1999). Although relatively little is known about how birds use riparian areas during the winter months, the winter season is an energetically stressful time for birds, and the winter ecology of birds may be relevant to biodiversity. During the winter season in southern Alberta, birds are exposed to extremely severe weather conditions, including low temperatures, low solar radiation levels, and high wind speeds. During winter, birds utilize different food sources than they do in the breeding season, and there is a much shorter period of daylight in which to forage. Also, during the winter months, the range sizes of some bird species expand and overlap, increasing the occurrence of interspecific and intraspecific competition (Grzybowski 1982).

The purpose of this study was to expand upon riparian bird biodiversity studies done along the Oldman River on breeding bird populations by Hurly et al. (1998) and Holgate (2003) by examining winter bird species richness and individual abundance in riparian habitats experiencing different intensities of cattle grazing. Eleven of the 12 sites used by Holgate (2003) were also examined in this study, and in the original riparian bird study conducted in this area, Hurly et al. (1998) used seven of the 12 sites used in this study. One of Holgate (2003)'s heavily grazed riparian sites was unavailable for use in

this study, so a new heavily grazed site was added to Holgate's study sites in the winter 2006 field season. I performed bird surveys throughout the winter season in ungrazed, moderately grazed, and heavily grazed habitats to determine the effect of cattle grazing on wintering birds.

Methods

Study Sites

I collected data in 11 cottonwood riparian sites in winter 2005 and in 12 cottonwood riparian sites in winter 2006. Of the 11 riparian sites used in 2005, 4 were ungrazed by cattle for at least 15 years, 4 had a history of moderate cattle grazing, and 3 had a history of heavy cattle grazing (Figure 2-1, taken from Holgate 2003). In 2006, I added a fourth heavily grazed site to the 2005 sites. All 12 sites were located along a 50 km stretch of the Oldman River between Fort Macleod and Lethbridge, Alberta (see Appendix 1 for a map of the study area). Of these 12 sites, six were city nature reserves, one site was located on federal government land, and five sites were located on private land.

Each riparian study site was a cottonwood forest site. The number of cottonwood trees did not vary between grazing treatments (Holgate 2003). The amount of shrub cover varied between grazing treatments (Figure 2-2, from Holgate 2003). The study sites varied in size, with approximate areas of 0.1 km² to 0.3 km². Study plots had previously been established in each of these sites by either Hurly et al. (1998) or Holgate (2003). Each study site had four circular plots, each with a radius of 50 m. I performed small mammal trapping and microclimate data collection within these plots in each study

site. Three of the ungrazed sites were located near each other (see Appendix 1), but this was the only option, as no other ungrazed riparian sites are available in this area. Two heavily grazed sites and two ungrazed sites were contiguous, but I was careful to ensure that data were collected separately.

Grazing Categories and Vegetation Surveys

Riparian sites had been previously classified into grazing categories based on a description of the grazing habitat and by information provided by landowners in response to a questionnaire about the site's grazing history (Holgate 2003). Landowners reported the average number of cattle they grazed in the riparian area, the average number of months the cattle grazed in the riparian area, the size of the riparian area the cattle were grazed in, and the months that cattle were grazed in the riparian area for the past fifteen years. Holgate (2003) carried out these surveys in 2002. In each grazed site, cattle grazing took place during the vegetative growing season. Animal-unit months were calculated by multiplying the number of cow-calf pairs by the number of months the animals spent in the riparian zone, divided by the size of the riparian zone in hectares. Heavily grazed sites had an average of more than 3 cow/calf pairs per hectare, moderately grazed sites had an average of less than 2 cow/calf pairs per hectare, and ungrazed sites had not been grazed by cattle for at least 15 years as of 2002 (Figure 2-1, Holgate 2003).

Holgate (2003) performed vegetation surveys in these sites in 2002 to determine if the vegetation structure varied among grazing categories. Within each site, 4 established circular plots were surveyed. Sampling occurred in 6 locations in each plot. At each sampling location, the percentage of ground covered by shrub was determined at ground

level, 0.5 m, 1 m, 2 m, 3 m, and greater than 4 m above ground. The cumulative percent shrub cover was determined by summing the percentage of ground covered by shrubs across the different sampling heights (Holgate 2003). The mean cumulative percent shrub cover was significantly higher in ungrazed sites than in heavily or moderately grazed sites (Figure 2-2, taken from Holgate 2003).

Bird Surveys

I performed bird surveys between 23 January 2005 and 16 March 2005, and between 19 December 2005 and 22 February 2006. Each survey lasted 60 minutes. During a bird survey, I walked transect lines parallel to the river along the length of the site. Parallel transect lines were approximately 50 meters apart. While walking transect lines, I recorded both the species of bird and number of individuals seen and heard. Between 23 January 2005 and 16 March 2005, I surveyed each of the 11 sites at least two times. I surveyed all 12 sites between 19 December and 30 December 2005. Between 2 January and 7 February 2006, I surveyed each of the 12 sites in sets a total of six times, with each survey set performed over the course of two to five days.

Data Analysis

I determined the average species richness values for both 2005 and 2006 by calculating the average number of species recorded across surveys in each riparian site, then calculating the average value of the four sites in a grazing category. In this way, I calculated mean species richness values for ungrazed sites, moderately grazed sites and heavily grazed sites for each of 2005 and 2006. I calculated the mean individual abundance values (number of individual birds) for data collected in 2005 and 2006 in the same way.

I used the Shannon-Wiener Diversity Index to compare bird species diversity between the three grazing categories. Shannon-Wiener Diversity Indices are calculated using the formula $H' = -\sum(p \cdot \ln(p))$ where H' is the measure of diversity and p is the relative abundance of each species, calculated as the proportion of individuals of one species to the total number of individuals in the community (Krebs 1989). I calculated the Shannon-Wiener Diversity Indices separately for each site in 2005 and 2006.

I analyzed data from 2005 and 2006 together using 2-way ANOVAs to compare the average number of bird species, the average number of individual birds, and the average value of H' found in each grazing category. The two factors used in the 2-way ANOVAs were grazing level and year. I tested the assumptions of a 2-way ANOVA (that the residuals are normal and homogeneous), and if the residuals were not normal, I Box-Cox transformed the Y variable.

I tested both the number of individual birds surveyed in 2005 and 2006, and the Shannon-Wiener Diversity Index values calculated for 2005 and 2006 using Page's test for ordered hypothesis (Page 1963). This powerful analysis tests the null hypothesis (that there is no difference in bird abundance or H') against the alternative hypothesis that there is an order to the categories being examined. Here, the hypothesis I used for the ranking of categories was that more individual birds and a higher H' would be found in sites that were ungrazed by cattle, fewer birds and a lower H' would be found in sites experiencing moderate amounts of cattle grazing, and the least number of birds and lowest H' found in sites that are heavily grazed by cattle. I calculated the test statistic L by ranking these categories from 1 to 3, then by ranking the average for each grazing category in each of the seven complete survey sets completed between 19 December

2005 and 7 February 2006. The rankings for each survey set were then added, and the sum of rankings was multiplied by the hypothesized rank for the category, and then added together to determine the L value. That is: $L = \sum(Y_j \sum X_{ij})$ where Y_j is the hypothesized ranking of the j th column, and $\sum X_{ij}$ is the sum of ranks observed in the j th column (Page 1963). I compared the value of L to a table of critical values for L (found in Page 1963) to determine the P-value for this test.

To determine if the species richness and individual abundance results hold true for individual bird species, I tested the seven bird species that were most often found to see if the number of birds found in 2006 varied between grazing categories. The seven species examined were black-capped chickadees, downy woodpeckers, flickers, great horned owls, hairy woodpeckers, magpies, and white-breasted nuthatches. I calculated the average number of birds found in a bird survey for each site, and I calculated the mean number of birds per survey for each grazing category. I used one-way ANOVAs to determine if bird numbers varied between grazing categories for these seven most commonly found species.

Results

Species Richness

Bird species richness was higher in ungrazed riparian sites than in grazed riparian sites in 2005 and 2006 (2-way ANOVA, $F_{2,17}=6.04$, $P=0.0104$) (Figure 2-3). Bird species richness was higher in 2005 than in 2006 (2-way ANOVA, $F_{1,17}=21.6$, $P=0.0002$) (Figure 2-3). There was no interaction between grazing category and year (2-way ANOVA, $F_{2,17}=0.654$, $P=0.533$).

Individual Abundance

Individual abundance of birds was not significantly different between grazing categories, (2-way ANOVA, $F_{2,17}=3.19$, $P=0.0665$), but there was a trend towards higher numbers of birds in ungrazed riparian sites than in grazed riparian sites (Figure 2-4).

Individual abundance of birds was higher in 2005 than in 2006 (2-way ANOVA, $F_{1,17}=5.58$, $P=0.0303$) (Figure 2-4). There was no interaction between grazing category and year (2-way ANOVA, $F_{2,17}=0.389$, $P=0.684$).

Shannon-Wiener Diversity Index

The Shannon-Wiener Diversity Indices were not significantly different between the three grazing categories (2-way ANOVA, $F_{2,17}=1.99$, $P=0.168$) (Figure 2-5).

Shannon-Wiener Diversity Indices were higher in 2005 than in 2006 (2-way ANOVA, $F_{1,17}=20.0$, $P=0.0003$) (Figure 2-5). There was no significant interaction between grazing category and year (2-way ANOVA, $F_{2,17}=1.37$, $P=0.280$).

Page's Test for Ordered Hypotheses

I tested bird abundance data from 2005 and 2006 and Shannon-Wiener Diversity Indices from 2005 and 2006 for ordered differences among the grazing categories. The hypothesized order of grazing categories for both bird abundance and Shannon-Wiener Diversity Index was that ungrazed sites would have the highest number of individuals and the highest H' , the moderately grazed sites would have fewer individuals and a lower H' , and the heavily grazed sites would have the least number of individuals and the lowest H' . Page's Test for Ordered Hypotheses (Page 1963) found significant differences in bird abundance between grazing categories (Page's Test for Ordered Hypotheses, $L=136.5$, $n=3$, $m=10$, $P<0.001$). Page's Test for Ordered Hypotheses also found

significant differences in H' between grazing categories (Page's Test for Ordered Hypotheses, $L=27$, $n=3$, $m=2$, $P<0.05$).

Most Common Bird Species

The mean number of birds counted per survey in 2006 did not vary between grazing categories for the seven most commonly found species (black-capped chickadees: ANOVA, $F_{2,9}=1.33$, $P=0.312$ (Figure 2-6a); downy woodpeckers: ANOVA, $F_{2,9}=3.34$, $P=0.082$ (Figure 2-6b); flickers: ANOVA, $F_{2,9}=3.17$, $P=0.0907$ (Figure 2-6c); great horned owls: ANOVA, $F_{2,9}=4.10$, $P=0.054$ (Figure 2-6d); hairy woodpeckers: ANOVA, $F_{2,9}=1.73$, $P=0.231$ (Figure 2-6e); magpies: ANOVA, $F_{2,9}=1.08$, $P=0.381$ (Figure 2-6f); white-breasted nuthatches: ANOVA, $F_{2,9}=2.06$, $P=0.183$ (Figure 2-6g)). I therefore pooled bird survey data among species. See Appendix 2 for a list of all bird species found during the 2005 and 2006 bird surveys.

Discussion

In a previous study which took place in the same riparian sites, Holgate (2003) found that the intensity of cattle grazing varied between these sites (Figure 2-1), and that the cumulative percent shrub cover decreased as grazing intensity increased (Figure 2-2). It seems clear that these grazing categories were still accurate representations of the grazing experienced in these sites in 2005 and 2006. Even if the grazing pressure had changed slightly, shrub growth rates are not high enough to result in a replenished shrub structure in only 3-4 years. Holgate (2003) found significantly more shrub cover at 0.5m, 1m, 2m, and 3m above ground in ungrazed sites than in grazed sites, and this was clearly the case in 2005 and 2006. These sites are therefore thought to be representative of

different livestock grazing pressures on the riparian habitats, and are useful for comparison of bird diversity and abundance.

In response to increasing cattle grazing pressure, the number of bird species and individual birds present during the winter months in the riparian zone declined significantly. The 2005 and 2006 data show strong statistically significant differences between bird species richness in the different grazing categories (Figure 2-3), and although initial ANOVA analysis of the 2005 and 2006 data show that bird abundance did not differ significantly between grazing categories (Figure 2-4), the more powerful Page's Test for Ordered Hypotheses (Page 1963) showed a highly significant influence of grazing on individual abundance.

In response to increasing grazing pressure, Shannon-Wiener Diversity Indices decreased, indicating that the bird species richness and individual abundance responses to grazing were in the hypothesized direction, and that bird biodiversity declined with increasing grazing pressure (Figure 2-5). The H' values calculated in this study (between 1 and 1.6 (Figure 2-5)) are low compared to reported measures of diversity, which approach 5.0 (Washington 1984), but are appropriate values for winter bird communities (Kricher 1972).

Bird species richness, individual abundance and diversity index results all show a significant year effect, with species richness, individual abundance, and diversity values higher in 2005 than in 2006 (Figure 2-3, Figure 2-4, Figure 2-5). It is possible that these differences in numbers are real, and that some species had not migrated in 2005, but migrated in 2006, but it seems more likely that these differences are due to early returns of migrant birds that were recorded in the late season surveys during the 2005 field

season. Bird species richness, and individual abundance data collected during the 2006 field season (data collected between 19 December 2005 and 22 February 2006) are likely more representative of the winter season than data collected during the 2005 field season (data collected between 23 January and 6 April 2005). Bird surveys conducted in late March and early April 2005 likely counted some early returning migratory birds, while in 2006 I limited data collection to the cold winter months. In addition, I collected more data in 2006 than in 2005, and collected 2006 data in 'survey sets', which meant that bird surveys were performed on all 12 sites over a short period of time. Due to these factors, the data I collected in 2006 are likely a better view of winter riparian habitat use by birds. However, even if late winter 2005 bird surveys did record some returning migrant birds, it is interesting to note that early returning migrant birds did appear to settle more often in ungrazed riparian habitats than in grazed riparian habitats (Figure 2-3, Figure 2-4), as was seen for migrant birds that returned to several of the same riparian sites later in the spring in 2002 (Holgate 2003).

Although the ANOVA tests of the number of the most common bird species did not show significant differences among grazing categories (Figure 2-6), some of these ANOVA tests were very close to the significant alpha level (downy woodpeckers, $p=0.082$; flicker $p=0.0907$; great horned owl, $p=0.054$), and for each of the seven most commonly found species, fewer birds were found in the heavily grazed sites than in the ungrazed sites (Figure 2-6). It seems worth noting that even though the tests of individual species were not statistically significant, the expected pattern of lower diversity in heavily grazed riparian sites did hold true.

Bird species richness and individual abundance are likely different during the winter in riparian areas with varied grazing histories due to the differences in shrub structure between the grazing categories. Holgate (2003) found that the percentage of ground covered by shrub in the riparian sites used in this study was highest in ungrazed riparian sites, and lowest in heavily grazed riparian sites (Figure 2-2). Holgate also reported that the gross habitat structure was not significantly different between these riparian sites, as the number of live and dead trees and the percentage of ground covered by bare ground, dead wood, grasses and forbs did not vary between the grazing categories (Holgate 2003). Since these other habitat measures are not significantly different across grazing categories, it seems likely that the shrub structure differences across the grazing categories are the reason for the bird biodiversity differences. Bird species richness and individual abundance have been found to increase with an increasingly complex vegetation structure in grazed and ungrazed habitats (Scott et al. 2003). Birds require shrubs during the breeding season for nesting (Knopf and Sedgwick 1992), shelter from the elements, and protection from predators (Ammon and Stacey 1997). During the winter season, birds require shrubs for shelter from the elements (see Chapter 4) and for food. Since birds require shrubs throughout the year, it is not surprising that decreases to the shrub understory due to livestock grazing seems to cause a decrease in bird species richness and individual abundance.

This winter study is unique in that no previous studies of winter bird biodiversity in grazed and ungrazed riparian habitats have been published. In a winter study in a grassland site, Bock et al. (1984) found that bird abundance did not differ between grazed and ungrazed grassland sites during the winter. In another winter study on upland sites in

Arizona, Bock and Bock (1999) found that the abundance of ground-foraging seed-eating birds was lower on grazed sites than on ungrazed sites (approximately 65% fewer birds on grazed sites during a winter season impacted by a drought, and approximately 39% fewer birds on grazed sites during the next winter field season when rainfall levels were normal), while the abundance of other birds was not significantly different between grazing categories. The ground-foraging and seed-eating bird species were also more affected by a drought than were the other bird species (Bock and Bock 1999). Unlike Bock et al. (1984), my study of riparian winter bird species richness and individual abundance did find significant differences in winter bird individual abundance (Figure 2-4), and species richness (Figure 2-3). Unlike Bock and Bock (1999), my study found relatively few ground-foraging seed-eating birds, as only 6 of 15 bird species counted throughout the 2006 field season were ground-foraging seed-eaters, several of those 6 species were omnivores that use more than one foraging strategy (Ehrlich et al. 1988), and the ground-foraging, seed-eating birds were found in all grazing categories (one species was found in all sites, two species were found only in one ungrazed site, one species was found only in one moderately grazed site, one species was found only in one heavily grazed site, and one species was found in one ungrazed, one moderately grazed, and one heavily grazed site). This indicates that the ground-foraging seed-eating bird species were not the only species affected by cattle grazing in this study. Perhaps the results of this study were different from the results of Bock et al. (1984) and Bock and Bock (1999) due to the very different winter climates that were examined in these studies. Both Bock et al. (1984) and Bock and Bock (1999) were studies of winter birds in Arizona. The lowest recorded temperature over thirty years of weather collection near

one of these study sites in Arizona was -3.0°C (Bock and Bock 1999). In Lethbridge, the average minimum January temperature is -13.9°C , with a lowest recorded January temperature of -42.8°C (in January 1950). Winter bird biodiversity is certain to be different and respond differently to grazing pressure when comparing studies carried out in the very mild desert climate of Arizona and the much colder prairie climate of southern Alberta.

A review of the possible causes for population declines of Nearctic migratory birds indicated that alteration of the wintering ground habitat was the best explanation for the population declines of these Nearctic birds (Rappole and McDonald 1994). One study of winter habitat quality found that male birds that over-wintered in a higher quality habitat arrived at the breeding habitat earlier than male birds that over-wintered in a poorer quality habitat (Norris et al. 2003). The same study found that birds that over-wintered in a higher quality habitat laid their first eggs earlier, fledged their first young earlier, and produced more offspring than birds that over-wintered in poorer quality habitats (Norris et al. 2003), suggesting that winter habitat quality is extremely important to reproductive success of these migrant birds. It is unknown if the same is true for winter habitat quality of non-migratory birds, but it seems possible that the quality of winter habitat for a resident bird is similarly important, and that birds that spend their winters in ungrazed riparian areas also lay their eggs earlier or produce more offspring than birds that spend their winters in heavily grazed riparian areas.

Holgate (2003) and Hurly et al. (1998) found that during the breeding season, bird species richness and individual abundance were lower in heavily grazed riparian sites than in ungrazed riparian sites in this area of southern Alberta. I found similar results

during the winter season. It is clear that declines in riparian habitat quality due to cattle grazing are detrimental to the birds that live in these habitats during the breeding season and the winter season. New range management techniques such as limiting the number of cow-calf pairs in riparian habitats and greatly reducing the amount of time cattle spend in riparian habitats would no doubt have a positive effect on bird species richness and individual abundance throughout the year.

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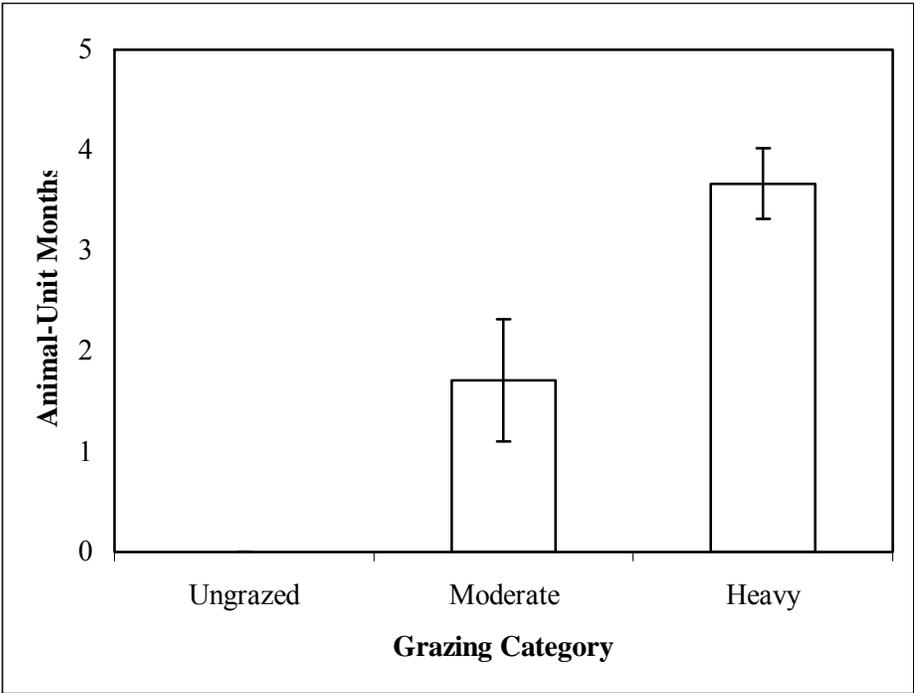


Figure 2-1. Mean (\pm S.E.) animal-unit months (AUM) per hectare for four riparian sites in each of three cattle grazing categories. Figure from Holgate (2003).

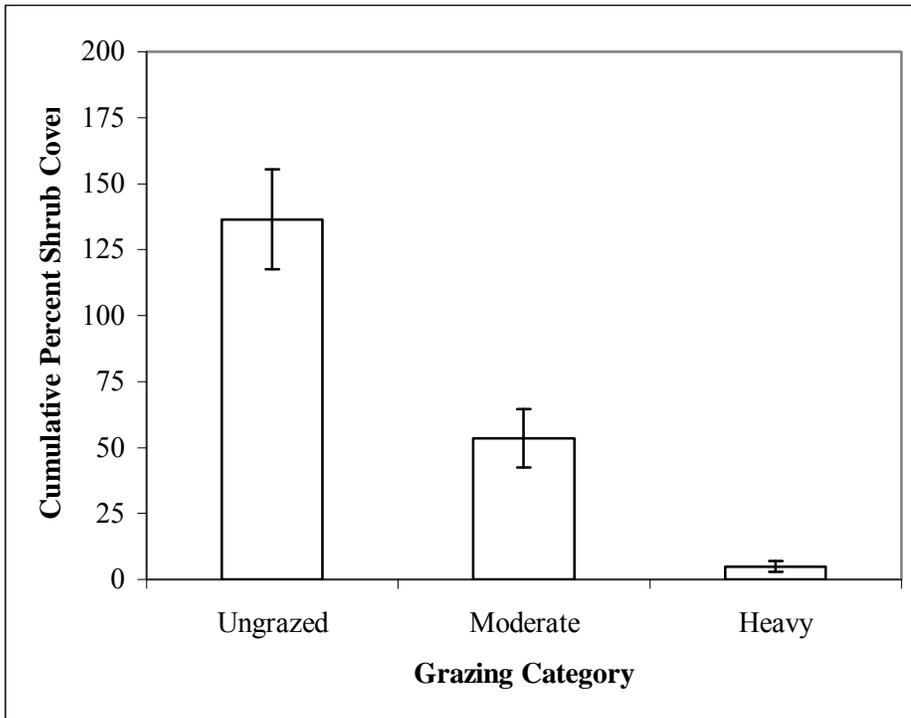


Figure 2-2. Mean (\pm S.E.) cumulative percent shrub cover for four riparian sites in each of three cattle grazing categories. Cumulative percent shrub cover is the sum of the amount of ground covered by shrubs at ground level, 0.5m, 1 m, 2 m, 3 m, and greater than 4 m above ground. Figure from Holgate (2003).

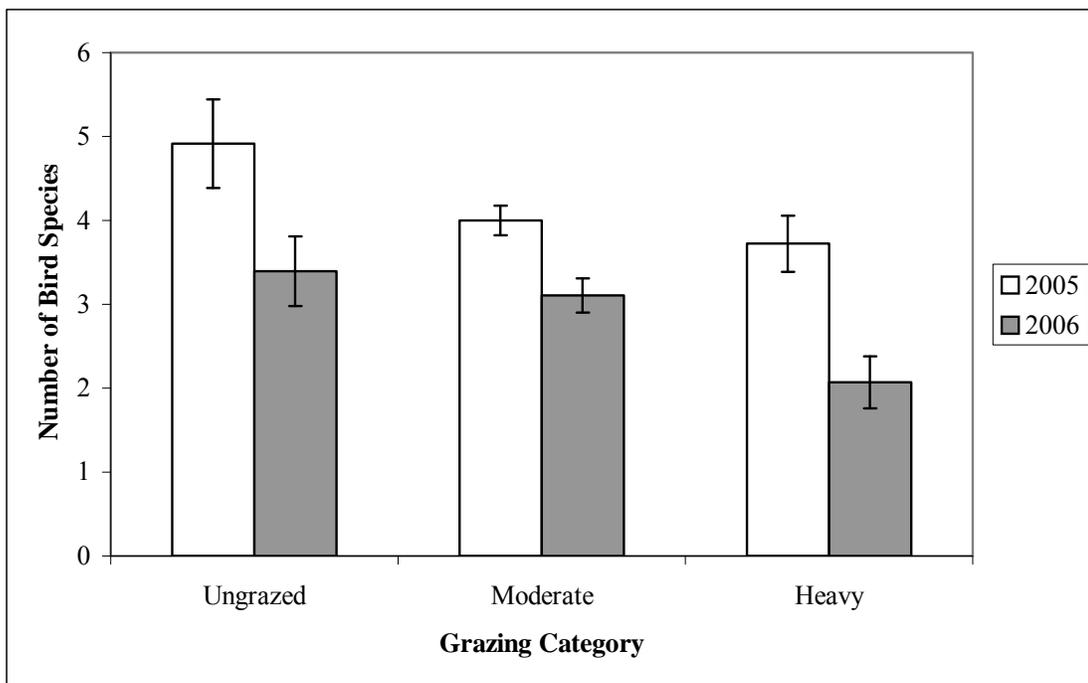


Figure 2-3. Mean (\pm S.E.) number of bird species surveyed at riparian sites in three grazing categories. 2005 data collected between 23 January 2005 and 16 March 2005 and were collected in four ungrazed sites, four moderately grazed sites, and three heavily grazed sites. 2006 data collected between 19 December 2005 and 7 February 2006, and were collected in the eleven field sites used in 2005, and in one additional heavily grazed site.

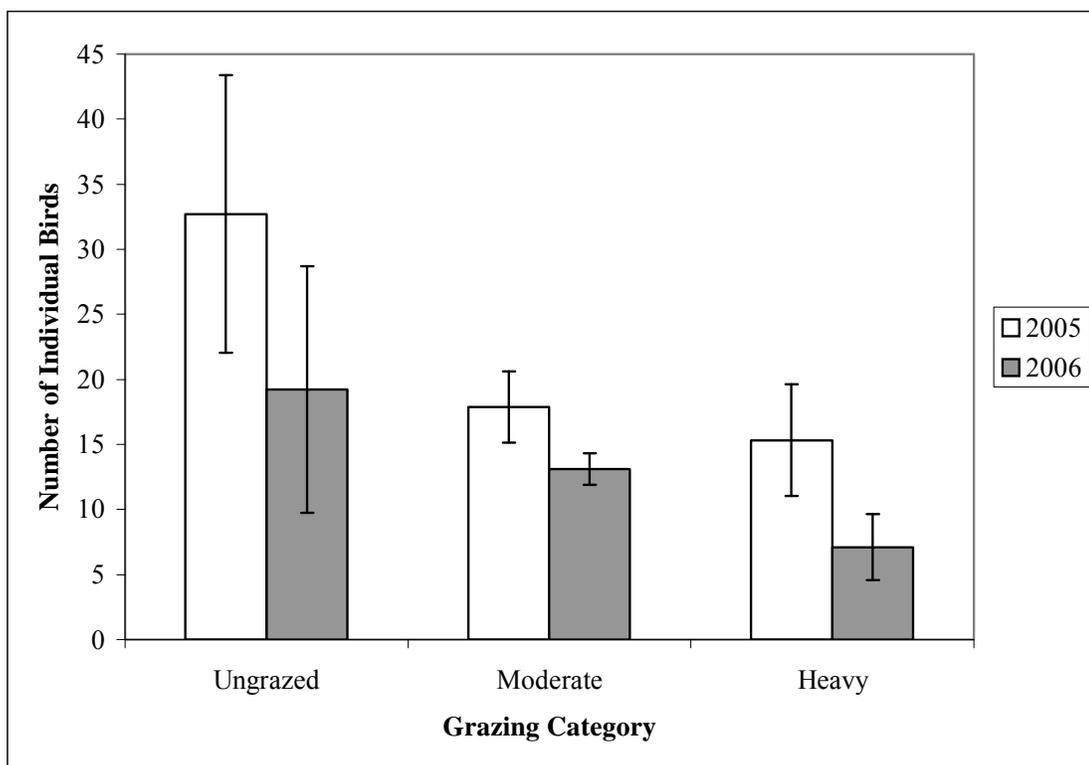


Figure 2-4. Mean (\pm S.E.) number of individual birds surveyed at riparian sites in three grazing categories. 2005 data collected between 23 January 2005 and 16 March 2005 and were collected in four ungrazed sites, four moderately grazed sites, and three heavily grazed sites. 2006 data collected between 19 December 2005 and 7 February 2006, and were collected in the eleven field sites used in 2005, and in one additional heavily grazed site.

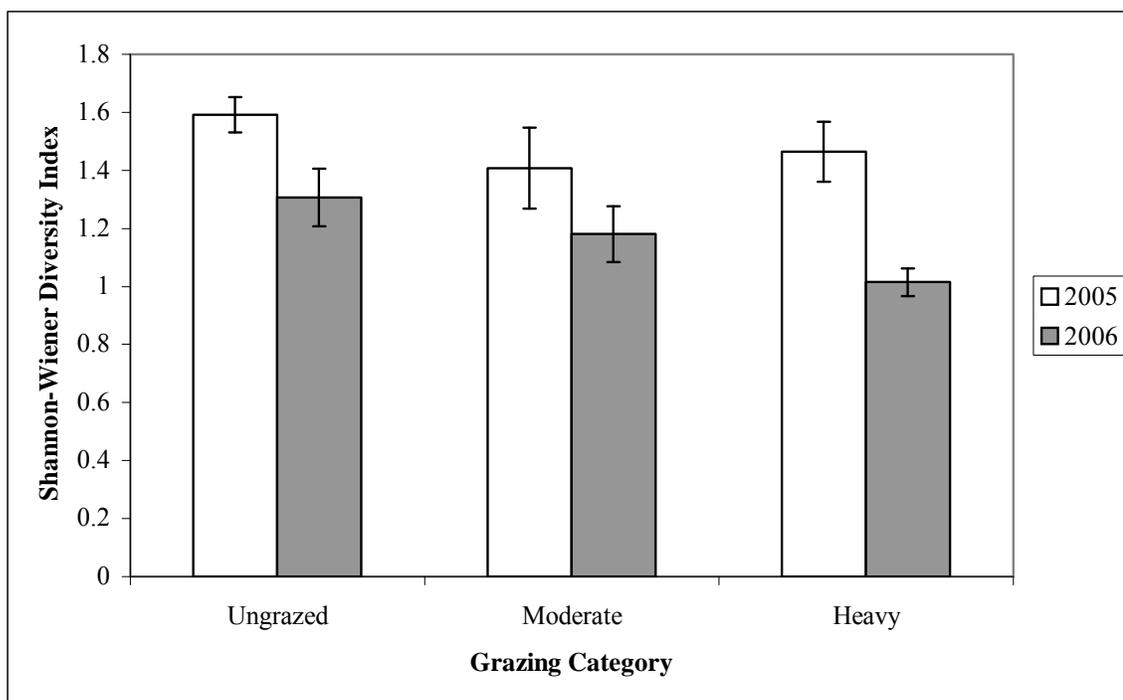


Figure 2-5. Mean (\pm S.E.) Shannon-Wiener Diversity Indices calculated for 12 riparian sites in three grazing categories. 2005 data collected between 23 January 2005 and 16 March 2005 and were collected in four ungrazed sites, four moderately grazed sites, and three heavily grazed sites. 2006 data collected between 19 December 2005 and 7 February 2006, and were collected in the eleven field sites used in 2005, and in one additional heavily grazed site.

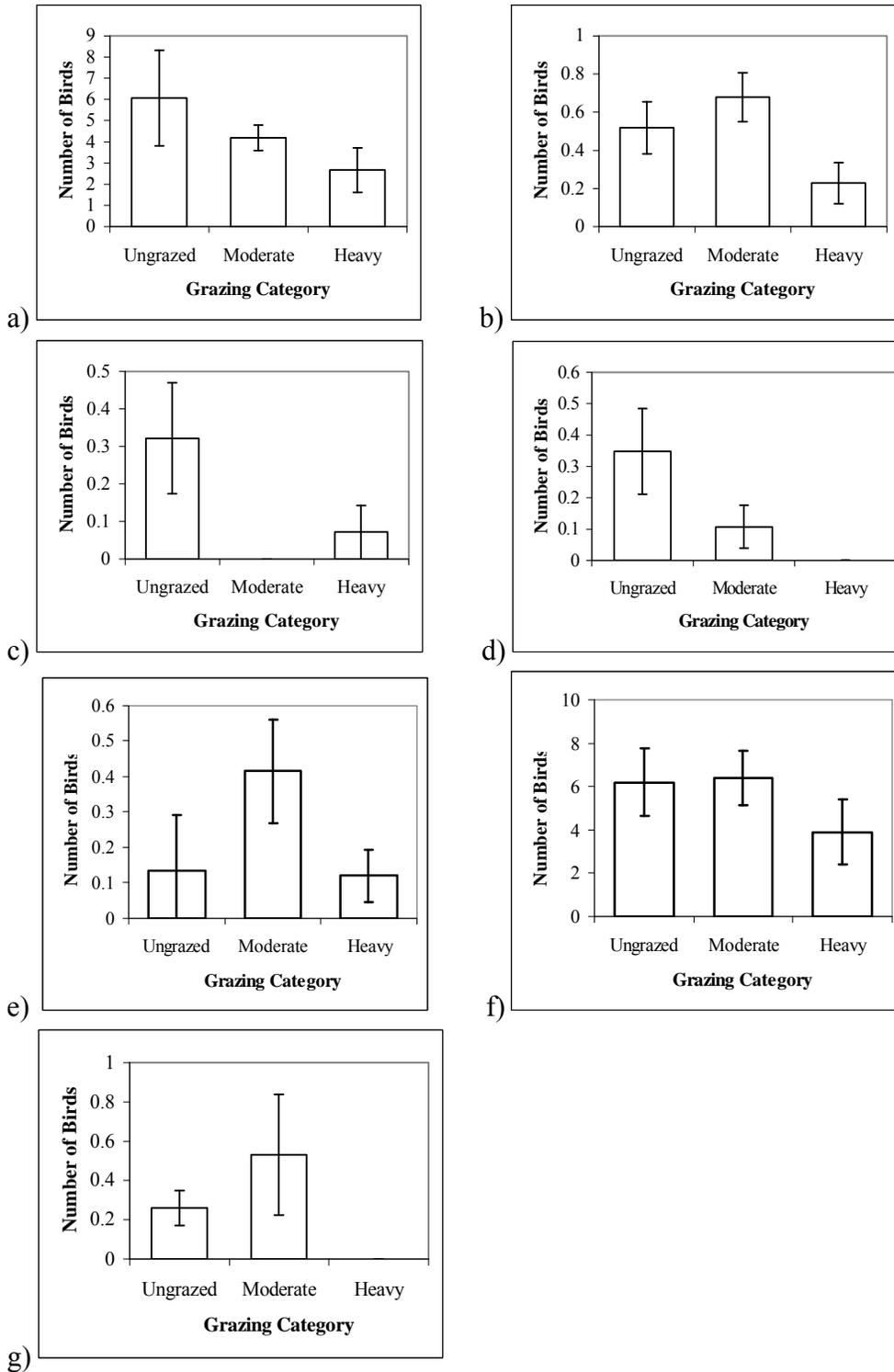


Figure 2-6. Mean (\pm S.E.) number of individuals of seven bird species surveyed in three grazing categories between 19 December 2005 and 22 February 2006. The seven species were the most commonly found bird species: a) black-capped chickadee, b) downy woodpecker, c) flicker, d) great horned owl, e) hairy woodpecker, f) magpie, and g) white-breasted nuthatch.

Chapter 3

Effect of Riparian Grazing on Biodiversity and Abundance of Small Mammals During Winter

Abstract

Livestock grazing in riparian habitats leads to large changes to the vegetation structure of the habitat, which has a detrimental effect on many resident wildlife species. Small mammals such as mice, voles and shrews live in riparian habitats in North America, and the impact of riparian livestock grazing on these species is somewhat unclear. This study examined winter small mammal biodiversity, population sizes, and over-winter survival in riparian sites with different histories of cattle grazing. I live-trapped small mammals in ungrazed, moderately grazed, and heavily grazed riparian habitats along the Oldman River during winter 2005 and during late fall 2005 and early spring 2006. Deer mice were trapped most often, and were trapped in each riparian study site. Masked shrews and meadow voles were both trapped less often, and masked shrews were only trapped in one site (heavily grazed site), while meadow voles were trapped in four sites (one ungrazed, two moderately grazed, and one heavily grazed site). During the 2005 field season, deer mouse population sizes were not significantly different between grazing categories. During the 2006 fall trapping session, deer mouse population sizes did not differ, but the 2006 spring trapping session deer mouse population sizes were higher on ungrazed than on heavily grazed sites even though over-winter survival rates of marked deer mice did not vary between grazing categories.

Introduction

Riparian habitats are home to a wide diversity of species and are a vital landscape feature for wildlife species, since approximately 70% of vertebrate species in a region use a riparian corridor at some point in their life cycle (Naiman et al. 1993). Livestock grazing is a widespread use of riparian habitats across North America, and results in the loss of biodiversity, decreases in population densities, and changes in community organization (Fleischner 1994). The effect of livestock grazing in riparian habitats on breeding birds has been well documented (e.g. Taylor 1986; Popotnik and Giuliano 2000), but fewer studies have examined the effect of livestock grazing in riparian habitats on small mammals. Since small mammals rely on understory vegetation for cover, and livestock grazing reduces the amount of understory vegetation in a habitat (Schulz and Leininger 1990; Holgate 2003), livestock grazing may have an effect on species richness and individual abundance of small mammals and other animals that rely on understory foliage (Ohmart 1996). Livestock grazing may also affect small mammals by contaminating water (Belsky et al. 1999), trampling individuals (Giuliano and Homyack 2004), trampling burrows and compacting soil (Hayward et al. 1997).

Small mammals such as mice, voles and shrews are an important component of the wildlife community in a riparian habitat (Medin and Clary 1989). They have significant roles in their habitats, and impact plants, soils, and other animals (as reviewed by Sieg 1988). Small mammals alter plant community composition and species distribution by grazing on plants and hoarding and consuming seeds. They increase the rate of decomposition of organic materials by adding feces and green herbage to the litter layer, and also by decreasing the sizes of the particles that are decomposing. Also, small

mammals alter soil structure and chemical composition by burrowing and adding feces and urine to the soil. Small mammals impact other animals by consuming invertebrates and other small mammals, and by being an available source of food for a variety of predators (Sieg 1988).

Previous studies of the impact of livestock grazing on small mammals have had varied results. Several studies have found that small mammal population sizes are lower in grazed habitats than in ungrazed habitats, or that biodiversity is lower in grazed sites than in ungrazed sites. For example, during one riparian grazing study in Nevada, researchers trapped almost twice as many small mammal species in the ungrazed habitat than in the grazed habitat, and trapped three times more individuals in the ungrazed habitat than in the grazed habitat (Medin and Clary 1989). Similarly, another riparian study found 1.7 times more small mammal species on ungrazed sites than on grazed riparian sites, and found 2.2 times more individuals on ungrazed than grazed sites (Giuliano and Homyack 2004). In a 10 year study of small mammal communities in ungrazed and grazed desert wetlands, Hayward et al. (1997) trapped approximately twice as many small mammals on ungrazed sites than on grazed sites. On the other hand, several other studies have found that small mammal biodiversity or population sizes are higher on grazed habitats than on ungrazed habitats. For example, Moulton et al.'s study of riparian small mammals found twice as many small mammal species in a grazed riparian habitat than in an ungrazed riparian habitat, although the total number of animals trapped was slightly higher on the ungrazed habitat (Moulton et al. 1981). Interestingly, many of the species Moulton et al. (1981) found in the grazed riparian habitat but not in the ungrazed riparian habitat were reported to be species that typically live in upland

habitats, not in riparian habitats. In another study, Schulz and Leininger (1991) trapped 7 small mammal species on both grazed and ungrazed riparian habitats, but found that the species composition was significantly different between the grazing categories. Although the direction of influence is disputed, it seems clear that cattle grazing influences the size and composition of small mammal communities.

Some have suggested that livestock grazing may limit the density of some small mammal species that prefer dense ground cover, but increases the density of mice in the genus *Peromyscus*, which are an abundant generalist species (Ohmart 1996). One riparian study found that deer mice (*Peromyscus maniculatus*) were more abundant in a grazed riparian habitat (15 individuals trapped) than in an ungrazed riparian habitat (1 individual trapped), and that western jumping mice (*Zapus princeps*), which prefer wet aspen and shrub-sedge savanna habitats with shrub cover, were more common on the ungrazed sites (22 individuals trapped) than the grazed sites (1 individual trapped) (Schulz and Leininger 1991).

Most of these studies of grazing effects on small mammal communities have taken place in the summer months, but the winter season is an energetically stressful time for small mammals, as they must regulate their body temperature while experiencing extremely cold temperatures and high wind speeds. Deer mice are active at night year-round, and are active at temperatures as low as -25°C (Conley and Porter 1986). Small mammal communities are less often studied in the winter, and the effect of livestock grazing in riparian zones on winter small mammal population sizes and biodiversity is unknown. The purpose of this study was to expand upon previous studies of breeding birds in grazed riparian habitats carried out along the Oldman River by Hurly et al.

(1998) and Holgate (2003) by examining the effect of riparian cattle grazing on small mammals. This study examined small mammal biodiversity and population sizes in ungrazed, moderately grazed, and heavily grazed riparian sites during two winter field seasons in southern Alberta. I monitored animals by live-trapping and releasing., and I compared biodiversity, population sizes and over-winter survival of deer mice between the three grazing categories.

Methods

Study Sites

I collected data in 11 cottonwood riparian sites in winter 2005 and in 12 cottonwood riparian sites in winter 2006. Of the 11 riparian sites used in 2005, 4 were ungrazed by cattle, 4 had a history of moderate cattle grazing, and 3 had a history of heavy cattle grazing. In 2006, I added a fourth heavily grazed site to the 2005 sites. All 12 sites were located along a 50 km stretch of the Oldman River between Fort Macleod and Lethbridge, Alberta. Of these 12 sites, six were city nature reserves, one site was located on federal government land, and five sites were located on private land. Sites were classified as either heavily grazed or moderately grazed based on information provided by private landowners in 2002 in response to a questionnaire about the site's grazing history, and calculations of animal-unit months (Holgate 2003). Appendix 1 shows a map of the area with the riparian study sites marked, and a further description of the study sites can be found in Chapter 2.

Small Mammal Trapping

Four circular plots, each with a 50 m radius, had previously been established in each of the twelve study sites by either Hurly et al. (1998) or Holgate (2003). I trapped small mammal in these circular plots. For the 2005 field season, I trapped small mammals between 22 January and 4 April 2005. In the 2006 field season, 'Fall Season' trapping occurred between 26 October 2005 and 18 January 2006, and 'Spring Season' trapping occurred between 23 March and 1 May 2006.

In 2005, I initially trapped in each site over two nights, but I increased trapping time to three nights at each site after trapping in four of the eleven sites resulted in few animals being caught. I carried out one extra night of trapping later in the season (total three nights of trapping) at one moderately grazed site where few animals were trapped during the initial two nights of trapping. I trapped one moderately grazed site for four nights, during extreme cold temperatures. I trapped for an additional two nights at this moderately grazed site in early April 2005 (total six nights of trapping at this site).

In 2006, 'Fall Season' trapping was over three nights at nine of the twelve study sites. Two sites (one moderately grazed, one heavily grazed) were trapped at for four nights and one moderately grazed site was trapped at for six nights when overnight temperatures were extremely low. During the 2006 'Spring Season' trapping session, I trapped over three nights at each of the twelve study sites.

In each circular plot, I set out clean Tomahawk live traps with clean polyester filament (for bedding material) and walnuts and a mixture of peanut butter and rolled oats for bait and nourishment. On the first night of trapping in a site, traps were set out in the four compass point directions from the centre of the plot, with 3 traps in each direction,

spaced 15 metres apart. On the next night of trapping, 3 traps were set out 15 metres apart in four lines rotated 45° from the compass point directions. On subsequent trapping nights the trap layout pattern alternated from compass points to rotated 45° from the compass points. Traps were spaced in this manner in an attempt to trap evenly throughout the plot. A total of 48 traps were set out each night, with 12 traps in each of the four circular plots. I set traps in late afternoon/early evening, and checked the traps early the following morning. I noted the species of trapped animal, and weighed, sexed, and tagged each animal with an individually numbered eartag (National Band and Tag Co. #1005-1 Monel tags). I also noted the location where the animal was trapped (e.g. plot 4, trap 2, along the south-east line).

Order of Trapping Across Study Sites

I performed small mammal trapping at each site according to an unbiased schedule. I established an unbiased pattern of grazing categories with each of the three grazing categories included in each sequential group of three. The sites within each grazing category were then randomly assigned to the pattern. For example, the pattern of grazing categories for 2006 Spring season trapping was: ungrazed site, moderately grazed site, heavily grazed site, heavily grazed site, ungrazed site, moderately grazed site, ungrazed site, moderately grazed site, heavily grazed site, heavily grazed site, ungrazed site, moderately grazed site. Within this pattern, the four sites in each grazing category were randomly assigned to the four positions for that grazing category. This pre-determined unbiased trapping order was occasionally adjusted by switching the order of the next two sites, as I did not trap in public parks on weekends.

Data Analysis

I estimated deer mouse population sizes for each study site using the formula:

$N = \frac{\sum(n_i * M_i^2)}{\sum(m_i * M_i)}$ (Schnabel method, Greenwood 1996) . In this formula, n_i is the total number of animals trapped on the i th trap night, M_i is the number of individuals that were tagged prior to the i th trap night (counting animals tagged during previous trapping sessions), and m_i is the number of tagged individuals captured on the the i th trap night.

I estimated population sizes for each of the 11 study sites used in the 2005 field season, and for both ‘Fall Season’ and ‘Spring Season’ trapping sessions in the 2006 field season. I used one-way ANOVAs to compare population sizes of deer mice across grazing categories. I also performed one-way ANOVAs on mouse population data with three sites (two classified as heavily grazed, one classified as moderately grazed) that are not currently grazed excluded. For data collected in 2005, I noticed a trend relating trap success to the lowest overnight temperatures. I used a regression to compare the lowest overnight temperature (collected by a weather station at the Lethbridge County Airport and retrieved from an online database at http://www.climate.weatheroffice.ec.gc.ca/climateData/canada_e.html). To reduce the effect of low overnight temperatures on trapping success, I only trapped on nights when forecasted overnight lows were above -10°C during the 2006 ‘Fall Season’ and ‘Spring Season’ trapping sessions.

I used a multiple regression test to determine whether mouse population sizes in 2005, Fall Season 2006, and Spring Season 2006 were linearly related to a measure of grazing intensity (grazing intensity was measured as animal-unit months (AUM), as determined for these sites by Holgate, 2003, and reviewed in Chapter 2) and a measure of

the percentage of ground covered by shrubs (the cumulative percent shrub cover was measured for these sites by Holgate, 2003, reviewed in Chapter 2). I carried out this multiple regression analysis for 11 of the riparian sites, as one of the heavily grazed sites in this study was not used by Holgate (2003).

For 2006 field season data, I estimated the over-winter survival of deer mice by determining the percentage of marked individuals who survived from the 'Fall Season' to be trapped again in the 'Spring Season' trapping session, and standardizing this survival rate per hundred days. For example, in one study site, 8 previously eartagged mice were trapped during the spring trapping session, a total of 35 individuals had been trapped during the fall trapping session, and 123 days passed between trapping sessions. Over-winter survival= $\left(\frac{8}{35} \times 100\right) / 123 \times 100 = 18.6\%$ survival). I used a one-way ANOVA to compare over-winter survival rates of deer mice across grazing categories.

I also estimated the percentage decrease in deer mouse population sizes over the winter months, and standardized this population decrease rate per hundred days. The percentage decrease in population sizes were estimated using the formula: $\left(1 - \left(\frac{\text{'Spring Season } N'}{\text{'Fall Season } N'}\right)\right) \times 100$ (with all population sizes estimated using the formula for N described above), and this percent decrease was then divided by the number of days that passed between the trapping sessions, and multiplied by 100 days. For example, in one study site, the 'Fall Season' population was 47, the 'Spring Season' population was 11, and 123 days passed between trapping sessions. The decrease in population= $\left(1 - \left(\frac{11}{47}\right)\right) \times 100 / 123 \times 100 = 62.3\%$. I compared the percent decreases in mouse population over the winter months between grazing categories with a one-way ANOVA.

Results

Small Mammal Species

Including recaptures, I trapped a total of 1229 small mammals in the 12 study sites over the course of this study. Three species of small mammals were trapped in these study sites: deer mice (*Peromyscus maniculatus*), masked shrew (*Sorex cinereus*), and meadow voles (*Microtus pennsylvanicus*). Deer mice accounted for 99.27% of the total number of animals I trapped, and were trapped in each of the twelve riparian study sites. Masked shrews accounted for 0.24% of the total number of animals I trapped, and were only trapped in one heavily grazed habitat in both the 2005 and 2006 field seasons. Meadow voles accounted for 0.49% of the total number of animals I trapped, and were trapped in one ungrazed habitat, two moderately grazed habitats, and one heavily grazed habitat in only the 2006 field season.

Population Sizes

The 2005 field season population sizes were not significantly different among grazing categories (ANOVA, $F_{2,8}=0.489$, $P=0.631$) (Figure 3-1). For this data set, when the three sites that are not currently grazed were removed from the analysis, the population sizes were not significantly different between grazing categories (ANOVA, $F_{2,5}=1.60$, $P=0.290$). The lowest overnight temperature recorded during a trapping night had a strong effect on the number of animals I trapped during the 2005 field season (Regression, $R^2=0.464$, $F_{1,32}=27.7$, $P<0.0001$) (Figure 3-2).

2006 'Fall Season' population sizes were not significantly different between grazing categories (ANOVA, $F_{2,9}=2.00$, $P=0.191$) (Figure 3-3). For this data set, when the three sites that are not currently grazed were removed from the analysis, the

population sizes were not significantly different between grazing categories (ANOVA, $F_{2,6}=4.27$, $P=0.0702$), but showed a trend towards a difference.

2006 'Spring Season' population sizes showed a trend towards significant differences between population sizes in the ungrazed and heavily grazed categories (ANOVA, $F_{2,9}=3.45$, $P=0.0772$) (Figure 3-4), and a means comparison test revealed that population sizes in ungrazed sites were significantly higher than population sizes in heavily grazed sites (student's t-test, $t=2.26$, $P<0.05$). For this data set, when the three sites that are not currently grazed were removed from the analysis, the population sizes were not significantly different between grazing categories (ANOVA, $F_{2,6}=1.96$, $P=0.222$).

Population Size and Shrub Cover/Grazing Intensity

The winter 2005 mouse population sizes were not related to the cumulative percent shrub cover or grazing intensity (multiple regression, overall model: $F_{1,8}=0.270$, $P=0.771$, $R^2=0.063$; percent shrub cover: $F_{1,8}=0.406$, $P=0.542$; grazing intensity: $F_{1,8}=0.129$, $P=0.729$). The 'Fall Season' 2006 mouse population sizes were not related to the cumulative percent shrub cover or grazing intensity (multiple regression, overall model: $F_{1,8}=0.659$, $P=0.543$, $R^2=0.141$; percent shrub cover: $F_{1,8}=0.093$, $P=0.769$; grazing intensity: $F_{1,8}=0.688$, $P=0.431$). The 'Spring Season' 2006 mouse population sizes were not related to the cumulative percent shrub cover or grazing intensity (multiple regression, overall model: $F_{1,8}=2.62$, $P=0.133$, $R^2=0.396$; percent shrub cover: $F_{1,8}=0.110$, $P=0.749$; grazing intensity: $F_{1,8}=2.10$, $P=0.186$).

2006 Over-winter Survival

Deer mouse over-winter survival rates did not differ among grazing categories (one-way ANOVA, $F_{2,9}=0.682$, $P=0.530$) (Figure 3-5). Percent decreases in population sizes over winter 2006 did not differ between grazing categories (ANOVA, $F_{2,9}=2.40$, $P=0.146$) (Figure 3-6).

Discussion

As only three species were trapped during this study, species richness did not vary across the grazing categories. I trapped two species in low numbers and in few study sites, with 3 masked shrews (*Sorex cinereus*) trapped in one heavily grazed site, and 6 meadow voles (*Microtus pennsylvanicus*) trapped in a different heavily grazed site, two moderately grazed sites, and one ungrazed site. The largest number of species I trapped in any one site was therefore two species, which were trapped in one ungrazed site, two moderately grazed sites, and two heavily grazed sites, with only one species (deer mice) trapped in each of the remaining seven study sites. By far, the most common small mammal in these riparian habitats was the deer mouse (*Peromyscus maniculatus*). Deer mice are the most widespread and generalized North American rodents, as they can live in a wide variety of habitats, including grasslands, deserts, forests, and swamps (Medin and Clary 1989). In a grassland study, Bock et al. (1984) found that several rodent species preferred ungrazed areas (e.g. hispid pocket mice (*Chaetodipus hispidus*), western harvest mice (*Reithrodontomys megalotis*), white-footed mice (*Peromyscus leucopus*), southern grasshopper mice (*Onychomys torridus*), and hispid cotton rats (*Sigmodon hispidus*)), and only one species (Merriam's kangaroo rat, *Dipodomys*

merriami) preferred a grazed riparian habitat to an ungrazed riparian habitat over 18 months of trapping. Although many species had a preference for ungrazed habitats, the highly generalized deer mice were equally abundant on the grazed and ungrazed sites (Bock et al. 1984). Unlike deer mice, the species that were more abundant on the ungrazed sites have been shown to prefer areas with substantial shrub and ground cover (Bock et al. 1984). Another study found that small mammal species diversity was higher on a heavily grazed upland habitat than on a lightly grazed or ungrazed upland habitat (Geier and Best 1980).

In my study, masked shrews were only trapped in one heavily grazed site, and were never trapped in any ungrazed or moderately grazed sites. Unlike the omnivorous deer mice and herbivorous meadow voles, shrews are insectivores. During the winter season, shrews eat isopods, and insect eggs, larvae, pupae, and dormant adults (Churchfield 1982). The presence of shrews on this heavily grazed site may be due to an increased abundance of insects on the heavily grazed site because of the presence of copious amounts of cattle feces (Giuliano and Homyack 2004). However, in one study of grazing effects on riparian small mammal communities, Giuliano and Homyack (2004) found no difference in the abundance of shrews on grazed and ungrazed sites, and in another study they found little difference in invertebrate communities between grazed and ungrazed sites (Homyack and Giuliano 2002). Surveys of the insect community on heavily grazed and ungrazed riparian sites are necessary to determine if insect populations vary among the grazing categories.

Deer mouse population sizes were not different between grazing categories in 2005 (Figure 3-1), or in 'Fall Season' trapping in 2006 (Figure 3-3), but ungrazed sites

had larger deer mouse populations than heavily grazed sites in ‘Spring Season’ 2006 (Figure 3-4). It is possible that population sizes in two of the heavily grazed sites and one moderately grazed site may have been similar to ungrazed site population sizes in 2005 and ‘Fall Season’ 2006 because these communities had recovered from livestock grazing after a long period of being ungrazed. Small mammal communities recover quickly after the relaxation of livestock grazing in riparian habitats. In one study, a riparian small mammal community recovered from grazing in 1-3 years (Giuliano and Homyack 2004). In the current study, two heavily grazed riparian sites had not been grazed for approximately 10 years (these sites are recently established city parks with a history of heavy cattle grazing). The small mammal communities in these sites may have recovered from the impact of livestock grazing in this period of time. Similarly, one moderately grazed site on government land had not been recently grazed, and this site may also have a recovered small mammal community. The inclusion of these not currently grazed sites in this study may account for insignificant differences in population sizes in the 2005 field season (Figure 3-1) and the ‘Fall Season’ 2006 trapping session (Figure 3-3). One test of this idea is to examine the multiple regression analysis of population size against grazing intensity and shrub cover, and see if the data points for these not currently grazed sites consistently fall above the plotted mean line, which would indicate that they were different from the average (currently grazed) sites. Also, one can look at the other moderately grazed and heavily grazed sites to see if the not currently grazed sites consistently have the highest population size. On the multiple regression analysis of population size against grazing intensity and shrub cover, these three sites did not always fall above the plotted mean line. For the winter 2005 data, the moderately grazed site and

one of these heavily grazed sites fell above the mean line. For the 'Fall Season' 2006 data, both of these heavily grazed sites fell above the mean line, but the moderately grazed site was below the mean line, and for the 'Spring Season' 2006 data, the moderately grazed site and one heavily grazed site fell above the mean line. Also, the moderately grazed site had the highest deer mouse population size of all the moderately grazed sites in both 'Fall Season' and 'Spring Season' 2006, but not in the 2005 field season, and these two heavily grazed sites had higher population sizes than the one other heavily grazed site in 2005, had higher population sizes than the two other heavily grazed sites in the 'Fall Season' 2006 trapping session, but did not have the highest heavily grazed population sizes in the 'Spring Season' 2006 field season.

To further examine if the not currently grazed sites had an impact on the non-significant population size tests, the not currently grazed sites were removed from the data analysis. When the three sites that were not currently grazed were removed from the analysis, the 2005 field season population sizes were still not significantly different between grazing categories. Similarly, when the same three sites were removed from the analysis, the Fall 2006 field seasons were still not significantly different between grazing categories, but the population sizes were closer to significance ($P=0.191$ when the sites with no current grazing were included and $P=0.0702$ when the sites with no current grazing are excluded). These tests indicate that the three sites that are not currently grazed may have small mammal populations that have recovered from the stress of cattle grazing.

Although 'Fall Season' 2006 population sizes did not differ among grazing categories (Figure 3-3), over-winter survival rates were not significantly different among

grazing categories (Figure 3-5), and the percentage by which the populations decreased over winter were not significantly different (Figure 3-6), the 'Spring Season' 2006 population sizes were significantly different between the ungrazed and heavily grazed categories (Figure 3-4). These significant differences in spring population sizes might indicate that deer mice in ungrazed sites reproduce earlier in the spring or that more of the offspring from the spring litters have survived in the ungrazed habitats. As the impact of cattle grazing in riparian areas on spring and summer deer mouse communities are not known in this area, further study of mouse populations in these seasons are required to address this issue.

Over-winter survival of small mammals is critically important for populations because breeding ceases over winter, and surviving individuals are able to reproduce in the spring (Gilbert and Krebs 1991). Over-winter survival of marked deer mice in this study did not vary among grazing categories (Figure 3-5), and ranged from 6% to 46% (Figure 3-5). These survival rates are similar to winter survival rates of mice in the *Peromyscus* genus over a long-term study in the Yukon, where values ranged from 0% to 66% over 13 winter seasons (Gilbert and Krebs 1991). My study of southern Alberta riparian habitats did not detect an effect of livestock grazing on over-winter survival of deer mice (Figure 3-5), winter deer mouse population sizes (Figure 3-1) or fall deer mouse population sizes (Figure 3-3). However, as this study did detect a negative impact of livestock grazing on early spring deer mouse population sizes (Figure 3-4), it is possible that livestock grazing in riparian areas does have a negative impact on small mammal populations throughout the winter, but this study was unable to detect this effect.

The minimum over night temperature on trapping nights during the 2005 field season had a significant effect on the number of small mammals trapped (Figure 3-2). Several other studies have found that small mammal activity levels are dependant on various weather variables, such as cloud cover (e.g. Doucet and Bider 1974), rainfall (e.g. Drickamer and Capone 1977), and the day-to-day changes in the weather (e.g. Gentry et al. 1965). Some other studies have found a similar temperature effect on trap success throughout the year. Getz (1961) found that temperatures below 0°C led to fewer captures of voles in a field site, but not in a marsh site, but low temperatures had no effect on the trapping success of a shrew species. Similarly, Vickery and Bider (1981) found decreased trap success for three small mammal species on cool nights during the summer months. The strong effect of temperature on trap success noted during the 2005 field season led to a modification to the trapping methodology used during the 2006 field season. In the 2006 field season, trapping occurred in fall and spring and trapping did not occur on nights when the forecasted lows were below -10°C throughout the 2006 field season.

I found only weak evidence of an effect of cattle grazing in riparian habitats on small mammal populations in the winter months, while the effect of grazing on winter bird populations was strong (see Chapter 2, Figure 2-3, Figure 2-4). Unlike bird populations, small mammal populations are very cyclical, and fluctuate over a period of several years (Oli and Dobson 2001). For example, Brady and Slade (2004) found a 3.5-year cycle in the population of prairie voles, and Gilbert and Krebs (1991) found a 3-4 year population cycle for northern red-backed voles. It is therefore possible that an effect of grazing on small mammal populations does exist, and a long-term study of these small

mammal populations in grazed and ungrazed riparian habitats would detect a grazing effect. It is possible that the short-term nature of this study, which occurred over only two winter field seasons, obscured any grazing effect that might exist. The spring 2006 trapping session did show a trend towards larger deer mouse populations in ungrazed habitats than in heavily grazed habitats, and it is possible that this trend would also be seen in a study of small mammal populations over 5-10 years in these riparian habitats.

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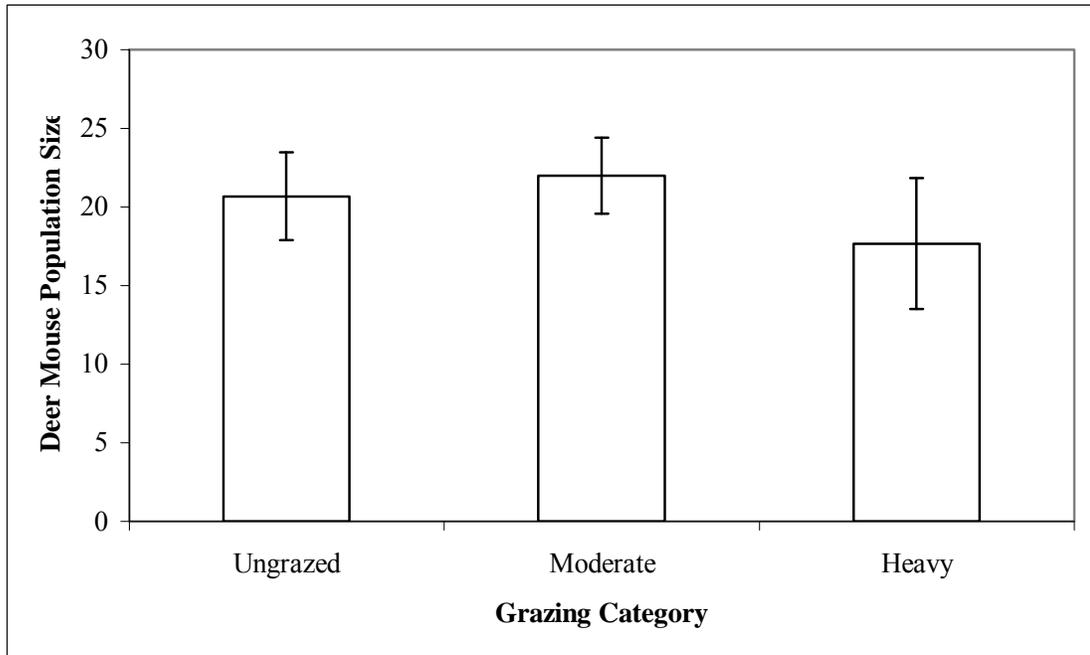


Figure 3-1. Mean (\pm S.E.) deer mouse population sizes estimated from live trapping during the 2005 field season. Data collected between 22 January and 4 April 2005. Graph shows data from four ungrazed riparian sites, four moderately grazed riparian sites, and three heavily grazed riparian sites.

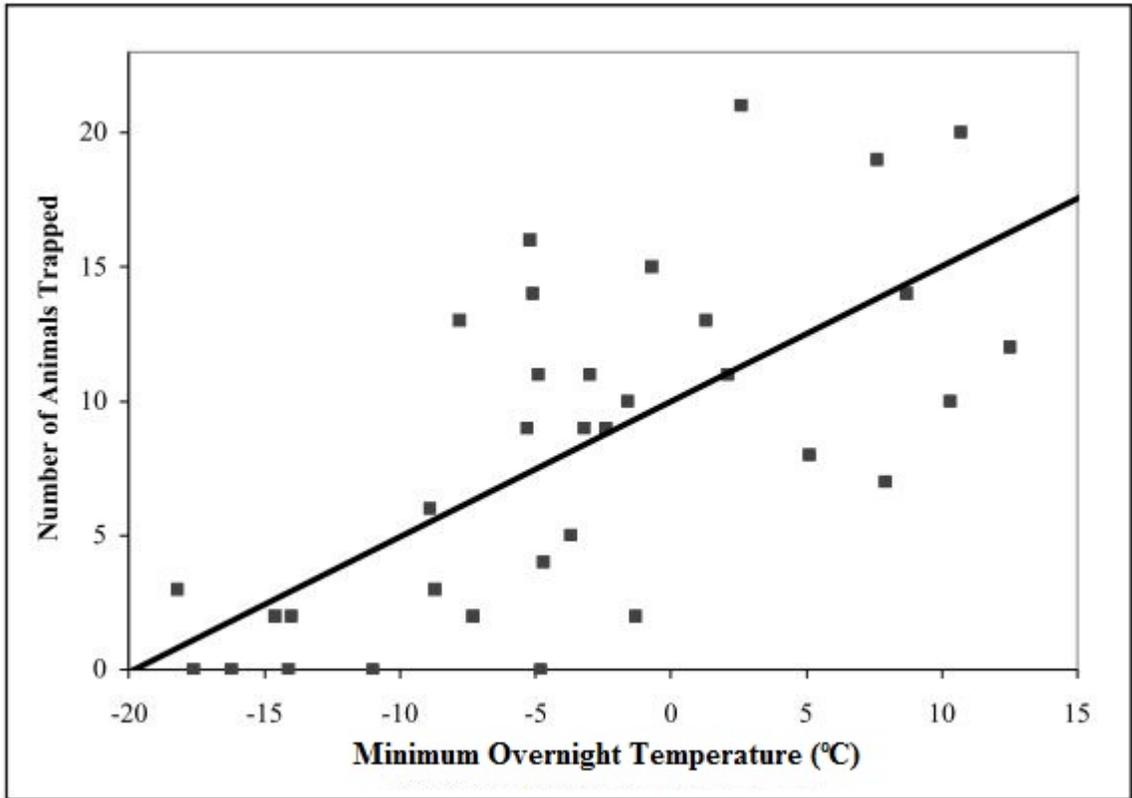


Figure 3-2. The number of individuals trapped on a trapping night during the 2005 field season was related to the lowest overnight temperature recorded in the area. Data collected between 22 January and 4 April 2005. Each data point represents one night of trapping in one site. For the line of best fit, $R^2=0.464$, and $y=0.5035x+9.975$.

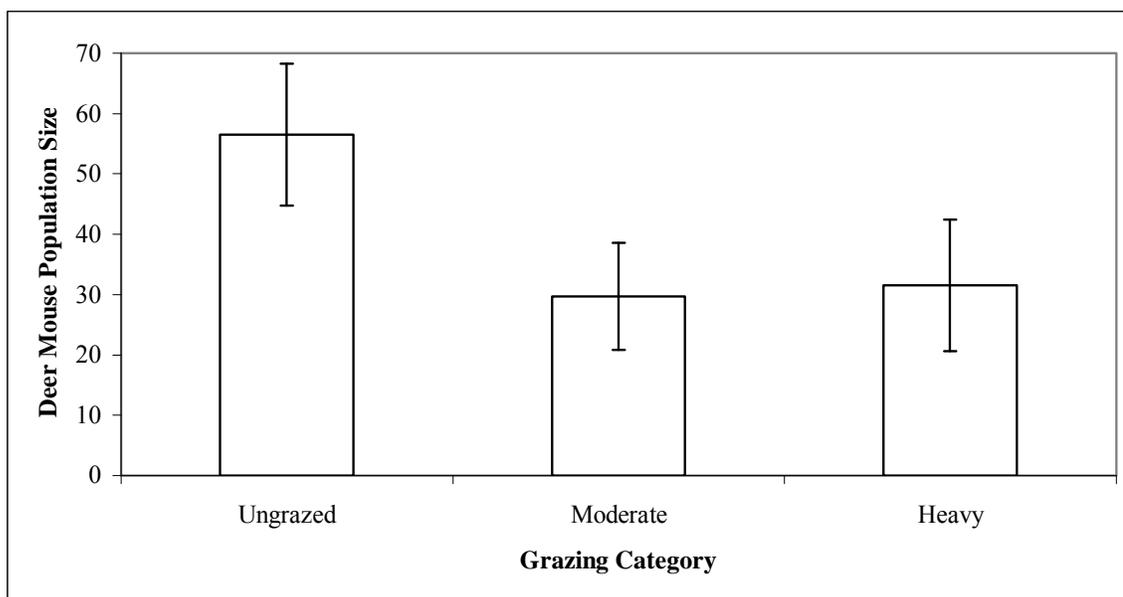


Figure 3-3. Mean (\pm S.E.) deer mouse population sizes estimated from live trapping during the 2006 'Fall Season' trapping session. Data collected between 26 October 2005 and 18 January 2006. Graph shows data from four riparian sites in each grazing category.

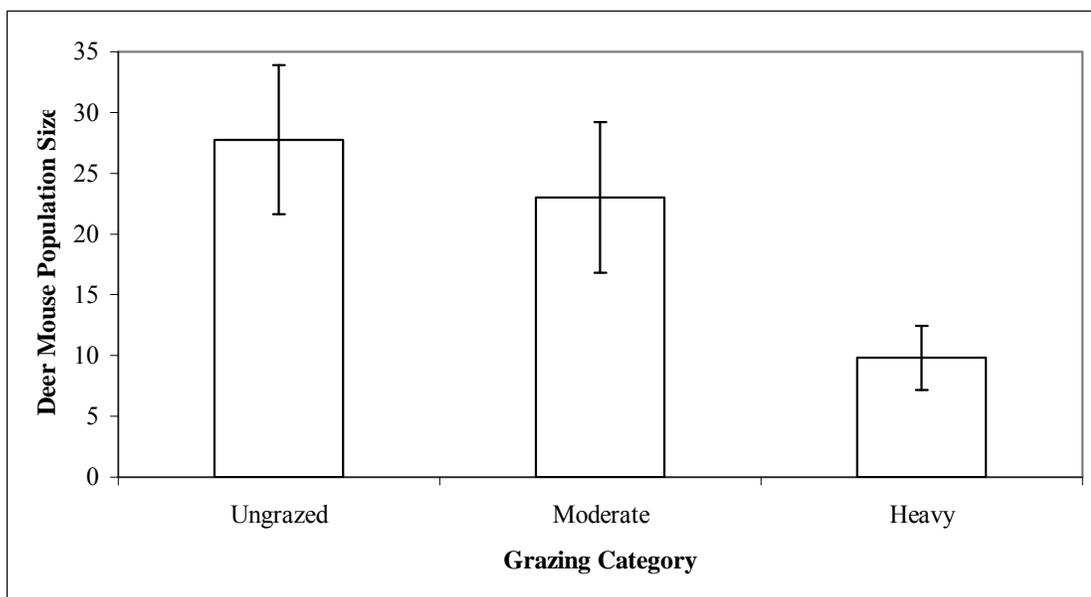


Figure 3-4. Mean (\pm S.E.) deer mouse population sizes estimated from live trapping during the 2006 ‘Spring Season’ trapping session. Data collected between 23 March and 1 May 2006. Graph shows data from four riparian sites in each grazing category.

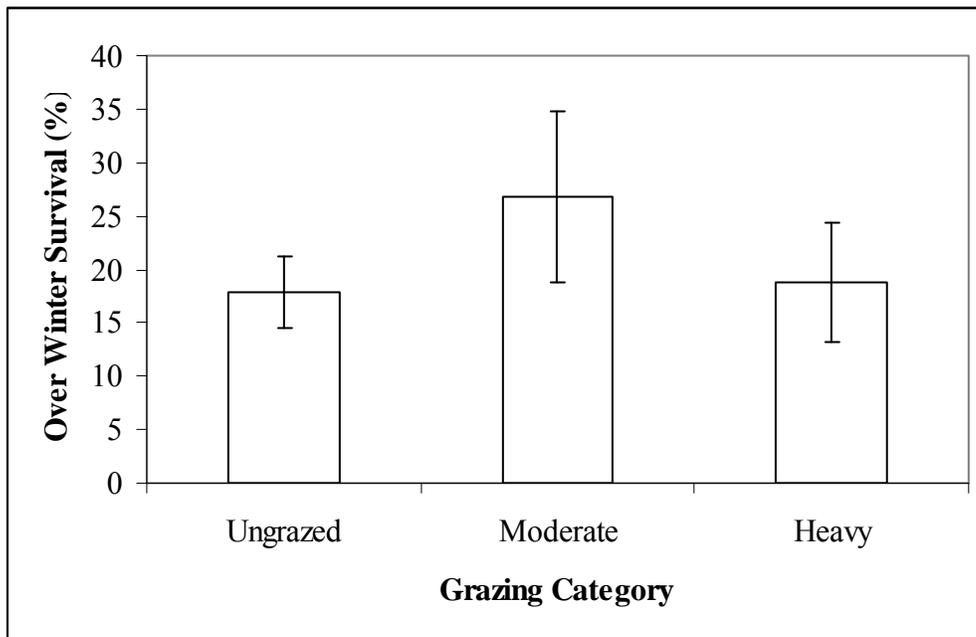


Figure 3-5. Estimated mean (\pm S.E.) over-winter survival of marked deer mice between ‘Fall Season’ 2006 and ‘Spring Season’ 2006 trapping sessions. Survival rate is standardized per 100 days. Graph shows data from four sites in each of three grazing categories.

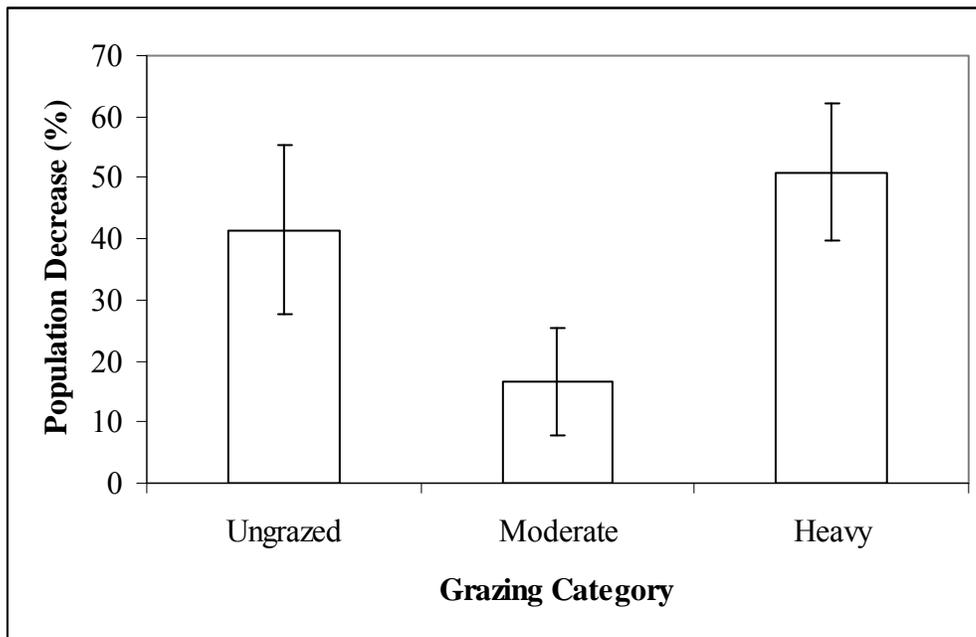


Figure 3-6. Estimated mean (\pm S.E.) percentage decrease in deer mouse population size between 'Fall Season' 2006 and 'Spring Season' 2006 trapping sessions. Percentage of population decrease is standardized per 100 days. Graph shows data from four sites in each of three grazing categories.

Chapter 4

Comparison of Microclimates between Ungrazed and Grazed Riparian Habitats and between Riparian and Upland Habitats

Abstract

Livestock grazing causes a reduction in vertical shrub structure in riparian habitats. The vertical shrub structure is an important source of shelter for birds and small mammals in riparian habitats. To determine if winter microclimates differ between grazed and ungrazed riparian habitats and if microclimates differ between riparian habitats and upland habitats, I collected microclimate data in ungrazed, moderately grazed, and heavily grazed riparian sites between 23 February and 31 March 2006. Riparian microclimate data were compared to weather data collected at a nearby upland site. Weather conditions were less severe in riparian sites than in the nearby upland sites, as air temperatures were higher and wind speeds were slower in the riparian sites. Wind speed data collected on the same day were compared between sites with a higher grazing intensity and sites with a lower grazing intensity. Wind speeds were faster in the more heavily grazed riparian sites than in sites with the relatively lower grazing intensity. The higher wind speeds in the more heavily grazed sites may account for the decreased bird biodiversity found in the more heavily grazed riparian sites in the winter.

Introduction

Climate refers to the characteristic meteorological conditions that occur at a given site, including the means, extremes and variability of weather conditions including air

temperature, soil temperature, wind speed, soil moisture, humidity, and solar radiation intensity. Meteorological conditions vary spatially, due to microscale terrain characteristics and differences in vegetation cover, and these differences in meteorological conditions in localized areas throughout a site are referred to as microclimates. For example, slope and aspect variation cause differential receipt of solar radiation in a given site (Bennie et al. 2008). Microclimate variables such as temperature, solar radiation and humidity have an impact on ecosystem processes and functions as they control plant growth, photosynthesis, respiration, net primary productivity, decomposition, succession, nutrient cycling, germination and enzyme activity (Brosofske et al. 1997). Spatial patterns of microclimate variables including air temperature, wind speed and wind direction influence horizontal and vertical patterns of foraging by Carolina chickadees and tufted titmice within woodlots during the winter season (Dolby and Grubb 1999).

Both natural disturbances and human disturbances can have significant impacts on microclimate. Fires are an example of a natural disturbance that causes an increase in wind erosion and soil temperatures, due to the elimination of vegetation (Vermeire et al. 2005). The activities of humans, including forestry and agricultural development, disturb the physical structure of ecosystems. These structural modifications result in alterations to microclimate factors such as moisture, wind, light, and temperature (Chen et al. 1999). For example, lumber harvesting in riparian habitats increases daytime air temperatures by approximately 7%, and increases daytime and nighttime soil temperatures by as much as 32% (Brosofske et al. 1997).

Some animal species are adapted to specific microclimate conditions, and changes to these microclimatic conditions could make a habitat unsuitable for those species (Brosnoff et al. 1997). In one study of foraging mountain chickadees (*Poecile gambeli*), Wachob (1996) found that the chickadees chose winter foraging sites that had higher air temperatures and lower wind speeds than surrounding available but unused foraging sites, enabling the birds to reduce their metabolic rates by an estimated 10-12% (Wachob 1996). Similarly, winter foraging Carolina chickadees (*Poecile carolinensis*) and tufted titmice (*Baeolophus bicolor*) were found to change foraging locations when wind speeds increased (Dolby and Grubb 1999). These chickadees and titmice reduced the height above ground at which they were foraging, and moved farther from the windward edge of the canopy when wind speeds increased (Dolby and Grubb 1999), thus remaining in suitable microhabitats by moving away from the windy areas. In another winter study, birds were more abundant in forests with higher solar insolation, as these habitats allowed birds to reduce energetic costs and maintain good body condition (Huertas and Díaz 2001). These studies support the hypothesis that when wildlife species are adapted to a certain microclimate condition, microclimate becomes an important factor in habitat selection by the species (Chen et al. 1999).

Livestock grazing alters a habitat's microclimate. A study performed on the Tibetan Plateau found that sites that were heavily grazed by livestock were warmer (soil temperatures were 2.0°C warmer and air temperatures were 0.7°C warmer) and 3% drier than sites with a history of light livestock grazing (Klein et al. 2005). A study of livestock grazing effects on microclimate in Eucalypt woodlands in Australia found no differences in relative humidity between grazing categories, but daily mean air and

surface temperatures were significantly higher in grazed sites than in ungrazed sites in both February-March and June. The temperature differences between grazed and ungrazed sites ranged from 0.7°C to 1.7°C (Yates et al. 2000). Soil temperature was 5.2°C higher in grazed sites than in ungrazed sites in February-March. In June, daily mean wind speeds were 0.32m/s higher at the grazed sites due to lower surface roughness (Yates et al. 2000).

Riparian habitats are home to a wider variety of wildlife species than the surrounding upland habitats (Naiman et al. 1993; Fleischner 1994), and riparian microclimates are generally different from the surrounding upland microclimates. During the summer, riparian habitats experience lower air temperatures, lower maximum air temperatures, and higher relative humidity than surrounding upland habitats (Brososke et al. 1997; Dwire and Kauffman 2003). Livestock grazing is common in riparian habitats, and grazing alters microclimates by causing soil compaction (which increases water runoff and decreases soil moisture levels), vegetation removal (which causes soil temperatures to rise and soil moisture levels to drop even further), and changes to the temperature of stream water (reviewed in Fleischner 1994). Livestock grazing in riparian habitats also reduces shrub height (Schulz and Leininger 1990; Holgate 2003). Shrubs not only provide nest locations for birds and foraging locations for birds and mammals, but they also have an important effect on microclimate, as the presence of shrubbery provides shelter from the wind, resulting in reduced wind speeds (Chen et al. 1995). Riparian habitats that have been grazed by cattle are therefore hypothesized to experience greater wind speeds than ungrazed riparian sites in the winter months due to the differing shrub structures in these habitats.

Animals with small bodies, including small mammals and birds, have high metabolic rates (McCafferty et al. 2003), and high energy requirements (Huertas and Díaz 2001). An endothermic animal maintains a relatively constant core body temperature by controlling heat gains and losses. Endothermic animals experience heat gains due to radiation and conduction, and can produce heat metabolically, while heat losses occur due to conduction, convection and radiation (Monteith and Unsworth 1990). Within a range of ambient temperatures referred to as the thermoneutral zone, an endothermic animal does not need to actively regulate its body temperature by raising its metabolic rate. For an endothermic animal to maintain a constant body temperature when environmental conditions are outside of the thermoneutral zone, the rate of heat production must increase with rising wind speeds or with falling ambient temperatures (Chappell and Holsclaw 1984).

An animal's energy budget can be significantly impacted by environmental factors such as solar radiation, air temperature, and wind speed (Wooden and Walsberg 2000), and many studies have found that individual animals choose microhabitats that have favourable wind speeds (e.g. Wachob 1996; Dolby and Grubb 1999), temperatures (e.g. Wachob 1996), and solar radiation levels (e.g. Huertas and Díaz 2001). Winter winds increase the rate of heat loss from an animal by disrupting the laminar boundary layer, the thin layer of still air surrounding the animal within which heat transfer is non-turbulent (Bakken 1991). This increases the coefficient of convection, a measure of how effectively air carries heat to and from a surface, and results in an increase in the amount of heat lost from the animal's body (Chappell 1980a; Chappell 1980b). Heat loss is a function of both wind speed and temperature. The amount of heat lost by an animal is

inversely related to temperature and directly related to wind speed. That is, as temperatures decrease or wind speeds increase, the amount of heat loss experienced by an animal increases. Wind speeds and air temperatures interact to produce a ‘wind chill temperature’, which is a measurement of the apparent temperature of the air felt by human skin (Osczevski and Bluestein 2005). The wind chill temperature effect has been determined because winds cause convective heat loss which makes air temperatures feel lower than the actual measured air temperature (Osczevski and Bluestein 2005). Although wind chill temperature calculations were developed to approximate the effect of cold temperatures and wind on human skin, and wind chill temperature calculations are deemed to be an accurate approximation of apparent temperatures only for humans, wind chill temperatures have often been used to examine the apparent winter temperatures felt by animals, including hummingbirds (Carpenter and Hixon 1988) and American woodcocks (*Scolopax minor*) (Stribling and Doerr 1985).

The winter season is an energetically stressful time for wildlife species in Alberta. Throughout the winter, animals must contend with low ambient temperatures and often with high wind speeds, while food is relatively scarce. In some areas, snow covers the ground for most of the winter season, and small mammals live in subnivean spaces at the base of the snow pack. In these air spaces in the snow, the wind speeds are lower and the temperatures are higher than they are on the snow’s surface (Taylor and Buskirk 1996), so a small mammal moving around a subnivean air space will be less energetically stressed than a small mammal living on top of a snow bank. In southern Alberta, snow rarely remains on the ground for more than a week, and the ground is bare throughout most of the winter, so subnivean habitats are not available. In the absence of subnivean

habitats, a small endothermic animal could greatly improve its energy budget by choosing foraging sites that are both warmer and less windy than other available sites. Small endothermic animals do seem to utilize foraging sites that are warmer or less windy. For example, the grass tunnels used by field voles (*Microtus agrestis*) in winter are up to 3°C warmer than the outside air temperature, and the entrances to these grass tunnels have significantly lower wind speeds than are found above the surface (McCafferty et al. 2003).

The purpose of this study was to provide quantitative data on the relationship between riparian cattle grazing and winter microclimate and to expand upon previous studies of riparian grazing and winter biodiversity. I measured microclimate near ground level, where small mammals live, at 1m above ground, where some birds and large mammals live, and at 3m above ground, where many species of birds live. I examined the hypothesis that microclimate differences between grazed and ungrazed riparian habitats and the associated differences in animal energetics are mechanisms influencing winter biodiversity differences between these grazing categories.

Methods

Study Sites

I collected microclimate data in 12 cottonwood riparian sites in winter 2006. Of these sites, 4 were ungrazed by cattle, 4 had a history of moderate cattle grazing, and 4 had a history of heavy cattle grazing. All 12 sites were located along a 50 km stretch of the Oldman River between Fort Macleod and Lethbridge, Alberta. Of these 12 sites, six were city nature reserves, one site was located on federal government land, and five sites

were located on private land. Sites were classified as heavily grazed or moderately grazed based on information provided by private landowners in 2002 in response to a questionnaire about the site's grazing history, and calculations of animal-unit months (Holgate 2003). Appendix 1 shows a map of the area with the riparian study sites marked, and a further description of the study sites can be found in Chapter 2.

I collected weather data at weather stations in two upland sites near the Oldman River. One upland site was near Lethbridge, and was close to the 7 riparian study sites that were located along the river near Lethbridge (3 ungrazed, 2 moderately grazed, and 2 heavily grazed sites). One upland site was near Fort Macleod, and was close to the 5 riparian study sites located along the Oldman River near Fort Macleod (1 ungrazed, 2 moderately grazed, and 2 heavily grazed sites). A map showing these locations is shown in Appendix 1. Weather data were collected at the upland weather stations to allow for comparison of the upland microclimates and the riparian microclimates.

Weather Data Collection

I collected microclimate data and weather station data between 23 February and 31 March 2006. I collected microclimate data with an Onset HOBO H8 Pro Series Temperature sensor (concealed in a solar radiation shield) and three Kestrel 2000 Pocket Weather Meters (portable wind speed meters). The temperature sensor and solar radiation shield were positioned on a mast at a height of 2m above ground (the standard temperature measurement height [Brosfeske et al. 1997]), and the Kestrel wind speed metres were placed at heights of 30cm (the lowest possible height for this setup), 1m, and 3m. The Kestrel wind speed meters were positioned 40cm from the mast. The

temperature sensor was programmed to record an instantaneous temperature reading every 60 seconds. Kestrel 2000 Weather Meters had to be manually read and reset.

I collected weather station data with an Onset HOBO brand Micro Station Data Logger. The sensors used were a high accuracy temperature sensor, concealed in a solar radiation shield, and three wind speed sensors. The temperature sensor and solar radiation shield were positioned on the mast at a height of 2m above the ground, and the wind speed sensors were placed at heights of 30cm, 1m, and 3m. Wind speed sensors were mounted 52cm from the mast. Lithium batteries were used to power the weather station, as they provide more accurate readings at low temperatures. I programmed the weather station to record temperature, wind speed and gust speed every 20 seconds. Weather data logged by both the HOBO Micro Station and the HOBO H8 Pro Series Temperature Sensor were offloaded using BoxCar Pro 4 (Onset brand) software.

Before microclimate data were collected each day, I set up the portable microclimate station at the upland site next to the weather station for 20 minutes to allow for calibration. After the 20-minute data collection at the base upland weather station, I transported the portable microclimate station to a study site and reassembled it in one of the four previously established 50m radius plots in the study site. The portable microclimate station was left to equilibrate with microsite environmental conditions for 5 minutes before data collection began. At each riparian site, I collected data for five minutes each at 9 locations selected by an unbiased schedule in each of the four plots. The first data collection period was at the centre of the plot. I then moved the microclimate station to eight locations around the plot. Specified distances from the centre of the plot (5m, 10m, 15m, 20m, 25m, 30m, 35m, 40m) were assigned to eight

directions from the plot centre (North, Northeast, East, Southeast, South, Southwest, West and Northwest) using an unbiased schedule. After five minutes of data collection at one location, I recorded the average wind speed, maximum wind speed, and temperature from the three Kestrel wind meters. The wind meters were then reset and moved to the next randomly selected distance in the specified direction from the plot centre. Once five minutes of weather data had been collected in each direction in the plot, I moved the portable microclimate station to the next plot, and data collection began again. In this manner, I collected five minutes of weather data in nine unbiased locations in each of four plots in each study site, for a total of 36 five-minute intervals of microclimate data collection in each riparian site on one day.

When the weather station was placed in the Fort Macleod upland location, I collected microclimate data in the five Fort Macleod-area study sites. I then moved the weather station to the Lethbridge upland location, and collected microclimate data in the seven Lethbridge-area study sites. I collected microclimate data on two separate occasions at each study site. The ‘first set’ of microclimate data were collected from 23 February – 20 March 2006 and the ‘second set’ of microclimate data were collected from 21 March – 30 March 2006.

Data Analysis

I determined daily ranges of temperature and wind speed (at 30cm, 1m and 3m heights) for each measurement date at a site. I calculated the daily mean temperature and wind speed values as the average recorded values on a day.

I compared riparian microclimates against upland base microclimates on individual days. I calculated daily mean wind speeds at ground level, 1m above ground

and 3m above ground, at both the riparian site and the upland base weather station on the same day. I also calculated daily mean temperature values for each day at both the riparian site and the upland base weather station on the same day. I used paired t-tests to determine if the daily mean temperatures or daily mean wind speeds were significantly different between the riparian site and the upland base site.

I calculated wind chill temperatures for air at 30cm above ground, 1m above ground, and 3m above ground. I calculated wind chill temperatures using the formula $WCT=13.12+0.6215*T-11.37*(V^{0.16})+0.3965*T*(V^{0.16})$, where WCT is the wind chill temperature in °C, T is the air temperature in °C, and V is the wind speed in km/hour (Osczevski and Bluestein 2005). I did not calculate wind chill temperatures if the air temperature was above +10°C or the wind speed was below 0.8 m/s. If the air temperature was above +10°C or the wind speed was below 0.8 m/s, I estimated the WCT to be equal to the ambient air temperature. The air temperature cutoff of 10°C was based on Osczevski and Bluestein's (2005) chart for wind chill temperatures. The wind speed cutoff of 0.8 m/s was based on trial calculations using different wind speeds at different temperatures. I carried out wind chill temperature calculations using the averaged five-minute wind speed recorded in each location in each plot, and the average temperature recorded during the same five minute interval by the Onset HOBO H8 Pro Series Temperature sensor concealed within the solar radiation shield.

I performed a sensitivity analysis of the wind chill temperature data by quantifying the change in wind chill caused by incrementally adjusting air temperature by 10% of the total temperature range at a constant wind speed, and by incrementally changing the wind speed by 10% of the total wind speed range at a constant air

temperature. I performed these calculations by keeping the wind speed constant at 2.8 km/h (0.8 m/s), and decreasing air temperature by 10% of the temperature range between air temperatures of -8.91°C and 15.23°C. I performed the same calculations using a constant wind speed constant of 14.04 km/h (3.9 m/s). I also performed these calculations with the air temperature constant at -8.91°C, and wind speeds increased by 10% of the wind speed range between 0 km/h (0 m/s) and 14.04 km/h (3.9 m/s), and with the air temperature constant at 15.23°C.

To determine if cattle grazing has an effect on wind speed at riparian sites, I used a Repeated Measures ANOVA test to compare the mean wind speeds recorded at three heights above ground in ungrazed, moderately grazed, and heavily grazed riparian sites. To better control for the great variation in temperatures recorded on different days, I also compared mean wind speeds between sites on the days when two different sites were monitored simultaneously. I made microclimatological comparisons between (i) heavily grazed and moderately grazed sites, (ii) heavily grazed and ungrazed sites, and (iii) moderately grazed and ungrazed sites. I performed a Repeated Measures ANOVA test on the paired wind speed data, to determine whether average wind speed differed (i) between the relative grazing levels or (ii) among the three measurement heights. The relative grazing level x measurement height interaction was included in the repeated measures ANOVA analysis.

Metabolic Rate Estimations

To estimate the different metabolic demands on deer mice and black-capped chickadees living in ungrazed, moderately grazed, and heavily grazed riparian sites, I calculated metabolic rate estimates for these animals. To estimate the metabolic rate of a

deer mouse, I used the equation $MR = (((0.116 - 0.003(T) + 0.0304(V^{0.5}))/60) * 20.1) * Wt$ (Chappell and Holsclaw 1984), where MR is the estimated metabolic rate in Watts, T is the air temperature in °C, V is the wind speed in m/s, and Wt is the mass of a deer mouse (here estimated to be 18g). To estimate the metabolic rate of black-capped chickadees, I used the equation $MR = ((40.69 - 1.85(T) + 4.5(V)) * Wt)$ (Mayer et al. 1982), where MR is the estimated metabolic rate in Watts, T is the air temperature in °C, V is the wind speed in km/h, and Wt is the mass of a chickadee (here estimated to be 13g).

Results

Riparian Microclimate Versus Upland Microclimate

On average, the riparian microclimate was warmer and less windy than that of the upland base sites. Across the ‘first set’ and ‘second set’ of microclimate data, daily mean air temperatures were 0.6°C higher in riparian sites than in base sites (Figure 4-1a). Daily mean ground level wind speeds were 0.3 m/s slower in riparian sites than in base sites (Figure 4-1b), while 1m and 3m wind speeds were respectively 1.4 m/s (Figure 4-1c), and 1.9 m/s slower in riparian sites than in base sites (Figure 4-1d).

‘First Set’ Microclimate Data

For the ‘first set’ of microclimate data collected, daily mean air temperatures were 0.7°C higher in riparian sites than in base sites (paired t-test, $t_{11}=3.17$, $p=0.0045$), daily mean ground level wind speeds were not significantly different between riparian sites and base sites (paired t-test, $t_{11}=1.61$, $p=0.0683$), daily mean wind speeds recorded 1m above ground were an average of 1.2 m/s slower in riparian sites than in base sites (paired t-test, $t_{11}=5.44$, $p<0.0001$), and daily mean wind speeds recorded 3m above ground were an

average of 1.8 m/s slower in riparian sites than in base sites (paired t-test, $t_{11}=6.62$, $p<0.0001$).

'Second Set' Microclimate Data

For the 'second set' of microclimate data collected, daily mean air temperatures were 0.6°C higher in riparian sites than in base sites (paired t-test, $t_{11}=3.95$, $p=0.0011$), daily mean ground level wind speeds were an average of 0.3 m/s slower in riparian sites than in base sites (paired t-test, $t_{11}=2.77$, $p=0.0091$), daily mean wind speeds recorded 1m above ground were an average of 1.5 m/s slower in riparian sites than in base sites (paired t-test, $t_{11}=6.45$, $p<0.0001$), and daily mean wind speeds recorded 3m above ground were an average of 2.0 m/s slower in riparian sites than in base sites (paired t-test, $t_{11}=6.79$, $p<0.0001$).

Riparian Microclimate Variation

During the riparian microclimate data collection period, temperatures at all riparian sites ranged from -8.91°C to 15.23°C (range of 24.14°C across the microclimate data collection period). On a single day at one riparian site, temperatures varied as much as 12.0°C (moderately grazed site on 17 March 2006, temperatures between -1.7°C and 10.3°C).

Average wind speeds measured at ground level at all riparian sites ranged from 0.0 to 3.9 m/s. On a single day at one riparian site, wind speeds at ground level varied as much as 3.5 m/s (moderately grazed site on 7 March 2006, wind speeds between 0.4 m/s and 3.9 m/s).

Average wind speeds measured at 1m above ground level at all riparian sites ranged from 0 m/s to 4.8 m/s. On a single day at one riparian site, wind speeds at 1m

above ground varied as much as 3.5 m/s (moderately grazed site on 7 March 2006, wind speeds between 1.3 m/s and 4.8 m/s).

Average wind speeds measured at 3m above ground level at all riparian sites ranged from 0 m/s to 6.5 m/s. On a single day at one riparian site, wind speeds at 3m above ground varied as much as 5.2 m/s (moderately grazed site on 7 March 2006, wind speeds between 0.4 m/s and 5.6 m/s). For reference, a wind speed of 5 m/s is equivalent to 18 km/h.

Wind Chill Temperature Sensitivity Analysis

I quantified the change in wind chill caused by raising air temperature by 10% of the measured temperature range while wind speed was kept at a constant value, and by raising wind speed by 10% of the measured wind speed range while air temperature was kept at a constant value. When wind speed was constant at 14.04 km/h (3.9 m/s), increasing air temperature by 10% of the temperature range resulted in wind chill temperatures increasing by 2.96°C. When air temperature was constant at -8.91°C, increasing wind speed by 10% of the wind speed range resulted in wind chill temperatures decreasing by an average of only 0.78°C.

The great range of temperatures experienced during the data collection period therefore had a large effect on the wind chill temperatures calculated, as wind chill temperature calculations are more sensitive to changes in temperature than to changes in wind speed. To decrease the influence of the highly variable day-to-day temperatures on the analysis, I examined wind speeds in these analyses rather than wind chill temperatures.

Wind Speed Variation Between Grazing Categories

I used a Repeated-Measures ANOVA test to compare wind speeds at the three heights in ungrazed, moderately grazed, and heavily grazed riparian sites. Overall mean wind speeds recorded at ground level, 1m above ground and 3m above ground were significantly different from each other (Repeated Measures ANOVA, $F_{1,11}=153$, $P<0.0001$) (Figure 4-2). Wind speeds recorded at each height were not significantly different between grazing categories (Repeated Measures ANOVA, $F_{2,9}=1.53$, $P=0.268$), but there was a slight trend towards lower wind speeds in ungrazed sites at ground level and 1m above ground (Figure 4-3).

To reduce the impact of variations in wind speed across multiple days, I used a Repeated Measures ANOVA test to compare wind speeds recorded simultaneously at two riparian sites on a single day. These comparisons were performed on microclimate data collected between 1 March and 30 March 2006. In each comparison, one of these riparian sites had a history of 'higher grazing intensity' and one site had a history of 'lower grazing intensity'. In this ANOVA, relative grazing intensity and height above ground were the two factors examined. Wind speeds recorded at each height above ground were significantly different between the relative grazing intensity levels (Repeated Measures ANOVA, $F_{1,8}=7.31$, $P=0.0270$) (Figure 4-4). Wind speeds recorded at the three heights above ground were significantly different from each other (Repeated Measures ANOVA, $F_{2,7}=9.27$, $P=0.0108$). There was no significant interaction between relative grazing intensity and height above ground (Repeated Measures ANOVA, $F_{2,7}=2.57$, $P=0.146$).

Estimation of Metabolic Rates

Estimated metabolic rates (in Watts) for deer mice and black-capped chickadees living in ungrazed, moderately grazed, and heavily grazed riparian sites are shown in Table 4-1. I calculated these values using formulas from Chappell and Holsclaw (1984) and Mayer et al. (1982), the mean wind speeds recorded in riparian sites at different grazing intensities at three heights above ground (Figure 4-3) and the mean temperature recorded in all riparian sites (Figure 4-1(a)).

Discussion

The wind chill temperature sensitivity analysis indicated that the great range of temperatures experienced during the data collection period had a large effect on the wind chill temperatures calculated. A small change in wind speed changed wind chill temperatures by only 0.78°C, but a small change in temperature changed wind chill temperatures by 2.96°C. The 24.14°C variation in temperatures recorded across the microclimate data collection period was too great to allow for comparison of wind chill temperatures across all grazing treatments. For this reason, wind speeds were used for data analysis rather than wind chill temperatures, and wind speeds on a single day in different grazing categories were compared.

Since microclimates in riparian sites were found to be less severe than the microclimates in the upland sites (Figure 4-1), there appear to be energetic benefits for an animal living in a riparian zone rather than in a nearby upland site during the winter months. In the riparian sites, the mean temperature was 0.6°C higher than the mean temperature in the upland base sites (Figure 4-1a), and the mean wind speeds at ground

level, 1m above ground and 3m above ground were each lower in the riparian sites than in the upland base sites (Figure 4-1b, Figure 4-1c, Figure 4-1d). Other studies have found similar differences between riparian and upland microclimates in the summer months, with milder temperatures (in summer, cooler temperatures) and lower wind speeds in riparian habitats than in upland habitats (e.g. Brososke et al. 1997). The differences in mean ambient temperatures and mean wind speeds recorded in this study are assumed to result in a lower average energetic cost for an animal living in a riparian site as compared to an upland site on the same day. The milder average microclimates recorded in riparian sites may be one reason why many animals prefer to utilize riparian sites rather than upland sites (e.g. Strong and Bock 1990; Knopf and Samson 1994; Fleischner 1994).

Wind speeds were faster in the more heavily grazed riparian sites than in sites with a relatively lower grazing intensity. The higher wind speeds in the more heavily grazed sites may account for the decreased bird biodiversity found in the more heavily grazed riparian sites in the winter (see Chapter 2, Figure 2-3). During the winter, an animal is likely able to select between microhabitats with different wind speeds. These animals might be more likely to choose the less windy riparian habitat rather than the windy upland habitat (Figure 4-1), and might be more likely to choose the less windy ungrazed or moderately grazed riparian habitat rather than the windy heavily grazed riparian habitat (Figure 4-3, Figure 4-4). Goldstein (1983) found that avian metabolic rates are a function of the square root of wind speed, while the mass of the bird and the ambient temperature affect the slope of this function. Webster and Weathers (1988) found that wind has a greater influence on avian metabolic rates at low temperatures than

at more moderate air temperatures. Several studies have found that birds choose to spend time in less windy sites. For example, Wachob (1996) examined the microclimates available to foraging mountain chickadees in winter, and found that the foraging sites used by the chickadees had 19.7% higher air temperatures and 9.6% lower wind speeds than available foraging sites that were unused by chickadees. The selection of foraging sites that are warmer and less windy was estimated to reduce a chickadee's metabolic rate by 10-12% (Wachob 1996). Mayer et al. (1982) calculated the metabolic rates of Carolina chickadees at different air temperatures and wind speeds, and determined that by selecting a more favourable microhabitat, a chickadee can reduce its nighttime metabolic rate by up to 50%. Selecting less windy microhabitats seems to be energetically favourable to animals in both daytime (Wachob 1996) and over night (Mayer et al. 1982).

To put the microclimate data into an animal context, I estimated the difference in the metabolic demands on deer mice and black-capped chickadees living in ungrazed, moderately grazed, and heavily grazed riparian sites. I calculated the metabolic rates (in Watts) for deer mice living at ground level, black-capped chickadees living at 1m above ground, and black-capped chickadees living at 3m above ground. I used the mean wind speeds recorded in riparian sites at different grazing intensities at three heights above ground (Figure 4-3) and the mean temperature recorded in all riparian sites (Figure 4-1(a)). To estimate the metabolic rate of a deer mouse, I used an equation from Chappell and Holsclaw (1984), and to estimate the metabolic rate of black-capped chickadees, I used an equation from Mayer et al. (1982). By living in an ungrazed riparian site rather than a heavily grazed riparian site, a deer mouse would reduce its metabolic rate by

approximately 5.1%. By living in an ungrazed riparian site rather than a heavily grazed riparian site, a black-capped chickadee 1m above ground would reduce its metabolic rate by approximately 10.2%. By living in an ungrazed riparian site rather than a heavily grazed riparian site, a black-capped chickadee 3m above ground would reduce its metabolic rate by approximately 5.8% (Table 4-1). The estimated energetic savings an animal would achieve by living in an ungrazed riparian habitat instead of a heavily grazed riparian habitat clearly illustrate the advantage that living in an ungrazed habitat would grant to an individual animal. These energetic savings may explain (i) why significantly more bird species and individuals were found in ungrazed riparian habitats than in heavily grazed riparian habitats (see Chapter 2, Figure 2-3, Figure 2-4), (ii) why small mammal populations were larger in ungrazed habitats than in heavily grazed habitats during the 'Spring Season' 2006 trapping session (see Chapter 3, Figure 3-4), and (iii) why a larger small mammal population was observed in ungrazed habitats during the 'Fall Season' 2006 trapping session (Figure 3-3). The estimated energy savings of 10.2% for a black-capped chickadee around 1m above ground are closely in line with the 10-12% energetic savings that Wachob (1996) estimated that mountain chickadees obtained by using calmer and warmer foraging sites in winter.

In addition to the harsh climatic conditions they must endure, small mammals and small birds that over-winter in cold regions experience decreased foraging time during the winter, due to the later sunrise and earlier sunset. Furthermore, these birds sometimes experience a decrease in the amount of food available when snow or ice cover the foraging substrate (Cooper 1999; Cooper 2000). For small mammals living on the ground and small birds foraging at approximately 1m above ground, the lower wind

speeds in riparian sites with a history of lower cattle grazing intensity (Figure 4-4) would be beneficial, as living in these less windy sites would confer an energetic advantage (Table 4-1). The lower metabolic rate of animals living in riparian sites with a lower grazing intensity may account for the higher number of bird species found in the ungrazed and moderately grazed riparian sites as compared to the heavily grazed sites (Figure 2-3).

Due to the negative relationship between surface roughness and wind speed at a given height, I expected to find higher wind speed in the more sparsely vegetated grazed riparian sites than in the ungrazed sites. Although the ANOVA test did not find a significant difference in wind speeds across grazing treatments, Figure 4-3 shows a trend towards lower wind speeds in sites with a lower grazing intensity at ground level and at 1m above ground. The difference in wind speed across grazing categories was expected to be minimal at 3m above ground because there is very little shrub cover at this height (Holgate 2003), so no shelter from the wind would be provided. Since recorded wind speeds were not significantly different between grazing categories at 3m above ground (Figure 4-3, Figure 4-4), no significant difference would be expected between the energetic costs of foraging in lightly grazed or heavily grazed riparian areas in birds that live at 3m above ground. The similarity of recorded wind speeds at 3m above ground in different grazing intensities might allow increased bird biodiversity in heavily grazed sites. Alternatively, it is possible that wind speeds are indeed different at 3m above ground, and this study was unable to detect this difference. Based on the recorded wind speeds at 1m and 3m above ground, I would predict that heavily grazed sites in winter contain fewer birds at approximately 1m above ground than ungrazed sites, and that the

number of birds found at approximately 3m above ground and higher are the same across these grazing categories.

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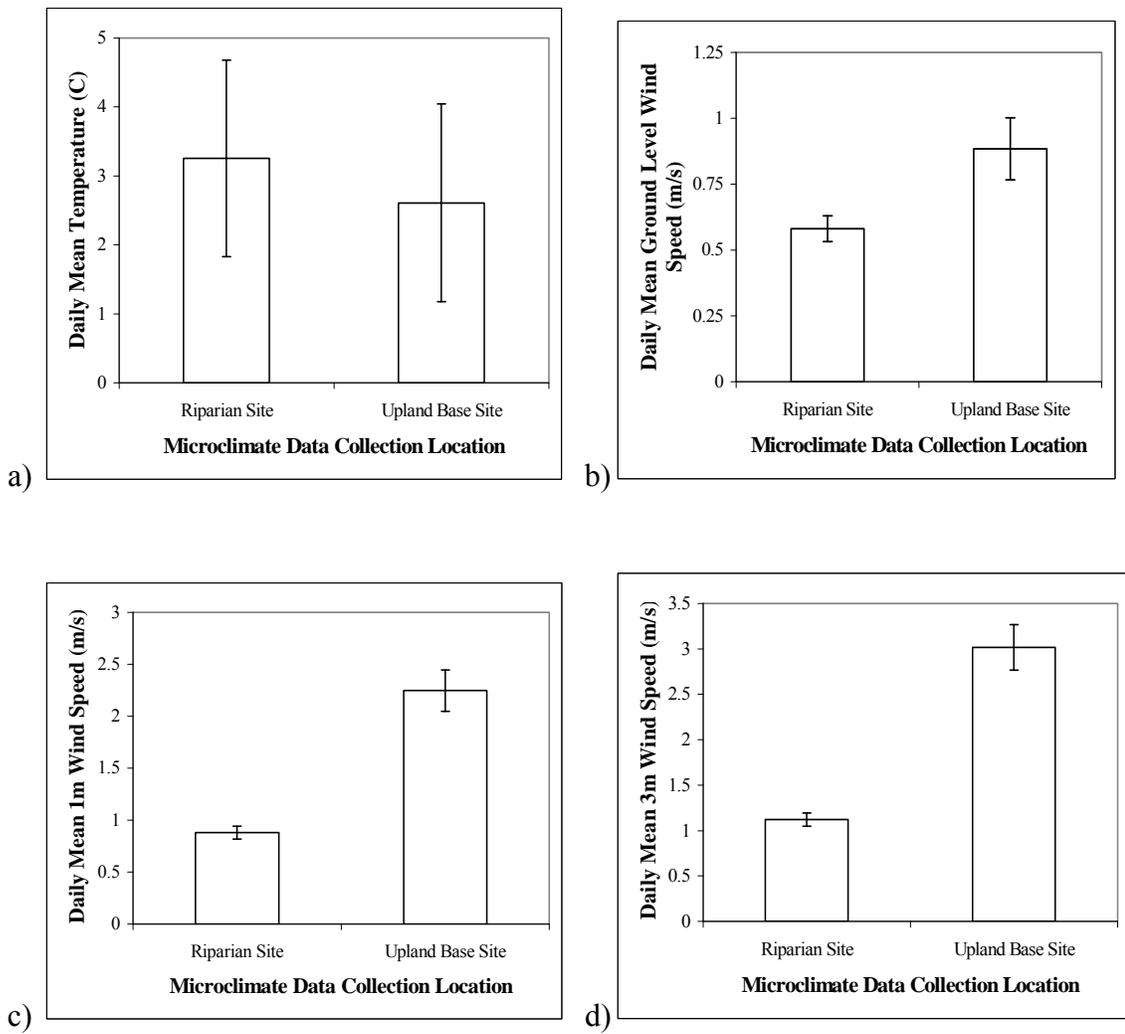


Figure 4-1. Mean (\pm S.E.) daily weather conditions at riparian sites and upland base sites. Weather data collected between 23 February and 31 March 2006. Graph a) shows average ambient temperatures ($^{\circ}$ C). Graph b) shows average wind speed recorded near ground level (m/s). Graph c) shows average wind speed recorded at 1m above ground (m/s). Graph d) shows average wind speed recorded at 3m above ground (m/s).

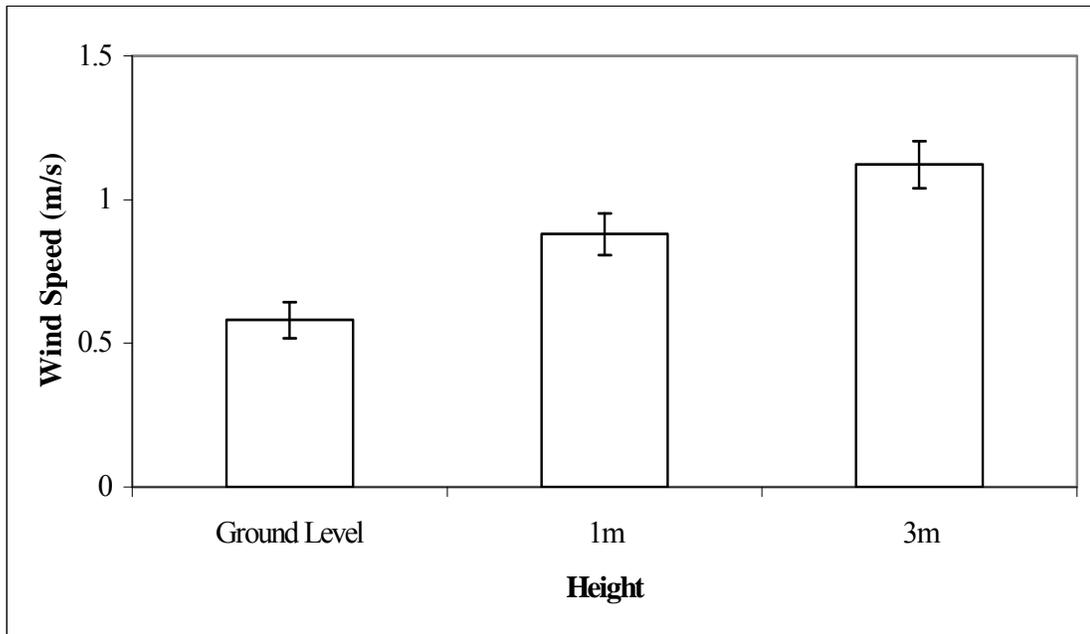


Figure 4-2. Mean (\pm S.E.) wind speed recorded at ground level, 1m above ground and 3m above ground in 12 riparian sites. Wind speed data were collected between 23 February and 31 March 2006. Four riparian sites had a history of heavy cattle grazing, four sites had a history of moderate cattle grazing, and four sites were ungrazed by cattle.

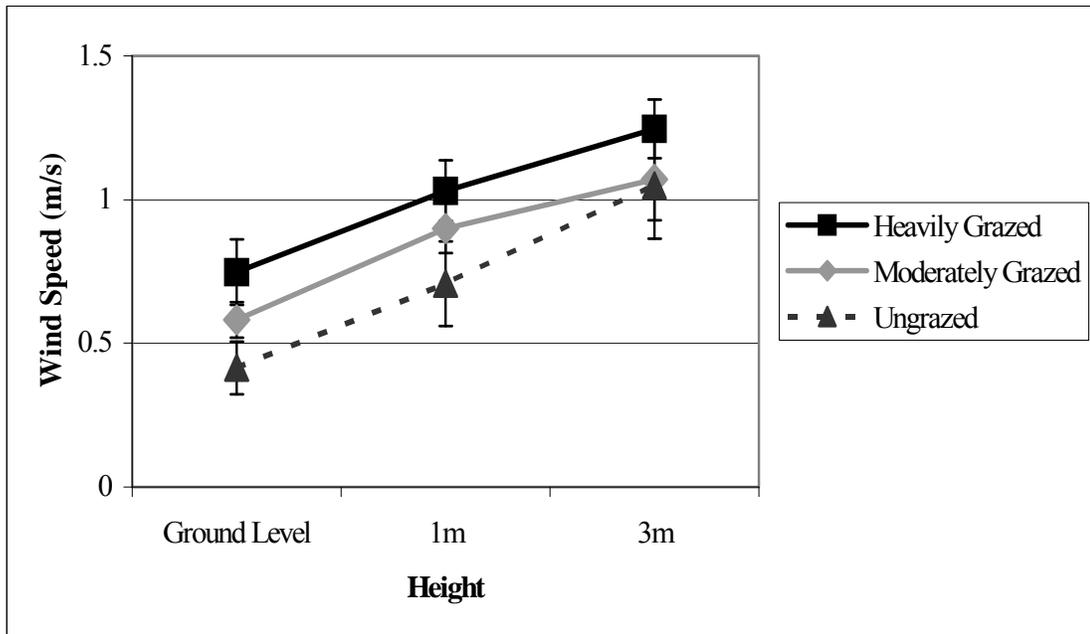


Figure 4-3. Mean (\pm S.E.) wind speed recorded in four heavily grazed sites, four moderately grazed sites and four ungrazed sites at three heights above ground. Wind speed data were collected between 23 February and 31 March 2006.

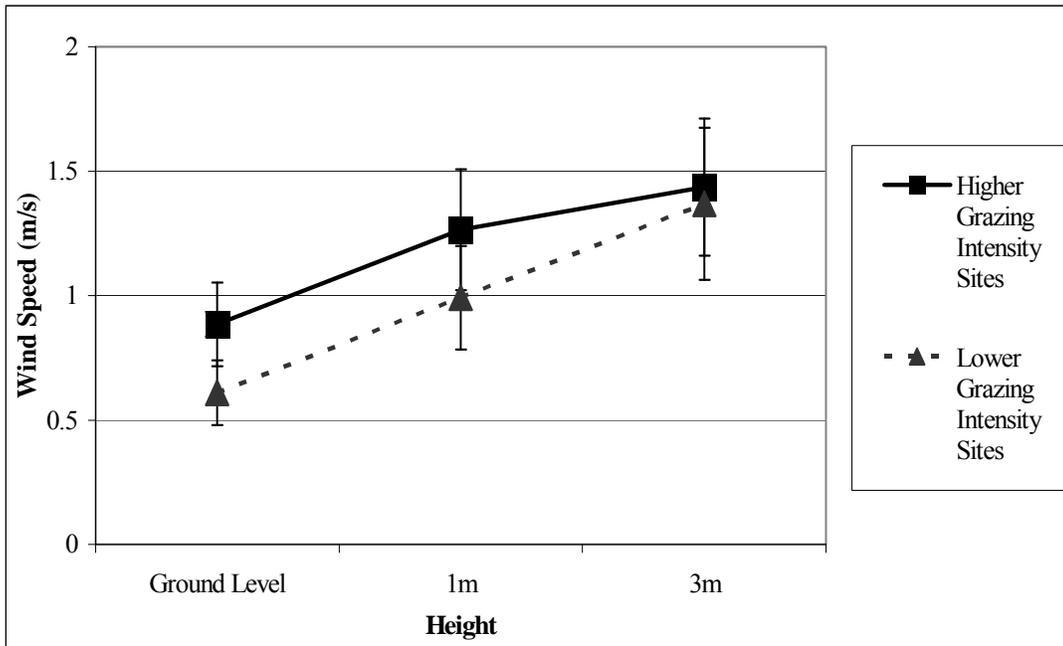


Figure 4-4. Mean (\pm S.E.) wind speed measured at three heights above ground in riparian sites that have experienced lower and higher intensities of cattle grazing. Wind speed data were collected at 10 riparian sites over nine days between 1 March 2006 and 30 March 2006.

Table 4-1. Estimated metabolic rates (in Watts) for deer mice, black-capped chickadees at 1m above ground, and black-capped chickadees at 3m above ground living in heavily grazed, moderately grazed, and ungrazed riparian sites. Deer mouse metabolic rates assume a mass of 18g, and are calculated using a formula from Chappell and Holsclaw (1984). Black-capped chickadee metabolic rates assume a mass of 13 g and are calculated using a formula from Mayer et al. (1982). Metabolic rates are calculated using mean wind speeds (Figure 4-3) and the mean riparian temperature (Figure 4-1(a)) recorded using a portable microclimate station between 23 February and 31 March 2006.

Animal	Metabolic Rates (W)		
	Grazing Intensity		
	Ungrazed	Moderately Grazed	Heavily Grazed
Deer Mouse	0.76	0.78	0.80
Black-Capped Chickadee at 1m	0.60	0.64	0.67
Black-Capped Chickadee at 3m	0.67	0.68	0.71

Chapter 5

Conclusions and General Discussion

Summary of Results

Cattle grazing in riparian areas along the Oldman River in southern Alberta had a negative impact on the number of bird species present during the winter (Chapter 2). Ungrazed riparian habitats held significantly more bird species than heavily grazed habitats during both winter 2005 and winter 2006 (see Chapter 2, Figure 2-3).

Similarly, cattle grazing had a negative impact on the number of individual birds present, as more individual birds were found in ungrazed riparian sites than in grazed riparian sites during the winter season in 2005 and 2006 (see Chapter 2, Figure 2-4). Grazing in these riparian habitats also had a negative impact on bird diversity index during the winter season in 2005 and 2006 (see Chapter 2, Shannon-Wiener Diversity Index). Ungrazed riparian sites had a higher level of avian diversity than grazed riparian sites (see Chapter 2, Shannon-Wiener Diversity Index, Figure 2-5).

To investigate the effect of cattle grazing on small mammal biodiversity in winter, deer mice, meadow voles, and masked shrew were trapped during winter 2005 and 2006. Over 99% of the animals trapped were deer mice (Chapter 3). Deer mouse population sizes were not different across grazing categories in winter 2005, or during the 2006 fall season, but there was a trend towards lower population sizes in heavily grazed sites in the 2006 spring trapping session (see Chapter 3, Figure 3-4). Deer mouse population sizes

were not related to grazing intensity (measured as animal-unit months), or cumulative shrub cover in winter 2005, fall 2006 or spring 2006.

I measured winter microclimates in riparian sites and upland sites to compare the habitats. Microclimates recorded in the riparian habitats were less severe than the microclimates recorded in the upland sites during winter 2006 (Chapter 4). The recorded air temperatures were higher in the riparian sites than in the upland sites, and the recorded wind speeds were generally lower in the riparian sites than in the upland sites (see Chapter 4, Figure 4-1).

The recorded air temperatures in the riparian sites varied greatly between days, so the recorded wind speeds were compared across grazing categories instead of comparing calculated wind chill temperatures, as slight temperature differences have a greater impact on wind chill temperatures than do slight wind speed differences. The overall mean wind speeds were different at three heights above ground, but were not different between grazing categories due to variation between days (see Figure 4-2, Figure 4-3). Wind speeds recorded at two different sites on the same day were compared across grazing categories, and wind speeds recorded in the higher grazing intensity riparian sites were faster than those recorded in the lower grazing intensity riparian sites (see Figure 4-4).

Estimated metabolic rates for black-capped chickadees and deer mice were calculated using formulae from Chappell and Holsclaw (1984) and Mayer et al. (1982). These calculations used the mean wind speeds recorded at three heights above ground in the riparian sites. For all three situations (deer mouse, black-capped chickadee at 1m

above ground, black-capped chickadee at 3m above ground), the estimated metabolic rates were higher at higher grazing intensities than at ungrazed sites (see Table 4-1).

Discussion

Cattle grazing in riparian habitats in southern Alberta was previously found to have a negative impact on the vegetation structure of the habitat (Holgate 2003), on breeding bird populations (Hurly et al. 1998; Holgate 2003), and on populations of migratory fall birds (Saunders and Hurly 2000). The current study indicates that cattle grazing in riparian habitats also has a negative impact on winter bird biodiversity in this area of southern Alberta (Figure 2-3, Figure 2-4, Figure 2-5), but that winter small mammal biodiversity is not strongly affected by cattle grazing in these riparian habitats (Figure 3-3, Figure 3-4).

This survey of winter bird biodiversity supports previous studies of both breeding bird communities (Hurly et al. 1998; Holgate 2003), and fall bird communities (Saunders and Hurly 2000) in grazed riparian habitats in this area of southern Alberta. Similar to the fall study, the results in this winter study are less robust than the results found by the breeding bird surveys. During the winter, there are fewer resident birds than during any other time of year, and resident winter birds are less territorial than breeding birds. For these reasons, the weaker grazing effect measured during the winter months is unsurprising.

While deer mouse populations were not significantly different between grazing categories in either 2005 or fall 2006, I did find a trend towards larger deer mouse populations in ungrazed sites than in grazed sites during the spring 2006 trapping session,

and the percentage of deer mice that died over the winter months was higher in the heavily grazed sites than in the ungrazed sites. These slight effects of cattle grazing on winter deer mouse populations indicate that it is possible that winter deer mouse populations are more affected by cattle grazing than this study was able to detect. It is possible that some of these populations had recovered from the effect of cattle grazing after a few years, as has been found to happen in riparian communities after the relaxation of cattle grazing (Giuliano and Homyack 2004). Alternatively, since small mammal populations typically experience a 3-4 year population cycle (Gilbert and Krebs 1991; Brady and Slade 2004), with the amplitude of the population fluctuation sometimes greater than two orders of magnitude (Oli and Dobson 2001), it is possible that any effect of cattle grazing may have been obscured by the fluctuating nature of small mammal populations in this two-year study.

To assess the mechanisms behind the diversity differences seen during winter, I tested the hypothesis that winter microclimate differences exist between riparian sites with varied cattle grazing histories. I found that the mean wind speed was faster at sites with a higher grazing intensity than at sites with a lower grazing intensity (Figure 4-4). A difference in wind speeds at different grazing intensities was predicted due to the greatly reduced vertical shrub structure in the heavily grazed sites (see Figure 2-2, data collected by Holgate 2003). The measured difference in mean wind speed at the different grazing intensities (Figure 4-3) translated into differences in the estimated metabolic rates for hypothetical animals living at different grazing intensities. The calculated metabolic rate for an animal living in a heavily grazed site was 5.0% to 10.4% higher than that of the same animal living in an ungrazed site, and 2.5% to 4.5% higher than that of an animal

living in a moderately grazed site (Table 4.1). The calculated metabolic rate for an animal living in a moderately grazed site was 1.5% to 6.3% higher than that of an animal living in an ungrazed site (Table 4-1). This predicted difference in metabolic cost at different grazing intensities may explain: (i) the lower bird biodiversity found in the grazed riparian sites than in the ungrazed sites (Figure 2-3, Figure 2-4), and (ii) the trend towards smaller deer mouse populations in grazed sites than in ungrazed sites recorded in the spring 2006 trapping session (Figure 3-4).

Another hypothesized mechanism for avian biodiversity differences at different grazing intensities is that there are differences in the amount of food energy available at the different grazing categories during winter. Holgate (2003) examined food-provisioning rates in breeding house wrens in southern Alberta, and found that house wrens in ungrazed riparian habitats delivered food to nestlings at a higher rate than did house wrens in grazed riparian habitats. This suggests that in these riparian sites during the summer, less food is available in grazed riparian sites than is available in ungrazed riparian sites. According to the Ideal Free Distribution model, animals are knowledgeable about food availability differences between habitats, animals are free to move to the habitat that holds the most food, and individual birds will disperse throughout habitat patches in a manner that is proportional to the amount of food available in each (Fretwell and Lucas 1970). The Ideal Free Distribution model would suggest that Holgate (2003) found more individual birds in ungrazed riparian habitats than in grazed habitats because there is a greater amount of food available in ungrazed sites. However, breeding birds are highly territorial during the summer, and a male bird sings to keep other males out of his territory. In this way, breeding birds are not 'free' to

move to the habitat that holds the greatest amount of food. In the winter, resident birds do not sing and are not territorial, so winter birds seem to be more free to move into the habitats that contain the most food. For this reason, it is possible that the Ideal Free Distribution model could be used to explain why more bird species (Figure 2-3) and more individual birds (Figure 2-4) were found in ungrazed habitats during the winter if more food is available at ungrazed sites than at heavily grazed sites during the winter.

Of the twelve riparian sites used in this study, six were public parks. These parks experienced substantially more foot traffic than the private sites, which could have an effect on the biodiversity found in these sites. It is possible that small mammal populations and bird populations could be smaller in busier areas due to the larger volume of people (and sometimes pets) traveling through these popular parks. Because the only ungrazed riparian habitats available in this area are riparian parks, all four of my ungrazed sites were public parks, and these ungrazed sites had higher bird biodiversity levels than did the grazed sites (Figure 2-3, Figure 2-4, Figure 2-5). Two of my heavily grazed sites were also public parks that also experience higher foot traffic, and biodiversity levels were not markedly different in these public heavily grazed sites than in the private heavily grazed sites.

Although cattle grazing is not a natural process in these riparian habitats, these habitats have long been grazed by various wildlife species, and the ‘ungrazed’ habitats in this study are certainly still grazed by some of these wildlife species. Mule deer and white-tailed deer were frequently seen in habitats at each grazing intensity. Deer seemed to be most common in the ungrazed sites, which is unsurprising given that the ungrazed sites used in this study were city parks where hunting is banned, and many of the grazed

sites were privately owned and likely hunted. Although deer were common in the ungrazed habitats, the intensity of grazing by deer and other wildlife species was much lighter than the intensity of cattle grazing in the grazed riparian sites. According to the United States Department of Agriculture, one 1000 lb cow with a calf (1 animal-unit month) is equivalent to either five mature mule deer, or 6.6 mature white-tailed deer, 1.6 elk, or 5.0 antelope (Table 5-1, from USDA 1997). A small group of ten deer grazing in one hectare of an ungrazed habitat is therefore less damaging to the habitat than is three cow-calf pairs grazing in one hectare of an adjacent habitat, especially when one considers that deer are able to leave the habitat, and fences are erected to keep cattle in the grazed habitats.

Natural ecosystems which have high levels of biodiversity are: (i) more resistant to disturbances such as fire, grazing, soil disturbances, nutrient inputs, trampling and fragmentation (Hobbs and Huenneke 1992) than communities with lower levels of biodiversity, (ii) are more resilient (i.e. they recover more fully from these disturbances) than communities with lower biodiversity (Tilman and Downing 1994), and (iii) have more stable and sustainable ecosystem services (e.g. higher annual biomass production) than altered ecosystems which have lower biodiversity (Tilman et al. 2006). An example of the increased resistance to disturbance due to high levels of avian species diversity can be seen in the periodic outbreaks of a New Brunswick insect that destroys boreal fir forest. The spruce budworm (*Choristoneura fumiferana*) outbreaks are controlled by interactions between the volume of trees that are susceptible to the insect, the densities of avian predators, and changing weather conditions (Peterson et al. 1998). At least 31 bird species prey on the budworms, and different bird species eat budworms from different

areas of a tree, with small birds eating budworms from the needles, and larger birds eating budworms from the branches. A variety of bird species with different foraging strategies therefore decreases budworm densities and by doing so, reduces the frequency of budworm outbreaks (Holling 1992; Peterson et al. 1998), resulting in an ecosystem that is more resistant to this type of disturbance.

The greater stability of ecosystems with higher biodiversity is referred to as the ‘insurance value of biodiversity’, as high levels of biodiversity can be seen as insurance against ecological uncertainties such as droughts, fires, and floods (Béné and Doyen 2008). In the case of agricultural lands, higher levels of wildlife biodiversity may also be related to higher economic yields for the producer. One study of grazed landscapes in England found that the estimated value of the yield obtained from the grazed land was low when plant biodiversity was low, and the value of the yields was higher at moderate and high levels of plant biodiversity (Hodgson et al. 2005). This finding is represented by the fitted line shown in Figure 5-1 (adapted from Hodgson et al. 2005). Since intermediate levels of grazing by cattle lead to an increased diversity of plant species (Milchunas et al. 1988; Olff and Ritchie 1998) and heavy levels of grazing lead to a decreased diversity of plant species (Milchunas et al. 1988; Fleischner 1994), the x-axis of Figure 5-1 can also represent decreasing intensity of cattle grazing.

If this model of yield value and plant biodiversity also holds true for grazed riparian habitats, a moderately grazed riparian area with a high diversity of plant species will be more resistant to disturbances (such as harsh climatic conditions, drought, flooding, and soil disturbance [Hobbs and Huenneke 1992]), more resilient (Tilman and Downing 1994), and more stable than a heavily grazed riparian area, and the monetary

yield taken from the moderately grazed habitats will be higher than the monetary yield taken from a heavily grazed habitat.

Because of this increased productivity, a well functioning moderately grazed riparian area would be a more valuable resource to the producer than a poorly functioning heavily grazed riparian area. To more closely examine this hypothesis, I modeled the amount of aboveground net primary productivity (ANPP) that would be consumed by cattle in heavily grazed, moderately grazed and lightly grazed sites. I imagined that sites in each grazing treatment were initially ungrazed by cattle, with an ANPP of $2570 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, the average ANPP for grassland sites (Milchunas and Lauenroth 1993).

Although some studies claim that cattle grazing at 50% of ANPP is sustainable and allows for the maintenance of range condition (e.g. Biondini et al. 1998), Holechek et al. (1999) reviewed 25 long-term grazing studies and found that heavy grazing (57% use of forage), moderate grazing (43% use of forage) and even light grazing (32% use of forage) resulted in decreases in ANPP (Table 5-2). I assumed that the ANPP of these grazed sites would gradually decrease to the mean ANPP for sites in each grazing treatment. These mean ANPP rates were calculated from 25 long-term studies of grazing intensity on vegetation and livestock (Holechek et al. 1999). The mean ANPP for each grazing treatment is shown in Table 5-2. I also assumed that the decrease to the mean ANPP would occur over a period of 5 years, based on a grazing simulation study which found a large decrease in ANPP after 2 years of simulated grazing at 7.5 AUM (Clark and Kinney 2002). As the heavy grazing treatment in my study was closer to 4 AUM, I conservatively estimated that the decline in ANPP would occur over 5 years of grazing. I then calculated the amount of ANPP consumed by cattle in each grazing treatment, where

32% of the available forage was consumed by cattle in lightly grazed sites, 43% of the available forage was consumed by cattle in moderately grazed sites, and 57% of the available forage was consumed by cattle in heavily grazed sites (Holechek et al. 1999).

The result of this model is shown in Figure 5-2a. In each grazing treatment, the forage consumption decreases over the first five years of grazing. The forage consumption rate drops most dramatically in the heavily grazed site (Figure 5-2a). After the initial drop in the mass of forage consumed per hectare, the annual amount of forage consumed by cattle is lowest in the lightly grazed site, and the annual forage consumption is slightly higher in the heavily grazed site than in the moderately grazed site (Figure 5-2a).

I then determined the amount of forage that is available to be consumed by each cow-calf pair (1 AUM) in each grazing treatment. I assigned grazing intensity values as follows: the lightly grazed site was grazed at 1.25 AUM, the moderately grazed site was grazed at 2.5 AUM, and the heavily grazed site was grazed at 4 AUM (these AUM values are similar to the AUM values for the moderately grazed and heavily grazed sites in my study [see Figure 2-1]). I divided the amount of forage consumed by cattle (i.e. $\text{kg}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$ values from Figure 5-2a) by the number of cow-calf pairs that would be grazing in that grazing treatment. I therefore divided the light grazing values in Figure 5-2a by 1.25, the moderate values by 2.5, and the heavy values by 4 to find the amount of food eaten by one AUM (one cow-calf pair) in each grazing treatment. The result is shown in Figure 5-2b. Over twice as much food is available to a single cow-calf pair on the lightly grazed land than on the heavily grazed land. Similarly, approximately 50% more food is

available to a single cow-calf pair on the moderately grazed land than on the heavily grazed land (Figure 5-2b).

I also determined the number of days that one hectare of grazed land can support one cow-calf pair. I divided the amount of forage that is consumed by one cow-calf pair (i.e. values in Figure 5-2b) by 11.8 kg, the amount of forage consumed by one cow-calf pair in one day (Table 5-1). The result of this manipulation is shown in Figure 5-2c. In each year, the lightly grazed site is able to feed the cow-calf pair for a longer period of time than the moderately or heavily grazed sites, and the heavily grazed site can feed the cow-calf pair for the shortest period of time. After 5 years, one hectare of lightly grazed land can feed the cow-calf pair for over twice as long as the heavily grazed land, and one hectare of moderately grazed land can feed the cow-calf pair for a full week longer than the hectare of heavily grazed land (Figure 5-2c).

The amount of ANPP consumed by cattle can roughly translate to economic yield, because the mass of food consumed by cattle is related to the amount of weight that cattle gain. Although the highest consumption of ANPP is by cattle in the heavy grazing treatment, this food is divided among a larger number of animals, so the mass gained by individual cattle in this treatment would be lower than in the moderate or light grazing treatments. Indeed, Holechek et al. (1999) report that the average weight gain per steer and the average calf weaning weight are lowest in heavily grazed sites (Table 5-2). Due to these differences in weight gain across grazing categories, Holechek et al. (1999) found that the net returns per animal and the net returns per acre were lowest for heavily grazed sites (Table 5-2). The net weight gains per animal and the monetary return per animal were highest in the lightly grazed sites, but the net monetary return per acre was

highest in moderately grazed sites (Table 5-2), indicating that moderate cattle grazing is more economically viable than is light or heavy cattle grazing. This economic argument for moderate grazing of riparian lands might be more compelling to producers than any ecological arguments for moderate grazing, and this economic argument might encourage producers to carefully monitor cattle grazing in riparian habitats to maintain it at light to moderate intensities so that wildlife biodiversity can be maintained.

Alternative grazing strategies are another option for reducing the impact of grazing on wildlife biodiversity. Alternative grazing strategies for riparian habitats aim to restore riparian habitats to more natural conditions. A functioning riparian habitat will have improved habitats for aquatic animals and other wildlife species, more stable stream channels, and improved water quality, and producers will benefit from having a more stable and productive forage supply for livestock, as well functioning riparian zones produce twice as much forage as poorly functioning, heavily modified riparian zones (Fitch and Adams 1998). Alternative grazing strategies to consider include: (i) controlling the distribution of animals and their access to water, (ii) controlling the seasons when grazing occurs, (iii) controlling the intensity of grazing, (iv) rotation of grazing area with planned rest years, and (v) total exclusion of grazing (Fitch and Adams 1998).

In the last several decades, scientists in several fields have attempted to develop models to determine the highest sustainable level of harvesting of a renewable resource. Maximum sustainable yield models were developed in the fisheries field (e.g. Deriso 1982), forestry field (e.g. Vanclay 1994) and agricultural field (e.g. Biswas and Benhi 1996; Bindraban et al. 2000). Maximum sustainable yield models for resource harvesting

are developed with the knowledge that: (i) a population's growth rate is determined by its size, (ii) the harvesting of the resource is equal to the growth rate at a steady state, and (iii) the resource has a reproductive surplus that can be harvested without driving the population to extinction (Nazarov and Li 2005). Unfortunately, these maximum sustainable yield models are imperfect, and the harvested populations often end up being over-exploited (Rosenberg et al. 1993; Hilborn 2007). More recently, models for harvesting an ecologically sustainable yield, defined as the yield that the ecosystem as a whole can sustain without losing functionality, have been proposed (Zabel et al. 2003). Ecologically sustainable yield models are more long-term, are intended to include careful monitoring of the ecosystem, and focus on the whole ecosystem by taking into account interactions between the species being harvested and numerous other species (Zabel et al. 2003). Although the implementation of the maximum sustainable yield models has been less than perfect in the past, the idea that a resource should not be fully exploited all the time is a good idea which can be applied to cattle grazing in riparian habitats. Since heavy grazing results in a significant decrease in the vertical shrub structure (see Figure 2-2, taken from Holgate 2003) and a decrease in plant species biodiversity (Milchunas et al. 1988), heavily grazing a pasture area results in a poorly functioning, heavily modified ecosystem. However, moderate grazing by cattle causes a less severe decrease to the vertical shrub structure (see Figure 2-2, from Holgate 2003), and increases plant species biodiversity (Olf and Ritchie 1998), so the negative impact on ecosystem health is lessened. Cattle grazing at moderate intensities (approximately 2 animal-unit months per hectare, see Figure 2-1) appears to be a more long term sustainable production method, and moderate cattle grazing levels may be a 'sustainable yield'.

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Table 5-1. The animal-unit month equivalent values for different animal species, ages, and sexes, and the mass of forage that each individual animal consumes in one day. These values are for mature animals unless otherwise noted. Table from USDA 1997.

Animal Description	Equivalent Animal-Unit Month	Forage Consumed per day (kg)
Cow with calf	1.00	11.8
Cow	0.92	10.9
1 year old cow	0.60	7.1
Bull	1.35	15.9
Horse	1.25	14.7
White-tailed deer	0.15	1.8
Mule deer	0.20	2.4
Elk	0.60	7.1
Bison	1.00	11.8
Antelope	0.20	2.4

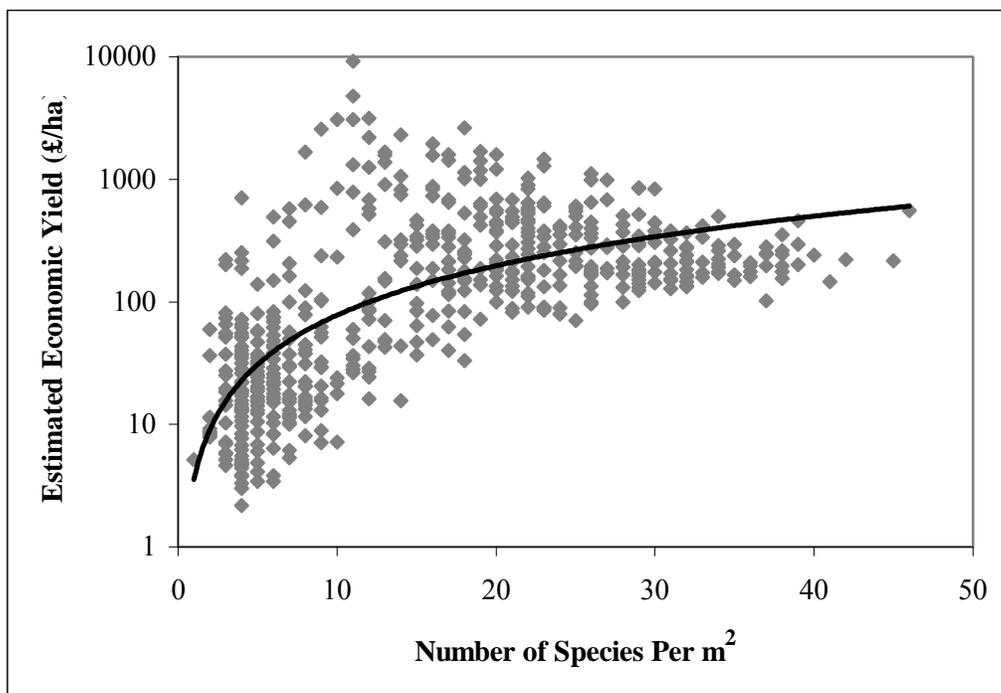
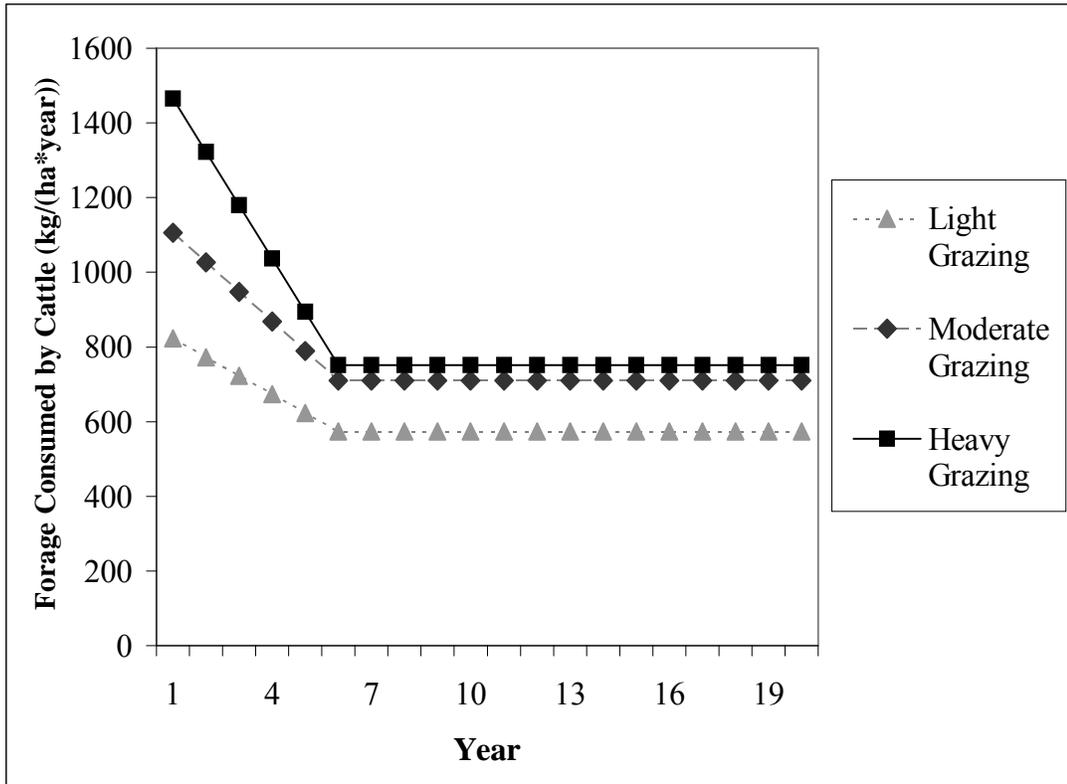


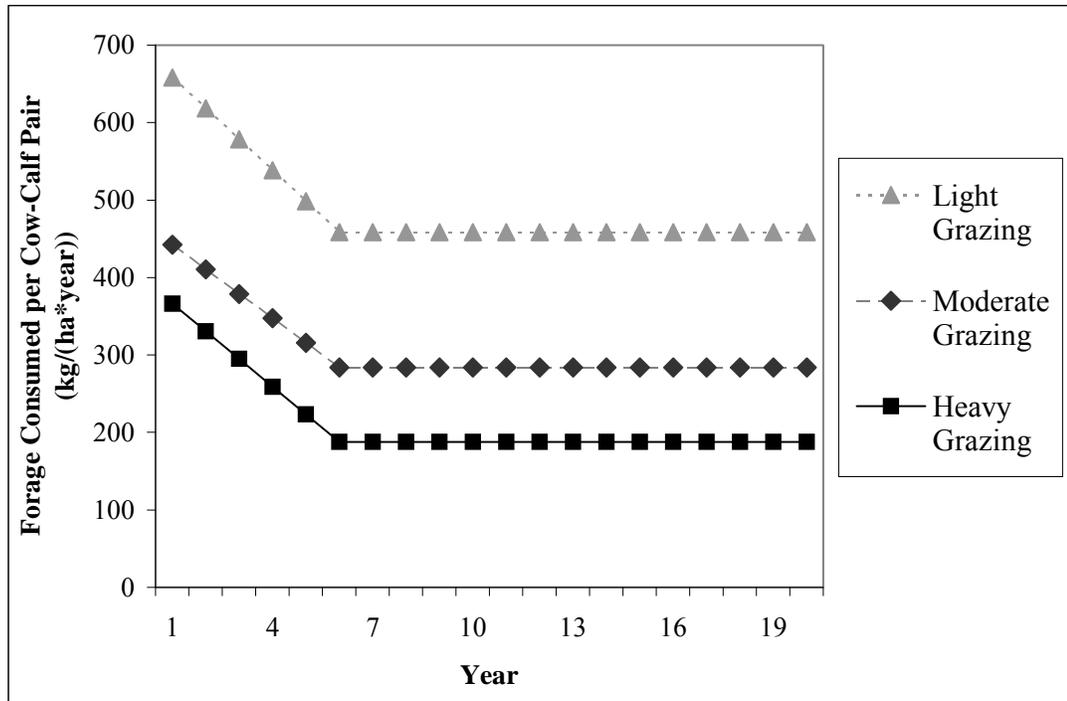
Figure 5-1. The relationship between plant biodiversity and economic yield on grazed upland habitat in England. The fitted line follows the equation $y=3.5366x^{1.3422}$, ($F_{1,544}=467.2$, $P<0.0001$, $R^2=0.462$). Note that the y-axis uses a logarithmic scale. Figure modified from Hodgson et al. 2005.

Table 5-2. Summary of data from long-term studies of the effects of grazing intensity on vegetation and livestock production. Only studies that examined light, moderate, and heavy grazing were included. Table modified from Holechek et al. 1999.

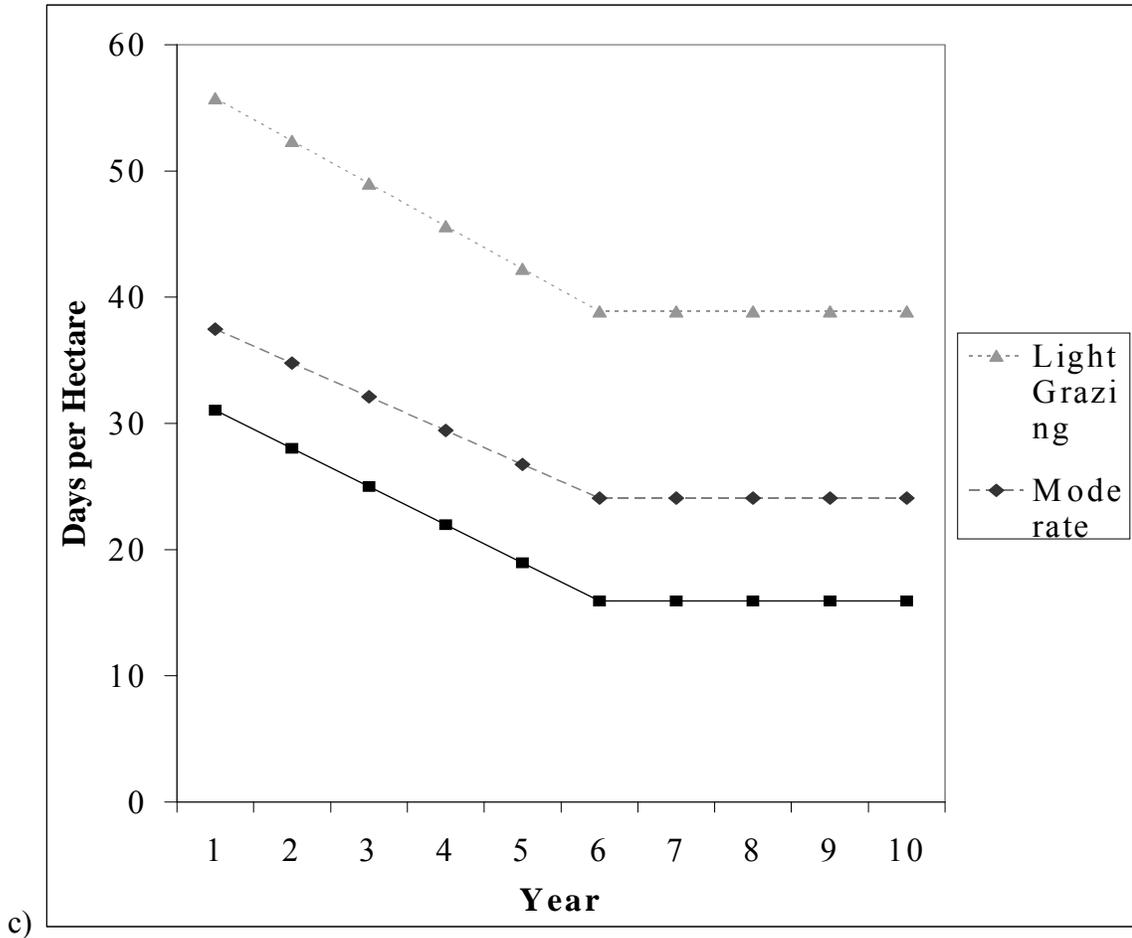
	Grazing Intensity		
	Light	Moderate	Heavy
Mean ANPP (kg•ha ⁻¹ •year ⁻¹)	1792	1652	1318
Mean % use of forage	32	43	57
Mean calf crop (%)	82	79	72
Mean calf weaning weight (kg)	196	188	173
Mean weight gain per steer (kg)	103	92	72
Net returns per animal (US \$)	58.89	51.57	38.06
Net returns per hectare (US \$)	5.86	6.45	3.19



a)



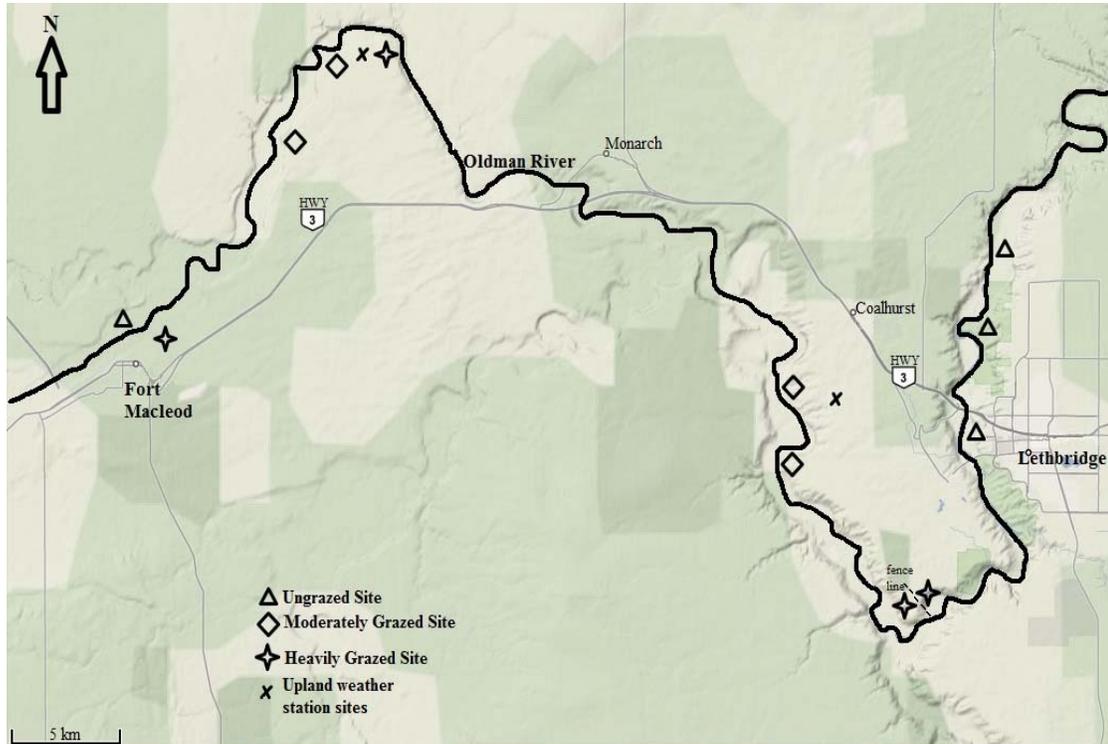
b)



c) **Figure 5-2.** A simulation of the amount of forage consumed by cattle over consecutive years of cattle grazing. In lightly grazed sites, cattle consume 32% of available forage, and grazing pressure is 1.25 AUM. In moderately grazed sites, cattle consume 43% of available forage, and grazing pressure is 2.5 AUM. In heavily grazed sites, cattle consume 57% of available forage, and grazing pressure is 4 AUM. Graph a) shows the amount of aboveground net primary productivity (ANPP) consumed by cattle each year. Graph b) shows the amount of ANPP consumed by each cow-calf pair (1 AUM). Graph c) shows the number of days per year that one cow-calf pair can be sustained by 1 hectare of land based on mean forage consumption rates (see Table 5-1).

Appendix 1

Map of study sites along the Oldman River in southern Alberta.



Appendix 2

List of All Bird Species Surveyed During 2005 and 2006 Field Seasons

* indicates a bird species found in only one site.

American tree sparrow *
Bald eagle
Black-capped chickadee
Bluebird
Bluejay *
Canada goose
Cedar waxwing *
Chipping sparrow *
Crow
Dark-eyed junco
Downy woodpecker
Flicker
Great horned owl
Hairy woodpecker
House finch
Hungarian partridge *
Magpie
Mallard *
Redpoll
American robin
Ruby crowned kinglet *
Starling
Swainson's hawk
Three-toed woodpecker
Tree swallow *
White-breasted nuthatch